Science Synthesis to Support Land and Resource Management Plan Revision in the Sierra Nevada and Southern Cascades

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1.0 Introduction

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Purpose and Objective

National Forests in the Sierra Nevada and Southern Cascades are preparing to review and revise their land and resource management plans (LMRPs). The three most southern national forests of the Sierra Nevada (Inyo, Sequoia, and Sierra) were selected to be the lead forests for Region 5 and are among the first of the 155 national forests to update their plans. The new planning rule requires the forests to consider the best available science and encourages a more active role for research in plan development.

To help meet this requirement, the Pacific Southwest Region (R5) Leadership asked the Pacific Southwest Research Station (PSW) to develop a synthesis of relevant science that has become available since the development of the existing LRMPs. Regional Leadership and stakeholders suggested that the GTR-220 report (North et al. 2009) served as a useful format, but that the content and scope of that report should be expanded to address additional biological, social, and economic challenges. In response to this request, a team of scientists from PSW and the Pacific Northwest Research Station (PNW) assembled to meet the goals of the effort and to engage with forest managers and stakeholders. Team members participated in the public Sierra-Cascades Dialog sessions and met with Forest Service leadership and managers and external stakeholders to learn about their concerns, interests, and management challenges.

Aware that a simple compilation or annotated bibliography of information would not meet management needs, the team discussed what format would make a synthesis more relevant and understandable. Most scientific research yields incremental steps forward, but those advances can be compiled to develop an understanding of larger systems. Many of the major environmental challenges that are likely to significantly affect ecosystem resilience, such as climate change, wildfire hazard, and air pollution, are best understood at large scales. To maintain and improve ecological integrity and associated ecosystem services (e.g., biodiversity, ecosystem health, water quality and quantity, recreation, economically viable communities) will require assessing and mitigating potential stressors in the near and long term across large forested landscapes. Therefore, the synthesis team chose to focus on synthesizing scientific information that would inform strategies that are likely to promote resilience of socioecological systems and sustain values at risk in the synthesis area over the short and long terms given expected stressors. This introductory chapter explains the components of that objective to help the reader understand what the synthesis was intended to address.

Synthesis Area

This synthesis presents recent science that is relevant to forest planning in the synthesis area, which includes the forested mountains of the Sierra Nevada, the southern Cascade Range, and the Modoc Plateau (Fig. 1). The synthesis primarily focuses on conifer-dominated forest ecosystems that constitute the vast majority of this area, although the Water Resources and Aquatic Ecosystems section includes chapters on forested riparian areas (6.2), wet meadows (6.3), and lakes (6.4). The broader concepts discussed in this document are likely to be useful beyond the area and ecosystems of focus. However, many of the specific examples may not necessarily be applicable to other areas, especially drier areas that are more representative of the Great Basin.

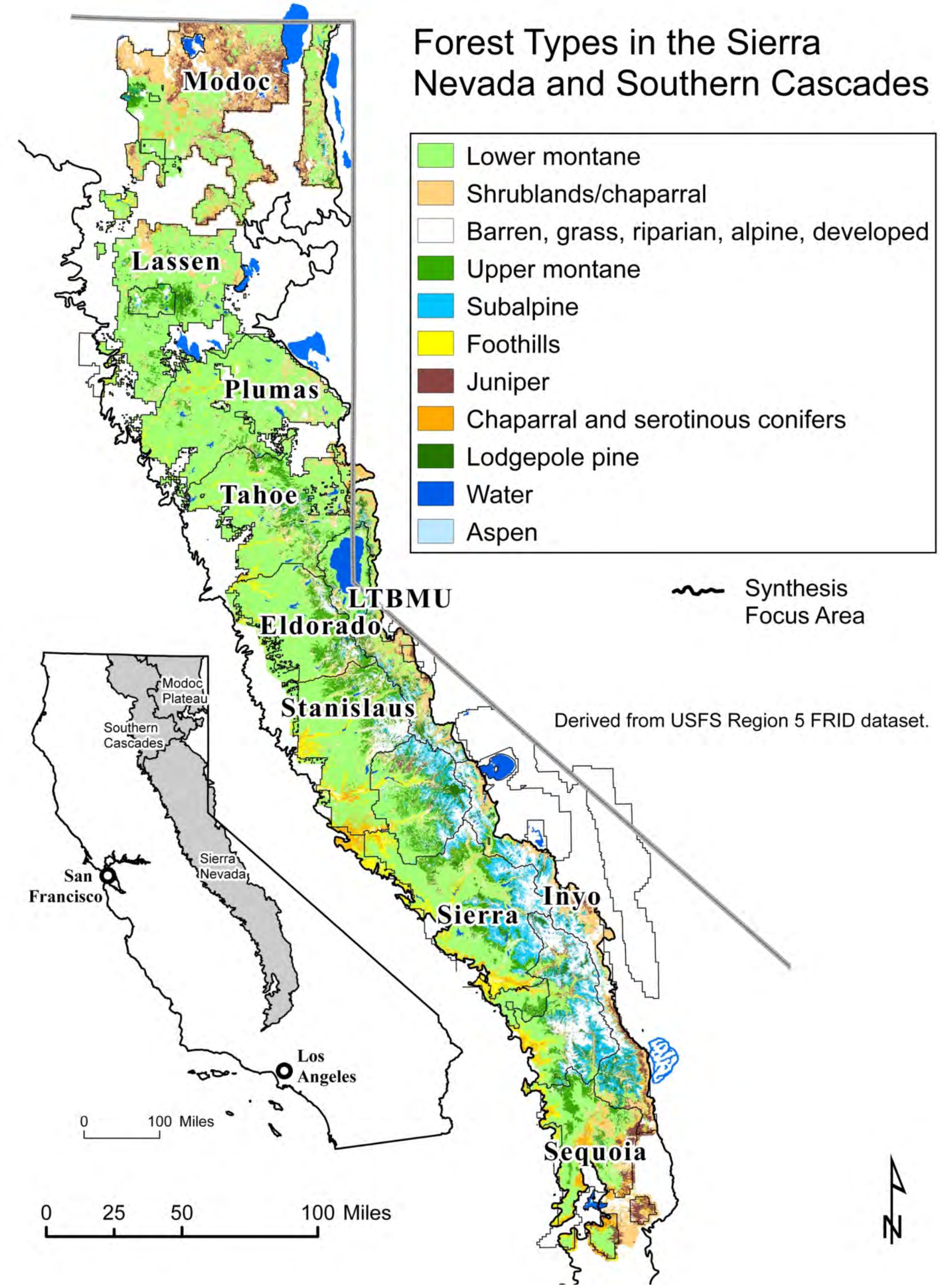

Figure 1: Focal areas of this synthesis are the conifer-dominated forests in the mountains of the Sierra Nevada, southern Cascades, and Modoc Plateau.

Scope and Approach

This synthesis emphasizes recent advances in scientific understanding that pertain to some of the most important issues facing managers across the synthesis area. These advances can help managers integrate ecological and social considerations across multiple spatial and temporal scales. The intent of this synthesis was not to create a comprehensive summary of the latest science, and chapters do not represent a complete review of all available literature. A number of recent management-oriented syntheses focused on different topics and disciplines have become available. These are referenced within the synthesis chapters and are also listed in an appendix.

The science synthesis team selected topics they considered most highly relevant to management in the focal parts of the synthesis area, based upon input from management, stakeholders, and reviewers, and to be consistent with priority topics highlighted in the planning rule:

> *"The planning rule is designed to ensure that plans provide for the sustainability of ecosystems and resources; meet the need for forest restoration and conservation, watershed protection, and species diversity and conservation; and assist the Agency in providing a sustainable flow of benefits, services, and uses of NFS lands that provide jobs and contribute to the economic and social sustainability of communities" (USDA Forest Service 2012).*

This synthesis is modeled in part after two prior synthesis reports, General Technical Report (GTR)-220 (North et al. 2009) and GTR-237 (North 2012), both published by the Pacific Southwest Research Station, which focused on management strategies for Sierra Nevada mixed-conifer forests. These reports provided a foundation for many of the broader strategies emphasized in this synthesis, and similarly emphasized a few wildlife species that have been management priorities.[1] This synthesis expands beyond terrestrial forest and fire ecology to include watershed and aquatic values and social systems, given their importance in the planning rule. Important themes running through the synthesis are the importance of scaling up from short-term, small-scale understandings to address long-term, landscape-scale processes, and to consider interactions within socioecological systems. In addition, the synthesis considers how changes in climate, air pollution, and other stressors are creating novel conditions that require broad adaptive approaches to management.

Like GTR-220 and GTR-237, this synthesis integrates findings from a range of scientific disciplines to inform the development of management strategies. The goal of this synthesis is to inform forest planning across the synthesis area rather than tactics at the project level. Strategic planning helps to define broad, integrative approaches that guide the goals, location, and timing of projects. Strategic

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[1] The two terrestrial wildlife chapters in this synthesis focus on three species that have been a priority for management and research: California spotted owl (*Strix occidentalis occidentalis*), fisher (*Martes pennanti*), and Pacific marten (*Martes caurina*). These species have been designated as Forest Service Sensitive Species by the Regional Forester. They are likely to be a focus of fine-filter analysis and monitoring under the new planning rule. In addition, they have had special habitat designations and they range across large areas; these attributes pose special challenges for landscape-scale management.

goals are often more conceptual and qualitative than the quantitative nature of project planning (Wood and Dejeddour 1992). The scales of space and time considered in strategic planning are usually more expansive (across broad landscapes and decades) than scales considered in project-level planning, which focus on a more localized place over a few years (Partidário 2007). Therefore, the resolution and precision of useful information often differ between these levels of planning.

Focus on peer-reviewed literature

This report is not an exhaustive review of the literature, a task which would have been beyond the scope and resources of the synthesis team. This synthesis focuses on published, peer-reviewed literature, with the majority of references published since the last round of science synthesis in the region, which included the Sierra Nevada Ecosystem Project in 1996 and a follow-up report on livestock grazing in 1999. Peer-reviewed literature is not the only valid source of information to inform management strategies, but a focus on that literature narrows the breadth to a more manageable level, highlights cross-cutting issues relevant at the scale of strategic planning, and reduces the burden of having to add an additional layer of peer review. Several of the sections also include gray text boxes that alert readers to recent or pending relevant studies that are expected but not yet published in peer-reviewed literature.

The emphasis on literature that has been clearly peer-reviewed is likely to leave out relevant scientific information that may be contained in reports by agencies, universities, and non-profit organizations, as well as in master's theses and dissertations. This restriction may pose particular concern for social, economic, and health issues. However, the plan revision process includes the parallel assessment phase, which is not limited to peer-reviewed literature.

In general, the team focused its scope to peer-reviewed research that occurred in the Sierra Nevada or in forest communities with relevant ecological or social conditions. Ecological and social research is always context specific, and there are few, if any, universal principles in either of these disciplines because place, time, and research scope all affect the data that are collected. Scientific studies are published with strict caveats about their spatial and temporal scales, making it difficult for managers and even other scientists to integrate and distill the information for particular management situations. This report tries to clarify the extent and limitations of available information, especially by highlighting various research gaps.

All chapters of the synthesis were reviewed by numerous individuals within Forest Service management and research, as well as by scientists from outside the Forest Service. This review process greatly helped to enhance both the content and readability of this report.

Structure

This report has several formats that reflect the effort to distill and integrate relevant research at different levels. The majority of the report is composed of chapters that summarize information or address key questions in specific topical areas (e.g., forest ecology, air quality, soils, and ecosystem services). These chapters address issues the authors considered highly relevant and ripe for synthesis, including topics suggested by managers, stakeholders, and reviewers.

The chapters in this first section have a different structure, which is designed to promote greater integration and generalization. The first chapter, Integrative Approaches (1.1), condenses much of the information from the different disciplines and summarizes themes that run through the topical chapters. The Synopsis of Emergent Approaches (1.2) and Synopsis of Climate Change (1.3) are highly condensed chapters that succinctly integrate and summarize central themes relevant to management of Sierra Nevada forests. Those two subjects were selected to address emerging challenges faced by the national forests. A final chapter in the integration section (1.4) focuses on adaptive management efforts and research gaps that also cut across the topical sections. Readers are encouraged to explore these different levels to understand connections across the various disciplines and topics.

Definitions of Resilience and Related Concepts

> "*Our goal is to sustain and restore ecosystems that can deliver all the benefits that Americans want and need. Due to changing climate, we may not be able to restore them to their original condition, but we can move them toward ecological integrity and health. The Forest Service recognizes that increasing the pace and scale of restoration and active management of the National Forests is critically needed to address these threats to the resiliency of our forests and watersheds and the health and safety of America's forest-dependent communities*" (Tidwell 2012).

> "*Our goal for the Pacific Southwest Region is to retain and restore ecological resilience of the National Forest lands to achieve sustainable ecosystems that provide a broad range of services to humans and other organisms"* (USDA Forest Service 2011).

Current goals for the US Forest Service policies (stated above) emphasize the concepts of restoration, resilience, and integrity. These terms are related and they are often used together, although their specific definitions have different emphases.

Restoration

Ecological restoration is commonly defined as "the process of assisting the recovery of an ecosystem that has been degraded, damaged, or destroyed" (Society for Ecological Restoration 1994: 132). The Forest Service has adopted the SER (1994) definition of ecological restoration while also incorporating the concepts of resilience and capacity to respond to future conditions by adding the following statement: "Ecological restoration focuses on reestablishing the composition, structure, pattern, and ecological processes necessary to facilitate terrestrial and aquatic ecosystems sustainability, resilience, and health under current and future conditions" (Office of the Federal Register 2012: 70).

Integrity

Originating from the field of water quality, ecological integrity has been defined as a combination of chemical, physical, and biological integrity, with integrity specifically defined as "the capability of supporting and maintaining a balanced, integrated, adaptive community of organisms having species composition, diversity, and functional organization comparable to that of natural habitats of the region" (Karr and Dudley 1981: 56). Ecological integrity can be seen as a state that allows an ecosystem to withstand and recover from natural and human-caused perturbations (Karr and Dudley 1981). The definition of ecological integrity in the recent Forest Service Planning Rule reflects this concept of a resilient state: “The quality or condition of an ecosystem when its dominant ecological characteristics (for example, composition, structure, function, connectivity, and species composition and diversity) occur within the natural range of variation and can withstand and recover from most perturbations imposed by natural environmental dynamics or human influence” (Office of the Federal Register 2012: 67).

Ecological Resilience

Ecological resilience is defined as the amount of disturbance an ecosystem can absorb without crossing a threshold to a different stable state, where a different range of variation of ecological processes and structures reigns (Gunderson 2000). This definition implicitly requires consideration of temporal changes relative to a reference condition, either backward to a past condition (or range of conditions) or forward to a desired future condition.

The ecological concepts of restoration, integrity, and resilience all converge around the concept of a system remaining close to a reference state (Safford et al. 2012). Such a reference need not necessarily include human influence; however, for the forested ecosystems of the Sierra Nevada, it is necessary to account for the fundamental historical role of Native Americans in maintaining a frequent fire regime and associated ecological communities (Nowacki et al. 2012).

Ecological Processes, Disturbances, and Stressors

Clarifying the meaning of ecological process, disturbance, and stressor may be helpful since terms associated with the concepts of equilibrium and succession can be problematic for describing how ecosystems change. A disturbance is commonly defined as a relatively discrete event that disrupts ecosystem structure and alters resource availability (White and Pickett 1985). Events such as a fire or flood can liberate resources and create conditions favorable to early seral species. However, Sugihara et al. (2006) contend that in many forests of California, fire was historically such a regular and essential ecological process that from an ecological perspective, it would be misleading to label it as a “disturbance,” or external disruption to the system. Stressors, on the other hand, are influences that act, either alone or in combination (e.g., climate change, air pollution, etc.), on ecosystems in ways that reduce their potential to be resilient to disturbance.[2] Though distinctions among these terms can be helpful, they fall along a gradient based upon the temporal and spatial scale of their influence relative to the ecosystem under consideration (Sugihara et al. 2006).

Berkes and Ross (2012) similarly suggest that the concepts of *resilient change* (remaining in a condition with essentially the same function, structure, identity, and feedbacks) and *transformation* (moving to a new state or “regime shift”) also describe a continuum rather than a dichotomy. Research to distinguish those two kinds of outcomes has been undertaken in subalpine whitebark forests (see Chapter 1.4). When differentiating uncharacteristic events, not only is their magnitude (i.e., size and intensity)

[2]The Forest Service has defined stressors in relation to ecological integrity as “factors that may directly or indirectly degrade or impair ecosystem composition, structure or ecological process in a manner that may impair its ecological integrity, such as an invasive species, loss of connectivity, or the disruption of a natural disturbance regime” (Office of the Federal Register 2012: 70).

relevant, but also the damage they cause to valued resources, including ecosystem services. Events that result in great damage over large areas represent one end of the continuum.

A comparison between fire regimes and flood regimes may be helpful in illustrating these points, since both have important roles in rejuvenating ecosystems by clearing dead plant material, redistributing nutrients, and exposing mineral soils. Small and frequent floods over streambanks often do not destroy much living vegetation or significantly alter channel structure. Less frequent floods typically do more work in terms of reshaping channel features and removing live biomass. Following unusual storms or severe wildfires, exceptionally large floods have potential to induce major ecological disruptions, particularly in streams with vulnerable and disconnected aquatic populations (Watersheds and Aquatic Ecosystems chapter (6.1)). Such floods also pose threats to life and property, which is why they are a priority for post-fire planning (see Post-wildfire Management chapter (4.3)). Resilience-based strategies often emphasize shifting a disturbance regime to promote more frequent small events and reducing the vulnerability of the socioecological system to large ones.

Integration of Social and Ecological Systems and Socioecological Resilience

A premise of this synthesis is that attempts to restore the integrity of ecosystems or maintain or increase the resilience of ecosystems to global change will depend on the extent to which those efforts can successfully integrate ecological and socio-economic concerns (Folke et al. 2010). An interdependent socioecological system ("SES") has been defined by (Redman et al. 2004) as:

1. a coherent system of biophysical and social factors that regularly interact in a resilient, sustained manner;
2. a system that is defined at several spatial, temporal, and organizational scales, which may be hierarchically linked;
3. a set of critical resources (natural, socioeconomic, and cultural) whose flow and use is regulated by a combination of ecological and social systems; and
4. a perpetually dynamic, complex system with continuous adaptation.

Key areas of emphasis in the synthesis flow from the SES concept, including the importance of understanding linkages across spatial and temporal scales; the interaction of biophysical and social factors; the flow of critical resources or ecological goods and services that are natural, socioeconomic, and cultural; and the dynamic and adaptive nature of systems. Scientists studying SES emphasize that dynamic and adaptive nature in characterizing socioecological resilience as the capacity of systems to cope with, adapt to, and shape change; to persist and develop in the face of change; and to innovate and transform into new, more desirable configurations in response to disturbance (Folke 2006). Similarly, the idea of adaptation is emphasized in a definition of community resilience as "the existence, development, and engagement of community resources by community members to thrive in an environment characterized by change, uncertainty, unpredictability, and surprise" (Magis 2010: 402).

An emphasis on dynamic, continuous adaptation implies a departure from narrower ecological definitions of resilience that emphasize a return to an equilibrium condition following disturbance (Folke 2006). Human communities have capacity to respond intentionally to create a desirable configuration, whereas ecological systems may often shift in ways that are undesirable for human communities. The definitions in this section point to important concepts that can be incorporated in plans to promote ecological integrity and social well-being. The next chapter goes deeper into the concept of socioecological resilience by describing some of the potential threats to critical resources that could shift systems in the synthesis area to less desirable configurations.

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1.1 Integrative Approaches: Promoting Socioecological Resilience

Jonathan Long, Carl Skinner, Malcolm North, and Lenya Quinn-Davidson with numerous contributions from the Science Synthesis Team

Contents

Introduction

This chapter begins by discussing current management challenges that emerged from multiple chapters of the full report. It then it considers integrative approaches to promote resilience, including general strategies that recognize the integrated nature of socioecological systems, the importance of promoting disturbance regimes upon which these systems have evolved, and opportunities to integrate social considerations into strategies (see the Introduction chapter (1.0) for definitions of key terms). It continues by outlining an adaptive management approach to scale up current practices so that planning and implementation are more congruent with the scales at which processes affect ecosystems in the synthesis area. The following chapter, a Synopsis of Emergent Approaches (1.2), focuses on three important themes that are touched on in this chapter; these themes emerged largely from synthesizing findings from the forest ecology, fire, and wildlife chapters. Chapter 1.3, Synopsis of Climate Change, provides a synopsis of climate change, which is another issue that cut across every topic in this synthesis. Chapter 1.4, Research Gaps, discusses a number of current adaptive management efforts and important topics that emerged as priorities for adaptive management and research. Altogether, the chapters in this section outline strategies to proactively respond to expected challenges in the synthesis area.

The Challenge of Multiple Stressors

The challenges facing national forests in the synthesis area have grown much more complex. They reflect long-standing natural processes (including fire, drought, and insects), decades of fire suppression and other practices that have altered ecosystems (e.g., hydrologic modifications, habitat fragmentation, loss of biodiversity, etc.), and increasingly, novel stressors associated with human activities within the region and even across the globe (Folke 2006, Steffen et al. 2007). Invasions of plants, animals, insect pests, diseases, and pollutants, combined with a legacy of human influences on climate, fire regimes, and species extinctions, are forming “novel ecosystems” (Hobbs et al. 2009), which do not have historical analogues upon which to base predictions or to serve as clear references for restoration. The remainder of this chapter focuses on several opportunities to promote system resilience to stressors.

There are many challenges to managing forests of the synthesis area in the 21st century, including an array of evolving, novel stressors:

- Dust from as far away as China may be causing snowpack to decline and polluting water bodies in alpine areas that have historically been regarded as relatively pristine wilderness (see Air Quality chapter (8.0)).
- The barred owl (*Strix varia*) is invading forests at the expense of the California spotted owl, and there are no clear solutions to prevent this incursion (Gutierrez et al. 2007) (see California Spotted Owl chapter (7.2)).
- The fisher is being poisoned by application of rodenticides by marijuana growers to protect their illicit crops (Gabriel et al. 2012) (see Forest Carnivores chapter (7.1)).
- Populations of priority amphibians face combined effects of climate change, introduction of predatory fishes, disease, pesticides, disrupted flow regimes, and other habitat impacts (see Lakes chapter (6.5)).
- Climate change is projected to shift precipitation from snow to rain, which may reduce seasonal water availability in forest soils, and negatively impact aquatic systems and associated ecosystem services by altering channel stability and stream hydrographs, especially by reducing summer baseflows (see Watersheds and Stream Ecosystems chapter (6.1)).
- Climate-driven projections suggest that forests will become more susceptible to insect attack and disease (Evangelista et al. 2011, Sturrock et al. 2011), and a complex interaction of climate change, altered fire regimes, and air pollution pose threats to forest resilience (see Air Quality chapter (8.0)). Research has already documented increased rates of insect attack, disease, and mortality in many Western forests that could portend vulnerability to substantial changes in forest structure, composition, and function (van Mantgem et al. 2009).
- Scientists report increasing frequency and extent of wildfires, along with the increasing occurrence of uncharacteristically severe wildfire in the synthesis area (Lenihan et al. 2003, Miller et al. 2012, Miller and Safford 2012, Miller et al. 2009, Westerling et al. 2011) (see Fire and Fuels chapter (4.1)).

Land management agencies have limited ability to prevent these impacts, but effective management actions can mitigate their effects. The new planning rule acknowledges the likelihood that some stressors may render it infeasible to maintain or restore ecological conditions to maintain a viable population of a species of conservation concern in a planning area. The existence of such stressors complicates management because it becomes harder to evaluate the effects of management actions without accounting for the stressor that cannot be controlled. Interactions between climate change, other stressors, and disturbances can induce positive feedbacks that threaten to push systems beyond key thresholds; these challenges should be considered as syndromes rather than as isolated problems (Rapport and Maffi 2011). Common indicators of such syndromes include losses of biodiversity, especially predators; simplifications of food webs; eutrophication of aquatic systems; and increasing prevalence of invasive species and diseases (Rapport and Singh 2006). Assessments, research studies, and management strategies that target these syndromes will be most effective if they consider multiple factors and their synergistic effects. Amphibians in lakes provide an example in the Sierra Nevada (see chapter 6.4) of how a response to a syndrome could include removing introduced fishes from lakes to help amphibians better withstand disease and climate change. Other strategies for assessing and responding to these syndrome impacts have been to develop highly integrated ecological indexes or state of the environment reports, which consider effects on both ecosystems and social systems, and emphasize opportunities for human actions to improve ecological health (Rapport and Singh 2006).

Wildfire Conditions that Reduce Socioecological Resilience

Periodic disturbance plays a fundamental role in the development of socioecological systems by facilitating reorganization and renewal (Cabell and Oelofse 2012, Folke 2006). However, people often regard such disturbances negatively because of their disruptive effects. Indeed, major shocks that push systems beyond critical thresholds can induce large and persistent loss in the flows of ecological services, and they can reduce the ability of a system to recover from future disturbances. Human actions have contributed to these regime shifts by reducing biodiversity, altering fire regimes, and inducing soil erosion, with negative impacts on livelihoods and societal development (Folke 2006). Stressors associated with anthropogenic activities, such as climate change, pollution, and species invasions, are critically important to consider from a socioecological perspective, although they are difficult to manage because they originate from outside local landscapes and/or do not recur frequently and predictably.

Fire is a fundamental ecological process that often repeats in relatively predictable ways across a landscape. Native Americans in the synthesis area historically lived with fire and used it to promote ecological outcomes to support their communities. Changes in forest fuel and habitat conditions over time can leave systems vulnerable to regime shifts (Agee 2002). If forests that have uncharacteristically large accumulations of living and dead fuels are not managed, when they inevitably burn there will be a loss of ecosystem services, including biodiversity and other social values (Franklin and Agee 2003). Consequently, various topical sections of this synthesis describe serious negative consequences of uncharacteristically large and severe fires on many socioecological values in the modern era. These impacts include:

1) High levels of tree mortality over large areas can forestall recovery of forested conditions and associated ecosystem services for long periods (decades to centuries) and may be a catalyst for regime shifts as climate change progresses. Even if these systems begin to regrow trees, they may be more vulnerable to effects of future fires. (Post-wildfire Management chapter (4.3))
2) Widespread tree mortality and persistent loss of trees may be associated with significant emissions of carbon, as forests are converted from carbon sinks into source areas for extended period (Dore et al. 2012). (Forest Ecology chapter (4.2))
3) Widespread patches of tree mortality may represent a loss of habitat for species such as California spotted owl, fisher, and Pacific marten. (California Spotted Owl chapter (7.2) and Forest Carnivore chapter (7.1))

4) Intense, large, and long-lasting wildfires are likely to cause exceedances of air quality standards instituted to protect human health. It is much more difficult to control air quality and other impacts from those wildfires than it is from prescribed fires. (Air Quality chapter (8.0))
5) Although aquatic systems often demonstrate relatively high levels of resilience to wildfire, very large and severe wildfires may induce significant channel erosion and reorganization that can extirpate vulnerable aquatic populations, degrade downstream water quality, reduce storage capacity of downstream reservoirs, and elevate flood risks. (see Post-wildfire Management chapter (4.3))

It is more difficult to identify critical thresholds beyond which the resilience of social systems substantially erodes (see Job Creation through Forest Management chapter (9.4)). However, fires can cause a range of impacts to social values, and much greater impacts are expected to result from fires that burn intensely, over large areas, and for long periods. Beyond the threats to life, property, and air quality, such fires can induce the acute stress of evacuation, as well as longer-term impacts to individual health and community well-being. Severe wildfires that cause widespread tree mortality impact socio-economic values, including timber flows that contribute to local economies and maintain their infrastructure and markets for forest products. Such fires also threaten non-use values of people well beyond California, as residents of New England expressed willingness to pay substantial sums to treat and protect old-growth forests associated with spotted owls from high-intensity wildfire (Loomis and Gonzalez-Caban 1998).

Uncharacteristically severe fires may constitute a threat to system resilience for some components of a socioecological system but not others. Not all fires that result in widespread tree mortality should be viewed as causing a loss of ecological resilience; in some cases, trees may have invaded areas that were much more open or even dominated by non-forest vegetation under a fire regime that existed prior to fire suppression (see Post-wildfire Management chapter (4.3)).

The ultimate measure of resilience is how systems respond to major shocks, so it can be a difficult property to evaluate except in hindsight. However, there may be useful indicators that point to vulnerabilities. The chapter on Research Gaps (1.4) concludes with further consideration of indicators of resilience. There may be some resources that decline in the absence of recurring fire. Some of these components may include yellow pines, sugar pine, and black oak in frequent fire mixed-conifer forests (see Fire and Tribal Cultural Resources chapter (4.2)) and wildlife species that depend on habitat created and maintained by such fire.

Risks of Non-treatment

North et al. (2012) have pointed out that large areas of the Sierra Nevada are unlikely to receive needed forest treatments. Foregoing treatments can result in lasting impacts to ecosystems, human communities, and myriad ecosystem services. For example, deferring tree harvest for extended periods can not only impose social and economic impacts, but it can also result in losses of key infrastructure needed to maintain capacity to conduct restoration treatments and preserve options for future forest management (see chapter on Managing Forest Products (9.5)). Furthermore, the global dimensions of economic and environmental issues mean that reducing harvests in local forests can have an unintended consequence of increasing environmental impacts much farther away (Berlik et al. 2002). The likelihood of major disruptions in the long term may increase if treatments are avoided. For instance, simulations by Scheller et al. (2011) found that the threat of large, severe wildfires to habitat of fisher over large areas outweighs the short-term negative effects of fuels treatments on fisher population size. Moreover, the analysis noted that the benefits of treatment would be even greater if climate change makes wildfires larger and more severe, as is expected. In a similar vein, Roloff et al. (2012) completed a risk analysis of fuels treatments for northern spotted owls in southwest Oregon, which suggested that active management posed fewer risks than no management in fire-prone landscapes, although they cautioned that this strategy requires testing through field evaluation under an adaptive management framework. For these reasons, lack of treatment may exact a higher cost than first imagined, and the desire to avoid short-term risk from an institutional perspective must be weighed against the larger social risks that may be compounded through inaction.

Recognizing and Resolving Scale Mismatches

Research to understand socio-economic and ecological processes is often restricted to a narrow range of influences, effects, localities, and time frames that facilitate study (see Table 1 for common spatial scales of ecological studies), but these constraints may not sufficiently reflect important processes that operate at larger scales. These types of scale mismatches have exacerbated debates over how best to manage national forests. In the Sierra Nevada, research has rarely been conducted in an interdisciplinary, cross-scale fashion that could enable better understanding of the dynamics and interactions of patterns over multiple scales of both space and time (Bissonette 1997). Many of the areas that have been designated for experimental approaches are relatively small (see Table 1 in Research Gaps chapter (1.4)). Likewise, there have been few attempts to craft a cohesive, interdisciplinary management strategy aimed at achieving multiple but seemingly disparate objectives. Forest management practices are often regulated by standards set for localized conditions at a single point in time, despite the fact that forest conditions continuously change in both space and time via stand development and disturbance processes. The integrated approach suggested in GTR-220 by North et al. (2009) took important steps forward in promoting a landscape strategy and collaboration across the disciplines of forest ecology, wildlife biology, and silviculture. The follow-up report, GTR-237, also edited by North (2012), extended those recommended considerations to include bark beetles, climate change, and various wildlife communities, and featured examples of collaboration and adaptive management experiments.

Integrated management strategies that consider effects at scales of 50 years or more, across local to large spatial scales, and across ecological and social dimensions, could help enhance socioecological resilience. Management approaches that seem sub-optimal from a stand-level perspective may be favored when seen from a landscape perspective (or vice-versa), because the effects of treating a stand may influence how the landscape as a whole responds to fire. For this reason, strategies that opportunistically target areas suggested by high fuel loads, low treatment costs, and reduced obstacles (such as regulations or additional planning requirements), can leave large parts of the landscape vulnerable to uncharacteristically severe wildfire under a management regime dominated by fire suppression. In a similar fashion, aquatic scientists have reinforced the importance of moving beyond reach-scale evaluations of conditions and projects to assessing how management shifts the cumulative distribution of stream conditions within a watershed over decades (Benda et al. 2003). The importance of a landscape perspective to promote forest resilience is detailed in the following chapter (1.2).

Table 1. Minimum scales needed to evaluate ecological data that can be collected at various spatial scales to answer research and management questions. For non-ecological data, see the scale discussion in the Social, Economic, and Cultural components Broader Context chapter (9.1).

Typical Minimum Scale of Data	Ecological attributes or processes
Plot (< 1 to 10 acres) to stand scale (100 acres)	Vegetation structure, composition, and regeneration Fire effects on plants, soils, insects, wildlife with small home ranges, etc. Effects of some mechanical and prescribed fire treatments and wildfires Soil structure and chemistry Soil erosion Wildlife with small home ranges, such as small mammals, birds, and amphibians Use of habitat patches by species with large home ranges (i.e., nest patch and foraging patch) Meadows Air pollution effects Tree genetics
Small landscape scale (100 to 1,000 acres), including headwater watersheds	Linkages between terrestrial watersheds and aquatic systems Stream water quantity and quality Benthic macro-invertebrates Sediment loads Fire effects on stands to small watersheds Fire history and stand structure reconstruction
Intermediate landscape scale (1,000 to 100,000 acres)	Terrestrial wildlife with large home range dynamics (e.g., raptors, forest carnivores and other large mammals) and fishes Fire history and stand structure reconstruction Fire severity patterns Fuel treatment effectiveness to reduce large, high-intensity wildfires Climatic influences on fire regimes and sub-basin hydrology
Large landscape scale (100,000 acres and larger)	Population dynamics of wildlife with large home ranges Landscape genomics Climatic influences on regional fire activity

Strategies to Promote Socioecological Resilience

The introduction to this synthesis (chapter 1.0) defines socioecological resilience as "the capacity of systems to cope with, adapt to, and shape change; to persist and develop in the face of change; and to innovate and transform into new, more desirable configurations in response to disturbance." **This**

synthesis focuses in particular on the long-term challenges posed by wildfire and climate change because of their potential to affect the resilience of socioecological systems throughout the region. This section considers several general strategies to address these kinds of challenges, beginning with several principles that emerge from a broad-scale perspective on ecological resilience.

General Strategies for Addressing Challenges

- **Recognize and address scale mismatches**—the temporal and spatial scales of management systems may not be well matched to the scales of environmental variation (Cumming et al. 2006).
- **Consider long-term (more than 50 years) risks** in addition to short-term (less than 10 years) expected outcomes. Management focused on avoiding short-term risks is unlikely to sufficiently account for infrequent disturbances such as severe wildfires, nor for the progressive effects of climate change.
- **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 (Hobbs et al. 2010). In particular, the occurrence of large fires is likely to impact plans.
- **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, decision-making tools, modeling, monitoring, and, where appropriate, research** to evaluate responses and better account for the risks and tradeoffs involved in management strategies. Although applications of such tools entail many caveats, technologies have advanced to facilitate concurrent analysis of many tradeoffs, such as effects on air quality, fire risk, wildlife habitat, water quality, water quantity, and cultural and economic values.

Promoting heterogeneity, emulating natural disturbances, and restoring processes

Actively promoting forest heterogeneity through silviculture and managed fire is an important restoration strategy, especially given the threat of climate change (see Forest Ecology chapter (2.0)). Current forest conditions are often relatively homogenous due to past management practices and the absence of fire. Forests that developed under the influence of frequent, mostly low- and moderate-intensity fires exhibited very heterogeneous conditions that were likely produced by interacting effects of site productivity, topography, and fire history. These forests were common historically, but are now very limited due to fire suppression. Researchers have suggested actively promoting greater diversity in stand structure, age, species composition, and genetic backgrounds within those species as a bet-hedging strategy to address uncertainty associated with climate change (Notaro et al. 2012). Treatments to reduce and promote variation in stem density and fuel loads should promote forest resilience to large disturbances associated with climate change, including droughts and insect outbreaks (Fettig 2012, Littell et al. 2012, Safford et al. 2012b).

North et al. (2012) have suggested that the most practical strategy for treating large areas is to significantly expand managed fire, while recognizing the importance of structural treatments to facilitate such a strategy. This approach builds on the principle of natural disturbance-based management (North and Keeton 2008). A recent review of the Fire and Fire Surrogate study concluded that fire should be maintained whenever possible, since mechanical treatments did not serve as surrogates for fire for most variables (McIver et al. 2012). Successful adaptive management strategies will anticipate these disturbances. For areas with frequent fire regimes like the Sierra Nevada, Hirsch et al. (2001) called for “fire-smart” management strategies that acknowledge the inevitability of wildfire. Under this approach, managers would be supported in reducing future wildfire risks by incurring immediate risks through use of prescribed fire. To be successful, adaptive strategies may require integration of land management plans and fire management plans to address short-term responses to wildfire as well as long-term objectives for large-scale fire restoration. Post-wildfire plans are important not only because fires are likely to be widespread agents of change, but also because wildfires can open windows of opportunity to learn and to take actions to promote future resilience (Littell et al. 2012).

There are many areas where silvicultural treatments to modify stand structures would help to facilitate returning fire as a primary disturbance mechanism (Miller and Urban 2000). Varying forest conditions with micro- and macro-topography can help increase heterogeneity and provide managers with a template for how and where to vary treatments. Recent studies provide information on how forest conditions and fire regimes varied according to topography when active fire regimes were operating historically (Beaty and Taylor 2001; Scholl and Taylor 2010; Taylor 2000; Taylor and Skinner 1998, 2003) and in landscapes where fire regimes have been partially restored (Lydersen and North 2012). Treatment strategies that build on the concept of emulating natural disturbance regimes would vary treatment type and intensity according to topographic position; for some landscapes in this synthesis area, such an approach might include reducing fuels preferentially on drier southern and western slopes, as compared to north slopes and canyon bottoms, and managing ridgetops for fuelbreaks (Weatherspoon and Skinner 1996).

Emphasizing process-based restoration and indicators

Because a resilience-based restoration strategy places so much emphasis on the dynamism of systems, it demands greater attention to functional processes. Conditions and processes are so interconnected that restoration has to address both; however, restoration ecology has placed increasing emphasis on restoration of dynamic ecological processes versus static targets for structure and composition (Harris et al. 2006). For example, scientists in the field of stream restoration have called for less emphasis on in-stream structural approaches in favor of reestablishing disturbances regimes (fires and floods), vegetation dynamics, coarse woody debris recruitment, and lateral and longitudinal stream connectivity that build in-stream habitat (see Palmer et al. (2005) and Watersheds and Stream Ecosystems chapter (6.1)). In terrestrial forests that experience frequent fires, researchers contend that ecologically based restoration depends on successfully restoring mostly low- to moderate-intensity fire as a keystone process, while recognizing that fire regimes and stand structures must be restored in an integrated way (see Fire and Fuels chapter (4.1) and Allen et al. (2002)). Therefore, structural indicators remain essential, but they have to be considered in light of dynamic processes, and there is a need for indicators and metrics that focus on process.

In addition to abiotic processes like fires and floods, it is also important to consider biotic processes as indicators of ecological resilience. For example, predation is an important process given the potential for trophic cascades when predators are lost (see Forest Carnivores chapter (7.1)). Using an example from Lake Tahoe, Vander Zanden et al. (2003) demonstrated how consideration of long-term changes in food webs can guide restoration efforts, in particular by targeting systems where such changes have been less extensive.

Researchers studying aquatic systems have asserted that management systems have tended to rely too much on indicators of acceptable habitat conditions and water quality standards rather than embracing system dynamics and disturbance regimes (Rieman et al. 2003). Some decision-making systems may provide incentives to treat priority species and water quality as constraints, with an emphasis on avoiding short-term potentially negative impacts. However, a resilience-based approach needs to consider opportunities to sustain ecological values over the long-run. Foundational components include the physical-chemical aspects of soil and water, which in turn support vegetation and habitat for terrestrial and aquatic organisms. Because foundational ecological processes, such as soil water storage, may not have explicit targets, there may be a tendency to undervalue, or even ignore them in decision making. Yet, as noted in the Watersheds and Stream Ecosystems chapter (6.1), forest treatments have the potential to enhance system resilience to multiple stresses by increasing soil water availability. Such treatments, along with meadow restoration (see Wet Meadows chapter (6.3)), also have potential to enhance the yield, quality, and timing of downstream water flows and resulting ecosystem services. Another approach emphasized in promoting resilience of fluvial systems is to reestablish reference hydrologic regimes, including overbank flows in wet meadows (see Wet Meadows chapter (6.3)) and natural hydrograph patterns in regulated rivers (see Watersheds and Stream Ecosystems chapter (6.1)).The strategic orientation of GTR 220 and 237, which focuses on restoring heterogeneity and landscape-scale ecological processes, can address aquatic resources by incorporating key hydrologic processes as treatment objectives rather than primarily as constraints.

Using fire regime metrics to evaluate performance

By addressing system dynamics, process-based indicators avoid some of the shortcomings that may be posed by structural indicators, but they still pose a risk of oversimplification. Carefully selected fire regime metrics can be useful for setting priorities and evaluating performance, since they focus on a key disturbance process. Sugihara et al. (2006) identify seven important attributes for characterizing fire regimes, including fire return interval, seasonality, size, spatial complexity, fireline intensity, severity, and type. The total amount of area burned in any given year does not necessarily indicate failure or success, since there has been such a deficit of fire on the landscape since the onset of fire suppression. The proportion of area burned at low, moderate, and high severity and how the fires threaten human life and property are more important indicators. Area burned at low to moderate severity could be an important indicator of progress, whereas the extent of high-severity fire could be a useful indicator of a problem (Weatherspoon and Skinner 1996). In terms of achieving restoration goals, expectations for particular areas would need to be based on historical variation and/or contemporary reference sites, current conditions, and projections of climate change and future disturbance (Safford et al. 2012a). Fire return interval departure (FRID) analyses can help evaluate departures from reference conditions at a large scale. However, FRID analyses may not provide sufficient detail to apply these metrics at the project scale, and fire recurrence intervals alone are insufficient to drive treatment priorities (see Fire and Fuels chapter (4.1)). For example, depending on values at risk and socioecological context, it may be more important to maintain a restored or minimally departed condition in one area than it would be to correct a significantly departed condition in another.

It is also important to consider the various dimensions of the fire regime other than simple averages of fire frequency, since the variation in fire regime characteristics within and among fires is a more important influence on landscape heterogeneity and biodiversity (Agee 2002). Individual low severity burns are generally insufficient to restore reference structure and process after long fire-free periods (Collins et al. 2011, Miller and Urban 2000, Skinner 2005). Consequently, a metric like time since last fire may be useful as an initial look or as a short-term indicator of management performance, but it should not necessarily be construed as an indicator of a restored fire regime (see Fire and Fuels chapter (4.1)). Unqualified measures of area burned or area treated would not be particularly useful indicators of restoration of ecological process. More multidimensional metrics are needed to evaluate effectiveness in reducing hazard or in restoring ecosystems.

Managing long-term post-wildfire outcomes

Uncharacteristically severe wildfires will continue to affect large areas of the synthesis area in coming decades. The Burned Area Emergency Response (BAER) program addresses short-term post-fire impacts to life, property, and ecosystems, but a longer-term strategy is important for promoting resilience of ecosystems within severely burned landscapes (see Post-wildfire Management chapter (4.3)). The Forest Service in California has recently developed a template to help guide national forests in planning for restoration and long-term management of post-wildfire landscapes. Post-fire conditions offer opportunity to realign ecosystem structure, function, or composition with expected future climate. Large areas of uncharacteristically severe fire may shift ecosystems into less desirable states that may persist for long periods, especially since climate change is also influencing those trajectories. Additional research and extensive monitoring are needed to ensure that treatments of those areas do not rely on untested approaches applied in a piecemeal fashion without consideration of landscape context and changing climate.

Social and Ecological Integration

The “triple bottom line” concept, which emphasizes social, economic, and ecological dimensions of sustainability (see the Broader Context for Social, Economic, and Cultural Components chapter (9.1)), underscores the understanding that human and natural ecosystems are interdependent (see also the chapter on Fire and Tribal Cultural Resources (4.2)). An important part of a strategy to promote socioecological resilience is to explicitly consider social effects of management strategies on near and more distant human communities, as well as how community capacity can facilitate management to promote resilience. Researchers studying socioecological systems note that no single approach to governance, including broader and more active participation by local communities, will solve problems in managing socioecological systems (Ostrom 2007), because human-environment relationships are so complex and vary from one place to another. However, there is growing recognition that engagement, capacity building, and participation are necessary components of strategies that promote resilience through social learning (Fernandez-Gimenez et al. 2008).

Summary of approaches for integrating social considerations into 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. Recognizing and measuring **ecosystem services and other sociocultural values** can help to **consider impacts to communities and ecosystems** as part of this approach.
- **Use participatory and collaborative approaches to facilitate adaptive responses and social learning**. Many of the topical sections of this synthesis note how scientists have moved toward such approaches as a way of promoting resilience, especially where management systems may be geographically and culturally distant from people who use the forests and their local knowledge systems (examples in the Collaboration chapter (9.6) include grazing management and incorporation of traditional ecological knowledge).

Recognizing ecosystem services and other sociocultural values

The shift to thinking about integrated socioecological systems has spurred efforts to value ecosystem services (see Ecosystem Services chapter (9.2)), because an ecosystem's capacity to generate such services is the foundation for social and economic development (Folke 2006b). Understanding changing demand for many ecosystem services at different scales is crucial for developing appropriate management strategies (Grêt-Regamey et al. 2012). An important component of a resilience strategy may be to moderate societal expectations for ecological services rather than trying to provide a constant or ever-increasing supply. The Sierra Nevada encompasses watersheds that support millions of people and a large part of the global economy; therefore, potential impacts to water quality and quantity are of great importance locally, regionally, and even globally. Impacts of treatments and wildfires on these services are an important research topic (see Watersheds and Aquatic Ecosystems chapter (6.1), as well as Research Gaps (1.4)).

The sociocultural value of ecosystems is not limited to direct uses by people, as it also extends beyond the Sierra Nevada and Southern Cascades region. Research has shown that people living far from the Sierra Nevada hold substantial values for the region's ecosystems and especially for their charismatic fish and wildlife (Loomis and Gonzalez-Caban 1998, Richardson and Loomis 2009). Ecosystems also support community identity and sense of place (see Broader Context for Social, Economic, and Cultural Components chapter (9.1)). These values resist quantification and commodification but may be critical to maintaining the sustainability of socioecological systems (Berkes et al. 2006, Ostrom 2007).

Emphasizing values sustained by the forests may help facilitate communications with diverse members of the general public, local residents, landowners, and other groups. Studies have shown that science-based planning and communication are important for improving acceptability of proposed actions, such as wildfire risk reduction treatments, biomass utilization, and salvage logging (see Managing Forest Products chapter (9.5)). Because local communities often play a role in management practices related to biodiversity enhancement, soil and water protection, and improving other ecosystem services, managing forest products on national forest lands to benefit those communities can in turn provide environmental benefits for forest and rangeland ecosystems across ownerships (see Managing Forest Products chapter (9.5)).

Considering impacts to communities and ecosystems

An integrated landscape-scale strategy can promote restoration in ways that benefit both local communities and ecosystems using specific approaches that are discussed within the Job Creation through Forest Management chapter (9.4). However, potential solutions may entail various tradeoffs between ecological and social impacts at multiple scales. For example, tools like stewardship contracts afford certain benefits and flexibility to promote ecological restoration, but under current policies, they can also incur potential impacts to communities by reducing payments to local governments. Redressing public policies that create disincentives for ecological restoration may be important in developing a successful long-term strategy.

An important issue raised in different topical sections of this synthesis concerns the potential to generate energy and fuel from forest biomass. This approach holds promise for simultaneously reducing greenhouse gas and smoke emissions, promoting renewable energy and U.S. energy security, and facilitating larger-scale forest treatment. There has been considerable debate concerning whether forest biomass should be regarded as carbon-neutral, and assessments of the overall impact of emissions hinge on assumptions about fire regimes (Winford and Gaither 2012). However, there is consensus that the utilization of “waste” biomass debris that would otherwise release carbon quickly into the atmosphere (through decay or pile burning) is likely to be carbon friendly (Johnson 2009). Therefore, encouraging a shift from burning debris from harvest or fire hazard reductions in piles to burning in biomass facilities could yield significant environmental and economic benefits. Researchers have sought to estimate a sustainable supply of biomass that represents a by-product of other management objectives, such as pre-commercial thinning and wildfire hazard reduction (Parker et al. 2010). However, development of biomass utilization in the Sierra Nevada requires consideration of an array of ecological, economic, and social factors, and the overall impact and acceptability of biomass initiatives depends heavily on local conditions (see Managing Forest Products chapter (9.5)).

Promoting collaboration and partnerships

Consistent with an all-lands approach, working at the landscape scale will require greater coordination and partnerships with private landowners, non-governmental organizations, and state and local governments. In addition, collaboration demands consideration of views and interests of stakeholders at broad scales, including people who may be farther away than those who have traditionally been included in planning. Although collaboration entails costs and complications, stakeholder input and participation from early stages can be crucial in outlining shared goals and objectives, facilitating shared learning and problem solving, and building trust (Bartlett 2012). Although reaching consensus may not necessarily be a goal of planning, research from other areas, such as the yellow pine forests of northern Arizona, suggests that diverse stakeholder groups are able to reach consensus about managing very large landscapes (Hampton et al. 2011, Sisk et al. 2006), particularly since treatments need to initially target only a portion of the landscape to effectively reduce the risk of large, intense wildfires (Ager et al. 2007, Loehle 2004, Schmidt et al. 2008, Syphard et al. 2011). Achieving such agreement may be easier in high-relief areas where topography has a strong influence on effective treatment options and more difficult in areas with high-profile values (e.g., sequoia groves and habitat for wildlife species of concern). However, there are no guaranteed outcomes from adopting a collaborative process, and, as outlined in the Collaboration chapter (9.6), cross-boundary collaboration may be particularly challenging in some communities. That section also suggests that fire could be a rallying point, since creating strategies to reduce fire risk across boundaries may enhance cooperation.

Science-based monitoring and feedback mechanisms that enable adaptive management practices are valuable for correcting course and building trust and cooperation (Cox et al. 2010, Fernandez-Gimenez et al. 2008). Such adaptive systems are important because there are significant gaps in scientific knowledge of the behaviors of these complex systems, as outlined in the Research Gaps chapter (1.4). Furthermore, these approaches embed capacity to identify and benefit from new information discovered as a result of monitoring, shifts in social systems or values, shifts in ecological systems or dynamics, or a change in their interactions. A wide range of collaborative approaches to adaptive management, including participatory research and monitoring, and collaborations with tribal groups to investigate effects of reintroducing traditional burning practices, is discussed in the chapter on Collaboration (9.6).

Institutionalized science-management partnerships are an approach to collaborative adaptive management that has been developed in the synthesis area. These partnerships have attempted to facilitate the dissemination of scientific information directly to managers, while providing researchers with a better understanding of the constraints currently faced by managers, including the challenges associated with climate change (Littell et al. 2012). Robust science-management partnerships may also provide the added benefit of building stakeholder trust and encouraging creative approaches to adaptive management. Regional examples of these science-management partnerships include the Tahoe Science Consortium, Southern Sierra Conservation Cooperative, Northern California Prescribed Fire Council, and California Fire Science Consortium.

Landscape-Scale Adaptive Management

Experimental approaches to landscape management, focusing on fire-related treatments in particular, have been tested on relatively small scales within experimental forests and other adaptive management areas in the synthesis area (see Chapter 1.4). Designation of large-scale demonstration areas, on the order of 100,000-200,000 acres, could provide a valuable opportunity to build upon such approaches. That size is an order of magnitude larger than existing experimental forests (Table 1 in Chapter 1.4). Such large areas would make it easier to evaluate treatment impacts on wildfire at the landscape scale and on a wide range of species with large home ranges, such as California spotted owls and fishers. The Dinkey Collaborative Forest Landscape Restoration Project has enabled observations of the effects of prescribed fire on fishers, but larger areas are needed to address questions of connectivity for forest carnivores. The Northwest Forest Plan set up ten adaptive management areas (AMAs) that ranged in size from around 92,000 acres to almost 500,000 acres, to afford managers opportunity to test new approaches at large scales and adjust standards and guides to local conditions. Scientists have noted that the plan's potential to facilitate large-scale experimentation has not been fulfilled due to disagreements over what constitutes adaptive management and a perceived or real lack of sufficient flexibility to test different strategies (Rapp 2008, Stankey et al. 2003). Nevertheless, the Goosenest Adaptive Management Area (GAMA), located within this synthesis area, has demonstrated some on-the-ground progress (Rapp 2008). Whereas managers used the larger area to explore strategies pertaining to raptors, goshawk, and spotted owls, the GAMA Ecological Research study was undertaken to specifically test a variety of treatments designed to achieve the AMA's goal of accelerating late-successional conditions in young growth forests.

Because priority wildlife species are a key concern in forested landscapes, and research on those species has proven to be one of the most costly components of adaptive management studies, one approach would be to leverage existing large-scale studies. Three areas, including the Plumas-Lassen, Eldorado, and Sierra national forests, have been selected as sites for extensive, long-term California spotted owl demographic studies. They also include experimental forests and research sites for multidisciplinary ecological research. These areas, which are well distributed across the Sierra Nevada, could be designated as experimental landscapes where adaptive management would be implemented based on ideas suggested in recent scientific literature. Although designating experimental areas would increase capacity for evaluating ecological effects, much of the land managed in the National Forest System is heavily woven into human uses, demands, and place meanings. The owl demographic study areas may not adequately represent the diversity of those concerns, so there would have to be special consideration given to how to translate findings from these experimental areas to socially more complex areas.

Promoting variation in canopy cover and habitat conditions

A strategy based on within-stand and across-landscape heterogeneity appears suited to deter the spread of high-intensity fire across the landscape while providing for a wide range of habitat conditions (Knapp et al. 2012). This strategy differs from more traditional treatments that reduce canopy continuity and crown fire spread by leaving widely and uniformly spaced trees. Traditional treatments could be modified to promote more variable density structure in order to be more compatible with desired wildlife habitat or forest restoration objectives (North and Sherlock 2012, North and Stine 2012). In turn, current stand-level canopy cover targets could be met while creating variable canopy closure conditions.

Relaxing constraints on treatments within special management zones

Within experimental landscapes, constraints on management practices associated with special management designations could be relaxed to facilitate achieving broader landscape-scale objectives; these measures would require monitoring to evaluate whether they meet the objectives, and they would require adjustments if deviation from objectives or negative impacts were detected. Applying the principle of emulating natural disturbance regimes would likely benefit riparian areas and habitat for species of concern over the long term. A patchwork of special management designations (e.g., protected activity centers (PACs), home range core areas (HRCAs), den site buffers, and riparian conservation areas (RCAs)), has imposed constraints on forest practices, especially mechanical treatments. Because areas within primary habitat for species of concern may be at relatively high risk for uncharacteristically severe wildfire (Ager et al. 2012), treatments within such areas may aid their long-term conservation (Scheller et al. 2011). Designing a landscape strategy requires carefully considering opportunities to promote wildlife, riparian, and aquatic habitat values rather than avoiding such priority areas. This type of active adaptive management approach would benefit from a robust partnership involving management, research, and stakeholder groups.

Riparian areas

Landscape strategies that consider fire regime and topography interactions in designing treatments should be able to accommodate riparian area concerns (North 2012, Skinner et al. 2006). Management to promote resilience of small to medium stream reaches that historically burned frequently like adjacent uplands would facilitate similarly frequent, low- to moderate-intensity fire (see Forested Riparian Areas chapter (6.2)), rather than being set aside as unmanaged buffer zones that are more susceptible to high fire intensity. However, in riparian areas that function differently than uplands, other tactics are warranted due to higher soil moisture and stronger connectivity to aquatic systems. Experimental, scientifically-informed harvesting and burning techniques could illuminate new ways to improve riparian conditions and improve understanding of treatment effects on water flow, water quality, soils, riparian and aquatic biota, and impacts from wildfire. This approach would require consideration of large woody debris loading, shading, stream channel stability, and nutrient inputs, among other factors (Burton 2005). As part of a long-term adaptive management strategy, experiments in riparian areas would help to address a significant research gap.

Special wildlife management areas

Areas of special management have been designated for several wildlife species, for example, protected activity centers (PACs) for California spotted owl and den buffer areas for fisher. In these areas, management restrictions have been linked to site location and hazard levels (wildland-urban interface versus wildlands), but treatments have generally been more difficult to implement. In wildland settings, mechanical treatments have often been restricted or not allowed, and approved treatments (e.g., prescribed fire and hand removal of fuels) have often been limited to specific timeframes or prescriptions. PACs were originally designated as an interim measure, but they have become long-term zones with little to no management (Berigan et al. 2012). In some cases, their boundaries have been revised based on changes in conditions and long-term monitoring data (Berigan et al. 2012), but in other cases, unoccupied areas have remained set apart from the general forest matrix. Under an experimental approach, core wildlife areas could be included within landscape-scale treatment plans while continuing to emphasize conservation of habitat for particular species of concern. Abandoned areas and margins of core areas could be used in experimental treatments as a proxy for occupied habitats, offering insights on how the habitat responds to treatment. Use of robust modeling tools as part of an adaptive management framework may highlight the ways in which landscape-scale strategies can afford sufficient protections and promote long-term improvements in habitat to reduce reliance on exclusion zones. The focus needs to move beyond effects easily seen at the stand scale to effects that are not as easily seen at the landscape scale, but that can still be modeled and validated through monitoring. This approach may aid conservation and recovery of additional species of concern as research is better able to evaluate the quality and connectivity of their habitats.

Promoting future habitat and appropriate habitat connectivity

Where landscapes appear to have deficits of priority species habitat compared to the likely historical range of variation (HRV), plans could be developed to guide management activities to restore high-quality habitat (North 2012) following conservation approaches suggested by Thompson et al. (2011) and Spencer et al. (2011). Landscape-level restoration strategies could be developed to promote desired habitat conditions where they currently do not exist, using concepts and tools like the HRV, climate adaptation strategies, and scenario planning (Nydick and Sydoriak 2011, Peterson et al. 2011). Landscape strategies would include treatment designs that consider and promote habitat connectivity appropriate to the landscape, keeping in mind the potential undesirable effects of connectivity, such as unwanted spread of severe fire or invasive species. The maintenance of habitat connectivity would be an important consideration as treatments progress through the landscape and as forest conditions change with stand development. Landscape analysis tools that can evaluate multiple objectives are well suited to help managers evaluate these tradeoffs (see the Forest Carnivores chapter (7.1) for examples).

Applying landscape-scale modeling

Because experimentation is so costly, difficult, and slow, modeling will be an important component of developing and adjusting an adaptive management strategy. The “fireshed” modeling approach demonstrated the potential for spatial analyses to evaluate complex tradeoffs across large areas over many decades (Bahro et al. 2007). Although the term “fireshed” may imply an emphasis on fuels reduction, the intent of the approach is to focus thinking at the broader landscape scale at which fire operates rather than at a more limited project scale. In this sense, fireshed is analogous to watershed except it is based on the scale at which fire operates informed by fire history, fire regimes, topography, vegetation, expected fire behavior, and the risk of problem fires (Bahro et al 2007). The objective of fireshed assessment would be to develop plans that limit the risk of large, high-intensity fires while considering a broad array of values—including watersheds, viewsheds, smokesheds, wildlife habitat quality and connectivity, ecosystem services and other social and economic values—in an integrated approach at an appropriate landscape scale. Landscape-scale simulations suggest that these broad treatment strategies may benefit wildlife species (Scheller et al. 2011). Combining multiple tools may be necessary to assess treatment effects on the distribution of seral stages/structural types, associated habitat values, and connectivity through time at multiple scales. An example of an integrated approach applied the Forest Vegetation Simulator (FVS) tool to model effects at the smaller scale of fisher home ranges (Thompson et al. 2011), while designing fuels treatments based on landscape analyses (e.g., Fireshed, Flammap), local knowledge of prevailing winds, and the general direction of historical large wildfires. Modeling habitat for priority species at the landscape scale would allow projections of the future arrangement of dense patches, matrix, and openings based on treatments and wildfires under different management strategies.

Monitoring effects on species of concern

Though habitat modeling will be important to evaluate potential outcomes at large scales, species monitoring will also remain a significant part of an overall resilience strategy. Effects of treatments on current habitat conditions would need to be monitored to estimate how species of concern are likely to respond. Although many species appear to either benefit from or be indifferent to fuels reduction treatments (Stephens et al. 2012), other species associated with high canopy closure and high structural complexity may be negatively affected by conventional treatments. However, even these species persisted within landscapes that historically had considerable amounts of open forest conditions and early seral habitat created and maintained by frequent, low- and mixed-severity fires (Perry et al. 2011). The Synopsis of Emergent Approaches (1.2) and Forest Carnivores chapter (7.1) focus more on this issue. Again, management strategies that emphasize heterogeneity, in addition to robust monitoring in treated areas, may be able to account for and address the needs of different priority species.

Monitoring plans are expected to include both a coarse-filter approach to evaluate landscape-scale habitat patterns and ecological processes, and a fine-filter approach to ensure that at-risk species are being conserved. Integrating modeling with monitoring of field conditions can help to evaluate how ecosystems are changing at broad scales where experimentation may be impractical. Noon et al. (2012) recommended targeting a small number of species based on management objectives and the species' ecological roles, sensitivity to change, and conservation importance; however, they recognized that multiple species approaches, as described by Manley et al. (2004), are appropriate for species that can be detected using the same protocols (for example, breeding birds, small rodents, and mesocarnivores).

Phased Strategies for Long-term Treatment

Because integrated planning must assess and forecast treatment effects on a host of resource concerns across time scales, approaching these issues in phases may help reconcile short-term and long-term priorities. For example, attempts to restore a more natural disturbance regime of fire as an ecological process without first securing vulnerable communities and resources could have catastrophic outcomes. Accordingly, the first phase would emphasize fire hazard reduction in strategic areas because of the current vulnerability of many landscapes to high-intensity wildfire, whereas subsequent phases would integrate broader considerations.

Phase One: Strategic defensive fuels reduction

The initial phase of an integrated landscape treatment strategy in the Sierra Nevada and southern Cascades would emphasize reducing fire hazard in strategic areas, with the objective of securing a margin of safety for implementing longer-term strategies. Areas would be selected to impede the spread of high-intensity wildfire to protect life, property, and public safety, and reduce the likelihood of compromised habitat and other resource concerns. There has been important progress in conducting strategic fuels treatment, facilitated in some cases through the Fireshed planning effort (Bahro et al. 2007). But in other cases, projects may have been undertaken to opportunistically reduce fire hazard without necessarily being configured and sized to provide community defense and facilitate larger landscape restoration. Special zones such as Defensible fuel profile zones (DFPZ) or community defense zones (CDZ) require sufficient area and vegetation conditions to be defensible (Weatherspoon and Skinner 1996). They can be implemented as strategically placed treatments across the landscape or they can target areas with the greatest risks for life and property, such as wildland-urban interface areas (WUIs), roads, utility rights of way, and other areas where human-caused ignitions are problematic and likely to occur (Schmidt et al. 2008). Involvement and cooperation of local communities is important in implementing these treatments (Weatherspoon and Skinner 1996), and research suggests that public support in these defense zones is likely to be high where the threat of wildfire is evident (Bright and Newman 2006). Effectiveness of these treatments may increase where community members complete and maintain complementary treatments on adjacent private lands in the WUI.

Although strategic defense treatments within the WUI appear critical in facilitating a larger restoration approach, they do not necessarily constitute restoration, since their intent is typically to alter fire behavior to aid suppression activities and minimize area burned rather than to produce conditions that reestablish fire as an ecological process. Hence, this approach relies strongly upon a resistance strategy (Millar et al. 2007), with decisions based on suppression response times and other defensive considerations. Removal of surface and activity fuels may be facilitated in instances where there is an economically viable market for the residual biomass. However, mastication, which creates novel conditions at least in the short-term, may be warranted in some instances given economics and the difficulty of employing prescribed fire in defense zones. However, it should be recognized that restoration opportunities for wildlife and other resources would often not be incompatible, even in intensively managed areas (Eitzel et al. 2012), and would be important considerations because those areas also harbor important biodiversity (Manley et al. 2006) and provide wildlife-related ecological services, including recreational opportunities.

Phase Two: Reclamation treatments in a fraction of the landscape

A major challenge going forward is to go beyond the first phase of fire hazard reduction and pursue a long-term plan of restoring conditions where fire can be returned safely as a key process in the landscape. This goal can be facilitated through treatments in the first phase, by affording managers and communities greater confidence in using fire within areas that are bounded by fire-resistant zones of sufficient size and structure to facilitate containment. However, the emphasis in a second phase of treatments is to promote landscapes that inhibit the potential for high-intensity fire to burn across the landscape in uncharacteristically large patches. Treatments in this phase would promote stand conditions that are representative of reference stand structures and resilient under the foreseeable climate. Topography, vegetation, expected fire behavior, and resource concerns would guide development of treatment strategies (Skinner et al. 2006). Multiple landscape modeling studies suggest that if treatments are strategically placed, an initial target of treating 10-25 percent of the landscape within a period of 5-10 years can effectively reduce the likelihood of unacceptably large, high-intensity fires (Ager et al. 2007, Ager et al. 2010, Finney et al. 2007, Schmidt et al. 2008, Syphard et al. 2011).

Treatments in this phase would represent a mix of strategies to promote conditions where wildfire can occur without unacceptably severe outcomes. Some areas may require sequential treatments before they would achieve that condition (Skinner 2005). Treatments would have to include some fire component to be considered fully restorative, even though structural treatments will in many cases be needed before fire can be safely applied. To incentivize restoration, it would be appropriate to recognize and accord greater weight to treatments that come closer to facilitating hypothesized fire regimes, considering patterns of frequency, seasonality, and severity. As the natural role of fire is restored, the seasonality of burns will become more significant.

Economic considerations are particularly important in this phase, because treatments of areas that have not been harvested for many decades may provide resources to restore parts of the landscape where harvest costs are likely to exceed biomass revenues (Hartsough 2003, North 2012). After initial entries remove some of the more merchantable trees, future economic returns are likely to be much smaller. Consequently, opportunities to receive greater returns for smaller tree biomass will be important in accelerating the pace and extent of restoration treatments.

Phase Three: Maintenance and rotation throughout the landscape

In the third phase, maintenance treatments would be needed to keep the areas treated in Phase Two in a fire-resilient condition and on a path to a more resilient fire regime. This phase may also rotate treatments into other parts of the landscape to create a mosaic of conditions, depending on the need to maintain or restore important habitat features. This step could be guided using predictive habitat models that are integrated with silvicultural and fire models. Tradeoffs between treating new areas and maintaining existing areas would need to be considered using models to account for costs of treatment, changing fire conditions, and other factors (Finney et al. 2007). Though a variety of approaches will be needed to extend effectiveness of treatments, incorporating managed fire will likely be an important component of short- and long-term strategies for promoting resilience in forest ecosystems (North et al. 2012). As outlined in the Synopsis of Emergent Approaches (1.2), increased use of managed wildfire would promote ecological resilience, particular in the many areas that are inaccessible to mechanical treatments. In addition, wildfires will likely happen within large planning areas, alter the priorities for treatment across the landscape, and create opportunities to influence ecological trajectories. As a consequence, greater integration with post-wildfire treatment plans, both before and after such fires occur, is another important facet of a resilience-based landscape strategy (see Post-wildfire Management chapter (4.3).

Management Implications

- Strategic placement and phasing across the landscape using a combination of prescribed fire, managed wildfire, and silvicultural treatments can accomplish the following:
    - Shift disturbance regimes toward patterns that are more consistent with how ecosystems evolved and promote resilience to stressors such as climate change.
    - Reduce undesirable losses from the terrestrial, aquatic, and socioeconomic components of socioecological systems that result from uncharacteristically large, severe, and dangerous wildfires.
    - Promote important values for the long-term, including habitat needs for species of concern, favorable water flows, traditional cultural resources, forest products and associated livelihoods and infrastructure, and other ecosystem services and social benefits.
- Approaches for reestablishing historical processes within aquatic ecosystems, in addition to terrestrial treatments, can include restoring incised channels in wet meadows, removing introduced fishes from lakes, and promoting more natural stream hydrographs below dams.
- Development and implementation of these approaches through collaborative, placed-based efforts can strengthen existing community capacities and reduce vulnerabilities to major disruptions.

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1.2 Synopsis of Emergent Approaches

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This synopsis presents three integrated themes that emerged from synthesizing information about biological resources. These themes become particularly important when managing forests to promote resilience at large landscape scales and long time frames. This synopsis summarizes ideas in the longer Integrative Approaches chapter (1.1) using a concise style where definitions, citations and elaboration of some key points are included in endnotes.

The central emergent theme for promoting resilience is working with and adapting to dynamic ecological processes at larger scales. From this broad perspective, two integral themes emerge: 1) restoring fire as an ecological process and 2) meeting fire hazard reduction objectives while sustaining wildlife habitat and restoring riparian ecosystems. Furthermore, these themes need to address the cohesion of socioecological systems, not only by considering potential management effects on economic, social, and cultural components, but also through collaborative and adaptive management. Issues that are particularly important for integrating socioecological components include selecting appropriate scales for planning in particular areas (see Broader Context for Social, Economic, and Cultural Components (9.1)), how to design forest treatments in ways that benefit local communities (see Strategies for Job Creation through Forest Management (9.4) and Managing Forest Products for

Community Resilience (9.5)), and how to consider local and traditional ecological knowledge and promote participation in monitoring programs (see Collaboration (9.6)).

Managing Forests for Resilience

Increasing forest resilience[1] in the Sierra Nevada will require management strategies that work with and adapt to dynamic ecological processes at larger scales. Current practices often concentrate on containing fire, sustaining large trees, and preserving wildlife habitat, attempting to maintain stasis with stand-level management. This approach is fundamentally at odds with dynamics in fire-dependent forests and will constrain rather than facilitate an adaptive response to climate change. Management might be better guided by evaluating how well it restores heterogeneous forest conditions that are congruent with how site productivity and historical fire intensity affected local growth and mortality.[2]

Many ecosystems processes[3] are complex and difficult to measure, forcing managers and scientists to use surrogate assessments, such as structural condition or indicator species presence. In the Sierra Nevada, management has often used indicators derived from other forest ecosystems, particularly the Pacific Northwest (PNW), such as old-growth forest characteristics and spotted owl viability.[4] Indicators from other regions, such as PNW old forests, which have profoundly different disturbance regimes and climate, are unlikely to be congruent with the reference heterogeneity and dynamism of Sierra Nevada forests. Proactive management could recognize that current old growth and spotted owl nesting habitat will change and plan for creating these conditions in future forest landscapes.

An emphasis on these indicators has often focused management at the stand scale, which can then get bogged down in identifying optimal forest structure on an acre-by-acre basis. Terrestrial management at small scales may also disconnect treatment strategies from affecting watershed and aquatic ecosystem processes on a scale effective for restoration. When desired habitat is managed at the scale of individual parcels, it can lose sight of the major ecological processes (e.g., growth, mortality, disturbance) that will continue to shape the larger landscape. These dynamics can render forest plans with static structural and habitat goals obsolete by the time they complete public and administrative review.[5] The new planning rule directs national forests to embrace and accommodate ecosystem change.[6]

Research suggests that a prudent approach may be to increase forest landscape heterogeneity at multiple scales with management practices that mimic the structure and composition that might have been produced by historical, frequent fire disturbance.[7] Sierra Nevada managers have been experimenting with GTR 220 principles using existing stand conditions and topography as a template to vary treatments while meeting fire hazard reduction, wildlife habitat, and forest restoration objectives. This approach is consistent with recent research showing that topography, site productivity, and fire history interact to influence burn intensity and forest heterogeneity.[8] Many modern forests are relatively homogenous, with much higher stem density and canopy cover than existed under an active fire regime.[9] Unless treated, these conditions will limit forest resilience to drought and climate change. Management activities that reduce stem density and move forests toward the range of conditions that would result from natural interactions between frequent fire and varying site productivity are likely to improve landscape resilience to both acute (e.g., high-severity wildfire, drought, etc.) and chronic (e.g., understory burning, climate change, bark beetles, air pollution, etc.) disturbance.[10]

Management Implications

- Forests managed to be congruent with what potential fire behavior and site productivity would produce will be more in sync with the two dominant processes—growth and mortality—that fundamentally shape Sierra Nevada forests.[11]
- Practices suggested in GTR 220 and discussed in GTR 237 may help create these conditions and increase the landscape heterogeneity needed for resilient terrestrial and aquatic ecosystems.

Restoring Fire as an Ecological Process

Although changing climate may be a chronic stressor, wildfire is a major catalyst through which its effects will be expressed. Managing fire in contemporary forests riddled with human development has significant risks. Notwithstanding these concerns, restoration of fire as an ecological process is the most efficient means of promoting forest resilience and rejuvenating aquatic habitat in much of the Sierra Nevada. In addition, there are large portions of wildland landscapes (e.g., steep slopes, wilderness, roadless areas, etc.) where mechanical treatment is infeasible. Thinning will always be a substantial component of forest treatments, however, where possible, fire should be used rather than preemptively dismissed as impractical. To increase the pace and scale of fuels reduction and forest restoration, management may need to enlarge project areas and incorporate fire at large scales. This will involve expanding burn windows, and in some instances, targets for allowable fire-caused tree mortality.

Fire must always be controlled in areas near homes, but in much of the forested wildland, there are opportunities for wider use of fire for fuels reduction and forest restoration. Current rates of fuels reduction, even when wildfire is included regardless of severity, treat less than 20 percent of the area that may have burned historically each year in the Sierra Nevada.[12] Research suggests that outside of the wildland urban interface[13] (WUI), a more practical objective is to reduce adverse fire effects and intensity rather than occurrence and size.[14] A recent comparison of fire severity and size between Forest Service and Yosemite National Park lands found that the park's policy of allowing most lightning fires to burn relatively unimpeded under a range of fire weather conditions had achieved fire patterns that were closer to desired historical conditions.[15] The pace, scale, and restoration benefits of fire would be significantly increased if national forests identified large, contiguous blocks of forest to be treated, and then moved these blocks out of fire suppression to be maintained with prescribed and managed wildfire.[16] Outside of the WUI, forests could be zoned for a range of wildfire responses consistent with desired effects and made a priority for managed fire use.[17] More creative and flexible ways of working with fire could help achieve restoration objectives. Greater use of wildland fire will require continued interagency coordination (especially between land management and air quality regulatory agencies), strategic monitoring, robust science-management partnerships, and increased support of fire management programs from agency leadership and the general public. Approaches that focus primarily on containing fire through suppression, regardless of burning conditions, sacrifice opportunities for using fire for ecological benefits and promise more dangerous and more destructive fires in the future.

Management Implications

- In mid-elevation forests, use of frequent, low- and moderate-intensity fire is the most effective management practice for restoring forest resilience in the advent of climate change. Treatment prescriptions could often be guided by what is needed to restore fire to the area.
- Outside of the WUI, each national forest could zone areas for different fire responses (e.g., let burn and monitor, containment but not suppression, allow surface but not crown fire, etc.) under specified weather percentile conditions.
- More remote firesheds could be identified, fuels treated in strategic locations, and desired conditions maintained by prescribed fire and managed wildfire.

Reconciling Fuels Treatment, Wildlife Habitat and Riparian Restoration

Current practices and regulations make it difficult to manage forested landscapes for large-scale processes. Most forests have a patchwork of designations (e.g., under the Sierra Nevada Forest Plan Amendment protected activity centers [PAC], habitat conservation areas [HCA], and riparian conservation areas [RCA]), within which forest practices are limited. Management working within these constraints often becomes triage, opportunistically treating forests with the highest fuel loads where economics, management options, and stakeholder support allow. Proactive, integrated landscape management will be needed to effectively reduce fire hazard while providing immediate and long-term wildlife habitat and restoring riparian ecosystems.

Recent success with several collaborative projects in the Sierra Nevada, including the Collaborative Forest Landscape Restoration Program (CFLRP) and Integrated Regional Water Management Plans (IRWMP), highlights some guidelines for moving forward. First, successful large-scale management efforts could be based on a collaborative process.[18] A key to successful collaboration is that participants define a desired condition and identify immediate and long-term objectives.[19] Second, science-based monitoring and its active incorporation in adjusting management practices are essential both for course correction and as a means of building trust and cooperation. A third guideline is the inclusion of rural community concerns and economic factors in decision making. Landscape management requires public support, and for long-term viability, a self-sustaining economic base. Projects that cultivate local community involvement and plan for generating sufficient revenue to support long-term management objectives could fund and build monitoring capacity that better accomplishes large-scale restoration.

Although GTR 220 principles suggest a general approach, optimal management of a landscape for all wildlife species while reducing fire hazards is still in a developmental stage. Using a fine-filter approach, current policy focuses on sensitive species and is weighed toward maintaining and creating high canopy cover, old-forest conditions. This focus does not adequately consider the habitat needs of a broader range of species and the shifting dynamics in frequent-fire forests. Management aimed at restoration of dynamic, large-scale processes that produce a range of vegetation conditions similar to those under which Sierra Nevada ecosystems evolved should help to conserve coarse-scale terrestrial and aquatic biodiversity. For terrestrial wildlife this approach would include developing variable habitat conditions for species associated with different seral stages from primary disturbance conditions (i.e., black-backed woodpecker and post-fire habitat associated species), to early-succession (fox sparrow, deer, etc.) through old forest conditions, and the diversity of prey species upon which top trophic predators depend. For riparian ecosystems, reductions in forest density and cautious fire use[20] could enhance soil water balance and help restore stream microclimate, nutrient and sediment processes that support aquatic diversity. The heterogeneity of conditions within Sierra Nevada riparian areas suggest riparian zones be defined using scalable widths based upon soil moisture, geomorphic settings, and other local landscape characteristics.

As management transitions toward a coarse-filter, all-species approach, sensitive species populations presently at risk still need to be maintained or increased. A prudent approach may be to first 'buy time' by strategically treating fuels outside of sensitive species core habitat so as to contain or reduce severity of likely wildfires. The second, often-uncompleted phase is to more broadly implement fuels treatments such that large trees, snags, and areas of high canopy closure are maintained during a large-scale burn, including within core habitat (i.e., PACs and rest sites). Treatments would need to be implemented gradually while sensitive species are monitored (see footnote[21] for a suggested approach). Recently developed fisher and spotted owl habitat models[22] can be used to evaluate different management alternatives and their potential influence on current and future habitat conditions. This would include designing and maintaining habitat connectivity[23] across a dynamically changing landscape.

Management Implications

- Consider establishing experimental, demonstration landscape laboratories around the northern and southern Sierra Nevada owl demographic areas. This will significantly reduce costs and 'startup' time because long-term baseline data are available (including fisher monitoring and instrumented watersheds in the southern Sierra) and research is already active at these sites.
- Using the CFLRP as a model, create a collaborative team of managers, stakeholders, and science consultants to draft a desired condition for each target landscape and a road map for how to get there. This may include the option, if agreed upon, of removing or modifying special land use designations within the experimental area.
- Using the collaborative team and the Regional Ecology program, develop a science-based, question-driven monitoring program.[24] Management could adapt by incorporating updated information.
- All management actions would be considered experimental and subject to initial evaluation against the best available sensitive species habitat and fire behavior models, and longer term evaluation from monitoring results.
- For long-term sustainability, many projects will need to generate their own revenue and support local economies. Without compromising ecological integrity, economic and social factors need to be explicitly included in planning, monitoring, and management to ensure long-term viability.

Endnotes

[1]Definitions of resilience have evolved as the concept has been adopted and more widely employed in ecology (e.g., Holling 1973, Walker et al. 2004. Folke et al. 2004). This synopsis uses the ecological (distinguished from the engineering) definition proposed by Holling (2010): "resilience is the magnitude of disturbance that can be absorbed before the system changes its structure by changing the variables and processes that control behavior."

[2]Recent research (Falk et al. 2006a) has stressed that restoration efforts should first assess whether structure, composition, or process measurements may provide the most efficient, albeit indirect, measure of ecosystem condition. A pattern in ecosystems with frequent disturbance regimes (e.g.,

fluvial plains, fire-dependent forests) is that measurements of disturbance processes are often the most effective metrics of restoration (Falk et al. 2006b).

[3]Ecological processes can be both biotic (e.g., competition, growth, nutrient cycling, etc.) and abiotic (e.g., fire, erosion, flooding, etc.). A general definition is "the physical, chemical and biological actions or events that link organisms and their environment." In many Sierra Nevada forests, the processes that appear to most strongly influence forest structure and composition are fire and site productivity (Lydersen and North 2012). Managing forests so that the conditions produced are congruent with these two processes is likely to maintain and restore other ecosystem processes that are much more difficult to assess.

[4]Management practices in the Pacific Northwest, such as clearcutting, truncated forest seral stages, largely eliminating the long disturbance-free period of old forest conditions. This reduced the amount of forest containing large structures and deep, multi-layered canopies, putting those conditions and the species associated with them at risk. Management practices in the Sierra Nevada also reduced the number of large trees in many areas, raising concerns for sensitive species associated with these stand attributes. Practices in the Sierra Nevada, however, often did not remove all large, old structures or completely reset forest seral stage. Perhaps a more pervasive management impact has been largely eliminating low-intensity fire, putting frequent change and the forest heterogeneity it produced at risk. The seral stage most imperiled in Sierra Nevada forests is that created by frequent, low-intensity fire.

Managers and scientists still have much to learn about frequent-fire forests in the Sierra Nevada by looking to active-fire regime landscapes within its borders (e.g., Illilouette Basin in Yosemite National Park, Sugarloaf Basin of Sequoia and Kings Canyon National Parks, Beaver Creek Pinery in the Ishi Wilderness) (Collins et al 2007, 2008; Collins and Stephens 2010; Taylor 2010) and to the south (i.e., Sierra San Pedro Mártir in Mexico) (Stephens 2004, Stephens and Fry 2005, Stephens and Gill 2005, Stephens et al. 2007). The research that has come from these areas is probably more directly applicable to Sierra Nevada forest dynamics than some of the information from the infrequent disturbance and relatively mesic conditions of Pacific Northwest forests west of the Cascade Mountains.

[5]Wildfire, beetle mortality, and drought stress often change forest condition, but change in wildlife habitat designations do not always follow suit. For example, managers can not decommission or retire PACs once they are established if there has not been "significant" change to the habitat, even if the PAC becomes unoccupied by owls. There is currently no threshold that defines "significant" change, leaving it unclear whether the designation should remain after moderate changes to habitat conditions that are common in dynamic ecosystems.

[6]Final Planning Rule, Section 219.8 (Sustainability): "The plan must provide for social, economic, and ecological sustainability within Forest Service authority and consistent with the inherent capability of the plan area, as follows… iv) System drivers, including dominant ecological processes, disturbance regimes, and stressors, such as natural succession, wildland fire, invasive species, and climate change; and the ability of terrestrial and aquatic ecosystems on the plan area to adapt to change."

[7]See North et al. (2009) and North (2012).

[8]See Lydersen and North (2012).

[9]An example: by one estimate (Forest Service Westcore data), tree density on Forest Service land averages 280 stems/ac. In contrast, Lydersen and North (2012) found stem densities ranging from 45 to 134 stems/ac on ridge and lower slope stands, respectively, in old-growth mixed conifer with restored fire regimes. They also found canopy cover ranged from 19 to 49 percent on ridges and midslope stands, respectively. We're not aware of any estimate of average canopy cover for the Sierra Nevada, but observation suggests current conditions are usually much higher and lack spatial variability.

[10]Betancourt (2012) suggests that landscape heterogeneity decreases the probability of synchronous high-intensity disturbance over large scales. In frequent fire forests, some processes (e.g., seed dispersal and microclimate amelioration) and forest conditions (plant and animals that require undisturbed refugia) may not be resilient to large increases in the patch size of high-severity fire.

[11]There is not a single structural condition that would always be produced by a set fire behavior. Rather, variation in weather and fuel conditions at the time of burn is likely to produce a range of outcomes that would give management general bounds within which to define a desired condition.

[12]See North et al. (2012).

[13]In the Healthy Forests Restoration Action of 2003, the WUI is defined as up to 1.5 miles from communities at risk or as defined in individual community fire protection plans. In general, the WUI is divided into two zones. The Defense Zone is the "area directly adjoining structures (i.e., 0.25 m) and evacuation routes that is converted to a less-flammable state to increase defensible space and firefighter safety. The WUI Threat Zone is an additional strip of vegetation modified to reduce flame heights and radiant heat. The Threat Zone generally extends approximately 1.25 miles out from the Defense Zone boundary. Actual extents of Threat Zones are based on fire history, local fuel conditions, weather, topography, existing and proposed fuel treatments, and natural barriers to fire and community protection plans, and therefore could extend well beyond the 1 1/4 mile."
The two zones are discussed in more detail at:
http://www.fs.usda.gov/detail/cleveland/landmanagement/planning/?cid=fsbdev7_016495

[14]See Reinhardt et al. (2008).

[15]See Miller et al. (2012).

[16]Recent research (Ager et al. 2012b) has developed models for optimizing fuels treatment locations across a landscape to facilitate managed wildfire and prescribed fire use, rather than the traditional allocation designed to aid suppression.

[17]Aplet (2006) suggests creating three zones with different levels of fire response: a Community Fire Planning Zone (analogous to a WUI), a Restoration Planning Zone, and a Fire Use Emphasis Zone. Dellasala et al. (2004) also suggested a similar three-zone approach.

[18]Collaborative teams by definition strive for consensus. However, it is not always possible to get 100 percent agreement. Effective and efficient collaboration may hinge on eventually voting on some issues and then moving forward following the majority's intent (Bartlett 2012).

[19]There are several collaborative groups that have made significant progress and can provide practical lessons, including all three CFLRPs, the Dinkey Landscape Restoration, the Amador-Calaveras Consensus

Group Cornerstone, and the Burney-Hat Creek Basin. Sagehen Experimental Forest has also had tremendous success with their collaborative efforts, including the implementation of demonstration plots to help visualize treatment options.

[20]Recent research suggests that riparian forests on many 1st and 2nd order streams in the Sierra Nevada may have had fire regimes comparable to adjacent uplands (van de Water and North 2010, 2011). This reinforces the idea that riparian areas should not be set aside when designing landscape-level treatments.

[21]Data from ongoing fisher research suggests a hypothesis to examine in these collaborative landscapes that is also detailed in the "Information Gaps" section of the Forest Carnivores chapter (7.1) of this synthesis. Fishers can tolerate some disturbance since they evolved in frequent-fire forests, but at present there is little information on the length of time and areal extent of habitat disturbance that can be tolerated. Research on the Sierra National Forest (C. Thompson and K. Purcell, unpubl. data), has surveyed several study areas equivalent to the average female fisher home range size (approximately 5 mi^2) and found that occupied areas include 2.2-16.9 percent of the area having some form of 'treatment' over 3 year periods (a reasonable estimate of fisher generation time). 'Treatments' included a combination of prescribed fire, salvage logging, and other forms of timber harvest. These data suggest fishers may tolerate a disturbance (fire or surrogate treatment) over about 10 percent of a 5 mi^2 area over a 3 year period (i.e., about 320 acres/5 mi^2 area/ 3 year period). Any treatment targets should also account for the suitability of habitat patches to be treated, as well as their contribution to habitat as a whole within the species' home range. These estimates align fairly well with the lower range of treatment area/year expected to reduce the likelihood of large, high-intensity fires (10-25 percent of a landscape over a 5-10 year period [Ager et al. 2007; Finney et al. 2007, Schmidt et al. 2008; Ager et al. 2010; Syphard et al. 2011]). This rate and extent of disturbance in fisher habitat may be worth testing in these experimental landscapes. Such a rate may lie within the tolerance of fishers for forest disturbance, especially since the fraction of the landscape needed to reduce wildfire risk would only partially coincide with occupied fisher areas.
A similar data-driven hypothesis might be developed for California spotted owls. Ongoing research on the Plumas, Lassen, Eldorado and Sierra National Forests has long-term demographic data in owl use areas that have experienced a range of wildfire severities and mechanical treatments.

[22]See Gaines et al. (2010), Zielinski et al. (2010), Thompson et al. (2011), and Ager et al. (2012a).

[23]Forest conditions that facilitate landscape connectivity vary between species, making it difficult to plan and manage 'corridors' for an array of wildlife. Certainly riparian areas should play some role, as research suggests that even under an active fire regime, historical riparian forests had higher stem density and canopy cover than upland forests (van de Water and North 2011). To maintain this high-cover corridor and avoid wildfire wicking, riparian forests could become a priority for light fuels treatment (i.e., surface and small ladder fuels). A more sophisticated approach, albeit focused on forest carnivores, is the multiple species habitat connectivity modeling that is nearing completion (collectively for fisher, marten, wolverine, and Sierra Nevada red fox) (Spencer and Rustigian-Romsos 2012). This effort received input, over several years, from species and connectivity modeling experts. Another more explicit modeling approach of note is the California Essential Habitat Connectivity Project (Spencer et al. 2010), which produced a coarse level of wildlife connectivity statewide.

[24]For developing an ecological monitoring program, one place to start would be to revisit and update the science-based, question-driven monitoring program in Appendix E of the Sierra Nevada Framework to

more fully address the social dimensions of socioecological resilience. Monitoring that evaluates the effects of management decisions on socioecological resilience might 1) reflect relevant ecological, social, and economic processes in a "triple bottom line" framework; 2) use metrics that are quantifiable, reasonably available to managers, and within the scope of management influence; 3) incorporate concerns of scientists, managers, and local experts; and 4) are linked to potential changes in management based upon the results of monitoring.

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1.3 Synopsis of Climate Change

Angela Jardine and Jonathan Long

Introduction

Changes in climate can interact with other stressors to transform ecosystems and alter the services those ecosystems provide. Scientific observations of climate variations in air temperatures and precipitation (type and quantity), for example, have been directly linked to changes in stream flows (amount and timing), fires (frequency and severity), and ecosystem structure and function over the past several decades. Future climate scenarios suggest a strong likelihood for increased exposure of socioecological systems to wildfire, droughts, intense storms, and other natural disturbances. This synopsis presents themes that run through the synthesis report regarding the impacts of a changing climate on the forests and waters of the Sierra Nevada and southern Cascades as well as long-term, large-scale, science-based strategies to promote system resilience to those impacts.

Observed and Predicted Climate Change in the Synthesis Area

Climate refers to the long term weather patterns for a given region. Observations of the components of climate (i.e., precipitation, temperature, humidity, sunshine, wind velocity, and associated weather phenomena, such as fog, frost, and hail storms) require a minimum monitoring period of 30 continuous years to reveal changes in climate (NASA 2005). Across the southwestern United States (California, Nevada, Utah, Arizona, Colorado, and New Mexico), temperatures since 1950 are reported to be the warmest in the past 600 years, with average daily temperatures in the most recent decade (2001 – 2010) being higher than any other decade since 1901 (Overpeck et al. 2012). Likewise, the spatial extent of drought from 2001 – 2010 covered the second largest area observed for any decade since 1901, and total streamflows in the four major drainages of the Southwest (Sacramento/San Joaquin, Upper Colorado, Rio Grande, and Great Basin) fell 5 percent to 37 percent below the 20th century averages during the 2001 – 2010 decade (Overpeck et al. 2012).

In the Sierra Nevada, warming temperatures since the 1980s are generally attributed to increasing nighttime minimum temperatures across the region; however, different elevations have experienced a range of temperature changes. For example, the annual number of days with below-freezing temperatures in higher elevations is decreasing, whereas the number of extreme heat days at lower elevations is increasing (Safford et al. 2012). Changing temperatures combined with elevation differences influence the type of precipitation received in the Sierra Nevada, which in turn greatly impacts regional hydrology and fire vulnerability.

Observations show an increase in the proportion of precipitation falling as rain instead of snow since the 1980s (Safford et al. 2012, Harpold et al. 2012); this change has manifested in spring snowpack decreases of at least 70 percent across the lower elevations of the northern Sierra Nevada, a trend that has not yet been observed in the higher elevation southern Sierra Nevada. By 2002, snowmelt was beginning 5 to 30 days earlier than what was typical in 1948, and peak streamflows were occurring 5 to 15 days earlier. These changes have, in effect, extended the fire season in the Sierra Nevada, particularly in low- to mid-elevation conifer forests (Safford et al. 2012). A longer fire season, associated with earlier drying and more cured fuels, has resulted in increases in the size and intensity of wildfires across the Western United States in general and the Sierra Nevada and southern Cascades specifically (Westerling et al. 2006, Miller et al. 2012, Safford et al. 2012, Miller and Safford 2013). These changes are a primary concern for forest and water resource managers across the synthesis region.

Furthermore, across the southwestern United States, seasonal and average annual temperatures are predicted to continue to rise throughout this century, and average annual and spring precipitation are projected to decline. This trend projects further decreases in mountain snowpack, earlier snowmelt and peak streamflows, and greater drought severity (Overpeck et al. 2012). Within the Sierra Nevada, models project a decrease in mountain snowpack of at least 20 percent and up to 90 percent (Safford et al. 2012) over the next century. This prediction is a major concern for water resource managers, who are already trying to balance various demands for water during periods of low flows. Flood potential is predicted to increase for high-elevation, snow-fed streams; this change is due to shifts toward earlier peak daily flows (driven by increasing temperatures), coupled with an increased proportion of precipitation falling as rain instead of snow (Das et al. 2011, Overpeck et al. 2012, and Safford et al. 2012).

Approaches to Promote Resilience to Climate Change

The Integrative Approaches chapter (1.1) summarizes strategic approaches to meet the challenges of promoting socioecological resilience. Current climate change impacts and those predicted to occur in the not-too-distant future challenge the ability to protect natural resources now and in the long term. The climate conditions that allowed ecosystems to thrive in the past are changing and in some cases, rapidly so. The following bulleted points highlight concerns, strategic approaches, and research needs from various chapters of this science synthesis that focus on climate change. Readers are encouraged to review the chapters listed at the end of each example (and the references therein) for greater detail.

- ***Recognize and address scale mismatches****—the temporal, spatial, and functional scales of management systems may not be well matched to the scales of environmental variation.*

Forest ecology research has concentrated on small spatial and temporal scales, however, effective planning for climate change should consider long temporal periods and large spatial scales to account for widespread changes in disturbance regimes. Designing treatments at larger scales allows strategies to better account for landscape-scale processes, such as wildfires and insect outbreaks, as well as species that have large ranges. **(Integrative Approaches 1.1)**

- ***Consider long-term (>50 years) risks*** *in addition to short-term (<10 years) expected outcomes.*

Large old forest structures, which provide vital habitat for a variety of fauna in the synthesis area, take decades or centuries to develop; landscape plans should promote recruitment of these habitat features by increasing growing space and reducing the risk of high-intensity fire. **(Forest Ecology 2.0; Forest Carnivores 7.1; and California Spotted Owl 7.2)**

Many of the ecological services afforded by mountain meadows are threatened by a warming climate, and these vulnerabilities appear to be particularly high in several central Sierra Nevada watersheds, including the American, Mokelumne, Tuolumne, and Merced. Restoration efforts in these systems may help to delay runoff and increase summertime low flows. **(Wet Meadows 6.3)**

Climate change can interact with wildfire to alter post-wildfire flooding and debris flows, which can have significant downstream impacts, including accelerated filling of reservoirs and other effects on water supplies, as well as significant and lasting impacts on vulnerable and isolated aquatic populations (**Watersheds and Stream Ecosystems 6.1).** Such debris flows can be difficult to mitigate—few options exist beyond reducing the potential for severe fires. (**Post-wildfire Management 4.3**)

- ***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.*

Eighty-five percent of known California spotted owl sites occur in moderate- or high-risk fire areas in the Sierra Nevada. Uncertainty exists regarding how increasing trends in the amounts and patch sizes of high-severity fire will affect California spotted owl occupancy, demographics, and habitat over longer time frames. Barred owls have replaced or displaced northern spotted owls over large areas of their range. Management needs to consider effects of multiple stressors on at-risk species, especially since conditions may change which options are prudent or feasible over time. **(California Spotted Owl 7.2)**

In the face of climate change, proactive conservation strategies for trout and amphibians should consider not only direct effects of climate, such as ameliorating high temperatures or low flows, but also reducing interactions with introduced species and other stressors. In some situations, there may be enhanced opportunities to deal with introduced species as climate change or wildfires alter conditions. (**Watersheds and Stream Ecosystems 6.1** and **Lakes 6.4**)

- ***Rely more on process-based indicators than static indicators of structure and composition, while recognizing that restoration of structure and process must be integrated.***

Sierra Nevada managers have been experimenting with GTR-220 principles, using topography as a template to vary treatments while meeting fire hazard reduction, wildlife habitat, and forest restoration objectives. Although climate change may be a chronic stressor, the catalyst through which its effects will be expressed is likely to be wildfire. **(Synopsis of Emergent Approaches 1.2)**

Manipulation of current forests to resemble historical forest conditions may not be the best approach when considering future climates. Rather, a prudent approach for maintaining forest ecosystems is to restore key processes such as wildfire that have shaped forest ecosystems for millennia, and associated structure and composition that are resilient to those processes and aid in their restoration. **(Fire and Fuels 4.0)**

Climate patterns strongly influence soil development and nutrient cycling processes. As elevation and precipitation increase, soil pH and base saturation tend to decrease due to greater leaching and decreased evapotranspiration. **(Soils 5.0)**

Climate change effects on flood regimes could alter sediment storage in floodplains, terraces, and colluvial hollows, which would in turn influence channel stability. Climate change is also expected to diminish summer low flows that could threaten aquatic life, especially cold water species. (**Watersheds and Stream Ecosystems 6.1)**

Because foundational ecological processes, such as soil water storage and vegetation evapotranspiration, may not have explicit targets, there may be a tendency to undervalue—or even ignore—them in decision making. Forest treatments have the potential to enhance system resilience to multiple stresses by reducing evapotranspiration and increasing soil water availability. In addition, such treatments have the potential to enhance the yield, quality, and timing of downstream water flows and resulting ecosystem services. (**Watersheds and Stream Ecosystems 6.1**)

- ***Integrate valuation tools, decision-making tools, modeling, monitoring, and, where appropriate, research*** *to evaluate responses and better account for the risks and tradeoffs involved in management strategies.*

Climate change may become a chronic stress in red fir forests in the lower parts of their present distribution; reductions in the extent of true fir forests could be particularly detrimental to martens; consequently, the potential influence on terrestrial and aquatic ecosystem processes in the fir zone constitute an important cross-cutting research gap. **(Forest Ecology 2.0 and Research Gaps 1.4)**

There is broad consensus for using common garden experiments or provenance tests to prepare for projected conditions by better understanding how genetic variability can improve ecological restoration. (**Forest Genetics 3.0**)

Rigorous assessment of the effects of future climate change on spotted owls will require dynamic models that incorporate vegetation dynamics and effects of competitor species. **(California Spotted Owl 7.2**)

Further research is needed to evaluate how nitrogen deposition and ozone affect carbon sequestration both aboveground and in the soil. This information will be critical to climate change mitigation efforts in the region. During severe fires, accumulated nitrogen in vegetation, litter, and surface soils will be released, and both thinning and prescribed fire can be used to proactively reduce the amount of plant matter available for combustion. However, long-term ecosystem protection and sustainability will ultimately depend on reductions in nitrogen deposition, and this is the only strategy that will protect epiphytic lichen communities. **(Air Quality 8.0)**

- ***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.*

Watersheds in the northern Sierra Nevada may be most vulnerable to decreased mean annual flow, south-central watersheds to changes in runoff timing, and the central portion to longer periods of low flow. Although the Kern River may be the most resilient watershed, the anticipated shifts in the hydrologic cycle will impact spring and summer water-based recreation and tourism and, more importantly, the California communities that depend heavily on Sierra Nevada water supplies. Projections highlight the importance of planning for increased flood events and developing contingency plans that provide approaches to mitigation and adaptation. (**Watersheds and Stream Ecosystems 6.1; Broader Context for Social, Economic and Cultural Components 9.1; and Sociocultural Perspectives on Threats, Risks, and Health 9.3**)

- ***Use participatory and collaborative approaches to facilitate adaptive responses and social learning****.*

Rural communities in the U.S. tend to be more vulnerable to climate change than urban communities, and people residing in the wildland-urban interface are particularly vulnerable to fire, making the concept of community resilience especially relevant in these contexts because of its focus on a community's ability to cope with change. **(Strategies for Job Creation through Forest Management 9.4)**

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1.4 Research Gaps: Adaptive Management to Cross-cutting Issues

Jonathan Long, Carl Skinner, Malcolm North, and Lenya Quinn-Davidson with numerous contributions from the Science Synthesis Team

Contents

Building on Adaptive Management Efforts

A number of studies within the synthesis area have been designed and implemented to better understand both more immediate and long-term effects of treatments, including the Blacks Mountain Ecological Research Project (Oliver 2000), Goosenest Adaptive Management Area Project (Ritchie 2005), National Fire and Fire Surrogate Study (McIver and Fettig 2010), Long-Term Soil Productivity Study (Powers 2006), the Teakettle Experiment (North 2002), the Kings River Experimental Watersheds (KREW) (Hunsaker and Eagan 2003), and the Sierra Nevada Adaptive Management Project (SNAMP) (Figure 1 and Table 1). Each of these studies has had difficulty maintaining funding after initial implementation, so the resulting information from the studies has been limited to responses over relatively short time periods. In a few situations, researchers have been able to study some long-term questions by taking advantage of a well-designed study that had been dormant or abandoned for some time but had been well archived by the original researchers (Dolph et al. 1995, Knapp et al. 2012). Such examples provide a valuable precedent for future research.

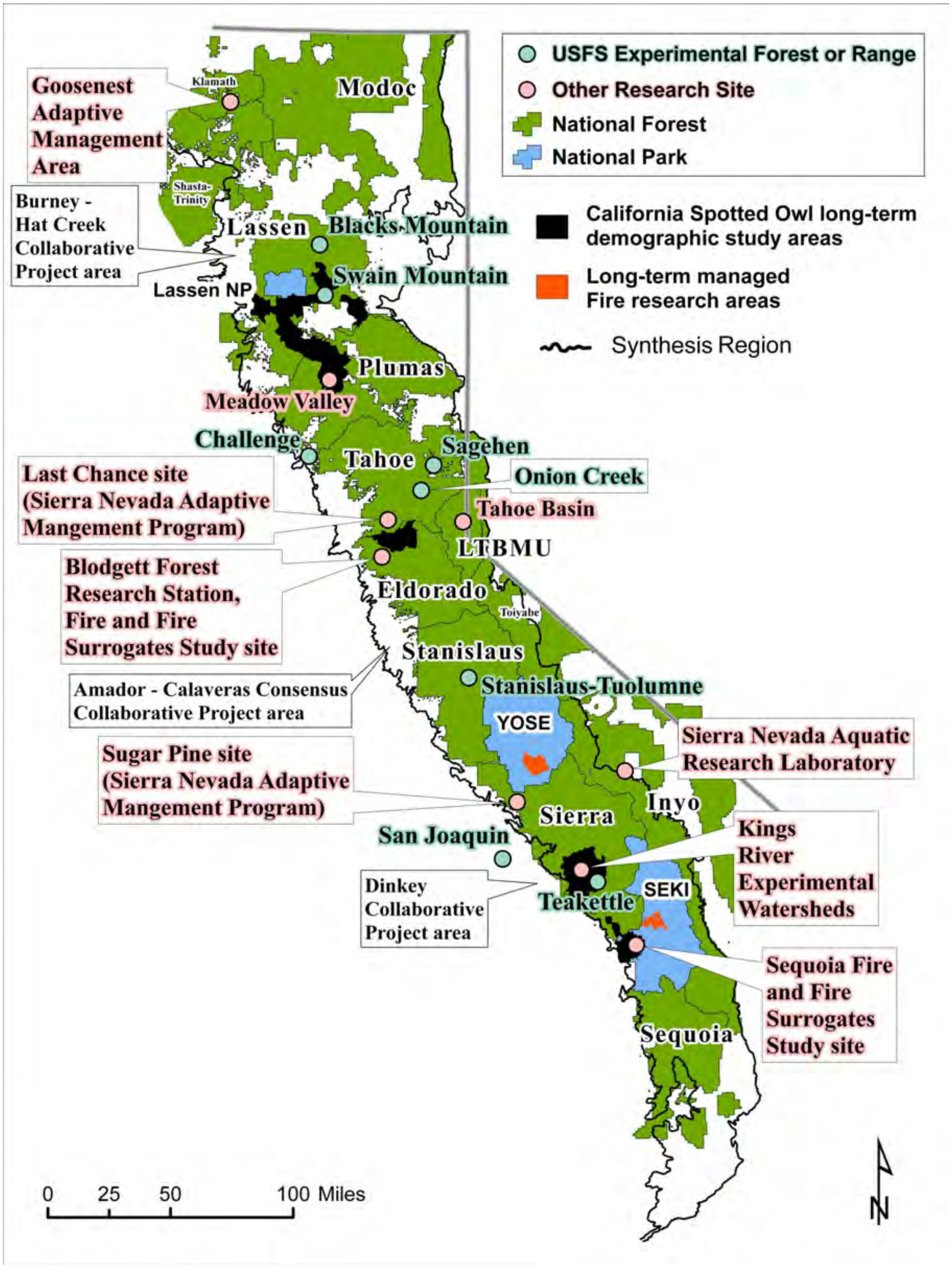

Figure 1: Map of experimental areas in the Sierra Nevada and southern Cascade Range, highlighting various adaptive management projects featured in this synthesis.

Table 1: USDA Forest Service Experimental Forests and Adaptive Management areas within the Sierra Nevada and Southern Cascades Synthesis Area

Research Area	Acres	Hectares
Blacks Mountain Experimental Forest	9,180	3,715
Challenge Experimental Forest	3,573	1,446
Goosenest Adaptive Management Area	172,000	70,000
Goosenest Ecological Research Study Area (also the Southern Cascades Site of the National Fire and Fire Surrogates Study)	3,000	1,200
Kings River Experimental Watersheds	46,604	18,860
Onion Creek Experimental Forest	2,965	1,200
Sagehen Experimental Forest	20,016	8,100
Stanislaus-Tuolumne Experimental Forest	1,500	607
Swain Mountain Experimental Forest	6,158	2,492
Teakettle Experimental Forest	3,212	1,300

Progress made on previous research topics will help to inform development of landscape strategies. Synthesis of research on the effects of forest fire hazard reduction treatments suggests that the threat of high-intensity fire can be significantly reduced with relatively benign impact on most wildlife species at project scales (Stephens et al. 2012). However, many of the less common species have not been a focus of study primarily because of statistical limitations with studying small populations or large home ranges. Most research associated with priority species, such as California spotted owl, has been on the small mammals that are their prey. Treatment areas within these studies have often not been large enough to make strong inferences about species with large home ranges, or to address patterns of habitat suitability and connectivity at the landscape scale. Additionally, these research projects generally have not been in place long enough to evaluate long-term effects. Adaptive management studies must overcome the challenge of maintaining long-term capacity and resources in order to promote social learning and system resilience.

Many experimental forests and other areas dedicated to adaptive management experiments are offering opportunities to improve understanding of how to design strategies to restore forests. A number of other large projects have been supported through the Collaborative Forest Landscape Restoration Program, including the Dinkey Landscape Restoration Project, the Amador Calaveras Consensus Group Cornerstone Project, and the Burney-Hat Creeks Basin Project. In particular, the Dinkey project is examining effects on fishers, and it has already yielded insights regarding approaches to promote successful collaboration (Bartlett 2012). A number of these projects are discussed throughout chapters of this synthesis; the following examples highlight projects in Forest Service experimental forests and adaptive management areas (Figure 1 and Table 2).

The Teakettle Experiment helped to understand the ecological effects of widely used forest treatments, such as understory and overstory thinning with and without prescribed fire (North 2002). Dozens of studies examined how these treatments affected different ecosystem components. Collectively, the research suggested that in fire-suppressed mixed-conifer forest prior to treatment, many ecological processes were impeded by competition for limited soil moisture and uncharacteristically high fuel and duff loading (North and Chen 2005). After treatment, patchy heterogeneity of forest conditions was associated with the greatest increases in species diversity and restoration of ecosystem functions (Ma et al. 2010, North et al. 2007, Wayman and North 2007). The researchers concluded that fire was essential to restoring many ecological processes but that understory thinning could play an important role in facilitating greater variability in burn effects and post-treatment forest heterogeneity (North 2006).

The objectives of several other experimental areas were to test particular hypotheses. For example, the Challenge area is included in the Long-term Soil Productivity study (LTSP), which looks at effects of different treatments on long-term soil productivity. Currents efforts on the Sagehen Experimental Forest are looking at the ecological effects of strategically placed treatments on a landscape. In addition, the Sagehen fuels reduction project was planned to protect and restore forest landscape heterogeneity, reduce fuels, and maintain and restore habitat for the Pacific marten. The consideration of habitat for a rare forest carnivore, early in a collaborative planning process, was viewed as key to the favorable prognosis for this project. Monitoring of martens and forest conditions as the treatments are implemented and beyond will help evaluate whether the expected outcomes develop.

The high diversity treatment at Blacks Mountain Experimental Forest (BMEF) is one of the most established efforts to study heterogeneity in forest structure and fuels through variable density thinning based on species composition and other factors. The primary objective of the BMEF study was to compare differences in ecological effects between stands treated for high structural diversity and stands treated for low structural diversity (Oliver 2000). Comparisons between the two types of treatments at the BMEF showed a large difference in short-term financial returns; the high diversity treatment yielded less revenue because it maintained most large legacy trees, whereas the low diversity alternative was based on a prescription that cut most larger trees while maintaining intermediate sized trees (Hartsough 2003). However, both types of treatments reduced fire behavior considerably when affected by an otherwise severe wildfire (Ritchie et al. 2007, Symons et al. 2008).

Treatments in the Stanislaus-Tuolumne Experimental Forest were designed to mimic historical reference conditions, including density, species composition, and age distribution (Knapp et al. 2012). To achieve these objectives, treatments based on stand conditions described by original detailed data and old maps have extended into some riparian areas and moved away from diameter caps in favor of selecting trees for removal to achieve the desired structure and spatial pattern. Future analyses of these treatments will facilitate a number of important comparisons of the effects of creating different stand structures to achieve both restoration and fire hazard objectives, including a comparison of timber volume and economic returns between variable density thinning and a more conventional even density thin. An extension of this research is examining effects of the experimental treatments on water yield.

The Goosenest Ecological Study Project and associated Fire and Fire Surrogate Study was focused on finding ways to accelerate late successional conditions through mechanical thinning and prescribed fire, including a comparison of a treatment that emphasized retention of pine trees and an alternative that emphasized retention of any large trees including firs; the study assessed treatment effects on fire hazard, vegetation, soils, small mammals, beetles, and birds.

Research Gaps

Appendix E of the Sierra Nevada Forest Plan Amendment provided a list of priority questions for monitoring and research to support adaptive management. Several of those priorities have yielded outcomes highlighted in this synthesis, including, but not limited to the following:

- Effects of fuel treatments on wildfire risk reduction (Fire and Fuels chapter (3.0));
- Continuation of watershed research at the Kings River Experimental Watershed (Watersheds and Stream Ecosystems chapter (6.1));
- Expansion of aquatic invertebrate monitoring (Watersheds and Stream Ecosystems chapter (6.1));
- Study of grazing effects on amphibians (Wet Meadow chapter (6.3));
- Expanded monitoring of fishers, particularly in the southern Sierra (Forest carnivores chapter (7.1)); and
- Continuation of the owl demographic study (California spotted owl chapter (7.2)).

Other topics recommended in that Appendix are likely to have been under addressed. In addition, some important areas discussed in this synthesis, such as social and economic components of resilience, were not emphasized in the appendix. Even one of the key premises of the Integrative Approaches chapter (1.1)—that a strategic landscape approach would promote resilience by avoiding potential traps of small-scale perspectives and constraints—is an important hypothesis to test through simulation modeling and an adaptive management framework. Revisiting and revising the list of questions from Appendix E would help to develop long-term strategy for research to address management challenges. The various chapters of this synthesis highlight gaps in knowledge from their respective disciplines, whereas the topics that follow emerged as important concerns across multiple chapters.

High-elevation forests

The management recommendations for the mixed-conifer forests presented in North et al. (2009) were not intended to extend to higher elevation forests with less frequent fire regimes. Considerations for red fir forests are discussed in the Forest Ecology chapter (2.0). Areas currently dominated by snow in winter, including alpine zones and subalpine forests, warrant increased attention due to the projected effects of climate change. Pacific marten (*Martes caurina*) are thought to be particularly vulnerable to habitat loss as a result of climate change (Purcell et al. 2012, Wasserman et al. 2012). Major changes in subalpine forest structure have occurred over the last century, and increasing tree densities may promote higher continuity of fuels, which could increase the future role of more intense fire, and greater density-related stress, which could increase forest susceptibility to outbreaks of insects and disease (Dolanc et al. 2012). Continued monitoring and research are needed to evaluate whether declines in whitebark pine (*Pinus albicaulis*) forests in California represent a “normative disturbance” indicative of a resilient ecosystem, or instead a "catastrophic" outcome (or transformation) resulting from the synergy of climate change, native insect pests, and novel stressors (Millar et al. 2012).

The zone of transition from wet mixed-conifer forests into red fir (*Abies magnifica*) is a particularly important focal area for forest management in the synthesis area. Multiple sections of this synthesis note that red fir forests are an important subject for research because they are broadly distributed in the region, support important values, such as habitat for priority species and water supply, and have not been extensively researched. Projected warming, shifts in precipitation from snow dominated to rain dominated, as well as associated increases in the incidence of severe wildfire, could result in disturbance effects that push systems in this transition zone beyond important ecological thresholds. For example, streams in this zone are expected to experience increases in and changes in seasonal timing of peak flows, and the freezing level in winter storms, which coincides with the moist mixed conifer/red fir transition, is expected to rise (Herbst and Cooper 2010, Safford et al. 2012a). As temperatures warm, trees in this zone are likely to grow all year rather than go dormant in winter, and their evapotranspiration will increase and likely reduce soil moisture and stream discharge (Bales et al. 2011). In addition, competitive interactions between martens and fishers may increase with decline in snowpack, which is projected to continue in the northern Sierra Nevada (Safford et al. 2012a).

Restoration of forested riparian areas

Forested riparian areas are highly valued yet have not been a focus for restoration research. The Water Resources chapter (6.2) suggests that more active use of mechanical thinning and prescribed fire would help to restore riparian ecosystems in the synthesis area, but effects on water quality, riparian soils, and priority riparian and aquatic species demand special consideration in experimental studies. However, for the Sierra Nevada, only one research experiment on prescribed fire in riparian areas has been conducted, and only one recent wildfire study has been completed for stream riparian areas. In the Tahoe basin, there have been studies on pile burning in streamside zones (see Soils chapter (5.0)) and pending research on silvicultural treatments in aspen stands, which are commonly found in riparian areas. The Kings River Experimental Watershed (KREW) study will provide new data from one experimental area in the southern Sierra Nevada over the next few years. Meanwhile, work as part of the Sierra Nevada Adaptive Management Program (SNAMP) will provide additional information on hydrologic effects in the central Sierra Nevada. Conducting experimental projects over extended periods (at least 10 years) and across the synthesis area, in combination with large-scale modeling, would help to guide practices to restore riparian areas and downstream aquatic resources.

Effects of wildfires, particularly long-term

Long-term effects of wildfire and treatments both pre- and post-wildfire remain a significant research gap (see Post-wildfire Management chapter (4.3)). Safford et al. (2012b) noted that the effectiveness of fuels treatments in reducing wildfire severity in frequent-fire forest types has been well established. There remains a need to evaluate effects of fires (along with effectiveness of forest treatments) in other ecosystem types, including riparian and montane hardwood forests. The Fire and Fire Surrogates Study (McIver et al. 2012, McIver and Fettig 2010) was designed for this purpose and would continue to provide important information if these sites were again emphasized. Moreover, the effects of fires (and post-fire treatments), especially in large severe patches over long periods, are not well understood.

Recent wildfires, such as the 2012 Chips Fire on the Plumas and Lassen national forests, present opportunities to learn how severe wildfires affect spotted owls and their habitat, because there is a decade-long monitoring dataset in the burned area. The Chips Fire burned through large areas previously burned by the 2000 Storrie Fire. The availability of data from existing plots in the Storrie Fire area will allow study of the effects of the reburn (see Post-wildfire Management chapter (4.3)). Among other objectives, these types of studies could help to evaluate the extent to which down woody fuel loads remaining from the Storrie Fire may have affected severity of the reburn.

Research on the effects of severe wildfire on aquatic systems has been quite limited in the Sierra Nevada (see Water Resources chapter 6.2). Given the particular importance of water quality as an ecosystem service, the potential impacts of increasingly severe wildfire in these systems, including debris flows (see Post-wildfire Management chapter 4.3) are an important research gap. One particular threat from wildfires that has been recognized in the Sierra Nevada is sedimentation of reservoirs, which can degrade water quality in the short-term and reduce storage capacity in the long-term (Moody and Martin 2004, Moody and Martin 2009).

Effects of post-wildfire treatments

Key questions remain concerning removal of burned trees and woody debris as part of post-wildfire treatments, given limited understanding of fuelbed succession following fires of different intensity. These questions are especially important given that the warming climate appears to be lengthening the fire season (Westerling et al. 2006), with associated increases in fire activity expected (Lenihan et al. 2003). There has been progress in evaluating effects of short-term Burned Area Emergency Response (BAER) projects on erosion control, although effects on channel processes remain a topic of concern (see Post-wildfire Management chapter 4.3). There is also some information on how dead trees are used by wildlife, such as black-backed woodpecker (*Picoides arcticus*). High-severity fire can provide habitat benefits in the form of standing dead trees, and post-wildfire management should consider the tradeoffs between habitat values and dead tree removal (see Post-wildfire Management chapter 4.3). Both social and ecological research is needed to evaluate the outcomes of accepting or influencing ecological trajectories of severely burned areas through salvage and other kinds of treatments. In addition, novel approaches may be needed to encourage regeneration of conifers and hardwoods where severities have exceeded expected conditions.

Effects of restoration treatments on ecological services and other social values

A common thread throughout this synthesis is the need for more integrated research that evaluates how ecological restoration efforts affect important socio-economic and cultural values. The Fire and Tribal Cultural Resources chapter (4.2), the Post-wildfire Management chapter (4.3), and the Wet Meadow chapter (6.3), all highlighted the gap in understanding effects on ecosystem services associated with wildlife, culturally important plants, and water resources (see Ecosystem Services chapter (9.2)). Although science suggests that there are opportunities for forest treatments to enhance water supply and mitigate some of the potential effects of climate change, research is lacking in the Sierra Nevada for how much and how long restoration treatments are likely to increase water yield and if water quality can be maintained (see Watersheds and Stream Ecosystems chapter (6.1).

Setting and applying guidelines

Benchmarks and performance criteria can be valuable tools for evaluating progress toward meeting broad restoration goals, and there are large efforts to develop integrated indexes of ecosystem health that consider ecological and social conditions (Rapport and Maffi 2011, Rapport and Singh 2006). However, at a broad strategic landscape level, it can be problematic to emphasize quantitative targets. As a result, resilience-based approaches tend to deemphasize fixed production targets in favor of plans to reduce vulnerability and strengthen capacity to respond and adapt. As an example of how metrics could be used, Fire Return Interval Departure (FRID) metrics can serve as an initial measure of restoring fire as an ecological process; however, strategic design of treatments should be informed by other important ecological and social factors that relate to vulnerability (see Fire and Fuels chapter (4.1)). As a result, restoration designed to promote broader societal interests will strive to include ecological criteria for evaluating success; however, efforts to quantify social criteria, such as cultural significance and community well-being, similarly run a risk of being overly reductionist (Higgs 1997). Instead, researchers have suggested using science-based guidelines to identify potential vulnerabilities and opportunities that may be helpful in promoting resilience in socioecological systems (Cabell and Oelofse 2012). As an example of such a guide, the Synopsis of Emergent Approaches (1.3) suggests that there may be a relatively low proportion of a landscape that could be strategically treated to avoid the most undesirable effects of wildfires while sustaining priority wildlife species. Such guidance needs to be considered as a hypothesis to be tested in an adaptive management framework that involves managers, stakeholders, and researchers. Such a reflective approach would engage the public and communities in evaluating the particular ecological and social characteristics of their landscapes, identifying vulnerabilities, and strengthening capacity to adapt to future disturbances and other stressors.

Summary of Cross-cutting Research Gaps

- **High-elevation forests, including the upper montane and sub-alpine zones, warrant increased attention and research due to the projected effects of climate change.** These forests provide important habitat and biodiversity values, and they face novel threats from shifts in precipitation patterns and increased likelihood of uncharacteristically severe wildfire.
- **Forested riparian areas are highly valued, yet they have not been a focus for restoration research**. Conducting experimental projects over extended periods and across the synthesis area, in combination with large-scale modeling, would help to guide practices to restore riparian areas and downstream aquatic resources.
- **Long-term effects of wildfire and treatments both pre- and post-wildfire remain a significant research gap.** Management could benefit from increased understanding of long-term fire effects in a range of ecosystem types (riparian, upland, aquatic, etc.) and after multiple fires.
- **Key questions remain concerning removal of burned trees and woody debris as part of post-wildfire treatments**, given limited understanding of fuelbed succession following fires of different intensity. Both social and ecological research is needed to evaluate the outcomes of accepting or influencing ecological trajectories of severely burned areas through salvage and other kinds of treatments.
- **There is a great need for more integrated research that evaluates how ecological restoration efforts affect important socio-economic and cultural values.** Although science suggests that there are opportunities for forest treatments to enhance water supply and mitigate some of the potential effects of climate change, research is lacking in the Sierra Nevada for how much and how long restoration treatments are likely to increase water yield and if water quality can be maintained.
- Benchmarks and performance criteria can be valuable tools for evaluating progress toward meeting broad restoration goals, and there are large efforts to develop integrated indexes of ecosystem health that consider ecological and social conditions. **However, at a broad strategic landscape level, it can be problematic to emphasize quantitative targets.** More research on these types of benchmarks and criteria would help to inform management goals and strategies.

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2.0 Forest Ecology

By Malcolm North

Contents

Two USDA Forest Service reports, PSW-GTR-220 (North et al. 2009a) and PSW-GTR-237 (North 2012a), summarize some of the recent research in forest ecology relevant to a science synthesis for the Sierra Nevada region. GTR-237 builds on concepts in GTR-220, providing new, more in-depth information on topics of “Forest Health and Bark Beetles” (Fettig 2012), “Climate Change and the Relevance of Historical Forest Conditions” (Safford et al. 2012a), “Marking and Assessing Stand Heterogeneity” (North and Sherlock 2012), “GIS Landscape Analysis Using GTR 220 Concepts” (North et al. 2012a), and “Clarifying [GTR 220] Concepts” (North and Stine 2012). The final chapter in GTR-237, “A Desired Future Condition for Sierra Nevada Forests” (North 2012b), highlights three topics (The limitations of stand-level averages, Economics and treatment scale, and Monitoring) where ecological research suggests a need for fundamental changes in how the Forest Service approaches ecosystem management.

Building on these GTRs, this Forest Ecology chapter has a different structure than the other chapters. It is focused on four subjects for which stakeholders and managers have suggested that a summary of existing information would be relevant to a regional science synthesis: tree regeneration and canopy cover, red fir (*Abies magnifica*) forests, forest treatments to facilitate fire heterogeneity, and carbon management in fire-prone forests. Furthermore, these four sections do not attempt to summarize and cite all literature relevant to each section. Rather, each section begins with one or two questions that motivated the subject’s inclusion in this synthesis. These questions provide the framework for how the relevant literature is selected and summarized. The chapter ends with a sidebar giving an example of how larger scales are often used in meeting forest management objectives.

Tree Regeneration and Canopy Cover

Within the last decade, there has been substantial new research on the light requirements and gap conditions associated with favoring shade-intolerant, fire-tolerant pines. Although managers have long known high-light environments are needed to favor pines, creating these gaps has sometimes been seen as conflicting with canopy cover targets suggested in the current standards and guides.

- What gap size and light conditions are needed to favor pine establishment and growth over shade-tolerant firs and incense cedar?
- Do open conditions created in these gaps reduce canopy cover below threshold guidelines?

To increase resilience in frequent-fire forests to a warming climate and wildfire, and to restore ecosystem functions, conifer regeneration across much of the Sierra Nevada needs to favor fire-tolerant pines. Sugar (*Pinus lambertiana*), Jeffrey (*p. jeffreyi*), and ponderosa (*p. ponderosa*) pine are all shade intolerant and require high-light environments to survive and grow more rapidly than fire-sensitive, shade-tolerant species, such as white (*Abies concolor*) and red fir. Two species, incense cedar (*Calocedrus decurrens*) and Douglas-fir (*Pseudotsuga menziesii*), are considered shade tolerant in the central and southern Sierra, but in the northern Sierra, southern Cascades, and Klamath Mountains, they are sometimes able to survive and thrive in high-light environments if there is sufficient precipitation. Studies have shown that firs and incense cedars can produce 20 to 30 times the amount of seed per unit basal area as many pine species (Gray et al. 2005, Zald et al. 2008). Even fuels reduction and forest restoration treatments that favor pine retention in mixed-conifer forests often retain enough large fir and incense cedar seed trees to perpetuate pre-treatment composition (Zald et al. 2008). Fire suppression, which has increased canopy cover and reduced understory light, has been in effect long enough that many shade-tolerant species are now large enough to survive low-intensity fire (Miller and Urban 2000, Collins et al. 2011, Lydersen and North 2012). Moderate-severity fire or mechanical thinning may be needed to sufficiently open the canopy. Repeated applications of low-intensity fire may also eventually shift tree regeneration and sapling composition toward pine, but at present, few forests burn with sufficient frequency to affect this compositional shift. In the absence of frequent fire, reducing canopy cover and planting pine seedlings may be the most effective means of overcoming the entrenched effects of fire suppression that favor shade-tolerant, fire-sensitive regeneration.

Recent work has examined what understory light level is needed to favor pine regeneration over fir and incense cedar (York et al. 2003, 2004; Bigelow et al. 2011). These studies suggest that in many forests, a minimum opening of 0.25 ac is needed to provide enough light (40 percent of full sunlight) to support pine regeneration within part of the opening. As gap size increases, so does the area with a high-light environment favoring pine, and the growth rate of the gap's pine seedlings (McDonald and Phillips 1999, York et al. 2004, Zald et al. 2008). A similar response occurs in smaller gaps when canopy cover is reduced on the southern side of a gap ('feathering the edge'). This may explain why pre-fire suppression gap sizes appear to have been relatively small (0.3-0.8 ac) (Knapp et al. 2012), yet most of these stands likely supported robust pine regeneration because reconstruction studies and old data suggest pine often contributed > 40 percent of mixed-conifer basal area (McKelvey and Johnson 1992). A recent study found that small gaps (0.1 ac) created in pile and burn treatments significantly increased stand-level light heterogeneity; that study also found greater ponderosa pine germination on ash substrates produced by the pile burns, compared to bare soil (York et al. 2012). At the Beaver Creek Pinery, a ponderosa pine forest with a modern history of low-intensity burns, Taylor (2010) found a small average gap size of 0.14 ac with high variability (range 0.02-0.6 ac), suggesting small gaps may be sufficient for shade-intolerant regeneration if the forest matrix surrounding the gap has a low density and low canopy cover.

Outside of gaps, Bigelow et al. (2011) found that thinning the forest matrix to a canopy cover of 40 percent provided sufficient light to support pine regeneration in about 20 percent of the treated area. In a recent study of old growth, mixed-conifer stands with restored fire regimes (Lydersen and North 2012), canopy cover averaged 44 percent, which supported a regeneration composition consistent with overstory composition (about 50 percent pine). These low estimates of canopy cover may seem at odds with the goal of providing habitat sufficient for some Forest Service sensitive species, such as fisher (*Martes pennanti*) and California spotted owl (*Strix occidentalis occidentalis*). However, canopy cover is a stand-level average of canopy conditions. In heterogeneous forests composed of tree groups, gaps, and a low-density matrix (North et al. 2009b, Larson and Churchill 2012), most of the gaps and some of the matrix will likely have low canopy closure (the percentage of the sky hemisphere covered with foliage when viewed from a single point [Jennings et al. 1999]). In contrast, canopy closure in tree clusters often exceeds 65 percent. Distinguishing between stand-level average measures of vertical porosity (canopy cover) and point-level measures of how much of the sky hemisphere is blocked by foliage (canopy closure) can improve assessments of canopy conditions (North and Stine 2012).

Point measures of canopy closure are probably best for assessing how much ‘protection’ and foliage cover there is over a patch or microsite. Spherical densiometers are often used to assess canopy conditions in the field. Practitioners should understand that densiometers do not measure canopy cover. Instead, they are designed to measure canopy closure (a large viewing angle represented by an inverted cone) over the point from which the readings are taken (Nuttle 1997). Of several methods available, closure is probably most effectively measured with a digitized hemispherical photograph that is analyzed with computer software. In contrast, canopy cover is probably most effectively measured with a siting tube or densitometer, which records whether, within a narrow view window approaching a point, the observer can or cannot see the sky. Multiple readings (often 100 or more) are taken throughout the stand of interest and the percent of readings where the sky is obscured is recorded as canopy cover. Canopy cover, however, is often indirectly estimated from plot data using the Forest Vegetation Simulator (FVS). Managers should be aware that these estimates are based on an assumption about how trees are distributed that does not account for actual conditions in the stand that is being modeled. If managers want to increase fine-scale heterogeneity, variability in canopy closure may provide a better assessment of conditions than canopy cover, a stand-level average. North and Sherlock (2012) suggest using both to provide estimates of stand-level conditions and within stand or patch conditions.

Distinguishing between canopy cover and closure may help resolve one problem often faced by foresters: how to provide high-light environments that favor pine regeneration and also meet canopy cover targets for Forest Service sensitive species. With high within-stand variability in canopy closure, managers can produce openings favoring pine regeneration and still attain a stand-level average of canopy cover high enough to meet canopy cover targets. Further discussion of this distinction and a figure illustrating the differences can be found in North and Stine (2012).

Red Fir Forests

Forest management in the Sierra Nevada has often focused more of its attention on forest types that historically had a frequent, low-intensity fire regime. With fire regimes now having been altered for over a century, some managers and stakeholders have asked whether red fir, generally the next higher forest type in elevation above mixed conifer, should receive more active management. In particular, there is interest in understanding whether these forests need fire and/or mechanical treatment to help restore ecosystem conditions and increase resilience to a changing climate.

- What was red fir's historical fire regime and how did it vary with site conditions?
- Is gap creation needed to facilitate red fir regeneration and development of younger tree cohorts?

According to a GAP (Gap Analysis Program) of forest types and ownerships in the Sierra Nevada, red fir forests are the fourth most extensive forest type. These forests cover 838,905 ac in the Sierra Nevada (or about 11 percent of the region's 7.8 million ac), of which 511,732 ac are on Forest Service land (Davis and Stoms 1996) (Table 1). It is the largest forest type in the upper montane zone (above 6000 ft to 7500 ft in elevation from the northern to southern Sierra Nevada, respectively), and it is often 'passively' managed (i.e., rarely receives active management treatments such as mechanical thinning, planting, or prescribed fire), because it is remote, in wilderness designation, or less of a fire danger to structures and humans. These forests are important habitat for many species, including the Pacific marten (*Martes caurina*) (see discussion in the Forest Carnivore chapter [7.1]), and they occupy the elevation zone with greatest snowpack depth (Laacke 1990). As a result, they may be significantly impacted by climate change, as most models suggest precipitation may often turn from snow to rain in much of this zone in the future (Safford et al. 2012a). It's unclear how this will affect red fir forests. Climate change may become a chronic stress in red fir forests in the lower parts of its present distribution, but the exact mechanisms of this stress and its potential influence on ecosystem processes are unknown. Historically, there was some timber harvested in red fir forests in the 1970s and '80s (Laacke 1990), but with increased designation of roadless areas and public controversy, there has been much less active management in many red fir forests since the 1990s. The concern with red fir is what type of management would best maintain or restore its ecosystem processes given a century of altered fire frequency and uncertain but probable future climate warming.

Management concepts in GTR-220 are applicable to forests that historically had frequent, low-intensity fire regimes. The historical fire regime in red fir is not as well defined as it is in lower elevation forest types, such as ponderosa pine and mixed-conifer, and it has often been classed as mixed-severity (Parker 1984, Skinner 2003, Sugihara et al. 2006). A recent review paper of all fire history studies on dominant woody species in California for different forest types lists 29 studies with some information on red fir fire return intervals (van de Water and Safford 2011). The review paper reports mean, median, minimum mean, and maximum mean fire return intervals of 40, 33, 15, and 130 years, respectively. Using a 40 year historical fire return interval, approximately 12,793 ac of red fir may have burned each year before fire suppression (about 2.3 percent of the historical annual burn acreage for all forest types) (Table 1). Many of the red fir fire history studies, like those at lower elevations, found few if any fires in their sample area in recent decades, suggesting some red fir forests have now missed more than one fire return interval (Stephens 2001).

A review of these studies suggests a wide range of fire regimes, possibly because many of the studies examine stands in which red fir is mixed with other species. Red fir is often found across a broad elevation band from mixed-conifer (generally 4500 ft to 7500 ft) to western white pine (*Pinus monticola*) (generally 8500 ft to 10,500 ft) forest types. Studies at lower elevations, where the tree species composition suggests drier site conditions, have generally found lower historical fire return intervals and age structures that suggest frequent pulses of regeneration. In contrast, higher elevation and more mesic site studies often document a mixed-severity fire regime with distinct recruitment pulses following fire events (Taylor 2004, Scholl and Taylor 2006). One study documented a strong linear relationship between fire return interval and elevation, possibly driven by snow pack and its effect on fuel moistures (Bekker and Taylor 2001). Another factor may be landscape context. Red fir forests that are well connected with lower elevation forest may have shorter intervals because fire could easily carry up into higher elevations under suitable weather and fuel moisture conditions (Skinner 2003). In contrast, some red fir forests grow in shallow 'flower pot' pockets surrounded by extensive exposed granite. These red fir forests likely had longer intervals because of their relative isolation.

Analysis of fire patterns in red fir indicates high-severity patches often occur (Pitcher 1987, Stephens 2000). A recent paper analyzing fire severity patterns between Yosemite National Park and adjacent national forest lands found that wildfires in red fir forests in Yosemite averaged 7.1 percent high severity and burned at significantly lower severity than wildfires on FS lands on the east and west sides of the Sierra Nevada crest (16.3 percent and 12.1 percent, respectively) (Miller et al. 2012). Given Yosemite's more extensive use of fire for resource benefit, Miller et al. (2012) suggested that the park's levels of high-severity fire may more closely mimic the area's historical fire regime. Another study in upper-montane mixed-conifer and red fir forests with a restored fire regime in Illilouette Basin (Collins and Stephens 2010) found higher levels (about 15 percent) of high severity. This paper also analyzed high-severity patch size, finding that most patches in that area were small (<10 ac), but about 5 percent of the total number of patches were large (150-230 ac). High-severity patches larger than the upper bounds of this range may be uncharacteristic of historical fire patterns. If fire burns at high intensity in red fir, larger patches can switch to montane shrub fields. This switch may persist for decades, as shrubs inhibit tree regeneration, slow their growth, and facilitate post-fire, small-tree mortality that favors shrub re-sprouting and dominance (Nagel and Taylor 2005).

Collectively, the research suggests two considerations for managing red fir forests. First, where feasible, fire restoration would benefit red fir ecosystems (Skinner 2003). For some remote areas, this may mean designation as managed wildfire areas and/or include the application of prescribed fire. Fire history studies suggest that many stands have 'missed' one to three burn events and, consequently, are likely to have increased fuel loading, higher stem densities, and less light in the understory (Taylor 2000). These changes have also reduced shrub cover, and the habitat that shrubs provide, to the low levels noted in some red fir studies (Selter et al. 1986, North et al. 2002). Fuel loads will need to be evaluated on a site-by-site basis (McColl and Powers 2003). With a mixed-severity fire regime, however, higher fuel loads may still be acceptable under moderate weather conditions, since some torching and large tree mortality may be a desired outcome. Fire appears to be the most effective means of ensuring natural red fir regeneration.

Second, in drier, lower elevation red fir forests, and in productive stands connected to lower elevation forests with frequent fire regimes, some fuels reduction may be needed to reduce risks to structures

and people (Zhang and Oliver 2006). Initial treatments in these areas could focus on surface fuels reduction and removal of some smaller trees. Canopy openings do not appear to be required for successful regeneration as long as canopy cover is low enough to allow sun flecking, which is associated with increased seedling survival (Ustin et al. 1984). Red fir is shade tolerant, so seedlings and saplings can persist in stands with high canopy cover (Selter et al. 1986, Barbour et al. 1998). Studies suggest, however, that recruitment and establishment are often linked to disturbance, particularly fire (Taylor and Halpern 1991, Taylor 1993, Taylor and Solem 2001).

Experimentation with mechanical treatments that create small openings (i.e., 0.1-0.5 ac) may be needed later as seedlings grow. There is some evidence to suggest that rates of sapling survival and growth are higher in areas where mixed-severity fire has killed overstory trees (Pitcher 1987, Chappell and Agee 1996). Long-term regeneration studies have found abundant natural seedling and successful red fir establishment in canopy openings created by mechanical thinning (Gordon 1970, 1973a, 1973b, 1979).

Forest Treatments to Facilitate Fire Effects Heterogeneity

Fire restoration in the Sierra Nevada is difficult due to many constraints including enforcement of air quality regulations, liability and safety concerns, and increased rural home construction (North et al. 2012b). Some fire managers and scientists have questioned whether prescribed fire constraints limit their intended ecological benefits, and in particular whether there is sufficient heterogeneity in intensity and severity.

- Given current limitations, how can prescribed fire be applied with different intensities to create forest structural heterogeneity, a common goal in forest restoration?

Recent ecosystem management approaches that emphasize increasing forest structural heterogeneity largely focused on mechanical treatments while stressing the benefits of reintroducing fire where possible (North et al. 2009b, North 2012a). Prescribed fire, however, can often only be used under certain weather and fuel moisture conditions during a limited 'burn window' allowed by air quality regulators. These constraints reduce fire effects variability because burns must often be quickly executed, which reduces the heterogeneity produced by slower moving burns that tend to be patchier. Furthermore, when fire has been absent for several decades, dense stands of young trees may not be killed by rapid, low-intensity prescribed fire, and structural homogeneity within the stand may be retained (Miller and Urban 2000). In contrast, accounts of historical fires and managed wildfires (often in wilderness) suggest that under less constrained conditions, fires burned for a long time and at different intensities, depending on changes in fuel and weather conditions (Nesmith et al. 2011). This variability likely created greater microclimate and habitat heterogeneity in the post-burn forest, producing bare mineral soil areas where fire burned at high intensity and other areas missed by fire that could provide refugia for tree saplings and some fire-sensitive understory plant species (Wayman and North 2007). With prescribed fire, this variability is often markedly reduced due to the constrained conditions of where and when fire can now be used.

In stands with constrained burn windows, managers might consider varying fuel conditions within treatment areas to help prescribed burning produce variable fire effects. In general, fuels treatments have been focused on removing ladder and surface fuels to facilitate fire containment, suppression, and reduced mortality of overstory trees (Reinhardt et al. 2008). When prescribed fire is only allowed to burn for a brief period, or when fuels have relatively similar moisture contents, creating surface and ladder fuel heterogeneity may help achieve some of the variable fire effects that would have been produced under less constrained conditions. Studies suggest surface fuel input rates and higher stem density associated with ladder fuels vary with site productivity (van Wagtendonk et al. 1998, Taylor and Skinner 2003, van Wagtendonk and Moore 2010). To create variable fuel conditions, managers might use small changes in productivity to guide spatial variation. In many mixed-conifer forests, productivity is often associated with available moisture. Higher surface fuel loads and some ladder fuels might be left in more mesic microsites and more extensively removed in more xeric conditions, such as ridge tops and areas with shallow soils. Metrics for evaluating prescribed fire effectiveness may also need to be adjusted. Desirable outcomes when creating variable fire effects will include limited areas of torching and some ground that has not been blackened. Safford et al. (2012b) recommend that prescribed burn projects plan for 5-15 percent overstory mortality. In mixed conifer, topography will naturally increase variability in fire effects, but given the time, weather, and fuel conditions, and the constraints associated with current prescribed fire policy, manipulations of fuel heterogeneity may be needed in some areas to produce the variable post-burn conditions likely created by historical fire regimes.

Carbon Management in Fire-Prone Forests

Forests store large amounts of carbon and through growth can become carbon sinks to offset anthropogenic emissions of CO_2. Wildfires release carbon back to the atmosphere, and the amount of release increases with fire severity. Fuels treatments can, in the event of a wildfire, reduce fire severity and consequent carbon release, but they come at a 'cost' because in the near-term, they also reduce forest carbon stores.

- Do young, fast-growing trees that are harvested for wood products provide greater long-term carbon storage than growing and retaining large, old trees?
- What are the carbon costs and benefits of fuels reduction in fire-prone forests?

Through growth and the long-lived nature of many trees, forests sequester carbon from the atmosphere. Recent policy and political attention has been focused on the potential to mitigate the effects of climate change through forest management. The most ready means of increasing forest carbon stores is through afforestation and reforestation of forest lands converted to other uses (i.e., agriculture, pasture, etc.) (IPCC 2005). Although developing countries often reduce their carbon stores as forestland is converted to other uses, forests in the United States have been a net carbon sink in the last century due to forest regrowth (particularly in the upper Midwest and New England) and, in some cases (see discussion below), fire suppression (Hurtt et al. 2002). For much of the U.S., where forestland cover is now relatively stable, the question has been whether different management practices could stabilize or increase the amount of carbon storage that forests presently contain.

In the past, some groups have suggested that converting old forests to young, fast-growing plantations, whose harvested wood products could store carbon for several decades, would create a net increase in long-term carbon stocks. This approach was based on the idea that old forests are slow growing and carbon neutral because respiration costs nearly balance carbon uptake (Odum 1969). More recent research generally does not support this idea, as a global survey of old forests found that many continue to sequester carbon and have stocks that far exceed young, managed forests (Luyssaert et al. 2008). In addition, there is some evidence (Sillett et al. 2010) that large trees may contain even more carbon than our current estimates predict. This is because a tree's carbon storage is estimated from its diameter and, unlike younger trees upon which most carbon allometric equations are based, old trees may be allocating most of their growth to the upper bole (Sillett et al. 2010). If young forest stocks could be efficiently harvested and their carbon sequestered in wood products for centuries, after several rotations they might match carbon stores in old forests dominated by large trees. However, this would be difficult with current wood use practices. The problem is not with the immediate carbon expense from machinery, because generally the amount of carbon loss from fossil fuel used in the forest operations (i.e., diesel and gasoline) is quite small (often <5 percent) compared with the carbon captured in the harvested forest biomass (Finkrel and Evans 2008, North et al. 2009a). The problem is that the carbon is not stored for long and often ends up, through decomposition, back in the atmosphere. A recent global analysis of the longevity of harvested forest carbon found that after 30 years, in most countries (90 of 169), less than 5 percent of the carbon still remained in longer storage, such as wood products and landfills (Earles et al. 2012). Most temperate forest countries with longer-lived products, such as wood panels and lumber, had higher carbon storage rates, with Europe, Canada, and the U.S. averaging 36 percent of the forest carbon still stored after 30 years (Earles et al. 2012). This higher rate, however, is still far short of what large long-lived trees would continue to accumulate and store over several decades to centuries.

In fire-prone forests, there has been substantial debate about whether carbon loss through fuels treatment (mechanical thinning and/or prescribed fire) is offset by lower carbon emissions if the treated stand is later burned by wildfire (Hurteau et al. 2008, Hurteau and North 2009, Mitchell et al. 2009, North and Hurteau 2011, Campbell et al. 2012). Different results from these studies and others are in part due to the spatial and temporal scale over which the carbon accounting is assessed, the 'fate' of the carbon removed in the fuels treatment, and whether long-term carbon emissions from dead trees are included (Hurteau and Brooks 2011). In general, treating forests often results in a net carbon loss due to the low probability of wildfire actually burning the treated area, the modest reduction in wildfire combustion and carbon emissions, and the need to maintain fuels reduction through periodic additional carbon removal (Campbell et al. 2012). Over the long term (i.e., centuries), Campbell et al. (2012) suggest that carbon stores in unthinned forests and those that experience infrequent high-severity fire will exceed those exposed to frequent low-severity fire. Forest location, however, is an important consideration, as some areas have much higher risk of ignition (e.g., road corridors, ridge tops) and carbon loss from wildfire than other areas. For most policy and economic analysis, Campbell et al.'s (2012) temporal scale is not as relevant as carbon dynamics over the next few decades (Hurteau et al. 2013).

Recent research has proposed the idea of carbon carrying capacity (Keith et al. 2009). This concept may be particularly relevant to forest managers because it emphasizes carbon stability and the level of storage that forests can maintain. In the absence of disturbance, a forest may 'pack' on more carbon as the density and size of trees increase. This additional biomass, however, makes the forest prone to disturbances, such as drought stress, pests, pathogens, and higher-severity wildfire, which increase tree mortality. This mortality reduces carbons stocks as dead trees decompose and through efflux, much of the carbon returns to the atmosphere. Carbon carrying capacity, therefore, is lower than the maximum storage potential of a forest, but represents the biomass that can be maintained given disturbance and mortality agents endogenous to the ecosystem. In frequent-fire forests such as Sierra Nevada mixed conifer, the carbon carrying capacity is the amount that a forest can store and still be resilient (i.e., have low levels of mortality) to fire, drought, and bark beetle disturbances.

One factor that would change this long-term balance is if management led to increased carbon storage by altering the amount and longevity of sequestered carbon. In Sierra Nevada mixed-conifer forests, two studies that examined historical forest conditions have suggested that this might be possible. Although historical forests were less dense due to frequent fire, they may have stored more carbon because the number and size of large trees was greater (Fellows and Golden 2008, North et al. 2009a) than in current forests that have fewer large trees, possibly due to increased mortality rates from increased stand density (Smith et al. 2005). Carbon stores are calculated from total tree biomass (a three dimensional measure) and will be much higher in a stand with a few large trees compared with a stand with many small trees, even if both stands have similar basal area (a two dimensional measure). Other studies (Hurteau et al. 2010, Scholl and Taylor 2010), however, have found higher carbon storage in modern fire-suppressed than in historical active-fire forests, suggesting that there may be considerable variability between different locations and levels of productivity. In general, forests managed so that growth and carbon accumulation are concentrated in large trees will also have longer, more secure carbon storage than in stands where growth is concentrated in a high density of small trees prone to pest, pathogen, and fire mortality.

Sidebar on Forest Management at Larger Scales

Most forest ecology research has been concentrated on small spatial and temporal scales that are not always relevant to managing forest landscapes over the long term. What research has occurred at larger scales is often context specific, providing case studies of particular landscapes and species. Although there are many relevant modeling studies, it is difficult to find empirical large-scale, long-term ecological research that is directly relevant to current management issues in the Sierra Nevada. In practice, it is often local managers who must make these decisions and must balance where and when to maintain current conditions versus treating forests to move toward a desired condition decades in the future. An example may help illustrate how different scales are often considered and how a stand's context might affect management decisions.

A manager might be faced with the choice of whether or not to thin around a large black oak (*Quercus kelloggii*) that is being overtopped by surrounding conifers. If thinning sufficiently opens up the canopy, the oak will likely survive and may produce acorns that could thrive in the high-light environment. If left alone, the oak will likely die within a few years, but even so, the tree can provide valuable resting and nesting habitat for sensitive species in the near future (as both a near-dead tree and, later, as a snag). Any manager faced with this decision will have to weigh current and future needs for habitat and oak regeneration, both locally and across the landscape in which the stand is embedded. The context of the forest's current condition forces consideration of larger scales. For example, how rare are sensitive species habitats and large oaks within the larger landscape, and how rare will they be in the future? How resilient will a large oak be to prolonged drought under different levels of stem density? There is no clear resolution to this situation. Communicating what the tradeoffs are and how decisions will be made may help stakeholders understand the effort to incorporate larger spatial and temporal scales into current, stand-level management decisions.

Table 1: Forest type, total area, fractional Forest Service ownership, historical fire return interval (HFRI), and estimated historical amount of area burned each year in the Sierra Nevada before fire suppression for Forest Service lands. HFRI was determined from three sources with extensive literature reviews of many fire history studies: Stephens et al. 2007, Van de Water and Safford 2011, and the fire effects information database (http://www.fs.fed.us/database/feis/plants/tree/. The extent of the Sierra Nevada is the Jepson (Hickman 1993) definition, which generally corresponds to the Plumas N.F. south through the Sequoia N.F., including the Inyo N.F. Table adapted and updated from North et al. 2012b. Forest type, total area, and fractional ownership are from Davis and Strom (1996).

Forest Type	Area (ac)	USFS Own.	USFS (ac)	HFRI mean (yr)	Hist burn ac/yr
Mixed-conifer	1,466,539	0.62	909,254	12	75,771
West-side ponderosa pine	1,087,734	0.53	576,499	5	115,300
Lower cismontane mixed conifer-oak	1,046,221	0.46	481,262	10	48,126
Jeffrey pine-fir	730,428	0.8	584,342	8	73,043
Jeffrey pine	484,563	0.75	363,422	6	60,570
East-side ponderosa pine	398,819	0.76	303,103	5	60,621
Black oak	268,598	0.6	161,159	10	16,116
White fir	133,434	0.7	93,404	25	3,736
Aspen	24,463	0.89	21,772	30	726
Sequoia-mixed conifer	17,544	0.31	5,439	15	363
Active Management Total	**5,658,343**		**3,499,655**		**454,371**
Red fir	838,905	0.61	511,732	40	12,793
Lodgepole pine	532,748	0.6	319,649	30	10,655
Red fir-western white pine	393,877	0.75	295,408	50	5,908
White bark pine-mountain hemlock	93,404	0.62	57,910	85	681
White bark pine-lodgepole pine	92,168	0.86	79,265	40	1,982
Upper cismontane mixed conifer-oak	64,493	0.48	30,957	15	2,064
Foxtail pine	58,810	0.21	12,350	50	247
Whitebark pine	54,115	0.68	36,798	65	566
Passive Management Total	**2,128,519**		**1,344,068**		**34,896**
All Land Total	**7,786,862**		**4,843,723**		**489,267**

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3.0 Genetics of Forest Trees

Jessica Wright

Contents

Introduction

Climate change is anticipated to cause dramatic shifts in climate across the Sierra Nevada, including increased frequency and severity of wildfires (Safford et al. 2012). Reforestation may be an important component of ecological restoration after severe wildfires. These wildfire events may be important opportunities to promote resilience to climate change, because interventions during the early stages of succession can be less costly and more effective than during later stages (Betancourt 2012). Ecological genetics, the study of genes and genotypes of natural populations in their environment, can inform restoration efforts with the goal of promoting more resilient forests.

Research in forest genetics strives to understand the distribution and structure of genetic variation within tree species across the landscape. This information can inform resilient forest management strategies. Studies include examining genetic variation in adaptive traits (growth and survival) in common garden studies (Conkle 1973, Mátyás 1994, Mátyás 1996, Rehfeldt 1999, O'Neill et al. 2007, O'Neill et al. 2008, Thomson and Parker 2008, Thomson et al. 2009, Ukrainetz et al. 2011), as well as using molecular genetic and genomic variation to characterize genetic variation (reviewed in Neale and Kremer 2011). Although the study of conservation genetics has informed the management of forest animal species (Avise 2004), this chapter focuses on the genetics of forest trees, as they are uniquely challenged in responding to climate change because they are long lived and cannot move once established, should the climate in a local area become intolerable.

Sidebar: Emerging genetic and genomic approaches

Recent advancements in molecular genetic and genomic approaches are making available powerful new tools, which allow for a better understanding of the genetic variation underlying important traits and the association between genetic variation and environmental variation. Recently Parchman et al. (2012) used a novel genomic approach and found 97,000 variable sites in a genome-wide survey of single nucleotide polymorphism (SNP) variation. Using genome-wide association (GWAS), they were able to identify 11 candidate loci that were strongly associated with serotiny in lodgepole pines (*Pinus contorta*) in Wyoming. Using approaches like this, one can find the loci that are associated with ecologically and economically important traits. In addition, landscape genomic approaches are starting to identify loci associated with climate variation. For example, Eckert et al. (2010) examined 1730 SNPs in loblolly pine (*P. taeda*) and found several of them associated with climate variables, suggesting that those loci are potentially adaptive. Association genetics and landscape genomics are beginning to provide important insights for understanding the underlying genetics of adaptive traits. Understanding how adaptive variation is distributed across the landscape will have the potential to help inform management decisions, including reforestation efforts.

An important question in forest management is how to use genetic tools to inform management responses to climate change. There are several potential strategies available to promote resilient forests. In 1992, Ledig and Kitzmiller proposed responding to climate change through assisted migration. They suggested moving tree seed sources uphill by an amount defined by the projected amount of increased temperature according to Hopkin's Law, which predicts that temperature goes down 1.4°C for every 1000 foot gain in elevation (Ledig and Kitzmiller 1992). However, they recommended waiting 10 years before starting any program of assisted migration in order to wait until "the signal for global warming becomes clearer" (Page 158). This simplistic approach to responding to novel climates has been argued against, particularly given the complex nature of mountain ecosystems (Millar et al. 2007). Indeed, uncertainty in predictions of the amount of temperature and precipitation changes in mountain ecosystems is a major hurdle in designing reforestation efforts to respond to climate change (Millar et al. 2007). One suggested approach is to sow mixtures of seed sources and, hence, use a bet-hedging approach to minimize the risk of failure (Crowe and Parker 2008, Millar et al. 2007). The topic of assisted migration (also known as managed relocation, see Schwartz et al. 2012) is highly complex, and is currently under intense debate. A recent review article describes a set of relevant ethical, policy, and ecological questions surrounding any managed relocation effort (Schwartz et al. 2012). There is currently no consensus for the use of managed relocation in managing California's forested ecosystems or in any other ecosystem.

In light of this uncertainty, Forest Service geneticists prepared a report entitled "Genetic resource management and climate change: Genetic options for adapting national forests to climate change" (Erickson et al. 2012). The report presents underlying principles for the role of genetics in responses to climate change; it outlines the need for "genetically diverse and adapted seed and planting stock" (Principle 1, page 10) for ecological restoration, and it emphasizes the importance of gene conservation. The authors recommend developing seed collection, storage, and nursery capacity, particularly for a broader range of species than historically deployed. They also propose working to establish new provenance and common garden studies, recognizing the need for the type of data that provenance tests can generate. They also put forward the idea of examining potential *in situ* and *ex situ* gene conservation plans.

Sidebar: Glossary

In situ gene conservation: populations of trees are conserved in a naturally occurring population that is part of a genetic conservation reserve.

Ex situ gene conservation: Seeds are collected from trees and stored in seed banks, either for long-term conservation or later deployment as part of ecological restoration activities. Seeds from the genus *Pinus* can generally be stored in optimal conditions for many years (Bonner and Karrfalt 2008). However, other species, such as oaks, do not store well after seeds are shed, so *ex situ* conservation for those species requires a living gene conservation archive plantation.

Common Garden Experiment: A study, generally in plants, that involves planting multiple sources of plants in a single common garden or across multiple gardens. When all sources of plants are planted in a garden located in their home environment as well as one or more other environments, this is called a *reciprocal transplant experiment*.

Provenance Test: A type of common garden experiment, generally in trees, where sources of trees (from a set of locations or provenances) are planted in a common garden or gardens. Ideally, these provenances are from across the entire range of the species, though many studies choose to focus on a particular part of the range.

Assisted Migration: Schwartz et al. (2012) define this as “Introducing a species into a new location by bringing propagules or individuals and releasing them” (Table 1, Page 733).

Managed relocation: Schwartz et al. (2012) offer this definition: “The intentional act of moving species, populations, or genotypes to a location outside of a known historical distribution for the purpose of maintaining biological diversity of ecosystem functions as an adaptation strategy for climate change” (Table 1, Page 733).

Forest genetics research has made a number of important contributions to land management practices in the Sierra Nevada. Perhaps none is more important than the creation of the California seed zone map (Figure 1) (Buck et al. 1970). The seed zone map indicates areas where seeds can be safely planted to preserve genetic structure and local adaptation. Forest Service silviculturists and geneticists use these seed zones to guide their seed collecting and planting decisions for ecological restoration projects that include reforestation. In California, there is one seed zone map for all tree species. The basic rule of thumb is that a reforestation project will use seeds from the same seed zone and within a 500 foot elevation band from the planting site (Buck et al. 1970). Recently, many of these reforestation projects have been organized as part of a post-fire response to help facilitate the recovery of forests after catastrophic wildfire.

Although the California seed zone map has done well in guiding reforestation efforts in the past, its future effectiveness is under question due to projected climate change (Erickson et al. 2012). The map was created under an assumption of a static climate, and that assumption is no longer sound (IPCC 2007). New information is needed to establish best management practices for reforestation efforts in the Sierra Nevada.

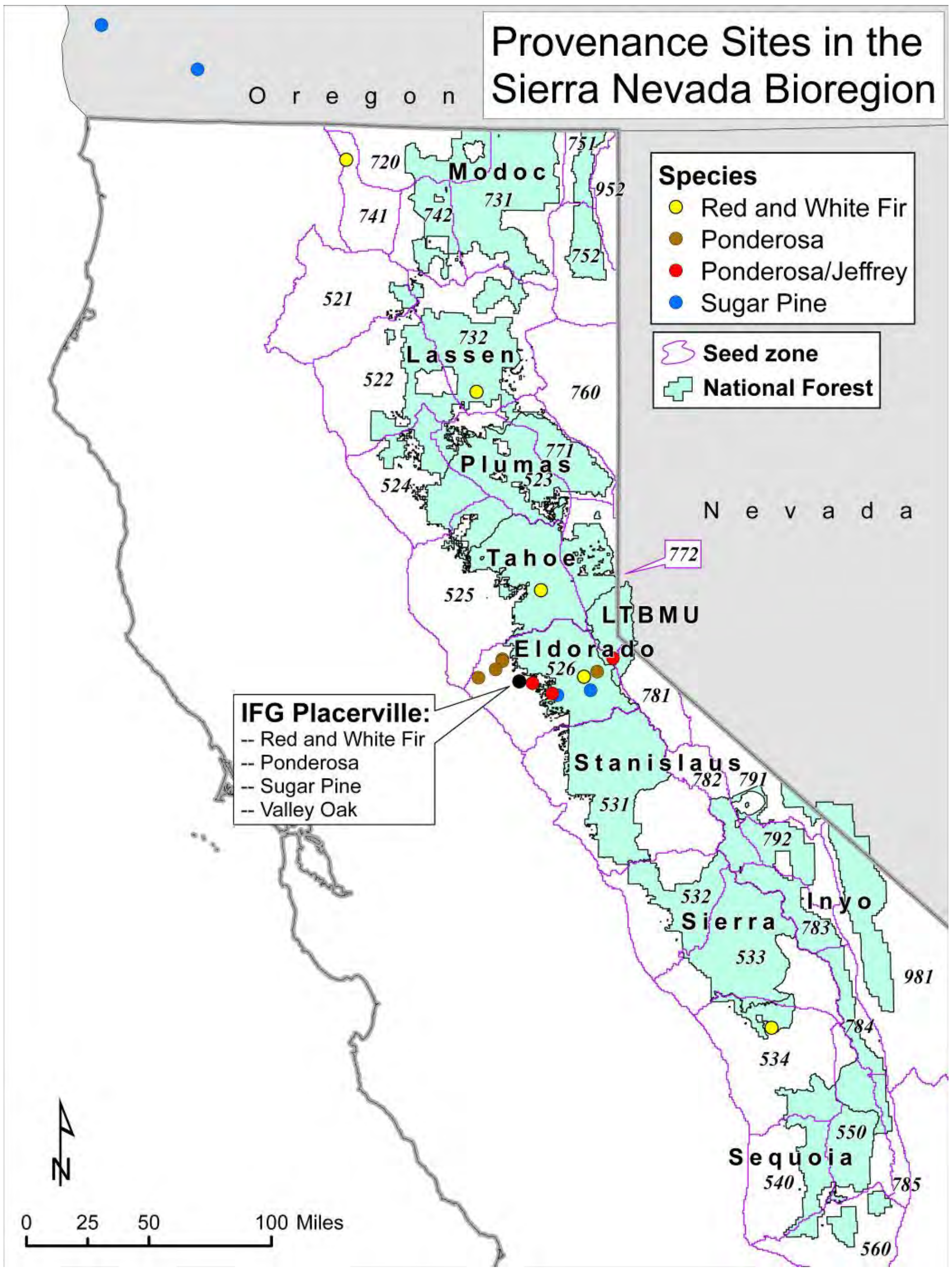


Figure 1: Seed zones (with purple outlines; the numbers identify each of the zones) and provenance tests established by PSW (circles) within the national forests (aqua areas) that are in the focus area of this review. The map includes two main seed zone series, 500 for west slope and 700 for east slope, within the Sierra Nevada

synthesis area. The colored dots show locations of provenance tests in a number of conifer and hardwood species.

Provenance Tests

Provenance tests are an important source of genetic information that can be used to inform land management. These are studies where seeds are collected from across the range of a species and grown in a common garden or gardens (ideally, multiple planting sites are used to allow a comparison between the different planting environments). Tests can be designed to focus either on variation within a species or variation among species by planting a number of different species in a single test.[1] Because seeds are moved from one climate environment to another, the response of each genotype to that novel climate can be measured. In general, provenance tests have revealed temperature to be the most important climate variable determining tree survival and growth (reviewed in Aitkens et. al 2008). However, some analyses do show that precipitation is an important factor in determining performance (e.g., Ukrainetz et al. 2011). Interestingly, for the Sierra Nevada, predictions from future climate models yield more consistent estimates for mean annual temperature (MAT) than for mean annual precipitation (MAP) (Safford et. al. 2012). Added to that is the geographic complexity found in the mountain ecosystems of the Sierra Nevada (Millar et al. 2007). The availability of water is dependent on MAP, but also on local, micro-scale topography and soils, which determine what happens to rain after it falls and washes down the mountain slope. As a result, two trees growing 10 meters from each other could have very different amounts of available water, but they would still be experiencing the same MAT. Clearly, even modeling current climate in a mountain environment is challenging.

An example of a very comprehensive provenance test is found in British Columbia, Canada. The Illingworth test in lodgepole pine (*P. contorta* spp*. latifolia, murrayana,* and *contorta*) was established in 1969, using 140 source populations, with 62 different test sites established, for a total of 69,120 seeds sown. The data from this test have been used to determine the response of populations to novel climates (Rehfeldt et al. 1999, O'Neill et al. 2008, Wang et al. 2010). Rehfeldt et

The Harrel provenance test in Sugar pine includes 124 difference sources of Sugar Pine, all grown in a single common garden.

[1] For a recently planted current example, see http://www.for.gov.bc.ca/hre/forgen/interior/AMAT.htm

al. (1999) performed a series of regression analyses to determine how the different source populations would respond to a changing climate. O'Neill et al. (2008) expanded this analysis to create a universal transfer function (UTF), which allows prediction of current and future forest productivity "for any population in any location" (Page 1041) using current climate and different models of climate change. Wang et al. (2010) modeled forest height for lodgepole pines in British Columbia under climate change, including both forests planted with local planting stock as well as planting stock predicted to have the best height growth using the UTF under the A2 "enterprise" climate scenario. They found very dramatic changes between the two models, with the modeled forests planted with the "ideal" planting stock being much taller than those that used the local planting stock.

This analysis was possible because of the existence of the Illingworth Provenance Test in British Columbia. In California, there are provenance test resources established by PSW for sugar pine, red and grand fir, ponderosa pine, and Jeffrey pine (e.g., Conkle 1973, Kitzmiller 2004, Kitzmiller 2005) (Figure 1). However, the largest number of planting sites for these tests was four—much fewer than the 62 sites used in Canada. For the provenances that were tested, similar analyses are being performed (J. Wright, Pers. Comm.). Past analyses have shown evidence for local adaptation in both sugar pine (Kitzmiller 2004) and ponderosa pine (Kitzmiller 2005). However, with limited existing provenance test resources, it is not possible to match the level of detail that can be achieved in British Columbia for lodgepole pine. If something approaching the level of detail achieved in Canada is desired for California, investment would be needed to establish additional tests in a broader range of species, as well as across a broad range of planting sites, including those outside of traditional planting areas. Region 5 has begun to establish climate adaptation plots that are designed to test the response of seedlings moving uphill within a seed zone. Plots were established within ecological restoration projects, with as much variation as possible in elevation. These sites have a great deal of potential to inform future ecological restoration projects by determining how far uphill trees can be moved within a seed zone and still survive and grow, however, they will require attention over the next several decades to obtain their full value. Because these test plots were all established using material from within the same seed zone, they are not useful for testing how seeds perform in different seed zones. While provenance tests take a long time to show results for 30-year growth in a particular climate, information can be begin to be obtained from tests starting on the day they are planted. Seedling growth and survival can be assessed from the very beginning of the experiment.

It is important, however, to point out the drawbacks and limitations of provenance test data. First are the silvicultural techniques used to establish each of the planting sites. Both to reduce environmental variation and to make sure as many seedlings survive the transplanting process as possible, sites are highly prepared, and seedlings are often watered until they are well established (Aitken et al. 2008). This can result in conditions that are very different from operational tree plantings, which often use the "plant and pray" method—seedlings are planted but not watered, and often not kept clear of competing vegetation. Another issue, particularly with historical tests, is that test sites were not established in marginal environments or outside of the range of the species (O'Neill et al. 2008). These tests were often established with silvicultural goals—finding the best seed sources for particular sites—and there was little point in learning about sites where trees did not thrive. However, under climate change, these could be some of the most important sites to include in future studies. Finally, provenance tests do not test the impacts of other important and potentially interacting factors besides climate, including fire, insect pests, or diseases (Aitken et al. 2008, O'Neill et al. 2008). A tree can be optimally adapted to grow in a certain environment, but it hardly matters if it is killed by beetles or is not resistant to a local pathogen.

Recognition and Management of Provenance Test Sites on National Forests

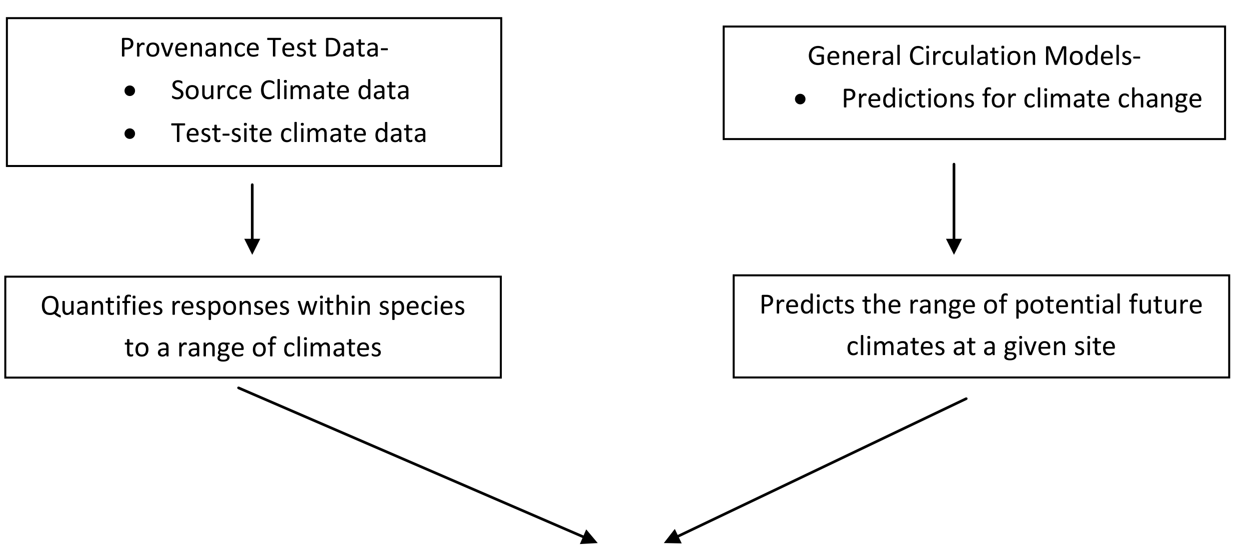
As a final note, it is important that land and resource management plans recognize the significance of provenance test plots located within national forests (Figure 1). Along with other kinds of special management areas designated for research purposes, these sites are investments whose value can be lost if they are inadvertently or randomly treated without direction from the researchers assigned to oversee them. Although sites should not necessarily be deferred from treatment, if treatments are needed, they should be carefully implemented to avoid impacts to the research value of the site.

Management Implications

There is active scientific debate about whether managers should move beyond local sourcing of revegetation materials toward the managed relocation of genotypes that appear better adapted to expected climate change; however, there is broad consensus for using common garden experiments or provenance tests to prepare for projected conditions by better understanding how genetic variability can improve ecological restoration (Figure 2).

Conclusion

In summary, combining data from provenance test studies with our current understanding of predicted climate change can be a powerful tool for informing reforestation efforts (Figure 2) (Mátyás 1994, Mátyás 1996). However, the limitations of both sources of data need to be understood in order to inform an approach to ecological restoration that reduces risk and promotes the highest chance of successful reforestation.



Model the performance of a given seed source at a given site in current and future climate conditions.

Figure 2: Flowchart illustrating how climate modeling and genetic information can help suggest seed sources to promote forests that are more resilient to climate change.

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4.0 Fire

All sections of the synthesis are concerned with fire because of its role as a dominant ecological process in the Sierra Nevada and southern Cascades. Fire has long influenced the diverse natural and cultural resources of the synthesis area, including air quality, human health, infrastructure, community well-being, soils, timber, terrestrial and aquatic wildlife, and water resources. The first chapter, Fire and Fuels (4.1), summarizes recent literature relevant to fire and forest management in yellow pine, mixed-conifer, and upper montane forest types in the synthesis area. It also discusses the historical role of fire in the region and describes potential outcomes of various management strategies—both currently and in the future. The second chapter, Fire and Tribal Cultural Resources (4.2), focuses on the pivotal role of fire in sustaining culturally important plants and opportunities to learn about the effects of Native American burning practices. The Post-wildfire Management chapter (4.3) considers both short-term responses to fire, including salvage logging, and longer-term management and restoration of post-fire landscapes. That chapter was added in response to concerns from managers, as well as the expectation that climate change will increase potential for more severe wildfires (see Synopsis of Climate Change (1.3)). As an increasing amount of forest land in the synthesis area has been affected by major wildfires, more restoration plans are being developed. The Forest Service in California recently developed a post-fire restoration strategy template to help guide national forests in planning for restoration and long-term management of burned landscapes. Together, the chapters in this section provide guidance for managing fire to minimize its undesirable outcomes while harnessing its power to rejuvenate ecosystems and increase their resilience.

4.1 Fire and Fuels

Brandon Collins and Carl Skinner

Contents

Introduction

Fire is an inherent process in most Sierra Nevada, southern Cascade Range, and montane Modoc Plateau forest types, where it has not only been a regulating mechanism, but it has also been the dominant force shaping forest structure within stands as well as patterns across landscapes (Riegel et al. 2006, Skinner and Taylor 2006, van Wagtendonk and Fites-Kaufman 2006). This chapter summarizes recent literature relevant to fire and forest management in several key forest types of the Sierra Nevada and southern Cascade Range: yellow pine (*Pinus ponderosa* and *P. Jeffreyi*) and mixed-conifer forest types. Red fir (*Abies magnifica*) forest types are addressed in the Forest Ecology chapter. The literature summarized and the implications discussed in the following chapter apply primarily to the forested area outside of the wildland-urban interface (WUI). Issues that are more specific to the WUI are addressed in the social science chapters (section 9.0).

Fire in Sierra Nevada Ecosystems

Historical Role of Fire

There are numerous studies demonstrating the integral role that fire played in shaping historical (i.e., pre-Euroamerican settlement) forest structure and composition in the focal area. These studies, which are largely from mixed-conifer, ponderosa pine, and Jeffrey pine forest types, demonstrate frequent occurrence of generally low- to moderate-severity fire over at least the last several centuries. The general consensus from these studies is that frequent fire maintained relatively open, patchy stands composed of primarily large, fire-resistant trees. Although this was likely the case for many areas within these forest types, to surmise that those stand conditions were ubiquitous throughout the Sierra Nevada would be a gross oversimplification. Recent studies of historical fire occurrence have gone beyond solely reporting fire frequency by reconstructing historical forest structure and characterizing spatial patterns resulting from more natural fire – forest interactions in the Sierra Nevada (e.g., Nagel and Taylor 2005, Scholl and Taylor 2006, Beaty and Taylor 2007, Beaty and Taylor 2008, Scholl and Taylor 2010, Knapp et al. 2012) and in the southern Cascade Range (Taylor 2000; Beaty and Taylor 2001; Bekker and Taylor 2001; Norman and Taylor 2003, 2005). These studies indicate a high degree of spatial complexity driven by heterogeneity in vegetation/fuels and topography and influenced by variability in climate, which mediates the timing, effects, and extents of fires over time. Notably, the great difference between the gentle topography of the Cascade Range and the more complex topography of the Sierra Nevada creates considerable differences in how fire functioned historically in the two mountain ranges. As a result of the differences between the two mountain ranges, the following discussion is mostly relevant to the Sierra Nevada and may not be as relevant to the Cascade Range or the Modoc Plateau. Please see Skinner and Taylor (2006) for a discussion of fire in the Cascades and Riegle et al. (2006) for the Modoc Plateau area. The complexity of factors influencing fire regimes in the Sierra Nevada makes it difficult to distill quantitative information relevant to restoring Sierra Nevada forests, but there are several general themes that may inform management activities as they relate to restoration and resilience. Note that these themes are generally applicable to the mixed-conifer, ponderosa pine, and Jeffrey pine forest types; however, there are moisture/productivity gradients within individual forest types, as well as across types, that influence key fire regime characteristics: frequency and severity.

- ***Topography*** : Several reconstruction studies demonstrate that topography strongly influenced historical fire regimes (Taylor and Skinner 1998, Beaty and Taylor 2001, Taylor and Skinner 2003, Beaty and Taylor 2008). This effect, however, appears to be moderated by the topographic complexity of a particular area, i.e., in landscapes with complex geomorphic structure, topography may have been the dominant influence driving patterns in fire effects, whereas in more gentle landscapes patterns in fire effects were driven more by the interactions between vegetation/fuel and topography (Skinner and Taylor 2006, Skinner et al. 2006). In more complex landscapes, upper slope positions tended to experience greater proportions of high-severity fire, whereas lower slope positions had lesser proportions. This pattern appears to exist almost independent of the vegetation/fuel structure in a particular landscape. The greater proportions of high-severity fire on upper slopes may have been exacerbated on south- and west-facing slopes, where more exposure and drying of fuels tends to coincide with more pronounced upslope and upcanyon winds. In more gentle landscapes of the Sierra Nevada, it appears that greater proportions of high-severity fire were associated with more mesic forest types (e.g., forests with greater component of fir (*Abies* sp.)). The mesic conditions could be a function of more northerly aspects and/or higher elevation. It should be noted that there are reconstruction studies that demonstrate no effect of topography on fire regime and forest structure characteristics (e.g., Scholl and Taylor 2010). It is unclear to what extent other factors may be masking more site-level influences (e.g., ignition sources/patterns, cold-air pooling).

- ***Riparian areas***: In many riparian sites, reconstruction studies have demonstrated historical regimes of frequent fire that do not appear to differ from adjacent upland areas (Skinner 2003, Van de Water and North 2010). However, results from these studies do suggest that perennial streams, which may have greater influence on understory vegetation, fuel moistures, and/or relative humidity, do have noticeably lower fire frequency than adjacent upland areas. It is suggested that these riparian areas may have acted as filters—not simply barriers—for fire spread, as fires tended to burn through these areas (or burn with enough intensity to scar surviving trees) only when conditions were more favorable for fire spread (e.g., during drought conditions or substantial wind events) (Skinner 2003).

- ***Eastside (Sierra Nevada) pine forests***: There are far fewer historical reconstruction studies in forests on the eastern side of the Sierran crest than there are for mixed-conifer forests on the west slope. Based on the few studies in eastside pine, it appears that fire frequency and inferred fire effects were generally similar between eastside pine and westside mixed-conifer forests (Taylor 2004, Moody et al. 2006, Gill and Taylor 2009, North et al. 2009b, Vaillant and Stephens 2009). There are, however, context-specific distinctions that suggest some differences exist in fire regimes between eastside pine and westside mixed-conifer: 1) In contrast to the larger expanses of contiguous forests on the westside, eastside forests are sometimes isolated in canyons or on benches in discrete stands (North et al. 2009b); this isolation results in longer fire return intervals for some eastside stands and greater variability in fire frequency and fire effects. 2) Several sampled stands in the eastside pine type maintained frequent fire regimes as late as the early- to mid-1900s (North et al. 2009b), whereas frequent fire in many westside mixed-conifer forests ceased around the 1880s. The structural changes associated with cessation of fire could be different as a result of these different cessation dates. The contemporary forest conditions in the Jeffrey pine-mixed-conifer dominated area of the Sierra San Pedro Mártir (Baja California) serve as a relevant reference site for eastside pine forests (Stephens and Fulé 2005). This area has experienced very little timber harvesting, and fire suppression only dates back to the 1970s (Stephens et al. 2003). This forest has an open, all-aged structure, with its most salient characteristic being high spatial variability (Stephens and Gill 2005, Stephens et al. 2008). This variability not only pertains to spatial arrangement and sizes of trees, but also to coarse woody debris and tree regeneration patches (Stephens and Fry 2005, Stephens et al. 2007).

- ***Cascade Range fire regimes***: The historical reconstructions of fire in these forests depict fire regimes considerably different than those of the Sierra Nevada. Further, many studies have focused on the upper montane forests (Taylor and Halpern 1991, Taylor 1993, Bekker and Taylor 2001, Taylor and Solem 2001) in addition to the mid-elevation pine and mixed-conifer forests (Taylor 2000; Norman and Taylor 2003, 2005). The more gentle topography of the Cascade Range affords conditions where fires are able to spread rather easily over large areas without significant interruption. Especially on the eastside of the range in the pine and mixed-conifer forests, pre-suppression era fires were not only primarily frequent, low- to moderate-intensity fires, but were also quite large. Fires of this type covering 10s to 100s of thousands of acres occurred on average once every 20 years (Norman and Taylor 2003). Though they burned less frequently than lower and middle elevation forests, the upper montane forests with mixed-severity fire regimes burned much more frequently than similar forests of the Sierra Nevada (Taylor and Halpern 1991; Taylor 1993, 2000; Bekker and Taylor 2001; Taylor and Solem 2001). The gentle topography of the Cascades, combined with continuity of vegetation (fuels) from lower to higher elevation, allowed fires to burn more regularly in the higher elevations (Skinner and Taylor 2006). This is in contrast to the very rocky, vegetatively broken landscapes of the Sierra Nevada, where it is more difficult for fires to move about so freely in the upper montane.

- ***Landscape heterogeneity***: Differential fire effects over the landscape, including stand-replacing patches, contribute to coarse-grained heterogeneity across landscapes. This has been demonstrated for historical fire regimes (Beaty and Taylor 2008) and for areas with more intact, contemporary fire regimes (Collins and Stephens 2010). These studies suggest that stand-replacing fire was a component of Sierra Nevada mixed-conifer forests, but at relatively low proportions across the landscape (~5 – 15 percent), consisting mostly of many small patches (<4 ha) with few large patches (~ 60 ha). Based on these studies, it appears that landscapes with active fire regimes included relatively dense, even-aged stands and shrub patches, as well as the often referenced open, park-like, multi-aged stands. Actual proportions in each vegetation type/structure are largely unknown due to the limitations of historical reconstruction studies, although several studies have made estimates based on reconstructed tree ages and density (Taylor and Skinner 1998, Beaty and Taylor 2001, Taylor 2004, Beaty and Taylor 2008, Scholl and Taylor 2010, Taylor 2010).

- ***Climate*****:** Variability in historical fire occurrence is linked to both short- and long-term fluctuations in regional and synoptic climate (Swetnam 1993, Swetnam and Baisan 2003, Stephens and Collins 2004, Taylor and Beaty 2005, Trouet et al. 2006, Taylor et al. 2008, Beaty and Taylor 2009, Gill and Taylor 2009, Trouet et al. 2009, Trouet et al. 2010, Taylor and Scholl 2012).

    - Short-term climatic variation (e.g., annual to decadal scale): Although climatic fluctuations do not appear to have moderated fire effects, climate (particularly variation in precipitation) has been shown to drive fire extent (e.g., widespread fire years coincided with regional drought years, and were sometimes preceded by regionally wet years).

- Long-term climatic variation (decades to century scale): Fire frequency, or chance of having fires, appears to be associated with variation in air temperature (Swetnam 1993, Swetnam and Baisan 2003), with higher temperature associated with more frequent fires and longer fire seasons (Westerling et al. 2006, Westerling et al. 2008, Westerling and Bryant 2009). Precipitation appears to be associated with fire extent (Swetnam 1993, Swetnam and Baisan 2003). Thus, moist years produce vegetation that is available to burn in the inevitable dryer years that occur during otherwise moist periods.
- The rain shadow effect on the eastside of the Sierra Nevada and the tendency for greater stand isolation, primarily in the southern portion, appears to somewhat de-couple fire in eastside pine forests from synoptic climatic conditions (North et al. 2009b).

Though there are many important lessons to learn from the past, we may not be able to rely completely on past forest conditions to provide us with blueprints for current and future management (Millar et al. 2007, Wiens et al. 2012). In particular, the nature and scale of past variability in climate and forest conditions, coupled with our inability to precisely reconstruct those conditions, introduce a number of conceptual and practical problems (Millar and Woolfenden 1999). Detailed reconstructions of historical forest conditions, often dendroecologically based, are very useful but represent a relatively narrow window of time and tend to coincide with tree recruitment in the period referred to as the Little Ice Age, which was much cooler than present (Stephens et al. 2010). Therefore, manipulation of current forests to resemble historical forest conditions may not be the best approach when considering future warmer climates (Safford et al. 2012a). Rather, restoring the processes (mainly fire) that shaped forests for millennia may be a prudent approach for hedging against uncertainties around maintaining fire-adapted forests (Fulé 2008). This is not to suggest that any incorporation of fire into these forests would be appropriate. A more suitable goal, albeit a more difficult one, would be to restore the forest stand and landscape conditions that would allow fires to function in what is generally believed to be a more natural way.

Altered Ecosystems

Past harvesting practices and livestock grazing, coupled with over a century of fire suppression, have shifted forest structure and composition within the ponderosa pine, Jeffrey pine, and mixed-conifer types of the Sierra Nevada. This shift is generally characterized by increased tree densities, smaller average tree diameters, increased proportions of shade-tolerant tree species, and elevated surface fuel loads relative to historical or pre-European settlement forest conditions (van Wagtendonk and Fites-Kaufman 2006, Scholl and Taylor 2010, Collins et al. 2011b). In addition to the stand-level changes within these forest types, fire exclusion and past management practices have led to considerable homogenization across landscapes (van Wagtendonk and Fites-Kaufman 2006, Hessburg et al. 2007, Perry et al. 2011). This homogenization is a product of several interacting processes: 1) widespread timber harvesting, primarily involving removal of larger trees left during earlier railroad and/or mining-related logging, 2) infilling of trees into gaps that were historically created and/or maintained by variable severity fire, and 3) forest expansion into shrub patches and meadows that were formerly maintained by fire. In addition to a loss of beta-diversity, these stand- and landscape-level changes have increased vulnerability of many contemporary forests to uncharacteristically high disturbance intensities and extents, particularly from fire and drought-induced insects/disease outbreaks (Guarin and Taylor 2005, Allen 2007, Fettig 2012). Following such disturbances, these forests and the species that depend on them have limited capacity to return to pre-disturbance states. This issue may be exacerbated if climate changes according to predictions in the next several decades, as large, high-intensity fires may become catalysts for abrupt changes in vegetation and associated species (i.e., type conversion).

Trends

Recent research has demonstrated increased proportion of high-severity fire in yellow pine and mixed-conifer forests the Sierra Nevada from 1984-2010 (Miller et al. 2009, Miller and Safford 2012). In addition, these studies demonstrate that fire sizes and annual area burned have also risen during the same period. The authors point out that these increases co-occur with rising regional temperatures and increased long-term precipitation. Westerling et al. (2006) also demonstrate increased area burned over a similar time period, which they attribute to regional increases in temperature and earlier spring snow melts. Despite these documented increases over the last few decades, California and the western U.S. as a whole are in what Marlon et al. (2012) describe as a large “fire deficit.” This is based on reconstructed fire occurrence over the last 1500 years using sedimentary charcoal records. Marlon et al. (2012) argue that the current divergence between climate (mainly temperature) and burning rates is unprecedented throughout their historical record. In other words, with temperatures warming as they have been over the last several decades, we would expect to see much higher fire activity, based on historical fire-climate associations. This divergence is due to fire suppression policies, which, as the authors point out, may not remain effective over the long-term if warming trends continue. It is likely, given increasing temperature and the precipitation patterns since the onset of fire suppression, that fire activity would have increased over the twentieth century rather than decreased had fire suppression not been implemented (Stine 1996, Skinner and Taylor 2006), further exacerbating the current fire deficit.

Notable increases in fire activity are predicted for California, and they are driven largely by projected increases in temperature and decreases in snow pack and, to a lesser extent, increased fuel production from CO_2 “fertilization” (Flannigan et al. 2000, Lenihan et al. 2003, Lenihan et al. 2008, Westerling et al. 2011). It remains unclear how these increases in fire activity would be manifested in Sierra Nevada forests (Safford et al. 2012a). Increased area burned does not necessarily result in increased proportions of high-severity fire (Miller et al. 2012a). However, one of the potential ramifications of decreased snowpack forcing longer fire seasons is that the probability of fire occurring on a given spot increases, potentially resulting in shorter intervals between successive fires. This may not be a problem if fire severity is generally low to moderate, with lesser proportions of high severity occurring in small patches. However, if high-severity proportions and patch sizes are elevated (Miller and Safford 2012), decreased time between successive fires could lead to type conversion or local loss of a particular plant association (Safford et al. 2012a). Further, even if proportions are not elevated but remain similar, this would translate into greater area burned at high severity as total burned area increases.

Effects of Ecosystem Management Strategies

Passive Management (No Action, with Continued Fire Suppression)

There is little evidence to suggest that passive management in Sierra Nevada forests will result in increased resilience to stressors (drought) or disturbance (fire, insects), and in fact, there is evidence to the contrary (Agee 2002). A recent study demonstrates that crown fire potential in untreated stands continues to increase over time (Stephens et al. 2012). Modeling studies at the landscape scale also predict much greater losses from wildfire in untreated scenarios than in fuels-treated scenarios (Ager et al. 2007, Finney et al. 2007, Schmidt et al. 2008, Ager et al. 2010a, Collins et al. 2011a). Stephens and Moghaddas (2005a), however, reported that relatively untreated mixed-conifer stands with little understory and ladder fuels had generally low torching potential. These stands, which were 80-100 years old, regenerated naturally after early railroad logging and were subjected to minimal or no silvicultural treatments throughout their development (except full fire suppression). However, stands with similar structure (closed stem exclusion phase, *sensu* O'Hara et al. (1996), with relatively low surface and ladder fuels) and management history are probably rare in the Sierra Nevada. The prevailing evidence, both from studies of fire effects following actual wildfires and from studies reporting modeled wildfire effects, demonstrates that untreated stands (no action) are more prone to crown fire initiation and high fire-induced mortality (Stephens and Moghaddas 2005b, Ritchie et al. 2007, Symons et al. 2008, Safford et al. 2009, Safford et al. 2012b).

Vegetation Management

Fuels reduction is becoming the dominant forest management activity in dry forest types throughout the western U.S. The primary objectives of these activities are to modify wildland fire behavior in order to protect private property and public infrastructure, minimize negative impacts on forests (Agee and Skinner 2005), enhance suppression capabilities (Agee et al. 2000), and improve firefighter safety (Moghaddas and Craggs 2007). In drier Sierra Nevada forest types, objectives for fuel reduction treatments can often be aligned with those aimed at increasing ecosystem resilience through restoration treatments (McKelvey et al. 1996, Weatherspoon and Skinner 1996). One key potential difference between a fire hazard versus restoration focus is the incorporation of variability in both residual stand structure and surface fuels, which for a restoration-focused treatment would involve creating more horizontal and vertical spatial variability that would include retaining clumps of trees and woody debris (North et al. 2009a). This clumpiness could result in local tree torching, and thus overstory tree mortality, under wildfire conditions. Torching potential within the denser clumps would likely exceed that in stands treated for fire hazard reduction, in which the goal is to more uniformly raise canopy base height and reduce surface fuels. Though research is underway to more directly assess the relative differences between the two treatment strategies (Knapp et al. 2012) there are no recent published results. However, early publications recognized the importance of spatial variability and was described by Show and Kotok (1924) this way: "The virgin forest is uneven-aged, or at best even-aged by small groups, and is patchy and broken; hence it is fairly immune from extensive devastating crown fires."

The activities carried out in fire hazard reduction- or restoration-focused treatments include fire (either prescribed or managed wildland fire), mechanical (e.g., thinning, mastication, chipping), or a combination of the two. In field-based experiments, Stephens and Moghaddas (2005b), Schmidt et al. (2008), and Stephens et al. (2009) all found that prescribed fire alone effectively reduces surface fuels, thus reducing modeled spread rates, fire line intensities, and flame lengths under a range of weather conditions. In addition, these studies also demonstrate substantial reductions in ladder fuels in areas treated with prescribed fire. However, as fire-killed trees fall and contribute to surface fuel pools, the overall effectiveness in reducing potential fire behavior can be short-lived (Skinner 2005, Keifer et al. 2006). It is likely that in dense, fire-excluded stands, multiple burns will be needed to achieve more long-lived effects (Stephens et al. 2009). Thinning effectiveness depends on the type of thinning performed and the subsequent treatment of activity fuels (Agee and Skinner 2005). In fire-excluded forests, fuel reduction prescriptions often aim to both reduce ladder fuels (increase canopy base height) and increase crown spacing (reduce crown bulk density), in combination with removing activity and existing surface fuels (e.g., piling and burning or broadcast underburning) (Agee and Skinner 2005, Stephens et al. 2009). Whole-tree harvests have also been shown to effectively reduce modeled fire behavior (Schmidt et al. 2008, Stephens et al. 2009) and actual fire effects (Ritchie et al. 2007, Symons et al. 2008). Data on tree mortality in thinned areas burned by wildfires, which demonstrate greater survivability in areas underburned following thinning, serve as real-world tests on the importance of treating activity fuels following thinning (see Raymond and Peterson 2005, Ritchie et al. 2007, Symons et al. 2008, Safford et al. 2012b). It is worth noting there are instances in which extreme fire behavior (e.g., plume collapse, extreme wind) can overwhelm even well-designed fuel treatments, and lead to high tree mortality (Finney et al. 2003, Werth et al. 2011).

One concern regarding treatments that reduce tree densities and increase canopy base heights is that more open stands could experience greater windspeeds and reduced fuel moistures (Countryman 1956). It has been suggested that these potential microclimatic changes could contribute to increased fire spread rates and surface fire intensities under wildfire conditions. However, a recent study by Bigelow and North (2011) demonstrated only modest increases in wind gust speeds and no significant differences in fuel moisture between treated and untreated stands. Findings from Estes et al. (2012) also demonstrate little to no effect of thinning on fuel moistures, particularly during peak fire season in northern California. The results from these studies suggest there is little evidence that the microclimatic changes associated with fuels treatments will result in noticeably increased fire behavior, at least not in

Mediterranean climates, where long dry periods desiccate fuels irrespective of stand conditions. Furthermore, reductions in fire hazard through well-designed fuels treatments are likely to compensate for any potential increases in fire behavior (Weatherspoon and Skinner 1996).

Plantations present a unique concern across managed landscapes in the Sierra Nevada. Plantations are generally dense, have uniformly low canopy base heights, and can often have shrub understories. These characteristics make plantations particularly susceptible to lethal fire, whether by high-intensity fire in tree canopies or from excessive heat produced by moderate-intensity surface fires (Weatherspoon and Skinner 1995, Kobziar et al. 2009, Thompson and Spies 2010). Recent research has demonstrated that prescribed fire treatments, either before plantation establishment (Weatherspoon and Skinner 1995, Lyons-Tinsley and Peterson 2012) or following establishment (Kobziar et al. 2009), can be effective at increasing tree survivability in wildfire. It should be noted that even under prescribed fire conditions, trees in plantations are fairly vulnerable to cambial kill or crown scorch (Knapp et al. 2011). Post-establishment mastication in plantations (shrubs and small trees) may be able to reduce fire behavior (flame length, rate of spread) under wildfire conditions sufficiently to aid in fire suppression activities, but does not appear to be very effective at reducing tree mortality (Kobziar et al. 2009, Knapp et al. 2011). However, if masticated fuel beds are allowed to decompose for a decade or so, fire hazards can be substantially reduced (Stephens et al. 2012).

Landscape-scale Considerations

The large wildfires that are occurring annually throughout the Sierra Nevada demonstrate the pressing need to scale up insights gained at the stand level to landscapes. The effort required for planning and analysis of alternatives tends to force larger project areas, encouraging actions at the landscape scale. Yet implementing fuels treatments across an entire landscape may not be consistent with desired conditions and/or may not be operationally feasible (i.e., funding, access, land designations – e.g., wilderness, etc.) (Collins et al. 2010). In response, fire scientists and managers have conceptually developed and are refining methods for the strategic placement of treatments across landscapes (Weatherspoon and Skinner 1996; Finney 2001, 2004; Stratton 2004; Finney et al. 2007). The basic idea is that an informed deployment of treatment areas—a deployment that covers only part of the landscape—can modify fire behavior and effects for the entire landscape. Due to the complexity of modeling fire and fuels treatments across landscapes (e.g., data acquisition, data processing, model execution, etc.), fuels treatment project design is often based on local knowledge of both the project area and past fire patterns. Recent studies in the northern Sierra Nevada and southern Cascade Range suggest that these types of landscape-level fuels treatment projects (where treatment arrangement is based more on local knowledge and fairly simple fire behavior modeling rather than intensive modeling associated with an optimization approach) can be quite effective at reducing potential fire behavior at the landscape scale (Schmidt et al. 2008, Moghaddas et al. 2010, Collins et al. 2011a).

Although only a few studies have explicitly modeled effectiveness of landscape fuels treatments using different proportions of treated area, there are some common findings: 1) although noticeable reductions in modeled fire size, flame length, and spread rate across the landscape relative to untreated scenarios occurred with 10 percent of the landscape treated, the 20 percent treatment level appears to have the most consistent reductions in modeled fire size and behavior across multiple landscapes and scenarios (Ager et al. 2007, Finney et al. 2007, Schmidt et al. 2008, Ager et al. 2010b); 2) increasing the proportion of area treated generally results in further reductions in fire size and behavior, however, the rate of reduction diminishes more rapidly when more than 20 percent of the landscape is treated (Ager et al. 2007, Finney et al. 2007); 3) random placement of treatments requires substantially greater proportions of the landscape to be treated compared to optimized or regular treatment placement (Finney et al. 2007, Schmidt et al. 2008); however, Finney et al. (2007) note that the relative improvement of optimized treatment placement breaks down when larger proportions of the landscape (~40 – 50 percent) are excluded from treatment because of land management constraints that limit treatment activities. It should be emphasized that this is not to preclude treating more than 20 percent of a landscape to achieve restoration, resilience, or other resource objectives. These studies suggest that when beginning to deal with fire hazard in a landscape, the initial objective would be to strategically reduce fire hazard on between 10 and 20 percent of the area to effectively limit the ability of uncharacteristically high-intensity fire to easily move across the landscape. This would buy time to allow restoration activities to progress in the greater landscape.

In designing landscape-level fuel treatment or restoration projects, there are often conflicts between reducing potential fire behavior and protecting/conserving other resources (Collins et al. 2010). One common conflict is habitat for wildlife species of concern (e.g., California spotted owl (*Strix occidentalis occidentalis*), Pacific fisher (*Martes pennanti*)). Often these species prefer multi-storied stands and/or closed canopies for nesting or denning habitat (Solis and Gutiérrez 1990, Weatherspoon et al. 1992, Spencer et al. 2008). Although it has been argued that fire suppression and past harvesting practices have created much of the habitat that is being called 'desirable' for many of these species (see Spies et al. 2006), the species-specific approach toward managing forests continues to prevail (Stephens and Ruth 2005). This approach limits the timing and intensity of fuel treatments. As a consequence, a manager's ability to modify potential fire behavior, particularly fast-moving, high-intensity fire, in forests with prolonged fire exclusion is restricted. Furthermore, regulations on forest management within and around nesting centers or natal dens (e.g., protected activity centers, or PACs), and riparian buffer zones affect the size and placement of fuels treatments across landscapes. Therefore, there is limited opportunity to apply "optimal" placement of fuels treatments to maximize the reduction in spread of intense fire across the landscape. Additionally, these protected areas are often highly productive and contain large amounts of live and dead fuel. Thus, these areas may be prone to exacerbated fire behavior, creating effects not only within these protected areas (Spies et al. 2006), but also carrying into adjacent stands.

The dynamic nature of forest ecosystems imposes an important temporal consideration on landscape fuel planning. A suite of fuels treatments deployed strategically across the landscape will have a characteristic lifecycle. As time since treatment increases, vegetation growth will contribute to fuel pools and rebuild fuel continuity (Agee and Skinner 2005, Collins et al. 2009). Thus, as stand-level treatments mature and become less effective at reducing fire behavior, the performance at the landscape-level will also decline (Collins et al. 2011a). Therefore, the design of landscape-level fuels treatments involves a trade-off between maximizing the fraction of the landscape area treated (if only once) and treating a limited area repeatedly to maintain treatment effectiveness (Finney et al. 2007). Empirical studies from wildfires (Collins et al. 2009; Martinson and Omi, in press) and studies based on modeled fire (Collins et al. 2011a, Stephens et al. 2012) suggest that treatments can be expected to reduce fire behavior for 10 to 20 years . Obviously, a number of factors contribute to this longevity: type and intensity of treatment, site productivity, forest type, etc. Ultimately, this balance between treatment longevity and landscape-scale effectiveness is going to be location specific, but it will require continual consideration in fire-adapted forest landscapes.

Fire Management

North et al. (2012) performed an analysis comparing current levels of fuels treatment across the Sierra Nevada to the estimated levels of historical burning throughout the range. They estimated that current treatment rates, which include wildfire area, account for less than 20 percent of the area that may have burned historically. Given that re-treatment intervals may need to be every 20-30 years depending on forest type, the authors argued that the current pattern and scale of fuels reduction and restoration treatments is unlikely to ever significantly advance restoration efforts, particularly if agency budgets continue to decline. Furthermore, because the estimate of treatment rates includes wildfire, regardless of severity, it is likely that North et al. (2012) overestimate current restoration efforts. Treating and then moving areas out of fire suppression into fire maintenance is one means of potentially changing current patterns. However, this approach would require a fundamental change in the objectives and scale of fuels treatments. Rather than treating areas to enhance fire suppression efficacy and continue to limit the spread of fire, which would only perpetuate the current hazardous fuels/fire deficit problem, the intent would be to implement fuels treatments that allow fire to occur such that fire effects were within a desired range across the landscape (Reinhardt et al. 2008). This type of strategy would not necessarily seek to achieve ubiquitous low-severity fire effects across a landscape. Instead, the aim would be to restore a fundamental ecosystem process that involves a range of fire effects consistent within the historical range of variability. Spatial fire modeling/fuels treatment tools have recently been developed to assist planning for transitioning toward a managed fire-dominated landscape (Vaillant et al. 2011, Ager et al. 2012). Minimizing ecological impacts associated with fire suppression activities (Backer et al. 2004) would be an additional benefit of transitioning towards increased use of managed fire.

A recent comparison of contemporary fire patterns (extent and severity) between lands managed by the Forest Service (FS) and National Park Service (NPS) in the Sierra Nevada revealed a significant distinction between the two agencies (Miller et al. 2012b). Across forest types analyzed, Miller et al. (2012b) demonstrated that the proportion of high-severity fire and high-severity patch size were smaller for NPS fires than for FS fires. In addition, their results demonstrate that overall fire extent was less on NPS lands. The authors point out that although in recent years the FS has begun to manage more wildfires for resource benefit, a policy of full suppression was in effect on most fires that occurred during their study period. In contrast, the NPS areas that they analyzed (all within Yosemite National Park) have a policy of only suppressing lightning-ignited fires when they occur outside their fire use zone or out of prescription, which resulted in most fires being managed for resource benefit. Miller et al. (2012b) suggested that by allowing most lightning fires to burn relatively unimpeded under a planned range of fire weather conditions, Yosemite has been able to achieve fire patterns that are closer to what may have occurred historically. This is not the case for the FS fires that were analyzed, which tend to burn under more extreme fire weather conditions, as these are the conditions under which fires generally escape initial fire suppression efforts (Finney et al. 2011). The authors do note that the NPS and FS lands included in the study have different land management histories, particularly with respect to timber harvesting, which have resulted in different contemporary stand structures, and could contribute to differential fire patterns. The NPS and FS lands in the study also had very different landscape contexts, with FS lands exhibiting a wider range of topographic and geomorphic configurations that would ultimately affect fire behavior. However, we must note that many of the FS fires analyzed were in the Cascade Range and Modoc Plateau where gentler, less complex landscapes more easily facilitate large fires and major fire runs (e.g., Fountain Fire 1992, Huffer Fire 1997) than the rocky, interrupted landscapes of Yosemite NP that were the focus of the Miller et al. (2012b) study.

Efforts to restore fire as an ecological process may be guided by metrics that help to quantify the effects of fire relative to reference conditions. One important set of metrics is the Fire Regime Interval Departure (FRID) geodatabase, which focuses on fire frequency (see sidebar). However, it is important to note that burning to achieve a particular interval between successive fires may not result in desired forest conditions. Clearly, fires were frequent in yellow pine and mixed-conifer forests of the synthesis area prior to Euro-American settlement. However, frequency alone did not appear to have generated the fine- and coarse-grained heterogeneity that has been associated with historical forest conditions. Rather, it seems that a range of fire effects over time, with a distribution skewed to the low- and moderate-severity, but including some stand-replacing effects, contributed to overall heterogeneity. The development of robust fire severity estimates derived from satellite imagery serves as a useful tool to quantify the distribution of fire effects both within individual fires (Collins et al. 2007) and across multiple fires throughout a region (Thode et al. 2011). It is important to emphasize that low-severity fire alone, even when applied multiple times, may not restore historical forest conditions (Miller and Urban 2000, Collins et al. 2011b). Reestablishing distributions of fire effects similar to historical conditions may prove difficult to achieve in the short-term in fire-suppressed forests, but it is a useful long-term goal for promoting socioecological resilience (SNEP 1996).

Sidebar: Fire Return Interval Departure (FRID) Metrics

Jonathan Long and Hugh Safford

Fire Return Interval Departure (FRID) is a measure of how much the frequency of fire has changed in recent years versus in a time before Euro-American settlement (Van de Water and Safford 2011). These data are fundamental for planning fuel treatments and restoring fire regimes, because they allow managers to identify areas at high risk of passing ecological thresholds due to altered fire regimes and their interactions with other factors (Van de Water and Safford 2011). High positive FRID values indicate areas that were characterized by frequent fire but have not experienced fire for many decades (Figure

1). FRID analyses can be combined with strategic considerations of fire behavior, topography, and values at risk to help identify priorities for fuel reduction and restoration of fire.

FRID maps are available for California from the US Forest Service Remote Sensing Lab. Unlike the national Fire Regime Condition Class (FRCC) program, which primarily measures departure from modeled conditions of vegetation structure, the California FRID data directly measure fire frequency departure. The FRID geodatabase includes several different metrics ("PFRID" metrics, based on percent departure) that account for the cumulative fire history of the national forests and adjoining areas since 1908. Another metric, the National Park Service (NPS) FRID index, compares the time since last fire against the pre-Euro-American fire frequency. The NPS FRID index is not structured to deal with areas experiencing more frequent fire today than under reference conditions (red areas in Figure 1), as is the case in much of low and middle elevation southern California and other areas where human activity and vegetation changes have made fire more frequent over time. The PFRID metrics extend into negative numbers to permit departure measurements under any scenario. Because the NPS FRID index weighs only the time since the most recent fire, it is most useful as a short-term performance measure, while the percentage-based metrics comparing long-term frequencies are better measures of actual fire restoration. Measures like mean, min, and max PFRID, which evaluate the influence of fire over a longer time scale, will be more helpful in targeting and tracking a longer term strategy to promote resilience to disturbance, a warming climate, and other stressors.

As with any simple metric, users should be cautious when interpreting the significance of FRID data or using them to plan treatments. FRID data do not account for non-fire silvicultural treatments and do not provide a measure of overall fire risk. Although FRID would be expected to be correlated to vegetation burn severity, FRID analyses do not directly account for expected fire intensity or burn severity. As a consequence, strategies need to consider other components of the fire regime, such as fire size, severity, and spatial pattern. Furthermore, a restoration strategy would take into account other factors, including forest productivity, aquatic ecosystems, wildlife habitat, social values, and other values at risk (Franklin and Agee 2003), as well as understandings of how fire may impact a landscape. As one example, mixed-conifer forests in areas of high productivity may be at more risk of uncharacteristically severe fire after missing only three or four fires than lower productivity ponderosa or Jeffrey pine forests that have missed more than four fires. However, because of the importance of fire frequency, FRID metrics can serve a useful role in measuring progress toward restoring a more natural role of fire as a dominant ecological process.

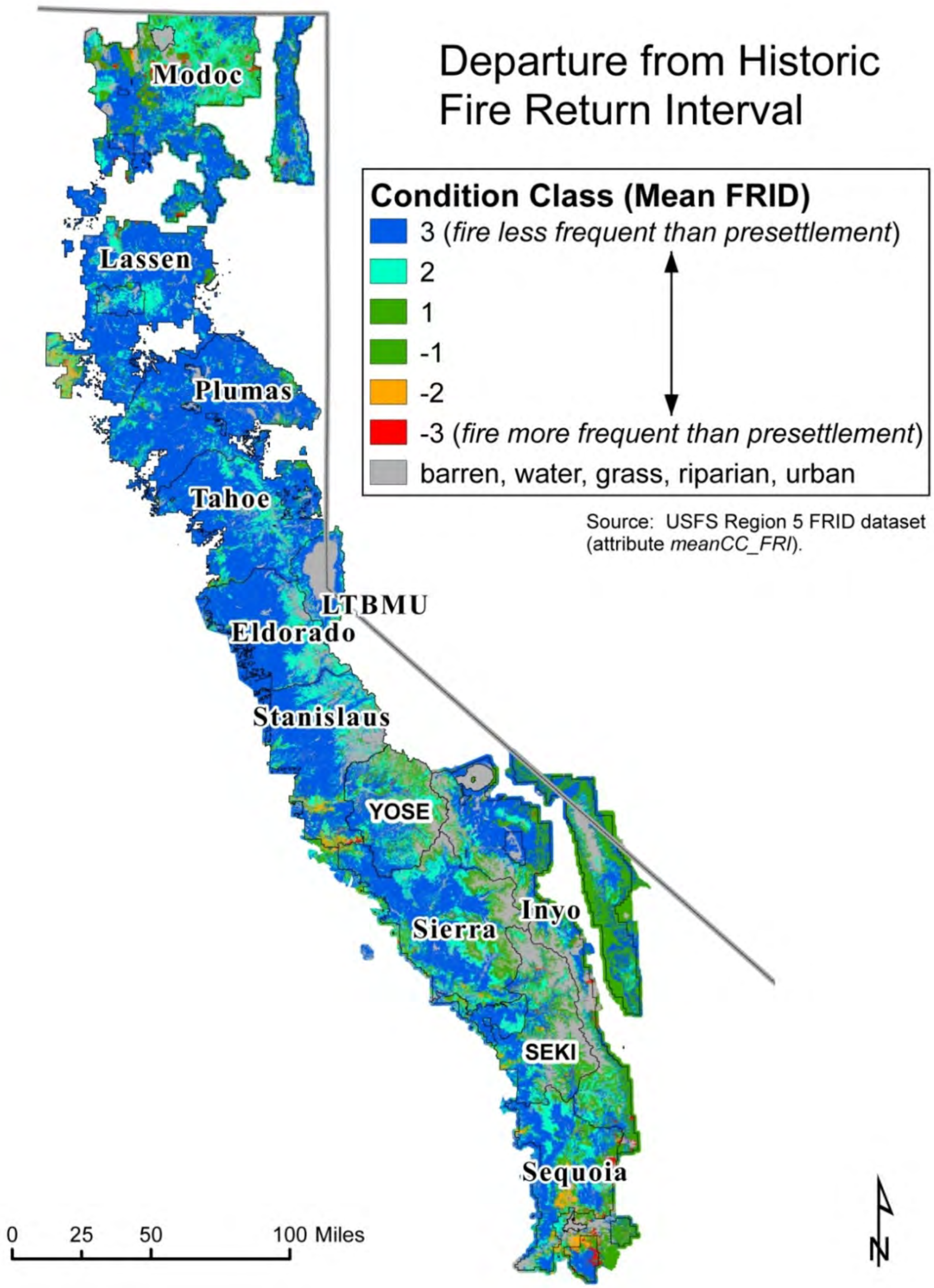

Figure 1: Mean percent fire return interval departure (mean PFRID) coded by condition class for the mountains of the Sierra Nevada and Southern Cascades. Negative condition classes indicate areas where fires have been burning more often than under pre-settlement conditions, while positive condition classes indicate areas where fires have been burning less often. Condition classes 1 and -1 are depicted with the same color because they both indicate conditions that are not greatly departed from the mean pre-settlement value. See Van de Water and Safford (2011) for more details regarding how the metric is calculated.

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4.2 Fire and Tribal Cultural Resources

Frank K. Lake and Jonathan W. Long

Contents

Executive Summary

Tribes regard plants that have evolved with frequent fire and other natural resources as living cultural resources that provide, water, food, medicines, and other material goods while also sustaining tribal cultural traditions. Collaborations between management agencies and tribes and other Native American groups can incorporate traditional ecological knowledge to facilitate placed-based understanding of how fire and various management practices affect such tribal cultural resources and values. Collaboration approaches reviewed in this chapter and in the Collaboration chapter (9.6) can foster restoration opportunities that would benefit Native American tribal communities and broader values. A strategy to promote socioecological resilience may include efforts to reestablish frequent fire regimes by emulating traditional burning practices, and to learn how the larger and more severe fires expected in the future may affect cultural resources and associated values.

Introduction

This chapter reflects several of the broader themes featured in this synthesis. First, it reinforces the perspective that humans are and have long been integral parts of ecosystems in the synthesis area. Therefore, to the extent restoration depends on reestablishing, at an appropriate scale, the disturbance regimes that have shaped ecosystems, it is important to consider opportunities to reestablish or emulate Native American forest practices, such as burning and harvesting. Second, this chapter emphasizes the importance of considering system dynamics at a range of scales, from individual organisms to large landscapes. Research has focused on small scale effects of tribal land management and traditional burning practices, such as how individual plants, patches, or sites respond, but the effects of tribal practices on larger vegetation communities and landscapes constitutes an important subject for further research (Anderson 2006). Lastly, this chapter recognizes that efforts to promote socioecological resilience would be incomplete if they did not consider how the widespread lack of fire in the synthesis area impacts contemporary uses of forest resources by Native Americans.

Many Native Americans [1] have a broad conception of cultural resources, which includes artifacts, structures, heritage sites, biophysical resources, and intangible resources (Welch 2012). Both wildfire and prescribed fire can affect cultural resources directly and indirectly (see Figure 1), with frequency, seasonality, extent, and severity of fires influencing those effects. Efforts to manage wildfires can also have lasting and detrimental effects on cultural resources through line construction, firing operations, and other suppression, mop-up, or post-fire rehabilitation activities (Ryan et al. 2012). Emphasizing the idea that critical resources have natural, economic, *and* cultural dimensions (see Social Section preface (9.0)), this chapter focuses on relationships between fire and tribal cultural resources, especially for living resources, such as plants, fungi, and animals, which have been actively managed to sustain them in their desired quantity and quality. Plants have been a particularly important focus of research on fire effects on cultural resources.

[1] Where this chapter focuses on cultural resources within the synthesis area, it refers to Native Americans. The term tribe is emphasized when discussing management strategies that are likely to be implemented through consultations, collaborations, or other interactions on a government to government basis. Those relationships, along with approaches to working with tribal traditional ecological knowledge, are considered in the Collaboration chapter (9.6).

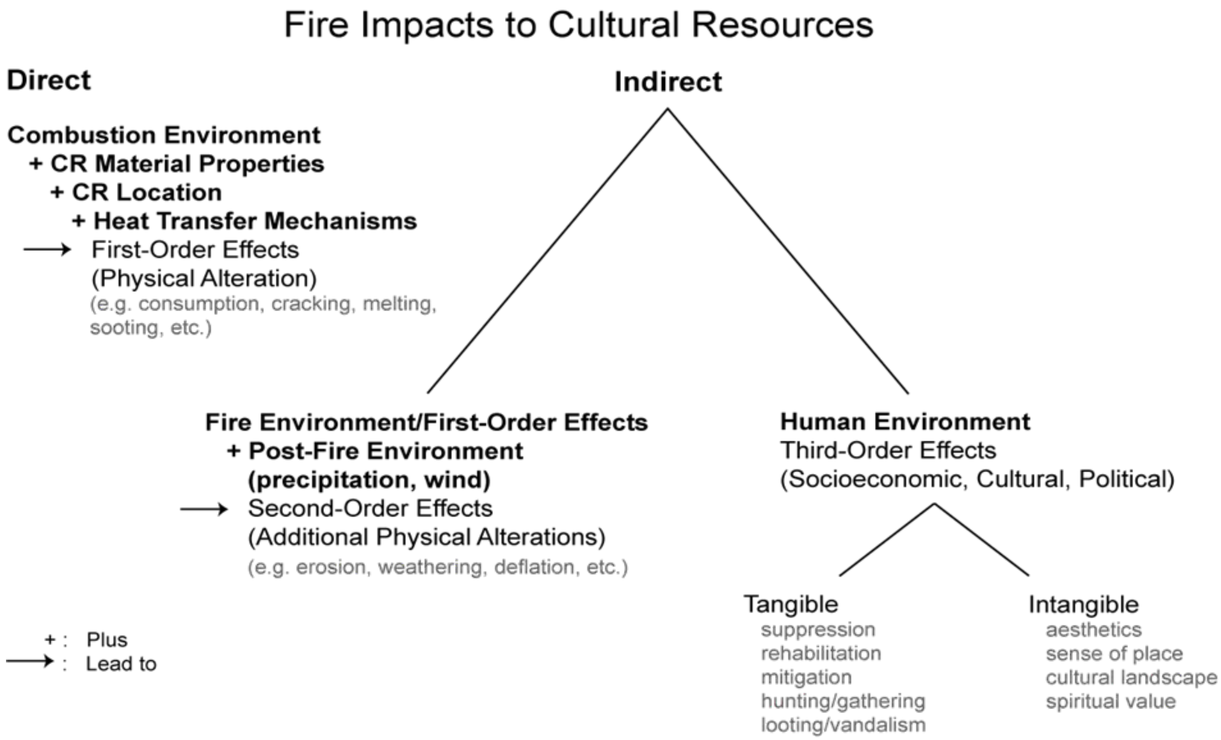

Figure 1: Direct and indirect fire effects on environment, cultural resources, and tribal values (Ryan et al. 2012: 12).

Burning to Promote Tribal Cultural Values and Ecological Restoration

Reestablishing traditional burning practices aligns well with ecological restoration goals, since many culturally valued plants produce berries, nuts, roots, and stems that support wildlife and provide other ecological services (Anderson 2006a, Anderson and Barbour 2003). Restoration efforts that support traditional tribal practices and subsistence activities can also promote other social and cultural values, such as native language, place names and maps, ceremonies, and other elements of cultural capital that perpetuate and maintain Native American traditions and associated ecosystems (Jordan and Shennan 2003, Long et al. 2003).

Understanding how Native American harvesting activities relate to ecological conditions at different scales may help management to promote valued cultural resources and broader restoration objectives. Native American practitioners today, as in the past, adapt and respond to areas of the landscape affected by fires to acquire resources of value (Anderson and Moratto 1996). Practitioners such as basketweavers harvest individual plants or patches at a variety of locations in a landscape (Anderson 2005). Securing enough resources for a tribal family requires access to different populations of target organisms distributed across diverse ecological communities. Consequently, small-scale treatments that enhance a few patches may only serve a small number of basketweavers (Anderson 1999). Tribal communities rely on access to a variety of ecological communities across a diverse landscape to maintain a variety of cultural traditions, including hunting, basket making, food gathering, and ceremonies (Figure 2). Using fire to promote cultural resources in different ecological communities at a larger landscape scale could enhance resource quality, diversity, and access for multiple tribal groups, since many Sierra Nevada tribes have similar or related cultural, language, and basketry traditions (Anderson 2006a, Jordan and Shennan 2003).

Figure 2: Traditional forest products, including sourberries and soup from black oak acorns, harvested for a native foods feast at the California Indian Conference, October 2011 (photo by Jonathan Long).

Effects of Fire on Culturally Valued Plant Resources

Many plant species that are used by Native Americans depend on fire both for persistence and for maintenance of desired growth forms and quality. In the absence of fire, many of these species will decline in abundance and/or mature to a condition where the plant material is not suitable for traditional cultural uses. Examples of these fire-associated plants valued highly by Native Americans and tribes are various shrubs, herbs, and graminoids used for basketry and cordage, including willows (*Salix*), Indian hemp (*Apocynum*), milkweed (*Asclepias*), sourberry (*Rhus trilobata*), sedges (*Carex*), deergrass (*Muhlenbergia rigens*), redbud (*Cercis orbiculata*), dogwood (*Cornus nuttallii*), and beargrass (*Xerophyllum tenax*); nut-producing trees, such as California black oak (*Quercus kelloggii*) and beaked hazelnut (*Corylus cornuta*); berry-producing shrubs and herbs, such as elderberry (*Sambucus*), strawberry (*Fragaria vesca*), and blueberry (*Vaccinium*); edible geophytes, including

snake lily (*Dichelostemma*), mariposa lily (*Calochortus*), and camas (*Camassia*); and plants for medicinal or ceremonial uses, such as wild tobacco (*Nicotiana*), among many others (Anderson 1994, 1999, 2006a). A lack of fire or undesirable applications of fire (including, but not limited to, uncharacteristically severe wildfire) can pose a threat to the sustainable production of these plants in the quantity and quality desired by Native Americans to sustain traditional lifeways and livelihoods. Three species that are culturally important and may have broader ecological significance are discussed below.

Beargrass

Beargrass is an important plant in the understory of conifer forests, where it has declined in abundance in part due to fire exclusion (Charnley and Hummel 2011, Shebitz et al. 2009). Across its range, beargrass provides food and habitat for several animals and pollinating insects, especially flies, beetles, and bees (Charnley and Hummel 2011, Hummel et al. 2012). A fire return interval of less than 20 years may be necessary to limit encroachment and maintain desired reproduction and growth of beargrass, as well as other valued resources associated with relatively open understories (Shebitz et al. 2008). Consequently, efforts to replicate traditional burning may be necessary to maintain these communities in a state similar to their historical condition (Hummel et al. 2012).

California black oak

California black oak is another example of a Sierra Nevada culturally significant species that is dependent on forest fire. Archaeological and ethnographic evidence indicate that black oak was historically one of the most important tribal food resources, and it remains an important species of concern to Native Americans in the Sierra Nevada (Anderson 2007, Haney 1992, Morgan 2008). The trees provide acorns for a variety of wildlife and valuable habitat for fisher and spotted owls (North 2012). Therefore, the tangible and intangible values of these traditional use sites connect past tribal use to contemporary cultural and ecological values, and they reveal the importance of promoting recovery and resilience of black oak in Sierra Nevada mixed-conifer forests. This species has several adaptive traits to survive repeated fires; in the absence of fire, conifers encroach, compete with the oaks for resources, reduce the crown openings needed for robust mast production, and increase fuel loads (Cocking et al. 2012). Mature black oak trees are susceptible to topkill by fire, although they generally resprout from the root collar (Cocking et al. 2012, Stephens and Finney 2002). Treatments focused solely on reducing fire hazard may not result in retention or recruitment of California black oak (Moghaddas et al. 2008). Consequently, management to promote resilience of black oak in the long term while mitigating short-term losses of mature trees is an important challenge when designing treatments to promote socioecological values.

Bearclover

Another plant that demonstrates complex interactions in managing fire for ecological restoration is bearclover, also known as mountain misery (*Chamaebatia foliolosa* Benth.). Bearclover is a low-stature shrub found in large areas of the Sierra Nevada, and it is commonly associated with black oak. Bearclover is a traditional medicine for Native Americans, provides for honey bees and native wildlife, fixes nitrogen, competes strongly with conifer seedlings, and provides highly flammable fuels to carry fires (McDonald et al. 2004). Treatments intended to reduce bearclover have promoted increases in grasses, including, where introduced, highly invasive cheatgrass (McDonald et al. 2004). Because bearclover is well adapted to burning and can be widespread, it is likely to have had an important role in maintaining fire regimes and affecting other forest understory species.

Landscape-scale Effects of Burning Practices

Traditional burning practices served as a smaller scale disturbance that not only maintained desired growth forms of individual plants, but also promoted desired plant communities across larger scales (Anderson 2006b). Though there has been debate about the extent of burning carried out by Native Americans in different regions (Keeley 2002), a comprehensive review suggests that the extent was large across various habitat types (Stephens et al. 2007). Skinner and Chang (1996) note that the frequency of fire necessary to perpetuate specific resources in conditions needed by Native Americans in the Sierra Nevada would have required extensive and intensive burning in important vegetation types. A primary mechanism by which fire contributes to the maintenance of culturally important plant species is by limiting the encroachment of trees and shrubs in meadow and woodland habitats (Anderson and Barbour 2003, Turner et al. 2011). Tribal land management practices served to maintain valued habitats and species diversity across landscapes, from riverine riparian areas to oak and mixed-conifer forests to montane meadows (Anderson 1994). Traditional burning practices occurred at different frequencies and during different seasons, with ignition strategies that varied according to the goals of fire use (Anderson 1999). These practices fostered a mosaic of vegetation types in different stages across landscapes that promoted food security (Charnley et al. 2008, Kimmerer and Lake 2001). Reintroducing traditional burning management practices would help increase heterogeneity in fuel conditions and reinstate finer-grained landscape patterns where burning by Native Americans was important in the past and is of value today (Anderson 1994, Anderson and Barbour 2003, Miller and Urban 2000).

Ecological Issues in Reestablishing Frequent Fire

Plans to restore frequent fire as an ecological process must consider various effects and interactions, especially generation of smoke. Prehistoric Native American burning practices are thought to have been significant contributors of smoke and carbon emissions in the Sierra Nevada (Anderson 1994, Stephens et al. 2007). However, these emissions should not necessarily be seen only as a pollutant, since smoke in appropriate seasons can provide ecological benefits, such as control of insect pests and enhanced germination of various plants, including beargrass (Shebitz and James 2010) and tobacco (Preston and Baldwin 1999). For plant germination, there are potential substitutes for natural smoke that may help compensate for fire deficits (Landis 2000, Shebitz and James 2010). However, smoke may provide other benefits that are not substitutable; for example, smoke from extensive fires has been hypothesized to be important in moderating water temperatures by diffusing direct solar radiation, which could in turn benefit cold water fisheries (Mahlum et al. 2011). Additionally, smoke particles can have physiological effects on plants that could have wider implications for ecosystem function (Calder et al. 2010).

Proposals to reintroduce frequent burning may generate other ecological concerns. Where non-native species are widespread, burning has potential to negatively impact native biodiversity, including culturally valued species (Brooks et al. 2004). For example, a study in Kings Canyon National Park cautioned that frequent burning in areas that have been invaded by cheatgrass might facilitate spread of the invasive grass (Keeley and McGinnis 2007). Further study would help to understand how season of burning influences these effects (Knapp et al. 2008) Ethnographic reports indicate that tribal burning occurred at various seasons, and may have differed from the natural lightning ignition season (Anderson 2006a). In addition, effects of fire frequency are an important subject for research. The Soils chapter (5.0) of this synthesis discusses the potential for frequent prescribed burning to deplete soil nitrogen in certain circumstances. However, at present, many locations targeted for cultural use burns are relatively nutrient-rich, have an abundance of nitrogen-fixing plants, and occur in areas along the western slope of the Sierra Nevada region, where nitrogen deposition rates tend to be elevated (see Air Quality chapter (8.0)). Currently, the areas treated with prescribed fire for tribal cultural purposes are so limited that concerns about nitrogen loss at the local to regional scale appear minimal. For those reasons, burning to emulate traditional burning is not likely to pose a risk of depleting nitrogen or adversely affect forest productivity.

Collaborations to Promote Traditional Burning and Cultural Resources

Productive, collaborative relationship between managers and tribal governments, communities, individuals (where appropriate), and organizations (e.g., California Indian Basketweavers Association) can help to prioritize forest treatments and promote alignment with tribal concerns. Collaboration, consultation, and other forms of engagement with local tribal governments and Native American communities help to incorporate tribal traditional ecological knowledge in research and management and to respect tribal needs and traditions regarding access and caretaking (see Collaboration chapter (9.6)). Several examples of collaborations are shown in the sidebar below.

Sidebar: Examples of National Forest-Tribal Collaborations

- Deergrass has been a subject of collaborative work by the Sierra National Forest and tribes (Anderson 1994, 1999, 2006a).

- As part of the proposed Sage Steppe/Dry-Forest Restoration Project, the Modoc National Forest worked with Cultural Advocates for Native Youth, an organization based in the Cedarville Indian Rancheria, to restore native tobacco plants at burn piles.
- Beargrass restoration has been a subject of collaborative restoration on the Plumas and Lassen National Forests involving Maidu tribes (Charnley et al. 2008), as well as studies on the Olympic National Forest involving the Quinault and Skokomish tribes (Shebitz et al. 2009).
- The Klamath and Six Rivers National Forests entered into an agreement with the Karuk tribe to manage the Katimiin Cultural Management Area, identified in the Klamath National Forest Land and Resource Management Plan, to allow for specific cultural management activities, including reintroduction of fire onto the landscape. The Karuk hold the culmination of their Pikyawish (world renewal) ceremonies in this area near Somes Bar, California. The Karuk, prior to

government fire suppression policies and efforts, used to ceremonially burn the mountain above Katimiin, a historical village site. This cooperative agreement between the Forest Service and the Karuk tribe may serve as a model for other federally recognized tribes in California.

These types of collaborations will assist managers in understanding which habitats, specific plants, or other valued resources can be perpetuated to serve tribal needs (Anderson and Barbour 2003). Each resource of interest (e.g., basketry material or food-producing plants) may have a favored season, frequency, or intensity of burning, and prescriptions may reflect a multitude of objectives for burning (Anderson 1999, 2006a). This information can assist restoration efforts that promote enhancement of cultural resources and address tribal values, and it can promote landscape resilience to climate change and detrimental wildfires. Landscape-scale modeling approaches (see Integrative Approaches chapter (1.1)) could incorporate Native American values and burning strategies. Riparian and meadow restoration activities in particular may provide important opportunities to promote habitat for culturally important species (see chapter on Wet Meadows (6.3)).

Research Gaps

Traditional burning regimes may have been an important factor in maintaining larger vegetation communities, such as open mixed-conifer forests with sugar pine, montane mixed-conifer forests with beargrass understory, montane meadows, and other relatively open riparian types. Evaluating the ecological outcomes of fuels and fire treatments that reinstate or emulate traditional burning practices in different habitats, as suggested in the Sierra Nevada Ecosystem Project report by Anderson and Moratto (1996), remains a potentially valuable avenue for research (Charnley et al. 2007). In addition, the historical and current responses of desired forest resources to fires of different size, season, and severity are an important research gap, especially given the likelihood of larger and higher severity burn patches in the future (see Integrative Approaches chapter (1.1)).

Management Implications

- Integrating cultural values into design of landscape-scale treatment strategies could increase access to tribally valued resources for food, materials, medicine, and ceremonial uses.
- Reintroducing traditional Native American burning practices at appropriate locations within the synthesis area may yield important social and ecological benefits including landscape heterogeneity. Research is needed to understand how factors such as season, frequency, and scale of burns (considering traditional burns, other kinds of prescribed burns, and wildfires) influence fire effects on social and ecological values.
- Collaborations and consultations with tribes and Native American groups can promote opportunities to learn about these fire effects and incorporate them into management practices and applied restoration efforts.

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4.3 Post-wildfire Management

Jonathan Long and Lenya Quinn-Davidson with contributions from Carl Skinner, Susan Charnley, Ken Hubbert, and Marc Meyer

Figure 1: Aerial view of the 2000 Manter Fire, Sequoia National Forest. Photo by Jan Beyers.

Contents

Executive Summary

Wildfires, especially uncharacteristically severe ones, exert major influences on socioecological systems and can result in undesirable outcomes. These events prompt decisions about post-wildfire management interventions, including short-term emergency responses, salvage logging, and actions to influence long-term ecological trajectories, including tree planting and treatment of fuelbeds. These interventions have been scrutinized with respect to their cost effectiveness and undesirable ecological effects. The outcomes of studies highlight the importance of targeting post-fire treatments to particular contexts where the expected benefits exceed the costs of interventions. These conditions are more likely to occur in high-severity patches that are larger than the range of expected variation. Further research is needed to understand effects of wildfires, and high-severity patches in particular, over long periods (and after multiple fires), including effects on fuelbeds, ecological trajectories, wildlife species associated with post-fire conditions and old forests, streams, watersheds, economic values, and social well-being. Increasingly, re-burns, such as the Chips Fire of 2012, may occur and present valuable opportunities to better understand long-term changes in ecological conditions and how to promote socioecological resilience through interventions before and after re-burn events.

Introduction

Wildfires, especially uncharacteristically severe ones, can have undesirable consequences that prompt decisions about management interventions. Because uncharacteristically large patches of high-severity wildfire will continue to occur in the synthesis area in coming decades, these choices may have significant implications for the resilience of socioecological systems. Post-fire situations entail several types of responses, including a short-term response through the Burned Area Emergency Response (BAER) program to protect life, property, water quality, and ecosystems; potential salvage logging of burned trees; and longer term restoration efforts. A range of options and approaches, many of which are outlined below, are available to address these issues. The Forest Service in California has recently developed a post-fire restoration strategy template to help guide national forests in planning for restoration and long-term management of burned landscapes.

Application of a long-term, landscape-scale, and integrated socioecological approach, as highlighted throughout the synthesis, is especially important in post-fire contexts, since wildfire is such a fundamental ecological disturbance in the synthesis area. A trend in a number of scientific publications has been to regard post-fire tree mortality and erosion as important disturbance mechanisms for rejuvenating habitats and promoting resilience to changing climates (Dellasala et al. 2004, Dunham et al. 2003). This perspective has combined with reports indicating that a variety of post-fire treatments have been relatively ineffective (Robichaud et al. 2000), as well as studies reporting unintended consequences of interventions, such as introduction of non-native species or genotypes, increased homogenization, and inhibition of natural succession processes (Dellasala et al. 2004). These issues highlight the importance of targeting post-fire treatments to particular contexts where the expected benefits exceed the costs of interventions.

Short-Term Management Actions and Recommendations

In the 1970s, federal agencies adopted Burned Area Emergency Response (BAER) programs as a coordinated approach to address short-term threats to life, property, and natural and cultural resources. Teams with members representing multiple resource concerns and disciplines collaborate to inventory damage, assess future impacts, identify values at risk from potential flood events and accelerated erosion, and recommend cost-effective mitigation treatments (Robichaud et al. 2000, Wohlgemuth et al. 2009). These responses are commonly limited to the first three years after a fire, which accords with the period during which flood and erosion risks may be particularly elevated. Short-term post-fire response varies depending on the values at risk, fire severity, topography, and other context-specific factors.

Recent reviews have examined techniques to mitigate erosion and other post-fire damage and found that many widely used treatments have demonstrated relatively low effectiveness at reducing erosion and sedimentation, or can induce unintended ecological consequences (Robichaud et al. 2000, Wohlgemuth et al. 2009). Some of the largest areas of short-term post-fire concern are post-wildfire flooding, erosion, and sedimentation, which are interconnected in complex ways, because water and sediment can be routed and stored in different places at different times. Post-fire flooding is a function of excess overland flow from hillslopes, although conditions of downstream channel networks can exacerbate flood hazards through potential failures of debris jams, culverts, and dams. Hillslope erosion can be a significant problem, although it may be greatly exceeded by erosion from channels (Moody and Martin 2001). However, rehabilitation strategies often target hillslopes, in part because of a general principle of treating the problems as close to the source as possible, and because hillslope treatments (see sidebar) appear generally more successful than in-channel treatments. Few post-fire instream treatments have been widely recommended because they are costly to engineer and susceptible to failure. Strawbale and log check dams are the two most common techniques, yet their effects are relatively short-term and failure rates are high (Figure 2) (Wohlgemuth et al. 2009).

Figure 2: Failure of check dams with strawbales and wattles following an extreme rain event. Photo by Jan Beyers.

Sidebar: Treatments for Hillslope Erosion

- **Straw mulch:** Straw mulch is a highly effective means of providing groundcover on hillslopes burned at high severity; it has consistently been shown to reduce post-fire erosion by 50 – 94 percent (Bautista et al. 1996, Robichaud et al. 2000). Straw mulching provides ground cover, protecting the topsoil against rainsplash, slowing surface flows, and helping control hydrophobic soil conditions. It is one of the cheapest and most widely employed post-fire erosion control techniques. However, it may be problematic to apply in areas with windy conditions.
- **Hydromulch:** Most hydromulch mixes consist of a bonded-fiber matrix combined with a non-water-soluble tackifier, which allows the aerially-applied mulch to penetrate into and bond with the soil substrate. Hydromulch can be highly effective, but it is several times more expensive than straw mulch (Hubbert et al. 2012). Treatment effectiveness decreases after the first year as the product breaks down (Robichaud et al. 2010).
- **Contour-felled log erosion barriers (LEBs) and fiber rolls (straw wattles):** Both logs and fiber rolls are used to improve infiltration, slow overland flow velocity by breaking up the slope length, and, to a lesser degree, capture and keep sediment on the slopes. Reviews show that effective use of these treatments requires a skilled workforce for proper placement; poor installation and foot traffic during installation can be a source of hillslope disturbance and rilling if water flows are concentrated (Robichaud et al. 2000). The fiber rolls may be easier to place and entail less ground disturbance than downing and placement of LEBs. Because both of these treatments are expensive and labor intensive, they are used primarily for protection of high-value areas (Robichaud et al. 2000). Both treatments lose effectiveness as the barriers fill with sediment, so they require maintenance. Use of LEBs in particular has decreased in recent years because other treatments are considered more effective (Robichaud et al. 2010).
- **Seeding:** Grass seeding for post-fire hillslope stabilization has decreased as a percentage of burned areas since the 1970s, although more has been spent on reseeding as area burned and use of native species for seeding have increased (Peppin et al. 2011b). Treatment may be unnecessary when

natural revegetation is sufficient or when erosive precipitation events do not occur, and treatment may fail when erosive precipitation events wash out the seeds. Therefore, there may be a relatively narrow window of conditions where seeds would successfully germinate and prevent erosion. As a result, well-controlled research studies generally show that this treatment is minimally effective for reducing erosion, and that there can be unintended detrimental impacts on native plants and annual fire-followers (Beyers 2004, Peppin et al. 2011a, Stella et al. 2010). Use of mulch with seeding increases the establishment of seeded grasses, but generally the combination does not provide greater cover than mulch by itself (Robichaud et al. 2010). The effectiveness of seeding for reducing non-native invasive plants after fire is mixed (Peppin et al. 2011a), and more research is needed.

Debris Flows

Debris flows are an important concern in BAER assessments because of the threat they can pose to life, property, and ecological values for a few years after wildfire. Additional discussion of debris flows is presented in the Watersheds and Stream Ecosystems chapter (6.1). Post-fire erosion associated with debris flows can negatively affect downstream public water supplies (Goode et al. 2012, Rhoades et al. 2011). Although impacts to channels from severe wildfires can rejuvenate aquatic habitats, they can also kill aquatic life and result in extirpation of vulnerable aquatic species that are not able to recolonize the affected streams (Neville et al. 2012). If post-fire landforms persist beyond the wildfire recurrence interval, successive wildfires will have an important cumulative impact on watershed morphology (Moody and Martin 2001). Because climate change is expected to increase the incidence of severe wildfire, high-intensity storms, and rain-on-snow events, the threat of post-wildfire debris flows is expected to increase and become more widespread (Cannon and DeGraff 2009). Some researchers suggest a general lack of good options to prevent post-fire debris flows (Goode et al. 2012). However, one post-fire study reports that a combination of well-designed, well-implemented, and well-maintained hillslope treatments (straw mulch, seeding, and LEBs) and channel treatments (check dams and debris racks) can reduce debris-flow volumes (deWolfe et al. 2008). Further research is needed to evaluate where and when such interventions are likely to be an efficient response. Because roads can concentrate runoff, obstruct stream flow, and alter other hydrologic processes, they are a focus of post-fire treatments. In addition, research from Western Oregon indicated that roads and debris flows can interact to facilitate spread of invasive plants (Watterson and Jones 2006), which is a particular concern after wildfires. A recent synthesis report (Foltz et al. 2008) provides information to guide road rehabilitation decisions after wildfires.

Identifying Erosion Hotspots Using Landscape Analysis Tools

High and rising treatment costs highlight a need to prioritize areas for treatment that are threatened by undesirable levels of erosion, debris flows, invasion of non-native species, and other hazards. The concept of designing forest treatments according to topography featured by North et al. (2009) has parallels in design of post-fire treatments from a water resources perspective. The spatial distribution of post-fire effects can be mapped in relation to landscape features using remote sensing and geographic information system (GIS) tools. In addition to topography, strategic design of mitigation treatments has to include a range of factors, including vegetation, precipitation, soil types, post-fire cover, and burn severity. However, research suggests that particular landscape areas, such as convergent swales, have greater potential for post-fire erosion (Benavides-Solorio and Macdonald 2005). Post-fire monitoring of burned watersheds can enhance understanding of post-fire erosional processes, help to develop models to inform and improve treatment strategies, and inform forest treatment planning by identifying areas that appear particularly vulnerable to post-fire erosion. Tools to predict post-fire debris flows in southern California and the Intermountain Region (Cannon and DeGraff 2009) could be tested and refined so that they could be used for BAER and longer term planning in the synthesis area. Evaluating risks at finer reach-scales can help to evaluate the likelihood of extirpation of aquatic life within basins by relating the probability and location of expected debris flows to occupied habitats.

Post-Fire Salvage Logging

One of the more controversial activities in the post-fire environment is salvage logging of fire-killed trees. Several of the more common reasons given for doing this work are (1) capturing the monetary value of the dead wood while it is still merchantable; (2) generating revenue for post-fire activities, such as site preparation and replanting of severely burned areas; (3) reducing the level of future fire hazard that may result as the dead wood accumulates on the ground as surface fuel; and (4) enhancing the ability of firefighters to safely control future fire, should it occur (Peterson et al. 2009, Ritchie et al. 2013).

Ecological Considerations

Salvage logging is controversial because few short-term positive ecological effects and many potential negative effects have been associated with post-fire logging (Peterson et al. 2009). Knowledge of the ecological effects of post-fire logging, most of which is short-term, has been summarized by McIver and Starr (2000), Lindenmayer et al. (2004), Lindenmayer and Noss (2006), Peterson et al. (2009), and Lindenmayer et al. (2011). These reviews note that general ecological concerns associated with salvage logging include impacts to soils; impacts to understory vegetation and recruitment; potential increases in surface fuel loads; reductions in such as snags and burned logs and their associated habitat values; and other influences on forest development. Impacts to tree recruitment have been observed when salvage logging has been delayed until after seedlings have become established (Donato et al. 2006, Newton et al. 2006). Salvage logging by helicopter is likely to avoid more of the ground disturbance. However, the economic feasibility of salvage logging in general, and especially more costly methods such as helicopter logging, may depend on removing larger, more merchantable dead trees. Such larger trees are likely to be disproportionately valuable for post-fire dependent wildlife (Hutto 2006); however, such relationships remain a subject of active research (including studies within the synthesis area). Information regarding reference snag densities and other habitat qualities from unburned forests may not translate to the habitat requirements of wildlife species that depend on fire (Hutto 2006).

Generally, effects vary considerably by forest type, fire severity, and the nature of post-fire logging (Peterson et al. 2009). Strategic guidance highlights the importance of understanding reference fire regimes and fuel loads, as Franklin and Agee (2003) suggested that salvage may be appropriate as part of an ecological restoration program in dry forests that had uncharacteristically high amounts of standing dead and down trees following stand-replacing fire. Other reviews have suggested retaining untreated patches and limiting the removal of biological legacies, especially larger structures (Lindenmayer et al. 2011). However, these reviews emphasize the lack of experimental research on post-fire logging, particularly regarding long-term effects, that would provide a basis for more specific management guidelines. The consequences of salvage logging following wildfire on carbon cycling and storage are another area of uncertainty (Bradford et al. 2012).

There is only one published experiment that was designed to study the effects of varying the intensity of post-fire logging within the synthesis area (Ritchie et al. 2013). The initial publication from this study focused on snag longevity and build-up of surface fuels over the first eight years following the Cone Fire (2002). The study found that most ponderosa pine (*Pinus ponderosa*) snags had fallen within eight years of the fire (only 16 percent were partially intact in the 30-45 cm class, compared to 41 percent in the >45 cm class). White fir (*Abies concolor*) snags were more durable (42 percent partially intact in the 30-45 cm class and 92 percent in the >45 cm class). Further, regardless of the intensity of post-fire logging, approximately 80 percent of the biomass of retained snags was on the ground eight years after the fire. Correspondingly, eight years after the fire, surface fuels were greater where greater basal area of snags had been retained. Additional studies would be needed within the synthesis area to understand these processes to account for variation in snag and log decay and other ecological factors.

Re-burns

The long-term ecological effects of re-burns, including potential ecological benefits of post-fire salvage of dead trees, are an important gap in our knowledge of post-fire management in the synthesis area. There are many conflicting accounts of fire behavior during re-burns in areas that had previously experienced different levels of fire severity and post-fire logging, but to date, no systematic studies have been done in forests similar to those in the synthesis area (Peterson et al. 2009). Studies in Oregon (Thompson et al. 2007, Thompson and Spies 2010) looked at re-burn severity in southwestern Oregon within the area that burned in the Silver Fire (1987) and burned again in the Biscuit Fire (2002). They found higher levels of crown damage in stands that had been salvage logged and replanted after the Silver Fire than in the areas that had been left unmanaged. They noted that the treatments had promoted relatively dense and homogeneous fuelbeds. They also found that high-severity burns from the Silver Fire led to a condition dominated by shrubs and regenerating trees that burned again at high severity during the Biscuit Fire. It is important to note that the area in that study was a relatively productive environment in which fuels could accumulate rapidly. Within the synthesis area, which is very different in both vegetation and climate, more long-term, site-specific research would help to understand post-fire recovery. Ongoing studies of the Storrie Fire (2000) may provide an opportunity for this type of research, since a large portion of that area was re-burned in the Chips Fire of 2012. Remeasurement of existing plots in the Storrie Fire area would yield more detailed data than were available to Thompson and Spies (2011). Studying the area burned by the Storrie and Chips fires will allow analysis of the effects of the re-burn on a variety of variables, including fire severity patterns, coarse woody debris, snags, regeneration of conifers and hardwoods, and response of understory shrubs and chaparral. This research could fulfill many objectives, including evaluating the extent to which down woody fuel loads remaining from the Storrie Fire may have affected fire severity.

Social Considerations

Despite controversy over the ecological effects of salvage logging, several studies have found a high level of public support for salvage logging in communities that have experienced a nearby wildfire, or are located in an area where the risk of wildfire is high. For instance, McCaffrey (2008) found that roughly two-thirds of sampled homeowners in a Nevada community near Lake Tahoe believed salvage logging was a fully acceptable option for fire hazard reduction. The study also found that respondents who had direct experience with wildfire were 15 percent more likely to find salvage logging acceptable than respondents who lacked experience with wildfire. Age was also positively associated with approval of salvage logging in this study. Individuals who found salvage practices unacceptable cited distrust of the commercial logging industry (McCaffrey 2008). Salvage logging was also viewed favorably by the majority of respondents in three communities in California, Colorado, and New Mexico who had recently suffered extensive wildfires (Ryan and Hamin 2009). Respondents cited concerns over aesthetics and public safety, and the potential for economic benefit, as reasons for their support. However, their support was contingent on salvage logging being environmentally benign. Another study by Ryan and Hamin (2006) found that residents in Los Alamos, New Mexico supported salvage logging after the Cerro Grande fire of 2000, but preferred not to have any new logging roads built. Their support was based on a perceived “wastefulness” of leaving burned trees standing. In this study, participants found salvage logging preferable to the logging of unburned forests.

Salvage logging has several economic benefits. It provides an option to recover economic value from dead and damaged timber, and helps prepare the damaged site for new investments (Prestemon and Holmes 2004). It can provide economic benefit to local timber industries, enabling them to recover some value from an otherwise economically catastrophic event (Prestemon et al. 2006). Additionally, salvage timber sales result in depressed timber prices, which are economically beneficial to the consumer, although they can be detrimental to producers of undamaged timber (Prestemon et al. 2006). Prestemon and Holmes (2004) stress the importance of focusing timber salvage operations on damaged stands that contain higher value materials in order to optimize economic benefits. The economic benefits of salvage logging may be jeopardized, however, if too much time passes in between the fire and the salvage. Dead and damaged timber decays and loses value by the day; delays in the salvage process can result in major economic losses unless timber salvage recovery plans are carried out as designed (Prestemon et al. 2006). There are two common reasons for harvesting delays in salvage situations: administrative delays and appeals and litigation, often resulting from environmental concerns (Prestemon et al. 2006).

In summary, social science research suggests that there is substantial social support for salvage logging in fire-prone communities because it has a number of benefits, such as increased aesthetic beauty, increased safety for recreational forest users, avoidance of waste, reduced fire risk, and economic benefits (McCaffrey 2008, Ryan and Hamin 2006, 2008, 2009). Nevertheless, ecological concerns over salvage logging often lead to contention (McCool et al. 2006, Prestemon et al. 2006, Ryan and Hamin 2009). A concerted research effort would be needed to resolve the many questions that remain concerning the short- and long-term effects of salvage logging.

Managing Long-term Post-wildfire Outcomes

A long-term post-fire management strategy is important for promoting resilience of ecosystems within severely burned landscapes. Post-fire landscapes offer opportunity to realign ecosystem structure, function, and/or composition with expected future climate and fire regimes. In some cases, severe fires may induce a reversion from forests back to shrubfields that were present under an earlier fire regime (Beaty and Taylor 2008, Nagel and Taylor 2005). These post-fire shifts in vegetation may be important for promoting or maintaining chaparral communities on upper and south-facing slopes, where they may have been dominant historically, and for restoring structural heterogeneity to the landscape (Crotteau et al. 2013, Nagel and Taylor 2005). For example, in the mixed-conifer forests of the Lake Tahoe Basin, (Nagel and Taylor 2005) estimated that fire suppression has reduced the average size of chaparral stands by more than 60 percent, greatly reducing biodiversity and habitat values in those areas. Likewise, high-severity fire may be a critical process in the restoration of heavily encroached black oak (*Quercus kelloggii*) stands. In locations where prescribed fire may not be intense enough to kill well-established conifers and widespread mechanical treatments may not be feasible, high and moderate severity fire may induce oaks to resprout, giving them a competitive edge over encroaching conifers (Cocking et al. 2012). However, uncharacteristically severe fire may also induce type conversions that may not have occurred had the forested areas been in a more resilient condition (Skinner and Taylor 2006). For example, Odion et al. (2004) noted persistent shifts from forest to shrublands in areas where multiple high-severity fires burned in the Klamath Mountains. These conditions may be of particular concern in large high-severity patches, where reseeding of conifers may be limited by the distance to live trees (Crotteau et al. 2013). Additionally, increased fire frequencies coupled with nonnative plant invasions may facilitate type conversions of forests and shrublands (e.g., chaparral, sagebrush) to nonnative annual grasslands (Keeley and Brennan 2012, Keeley et al. 2011).

Many parts of the landscape, such as steep terrain and roadless or wilderness areas, may be largely beyond the capacity of managers to treat. Moreover, refraining from active treatment may be appropriate in areas where the ecological trajectory of a post-fire landscape seems to align with desired conditions, which may include regeneration of hardwoods, chaparral or other shrubs (Taylor 2004). In other areas where particular resource values are highly threatened, active restoration may be warranted to guide succession toward desired conditions. Approaches that consider landscape context and the changing climate (Briles et al. 2011, Nydick and Sydoriak 2011, Peterson et al. 2011) could help to identify desired conditions, target treatments and associated monitoring, and identify species and genotypes appropriate for post-fire planting efforts. Guiding principles from other parts of the synthesis can be applied to post-fire restoration strategies.

Landscape Approach and Multi-scale Heterogeneity

When designing a management strategy for large, severely burned areas, it is important to consider neighboring unburned and low- to moderate-severity burned areas within contiguous watersheds and adjacent habitats to provide a geographic context, refine priorities, establish reference locations, and offer increased opportunities to promote landscape resilience. For example, unburned and lower burn severity patches within or adjacent to large, severely burned areas often contain substantial seed source for natural tree regeneration. Furthermore, habitat connectivity could be fostered by designing treatments to maintain desired habitat patches and corridors within or surrounding the fire perimeter.

In addition to facilitating development of landscape-scale heterogeneity in vegetation types, plans may be able to promote smaller-scale heterogeneity by planting trees in spatially variable arrangements rather than on more equal spacing. Another strategy that might be considered would be to select tree stock based on projected resilience to climate change (see chapter on Genetics of Forest Trees (3.0)). These strategies may include planting fire-resistant tree species, planting trees at variable densities (i.e., groups, single stems, and openings) that promote heterogeneity (Larson and Churchill 2012), and managing plantations with the intention of introducing fire as a management tool (Kobziar et al. 2009).

Planning and Collaboration

A key challenge is to develop sufficient capacity for post-fire restoration plans to be implemented quickly before various thresholds (e.g., soil erosion, vegetation development, or use of harvested materials) have already passed (Littell et al. 2012). Increased collaboration and integration of post-fire considerations into management plans in anticipation of potential fires may facilitate implementation of post-fire restoration. Prestemon et al. (2006) suggested that there could be substantial economic benefits from advance planning by reducing the potential for delayed salvage.

Monitoring and Research

Monitoring of post-fire landscapes can help managers determine the likely trajectory of ecosystem recovery in the post-fire environment, prioritize areas for treatment, and evaluate when important thresholds might be crossed. Some fires may warrant investments in more rigorous study designs to answer important research questions, such as impacts to priority species. A specific research gap concerns the habitat needs of species associated with post-fire conditions. Another gap concerns tree the effects of climate change large-scale failures for trees to regenerate. Also, efforts to promote resilience of socioecological systems to future wildfires may be enhanced by including socio-economic considerations within the scope of research and monitoring plans.

Management Implications

- Targeting post-fire treatments to particular contexts where the expected benefits exceed the costs of interventions may help to avoid cost-ineffective practices and unintended negative effects. These conditions are more likely to occur in high-severity patches that are larger than the range of expected variation.
- Further research is needed to understand effects of wildfires, and high-severity patches in particular, over long periods (and after multiple fires), including effects on fuelbeds, ecological trajectories, wildlife species associated with post-fire conditions and old forests, streams, watersheds, economic values, and social well-being. The 2012 Chips Fire, which burned through study areas previously burned in the 2000 Storrie Fire, presents a valuable opportunity to study effects on California spotted owls and many other ecological attributes.

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5.0 Soils

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Contents

Executive Summary

When managing for resilient forests, each soil's inherent capacity to resist and recover from changes in soil function should be evaluated relative to the anticipated extent and duration of soil disturbance**.** **A**pplication of several key principles will help ensure healthy, resilient soils: 1) minimize physical disturbance using guidelines tailored to specific soil types; 2) evaluate changes in nutrient capital and turnover, perhaps using simple balance sheets; and 3) recognize effects on organic matter and soil biota. Due to fire suppression, accumulations of litter and duff in many Sierra Nevada forests that evolved with frequent fires may exceed levels that occurred historically. Repeated prescribed burns may be designed to consume fuels in patches to temper nutrient losses and other undesired effects. Extensive areas of high-severity fire pose risks to long-term soil quality by altering soil bulk density, structure, water-holding capacity, and nutrient content in ways that ultimately contribute to declines in soil resilience. A pending synthesis report published by the Pacific Southwest Research Station (see Appendix A) provides a current review of soil science relevant to forest management.

Introduction

Soil is in many ways the lifeblood of nearly all terrestrial ecosystem functions. Beyond just a growth medium for plants, soils store and mete out water and nutrients, fostering growth of vegetation, animal, and human communities. Soil also degrades toxins, sequesters carbon, and is home to an unimaginable number and diversity of organisms, each of which contributes to soil processes and functions. Enthusiasts richly describe soil as the "porous rind," "living mantle," and even the "ecstatic skin of the earth" (Logan 1995). Soil is easily manipulated, and management actions can simultaneously have a mix of positive and negative impacts on soil functions or plant growth. By and large, many management and disturbance effects on long-term soil sustainability remain unknown (Powers et al. 2005). There is still much to learn about basic nutrient storage pools, appropriate sampling schemes (Harrison et al. 2011), and the chemical importance of rocks within and below the soil (Johnson et al. 2012, Morford et al. 2011). With up to 20,000 km (12,427 mi) of fungal mycelia in a cubic meter (1.3 cubic yards) of soil (Pennisi 2004), it can be difficult to untangle the many complex processes that encourage plant growth and distribute water and solutes through the soil and among roots. However, it is known that if we manage our soil poorly, civilizations themselves may ultimately erode (Montgomery 2012).

Following a perturbation, the functional or structural integrity of a soil may change. The magnitude of change reflects the resistance of the soil, with more resistant soils showing little change in soil function due to a disturbance. The degree and rate of recovery describe the soil's resilience. Together, the concepts of soil resistance and resilience can be useful when considering management impacts on vital soil functions (Seybold et al. 1999). The temporal and spatial scales of resistance and resilience may also influence management decisions. For example, creating a parking lot at a trailhead will cause a significant change in soil hydrologic function due to vegetation loss and compaction, but it may be considered allowable or desirable if it impacts only a very small proportion of a stand or watershed, or reduces the overall impact of a user-created parking area. On the other hand, multiple timber entries may reduce the soil hydrologic function across a broad area if increasingly more land is compacted without allowing the soil's structural integrity to recover between entries. When managing for resilient forests, each soil's inherent capacity to resist and recover from changes in soil function should be evaluated relative to the anticipated extent and duration of soil disturbance.

Sierra Nevada Geology and Soils

The Sierra Nevada, often described solely by its massive granite core, in fact embodies a rich and complex geologic history. In general, the range comprises three rock groups: the famed granitic batholith, older metamorphosed rocks that were invaded by the batholithic magma, and younger volcanic and sedimentary post-batholithic rocks. Sierra Nevada granite formed as igneous magma that intruded below the surface rock and cooled beneath it. Millions of years of erosion and weathering have removed much of the older surface layers, exposing the granite core. In the northern half of the range, the older metasedimentary and metavolcanic features are seen in an elongated band along the western flank. These highly varied features include slate, schist, quartzite, greenstone, serpentine, and many other rock types. The southern Sierra has undergone greater uplift, so much of the older rock surrounding the granite intrusion has been removed, and granitic rocks dominate the terrain. In some places, however, roof pendants of the ancient metamorphic rocks can be seen atop their granite base. In more recent geologic history, volcanic eruptions deposited tuff and andesite flows upon the older granitic and metamorphosed basement. These surfaces predominate east of the Sierra crest as well as in the north, extending to the eastern Cascade mountains and the Modoc Plateau. Uplift, faulting, and repeated glaciations have further sculpted the Sierra Nevada landscape, carving the rock and depositing till and sediments in their wake.

Sierra Nevada soils are highly varied, reflecting the combined influence of climate, topography, biological activity, and parent material over millennia. The resulting soil landscape is a diverse mosaic of varying soil color, depth, texture, water-holding capacity, and productivity. Generally speaking, many soils of the Sierra Nevada are weakly developed and classified as Entisol or Inceptisol soil orders. These often occur at higher elevations and ridge positions, where cold temperature regimes and steep topography limit soil development, but they also occur on resistant parent materials at lower elevations. Developmentally young soils are typically shallow and coarse textured, with little clay development and rapid infiltration rates. Many form on granitic bedrock. On the west slope, mid-elevation soils typically exhibit greater development, support the most productive forests of the range, and include Alfisol and Ultisol soil orders. These soils are deeper, have greater structure and color development and fewer rock fragments, and are enriched with clay. These characteristics enhance the soil's ability to store nutrients and water.

Climate patterns strongly influence soil development and nutrient cycling processes. As elevation and precipitation increase, soil pH and base saturation tend to decrease due to greater leaching and decreased evapotranspiration. Soil carbon tends to increase with elevation. Soil depth, color development, and organic horizon thickness and decay rates reach a maximum at mid elevations, with decreases both above and below (Dahlgren et al. 1997). Microbial activity tends to be greatest when soils are both warm and moist. The Mediterranean climate of the Sierra Nevada produces a prolonged summer drought, limiting decomposition rates due to moisture limitation during the warmest months. Aspect also influences soil development and processes, with more weathering and deeper, richer soils forming on mesic north slopes compared to xeric south-facing slopes. Besides water, nitrogen (N) is typically the most limiting factor to plant growth in forest systems (Vitousek and Horwath 1991). In California's forests, the mineral soil is the primary N reserve, storing 65 – 90 percent of ecosystem N capital (Johnson et al. 2008, 2009). Within forest soil profiles, both N and carbon are concentrated at the surface, and typically decline with depth (Zinke and Stangenberger 2000). Recent research has shown that nutrient hotspots occur at sites on both the west (Johnson et al. 2011) and east (Johnson et al. 2010) slopes of the Sierra, with point-scale increases in available N and other important nutrients.

Though simplified patterns of Sierra Nevada geology and soil properties can help describe the regional setting, a tremendous variety of local conditions exists throughout the range. Parent material can be an important factor in soil conditions. For example, soils formed on serpentine have unique nutrient properties, including a low calcium-to-magnesium ratio and high accumulations of heavy metals, which tend to support sparse and sometimes endemic vegetation. Likewise, bands of ancient metasedimentary slate may underlie highly leached and weathered soil, with low levels of base cations and low productivity, whereas an adjacent soil on a more recent andesitic flow may support a robust stand with rich nutrient reserves. Parent material has also long been used as an index of soil erodibility. In the Sierra Nevada, soils formed from decomposed granite tend to be highly erodible (André and Anderson 1961), whereas metasedimentary soils are more stable. Local knowledge of geology and soil conditions is essential in understanding the potential of, and management concerns in, a particular landscape.

Priorities in Soil Management

Regardless of overall land management strategies, application of several key principles will help ensure healthy, resilient soils. Though the list below is not by any means exhaustive, the following considerations are straightforward, easy to grasp, and easy to apply in practice.

Prevent Soil Loss

Maintaining soil in place is often the highest soil management priority. Soil erosion is a natural process—over the ages, mountains erode, alluvial valleys form, and lakes fill in. However, in human timescales, soil erosion is considered acceptable when it is in equilibrium with rates of soil formation. Soil formation rates vary by location, but have been estimated around 2 – 4 Mg per hectare per year (1-2 t/ac/yr) for forest soils. When spread uniformly across an area, this represents an annual gain of a few tenths of a millimeter (0.01 in). Visually, sheet erosion at this rate may be imperceptible, though modeling programs, such as the Water Erosion Prediction Project, are frequently used to estimate losses through model simulations. Accelerated erosion, which exceeds the background rate of soil formation due to management activities, is typically considered unacceptable. Due to the timescale at which soils form, prolonged soil erosion is perceived as effectively irreversible. When soil is lost, so is the rooting medium in which plants grow, as well as nutrients, carbon, organic matter, and the ability to hold water (Page-Dumroese et al. 2010). These properties are generally concentrated at the soil surface, so surface erosion can have greater impacts than soil loss from lower horizons (Elliott et al. 1999). These losses can permanently impact site quality where soil is removed, yet productivity may be enhanced where it is deposited. Excess sedimentation into lakes and streams, however, can reduce water quality and aquatic habitat. Maintaining soil cover is the easiest way to prevent accelerated erosion. Using model simulations, Page-Dumroese et al. (2000) found that in many cases, 50 percent ground cover could prevent accelerated erosion rates. Citing several other studies, Robichaud et al. (2010) suggested that levels of exposed bare soil less than 30 – 40 percent following forest thinning can generally keep soil erosion rates “acceptably low.” Maintaining soil on site is essential to continued ecological function, and time frames for recovery of lost soil and the functions it provides are far greater than human lifetimes.

Minimize Physical Disturbance

Forest management practices, especially those using mechanized equipment, may disturb the soil. Many soils are easily compacted by heavy machines, which also displace organic and mineral horizons during turning maneuvers. Forest floor displacement, especially when combined with compaction, leaves soil vulnerable to erosion. Mineral soil displacement can impact soil quality by removing surface material, which is generally richer in nutrients, organic matter, and habitat than underlying subsoil. In cases where a residual canopy exists, litter accumulation and recovery of lost or displaced soil cover can be achieved in a matter of years. Compaction effects on forest soils have been studied for over 60 years (Munns 1947, Steinbrenner and Gessel 1955), and they remain an important management concern today. Physical soil changes due to compaction have been enumerated by many (see Page-Dumroese et al. 2006), and can include decreases in soil porosity, rooting volume, and aeration, and increases in soil bulk density, strength, water content, runoff, and erosion. Compaction impacts are site-specific, with varied effects on forest stand productivity (Gomez et al. 2002a). One of the easiest ways to prevent compaction is to operate machines when soils are at their driest. This is especially true of soils with high clay content, which develop high soil strength—and compaction resistance—as they dry. Operationally, treatment operations can be timed to delay more vulnerable sites to later in the summer period to allow for greater soil drying. Once compacted, recovery of bulk density or soil strength can take many decades. Recovery rates are influenced by management history, including the number of harvest events in a stand and soil moisture conditions during the harvest, as well as soil and site attributes, such as soil texture, rock fragment content, and freeze-thaw cycles (Page-Dumroese et al. 2006). Equipment and operating conditions can be specified to limit soil compaction. These well-known guidelines are typically tailored for soil texture, rock content, and organic matter, and they include using low-ground pressure equipment and operating when soils are dry, frozen, or under substantial snow (for example, Table 1). Compaction can be mitigated by techniques such as subsoiling, which typically uses a winged implement to lift and shatter the compacted layer without inverting the soil. When properly applied, subsoiling can increase soil infiltration, allow deeper root elongation, and foster increased plant growth. These practices are not without their own risks however, and may cause rilling and erosion when improperly applied on moderate to steep slopes. Preventing or limiting compaction is quite feasible, and in most cases is preferable to relying on post-compaction mitigation practices.

Table 1. Compaction risk ratings based on texture class and coarse fragments. Adapted from Forest Service Region 5 Detrimental Compaction Risk Rating Guide (USDA Forest Service 2006).

Compaction hazard	Texture class	Coarse fragments > 2mm
Low	sandy	any amount

	any texture	greater than 70 percent
Moderate	loamy texture	any amount
High	clayey	any amount
	silty	less than 35 percent

Evaluate Changes in Nutrient Capital and Turnover

Forest management can directly and indirectly change nutrient stores at a site. Vegetation harvest removes nutrients in wood and/or crowns, immediately affecting local nutrient pools (Powers 2006). Reductions in canopy cover and altered microclimate can also change the rates at which organic matter decomposes and nutrients cycle from organic to inorganic forms. Fire short-circuits this decomposition pathway, rapidly cycling nutrients tied up in organic matter (Knoepp et al. 2005). Heat from prescribed fire operations volatilizes nutrients, including N, most of which is typically lost as gas during forest floor combustion. Some N may move downward into the soil in forms chemically available to plants and microbes. In order to evaluate the nutrient impacts of different treatment strategies, forest managers may find it useful to assess the scale of nutrient removal relative to existing pools, and the local mechanisms and rates of nutrient replenishment. Understanding sources and rates of nutrient inputs and outputs allows estimations for future condition and potential recovery, and these are discussed in more detail in the following sections.

Recognize Effects on Organic Matter and Soil Biota

Organic matter is considered a cornerstone of soil quality, enabling soil to perform important biological, chemical, and physical functions: as a habitat and nutrient source, organic matter supports soil biota; it also has an extremely high capacity to retain and exchange water and nutrients, and it contributes to soil structure, aggregation, and stability. Soil biotas are essential to many basic soil processes, including formation of soil structure and porosity, organic matter decomposition, atmospheric nitrogen fixation, and enhanced nutrient uptake by plants. From single-celled bacteria to complex arthropods and vertebrates, soil organisms are the lifeline between plants and mineral soil. Soil inhabitants tend to concentrate near their food sources at the soil surface, where organic matter and roots are most abundant. Soil biological indicators can be used to detect environmental changes, but their use in land management is limited by taxonomic challenges and their inordinate numbers, and by inefficient analytical protocols and a lack of understanding about them (Andersen et al. 2002). Our understanding of soil biodiversity is in its infancy. For example, less than 10 percent of soil microarthropod populations have been explored (André et al. 2002), and more than 1,000 species of new fungi are described each year (Hawksworth 2001), though not all exclusively occupy soil habitat. Symbiotic associations between plants and fungi, known as mycorrhiza, are well known, but how these intricate mycelial pipelines operate to transmit water and influence plant establishment remains under investigation (Plamboeck et al. 2007). Soil organisms are generally outside the scope of forest management. However, managing for organic matter is a complementary strategy to ensure biologically healthy soil.

Management Effects on Soils

Mechanical Forest Restoration and Fuel Treatments

Thinning to reduce hazardous fuels and/or improve forest health in dense Sierra Nevada stands often involves the removal of hundreds of stems per hectare. Though these are typically small-diameter materials, the intensive mechanical operations used to harvest and treat them have raised questions about long-term soil impacts, including compaction, erosion, and nutrient removal. These concerns are not new to forest management, but novel treatments in managed landscapes require careful evaluation of new and cumulative impacts on soil quality.

Physical soil disturbance

Mechanical thinning treatments in the Sierra Nevada typically use conventional harvest techniques, including heavy equipment operating as harvesters (feller-bunchers and cut-to-length harvesters), skidders, or forwarders. In contrast to traditional commercial stand thinnings that remove fewer, larger trees, forest restoration treatments often aim to remove or process large quantities of small stems. Operationally, this may require that equipment traverse a large proportion of the treatment area in order to access and remove material. Though this equipment footprint can increase the risk of soil compaction from vehicle traffic and soil displacement due to vehicle turning, careful operations and application of best management practices can avoid excessive disturbance.

Mechanical fuels treatments and restoration thinning can be conducted with minimal exposure of bare mineral soil. In several southern Sierra studies, bare soil exposure 1 – 3 years after treatment did not differ between treated stands and controls, regardless of whether slash material was left on site (Wayman and North 2007) or piled and burned (Berg and Azuma 2010). Though bare soil can contribute to surface erosion, Berg and Azuma (2010) found no evidence of rilling following forest thinning on predominantly granitic soils. In an erosion study focused on the northern and central Sierra, Litschert and MacDonald (2009) studied mechanical harvest units thought to have erosion or sedimentation problems. They evaluated approximately 200 of these units on a range of parent materials and found evidence of soil erosion (i.e., rills or sediment plumes at the lower unit boundary) in only 19 instances. In all thinning units, the erosion was traced to a skid trail rather than the harvest area in general.

Steep slopes are typically more vulnerable to runoff and erosion, but they are increasingly identified as high priority areas for thinning treatments. To prevent excessive soil disturbance, most public lands in the Sierra Nevada exclude mechanical operations from slopes with gradients above 30 – 35 percent. No studies have examined ground-based operations on steep slopes of the Sierra Nevada, but Cram et al. (2007) studied disturbance and erosion on intermediate (10 – 25 percent) and steep (26 – 43 percent) slopes in a thinned New Mexico mixed-conifer forest. Though operations on steep slopes generally cause more soil disturbance, they found that maintaining soil cover and minimizing large areas of bare soil can be sufficient to prevent increased erosion and sedimentation levels. Novel equipment has been developed to operate on steep and sensitive areas. For example, John Deere developed a prototype of a "walking harvester," which has six legs rather than tires or tracks, can navigate steep and uneven terrain and turn easily, and purportedly leaves minimal physical impacts due to its small footprint. Two prototypes are on display at forestry museums, but the machine never made it into production (John

Deere 2012). Beginning in 2013, the US Forest Service Technology and Development Program will evaluate soil impacts by this type of “walking” equipment on steep slopes, based on a field trial on the Lassen National Forest.

In addition to bare soil exposure, skid trails and associated compaction are likely the greatest physical impact of mechanical forest restoration and fuel reduction operations. The spatial extent and arrangement of skid trails depends on the material removed as well as slope, terrain, and proximity to temporary or permanent roads. Much of the compaction on skid trails occurs during the first few machine passes (Williamson and Nielson 2000). Cut-to-length harvest systems, in which only tree boles are taken off site, have been shown to reduce the amount of compaction when boles are forwarded over slash-covered trails. Compared to whole-tree harvests that were yarded with skidders, cut-to-length systems produced a smaller compacted footprint with a lower degree of bulk density change (Han et al. 2009). Though heavy slash mats help buffer the impacts of machine traffic, they break down after multiple equipment passes and their effectiveness at minimizing compaction decreases (Han et al. 2006). Many Sierra Nevada stands have legacy skid trails from previous harvest entries, and re-use of existing trails could limit cumulative compaction impacts. This can be problematic, however, when previous skid trails or landings are located in drainages or sensitive areas, do not access the necessary part of the unit, or are poorly suited for contemporary harvest methods. Roads are often a large contributor to cumulative watershed effects due to compaction, erosion, and sedimentation to streams. Erosion from roads is discussed in the Water Resources and Aquatic Ecosystems section of this synthesis.

Mastication treatments in particular have raised concerns about soil compaction, since masticators may need to operate well away from skid trails to reach standing trees, shrubs, or slash. Like slash mats, masticated material may help buffer the compacting forces of heavy equipment (Moghaddas and Stephens 2008), but soil moisture remains a key factor in susceptibility to compaction. Recent findings in the Sierra have shown that compaction effects on tree growth are complicated, and can vary with soil texture. Growth in pine plantations less than 10 years old was negatively impacted in compacted clay soils, responded neutrally to compacted loam, and increased in compacted sandy loam soil (Gomez et al. 2002a, 2002b). Compaction compresses large pores, so coarse-textured soils may have more capillary ability when compacted, holding more water and enhancing tree growth. Soil compaction is a reversible process, but recovery can take many years. Soil recovery is greatly enhanced by freeze-thaw processes, but soil texture may also play a key role in recovery rates. In plantations grown in compacted soils, bulk density recovery after five years was slower in fine-textured soils than coarse-textured ones (Page-Dumroese et al. 2006).

Sidebar: Compaction

Soils are easily compacted, even with just a few machine passes. Recovery following compaction is a slow process, often requiring decades. Mitigation techniques, such as subsoiling, are not without their own risks and effects, and should be considered far less desirable options than preventing compaction in the first place. Operationally, compaction can be minimized by:

- Operating when soils are dry; moist soils are more susceptible to compaction and will compact to a greater depth than dry soils
- Operating when soils are frozen or under deep snow
- Using equipment with minimal ground pressure
- Limiting equipment to designated trails, and reusing trails where feasible
- Using boom-mounted equipment, which requires less ground travel
- Travelling over deep slash layers where feasible

Results from the Long Term Soil Productivity Experiment show that compaction can increase soil water availability in sandy soils, leading to improvements in vegetation growth. However, growth may be inhibited in compacted clay soils (Gomez et al. 2002a, 2002b).

Effects on soil nutrients

Forest restoration thinning or fuel treatments are commonly achieved through whole-tree harvest techniques. For more than 40 years, this practice has raised concerns about nutrient loss and long-term site productivity because branches and foliage are removed along with the tree stems (Kimmins 1977; Tamm 1969). However, most research has evaluated whole-tree clearcut harvests, and surprisingly few studies have addressed impacts of whole-tree thinning. Thinning typically removes far less biomass than clearcut prescriptions. For example, fuels reduction thinning projects in dense Sierra Nevada stands removed an average of 12 percent of the standing live volume (Collins et al. 2007), which was equivalent to 21 percent of the basal area in those areas (Stephens and Moghaddas 2005). Fuels reduction treatments in dense stands typically reduce basal area by 20 – 45 percent (Boerner et al. 2008b), while retaining a majority of the standing volume on site. Fuels are often thinned from below, so understory, suppressed, and intermediate trees are removed before codominant or dominant trees. These lower crown positions have proportionately less canopy biomass (Reinhardt et al. 2006) and, therefore, fewer canopy nutrients, than the dominant overstory. Using clearcut-based studies to infer nutrient loss impacts following fuel reductions could grossly overestimate effects on soil nutrient pools and stand productivity. In any case, studies of stands greater than 15 years old suggest that whole-tree clearcut impacts to soil C and N stocks diminish with time (Jandl et al. 2007, Johnson et al. 2002, Jones et al. 2008, Walmsley et al. 2009), and nutrient recovery in thinned stands would likely be much quicker.

Few whole-tree thinning studies have been conducted in U.S. forests, and studies on fuels reduction thinning in the Sierra Nevada are even fewer. Johnson et al. (2008) compared whole-tree and cut-to-length thinning in a Sierran east side pine forest. Whole-tree harvest methods removed approximately three times more N than cut-to-length methods. Because limbs and tops were left on site, the cut-to-length system left two to three times more C and N content in the forest floor than the whole-tree harvest. However, neither harvest system removed more than a few percentage points of the ecosystem N capital of the sites (Johnson et al. 2008). In a fuel reduction study in dry forests of central Oregon, Busse et al. (2009) compared the effects of whole-tree harvest, bole-only removal, and thinning without biomass removal on vegetation responses. Through periodic measurements in the 17 years following the treatments, they found no differences in tree growth, shrub cover, or herbaceous biomass among treatments. Busse and Riegel (2005) estimate that this whole-tree thinning removed 4 percent of ecosystem N, whereas bole-only harvest removed 1 percent. Compared to the other residue treatments, the whole-tree harvest did not reduce the site potential or soil nutrient status of the relatively infertile sites included in that study (Busse and Riegel 2005). It's likely that many Sierra Nevada forests are similarly resistant to N reductions, but the potential losses must be considered in the context of the total amount of N on site and sources of N inputs over time. Fertile sites with deep, rich soils tend to be more resilient to whole-tree harvests than poor sites, such as shallow soils over bedrock or coarse-textured soils (Raulund-Rasmussen et al. 2008). More nutrients are generally exported during whole-tree harvests from fertile sites than from nutrient-poor sites, but higher levels of nutrient inputs and cycling rates often allow for rapid replacement of the lost nutrients.

Balance sheets are useful to compare nutrient inputs, outputs, and on-site reserves (Smith 1986). Forest managers can estimate the amount of nutrients removed during whole-tree fuels reduction by first taking stock of what's held in tree crowns and boles and then determining how much material will be removed during treatment. For example, canopy fuels in a dense, mid-elevation west slope Sierra Nevada stand with high fire hazard conditions were estimated at 17 Mg/ha (8 t/ac) (Reinhardt et al. 2006). Estimates from other similar studies in the western U.S. found crown fuel loads between 9 and 21 Mg/ha (4-9 t/ac) (Cruz et al. 2003, Fulé et al. 2001). Assuming that half the crown volume of a stand is removed and that its foliar N content is 1.4 percent (Carter 1992, Garrison et al. 2000), approximately 65 – 150 kg N/ha (0.029 – 0.067 t N/ac) might be removed as crown material during fuels treatments. Assuming an equal portion of N is stored in the wood, 65 – 150 kg N/ha (0.029 – 0.067 t N/ac) may also be removed as bole material. For many Sierra Nevada sites, 130 – 300 kg N/ha (0.058 – 0.130 t N/ac) represents only a small percentage of the N stored in the soil (Zinke and Stangenberger 2000) and an even smaller proportion of total ecosystem N on site. Furthermore, the actual amount of N removed will vary by stand and thinning treatment. At their east side Jeffrey pine site, Johnson et al. (2008) reported N removal of 50 and 162 kg N/ha (0.022 and 0.072 t N/ac) in areas treated with cut-to-length and whole-tree thinning, respectively. It is important to remember that N is chronically added to forest stands by deposition and lost by leaching, and these processes vary by geographic region. By estimating local deposition and leaching rates, one can consider harvest N removals in the context of how quickly they will be replenished (Table 2). Though useful as a simple tool to gain perspective on long-term nutrient changes, this nutrient budget approach can't account for complex and unpredictable changes in nutrient cycling or rates of availability (Powers et al. 1990).

For particularly sensitive sites, there are a number of options to minimize or compensate for nutrient losses due to whole-tree harvesting. More nutrients can be kept on site by harvesting in the fall or winter. Wood becomes more brittle during this time and is more likely to break during thinning activities. Leaving broken branches or tops in the stand will reduce the nutrients exported off site. Harvesting trees during their dormant period may also reduce nutrient exports, as leaves translocate their nutrients to the roots and other components at senescence (Nambiar and Fife 1991). Thinning deciduous trees after leaf drop can also reduce foliar export out of the stand. Marking guidelines that account for species nutrient requirements can also help offset nutrient removals; thinning trees with high nutritional needs while retaining more frugal species may export a relatively larger amount of nutrients, but the overall nutrient demand in the residual stand will be reduced. Where species exhibit great differences in nutrient needs, these nutrient requirements could become a consideration in marking guidelines for fuels reduction, in addition to crown spacing, shade tolerance, and fire behavior characteristics. Five years after whole-tree thinning to reduce fuels and restore stand structure, units where large trees were preferentially retained, regardless of species, had greater levels of soil N, and thus greater short-term potential for increased growth and productivity, than units where pine species were preferentially retained. However, the pine-retention stands had higher levels of forest floor N, suggesting greater potential for nutrient availability in the future (Miesel et al. 2008).

Table 2. Example soil nutrient capital and nutrient balance accounts for a site thinned with a whole-tree harvest approach. Nitrogen removal from whole-tree harvest is equal to less than 3 percent of the total soil pool of N. Assuming deposition and leaching rates remain constant, N is added at a rate of 5 kg ha^{-1} yr^{-1}. At that rate, the N removed from the thinning treatment would be replenished after approximately 40 years, or sooner, if the abundance of N-fixing vegetation increased after treatment.

Credit/debit	N kg ha^{-1}	Explanation
Soil pool	9800	The soil acts as a large reservoir to store N. Most soil N occurs in organic form, which is not readily available for plant uptake and cannot be easily leached. Plant-available inorganic N is slowly released through decomposition and mineralization pathways.
Harvest export	-200	Whole-tree thinning removes some of the N capital from the site. The time required to replenish lost N depends on rates of inputs and outputs. Harvesting alters soil microclimate, which can increase or decrease the amount of N available for plant uptake by altering rates of decomposition and N mineralization.
Annual Deposition	6	Nitrogen is continually added to terrestrial systems as both dry and wet deposition.
Annual Leaching	-1	Nitrogen leaching losses typically occur as plant-available nitrate.

In many cases, whole trees are skidded to a landing where processors, such as delimbers, remove the non-merchantable material from the bole. Rather than chipping or burning the slash, skidders could backhaul some or all of it into the unit to redistribute the nutrients on site. This would allow efficient harvesting equipment (e.g., feller-bunchers) to fell trees while reducing nutrient losses. Rich (2001) addressed some operational constraints and practices to help make backhauling of slash a feasible option. However, fuel loading should be a consideration in plans that include redistribution of slash.

Prescribed Fire Treatments

Prescribed fire operations designed to reduce ground and surface fuel loads are common in the Sierra Nevada, and many basic prescribed fire effects on soils have been well described in review or meta-analysis publications that use references from around the globe (for example, see Carter and Foster 2004, Certini 2005, Johnson and Curtis 2001, Nave et al. 2011, Neary et al. 1999, Neary et al. 2005, Raison 1979, Wan et al. 2001). Like mechanical vegetation removal, fire reduces the nutrient capital on site, but through a very different mechanism. Whereas harvests physically remove nutrients contained in biomass, fires volatilize and transform nutrients through heating and combustion. Similar effects are generally found regardless of whether or not sites are thinned prior to burning, but the magnitude of those effects can vary. Management practices that alter the forest floor will similarly alter fire behavior and effects. Conditions that lead to greater fuel consumption have the potential to increase impacts on soils. Slashmats left in yarding trails during cut-to-length harvests create continuous fuelbeds that burn more than adjacent areas, whereas skid trails tend to disrupt fuel continuity and burn less than adjacent areas.

Recent research has focused on comparing the individual and combined effects of thinning and burning treatments on soil. Boerner et al. (2008a, 2008b, 2008c, 2009) have conducted soil meta-analyses for 12 North American forest sites in which the same study design was used to examine forest thinning, burning, and combination treatments. A number of similar forest restoration or fuels reduction studies using prescribed fire have been implemented across and near the Sierra Nevada, and these form the basis for this section. They include treatments in mixed-conifer stands in the Goosenest Adaptive Management Area in the southern Cascades, Blodgett Forest Research Station in the central Sierra, Teakettle Experimental Forest and Sequoia National Park in the southern Sierra, and a Jeffrey pine forest east of the Sierra crest.

One of the most significant changes to the soil system caused by fire is the loss of mass and nutrients from the forest floor. Fuel consumption typically varies across a burned area due to microsite differences in fuel moisture, loading, and continuity. In unharvested stands, Knapp and Keeley (2006) found that 70 percent of the treatment area burned during early season prescribed fires when fuels and soil were moist, whereas 88 percent burned during late season fires when conditions were substantially drier. Pre-fire harvests can both increase and decrease fuel continuity and burn extent. Cut-to-length harvesting, in which activity slash is placed in the yarding trail, can result in higher levels of fuel consumption within slash mat features; in one study, fire covered 60 percent of the area outside slashmat features, while 70 percent of the area burned within the slashmat trails (Murphy et al. 2006a). In contrast, skid trails typically expose large amounts of bare soil, reducing the percentage of area burned. Following whole-tree harvest, Murphy et al. (2006a) found that fire covered 77 percent of areas outside skid trails, but only 30 percent of the area within them. Similarly, Moghaddas and Stephens (2007) found that, in thinning units where both logging slash and masticated debris were left on site, 95 percent of areas outside skid trails burned, but only 48 percent of the area within skid trails burned. Where skidders were used, the combination of thinning and burning exposed more bare soil than either one alone (Moghaddas and Stephens 2007, Wayman and North 2007).

Figure 1: Before and after photos of a Fire and Fire Surrogate Study site at UC's Blodgett Forest that was thinned and burned. Before photo was taken in 2001, and after photo was taken in 2003. Photo by Emily Moghaddas.

When litter and duff layers burn, most of the N they contain is lost to the atmosphere in gaseous form. The amount of N lost during fires is positively correlated to the amount of material burned, which can vary tremendously across the Sierra. For example, prescribed fire in a highly productive mixed-conifer forest reduced the forest floor mass by 87 percent, causing N losses of 725 – 750 kg/ha (0.32 – 0.33 t/ac) (Moghaddas and Stephens 2007), which represents well under 10 percent of the ecosystem N on site (Boerner et al. 2008c). But burning in east side Jeffrey pine forests reduced the forest floor mass by 60 percent – 75 percent, with concomitant N losses of only 100 – 250 kg/ha (0.04 – 0.11 t/ac) (Murphy et al. 2006a), equivalent to less than 5 percent of ecosystem N there (Johnson et al. 2008). At each locale, the greater losses occurred in stands where logging slash was present. The forest floor was reduced by more than half at both sites, causing a substantial relative reduction in surface N capital. Carbon losses from the forest floor follow similar patterns to N, with proportionately more C lost as more forest floor is combusted.

Despite huge changes in the forest floor, total C and N pools in mineral soil often remain unchanged following prescribed fire, though both decreases and increases have been observed. Soil total C and N were unchanged following burning in both examples described above (Johnson et al. 2008; Moghaddas and Stephens 2007). Similarly, burning and thin-burn treatments in a southern Sierra site did not alter soil C pools (North et al. 2009) or C and N concentrations relative to control plots (Wayman and North 2007). Soil pools may remain largely unaffected, because so much nutrient capital exists in the soil that fire-induced changes are relatively small by comparison (Wan et al. 2001), and because prescribed burns do not often reach high enough temperatures to combust soil organic matter beyond shallow surface layers (Johnson et al. 2009). At another southern Sierra mixed-conifer site, 74 percent of the forest floor mass was consumed during early season burns (Knapp et al. 2005), with no change in soil total C or N pools (Hamman et al. 2008). In contrast, late season burns at the same site consumed 94 percent of the forest floor (Knapp et al. 2005), reducing soil C levels for at least 3 years, but causing no change in total N. The drier fuel and soil conditions during late season burns can contribute to greater burn severity and a more prolonged soil effect (Hamman et al. 2008). In the southern Cascades, Miesel et al. (2007) reported reduced concentrations of soil C in unthinned, burned stands, but no change in areas that had been whole-tree harvested prior to burning. The decrease was attributed to greater soil heating and organic matter combustion in the unthinned areas. In some cases, soil C can increase following prescribed fire, due to incorporation of charcoal (Johnson and Curtis 2001), but no examples of this have been documented in the Sierra Nevada.

Sidebar: Putting Nutrient Removals in Perspective

Due to fire suppression, accumulations of litter and duff in many Sierra Nevada forests that evolved with frequent fires may exceed levels that occurred historically and may now represent novel conditions. As a result, proportionately higher pools of nutrients may exist aboveground than in the past. Both forest thinning (harvest) and prescribed fire cause nutrient losses, though through very different mechanisms. Whereas nutrients are directly exported in boles (and tops and limbs, in the case of whole-tree harvest) during thinning, fires remove carbon and nitrogen largely through combustion and volatilization. Cumulatively, harvest and burning treatments will remove more nutrients than either treatment alone. It is a good idea to assess the scale of nutrient removal relative to existing pools, in order to evaluate potential risks to forest productivity and consider soil resilience to nutrient losses. Soils are widely variable in space, and knowing if a specific soil has vast reserves of N and other nutrients or is shallow and nutrient-poor can be critical in assessing the consequences of nutrient removal. This is especially true if repeat or cumulative treatments are being considered. Management goals may not seek to replenish these nutrients if they are removed through harvest or fire, but understanding the magnitude of change and rate of recovery can provide important ecological perspective. Balance sheets can provide ballpark estimates of nutrient inputs and outputs and give managers a sense of the scale of impact different treatment alternatives may carry (see Table 2). After treatments, periodic site visits should be used to assess the cover and type of nitrogen-fixing shrubs; these visits can further refine estimates of N replenishment over time.

A short-term increase in soil pH is often, though not universally, observed following prescribed fire due to deposition of ash rich in base cations (Raison 1979). Studies in the Sierra Nevada that measured pH show that thinning and burning treatments elicit the same soil response as burn-only treatments, whether that response is an increase in pH (Moghaddas and Stephens 2007, Ryu et al. 2009) or no change in pH (Murphy et al. 2006a). Fire also tends to increase inorganic N levels in the soil (Miesel et al. 2007, Moghaddas and Stephens 2007). In both the southern Cascades and the central Sierra, thinning before burning resulted in greater inorganic N increases. At the latter location, the increase was due primarily to large increases in ammonium, which can result from oxidation of organic matter during burning and increased N mineralization. Presumably, the organic matter and nitrogen source was the combusted forest floor, as volatilized N can condense and move down into the mineral soil. The increase in inorganic N, however, represented only 5 percent of the N lost from the forest floor (Moghaddas and Stephens 2007). Nitrogen-fixing plants, which occur at both of these study sites, can also contribute inorganic N to the soil. Despite increases in mineral N, no changes in mineralization rates were detected following burning in the southern Cascades (Miesel et al. 2007) or in central Sierra (Moghaddas and Stephens 2007) or southern Sierra (Hamman et al. 2008) sites, and changes in nitrification rates were variable. Although N turnover rates often increase following fire, microbial activity is strongly linked to substrate availability, organic matter quality, and microclimate—factors that will vary under different site and burn conditions. When changes do occur following prescribed fire, they tend to diminish within several years (Wan et al. 2001).

Woody fuels are piled and burned in some areas due to infeasibility or restrictions against underburning or mechanical operations. Burn piles concentrate soil heating effects into relatively small, confined areas beneath the pile footprint. Soil heating during pile burning can be extreme. In southern Colorado, Massman and Frank (2004) measured soil temperatures of 400°C (752°F) beneath a large slash pile, and temperatures remained elevated for several days. Significant changes in the physical, chemical, and biological properties of soil are likely under these conditions, but not all pile burns result in extreme soil temperatures or soil damage (Busse et al., in press). The severity of an individual burn plays a large role in subsequent soil impacts, which may include changes in organic C and N, available nutrients, water repellency, microbial activity, soil texture, mineralogy, bulk density, and porosity. Pile burning is also responsible for the so-called "ash-bed effect," in which the release of nutrients (particularly N, calcium, magnesium, potassium) from organic materials can temporarily augment soil fertility. In the Sierra

Nevada, York et al. (2009) found that 10-year height and diameter growth of conifer seedlings was up to 50 percent greater within pile burn perimeters than on adjacent, unburned ground.

Fuels managers face decisions about the appropriate size and density of piles for given treatment units. Although larger piles will generate more heat overall, they do not necessarily increase the degree of soil heating. For example, Busse et al. (in press) found no significant relationship between pile size and maximum soil temperature or heat duration for piles ranging from 1.8 – 6.1 meters (6 – 20 ft) in diameter in the Lake Tahoe basin. Rather, fuel composition was the key factor in soil heating at these sites. Piles containing high levels of large-diameter bole wood reached greater soil temperatures for longer durations than piles containing smaller diameter materials and limbs. Under all piles, the most extreme heating was limited to the surface 5 – 10 cm (2.0 – 3.9 in) of mineral soil below the pile. Extreme soil heating may be of little concern if it occurs beneath widely spaced piles that occupy little of the total land surface. The greater the density or total ground coverage of piles, the greater the potential to impact soil quality due to extreme soil heating. Across 71 sites in the Lake Tahoe basin, ground coverage by piles averaged 8 percent, but reached as high as 35 percent where larger diameter insect-killed trees were bucked and piled (Busse et al., in press). These findings suggest that, in most cases, decisions regarding the optimal size and number of piles per treatment unit can be based on operational factors, including cost effectiveness, fire risk, and operator safety, rather than potential soil effects.

Soil resilience and repeat burning—fire as a restoration tool

Most terrestrial areas in the Sierra Nevada evolved with some periodic influence of fire, including the changes in nutrients and soil processes that fire causes. The reintroduction of fire is often recommended as a restoration and maintenance tool. There are very few data available about repeat burning effects on soils in the Sierra Nevada. Perhaps the most extensive research on the soil impacts of long-term, frequent prescribed fire programs has come from the southeastern U.S. Soil studies in southern pine forests have examined impacts following decades of annual burning and made comparisons to less frequently burned or unburned stands. As with single-entry prescribed fires, repeat burning results in reductions in the forest floor and the nutrients contained therein. The more frequent the burns, the greater the reduction in forest floor N content. For example, following 30 years of prescribed fire treatments in the Coastal Plain of South Carolina, forest floor N mass was reduced by 29, 60, 60, and 85 percent following fires every 4, 3, 2, and 1 years, respectively, relative to the 480 kg N/ha (0.21 t N/ac) in the control stand (Binkley 1992). Similarly, annual burning at another South Carolina site reduced forest floor N by 68 percent, and a fire return interval of 7 years resulted in a 32 percent loss of N relative to the 408 kg N/ha (0.18 t N/ac) in the control (McKee 1982). Although climate, soils, and forest type differ between the Sierra Nevada and the Southeast, the concept that more frequent fire results in greater cumulative forest floor N losses transcends these differences. Johnson et al. (1998) developed a nutrient cycling model to predict N changes following frequent prescribed fire at a site in the eastern Sierra Nevada. Using local litterfall rates, N concentration, and litter decay rates, they simulated the forest floor biomass and N content under varying fire frequencies and levels of fuel consumption. They showed that over a 100-year period, prescribed fires every 10 years would result in 35 percent more N loss than fires every 20 years, assuming half the forest floor is consumed. Allowing litter to accumulate for 100 years before it is completely consumed by wildfire would result in less than half the N loss modeled for a 10-year prescribed fire interval (Johnson et al. 2009).

Over time, repeated fires can lead to a gradual reduction in the N capital of a site. This does not suggest, however, that infrequent fire is the most desirable management strategy. Though more N is conserved under infrequent fire regimes, that scenario places overall soil resilience at risk. In that case, the forest floor N pool slowly swings between extreme states—unprecedented accumulation in thick duff and litter layers, then complete loss following wildfire. Furthermore, fires that completely consume the forest floor leave the mineral soil vulnerable to erosion and associated losses of nutrients and organic matter. Rather than broadly excluding fire to preserve on-site N pools, managers may choose to consider local N replenishment mechanisms following periodic fire and factor that into planning efforts. Soil resilience to N loss depends on the rate of N recovery. Predominant N input sources include atmospheric deposition and N fixation. Levels of N deposition depend largely on air pollution and weather patterns, which vary across the Sierra Nevada. The National Atmospheric Deposition Program maintains a long-term record of wet deposition chemistry and may be a useful starting point for managers to approximate deposition levels. In relatively unpolluted areas, N fixation is the most important N input source. Johnson et al. (2004) studied nutrient changes following the stand-replacing Little Valley wildfire in the eastern Sierra Nevada. Although 71 percent of above-ground N was consumed in the 1981 fire, additions from the N-fixing shrub *Ceanothus velutinus* (snowbrush) had more than made up for the losses 16 years later. If inputs were limited to deposition alone, the lost N would not be replaced for more than 1000 years at this relatively unpolluted site (Johnson et al. 2004). Other N-fixing species contribute far less to ecosystem N recovery. Slow growing N-fixing shrubs in northeastern California and central Oregon, including *Purshia tridentada* (bitterbrush) and *Ceanothus prostratus* (mahala mat), probably fix enough N to meet their own needs, but are unlikely to contribute enough to compensate for N losses following disturbances such as fire (Busse 2000). Johnson et al. (2009) suggest that frequent prescribed burning (with intervals less than 10-20 years) has potential to result in substantial N losses over time. It follows that historical frequent fire regimes would have also caused cumulative N losses, potentially reducing productivity over time. Similar losses from frequent prescribed fire might raise concerns about contemporary site productivity, but they may also be a desirable outcome, especially in watersheds with nutrient-sensitive waterbodies, such as Lake Tahoe and areas with heavy N pollution. Where cumulative N loss from repeat prescribed burns is a concern, techniques to increase burn heterogeneity, in which areas of the forest floor remain unburned, may be useful. In general, research on the effects of frequent prescribed burning in the Sierra Nevada is needed, as the effects of long-term fire suppression on forest soils in this region are not well understood (Miesel et al. 2011).

Effects of Wildfire

The impacts of wildfire on soil physical characteristics and factors involved in post-fire erosion are discussed in detail within the Post-wildfire Management chapter (4.3). Post-fire erosion often results in sedimentation to streams, which is discussed in the Watersheds and Stream Ecosystems chapter (6.1). Though a number of soil impacts are expected following a wildfire, including loss of the forest floor and associated C and N, increased erosion due to exposure of bare soil, and potential changes in soil structure and biota, there are very few studies of wildfire effects on soils that include comparisons of pre- and post-fire data. Only one such study exists in the Sierra Nevada; it took place on the southeast side of Lake Tahoe, where soil research plots burned in the 2002 Gondola Fire (Johnson et al. 2007, Murphy et al. 2006b). That fire resulted in a 20 percent reduction in ecosystem C and a 15 percent reduction in ecosystem N, due primarily to combustion of vegetation, the organic soil horizon, and large woody debris. Though the wildfire had no statistically significant effect on soil C and N, about one-fifth of the N lost was from mineral soil (Johnson et al. 2007). This is in contrast to most prescribed fires, which do not typically reach high enough temperatures to volatilize N in the soil. Unfortunately, no data on the intensity of the Gondola Fire were reported. Some of the C and N losses may have been removed by erosion. A few weeks after the fire, a high-intensity precipitation event (15 mm; 0.59 in) led to runoff and erosion of up to 1.4 cm (0.55 in) of soil from the study area (Murphy et al. 2006b). The ecosystem C is unlikely to be replenished until a mature forest is established at this site, whereas lost N may recover within a few decades if N-fixing shrubs, such as *Ceanothus velutinus,* colonize the site (Johnson et al. 2007).

A similar study of pre- and post-fire soil conditions was conducted in southwestern Oregon in the area that burned in the 2002 Biscuit Fire. However, the Biscuit Fire burned at high intensity, reaching temperatures >700°C (>1292°F), as evidenced by melted aluminum tags in the research area (Bormann et al. 2008). This study represents the first direct evidence of significant mineral soil C and N losses due to wildfire. Unlike the Gondola Fire, which did not affect mineral soil C pools, 60 percent of C lost from the organic and mineral horizons in the Biscuit Fire came from mineral soil. Similarly, 57 percent of the N lost from the organic and mineral horizons in the Biscuit Fire came from the mineral soil. The Biscuit Fire caused substantial losses of fine soil, totaling up to 127 Mg/ha (57 t/ac). This loss was likely caused by water erosion as well as convective transport in the fire's smoke plume (Bormann et al. 2008). The loss of soil organic matter may affect soil bulk density, structure, water-holding capacity, and nutrient content, contributing to declines in soil resilience. As a general conclusion regarding effects of wildfire, management strategies that reduce the potential for uncharacteristically severe wildfires in Sierra Nevada forests will help limit erosional losses and conserve essential soil functions.

Sidebar: Heavy Metals and Mercury

Heavy metal accumulation is a nation-wide environmental health hazard. Even relatively remote forest ecosystems are not immune from this problem—in fact, it is suspected that extensive fire suppression in the Sierra Nevada has led to a buildup of heavy metals, particularly mercury, in forest litter (Obrist et al. 2009). Of immediate concern is the potential for redistribution of litter and sediment-bound metals during fire, leading to unwanted pollution of lakes and reservoirs and, ultimately, heavy metal bioaccumulation in fish (Obrist 2012).

The environmental fate of mercury is reasonably well studied in Sierra Nevada soils and offers insight to the fates of other heavy metals such as lead, chromium, cadmium, nickel, and zinc. Mercury accumulation has resulted largely from human activity, coinciding with the start of the industrial revolution. For example, Drevnick et al. (2010) reported estimates of mercury flux to Lake Tahoe at 2 μg/m^2/year for preindustrial sediments and 15-20 μg/m^2/year for modern sediments. Key points relevant to the fate of mercury in Sierra Nevada forests are:

- Ninety to 98 percent of the total mercury in Sierran forests is found in mineral soil (Engle et al. 2006, Obrist et al. 2009, Obrist et al. 2011)
- Mercury is essentially inert in mineral soil and unaffected by wildfire or prescribed fire (Schroeder and Munthe 1998, Engle 2006). Large post-fire erosional events that transport mercury-bound sediments to streams, lakes, and reservoirs may be of concern (Caldwell 2000, Driscoll et al. 2007, Burke et al. 2010)
- Carbon and nitrogen rich soils (highly fertile) typically contain the highest concentrations of mercury (Obrist et al. 2009, Obrist et al. 2011). The corollary is that **many low-fertility Sierran soils are at low risk for mercury contamination**
- Combustion of forest litter is the primary source of mercury transport during fire (Engle et al. 2006, Obrist et al. 2009). Thus, **severe burning with complete combustion of the forest floor represents the greatest risk for mercury redistribution and potential bioaccumulation in Sierran waters**

Knowledge Gaps

Mastication is a technique used across the Sierra Nevada to thin forested stands and plantations and rearrange woody fuels. Mastication produces a layer of woody residue on the forest floor that has no natural analog. Debris size, depth, and density depend on the characteristics of the mastication equipment and masticated materials, as well as the time since treatment. Residues effectively serve as a mulch layer, reducing soil heating caused by solar radiation and retaining soil moisture due to reduced evaporation. But few studies have examined mastication effects on soils, particularly long-term responses as the debris settles and decays. Soil scientists have concerns about deep residues, and how they may impact rates of nutrient cycling, nitrogen availability, or soil aeration. Depending on the depth, density, and continuity of masticated debris, fire treatments in masticated stands may result in more severe effects to soils. No long-term studies exist to address these issues.

Many ground-based mechanical operations in the Sierra Nevada are limited to slopes less than 35 percent, but there is an increasing desire to treat steeper slopes. Operational knowledge is needed to effectively treat steep ground without substantially increasing the risk of soil loss, erosion, and sedimentation into streams. Existing equipment innovations may render the 35 percent slope restriction obsolete, and field-based trials and studies are needed to inform and enhance managers' options in this sensitive terrain.

The topic of coarse woody debris has received much attention as a habitat component for wildlife and a structural attribute of aquatic systems. Less is known about the importance of large wood for overall soil resilience, or what levels and types of woody material are desirable for Sierra Nevada ecosystems. Woody debris acts as a barrier against soil erosion, provides water to plants and microbes during summer drought, and contributes to nutrient cycling processes. However, actual ground cover of down wood is typically so low that the importance of these services may either be viewed as trivial or as highly valuable due to their relatively rare occurrence. Evaluations of the contribution of woody debris to soil ecosystems in the Sierra Nevada is needed to help establish desirable woody debris conditions, including size, quantity, and decay class distributions.

Soil biotas are essential to many basic soil processes, but our understanding of soil biodiversity—both composition and function—is limited. However, healthy soils are an important component of forest health, and further research in this area would complete management efforts in the synthesis region.

Sidebar: Dynamic Soils, Dynamic Data?

Soil survey data is a tremendous asset to land managers. However, soil resource inventories provide only a single snapshot in the life of a soil. Soils change over time, and long-term monitoring can inform adaptive management strategies to achieve sustainable, resilient forests. To this end, the Forest Inventory and Analysis (FIA) and Forest Health Monitoring programs of the USDA Forest Service have incorporated soil measurements into their national assessment scheme. On a very coarse spatial scale (1 soil plot per 38,450 ha (95,012 ac)), soil data are collected to monitor erosion, surface disturbance, and chemical and physical properties (O'Neill et al. 2005). Plots are to be re-measured every five years to capture changes in soil characteristics and condition. Over time, this ambitious undertaking will provide a wealth of data to conduct trend assessments and provide broad insights for management strategies and on long-term climatic influences. Already, the data have allowed a national assessment of forest floor C stocks (Woodall et al. 2012). One legitimate criticism of the FIA protocol is its focus on the upper soil, which neglects material greater than 20 cm deep (Harrison et al. 2011).

Management Implications

Prevent soil loss

- Maintaining soil in place is paramount to current and future soil quality, resilience, and health. Recovery of severe erosion is beyond human timescales.
- With proper design and BMPs, mechanical treatments and prescribed fires can be implemented with low risk of soil erosion.
- Severe wildfire, particularly at large scales, poses a high risk of post-fire soil loss through erosion.

Minimize physical disturbance

- Bare soil exposure can be minimal following mechanical treatments, but compacted skid trails can contribute to decreased soil function and to downstream sedimentation.
- Compaction may have beneficial soil impacts in sandy soils. In other cases, operational restrictions, such as soil moisture guidelines or equipment specifications, can be tailored to the specific soil type to limit compaction.
- Prescribed fire can greatly reduce the mass and depth of the forest floor. Needlecast from scorched trees can quickly replace lost soil cover.
- Combined thinning and prescribed fire treatments typically expose more bare soil than either practice alone.
- In most cases, the size and density of burn piles can be based on operational factors rather than potential soil heating effects.
- Severe wildfire can remove the forest floor and woody debris, expose bare soil, and alter soil structure and bulk density.

Evaluate changes in nutrient capital and turnover

- Whole-tree harvest techniques transport more nutrients off site than bole-only methods, but many Sierra Nevada sites have large soil N reservoirs and are fairly resistant to N loss regardless of thinning method.
- Prescribed fire removes C and N by combusting the forest floor, but C and N pools in the mineral soil typically remain unchanged
- Nutrient cycling models show that frequent, low-severity fire will cause greater overall nutrient loss than infrequent, high-severity fire where fuels have accumulated over many decades. At face value, reduced nutrient loss seems beneficial to soils, but extensive high-severity fire in fact poses far greater risks to long-term soil quality and resilience.
- Design repeat burns to produce patchy fuel consumption to temper nutrient losses from frequent fires.
- Simple balance sheets are useful to gain perspective on nutrient losses relative to existing pools and inputs or outputs over time.

Recognize effects on organic matter and soil biota

- Due to fire suppression, accumulations of litter and duff in many Sierra Nevada forests that evolved with frequent fires may exceed levels that occurred historically.
- Biologically healthy soil is critical to sustaining resilient forests, but predicting and quantifying management effects on soil organisms is generally beyond the reach of forest managers.
- Severe wildfires consume soil organic matter. This loss can affect soil bulk density, structure, water-holding capacity, and nutrient content, ultimately contributing to declines in soil resilience.

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6.0 Water Resources and Aquatic Ecosystems

Water resources and aquatic ecosystems in the Sierra Nevada and southern Cascades support critical ecological and socioeconomic values both within and well beyond the region. This section contains four chapters on different types of water-based ecosystems in the synthesis area. Chapter 6.1, Watersheds and Stream Ecosystems, considers challenges and threats facing those systems, including climate change and wildfire, before turning to recent research on water quantity and water quality, including how macroinvertebrates serve as indicators of water quality. Chapter 6.2, Forested Riparian Areas, focuses on the ecologically important transition zones between upland forests and streams. It discusses current understandings of the role of fire in riparian ecosystems, as well as findings about opportunities for management to restore those areas. Wet meadows, the subject of chapter 6.3, have been a special focus of restoration efforts and research in the synthesis area and in other regions. Chapter 6.4, Lakes, discusses recent research and restoration strategies for high-elevation lake ecosystems; it examines a multitude of stressors, including climate change, pollution, introduced fishes, and diseases. Though these different kinds of systems are related through the flow of water, they have distinct ecological issues and management challenges. Taken together, these chapters feature strategies to promote resilience that complement the broader themes of the synthesis, including an emphasis on promoting or emulating natural disturbance regimes, considering the larger spatial and temporal contexts of these systems, and understanding linkages between ecological processes and social values. As water travels, it integrates landscape influences, so that downstream waterbodies and their aquatic organisms reflect the condition of terrestrial and aerial environments. Accordingly, these chapters emphasize the connections between aquatic ecosystems and other forest components that are discussed in the chapters on Forest Ecology (2.0), Fire and Fuels (4.1), Fire and Tribal Cultural Resources (4.2), Post-wildfire Management (4.3), Soils (5.0) and Air Quality (8.0).

6.1 Watershed and Stream Ecosystems

Carolyn Hunsaker and Jonathan Long with contribution from David Herbst

Contents

Introduction

All life depends on an adequate supply of water. When the Forest Service was established through the Organic Act of 1897, one of the primary reasons for establishing the public forest reservations was "for the purpose of securing favorable conditions of water flows." How much and where water occurs is a direct function of climate and weather patterns. Soils, topography, and vegetation affect how water is partitioned in the landscape, and these factors, along with human activities, air quality, and ecosystem disturbances, affect the quality of water. Measurements of physical, chemical, and biological characteristics serve to characterize the condition or health of water resources and aquatic ecosystems. Monitoring environmental attributes at different scales or by doing research, especially by designed experiments, help to gain knowledge about effects of land use practices. The Adaptive Management Plan, Appendix E, in the Sierra Nevada Framework (USDA 2001, 2004) described the need for status and trends monitoring and research; it also identified priority questions and knowledge gaps that required new information to improve Forest Service management of water resources and aquatic ecosystems. Since that time, most Forest Service efforts for aquatic resources in the Sierra Nevada were directed to studies of amphibians, grazing practices, and invertebrates. More recent attention has been given to meadow restoration. A long-term watershed research project in the Sierra Nevada was established by the Pacific Southwest Research Station in 2000 at the Kings River Experimental Watersheds (KREW), which includes a portion of the Teakettle Experimental Forest. This research site has attracted National Science Foundation funding for the establishment of the Southern Sierra Critical Zone Observatory,[1] which is starting to provide new information on hydrology and geosciences in the Sierra Nevada. Older watershed research sites with long investments exist at California's Blodgett Forest Research Station and Sagehen Experimental Forest (see Figure 1 in Chapter 1.4).

This chapter begins with a review of values and services associated with aquatic ecosystems. It then considers climate change and wildfire before turning to recent science on water quantity, water quality, and macroinvertebrates as indicators of water quality. This chapter concludes with a discussion of management strategies to promote resilience of aquatic ecosystems.

[1] http://criticalzone.org/sierra

Values and Services of Aquatic Ecosystems

> *Water, in all its forms, is indeed the crowning glory of the Sierra. Whether in motion or at rest, the waters of the Sierra are a constant joy to the beholder. …Above all, they are the Sierra's greatest contribution to human welfare.*
>
> History of the Sierra Nevada, F.P. Farquhar (1965: 1)

National forests supply 45 percent of California's water, and most of the state's surface water originates in the Sierra Nevada. For the Pacific Southwest Region, one of the most valuable ecosystem services that the national forests provide is an adequate supply of good water for aquatic ecosystems and human needs. California's economy is highly dependent on agriculture, and much of the country relies on the fresh fruits and vegetables produced in California. Water is therefore pivotal to California's economy and the country's food supply. Furthermore, human recreation is highly influenced by the presence of water bodies.

Streams, riparian areas, and wet meadows support a wide range of social, cultural, and ecological values, including plant and wildlife diversity, water quality, water quantity, cultural values, aesthetic values, sport fishing, and tourism. Native American cultural resources are often concentrated along perennial streams due to availability of water and culturally important plants, travel corridors, and other patterns that facilitated settlement (Jackson 1988). Efforts to restore stream and meadow systems should consider activities such as timber harvest, recreation, and livestock grazing (see Managing Forest Products for Community Resilience chapter (9.5)) that influence the condition of riparian and meadow areas. These areas can sustain a diverse array of ecosystem services and ameliorate effects of climate change (see Wet Meadows chapter (6.3)).

The value of riparian ecosystems (termed *aquatic-terrestrial ecotones* in the international literature) has been written about extensively (Malanson 1993, Holland et al. 1991, Pinay et al. 1990, Petts 1990). Historically, riparian ecosystems were valued for their economic uses: transportation corridors, water supply and electricity, construction materials and waste disposal, agriculture and livestock, and settlement. The more recently recognized economic, social, and biological values of riparian ecosystems are listed in Table 1. Riparian areas are unique environments because of their position in the landscape; they are both ecotones between the terrestrial and aquatic zones, and corridors across regions (Malanson 1993). The term *ecotone* was first used in 1905 by Clements; with the development of the discipline of landscape ecology, there was a renewed interest in the ecotone concept in Europe around 1990, and it was explored by the Man and Biosphere program (Holland et. al 1991, Naiman and Decamps 1990). An ecotone was then defined as "a zone of transition between adjacent ecological systems, having a set of characteristics uniquely defined by space and time scales and by the strength of interactions between adjacent ecological systems" (Naiman and Decamps 1990: 3)

Table 1. Values of riparian ecosystems from the referenced literature.

Economic
Reduce downstream flooding
Recharge aquifers
Surface water supply in arid regions
Support secondary productivity, e.g., for fisheries
High yields of timber
Social
Recycle nutrients
Store heavy metals and toxins
Filter of diffuse pollution from uplands
Accumulate organic matter as a sink for CO_2
Intermediate storage for sediments
Natural heritage
Recreation
Aesthetics
Natural laboratories for teaching and research
Biological
Special habitat for some endangered or threatened species
Habitat for aquatic species
Refugia and water for upland species
Corridors for species movement

Resilience and Degradation in Stream Ecosystems

The definition of ecological resilience in the Introduction chapter (1.0) is the amount of disturbance an ecosystem can absorb without crossing a threshold to a different stable state, where a different range of variation of ecological processes and structures reigns (Gunderson 2000). This general approach is also reflected in the concept of "dynamic equilibrium," which Heede (1980) described as the capacity of streams to adjust to perturbations within a few years. These concepts clearly depend on the timeframe being considered and the range of variation in processes and structures. Understanding thresholds of erosion beyond which long-term sustainability is jeopardized requires extensive monitoring and understanding of reference conditions and natural variability. Developing site-specific restoration and management strategies therefore requires consideration of the evolutionary history of particular sites (Miller et al. 2001).

Some scientists have challenged the concept of dynamic equilibrium by arguing that many fluvial systems are inherently unstable (Lave 2009). Fluvial systems in Mediterranean climates in particular have been characterized as highly variable and ever-changing (Kondolf et al. 2012). Reflecting this view, scientists in recent years have challenged efforts that emphasize promoting channel stability, and they cautioned that management and restoration approaches are often overprotective in seeking to avoid disturbances and erosion. They pointed out that channel instability may have desirable consequences;

for example, erosion and deposition following major disturbances, such as wildfires, can be important for maintaining stream functions and biodiversity (Bisson et al. 2003). Florsheim et al. (2008) outlined the various benefits of steambank erosion for maintaining aquatic habitat diversity and reiterated that total elimination of bank erosion should not be a goal when restoring streams.

Climate Change Effects on Watersheds and Stream Ecosystems

Effects on Hydrology

Anticipating that a changing climate in California will substantially affect water resources and aquatic ecosystems, strategies for assessing the impacts of altered stream flows need to be developed. Changes in the Sierra Nevada, the primary source area of water in the state, are of particular concern. Warming has produced a shift toward more precipitation falling as rain than snow, and this reduces snowpack water storage, causes earlier runoff, increases the frequency of major floods through rain-on-snow events, and diminishes late season flows and the stability of headwater habitats that are important for maintaining watershed hydrological and ecological function (Figure 1). The California Department of Water Resources (DWR 2006) identified some other potential effects of climate change in California on water resources, including changes in vegetation, increased incidence of wildfires, increased water temperatures, and changes in human water demand.

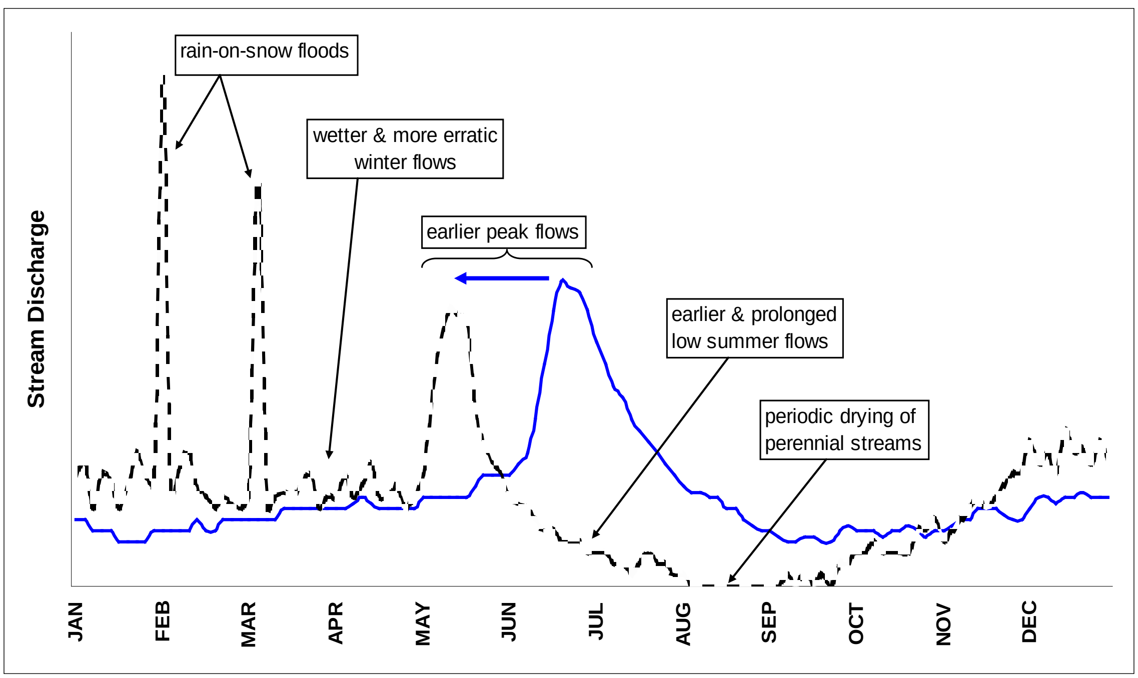


Figure 1: Conceptualization of the climate-driven changes (gray line) to the natural hydrograph (blue line) of a Sierra Nevada stream (from D. Herbst).

The water resources of the western United States depend heavily on snowpack to store part of the winter precipitation into the drier summer months. A well-documented shift toward earlier runoff in recent decades has been attributed to more precipitation falling as rain instead of snow and earlier snowmelt (Knowles et al. 2006). The start date of snowmelt is earlier now by about 15 days, based on data from 1960 to 2000 (Figure 2). A decline in the mountain snowpack of western North America has also been documented (Mote et al. 2005, Barnett et al. 2008). A publication by the California Department of Water Resources (DWR 2006) shows how hydrologic patterns by river basin have already changed in California during the past 100 years. There is a slight decreasing trend in precipitation in central and southern California and increased variability in precipitation. There is also a difference between changes in northern and southern California. For example, the total annual water year runoff has been increasing for the Sacramento River basins (northern and central Sierra Nevada) and decreasing for the San Joaquin River basins (central and southern Sierra Nevada). However, both areas experienced decreases in spring runoff; runoff from April through July declined by 9 percent for the Sacramento River basins and declined by 7 percent for the San Joaquin River basins.

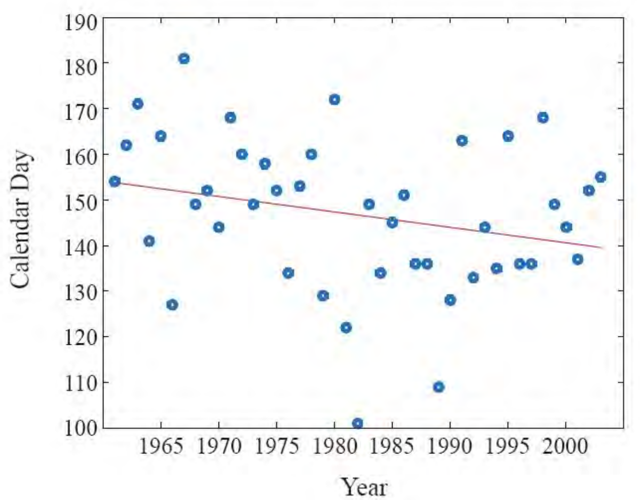


Figure 2: Trend in the timing of snowmelt discharge based on the day of maximum daily discharge, Kern River, California (from Peterson 2008).

Snowpack provides 20 percent of California's total runoff and 35 percent of its usable surface water. Climate modeling predicts a loss of snowpack for the mountains in California (Table 2), with a greater effect in the northern Sierra Nevada, where the mountains are lower in elevation than the southern Sierra (Knowles 2002). A change in surface water quantity of this magnitude will affect aquatic ecosystems and human uses.

Table 2. Temperature increase and effect on Sierra Nevada snowpack (Knowles 2002)

Temperature increase (°C)	Loss of snowpack (%)	Predicted year of effect
0.6	5	2030
1.6	33	2060
2.1	50 (43 in south and 66 in north)	2090

In the past 10 years, many publications and modeling efforts have focused on predicting climate change effects on air temperature and precipitation. Global climate models use a grid that is too coarse to adequately depict the complex structure of temperature and precipitation in California, especially the

Sierra Nevada. A statistical technique allows coarse data to be “downscaled” to a finer level of detail, and a grid scale of 12 kilometers (7 miles) was available by 2006 (Cayan et al. 2006). More recent work is downscaling data to even finer grids that allow predictions on possible changes to other attributes, such as stream discharge, water quality, and erosion. For example, Ficklin et al. (2012) developed and applied a hydroclimatological stream temperature model within the Soil and Water Assessment Tool (SWAT) to mountain areas of the western United States. These scientists are also working on projections of (1) future hydrologic flow components for the major river basins of the Sierra Nevada using an ensemble of general circulation models, and (2) the effects of climate change on water quality (stream temperature, dissolved oxygen concentration, and sediment concentration) in the Sierra Nevada. The U.S. Geological Survey in Sacramento has an ongoing study of the effects of climate on snowmelt and water availability in the southern Sierra Nevada (Tuolumne, Merced, San Joaquin, King, and Kaweah river basins). These new downscaling efforts and predictions at the river basin scale and for smaller watersheds will be useful to forest managers in considering climate change effects on water resources. For example, potential losses of the ecological services afforded by mountain meadows appear to be particularly high in several central Sierra Nevada watersheds, including the American, Mokelumne, Tuolumne, and Merced (Null et al. 2010). The Kings River Experimental Watersheds (KREW) research can also help to understand climate change effects for the southern Sierra Nevada, since five of the KREW streams are located in the rain-snow interface zone and five are in the snow-dominated zone. The current functioning of the lower elevation streams provides valuable insight into what can be expected for higher elevation streams with a 2°C air temperature shift (Hunsaker et al. 2012, Bales et al. 2011a).

Bales et al. (2006) identified three pressing hydrologic information needs for the western mountains of the United States given climate change, population growth, and land use change:

- To better understand the processes controlling the partitioning of energy and water fluxes within and out from these systems
- To better understand feedbacks between hydrological fluxes and biogeochemical and ecological processes
- To enhance our physical and empirical understanding with integrated measurement strategies and information systems

New information is being developed on these topics with field instrumentation and data analyses by the Sierra Nevada Adaptive Management Project (SNAMP), the Southern Sierra Critical Zone Observatory (SSCZO), and the Kings River Experimental Watersheds (KREW).

Effects on Channel Processes

Because climate change is expected to increase rainfall and storm intensity (Moody and Martin 2009) and the occurrence of uncharacteristically severe wildfire (Miller et al. 2009), flooding and sediment movement may increase due to the incidence of rain-on-snow events or post-wildfire floods, which could in turn reduce channel stability and habitat quality. Negative impacts of climate change may be especially pronounced in high-elevation, currently snow-dominated systems that shift toward more winter rainfall (Battin et al. 2007). Herbst and Cooper (2010) suggested increased rain-on-snow floods might pose a particular threat to streams that are already degraded (see Effects of Floods section in this chapter). Riebe et al. (2001) concluded that outside of glacial transition periods, climate change is unlikely to substantially affect watershed-scale erosion rates in the Sierra Nevada; however, they cautioned that climate change could alter sediment storage in floodplains, terraces, and colluvial hollows, which would affect short-term sediment delivery and channel stability. In reviewing effects of climate change on streams in the mountains of Idaho, Goode et al. (2012) contended that sediment yield could increase tenfold compared to recent historical levels due to increases in post-fire debris flows. Because climate change is expected to increase the incidence of severe wildfire, high-intensity storms, and rain-on-snow events, the threat of post-wildfire debris flows is expected to increase and become more widespread (Cannon and DeGraff 2009). If post-fire landforms persist beyond the wildfire recurrence interval, successive wildfires will have an important cumulative impact on watershed morphology (Moody and Martin 2001).

Debris Flows

Intense storms, in many cases following wildfires, can trigger debris flows.[2] Most debris-flow activity occurs within about two years following a fire, since revegetation tends to quickly stabilize hillslopes; however, substantial hazards from flash flooding could remain for many years after a fire (Cannon and Michael 2011). In studies of post-fire debris-flow processes throughout the western United States, the great majority of fire-related debris flows initiate through a process of progressive bulking of storm runoff with sediment eroded both from hillslopes and from channels, rather than from infiltration-triggered landsliding. Statistical-empirical models have been developed to estimate the probability and volume of debris flows that may be produced from burned drainage basins as a function of different measures of fire severity and extent, gradient, and soil physical properties in the basin (Cannon and Michael 2011). The probability model was developed using data from 388 basins in 15 recently burned areas of the western United States, and the volume model was developed from 55 debris-flow-producing basins burned by eight different fires where the volume could be attributed to a single storm. This modeling work used a 30-minute-duration, 10-year-recurrence rainstorm of 0.73 in to trigger an event. Intense rainfall events, rain-on-snow storms, and rapid snowmelt are all associated with debris flow occurrence in the Sierra Nevada. Cannon et al. (2008) summarized research on rainfall thresholds that result in significant movement of runoff and sedimentation events from recently burned watersheds and found that a 30 min peak rainfall intensity greater than 10 mm/h resulted in significant increases in runoff and intensities greater than 20 mm/h resulted in significant sediment movement. The association between wildfire, debris flow, and floods is well established in the southern Sierra Nevada (DeGraff et al. 2011), and the modeling work by Cannon and Michael (2011) enables risk

[2]Land and rock slides are another geomorphic disturbance that can deliver sediment to stream networks in the synthesis area. However, compared to some mountain ranges, such as the European Alps or the Himalayas, the Sierra Nevada generates relatively infrequent massive rock slides. In the southern Sierra Nevada, nine slides have been documented from prehistoric times to 1997, ranging in size from 23,000 m^3 to 11 million m^3 (Harp et al. 2008). Such slides have had severe impacts on people, communities, and infrastructure and can create dams in steep river canyons.

potential and volume of sediment from wildfires to be estimated for comparison to sediment from management activities.

Large debris flows down channels may be among the most persistent effects of wildfires (Benda et al. 2003, Goode et al. 2012, Moody and Martin 2001). Debris flows are a major concern in southern California and the Intermountain Region, and those flows tend to be larger than flows in more humid climates, such as the Pacific Northwest (Santi and Morandi 2012). The wetter western slopes of the Sierra Nevada have experienced debris flows associated with landslides following high rates of rainfall, rapid snowmelt, or rain-on-snow events (DeGraff 1994). In addition, there have been instances of post-wildfire debris flows from burned watersheds upslope from El Portal, CA, near Yosemite National Park (Cannon and DeGraff 2009). In the Southern Sierra Nevada, monsoonal storms on July 12, 2008 produced intense rainfall that triggered large debris flows in the southern Sierra Nevada (Figure 3) (DeGraff et al. 2011). One flow traveled down the north and south forks of Oak Creek through the town of Independence on the east side of the Sierra Nevada; it resulted in substantial damage to homes, a Forest Service campground, and other infrastructure. The other flow traveled down Erskine Creek through the town of Lake Isabella, CA and into the Kern River on the southern end of the Sierra Nevada. The Inyo Complex fire had burned 30 percent of the Oak Creek watershed at high or moderate severity in 2007, and the Piute Fire had burned 15 percent of the Erskine Creek watershed at high or moderate severity, but the two events shared relatively intense rainfall (16-30 mm/hour) (DeGraff et al. 2011). These incidents demonstrate that post-fire debris flows are a significant concern in the southern and eastern parts of the Sierra Nevada, which experience high-intensity rain storms. Further research would be needed to evaluate risks within the synthesis area, given high amounts of variability in these watershed processes. A comparison of rainfall regimes by Moody and Martin (2009) showed that the region that includes the Sierra Nevada experiences less intense rainfall than the mountains of Arizona, but more intense rainfall than in the Great Basin. However, they found very high variability within the Pacific region and poor correlation between post-wildfire sediment yields and rainfall intensity (measured as the average 2 year event over 30-minute periods).

Figure 3: Damage to residences along Oak Creek following the post-fire debris flow incident of 7/12/2008 on the Inyo National Forest. Photo by Jerome V. De Graff.

Effects on Aquatic Ecosystems

Projected effects of climate change on aquatic ecosystems include hydrologic effects discussed above (especially lower summer baseflows, earlier runoff, and higher summer water temperatures), and well as biological effects, such as increased isolation of native aquatic species and spread of invasive species (Viers and Rheinheimer 2011).

Impacts on trout

The projected impacts of climate change on trout and salmon species are a particular concern because of the vulnerability of those species to increased stream temperatures (Moyle et al. 2011). Although the middle of the Sierra Nevada includes a large area that was historically fishless, the northern and southern Sierra Nevada support several endemic strains of native trout (Figure 4). Several varieties of redband or rainbow trout (*Oncorhynchus mykiss*) evolved in the Sacramento-San Joaquin drainages of the Sierra Nevada, while varieties of cutthroat trout (*Oncorhynchus clarkii*) evolved within the interior east-side drainages of the Lahontan basin (Behnke 2002). Due to stocking of non-native trouts, translocations of trouts outside of their native streams, and impacts to habitats, most of these native trout have become confined to relatively small streams, leaving them vulnerable to the effects of climate change and wildfire (see Research Gaps at the end of this chapter for recent reports on this subject). Wenger et al. (2011) forecasted significant declines in trout habitat and associated socioeconomic consequences across the interior western United States over the next 60 years.

Post-wildfire floods that reorganize channel habitats can have significant impacts on fish populations, including extirpation of isolated native trout populations in headwater streams in the Southwest (Brown et al. 2001). A study by Isaak et al. (2010) in Idaho demonstrated that severe wildfires followed by channel-reorganizing floods can increase rates of stream warming over long periods. These events can cause streams to warm by removing vegetation and widening channels; those effects may offset the potential of such events to lower temperatures by increasing base flow (a potential short-term effect due to reduced transpiration in the watershed) (Sugihara et al. 2006), or by increasing heat exchange with colder groundwater (Dunham et al. 2007). In contrast, native fish populations that inhabit relatively intact stream networks in the Northwest and the northern Rocky Mountains have demonstrated resilience following wildfires (Gresswell 1999, Neville et al. 2009). However, the responses of aquatic systems to wildfire and climate change observed in other regions may not transfer well across the synthesis area due to variations in climate, topography, extent of non-native competitors, and connectivity of aquatic populations.

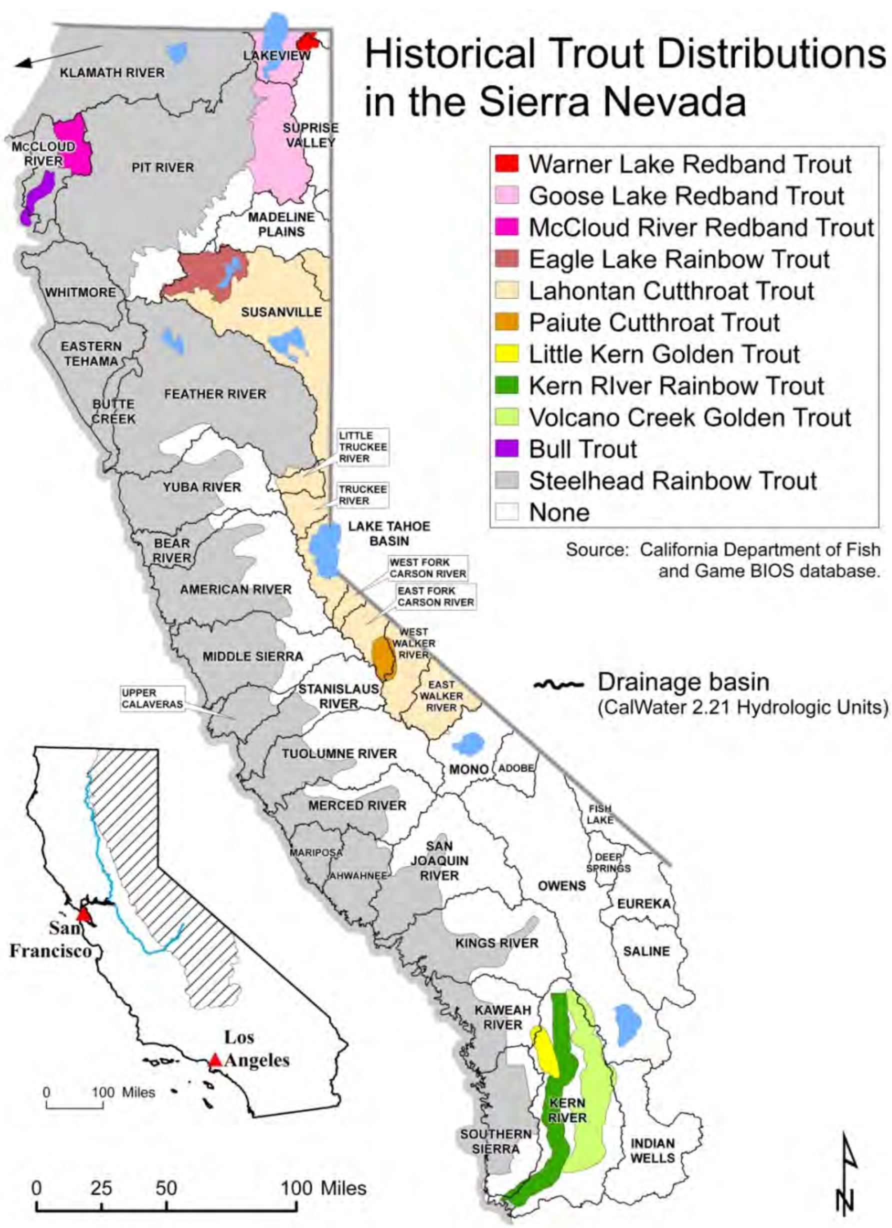


Figure 4: Historical distribution of native trouts within the synthesis area.

Sidebar: Reports on threats from climate change and wildfire on aquatic species

Trout Unlimited has generated a series of reports under its Conservation Success Index (CSI) program (Williams et al. 2007) that characterize risks for each of the native trout and salmon species due to changes in climate and fire regime within their ranges.

Even more recently, researchers at UC Davis prepared a white paper report on the effects of future climates on freshwater fishes (Moyle et al. 2012).

In addition to the recently published study of post-fire debris flows by (DeGraff et al. 2011), the Forest Service has conducted monitoring of impacts to aquatic ecosystems following recent fires, including the Moonlight Fire (2007), Cub Fire (2008), and Lion Fire (2011). These observational efforts should afford opportunities to evaluate resilience of streams in the synthesis area to wildfires of different severities.

Effects on Biological Indicators of Water Quality

Understanding particular effects of climate change on the biology of mountain streams would be valuable, since management agencies use biological criteria to measure trends of ecological health, water quality, and the integrity of ecosystem function. In particular, streams in relatively undisturbed watersheds serve as references to evaluate condition. Against a background of climate-driven alteration to the ecology of streams across the Sierra, the biological integrity of reference streams may decline. Biological diversity in confined headwater and alpine streams may be especially sensitive to shifting hydrologic patterns. Even though all streams and lakes are affected by climate change, the reference habitats may have more to lose than disturbed streams that have already been impacted by localized sources of pollution and other forms of degradation. If reference streams lose a higher proportion of aquatic life due to warming and hydrologic disruption, then the “signal” or difference relative to disturbed test sites would be decreased. The reference condition for streams is typically developed based on many sites sampled over many years, so if these streams slowly degrade, the range of variability or “noise” in the cumulative reference condition will increase. The net effect of a declining reference condition is that it will be harder to detect degradation by non-climate factors. Establishment of current conditions and quantification of climate-induced drift would help to monitor conditions within the Sierra Nevada and to recalibrate standards as climate conditions change.

Sidebar on Stream Monitoring Network for Climate Change

In 2010, the Management Indicator Species Program of the Pacific Southwest Region funded the establishment of a stream observatory network, designed by David Herbst of the Sierra Nevada Aquatic Research Laboratory, to detect the ecological impacts of climate-induced changes in hydrologic balance and temperature of Sierra Nevada streams and to provide a historical context for recovery of degraded ecological values. The network includes 12 sites (6 in the southern Sierra Nevada and 6 in the northern Sierra Nevada) that serve as undisturbed reference sites for streams that are expected to have high and low risk for climate-induced loss of snow cover and hydrologic stability, in combination with high and low resistance to climate change. This network also sets up a natural experiment within which differing hypothesized risks based on forecasted climate conditions and hydrographic susceptibility can be contrasted. The sites are broadly representative of Sierra Nevada streams across a range of elevations from 4,000 to 12,000 ft. Measurements at these sites include stream invertebrates, algae periphyton, water chemistry, geomorphic characteristics, stage height, riparian cover, and water and air temperature. Benthic invertebrate samples from this network partition into two community groupings—those sites north of Yosemite and those south of Yosemite. A possible difference is that southern Sierra Nevada streams have less groundwater recharge based on a comparison of their chemistry with that of northern streams, and they are thus more susceptible to low flows and drying and support less biological invertebrate diversity. Southern Sierra Nevada streams with longer upstream length may be less prone to drying and therefore have a higher level of diversity.

Water Resources

This subchapter includes a discussion of ecosystem processes and issues that are important to water quantity and quality. Stream benthic macro-invertebrates are included here as important biological indicators of water quality.

Water Quantity

Streamflow response to a change in forest density is strongly related to climate, species composition, and the percentage change in vegetation density. Troendle et al. (2010) provided a review of the many studies on this topic and made the following observation (pages 126-127): “The data from 95 watershed experiments conducted in the United States show that, on average, annual runoff increases by nearly 2.5 mm for each one percent of watershed area harvested (Stednick 1996). Because runoff is quite variable from year to year, the general conclusion is that approximately 20 percent of the basal area of the vegetation must be removed before a statistically significant change in annual runoff can be detected (Bosch and Hewlett 1982, Hibbert 1967, Stednick 1996)." Because most of these paired watershed experiments imposed a partial or complete clearcutting of the mature trees, our current understanding of the hydrologic impacts of thinning and prescribed fire comes from inference supported by some plot and process studies.

Many fuels management treatments or forest restoration efforts remove less than 20 percent of the basal area of trees; although this may result in a change in flow, it likely will not be detectable, especially in dry years. With best management practices (BMPs), which should not cause overland flow from skid trails or soil compaction, there should be little or no detectable effect on peak discharges. Any change will be short-lived because of vegetation regrowth, except in cold snow zones (Rocky Mountain region). Prescribed fire by itself is less likely to influence water yield than mechanical treatments because of the smaller reduction in basal area and lack of ground disturbance by heavy machinery (Troendle et al. 2010).

During the past decade, a better understanding of hydrologic processes has developed for the southern Sierra Nevada. Hunsaker et al. (2012) characterized the climate and hydrologic patterns for eight headwater catchments, including both the rain-snow transition zone and the snow-dominated zone of the southern Sierra Nevada. A water-balance instrument cluster at these rain-snow catchments enabled an estimate of total annual evapotranspiration at 76 cm in 2009, a value higher than previous estimates for the Sierra Nevada (Bales et al. 2011a). Water loss rates from soil were estimated to be 0.5 to 1.0 cm d^{-1} during the winter and snowmelt seasons. Soon there will be data on the effect of both prescribed fire and mechanical removal of vegetation on streamflow (see sidebar for Kings River Experimental Watersheds).

Engle et al. (2008) provide the only new experimental data on streamflow response to prescribed fire in the Sierra Nevada. The Tharps Creek watershed (13 ha) was burned after having no fire for at least 120 years; the pre-burn surface fuel load was 210 Mg ha^{-1} and fuels were reduced by 85 percent as a result of the fire. After fire, runoff coefficients increased by seven percent in dry years and 35 percent in wet years. (The runoff coefficient is the relationship between the amount of runoff to the amount of precipitation; the value is large for areas with low infiltration and high runoff.) Mean annual runoff in the 50 ha Log watershed (control) during the dry years was 29 percent of precipitation; mean annual runoff was 56 percent of precipitation when drought years are excluded. Runoff coefficients in the Tharps watershed were consistently lower than in the Log watershed, averaging 51 percent during wet years and eight percent during dry years. Nine years after the burn, there was no evidence that runoff in Tharps watershed was returning to pre-fire levels.

Sidebar: Water Yield Predictions

A recent report based on modeling suggests a somewhat different conclusion from Troendle et al. (2010) about the ability to increase water yield through forest harvesting. Bales et al. (2011b) suggest that reducing forest cover by 40 percent of maximum levels (based on leaf area index (LAI)) across a watershed could increase water yields by about 9 percent. Treatment proposals at the Onion Creek Experimental Forest on the Tahoe National Forest estimate that treatments could increase water yield by as much as 16 percent and extend snow storage (i.e., delay snowmelt) by days to weeks. Recent studies in the Sierra Nevada report potential increases in snow accumulation of 14 to 34 percent due to forest harvest (see Bales et al. 2011b).

Water Quality

General condition

Sierra Nevada waters are impacted by both recent human activity and “legacy” impacts; the latter tended to be more severe (historical mining, intense logging, railroad construction, etc.). This discussion focuses on issues we believe are common across the synthesis area and need to be monitored into the future for maintenance of good water quality or improvement in quality. We recognize that legacy impacts (e.g., mercury contamination from historical mining, high sediment loads from old and poorly constructed roads, or human health hazards from old septic systems) can be substantial in local areas, but they are not a focus of this review. Some human health issues with forest water quality are discussed in chapters 6.3 (Wet Meadows) and 9.3 (Sociocultural Perspectives on Threats, Risks, and Health).

The Sierra Nevada is made up of 24 major watersheds, with 16 watersheds on the west side and eight watersheds on the east side. Water from the Sierra accounts for 60 percent of the total dollar value of all natural products or services produced by the entire region—more than forest products, agricultural products, recreational services, or even residential development (SNEP 1996). A number of indicators can be used to characterize water quality, including chemical indicators (nutrients, conductivity, pH, metals, pathogens, pesticides, and organics), physical indicators (temperature and sediment), biological indicators such as stream invertebrates, and human exposure indicators (swimmable, fishable, drinkable). A general overview of water quality for major Sierra Nevada river basins is based on publicly available data about various indicators—some quantitative and some qualitative (Timmer et al. 2006).

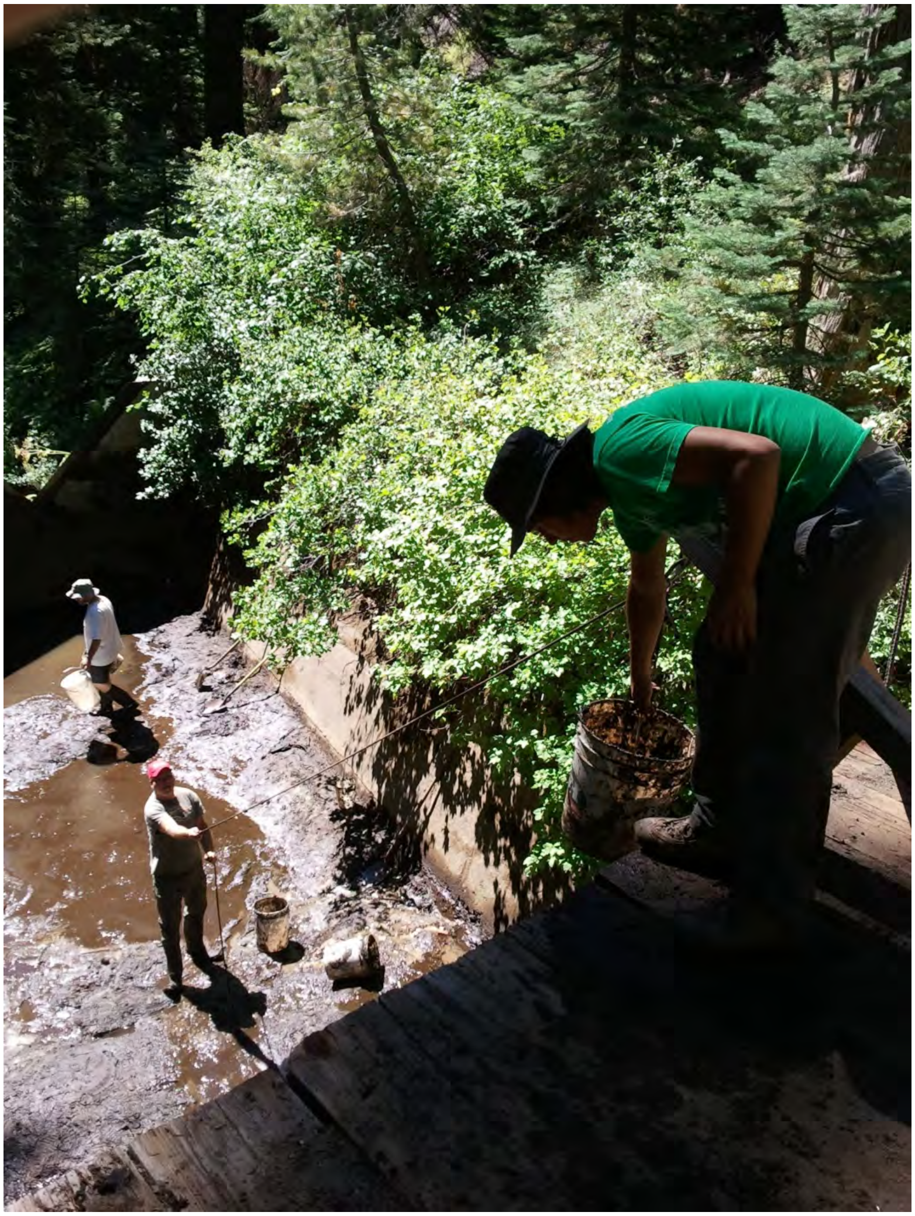
The U.S. Environmental Protection Agency (EPA), State of California, and Forest Service all use macro-invertebrates as a biological indicator of water quality along with measures of stream physical habitat and water chemistry. An ecological condition assessment (2000 through 2006) of California’s perennial wadeable streams provides an overview of water quality for forested lands (Ode 2007). This assessment reported that forests had 80 percent of the monitored stream segments in an *unimpaired* condition, as compared to 80 percent being *impaired* for agriculture and urban land use. When a forest stream segment was very impaired, the following stressors were associated the most with that poor condition: total nitrogen (30%), chloride (20%), total phosphorus (10%), lack of habitat complexity (20%), and riparian disturbance and streambed stability (10%).

National forests are mostly in the headwaters of Sierra Nevada river basins; often the impaired portion of a river or stream is below national forests or associated with reservoirs or other impoundments. As an example, Timmer et al. (2006) reports in their *State of Sierra Waters* the following information for the

Kings River watershed. The upper North Fork has been listed as *impaired* for wetland habitat and flow alterations by the EPA and listed as threatened for habitat, fishery, and freshwater by the California State Water Resources Control Board. The Main Fork Kings River is listed by the EPA as *impaired* for flow alterations and threatened for habitat, fishery, and freshwater. Timmer et al. (2006) listed the probable sources for these detrimental impacts as construction, agriculture or nursery operations, and modification of the streambed. At Pine Flat Reservoir, EPA listed the Kings River as *impaired* for pathogens, habitat, and freshwater and as *threatened* for swimming, fishing, fish tissue concentrations, and recreation user days. The *threatened* designation means the water currently supports designated uses, but may become impaired in the future if pollution control actions are not taken. *Impaired* means a designated water use is not supported. This report also indicates if a water body is impacted by a particular metal and if human exposure is a concern. Similar general condition information exists for all major watersheds in the Sierra Nevada (Timmer et al. 2006).

Stream sediment and erosion

Undisturbed forests are an important source of the clean water that is necessary for ecosystem health as well as urban and agricultural uses. By altering infiltration rates and evapotranspiration rates and disturbing the soil, forest management activities (including road construction, timber harvesting, site preparation, fuels reduction, and prescribed fire) can increase overland flow rates and sediment yields. Sediment yields are dependent on many factors: climate, topography, soil type, vegetation, historical land use, and the dominant erosion processes (Stednick 2000). Robichaud et al. (2010) provided a review of fuels management effects on erosion. Reported sediment yields from undisturbed forests in the western United States are 0.003 t ac^{-1} (0.007 Mg ha^{-1}), but values up to 11 t ac^{-1} (25 Mg ha^{-1}) have been reported (Stednick 2000). Hunsaker and Neary (2012) reported an average of 16 ± 21 kg/ha (0.016 ± 0.021 Mg ha^{-1}) over seven years of measurement at the undisturbed Teakettle Experimental Forest in the headwaters of the Kings River (Mg = 10^6 grams or 1 metric tonne). Breazeale (1972) reported mean annual sediment rates of 110 t mi^{-2} for the North Fork of the Kings River, and Dunne and Reid (1985) reported 43 t mi^{-2} for the Teakettle Experimental Forest.



Research to better understand cumulative watershed effects in the Sierra Nevada has focused on collecting sediment data in the Eldorado National Forest (MacDonald et al. 2004). The median sediment production rate from roads was 0.2 kg m^{-2}, nearly an order of magnitude higher than any of the other sources measured by 91 sediment fences (skid trails, off-road vehicle trails, hillslopes burned by prescribed fire and wildfire, undisturbed).

Any type of land use change that causes soil disturbance or vegetation removal (timber harvest, brush clearing for fuels reduction, fire, and road construction, use, and decommissioning) has the potential to cause erosion and subsequent sediment delivery to water bodies. Historically, roads have been considered the primary source of sediment and a significant problem in many landscapes. For the Sierra Nevada, at this point in time, the potential for erosion and sediment effects on water quality and aquatic habitat from forest management and wildfire needs to be evaluated with current knowledge and practices. Coniferous forests across western North America are experiencing widespread mortality as a result of drought, insect outbreaks, and wildfire associated with climate change. In many of these landscapes, wildfires and subsequent storms commonly result in the delivery of large, infrequent pulses of sediment to water bodies. Goode et al. (2012) suggested that sediment yields may be roughly ten times greater with climate-modulated processes than those observed during the 20th century. Although coarse sediment is important for forming geomorphology and aquatic habitat, an order of magnitude increase may have undesirable impacts to aquatic organisms and reservoir management and life expectancy.

Gucinski et al. (2001) synthesized scientific information on forest roads and noted that the Forest Service has a framework (USDA FS 1999) in place for evaluating benefits, problems, risks, and tradeoffs of roads. On November 9, 2005, the United States Forest Service published the Final Travel Management Rule (70 Federal Register (Fed. Reg.) 216, November 9, 2005; p. 68264-68291), which required designation of roads and trails for motor vehicle use. Implementing travel management plans to meet this requirement should help reduce sediment from roads. Despite the size of the forest road network, road effects have been examined in only a few places, especially in the Appalachians, Pacific Northwest, and Rocky Mountains. Given the wide variability in road history, age, construction methods, and use patterns in relation to topography, climate, and social setting, the narrow geographical scope of these studies limits their extrapolation to other regions or their usefulness in addressing more subtle effects. In forests along the west side of the Sierra Nevada, major roads were built along broad ridges, with secondary roads leading down into headwater areas. In general, Sierra Nevada roads create less erosion and landslides when compared to roads in western Oregon forests that usually entered watersheds along narrow stream bottoms and then climbed the adjacent steep, unstable hillslopes to access timber extending from ridge to valley floor. Road placement in the landscape, combined with local geology and climate, resulted in different effects of roads on watershed, vegetation, and disturbance processes in the western U.S. (Gucinski et al. 2001). A summary of points discussed by Gucinski et al. (2001) about road erosion effects is included here.

- Although mass erosion rates from roads typically are one to several orders of magnitude higher than from other land uses based on unit area, roads usually occupy a relatively small fraction of the landscape, so their combined effect on erosion may be more comparable to other activities, such as timber harvest.
- Roads interact directly with stream channels in several ways, depending on orientation to streams (parallel, orthogonal) and landscape position (valley bottom, midslope, ridge).
- The geomorphic consequences of these interactions, particularly during storms, are potentially significant for erosion rates, direct and off-site effects on channel morphology, and drainage network structure, but they are complex and often poorly understood.
- Encroachment of forest roads along the mainstem channel or floodplain may be the most direct effect of roads on channel morphology in many watersheds.
- Poorly designed channel crossings of roads and culverts designed to pass only water flow also may affect the morphology of small tributary streams, as well as limit or eliminate fish passage.
- Indirect effects of roads on channel morphology include the contributions of sediment and altered streamflow that can alter channel width, depth, local gradients, and habitat features (pools, riffles) for aquatic organisms.
- Extensive research has demonstrated that improved design, building, and maintenance of roads can reduce road-related surface erosion at the scale of individual road segments.

Although poorly constructed roads in the Sierra Nevada can cause soil erosion and increase sedimentation to water bodies, their effects have not been studied very much in the synthesis area.

Road impacts to water quality and aquatic habitat should be less in the future because very little new road construction is expected and knowledge exists about how to construct and maintain roads to lessen impacts. Larger sources of soil erosion may include increased wildfires (see Post-wildfire Debris Flows in this chapter), as well as lack of road maintenance that results in progressive degradation of road-drainage structures and functions (Furniss et al. 1991).

Goode et al. (2012) projected that climate change would increase sediment yield in semi-arid basins, primarily through changes in temperature and hydrology that promote vegetation disturbance (i.e., wildfire, insect/pathogen outbreak, and drought-related die-off). Although their case study took place in central Idaho, it is relevant for the Sierra Nevada because of similarities in conditions, such as coarse-textured, granitic soils and forests on steep mountain terrain. Istanbulluoglu et al. (2004) demonstrated that the mechanism driving higher long-term sediment yields in smaller catchments (<25 km) is rare, post-fire erosional events that are typically two orders of magnitude larger than the long-term average yields of 146 T km^{-2} yr^{-1} (determined from cosmogenic dating). Sediment yields from experimental basins with roads are on the order of 10^1 T km^{-2} yr^{-1}, whereas yields from individual fire-related events in this region are three orders of magnitude greater (10^4 T km^{-2} yr^{-1}). An experiment (Ketcheson et al. 1999) showed that for 1-2 km/km^2 of new road, the amount of sediment yield increased by 7 to 12 T km^{-2} yr^{-1} compared with 2.5 T km^{-2} yr^{-1} for the control basin (more than a doubling during the four study years). Goode et al. (2012) concluded that road maintenance and decommissioning are generally effective and beneficial for water quality, but will not mitigate an increase in sediment yields from increased wildfire frequency. They also highlighted the substantial uncertainty about the efficacy of post-fire treatments for vegetation and hillslope erosion in forest mountain basins (Robichaud et al. 2000) and the growing body of literature discouraging further interference in natural landscape disturbance processes because the dynamic response to such disturbances may help maintain more diverse ecosystems that are more resilient to changed climates (DellaSala et al. 2004). Therefore, work to reduce the magnitude and frequency of wildfire is likely important to influence total sediment yields from forests in the Sierra Nevada drainage basins.

Sidebar: Kings River Experimental Watersheds

The Kings River Experimental Watersheds (KREW) is a watershed-level, integrated ecosystem project for headwater streams in the Sierra Nevada (www.fs.fed.us/psw/topics/water/kingsriver). Eight watersheds at two study sites are fully instrumented to monitor ecosystem changes. The KREW project was designed to address several of the information gaps for water resources and aquatic ecosystems included in the Monitoring and Adaptive Management Plan for the Sierra Nevada (Appendix E, Sierra Nevada Forest Plan Amendment 2001, 2004). A few examples of these questions are.

- What is the effect of fire and fuels reduction treatments (i.e., thinning of trees) on the physical, chemical, and biological conditions of riparian areas and streams?

- Does the use of prescribed fire increase or decrease the rate of soil erosion (long term versus short term) and affect soil health and productivity?

• How adequate and effective are current stream buffers (areas on both sides of a stream with restricted uses) at protecting aquatic ecosystems?

Prior to 2000, when KREW was designed, there was no long-term experimental watershed study in the southern Sierra Nevada to guide future land management activities. KREW has a site in the rain-snow zone and a site in the snow-dominated zone of Sierran mixed conifer. Data have been gathered for a 9-year pre-treatment period (Hunsaker and Eagan 2003, Hunsaker et al. 2007, Hunsaker et al. 2012, Hunsaker and Neary 2012, Liu et al. 2012, Johnson et al. 2011a, Brown et al. 2008). Tree thinning was completed in 2012, and prescribed underburns are planned for 2013 and 2014; the experimental design will allow partitioning of effects between thin only, underburn only, and the preferred treatment of thin and burn.

This research is evaluating the integrated condition of the streams and their associated watersheds (i.e., physical, chemical, and biological characteristics).

• Physical measurements include upland erosion, turbidity (suspended sediment), stream temperature, streamflow, channel characteristics, and weather conditions.

• Chemical measurements for stream water, shallow soil water, precipitation, and snowmelt include nitrate, ammonium, and phosphate (primary biological nutrients); chloride; sulfate; calcium; magnesium; potassium; sodium; pH; and electrical conductivity.

• Biological measurements include stream invertebrates (like dragonflies and mayflies), algae, and riparian and upland vegetation (herbs, shrubs, and trees). Yosemite toads are also being studied at the Bull Creek site.

Unique aspects of KREW include the following:

- An integrated design of physical, chemical, and biological components being measured at the same locations and at several spatial scales.
- A control watershed that can provide the "natural range" of variability (no roads or timber harvesting).
- A designed comparison of fuels and vegetation for riparian and upland parts of the watersheds.
- Use of a long-term data set for stream invertebrates to evaluate water quality effects after management treatments at a designed BACI experiment.
- A comparison of adult Yosemite toad (*Anaxyrus (=Bufo) canorus*) movement before and during treatments.
- A comparison of rain-snow and snow-dominated watershed function before and after land disturbances.

Nutrients and chemistry

Water chemistry is of interest for various reasons: human health, aquatic ecosystem condition, and agricultural and industrial uses downstream of the forests. Although the chemistry of water is usually very good within national forests, it is prudent to monitor some characteristics. The EPA and State of California use chloride, specific conductance, and total nitrogen and phosphorus as indicators of stress in perennial, wadeable streams (Ode 2007). Additional information on chemistry processes in forests can be found in the Soils (5.0) and Air Quality (8.0) chapters of this report. This discussion focuses on nitrogen (N), which is a necessary nutrient for vegetation, but in high enough amounts can be a substantial stressor on both terrestrial and aquatic ecosystems. N is an important indicator of the overall health of a forest, and knowing its concentration over time in atmospheric deposition, vegetation, soils, and stream water provides a useful assessment tool (see Soils chapter (5.0)). Usually Sierra Nevada stream water has very low N concentrations, almost at detection limits (Hunsaker et al. 2007, Engle et al. 2008). However, atmospheric deposition (as discussed in the Air Quality chapter (8.0)) is high and moderately high in the southern and central Sierra Nevada, respectively. Nitrogen leaching from soils to water, which can lead to acid conditions and a fine root biomass loss of 26 percent, are expected at N deposition levels of 17 kg N ha^{-1} yr^{-1} (Fenn et al. 2008). High N deposition, coupled with recent findings of high N content in mineral soils and water flowing thorough forest floor litter in both the western and eastern Sierra Nevada (Johnson et al. 2010, 2011a, 2011b; Miller et al. 2005), are reasons to have long-term measurements on N partitioning in the forest ecosystem.

The long-term (decade-scale) effects of fire on watershed chemical balances relate to changes in vegetative cover and include nitrogen (N) fixation and the accumulation of elements in aggrading plant biomass (Johnson et al. 2005). Tree mortality, crown scorch, and stand replacement can affect canopy-related processes that are important in watershed balances of water and nutrients, such as interception of precipitation and cloud water, scavenging of aerosols and gases, and transpiration. The changes in nutrient budgets expected soon after fire (first few years) are a result of many processes, as listed by Engle et al. (2008): wind- and water-driven sediment export, changes in the physical properties of soil, dissolution of ash, shifts in soil water pH, changes in microbial biomass and activity, increased decomposition, and changes in biological demand for water and nutrients.

Johnson et al. (1998) discussed that fire and post-fire N fixation are more important than atmospheric deposition and leaching for N fluxes in most semi-arid forests of the southwestern United States. Exceptions may occur in areas with high atmospheric inputs of N due to local air pollution. They argued that existing literature shows that the nutrient cycling paradigm established for humid forest ecosystems, which emphasizes fluxes into and out of the ecosystem by water, needs modification for semi-arid forests. Odum et al. (1994) proposed that a realistic paradigm for natural systems is one in which the system is subjected to regular pulses of N from processes such as fire rather than the concept of steady-state and one-dimensional, vertical nutrient cycling. Nitrogen cycling studies in semi-arid forests require a long-term, landscape-scale perspective that encompasses episodic fire and periods of intensive post-fire N fixation. Johnson et al. (1998) concluded that the frequency of fire and the occurrence and duration of post-fire N fixation are crucial factors that determine the long-term productivity of semi-arid forest ecosystems and require more study; data sets comparing N fluxes via fire and water at the same site are very rare.

Fire has both short- and long-term effects on nutrient availability and cycling in forest ecosystems. Because of its low volatilization temperature (200°C), nearly all N in burned biomass is lost in gaseous form, and N losses are disproportionately large compared to carbon (C) losses. Fire can also result in the losses of other nutrients, including sulfur and phosphorus, by volatilization, though to a lesser extent than for N (Raison et al. 1985, 1990). The mean and median values for N losses during wildfire and prescribed fire are 360 and 280 kg ha^{-1}, respectively, which equal approximately 500 to 12,000 years leaching loss of N from semi-arid forests (measured rates 0.1 to 0.6 kg ha^{-1} yr^{-1}). Data exist for the east side of the Sierra (Little Valley, Nevada) to compare fluxes of N via deposition and leaching versus wildfire (Johnson et al. 1997), and they show that at a 100-year interval, wildfire was the dominant factor in long-term N losses, exceeding leaching losses by more than two orders of magnitude (3 to 6 compared to 0.03 kg ha^{-1} yr^{-1}).

Some combination of restoring natural fire frequency, vegetation conditions, and/or fuel loading to landscapes is usually the goal of prescription burning and forest restoration, thus it is important to understand how watershed balances respond to fire on timescales that match target fire return intervals (FRIs). Tree ring studies indicate that from 1700 to 1900, natural FRIs in the region of Sequoia National Park averaged 10 to 20 years (Engle et al. 2008); this is an accepted FRI for mixed-conifer forests in the southern Sierra Nevada.

Engle et al. (2008) provided the only new long-term research on stream chemistry before and after prescribed fire in the Sierra Nevada. Their research is for a 16-year paired catchment study in sequoia-mixed-conifer forest, Sequoia National Park. Seven years of pre-fire chemistry data were compared to nine years of post-fire chemistry data for two adjacent headwater streams—the intermittent Tharps Creek (13 ha) and the perennial Log Creek (50 ha) in the rain-snow zone of the southern Sierra Nevada. This study provides an excellent opportunity for the comparison of water chemistry data collected before and after thinning and burning treatments at KREW; these two long-term research projects should provide the necessary information for managers to understand small stream response and recovery processes to forest restoration practices in the southern Sierra. Measurement of ecosystem outputs after fire using gauged streamflow is rare (Engle et al. 2008).

Inorganic N was elevated in stream water for three years after fire in the Tharps Creek watershed. Increased export of water, SO_4^{-2}, Cl^-, SiO_2, and base cations continued through the end of the study. This loss was calculated to be less than one percent of the N, up to one-third of the Ca and Mg, and up to three-fourths of the K contained in the forest floor prior to combustion. Changes in watershed balances indicated that low-end natural FRIs may prevent complete re-accumulation of several elements between fires. However, this result needs to be considered in the context of the high fuel loads that had built up over 120 years. Unfortunately, we don’t have data that relate nutrient losses to a more historical or natural fuel loads that would have built up under an FRI of 10 to 20 years.

Spatial variability of nutrients on the landscape

Understanding soil nutrient hot spots is important for water quality and plant nutrition. "Hot spots" are areas (or patches) that show disproportionately high reaction rates relative to the surrounding soil area (or matrix). In semiarid soils, these patches have long been recognized where "islands of fertility" occur near widely spaced shrubs or patches of vegetation (Johnson et al. 2011b). Schimel and Bennett (2004) highlighted the importance of hotspots as sources of nutrients to plant roots, which are otherwise outcompeted by microbes for nutrients in the rest of the soil matrix. A review of data sets for forests in the eastern Sierra Nevada mountains showed N hotspots in soils, resin lysimeters, and resin capsules; other measured nutrients (extractable P, Mg^{2+}, K^{+}, SO_4^{2-}, and Ca^{2+}) also showed positive skew and outliers, but less so than N (Johnson et al. 2010). A recent study at KREW on the western side of the Sierra showed that nutrient hotspots occur in mixed-conifer forests for nearly all measured nutrients; these nutrients were measured using soil cores, resin collectors, resin probes, and resin capsules in 6 x 6-m plots (Johnson et al. 2011b).

Recent literature shows that the lack of rooting in the O horizon of semiarid forests (due to extreme summer drought) results in a spatial decoupling of the processes of decomposition/nutrient mineralization and vegetation uptake, and a lack of the intense competition for N between roots and decomposers (Johnson et al. 2011). Because of this vertical decoupling, nutrients released during decomposition in O horizons are not immediately taken up and are solubilized by rain or snowmelt, creating solutions with very high inorganic N and P concentrations, which presumably infiltrate the soil at preferential flow paths and contribute to hotspots and possible leaching to surface water (Miller et al. 2005). Nutrient-enriched O horizon interflow has been quantified in both the eastern (Miller et al. 2005) and western (Johnson et al. 2011b) Sierra Nevada. At KREW, values for ammonium-N ranged from less than 0.1 to 456 µmol L^{-1} (<0.1-6.3 mg N L^{-1}), NO_3-N concentrations ranged from less than 0.1 to 622 µmol L^{-1} (<0.1-8.8 mg N L^{-1}), and ortho-P concentrations ranged from less than 0.1 to 98 µmol L^{-1} (<0.1-3.1 mg P L^{-1}), values that exceed those found in soil solutions and streamwaters at these sites by 10 to 100 fold (Hunsaker et al. 2007).

Stream benthic macro-invertebrates

This review examines studies where the ecology of aquatic invertebrates in streams has been used to monitor and/or evaluate management actions, stressors from different kinds of disturbance, and natural processes in stream ecosystems. Informed decision making in the Forest Service or other agencies is founded on reliable science, so the studies reported here emphasize practical applications for management planning and design. The studies cited in this section come exclusively from the Sierra Nevada, have not been covered in previous summaries (such as Erman

1996), and include only those with some invertebrate data component.

Development of biomonitoring tools and bioassessment programs in California

Use of stream invertebrates as biological indicators has become one of the most common water quality tools of regulatory agencies (Rosenberg and Resh 1993, Allan 1995). Invertebrates are especially useful indicators in small, wadeable streams in the Sierra Nevada, and in headwaters and fishless or intermittent streams. These organisms can be used as sentinels to show how much the ecological integrity of watersheds is changing, and how effective management may be in protecting these natural resource values. Adaptive management requires monitoring tools for tracking the progress of desired outcomes. Bioassessment sampling by California's Surface Water Ambient Monitoring Program (SWAMP) provides extensive new data on community composition and indicators of environmental quality. The database, which is under construction, compiles surveys from portions of the Sierra, and these will be used to develop quantitative numeric biological objectives for use in water quality monitoring programs of the State of California (Ode and Schiff 2009). The Southwest Association of Freshwater Invertebrate Taxonomists has summarized various regional taxonomic updates and species descriptions from the broader region.[3] Online documents provide lists by state, taxonomy resources, compilation of tolerance values, and functional feeding groups as sources for bioassessment, but those documents cannot at present be compiled just for the Sierra (though the SWAMP database can). Benthic macroinvertebrates have been adopted as "management indicator species" (MIS) for monitoring of status, trend, and health in Region 5, US Forest Service. (Management indicator species are chosen either due to their rarity, their restriction to a particular unique habitat, or because they are more common and widespread. The health of their populations is supposed to provide an "indicator" of each habitat's health in response to management activities (USDA 2001, 2004).) The MIS program has been collecting bioassessment survey data for several years from streams in the Sierra using probabilistic sampling (Joseph Furnish, coordinator).

Hawkins et al. (2000) provided the foundation for developing predictive models (RIVPACS) for assessing stream health in the Sierra Nevada and Klamath Mountains, and those models are now being used in SWAMP to establish biological objectives for streams across California. Results suggest logged sites had subtle losses of diversity (10 percent) compared to references areas (<5 percent basin logged), with losses related mostly to reduced riparian cover rather than amount of area logged or number of roads (logging intensity and type were unfortunately not specified in the analysis).

Herbst (2004) provided an overview of stream survey work done in the Sierra through 2002, including initial steps in identifying reference stream standards, monitoring of grazing and mining practices, and gaps in understanding of stream invertebrate ecology. An eastern Sierra Nevada multimetric index (IBI) and predictive models (RIVPACS) show how differing methods and analytic tools are robust in giving the same assessments of loss of stream health related to channel modifications or livestock grazing (Herbst and Silldorff 2006).

[3] www.safit.org

Natural patterns of variation in stream communities in space and time

Carter and Fend (2001) found that differences in the richness of invertebrates in riffle and pools appear to depend on annual discharge regime, and are more pronounced during low discharge years and disappear when flow is higher. This study, which took place on the Merced River in the Yosemite Valley, suggests that differences in erosional and depositional features between riffles and pools diminish when flows increase, so communities become more similar. An implication of this is that it is possible that flow regulation may influence the natural variations in habitat-based diversity and that channelization (eliminating riffle-pool geomorphology) may also produce less diverse assemblages of aquatic life and more limited ecosystem processes (such as nutrient recycling, productivity, organic matter transport, conversion, and decomposition). Beche and Resh (2007) found that traits related to adaptations to the environment vary in response to gradients of flow between years, from dry (drought) to wet (above average) conditions. Traits that provide adaptation to drying (e.g., desiccation resistance, aerial respiration) were more common in drought years, whereas traits permitting survival during high flows (e.g., flat body shape, drift dispersal) were more common in wet years. Prolonged drought or wet conditions result in shifts in trait composition. Despite this, there is redundancy in traits among taxa, so taxa can be replaced without loss in represented trait diversity. This suggests that it may be more difficult to conserve species diversity than trait diversity in the face of changing climate regime.

Erman (2002) studied the invertebrates of spring and springbrook (outflows) communities over 20 years to describe the biota and physical/chemical properties of Sierra Nevada cold springs. Results showed the individualistic nature of springs even within the same stream basin. Spring invertebrate assemblages differed greatly from one spring to another, as did timing of insect emergence and abundance of species. Invertebrate species richness was greater in deeper, more permanent springs, which were distinguished by high concentrations of dissolved ions, especially calcium. Spring permanence was also determined by direct observations over time, measurement of discharge variability, correlation of discharge with ionic concentration, and water dating. This study shows the high conservation value of spring habitats and the high levels of diversity that can serve as a biodiversity refugium in cold-water environments.

Fire effects

Beche et al. (2005) found minimal effects from prescribed fire on stream invertebrates in a Sierra Nevada study. Prescribed fire altered BMI community composition only within the first weeks post-fire, but there were no lasting (1 year) impacts on BMIs. Densities and percentage of sensitive taxa were significantly reduced after an intense wildfire on Angora Creek in the Lake Tahoe Basin, but there were no consistent changes in taxonomic richness or diversity (Oliver et al. 2012). Canopy cover and bank stability declined dramatically following the wildfire and substrate also changed substantially, with fine sediment more abundant and cobble less abundant post-fire. There were large reductions in relative abundances of shredder and scraper taxa, whereas collector-gatherer abundances increased. Community composition shifted away from pre-fire configurations, and continued to diverge in the second year following the fire. Scores from a regionally derived index of biotic integrity (IBI) were variable, but overall they were much lower in post-fire samples and did not show recovery after 2 years. This study demonstrated substantial post-fire effects to aquatic ecosystems even in the absence of large flooding or scouring events, and it showed that these effects can be transmitted downstream into unburned reaches. These findings, when compared to those from Beche et al. (2005), suggest that fire effects are strongly related to fire intensity.

Forest management practices

Although stream invertebrates have been an MIS for the Forest Service, little published information exists regarding effects of mechanical forest management practices (road building and maintenance, tree thinning and commercial harvesting, tractor piling of slash and burning) on stream invertebrates in the Sierra. A few publications exist on prescribed fire effects on stream invertebrates (see previous discussion). The usefulness of stream invertebrates for monitoring aquatic ecosystem condition and associated information gaps were recognized in the Adaptive Management Plan, Appendix E, of the Sierra Nevada Framework (USDA 2001, 2004), and one new research experiment exists (see sidebar on KREW above). McGurk and Fong (1995) found there was reduced diversity and increased dominance (most common taxa increase in fraction represented) in stream invertebrate communities in the Sierra Nevada when equivalent roaded area exceeded 5 percent (equivalent roaded area is an expression of landscape disturbance from roads and combined developed areas and logging).

Flow regulation and impoundments

Aquatic organisms have evolved life history strategies to take advantage of high flood predictability and associated seasonal processes. The timing of the spring snowmelt recession and the shape of the recession hydrograph contribute to reproductive cues for many riparian and aquatic species, such as cottonwoods, willows, mayflies, amphibians, and salmonids. Yarnell et al. (2010) developed a conceptual model about snowmelt recession that provides some testable hypotheses about regulated flows in streams and climate change effects on the hydrograph. As flows gradually decrease through spring, the hydrograph passes through windows of biological opportunity at magnitudes that support habitat (i.e., availability) in sufficient condition (i.e., suitability) for species persistence. Shifts in the timing of the recession or changes to the shape of the recession hydrograph that preclude suitable habitat during a particular species' window of reproduction can lead to a lack of success. Shifts in the timing of the recession may push periods of reproduction out of phase with the availability of suitable habitat. Shifts in the rate of the recession affect both abiotic and biotic conditions, creating the largest observed changes to the stream ecosystem. The effects of climate warming on aquatic ecosystems in Mediterranean-montane climates will be profound, with shifts in each of the three primary components of the recession (magnitude, timing, and rate of change). Shifts in the timing at the start of the recession and decreases in the magnitude of the flow, coupled with a shorter duration resulting from a relatively small increase in the rate of change, will alter in-stream and riparian species compositions, forcing cold-water aquatic species to inhabit higher elevations, and leading to a higher abundance of non-native species. Shifts in the spring recession as a result of flow regulation can create similar patterns. On the basis of this conceptual model, the authors found that managed hydrographs with a flashy, short-duration spring snowmelt recession overlying a steady base flow can create channel conditions reflective of the two observed extremes in discharge, flood, and base flow. Aquatic and riparian species will be reflective of the homogeneous channel conditions and lack diversity. Rehn (2008) used a reference stream dataset to establish a multimetric IBI for Sierra westslope streams to evaluate the effects of hydropower releases on benthic macroinvertebrates. Degradation of the invertebrate community (quantified through comparison with upstream sites) was found within three km of dams and was mostly related to flow regulation and constancy below dams (i.e., loss of natural flow regime). Rehn's findings support the conceptual model and hypotheses in Yarnell et al. (2010).

Effects of floods

Herbst and Cooper (2010) evaluated conditions before and after the 1997 New Year's Day floods for 14 small, headwater streams in the eastern Sierra. The streams showed loss of bank stability and riparian cover due to scour. Densities of BMIs in previously disturbed habitats increased, with increases mainly consisting of small opportunistic species (rapid-growing colonizers) feeding on fine particulate organic matter. These results show a shift to common taxa with generalized food habits. Undisturbed reference streams changed little from 1996 to 1997, suggesting that these more diverse and stable communities persist even in the face of short-term catastrophic flows, and are important biodiversity refugia for downstream habitats where flooding may be more severe. These results highlight the importance of protecting the integrity of less disturbed headwater stream habitats.

Introduced invasive species

A paired watershed study of fishless streams with adjacent trout-stocked streams in Yosemite National Park showed that trout reduce native grazers and permit more dense growth of algae on stream rocks (Herbst et al. 2008). This study showed losses of 20 percent of BMI taxa richness, mostly as losses of endemics and native montane species, in the presence of trout. The higher algae cover in streams with non-native trout corresponds to more collector-gatherers and fewer predators and grazers. Conserving biodiversity and restoring natural foodwebs likely depends on removal of introduced trout.

The New Zealand mud snail (NZMS) has caused significant disruptions in stream food chains across many trout streams of the western United States. Herbst et al. (2008) suggest that specific conductance levels may control which streams NZMS can colonize. In streams with specific conductance below 50 uS, snails do not survive, and at levels below 200 uS, their growth and survival are inhibited. Invasive herbivores like NZMS can have strong top-down and bottom-up influences on invaded ecosystems, but these impacts can be extremely variable across time and space.

Information gaps

The Sierra Nevada Ecosystem Project (SNEP) provided a summary on the status of invertebrates, highlighting high endemism (among caddisflies and stoneflies in particular) and dependence of diversity on habitat quality, but it provided little information on ecological structure, function, and ecosystem processes in streams (Erman 1996). The distribution and abundance of aquatic invertebrates in the Sierra is still mostly unknown. Many studies are localized at research areas (see Figure 1, Chapter 1.4), such as the Sierra Nevada Aquatic Research Laboratory (SNARL) on Convict Creek, Sagehen Creek Experimental Forest, the Blodgett Forest Research Station, and Kings River Experimental Watersheds, or where restoration project monitoring has occurred. Recent sampling done under the California Surface Water Ambient Monitoring Program (SWAMP) provides more survey data, but important geographic gaps remain. High elevations, intermittent streams, springs, remote regions, and whole catchments remain poorly characterized (most surveys represent only 100-200 m reaches within larger basins). Data from some studies show an emerging pattern of north-south distinctions in biogeography.

We still do not know a lot about the biodiversity of aquatic invertebrates in the Sierra Nevada. Erman (1996) reported that species-level information was lacking for many taxa and that inventories or lists were incomplete; this statement remains true. Erman reported approximately 400 taxa from streams and lakes, approximately 20 percent of which were endemic to the Sierra. From surveys in eastern Sierra streams alone, the database developed at SNARL has in excess of 500 distinct taxa from about 200 stream surveys.[4] More comprehensive listings of taxa from high-elevation streams in the western Sierra Nevada, and at species-level resolution (many are only at genus level currently), will likely place this total closer to 1,000 species or more. Reference specimen collection cataloguing is underway at SNARL for hundreds of surveys in the Sierra Nevada and Great Basin. Hidden biodiversity exists in the genetics of distinct metapopulation variants. With the advent of "bar-coding" DNA technology, genetic diversity and inter-population distinctions will likely be found. Studies of the large perlid stonefly *Doroneuria baumanni* (the most common large insect predator in the high Sierra Nevada) show genetic variation in isolated Sierra Nevada and Great Basin populations of this native predator (Schultheis et al. 2012).

Strategies to Promote Resilience of Water and Aquatic Ecosystems

Promoting Favorable Water Flows

Given changing climate conditions, maintaining or improving water quantity and quality in low-order streams through fuels reduction activities (mechanical thinning and prescribed fire) can be considered as a management opportunity rather than a constraint. For water quality, this approach requires a balancing of short-term, low-intensity effects against high-intensity effects from more catastrophic events like wildfire. For example, wildfire frequency is increasing with climate change. Since fuels reduction activities are expected to reduce the risk of wildfires, they can be considered as an opportunity to maintain or improve water quality because wildfires can have significant impacts through increased sediment loads and phosphorus concentrations and debris flows. Also, with warming temperatures, trees are expected to decrease soil moisture and increase evapotranspiration, thus leaving less water for movement to streams; mechanical thinning of trees and low-intensity underburning of vegetation would reduce evapotranspiration and help maintain soil and stream water amounts. The strategic orientation of GTR-220 and 237, which focuses on restoring heterogeneity and landscape-scale ecological processes, can be extended to aquatic ecosystems. The incorporation of key hydrologic and nutrient processes as treatment objectives facilitates a more holistic forest restoration effort.

Because nitrogen concentrations are a good indicator of forest health (both productivity and stress), it would be beneficial to have a few locations where nitrogen is measured periodically in wet and dry atmospheric deposition, mineral soil, and soil and stream waters. Such measurements are most likely to

[4] http://vesr.ucnrs.org/pages/Herbst_Research.html

be done at research locations (Figure 1, Chapter 1.4), and they could continue through collaborative activities between research and management.

The scientific literature indicates that increased wildfire frequency is the future area of concern for soil erosion, sedimentation, and impacts to water resources (Goode et al. 2012). To mitigate potential impacts to water bodies, it is important to evaluate overall watershed potential for erosion and sedimentation due to wildfires, as well as fuels reduction activities; timber harvest; and road construction, use, improvement, and decommissioning. Some research exists and can be augmented with monitoring data from future forest management projects.

Addressing roads

Although the ability to measure or predict the hydrologic consequence of building or modifying a specific road network might be limited, general principles and models are available to decrease the negative effects of roads. These principles can be useful during upgrading or decommissioning of roads to meet various objectives. A list of principles provided by Gucinski et al. (2001) includes the following:

- Locate roads to minimize effects by conducting careful geologic examination of all proposed road locations.
- Design roads to minimize interception, concentration, and diversion potential, including measures to reintroduce intercepted water back into slow (subsurface) pathways by using outsloping and drainage structures rather than attempting to concentrate and move water directly to channels.
- Evaluate and eliminate diversion potential at stream crossings.
- Design road-stream crossings to pass not just water but also woody debris, sediment, and fish.

Assessing water quality using stream macroinvertebrates

There is much work still to be done to characterize the health of Sierra Nevada streams using stream invertebrates. Mapping invertebrate distributions would provide a basic understanding of biodiversity patterns, hot spots, and biogeographic regionalization of aquatic invertebrate fauna. Future planning efforts will benefit when forest-wide analyses and multi-forest syntheses can be done on stream invertebrate data and stream physical properties data (Stream Condition Inventory) that already exist for the synthesis area.

Spatial analysis of watersheds, from headwaters to major river systems, would provide information on how the ecological health of interconnected stream systems changes as a function of land use disturbances, habitat fragmentation, and reservoirs/dams. Combining data from SWAMP, the region-wide Management Indicator Species Program, PSW, and other organizations like SNARL could yield valuable information to improve assessments of cumulative watershed affects. Furthermore, distribution data on fish and amphibians could be combined with the accumulating data on aquatic invertebrates to assist in identifying areas of high biological value (richness, endemism, index of biological integrity) for conservation management to provide a means for identifying and protecting reference areas of biological integrity. Knowledge about forest stream condition (as shown by Ode 2007) can improve substantially if invertebrate monitoring occurs at selected management projects that have not yet proceeded to an implementation phase. It is possible to use bioassessment tools to gather data using a before-after and control-implementation (BACI) statistical design to evaluate project outcomes in terms of aquatic invertebrate indicators of desired ecological improvements. These case histories could provide a foundation for adaptive management (monitoring informs decisions on how to proceed with actions) and advance restoration as a prescriptive science (what works and where and how long it takes) (see KREW sidebar). Opportunities exist at the Sierra Nevada Adaptive Management Project, the Dinkey Collaborative Project on the Sierra National Forest, and for similar projects where planned management affords an opportunity to learn more about prescribed fire, selective logging practices, or fuels reduction in protecting aquatic ecosystems within the context of forest ecosystem health.

Promoting Resilience of Aquatic Ecosystems

Landscape-scale consideration of tradeoffs in managing forests for wildfires

Although wildfire can have negative or neutral impacts on fish, wildfire-related disturbances can also help to maintain diverse and productive habitats (Rieman et al. 2003). Many of the processes associated with wildfire disturbance have potential to benefit aquatic habitats, including contributions of nutrients, wood, and coarse substrate; reorganization of in-channel habitat structure; increases in streamflow; and increases in temperature, light, and in-stream food production (in systems that are below optimum growth levels) (Gresswell 1999). The overall impact of a wildfire on aquatic organisms depends on the specific context of that event, however, scientists have come to a general conclusion that fishes in large habitat networks are more likely to benefit even after relatively severe wildfires, whereas fishes in small, isolated systems are more vulnerable to losses (Rieman et al. 2010). A recent synthesis of short-term effects of wildfire on amphibians yielded similar conclusions, specifically that 1) wildfire can provide important benefits for amphibian diversity overall; 2) wildfire can pose threats to small, isolated, or stressed populations, particularly in the Southwest; and 3) negative effects on populations or individuals are greater in fire-suppressed forests, and high-severity burns cause greater negative effects on populations or individuals (Hossack and Pilliod 2011). In addition, some research suggests that native fishes may be better adapted to fire-associated disturbances than non-native competitors. This relationship, combined with the potential for wildfire to extirpate or greatly reduce non-native trout species, suggests that wildfires, even uncharacteristically severe ones, could provide important opportunities to enhance native species.

Forests may be treated to reduce the threat of uncharacteristically severe wildfire or to emulate some of the desired effects of natural disturbances, consistent with the principle of disturbance-based management (see Integrative Approaches chapter (1.1)). Burton (2005) and Rieman et al. (2003) emphasized the need to carefully weigh the costs and benefits of such treatments, because road networks and stream crossings that may be used to implement the treatments have potential to perpetuate impacts to streams and aquatic populations. Collectively, these studies recommend spatially explicit analysis of risks for aquatic species in the synthesis area.

Stream and riparian restoration to promote resilience to climate change

Restoration of stream and riparian ecosystems is a core strategy for enhancing ecological resilience to post-fire impacts and climate change (see Introduction chapter (1.0)), due to the important roles of streams in providing linear habitat connectivity, laterally connecting aquatic and terrestrial ecosystems, and creating thermal refugia for cold water species such as salmonids (Seavy et al. 2009).

Promoting connectivity

Increasing longitudinally connected networks from mainstem rivers to headwater tributaries has been recommended to help native trout species cope with the threat of wildfire under projected climate change (Haak and Williams 2012). However, because many remaining native fish populations have been purposefully isolated from non-native invaders, efforts to reconnect isolated populations could leave populations exposed to potential invaders (Fausch et al. 2009). For that reason, Williams et al. (2009) observed that a shift away from an isolation strategy would require increasing efforts to reduce non-native fishes.

Ameliorating high temperatures

An analysis by Wenger et al. (2011) suggested that proactive trout conservation strategies in the face of climate change should include ameliorating high temperatures along with reducing interactions with non-native species. Stream restoration has potential to ameliorate increases in stream temperatures, reductions in base flows, and other projected effects of climate change. Restoration efforts can promote vegetation growth and channel narrowing that reduce solar exposure, and they can also promote channel complexity and associated hyporheic exchange (where surface water mixes with shallow groundwater) by developing riffles, secondary channels, and floodplain sediments (Hester and Gooseff 2010, Kondolf 2012, Poole and Berman 2001). As an example of this strategy, researchers are investigating how wet meadow restoration could increase summer discharge and reduce water temperatures to help sustain California golden trout (*Oncorhynchus mykiss aguabonita*) (see Wet Meadows chapter (6.3)).

Restoring flow regimes on regulated rivers

Management of reservoirs and regulated rivers were not a focus of this synthesis, although these systems are clearly important as a source of ecosystem services in terms of recreation opportunities, flood control, water supply, and power generation (Null et al. 2010). Recent research demonstrates that the cross-cutting theme of disturbance-based management relates strategies for river management to conservation of endemic species. A recent study examined time series data for the foothill yellow-legged frog (*Rana boylii*) and California red-legged frogs (*R. draytonii*); the findings suggest that flow management that emulates natural flow regimes is likely to promote resilience of populations of these native frogs (Kupferberg et al. 2012). Dams generally reduce overall heterogeneity in flow regime, while also permitting unnaturally rapid changes in flow, such as sharp decreases in flow following spring runoff (Moyle and Mount 2007, Kupferberg et al. 2012). Climate change may also alter hydrologic regimes in ways that are similarly detrimental to these aquatic species by reducing snow packs, inducing earlier and more rapid snowmelt in the spring, and by extending periods of low flow in the summer and fall (Null et al. 2010). Those changes may harm species that are adapted to gradual spring recession flows (Yarnell et al. 2010), and also reduce whitewater boating opportunities. Kupferberg et al. (2012) reported a negative association between hydrologic modification, as suggested by dam height, and persistence of foothill yellow-legged frog populations. In regulated rivers, a management strategy to protect native riverine species is to emulate natural flow patterns, especially by limiting rapid fluctuations in water levels; this strategy could benefit the species directly, by helping them avoid mortality, and perhaps indirectly, by promoting changes in channel morphology and in-stream habitat (Yarnell et al. 2012). Another component of a management strategy to benefit these species could be meadow restoration efforts, to the extent that they can extend baseflows longer into the summer (see chapter on Wet Meadows (6.3)).

Restoring fluvial processes

Researchers have emphasized the fundamental importance of restoring fluvial processes in dynamic stream systems (for example, by removing levees or other artificial structures) as a strategy to restore fluvial form and aquatic habitats (Kondolf 2012, Kondolf et al. 2012). This approach shares close parallels with the idea of restoring fire as an ecological process, and as such, its success will depend on careful articulation of goals and understanding of the ecological and social context of the system. In particular, the more passive approach of setting aside a “zone of liberty,” where natural riverine processes of deposition and erosion can occur freely, is more likely to succeed in relatively large and powerful streams or rivers (Kondolf 2012).

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6.2 Forested Riparian Areas

Carolyn Hunsaker and Jonathan Long with a contribution from Malcolm North

Contents

Figure 1: Forested riparian area on the Stanislaus-Tuolumne Experimental Forest

Introduction

Riparian areas are important transition zones between terrestrial and aquatic ecosystems that can modulate effects from the watershed. This chapter uses the term *riparian area* broadly to describe the "stream-riparian corridor," which consists of the stream channel, adjacent floodplains, and the transitional upland fringe, as defined by Dwire et al. (2010).[1] Consideration of the potential effects of forest management activities on ecological functions of riparian areas is an important part of determining cumulative watershed effects. Dwire et al. (2010) synthesized the state of knowledge about the potential impacts of streamside and upland fuels management on riparian areas and found that most information was derived from studies on the effects of forest harvest or wildland fire. Although research about fire history in particular strongly suggests a need for treatments within many riparian areas, limited information about the effects and effectiveness of mechanical treatments and prescribed fire treatments currently limits guidance for managing these valuable riparian ecosystems. As a consequence, these systems present an important opportunity for research on riparian responses to treatments as well as to fires of different severities.

Fire History and Behavior in Riparian Areas

Riparian plant communities evolved within the ecological context of regional fire regimes. A broader review of fire and fuels in the synthesis area is provided in the Fire and Fuels chapter (4.1). Research in the Sierra Nevada suggests that riparian forests have higher fuel loads than adjacent uplands, and that on smaller and more incised streams, forested riparian areas have fire histories similar to adjacent uplands (van der Water and North 2010, 2011). Conducted at 36 sites in the northern Sierra Nevada (Lassen National Forest, Onion Creek Experimental Forest, and Lake Tahoe Basin), these studies developed dendrochronological fire records in adjacent riparian and upland areas across a variety of forest and stream conditions. They sampled first through fourth order streams, with a particular focus on first- and second-order streams. Riparian and upland fire return intervals (FRIs) were significantly different in only one quarter of the sites they sampled. They found that the historical seasonality of fire did not differ between riparian and upland areas; in both, fires typically occurred in late summer to early fall. Riparian FRIs ranged from 8.4 to 42.3 years. Fire return intervals were shorter in forests with a higher proportion (>23 percent) of pine species, sites east of the Sierra crest, lower elevation sites (<1944 m), and riparian zones bordering narrower, more incised streams (width/depth ratio <6.2).

A recent study of two fires in southern Oregon similarly reported that smaller headwater streams had characteristics similar to adjacent uplands (such as low composition of riparian deciduous hardwoods) that were associated with high riparian fire severity (Halofsky and Hibbs 2008). Research in dry inland forests of Oregon also showed that historical fire frequencies in riparian areas were comparable to those in adjacent uplands (the differences were not statistically significant), but high patchiness and mixed severity meant that many fires occurred only at a riparian plot or only in an upslope plot within a pair,

[1] Riparian areas have been defined in the planning rule by the Forest Service as "three-dimensional ecotones of interaction that include terrestrial and aquatic ecosystems that extend down into the groundwater, up above the canopy, outward across the floodplain, up the near-slopes that drain to the water, laterally into the terrestrial ecosystem, and along the water course at variable widths" (Office of the Federal Register, 2012: 1411).

but not at both (Olson and Agee 2005). In some areas, riparian fires may also be less frequent but more severe than those in uplands (Arno 1996, Everett et al. 2003). Aspect may be an important factor within landscape areas, as Everett et al. 2003 found that fire frequencies were more similar across site types on north-facing aspects (higher moisture and cooler temperatures) than on south-facing slopes. These studies demonstrate the wide variation in relationships between fire regimes across the riparian-upland interface.

Wildfire Effects on Riparian Areas

Kobziar and McBride (2006) studied the relationships between wildfire burn patterns, stream channel topography, and the short-term response of riparian vegetation to the Lookout Fire along two creeks in mixed-conifer forest in the northern Sierra Nevada (Plumas National Forest). The study streams were perennial (3 m wide) with 7.4 to 9.9 m-wide riparian corridors on their southern aspects. One stream burned at lower severity, with 53 percent of transects at low to moderate severity and 47 percent at moderate to high severity. In the other stream, 86 percent of transects burned at low to moderate severity and 14 percent burned at moderate to high severity. The entire riparian corridor burned only 14 to 26 percent of the time, and one-third of the study transects were not burned. The authors noted that wider floodplain terraces supported mountain alder, which has been shown to slow backing wildfires moving toward streams. That study found that post-fire seedling recruitment and sprouting allowed riparian vegetation to be resilient and maintain stream quality even following high-severity fire. Wildfire effects on streams and aquatic systems are discussed more in chapter 6.1 (Watersheds and Stream Ecosystems).

Influence of Stream Order

Distinctions between headwater streams and larger stream orders may be relevant for predicting fire effects and for disturbance-based management. The definition of headwater streams often varies, although first through third order may be a reasonable division for parts of the synthesis area. For example, streams of those orders often have very narrow riparian areas (1-3 m on a side in the Kings River Experimental Watersheds (KREW)) that have a unique plant community from the adjacent uplands (Dolanc and Hunsaker 2007). These distinctions may have an influence on management plans, since first- to third-order streams represent approximately 90 percent of all streams in the continental U.S. (Leopold et al. 1964). Agreement on delineation rules and verification of stream order and flow regime in the field is necessary to determine the extent of different stream types and direct management to protect water quality and aquatic habitats (Hansen 2001). Streams at the fourth-order size up to large rivers usually support wider riparian areas and create a larger, moister microclimate; these downstream riparian areas likely impeded some fires from burning all or some of their vegetation or crossing their stream channels.

Microclimate Effects

Riparian areas are supported by a moister, three-dimensional air and soil microclimate as compared with adjacent uplands. Rambo and North (2009) compared microclimate (air temperature and humidity) gradients in trees from near the forest floor up through the canopy for both upland and riparian-influenced forest trees (three trees for each landscape type). The study area was in the Teakettle

Experimental Forest in old-growth mixed-conifer forest, that received one of three treatments, none, understory or overstory thinning. Measurements were made at 5, 15, 25, 35, and 45 m above the forest floor. Riparian microclimate had significantly lower minimums and means, and greater daily ranges of temperatures and humidity. The largest temperature and humidity ranges were near the stream and forest floor. In summer, steep slopes cause drainages to be warmer than ridge tops from upslope winds in daytime and cooler at night due to downslope flow of cold air from surrounding higher terrain. Accumulation of cold air at night can result in a local temperature inversion in drainages. This phenomenon acts in conjunction with stream influence, which directly cools air temperature and indirectly supplies water for daytime evaporative cooling via plant transpiration. In another study assessing changes in microclimate conditions both vertically and horizontally from the Teakettle creek, Rambo and North (2008) found a very narrow area around the stream (< 5.0 m vertically and < 7.5 m horizontally) in which microclimate conditions differed from upland.

Recent Research on Management in Riparian Areas

Prescribed Burning

Beche et al. (2005) published one of the few studies that focused on effects of prescribed fire in a Sierra Nevada riparian area. They examined prescribed fire effects in a mixed-conifer forest of the northern Sierra Nevada by comparing characteristics of the stream and its riparian zone in the burned watershed with those of five unburned watersheds (first- and second-order streams of low gradient). Effects were measured immediately and up to one year after the fire and compared with conditions one to seven years pre-fire. They concluded that the prescribed fire either had no or short-lasting (≤1 year) impacts on the stream and its riparian zone. The prescribed fire in the riparian zone was patchy in terms of intensity, consumption, and severity; it consumed 79 percent of pre-fire fuels, 34 percent of total surface fuels, and 90 percent of total ground fuels. The prescribed fire significantly reduced percent cover of surface vegetation and plant taxa richness in comparison with unburned sites, but not plant diversity (Simpson's D). Community composition of understory riparian vegetation changed post-fire, most likely as a result of the reduction in taxa richness and cover. Post-fire riparian tree mortality (>11.5 DBH) was only 4.4 percent. No post-fire change occurred in large woody debris volume and recruitment or in the amount of fine sediment in pools. Some water chemistry parameters increased (SO_4^-, total P, Ca^{2+}, and Mg^{2+}), and periphyton biomass decreased; however, these changes were short-term (≤1 year). Macroinvertebrate community composition was affected 10-19 days post-fire, but density, richness, and diversity were unaffected; furthermore, composition recovered within one year. These effects are discussed in more detail in chapter 6.1 (Watershed and Stream Ecosystems). Beche et al. (2005) explained that the limited observed impacts may be a result of the small portion (<20 percent) of the watershed area that burned, moderate topography, the low to moderate severity of the fire, and the below average precipitation year that followed the fire.

In a study from the Lake Tahoe Basin, prescribed burning in areas that included some ephemeral channels showed short-term (3 month) increases in calcium and pH but not a significant increase in the amount of soluble reactive phosphorus in stream waters (Stephens 2004: 258).

Recent research in the Tahoe Basin that examined effects of pile burning in riparian areas suggested that in most management settings, potential soil effects did not appear to be an overriding concern (see Soils chapter (5.0) for details on soil heating, although water quality results of that study are still in review).

Aspen Management

Aspen is an important vegetation type associated with riparian areas in the Sierra Nevada where treatments and research have been conducted. Recent studies have demonstrated the benefits of selective conifer removal in aspen stands; studies have taken place on the Eagle Lake Ranger District (ELRD) and the Lassen National Forest (Jones et al. 2005, Jones et al. 2011), and there is also an ongoing study in the Tahoe Basin (Berrill and Dagley, in review). At the sites in Lassen, the removal treatments were conducted in concert with control of heavy grazing pressure, and harvest was selected over the use of fire to avoid damage to the aspen trees. They reported that hand pile burning within the treated stands killed aspen roots and appeared to inhibit regeneration (Jones et al. 2005).

Research Gaps and Management Implications

Dwire et al. (2010) concluded that there is little information about specific and cumulative impacts of different fuels reduction treatments within riparian systems. Study results are often quite variable and confounded by local effects of other past and current management activities (Wondzell 2001). Stone et al. (2010) concluded that additional experimental studies of fuels treatment effects on aquatic and riparian ecosystems are needed before generalizations can be made across different forest types and local conditions. After reviewing the literature, Dwire et al. (2010: 194) reached a similar conclusion: "Current knowledge on the effects of fuel reduction treatments on riparian areas is limited, and research is needed to address the impacts of fuel treatments on watershed processes, riparian functions, and aquatic resources."

The 1996 Sierra Nevada Ecosystem Project report included a number of recommendations on riparian management, including a prohibition on vegetation removal and ground disturbance within riparian zones, which was intended to benefit both riparian and aquatic habitats. That section emphasized the importance of riparian tree canopies within first- and second-order streams in blocking summer sun and moderating water temperatures, as well as stream loading of large wood and other organic matter from riparian trees. It also suggested a fixed buffer width of 150 feet based on typical tree heights in the Sierra Nevada, and it recommended adopting wider, variable buffer widths that could be increased to account for variation in the riparian community and hillslope and soil erodibility. They asserted that "even the natural role of disturbance…does not require, in most situations, active restoration of the landscape in order to secure the habitat conditions necessary for the area" (Kondolf et al. 1996: 1026).

However, recent science has shown that higher stem densities and fuel loads in riparian forests can serve as a wick for high-intensity fire to move within treated upland forests under some conditions, such as the Angora Fire in the Tahoe Basin (Murphy et al. 2007, Pettit and Naiman 2007, van der Water and North 2011). More studies of variation across riparian areas are needed, but limited evidence does suggest some of these forests are vulnerable to uncharacteristically high-severity fires under severe weather conditions; as a result, scientists have noted the importance of considering treatments in riparian areas as part of landscape-scale restoration strategies (Messier et al. 2011, van der Water and North 2011).

Broad principles based upon recent science discussed in this synthesis suggest that more active management within riparian areas, including mechanical harvest, could promote resilience to uncharacteristically severe wildfire. The principles of restoring upland forests described in the Integrative Approaches chapter (1.1) can extend to riparian areas. For example, it may be appropriate to design treatments to increase heterogeneity where it has been reduced. Customization to local conditions and consistency with principles designed to promote resilient soils (see Soils chapter (5.0) would help to develop specific treatment approaches.

Effects of fire suppression and lack of active treatment have contributed to high fuel loads, increased tree density, and shifted vegetation composition to less fire-resistant species in riparian areas as well as in uplands. Treatments should reduce the likelihood of high-severity wildfires where they are not characteristic of the landscape. Riparian areas support important resource values, they are well adapted to recovery from disturbance, and even uncharacteristically high-severity fires may not necessarily impair long-term recovery of key functions. Outcomes may depend on the extent and severity of fire in the surrounding landscape and the vulnerability of downstream aquatic resources (see Watersheds and Stream Ecosystems chapter (6.1)). Better information is needed to understand how uncharacteristically severe fire may alter trajectories in riparian areas over a range of time scales relevant to understanding particular ecological processes (such as aquatic life cycles, channel organization, recruitment of woody debris, etc.).

Rieman et al. (2003) stated that objectives for fuels reduction treatments should include the return to fuel loads that support ecosystem processes and natural disturbance regimes and incorporate short- and long-term targets for the vegetation condition of uplands and riparian areas. Fuel loads in many riparian forests are so high that mechanical treatments may be needed to reduce fuels to levels that facilitate safer reintroduction of fire. Studies in uplands show that mechanical fuels reduction treatments, if conducted properly (i.e., reducing surface and ladder fuels), can effectively reduce fire severity under most weather conditions (Safford et al. 2012). These treatments should work just as effectively in riparian areas, although higher productivity in riparian areas may necessitate more frequent maintenance.

Riparian treatments would need to be evaluated and monitored to assess impacts and guide approaches in the future. There may be valuable opportunities to better link management and research. For example, Stone et al. (2010) interviewed USDA Forest Service Fire Management Officers in 11 western states and found that 43 percent were conducting fuels reduction treatments in riparian areas (California had 7 of 12 districts with riparian treatments). Although 88 percent of the districts reported monitoring activities to evaluate the effectiveness or ecological effects of the fuels reduction treatments in riparian areas, most monitoring was qualitative or not collected with sufficient spatial and temporal replication for quantitative summaries.

The special nature of these systems warrants developing localized prescriptions based in part upon historical fire regimes. For instance, approaches should differentiate riparian areas that function similar to upland landscapes in terms of fire frequency and spread; as discussed earlier, stream order may be a useful distinguishing characteristic. Van de Water and North (2010: 394) suggest that the following riparian types could probably be treated similarly to upland areas, including;

- Lower elevation riparian areas;
- Riparian areas adjacent to small, incised headwater streams that historically experienced fire at frequencies similar to those of upland areas; and
- Riparian areas surrounded by forests with a high proportion (about one-third of the basal area or greater) of fire-tolerant pines, especially those on the east side of the Sierra Nevada.

For other kinds of forested riparian areas, including those at higher elevations and those bordering wider streams, they recommended considering less intensive treatments, such as hand thinning and pile burning small trees (Van de Water and North 2010).

An ongoing research experiment in eight Sierra Nevada watersheds in the mixed-conifer zone will provide new insight into restoration treatments in headwater riparian areas for both mechanical thinning and prescribed fire (see sidebar on KREW in Watersheds and Stream Ecosystems chapter (6.1)). However, the research gap is so large that more adaptive management research is needed to develop guidelines for mechanical and prescribed fire treatments in riparian areas within the synthesis area. Consequently, the approach of large experimental areas outlined in the Integrative Approaches chapter (1.1) might incorporate adaptive management experiments within riparian areas to help fill this gap.

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6.3 Wet Meadows

Jonathan Long with contributions from Karen Pope and Kathleen Matthews

Contents

Executive Summary

Wet meadows help to sustain favorable water flows, biological diversity, and other values; consequently, restoration of degraded wet meadows is an important part of a strategy for promoting socioecological resilience. This chapter focuses on high-elevation wet meadows that are associated with streams; thus, restoration of such meadows may be considered a subset of stream restoration. Evaluations of wet meadow restoration efforts within the synthesis area have demonstrated gains at specific sites in certain functions, including water quality, water quantity, and macroinvertebrate diversity. Broader reviews in the past decade indicate that stream restoration efforts have often fallen short in demonstrating anticipated benefits, especially in terms of wildlife and fishes. These shortcomings may reflect a variety of causes, including incomplete documentation of projects, inability of treatments to address limiting factors, and limitations on monitoring resources, experimental designs, and timeframes. Consequently, researchers have cautioned against overstating the benefits of treating small sites rather than addressing conditions at the watershed or landscape scale. These findings reinforce the need for continued and increased monitoring of treatment outcomes, use of rigorous experimental designs, and use of conceptual models when evaluating the potential for improving site conditions, designing treatments, setting restoration objectives, and evaluating outcomes. A general trend consistent with the synthesis has been to emphasize restoration of processes, such as flooding, and avoiding more rigid structural approaches that may unnaturally restrict processes of bank erosion and vegetation. Active site-specific restorations may be warranted where local factors have caused degradation to a point where natural recovery is likely to be extremely slow. In particular, monitoring the rate and extent of channel incision is important to avoid losses of socioecological values in streams and meadow ecosystems associated with erosion and lowering of water tables. However, considering broader landscape influences on meadows may be more important over the next several decades, given that climate change, wildfire, and spread of non-native species may alter key ecological processes in meadows, such as aggradation and vegetation growth. Considering a wide range of ecological, social, cultural, and economic interactions is important in any restoration effort, but will be increasingly complex at broader scales. Designing, conducting, and evaluating restoration strategies in an adaptive management framework will benefit from broad participation by managers, researchers, and community members.

Figure 1: Wet meadow restoration site on Angora Creek in the Lake Tahoe Basin Management Unit. Photo by Jonathan Long.

Introduction

This chapter addresses wet meadows, and in particular, high-elevation wetlands that have fine-textured soils and have shallow groundwater tables in the summer. These conditions support wetland vegetation, predominantly herbaceous plants, including sedges, other graminoids, and forbs, but also woody plants, such as willows that can tolerate anaerobic conditions (Ramstead et al. 2012, Weixelman et al. 2011). This chapter focuses on meadows that are associated with defined stream channels (Figure 1). It does not focus on headwater fens and other peatlands, which are relatively uncommon and valuable systems; recent publications provide guidance for assessing their conditions (Weixelman et al. 2011).

Stream restoration in general is an important part of an overall restoration strategy for the synthesis area (USDA Forest Service 2011a). Restoration of wet meadows provides important opportunities to promote ecological resilience and benefit social values (Weixelman et al. 2011). Wet meadow restoration is expected to have an important role in securing favorable flows of high quality water (Viers and Rheinheimer 2011), mitigating anthropogenic carbon and nitrogen (Norton et al. 2011), and supporting biodiversity including pollinators (Colloran et al., in press).

Restoration of streams has been a focus of research across the Sierra Nevada region, the state of California, and the United States within the past decade. Earlier synthesis reports for the region, including the Sierra Nevada Ecosystem Project report (SNEP Science Team 1996), the 1998 Science Review (Sierra Nevada Science Review Team 1998), and the 1999 report "Sierra Nevada Ecosystems in the Presence of Livestock" (Allen-Diaz et al. 1999), remain relevant and useful because they address a broader range of meadow ecosystems and related topics, including conservation of aquatic biodiversity, sustaining stream flows for wildlife and human uses, and grazing management on public lands. A more recent synthesis intended for Great Basin ecosystems by Chambers and Miller (2011) explains that strategies to restore streams and meadows should consider a wide range of watershed impacts, including roads, grazing, and water diversions. Restoration strategies may be most effective if they consider where and when addressing these watershed influences is necessary to promote restoration, and in which cases active interventions are warranted and cost effective (Hobbs and Cramer 2008, Kauffman et al. 1997).

Promoting Resilience in Wet Meadow Ecosystems

The first chapter in this section, Watersheds and Stream Ecosystems (6.1), provides a definition of resilience and emphasizes the importance of restoring natural fluvial processes. Although these concepts apply generally to montane wet meadow systems, a strategy of relying on natural disturbance processes may be less effective in these less physically dynamic systems because they have relatively small watershed areas and reduced stream power. There is widespread recognition that channel headcutting in headwater systems can be indicative of a more persistent disequilibrium, in part because the process is difficult to reverse through natural deposition in such small systems. In the Great Basin, many streams have a natural tendency toward incision and may be prone to channel entrenchment for thousands of years (Germanoski and Miller 2004). However, meadow systems on the western slope of the Sierra Nevada appear to have been stable in recent evolutionary time (Benedict 1982). Reference erosion rates appear very low in many small headwater streams and in wet meadow systems that have intact streambank vegetation, as reported by studies in the Sierra Nevada (Micheli and Kirchner 2002, Simon 2008). As a consequence, high rates of incision and bank erosion are more likely to be outside the range of historical variation in these systems. The presence of these uncharacteristic conditions would strengthen the rationale for active intervention.

To evaluate what kinds of interventions, if any, are warranted in a particular ecological system requires analysis to determine whether abiotic or biotic thresholds have been passed (Hobbs and Cramer 2008). In considering these questions, Sarr (2002) offers a useful description of recovery trajectories following past damage by overgrazing, although it is reasonable to apply this resilience-based framework to disturbances more generally:

1. The "rubber band model": systems can recover quickly and predictably; this model is typically found in productive sites where soils and geomorphology are intact;
2. The "Humpty Dumpty model": systems fail to recover due to changes in system structure or function; and
3. The "broken leg model": systems recover slowly and remain more sensitive to impacts than they were prior to the disturbance.

In the rubber band model, recovery trajectories reflect a simple reversal of the degradation process. The two latter models describe a shift to alternative states marked by channel incision (discussed in more detail below), lowering of local water tables, reduced connectivity of channels to broad floodplains**,** and encroachment of non-hydrophytic woody plants (particularly conifer trees and sagebrush) in formerly wet riparian areas (Sarr 2002). Because these changes are often interrelated, active restoration efforts to remove woody vegetation, raise groundwater levels, and reestablish burning regimes may be needed to restore native herbaceous communities (Berlow et al. 2003). Sagebrush encroachment is addressed in detail by Chambers and Miller (2011). Active revegetation measures, such as transplanting sedges, are more effective in settings where groundwater tables are sufficiently high, which fits the description of the rubber band model (Steed and DeWald 2003). Since restoration potential appears to be very site-specific, studies of geology, hydrology, and soils attributes, as well as assessment tools discussed in the section on monitoring and evaluation near the end of this chapter, are important to determine site potential and to select appropriate treatments (Ramstead et al. 2012).

Channel Incision

Channel incision can cause a profound loss of productivity in wet meadow ecosystems. Shields et al. (2010) described incision as a syndrome that threatens many ecosystem services by triggering a cascade of geomorphic, hydrologic, and biological effects, including bank instability, channel erosion, perturbated hydrology, non-point source pollution, conversions from wet to dry meadow vegetation, degradation of aquatic habitat, and reduced fish species richness. Causes of incision may be natural, such as geological uplift, lowering of channel base levels associated with changing climate, or extreme runoff events associated with wildfires or storms; or due to more specific human actions, such as blocked culverts that impair sediment movement or overgrazing that removes protective vegetation or substrates from stream channels. These influences result in excess capacity of a stream to transport sediment relative to the supply of sediment from upstream reaches (Simon and Rinaldi 2006). In theory, long-term reductions in sediment supply due to fire suppression could also leave a system more vulnerable to incision.

Studies within the synthesis area have reinforced the importance of addressing channel incision. The potential for channel incision to pierce low-permeability layers and alter stream hydrology is a particular concern; such layers may be associated with peat or with compacted soils that are a legacy of historical heavy grazing (Hill and Mitchell-Bruker 2010). Local studies have also quantified some impacts from incision; for instance, a study of Monache Meadow in the southern Sierra found that banks without wet meadow vegetation are approximately ten times more susceptible to erosion than banks with herbaceous wet meadow vegetation, such as sedges and rushes (Micheli and Kirchner 2002). Where channels have active headcuts, herbaceous vegetation may not be effective in preventing bank erosion (Zonge et al. 1996). Moreover, physical changes associated with channel widening or incision, including increased temperatures, are not readily changed through restoration of riparian vegetation alone (Poole and Berman 2001). Because of the profound losses in ecosystem functions that can occur as a result of incision (Sarr 2002), management strategies and monitoring would benefit from focusing on this process. Although incision is likely to be a predominant problem of concern, other processes in meadows are important, such as channel widening (Loheide et al. 2009).

Chambers and Miller (2011) proposed a general framework for addressing incised meadows based on the degree of incision (Table 1). Their synthesis is particularly useful for considering issues that are important for east-side systems, such as sagebrush encroachment and management of the fire-adapted, invasive cheatgrass.

Table 1: Framework for addressing incision within wet meadow systems by Chambers and Miller (2010).

Condition	Indicators	Treatment Approach
Low to moderately incised	Channel has incised to 0-2 times bankfull channel depth	In-stream structures and bank stabilization measures to prevent knickpoint migration and maintain meadow vegetation
Highly incised	Channel has incised to >2 times bankfull channel depth	Careful design of in-stream structures to minimize further incision of the main channel and to maintain springs
Fully incised	Channel has previously incised but is no longer actively incising	Actively manage area to maintain meadow vegetation based upon knowledge of groundwater tables and riparian vegetation

In-stream Structural Approaches

Because waiting for streams to stop incising may result in extensive erosion, structural interventions are often proposed for incising systems. Structural treatments include check dams; grade-control structures; headcut revetments; streambank armoring; and channel reconfigurations, including diversion, filling, or plugging of existing channels and excavation of new ones. In-stream structures have long been utilized to enhance aquatic habitat, but there remain serious concerns about their potential for failure. For example, Stewart et al. (2009) concluded that managers should be circumspect in using in-stream engineered devices because evidence does not support their effectiveness, although failures are more common in larger streams. Common failures include erosion around or under the structures, and in-channel deposition and flood flows over the meadow surface that can cause incision elsewhere in the meadow. Consequently, in-stream structures require long-term monitoring and maintenance. An important trend in restoration strategies is to move from permanent structures toward temporary protection and enhancement that allows natural vegetation, sedimentation, and erosional processes to reestablish (Miller and Kochel 2010). These softer or "deformable" treatments may be somewhat more challenging to evaluate than harder, in-stream structures because by their nature, they are intended to be overtaken by natural recovery processes. Although montane wet meadow systems are less dynamic than low-elevation riverine systems, the general principle of designing deformable treatments is still relevant.

Channel filling and plugging

Filling of incised channels has been conducted in a number of sites in the synthesis area (Loheide et al. 2009, Ramstead et al. 2012). For sites that are thought to have historically lacked defined channels, flows may be directed over the meadow surface (examples include Halstead Meadow in Kings Canyon/Sequoia National Park and Wawona Meadow in Yosemite National Park). In other cases, they may be diverted into one or more remnant channels that have a more desirable geomorphic configuration and vegetation. Either approach requires careful attention to protecting and/or restoring native vegetation and hydrology to prevent re-incision. Where remnant conditions are not suitable for reintroducing flows, practitioners have often constructed new channels.

To reduce the volume of material needed to refill incised channels, practitioners have developed the "plug and pond method," wherein materials are excavated within the meadow, creating ponds, and then the channel is plugged at various locations using the excavated materials. This method has been the subject of studies in the Feather River watershed, which provide evidence that this method is effective in restoring many attributes of these systems (Loheide et al. 2009), as described further in the next section. However, researchers have noted concerns about these treatments:

- Plug and pond creates novel conditions of deep ponds, which can become habitats for invasive aquatic species, such as bullfrogs (*Rana catesbeiana*) and green sunfish (*Lepomis cyanellus*) (Adams and Pearl 2007, Ramstead et al. 2012).
- Channel reconstruction or plug and pond methods may be inappropriate in systems with fine-grained confining units, since the process of excavating alluvial materials could disrupt the meadow hydrology (Chambers and Miller 2011).

Evaluating Benefits of Meadow Restoration

Water quantity and quality effects of meadow restoration have been undertaken at a relatively small number of sites in the Sierra Nevada within the past decade, with considerable emphasis on large, low-gradient meadows along tributaries of the Feather River and streams in the Lake Tahoe basin. Published studies suggest that active meadow restoration designed to remedy incised channels has increased groundwater levels and subsurface storage, which in turn promotes wetland vegetation (Hammersmark et al. 2010); increased frequency and duration of floodplain inundation, which in turn may filter sediment and nutrients; attenuated peak flows and increased mid-summer baseflows (Hammersmark et al. 2008, Tague et al. 2008); and reduced maximum water temperatures (Loheide and Gorelick 2006). Stream flow below restored meadows may be affected by higher evapotranspiration rates in the rewetted meadows (and any created ponds) and increased subsurface storage (Hammersmark et al. 2008, Loheide and Gorelick 2005). Research has helped to understand how site qualities influence response, including the presence of impermeable layers that can maintain high water tables but also can inhibit groundwater from upwelling to the meadow surface (Booth and Loheide 2012). The water quality and water quantity benefits of wet meadow restoration are an important topic for which the National Fish and Wildlife Federation has initiated a major research initiative in Region 5 (Viers and Rheinheimer 2011).

Restoration of meadow hydrology and vegetation should generally result in a cascade of higher order functions, including increases in soil carbon and improvements in fish and wildlife habitat. Nevertheless, a number of reviews have suggested caution not to oversell these benefits without further monitoring and research to quantify them. This concern has emerged in light of the popularization and commercial expansion of stream restoration in parts of the United States (Lave et al. 2010). Several reviews have recommended more rigorous application of ecological theory and greater emphasis on monitoring outcomes (Palmer 2009, Ramstead et al. 2012). Bernhardt and Palmer (2011) noted that research in recent years has progressed from asking “Why don't we know more about river restoration success?” to asking “Why aren't river restoration projects more effective?” This general trend also appeared to unfold in California, where Kondolf et al. (2007) highlighted a lack of information needed to evaluate projects. In a meta-analysis of effects of stream restoration projects on macroinvertebrates, Miller et al. (2010) did not include any studies from the Sierra Nevada, presumably because they did not find ones that met their criteria for a controlled research design. A recent evaluation of wet meadow restoration in the Southwest, which included studies of about a dozen projects from the synthesis area, concluded that although there has been significant progress in restoring morphology and vegetation, there remains a need for long-term and better-designed monitoring programs (Ramstead et al. 2012). These reviews noted lack of controls and confounded treatments as a common problem in evaluating project effects. For example, passive restoration through changes in grazing management are often confounded with structural restoration treatments, and sites that have not been treated recently may have an older history of treatments. Others have noted the potential for a publication bias in favor of reporting more successful projects (Ramstead et al. 2012, Stewart et al. 2009).

Demonstrating benefits of stream and meadow restoration becomes more challenging when evaluating benefits to higher order ecosystem services, including biodiversity. Researchers have criticized shortcomings of some restoration projects as relying upon the “fields of dreams hypothesis” that “if you build it, they will come,” in which “it” refers to physical structure, hydrology, and/or vegetation, and “they” refers to the desired biological community, usually wildlife (Hobbs and Cramer 2008, Palmer et al. 1997). Defenders of that approach may counter that projects that fell short may have lacked restoration of critical ecosystem processes, such as overbank flooding and fire, so in effect, they did not “rebuild it.” Nevertheless, researchers contend that this hypothesis needs to be rigorously tested for different habitats and different species (Palmer et al. 1997). In recent years, researchers have reviewed stream restoration efforts nationwide to evaluate this hypothesis. Bernhardt and Palmer (2011) cautioned that channel reconfiguration efforts may reduce bank erosion and increase sinuosity, but that evaluations have found little evidence for benefits to sensitive taxa and water quality (in particular, reduction of nutrients). They cautioned that many projects in the United States are undertaken at sites where watershed degradation is a key factor, so reach-specific channel restoration treatments do not treat the underlying causes of degradation. However, their review included many urbanized streams and other sites in heavily altered watersheds. Sites on national forests in the Sierra Nevada are less likely to have experienced severe watershed-scale impacts (although dams may have significantly altered hydrologic processes in some watersheds), and many restoration projects have targeted streams and meadows that have been significantly affected by localized road, channelization, or grazing impacts. Nevertheless, this research firmly underscores the importance of long-term monitoring and research to evaluate more complex, higher order outcomes of restoration.

Researchers have emphasized the importance of conceptual models to explicitly state and test the strength of linkages between various fundamental changes, such as modifying channels to reduce entrenchment and increase the areas flooded during frequent floods, to vegetative effects and higher order effects on fish, amphibians, and terrestrial wildlife. Through a national meta-analysis of two dozen studies, Miller et al. (2010) concluded that although habitat restoration may promote biodiversity and ecosystem resilience, its ability to increase biomass of macroinvertebrates for the benefit of higher tropic levels (e.g., fish, amphibians, and birds) was still uncertain. That study noted that channel reconfigurations yielded highly variable invertebrate community responses. Studies in the Catskill Mountains of New York showed significant responses to channel reconfiguration projects for fish populations but not invertebrates; the researchers hypothesized that responses to reconfiguration may be stronger in pool habitats where fish reside than in the riffle habitats where the macroinvertebrate samples were collected (Baldigo and Warren 2008, Ernst et al. 2012). On the other hand, in a study from the Sierra Nevada, Herbst and Kane (2009) reported that active channel restoration yielded a rapid shift in macroinvertebrate communities toward reference conditions. Conceptually, restoration of wet meadow hydrology should yield benefits for a variety of wildlife species, including willow flycatcher (*Empidonax traillii* Audubon) (Cocimano et al. 2011). However, many of these higher order biological objectives may prove hard to achieve (or to demonstrate) in short timeframes because of confounding or limiting factors, including legacy effects of past management, including historical overgrazing, soil compaction, mining, and stocking of non-native trout. For example, the stocking of trout into fishless systems has affected amphibians, reptiles, and birds by altering foodwebs in lakes and streams (Eby et al. 2006, Epanchin et al. 2010). A Sierra Nevada study by Purdy et al. (2011) found that fundamental indicators of vegetation and physical habitat tend to classify meadows as being in better condition than the aquatic indices do, especially the native fish and amphibian index. This finding could reflect a variety of causes, including legacy effects, time lags in these indicators, and controlling influences that are beyond the site.

Grazing Management and Wet Meadow Restoration

Livestock grazing involves a complex interplay of social and ecological factors (see Managing Forest Products for Community Resilience chapter (9.5)). Although grazing is only one of many land uses that impact streams and wet meadows, grazing management and hydro-geomorphic condition appear to be critical determinants of meadow restoration outcomes (Ramstead et al. 2012). In a recently published review of rotational grazing from a broad socioecological perspective, Briske et al. (2011a) offer frameworks to promote effective management of grazed systems, including adaptive management with an emphasis on stakeholder participation (Fernandez-Gimenez et al. 2008) as well as targeted grazing that explicitly emphasizes management outcomes, such as weed control, fire hazard reduction, and wildlife habitat improvement. The latter approach suggests that grazing management could be an important tool for promoting socioecological resilience in systems that evolved with grazing animals. This approach embodies the logic of disturbance-based management as described in North and Keeton (2008), and recognizes that grazing, like fire, can be a tool for rejuvenating areas by reducing accumulated vegetation. It is important to recognize that because different kinds of domesticated livestock (i.e., cattle, horses, and sheep) have different grazing behaviors and influences, they are not interchangeable with each other or with the prehistoric assemblage that may have grazed particular landscapes. Researchers have discussed the utility of grazing in “novel systems,” where grazing has a long history and non-native species have become dominant (Hobbs et al. 2009). In such systems, carefully managed livestock grazing may be a useful, albeit often controversial, tool for maintaining biodiversity and ecological services (Hobbs and Cramer 2008). For example, studying spring systems in Sierra Nevada foothills, Diaz et al. (2004) found that removing livestock grazing may allow dead plant material to accumulate, which in turn can increase levels of nitrate in wetland waters and decrease plant diversity. Similar findings have come out of research in vernal pool systems (Marty 2005). The ecological benefits of grazing-based management approaches to less invaded, high-elevation wet meadows of the Sierra Nevada are less clear. A report by the Sierra Nevada Ecosystem Project indicated that it was unknown whether grassland ecosystems in the Sierra Nevada were adapted to disturbance by prehistoric megafauna, and it suggested that more intensive grazing practices, such as active herding, could avoid many undesirable impacts (SNEP Science Team 1996).

A recent comprehensive report on riparian management practices provides an overview of prescribed grazing effects on a wide range of resource values, including wildlife habitat, water quantity and quality, stream bank and soil stability, carbon storage, plant and animal diversity, composition and vigor of plant communities, forage for grazing and browsing animals’ health and productivity, riparian and watershed function, soil condition, and fine fuel loads (Briske et al. 2011b). In a companion chapter on riparian management practices, George et al. (2011) found that grazing practices that result in heavy use of riparian vegetation, are too long in duration, or are poorly timed can be detrimental to aquatic values, such as fisheries and stream bank stability. They found support for grazing exclusion as a restoration strategy for degraded riparian systems because it promotes recovery of riparian plant community composition. However, they noted that other techniques for manipulating livestock distribution, including herding, supplement placement, water development and fences, are effective in reducing livestock residence time and utilization in the riparian zone.

Within the synthesis area, a recently completed 5-year study addressed the effectiveness of excluding cattle from breeding areas of Yosemite toads (*Bufo canorus* Camp). The researchers found no detectable differences in toad occupancy, toad density, or water quality between grazed and non-grazed meadows when livestock grazing met current standards, including 30-40 percent use (Roche et al. 2012a). In addition, they found that meadow hydrology there influences occupancy by toads, and cattle grazing intensity does not (Roche et al. 2012b). Recent studies on national forests of the Sierra Nevada have focused on levels of fecal coliform bacteria (*Escherichia coli*, specifically), reporting exceedances in several meadow streams with cattle grazing as well as recreational use in some cases (Derlet et al. 2012, Myers and Kane 2011, Myers and Whited 2012). Additional research on a number of sites in the synthesis area should help to put these findings within a broader context of national forest management (see sidebar under Research Gaps below).

Much of the research on grazing has had limitations on experimental design that constrain the range of inference to contexts that may not necessarily match conditions on national forest lands. Many studies of grazing impacts are difficult to translate to grazing management strategies when they lack details such as stocking rates or utilization levels (Briske et al. 2008). Many studies of grazing in the Western

United States, including the Sierra Nevada, have provided a dichotomous view of grazing by comparing differences or trajectories of vegetation and channel morphology inside and outside of exclosures (examples from the Sierra Nevada include Kondolf (1993) and Knapp and Matthews (1996)). Studies have commonly reported that where physical thresholds had not been exceeded (for example, channel incision that had lowered groundwater tables below the rooting zone), long-term grazing exclusion or reduction has facilitated substantial growth of native wetland herbaceous and woody vegetation such as willows (Ramstead et al. 2012). A review of exclosure studies on the Kern Plateau by Sarr (2002) noted that particular vegetative and channel responses to exclusion vary due to a host of factors, including watershed stability, climate, subsurface moisture availability, soil organic content, proximity of willow propagule sources, and degree of channel incision.

Variation in responses has also been reported in some studies on higher order responses within the past decade. Studies assessing the impacts of cattle on amphibians have often been correlative and have yielded mixed results; for example, Bull and Hayes (2000) found no evidence of negative effects of grazing on Columbia spotted frog (*Rana luteiventris* Thompson), but they noted their inability to control for wide variation in grazing intensity and other landscape variables. More experimental studies using cattle exclosures have also reported mixed results with specific implications for particular taxa. For example, reporting from a study in Tennessee, Burton et al. (2009) suggested that fencing cattle from wetlands may benefit ranid frogs, and controlled grazing may benefit toads in the genus *Bufo*.

Quantifying the influence of livestock grazing in stream and meadow ecosystems has been difficult because experimental designs may not sufficiently address ecological variation. Sarr (2002) identified common problems in evaluating responses to livestock grazing and exclusion, including lack of proper controls and the small size of exclosures. Research experiments are often conducted at too small a scale to properly evaluate effects (Briske et al. 2008). A study in a Sierra Nevada meadow found that grazing effects on soil nutrients were more pronounced at the edge of the forest than at the stream edge (Blank et al. 2006), demonstrating one mechanism by which whole meadow exclusion would have different effects.

Adaptive Management, Monitoring, and Research Gaps

Monitoring and Evaluation

Recent stream restoration research indicates that monitoring remains very important in evaluating whether restoration is achieving desired objectives and in improving practice. The Forest Service Watershed Condition Framework establishes a goal of a having “a comprehensive monitoring effort in place within 5 years” (USDA Forest Service 2011b). Performance metrics, such as number of “stream miles restored or enhanced,” may create an unintended incentive for intervention, particularly in reaches that may have lower treatment costs. Therefore, it is important to measure performance in terms of ecological condition. Furthermore, metrics based primarily on physical structure (such as high sinuosity and low width-depth ratios) may underemphasize ecological functions (Bernhardt and Palmer 2011) . Consequently, measurements of ecological processes (such as overbank flooding) and services (e.g., improved water quality, changes in seasonal water tables, dampened floods, and improvements in the diversity or abundance of target taxa) may be more appropriate for tracking progress (Bernhardt and Palmer 2011). This reflects another theme of this synthesis (Integrative Approaches chapter (1.1)). In addition, monitoring is important in evaluating project outcomes over a long period during which floods and vegetative development are expected to alter conditions. Finally, adopting a landscape perspective may be important in promoting socioecological resilience by giving increased weight toward meadows with greater potential to yield desirable outcomes. For example, stream reaches that are located close to less disturbed areas are more likely to be successful in reestablishing aquatic organisms (Bernhardt and Palmer 2011).

The Watershed Condition Framework considers channel incision within criteria for channel shape and function. Developing more specific quantitative criteria for this critical threshold of ecological function may be possible in different regions within the Sierra Nevada; for example, Micheli and Kirchner (2002) suggested that a bank height of 1 meter in a southern Sierra meadow represents a threshold for shifting to dry meadow species and less stable streambanks, and Chambers and Miller (2011) suggested a threshold of incision occurs when channel depths exceed twice the bankfull depth.

A recent study in the Sierra Nevada concluded that soil properties correspond with rapid assessments of meadow condition using the Proper Functioning Condition methodology; specifically, they found that meadows categorized as “properly functioning” have greater nitrogen and dissolved organic nitrogen than “non-functioning” or “at-risk” meadows, and greater carbon than “non-functioning” meadows (Norton et al. 2011). To evaluate grazing management in wet meadows, Blank et al. (2006) suggested using root-length density as an indicator of ecological function. Root depth is another useful indicator of functional condition, and these types of qualities can be related to vegetation cover and composition data that is collected more routinely (Weixelman and Zamudio 2001, Weixelman et al. 1999). Collectively, these studies provide a basis for rapid assessments that focus on channel incision and shifts in vegetation away from native obligate graminoids; however, they also point to more quantitative metrics and possible threshold values for monitoring.

Stewart et al. (2009) contended that managers need more research-based information about in-stream structural treatments before widespread use can be recommended. Kondolf et al. (2007) pointed to the importance of post-project evaluation, monitoring, and adaptive management for advancing stream restoration and learning from both successes and failures. They further noted that restoration projects in California rarely provide for monitoring beyond three years, which is likely inadequate to observe effects of large, infrequent events. They also argued that projects do not always meet the standards needed to evaluate the restoration outcomes articulated by Bernhardt et al. (2007), including a clearly defined goal, objective success criteria, and use of BACI (before-after, control-impact) monitoring designs. Ruiz-Jaen and Aide (2005) recommended including measures of diversity, vegetation structure, and ecological processes, and monitoring more than one reference site, to account for the temporal and spatial dynamics of ecosystems. Identifying suites of streams and watersheds that are in reference condition or could be brought into reference condition would facilitate evaluation of restoration potential and success in the face of anticipated stressors.

Monitoring frameworks need to consider temporal scale to account for effects of disturbances on key indicators (Berkes and Folke 2002, Bryant 1995). Because recovery of stream channels and floodplains may be limited by the episodic nature of flood disturbances (Sarr 2002), pulsed monitoring to coincide with climate and flood dynamics has been proposed as an efficient way to evaluate stream condition (Bryant 1995). Furthermore, interpreting indicators of stream condition relative to flood dynamics may be more appropriate than evaluating annual trends in systems where less frequent floods drive key ecological processes. The idea of quantifying a threshold to aid determination of when geomorphic monitoring is warranted was proposed by Florsheim et al. (2006) for dynamic lowland rivers. Even in relatively stable wet meadow ecosystems, environmental conditions can vary significantly from one year to the next, so reliable evaluations need a relatively long timeframe for monitoring, including pre- and post-treatment data, to demonstrate trends (Kiernan and Moyle 2012, Ramstead et al. 2012). These temporal dynamics further complicate efforts to evaluate impacts of grazing and rest, so one strategy is to consider long rest periods that provide opportunities to evaluate influences of grazing from multiple perspectives (Briske et al. 2011a).

Standardized monitoring and classification protocols could facilitate collection and comparison of data at larger spatial scales. Katz et al. (2007) provided a number of recommendations to facilitate evaluation of project effectiveness, including standardized metrics and a common reporting system for tracking restoration projects that includes common semantics for project type, location, timing, and magnitude. One tool that is becoming more widely used for evaluating site condition and trends over large management units, as well as tracking restoration performance, is the California Rapid Assessment Methodology (CRAM) (Stein et al. 2009). A CRAM module for assessing wet meadows is currently being developed and tested in the Tahoe basin. Validation is important to ensure that the methods are effectively capturing important information about condition and trend.

Many tools, such as CRAM and the Watershed Condition Framework, are intended to be applied over large areas. Classification systems are helpful in addressing heterogeneity within these areas. A field key by Weixelman et al. (2011) provides a tool for classifying meadow types based on several hydrogeomorphic factors, including soils, water source, and gradient. Loheide et al. (2009) developed a framework for predicting potential benefits of restoration based on several factors, including elevation, soil texture, and the degree of stream incision. Use of these classifications in project reporting and evaluation could help evaluate restoration treatment effectiveness across the Sierra Nevada by identifying geomorphic settings and hydrologic characteristics that appear particularly sensitive to threats or responsive to treatments.

Research Gaps and Pending Research

Many topics that influence the success of stream restoration warrant further research, including groundwater interactions between meadow aquifers and surrounding systems and the effects of meadow properties and various treatments on hydrologic responses to treatment (Hill and Mitchell-Bruker 2010). Long-term studies of effects of stream and meadow restoration on higher order values, such as water flows and aquatic life, remain a topic for further research, particularly in light of anticipated effects of climate change. However, a host of pending research projects will be helping to fill some of the important gaps in knowledge regarding wet meadow restoration (see sidebar below).

Regarding biodiversity, there has been some research on fire effects on amphibians, yet the combined effects of varying fire severity and interactions with timber harvest on amphibians has been raised as an important research topic (Hossack and Pilliod 2011). Appendix E of the Sierra Nevada Forest Management Plan amendment identified effects of fuels treatments as an important research topic, with a focus on site occupancy by the foothill yellow-legged frog. Other species that may be important to consider are terrestrial salamanders in the genus *Batrachoseps*, within which several new species have been recently described in the Southern Sierra (Jockusch et al. 2012). A synthesis of wildfire effects on amphibians found that four studies that include plethodontid salamanders reported negative effects on populations or individuals, and that these effects were greater in forests where fire had been suppressed and in areas that burned with high severity (Hossack and Pilliod 2011).

The introduction of beaver (*Castor canadensis*) has been suggested as a strategy for restoring wet meadows through their potential to induce sediment deposition, raise water tables, and alter relatively large habitat patches (Johnston and Naiman 1990). Beaver introductions in Yellowstone have shown potential to promote increased water surface area, wetland herbaceous vegetation, and riparian shrubs (McColley et al. 2012). However, the complex interactions between beaver activity, wetland hydrology and vegetation, and human infrastructure have to be considered, especially given the potential impacts of beaver dam failures (Beier and Barrett 1987, Butler and Malanson 2005). Recently published evidence that beaver were native to at least some watersheds in the Sierra Nevada suggests that other areas warrant more in-depth investigation (James and Lanman 2012, Lanman et al. 2012). These findings heighten the importance of research to determine the conditions under which beaver reintroductions may promote meadow restoration.

Beyond these more narrow research questions lies the poorly charted territory of large-scale restoration efforts to better incorporate disturbance regimes and multiple successional states (Lake et al. 2007). Proposals to study landscape-scale effects of forest treatments on multiple resource values would benefit from including aquatic resources, especially to evaluate effects of managing riparian areas and impacts from wildfires. Many of the key cause and effect questions and information gaps identified in the Sierra Nevada Forest Plan Amendment Appendix E (USDA Forest Service 2004) focused on impacts of livestock grazing practices. Sarr (2002) described many of the challenges in researching those effects; in response, he suggested that managers evaluate treatments at watershed scales on experimental rangelands, and in modest-sized exclosures as part of an ongoing adaptive management process. This strategy is reflected in the recent study by Herbst et al. (2012), which suggested that treatments at larger scales may yield different outcomes than smaller riparian exclosures.

Current and Pending Research

Golden trout and climate change adaptation

- The US Forest Service Pacific Southwest Research Station (PSW) has a current project funded by the National Fish and Wildlife Foundation to examine the resiliency of stream habitats in the Golden Trout Wilderness to future climate warming. This project will spatially analyze stream temperatures and shading in restored and degraded sections of Mulkey and Ramshaw meadows to estimate what proportion of the habitat will be resilient to climate change and what proportion should undergo restoration treatments. The project is using peak temperature threshold values of 23°C to trigger management responses, with a long-term goal of keeping streams below 20°C so they will be resilient to future climate warming. Monitoring of degraded and recovering stream sections will enable comparisons of temperature, dissolved oxygen, stream depth and width, and shading, and it will be used to guide restorative management actions. Chytridiomycosis continues to be a major conservation concern for frogs in mountain lake ecosystems. Researchers are working to develop treatment options to help infected populations.

Grazing management in national forests

- Researchers at UC Davis, led by Kenneth Tate, are completing a study across several national forests to determine whether grazing best management practices avoid exceedance of water quality standards.
- A joint Region 5 and UC Davis project, also led by Kenneth Tate, will publish long-term (1999 to present) rangeland condition and trend monitoring data on over 800 permanent plots from throughout the state.

Hydrologic effects of meadow restoration

- The US Forest Service Region 5 is preparing an assessment of meadow restoration and meadow hydrology for the Sierra Nevada. It is expected to be published in 2013.

Meadow restoration guide

- "A guide for restoring functionality to mountain meadows of the Sierra Nevada" (Stillwater Sciences 2012).

Meadow restoration and native trout reports

- Meadow restoration to sustain stream flows and native trout: a novel approach to quantifying the effects of meadow restorations to native trout (Henery et al. 2011).
- UC Davis researchers are in the process of releasing a study that examined the number and size of meadows in the Sierra Nevada.
- UC Davis researchers recently presented preliminary findings from a pending study of amphibians and climate change: “Using sensitive montane amphibian species as indicators of hydroclimatic change in meadow ecosystems of the Sierra Nevada, California.

Evaluation of meadow restoration

- A recent PSW meadow restoration study includes sites from the southern Cascades to the Stanislaus National Forest (Figure 2). Response variables of interest included basic physical and vegetative indicators of wetland condition, including soil moisture, soil carbon, vegetation cover and biomass, channel depth, and presence of headcuts.

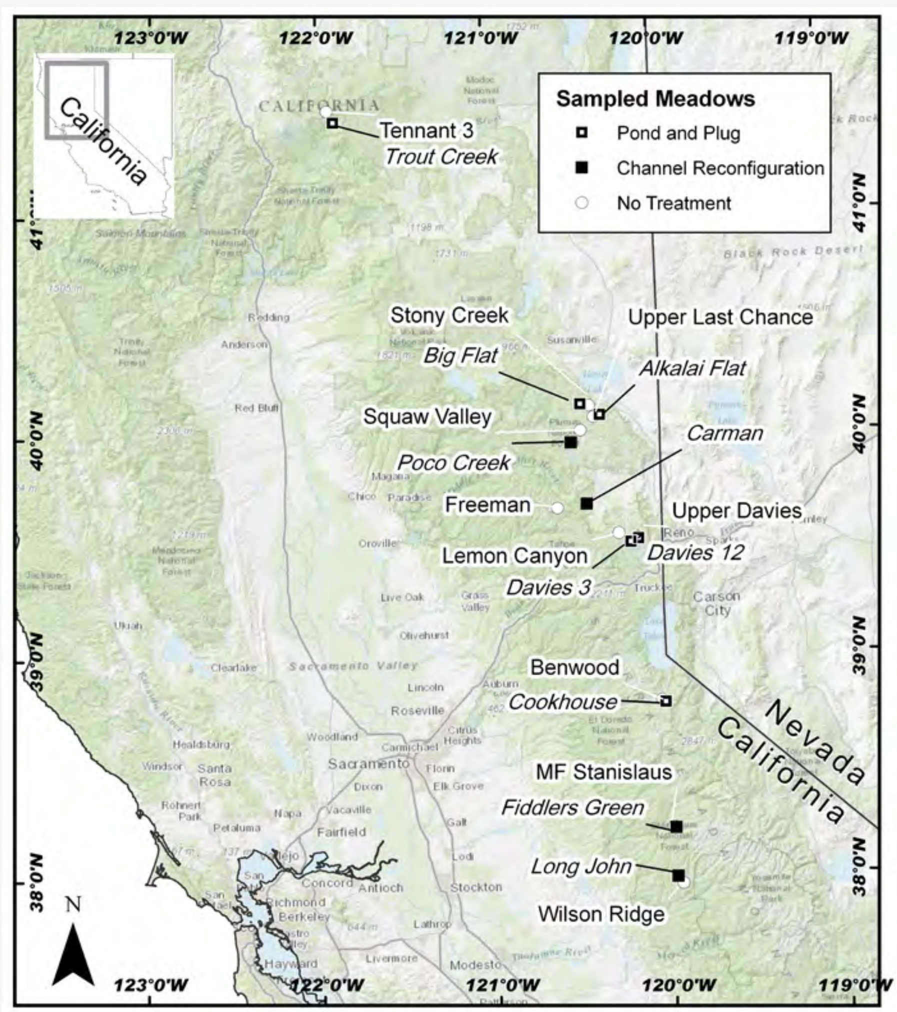

Figure 2: Map of sites in the meadow restoration study led by the Forest Service Pacific Southwest Research Station.

Integrated Socioecological Approaches to Stream and Meadow Restoration

Because streams and meadows provide important ecological services, such as promoting favorable water flows, and other socio-cultural values, they present opportunities to pursue restoration that integrates ecological and social considerations. Therefore, in addition to considering spatial and temporal scales, restoration and monitoring efforts should consider the potential benefits of heightened community involvement. Bernhardt et al. (2007) observed that projects involving local community members and advisory committees are more likely to meet their standards, which suggests that those projects may have more resources and support and/or more expectations to demonstrate results. Community participation in planning may also help to increase the success and social benefits of stream restoration (Golet et al. 2006). Other researchers note that advancing management will depend not on experiments that test whether a particular approach is compatible with specific ecological standards, but rather on a participatory or collaborative adaptive management program that accounts for both social and ecological objectives and promotes learning to meet those objectives (Briske et al. 2011a). This approach builds upon the distinction drawn by Higgs (1997) between "effective restoration" that is focused on meeting technical performance criteria and "good restoration," which addresses the value of the restoration in socio-cultural contexts.

Figure 3: Fly-fishing along Trout Creek, Lake Tahoe Basin Management Unit, represents one of the social benefits of functioning wet meadows.

Increasing interest in the idea of payments for ecosystem services through restoring montane meadows of the Sierra Nevada has generated excitement about the potential to accelerate the pace of restoration

(Viers and Rheinheimer 2011). However, some researchers have worried that the increasing importance of stream restoration as a business and potential market for ecosystem services could be distortionary (Lave et al. 2010). Kondolf et al. (2007) point out that although quantitative criteria are important measures of success, projects may also have broader goals regarding stakeholder involvement. Many meadow sites have strong cultural value to tribes and may have been managed by Native American tribes (Ramstead et al. 2012). A strategy discussed in the Fire and Tribal Cultural Resources chapter (4.2) focuses on partnering with tribes to reestablish more frequent fire regimes and enhance growth of culturally desirable plants, such as sedges, willows, and various geophytes, including beargrass, that are commonly associated with riparian and wet meadow habitats.

Management Implications

- Wet meadows can be vulnerable to transformations that result in diminished socioecological value. The flip side of that coin is that restoration of these systems holds great potential to provide multiple ecological and social benefits, despite their small share of the landscape. Research to date suggests that projects can promote important benefits; however, additional long-term monitoring and research would help to evaluate those benefits and prioritize investments in restoration.
- In particular, long-term studies of effects of stream and meadow restoration on higher order values such as water flows and aquatic life remain a topic for further research, particularly in light of anticipated effects of climate change.
- Assessments of the number, size, location, current condition (especially extent of incision), and recovery potential of degraded wet meadows in the synthesis area will help managers to target and prioritize structural interventions, changes in grazing practices, or other restoration treatments.
- In addition to site-specific assessments and treatments, examination of disturbances (e.g., wildfire) and management practices (e.g., prescribed grazing practices) on a larger, watershed scale, could aid the design of more effective strategies to promote long-term resilience of these valuable systems.

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6.4 Lakes: Recent Research and Restoration Strategies

Karen Pope and Jonathan Long

Contents

Figure 1: Dusy Basin Lake, Kings Canyon National Park. Photo by Kathleen Matthews.

Executive Summary

Mountain lakes in the Sierra Nevada and southern Cascades (Figure 1) have high ecological value because they provide food for terrestrial and aquatic predators and support rare fauna such as the mountain yellow-legged frog (*Rana muscosa/R. Sierrae*). Lakes also have high social value, in particular by serving as destinations for hiking, camping, swimming, and fishing. A number of stressors interact to impact lake ecosystems in the synthesis area. Climate change is expected to affect lakes by altering physical processes and reducing water levels. In shallow lakes and ponds, reduced hydroperiods could directly reduce the amount of available habitat for lentic amphibians and increase the instances of stranding mortality of eggs and tadpoles. Introductions of fish into lakes have altered foodwebs and particularly impacted native amphibians. Chytridiomycosis, an amphibian-specific fungal disease caused by *Batrachochytrium dendrobatidis* (*Bd*), has caused significant declines and extirpations in populations of amphibians native to the Sierra Nevada and southern Cascades. Research is ongoing to determine ways to reduce impacts of *Bd* on native amphibian populations. Air pollution has potential to negatively affect lake-dwelling amphibians, especially due to interactions with other stressors; however, recent studies from the synthesis area did not find associations between frog population status and measured

pesticide concentrations. A metric using the ratio of the number of taxa observed at a site to that expected can be an effective tool for assessing resistance and resilience of expected native taxa to threats. Successful restorations will likely depend on the elimination of introduced fish, the presence and virulence of *Bd*, and habitat conditions that help frogs to withstand these and other stressors.

Socioecological Significance of Lakes

The Sierra Nevada and southern Cascades contain an abundance of natural lakes that provide both ecological and recreational values. In high-elevation environments characterized by low productivity, lakes provide food and energy subsidies to terrestrial consumers, such as birds (Epanchin et al. 2010) and snakes (Matthews et al. 2002, Pope et al. 2008). In addition, native endemic species, such as the Sierra Nevada yellow-legged frog (*Rana sierrae*) (Figure 2) and Yosemite toad (*Anaxyrus* [=*Bufo*] *canorus*), are found in California's mountain lakes. These species, in addition to the southern mountain yellow-legged frog (*R. muscosa*), are candidates for listing on the federal endangered species list (USDI Fish and Wildlife Service 2011).

Lakes also serve as important destinations for recreational activities, including hiking, camping, swimming, and fishing. Historically, nearly all lakes in the high Sierra Nevada (above 1800 m) were fishless, but the introduction of non-native trout has been a common practice since the early 1900s (Knapp 1996). Currently, the majority of large, deep lakes support introduced populations of fish, including brook trout (*Salvelinus fontinalis*), rainbow trout (*Oncorhynchus mykiss*), and brown trout (*Salmo trutta*) (California Department of Fish and Game 2011, Knapp and Matthews 2000). This fishery supports a multimillion dollar recreational industry by bringing anglers to the region.

Figure 2: Recent metamorph and adult forms of Sierra Nevada yellow-legged frog. Photo by Kathleen Matthews.

Conservation Issues and Restoration Options

Climate Change

Climate models estimate that temperatures in California will increase 1.5 – 4.5° C by the end of the 21st century (Cayan et al. 2008). Summertime lake surface temperatures may be warming more rapidly than air temperatures. Using thermal infrared satellite imagery, Schneider et al. (2009) found that six large lakes in California (including Lake Tahoe) have been warming an average of 0.11 ± 0.02 °C per year since 1992. These rates of change are about twice as high as regional trends in air temperature (Schneider et al. 2009). As a consequence of climate change, increased surface water warming rates will likely impact lake ecosystems of the Sierra Nevada, affecting large reservoirs and Lake Tahoe not only by potentially decreasing lake levels, but also by altering critical physical processes, such as stratification and deep mixing (Sahoo et al. 2012).

Warmer air temperatures are predicted to result in less annual snowpack in the Sierra Nevada (Cayan et al. 2008, Young et al. 2009), which in turn will affect the hydroperiod of small lakes and ponds. Reduced hydroperiods could directly reduce the amount of available habitat for aquatic amphibians and increase the instances of stranding mortality of eggs and tadpoles (Lacan et al. 2008). Warmer temperatures could also affect the distribution of pathogens and their vectors, exposing frogs to new pathogens (Blaustein et al. 2001). Sierra Nevada yellow-legged frogs and southern mountain yellow-legged frogs (hereafter mountain yellow-legged frogs) have a 2-3 year larval stage and may experience an interactive effect of fish and climate change, with fish preventing breeding in the majority of deep lakes (Vredenburg 2004) and climate change increasing the likelihood of drying of shallow lakes and ponds that do not support fish (Lacan et al. 2008). Based on this expected interaction, mitigations for climate change effects on native lentic species include removing fish from some larger lakes to provide additional fish-free, permanent, cool water refuge habitat (see Introduced Fishes section below).

Introduced Fishes

The broad-scale introduction of non-native trout into the majority of larger lakes has had significant negative effects on priority native species that rely on these lakes. For example, deep, permanent waters are critical as overwintering and breeding habitat for mountain yellow-legged frogs (Bradford 1989, Knapp and Matthews 2000). Introduced trout are significant predators of these frogs (Grinnell and Storer 1924, Vredenburg 2004), and many studies have found that breeding populations of mountain yellow-legged frogs rarely co-occur with non-native trout (Knapp and Matthews 2000, Knapp 2005). Both the federal and state status evaluations of mountain yellow-legged frogs consider introduced fish to be a primary cause of their range-wide declines (California Department of Fish and Game 2011, USDI Fish and Wildlife Service 2003).

Aquatic invertebrates are major components of both freshwater communities and adjacent terrestrial habitats. Larval insects and zooplankton serve as prey for larger aquatic insects and amphibians, and the winged adult stages of insects feed terrestrial predators, such as birds, bats, and spiders (Nakano and Murakami 2001, Sanzone et al. 2003). In high-elevation lakes, introduced trout can produce strong top-down effects on aquatic invertebrates (Vadeboncoeur et al. 2003, Knapp et al. 2005, Pope et al. 2009). This is important because changes in invertebrate abundance and composition can have cascading consequences for nutrient cycling (Schindler et al. 2001) and terrestrial communities (Knight et al. 2005, Finlay and Vredenburg 2007, Epanchin et al. 2010). For example, Knight et al. (2005) found that fish reduce dragonfly emergence, with subsequent consequences on native pollinators and terrestrial plant reproduction. In the Sierra Nevada, Epanchin et al. (2010) linked introduced fish to dramatic decreases in mayfly emergence, with indirect consequences for feeding Gray-crowned Rosy-Finches (*Leucosticte tephrocotis dawsoni*). These studies show that the effects of introduced trout permeate beyond the lake boundary and have ramifications for neighboring terrestrial ecosystems.

Recent policy changes have been implemented in California by the California Department of Fish and Wildlife (CDFW, the agency responsible for fish stocking) regarding fish stocking in water bodies where sensitive species occur or may occur. Prompted by a 2006 lawsuit, the California Superior Court ruled in 2008 that the CDFW must consider the effects of hatchery operations and fish stocking on sensitive aquatic species when making stocking decisions (Pacific Rivers Council & Center for Biological Diversity v. California Department of Fish and Game. 2007. Case number 06 CS 01451, California Superior Court of Sacramento County). Regardless of current stocking practices, introduced fish are nearly ubiquitous in Sierra Nevada lakes and are likely to remain so unless management actions are undertaken to remove them from key water bodies. For example, Armstrong and Knapp (2004) found that lakes in the John Muir Wilderness with >2.1 m of spawning gravels that were <3520 m in elevation nearly always showed signs of supporting self-sustaining populations of rainbow and golden trout.

The stocking of fish in aquatic water bodies represents a management action over which participating agencies have the ability to exert both direct and indirect controls. Because stocking has occurred throughout the Sierra Nevada and southern Cascades, actions taken on this factor have the potential to be far reaching. Further, the rapid recovery of native frog populations and other native species following fish removal in lentic systems (Vredenburg 2004, Knapp et al. 2007, Pope 2008, Knapp et al. 2001) indicates that fish removals have the potential to be effective restoration tools. Both CDFW and the National Park Service have integrated strategic fish removal projects into their resource management plans, and the Forest Service is implementing fish removal projects in the Sierra Nevada (e.g., Desolation Wilderness, El Dorado National Forest). Projects to remove fish from Sierra Nevada lakes have primarily used non-chemical methods, such as setting gill nets and electrofishing in inlets and outlets, to reduce impacts to non-target organisms.

Research to understand the mechanisms of recovery following fish removals can help managers determine characteristics of lakes and amphibian populations best suited for recovery. For example, Pope (2008) found that the high reproductive output of Cascades frogs may allow rapid population growth, even with only a small number of breeding-aged frogs onsite. Knapp et al. (2007) found that following rapid population increases, mountain yellow-legged frogs disperse to neighboring suitable habitat if it is available. Determining lake basins to focus on native amphibian restoration involves science and management working together to maximize the positive outcomes of restoration actions while maintaining recreational fisheries. Important basin characteristics include the presence of target amphibians and additional nearby suitable habitat, the ability to successfully eliminate introduced fish, and the level of use by anglers. Because it is extremely difficult to remove fish from streams without the use of toxicants (e.g., rotenone), lakes with natural fish barriers near their inlets and outlets are better targets than lakes without natural fish barriers (e.g. Knapp et al. 2007, Pope 2008).

Amphibian Disease

Although amphibians are susceptible to a wide array of diseases, one disease has emerged as the greatest conservation concern for amphibians in the Sierra Nevada (and the world). Chytridiomycosis, an amphibian-specific fungal disease caused by *Batrachochytrium dendrobatidis* (*Bd*), has been implicated in declines and extinctions of amphibian populations worldwide (e.g., Bosch et al. 2001, Lips et al. 2004, Muths et al. 2003), and has contributed to widespread declines of mountain yellow-legged frogs throughout the Sierra Nevada (Rachowicz et al. 2006, Briggs et al. 2010, Vredenburg et al. 2010). Mass die-offs of frogs and population extirpations have been observed soon after the arrival of *Bd* in mountain yellow-legged frog populations (Vredenburg et al. 2010). Although many populations have been driven to extinction by *Bd*, some populations have survived the population crash and persist with the disease (Briggs et al. 2010, Knapp et al. 2011). Research is ongoing to determine factors that allow some populations to persist while others go extinct (Knapp et al. 2011). One important finding is that large populations have a higher likelihood of persisting with the disease (Knapp et al. 2011). This finding suggests that although *Bd* can be devastating to populations even where fish have been removed, these “recovered” frog populations have a better chance of persisting in the long term than those facing the additional stressor of introduced fish.

Options to help ameliorate the impacts of *Bd* include developing protected populations and prophylactic or remedial disease treatment (Woodhams et al. 2011). Because sustainable conservation in the wild is dependent on long-term population persistence, successful disease mitigation would include managing already infected populations by decreasing pathogenicity and host susceptibility so that co-evolution with those potentially lethal pathogens can occur (Woodhams et al. 2011). Currently, researchers are working to identify mechanisms of disease suppression and develop adaptive management strategies to be tested in field trials with natural populations. Antimicrobial skin peptides and microbes that inhibit infection by *Bd* have been identified from the skin of mountain yellow-legged frogs and may be useful tools for increasing the frog’s resistance to *Bd* (Woodhams et al. 2007, Harris et al. 2009).

Pollution

Exposure to pesticides transported to the Sierra Nevada by prevailing winds out of California’s Central Valley has been hypothesized as a cause of population declines of native amphibians, primarily along the west slope of the Sierra Nevada (Davidson 2004, Fellers et al. 2004). Pesticide residues from Central Valley agricultural areas have been found in samples of air, snow, surface water, lake sediments, amphibians, and fish across the Sierra Nevada (Cory et. al 1970, McConnell et al. 1998, Fellers et al. 2004, Hageman et al. 2006), and windborne contaminants have been linked to patterns of decline of mountain yellow-legged frogs (Davidson and Knapp 2007). However, a recent study that compared concentrations of historically and currently used pesticides with the population status of mountain yellow-legged frogs in the southern Sierra Nevada found no association between frog population status and measured pesticide concentrations (Bradford et al. 2011). In addition, in both the Sierra Nevada and southern Cascades, pesticide concentrations in water and amphibian tissue were consistently below concentrations found to be toxic to amphibians (Bradford et al. 2011, Davidson et al. 2012). Low concentrations of pesticides could, however, interact with other stressors and contribute to adverse effects on frogs. For example, the pesticide carbaryl was found to reduce production of amphibian skin peptides that inhibit the growth of (*Bd*), the fungus that causes chytridiomycosis (Davidson et al. 2007). For practical reasons, most field studies continue to focus on single stressors; however, the few studies assessing interactive effects of low-level contaminants with other stressors highlight the need for additional multifactor studies.

Metrics for Lake Assessments

A metric using the ratio of the number of taxa observed (O) at a site to that expected (E) to occur at the site in the absence of anthropogenic impacts (abbreviated as O/E) has been applied to lakes with fish in the Sierra Nevada (Knapp et al. 2005). The authors found that amphibians, reptiles, benthic invertebrates, and zooplankton have relatively low resistance to fish introductions, but all taxa recover when lakes revert to a fishless condition. This metric proved effective at assessing resistance and resilience of expected native taxa in Sierra Nevada lakes to one threat (introduced fish), and it may be

Recent and Pending Studies on Frogs and Chytrid Disease

Pending research may suggest additional considerations to help frogs withstand the novel threat posed by the fungal disease chytridiomycosis, which is caused by *Batrachochytrium dendrobatidis* (*Bd*). For example, a recent lab study comparing the effects of *Bd* isolated from two localities in northern California found dramatic differences in the virulence of the isolates on Cascades frogs (*Rana cascadae*), with one isolate causing nearly complete mortality in test animals within six weeks and the other causing a mortality rate only slightly higher than that of unexposed animals (Piovia-Scott et al., in prep). This result suggests that in addition to environmental conditions and local host population size, the local strain of *Bd* may also play an important role in determining host population dynamics.

Furthermore, preliminary field testing of supplementing microbes on the skin of mountain yellow-legged frogs have shown promising results in boosting survival with the disease (Vredenburg et al., unpublished data).

A recent study by the Pacific Southwest Research Station and UC Davis collaborators found that all the remnant populations of Cascades frogs in the southern Cascades survive with *Bd* and all occur in habitats where adult frogs commonly move among different water bodies for breeding, summer feeding, and overwintering. When the frogs are in the breeding sites, they tend to have high prevalence and loads of *Bd*, but when they move to streams and channels, loads of *Bd* are dramatically reduced or eliminated (Pope et al., in prep). One hypothesis is that the movement away from breeding sites where the disease may thrive allows the frogs to behaviorally eliminate the disease, and thus, the population is able to persist. If this is true, restoration of lake basins with additional stream and meadow habitats in close proximity may be most effective where *Bd* is a concern.

effective at assessing other threats. Since then, Van Sickle (2009) has suggested an alternative statistical approach that could be more sensitive to anthropogenic stressors; this approach uses BC, an adaptation of Bray-Curtis distance, instead of O/E to assess compositional dissimilarity between an observed and expected assemblage.

Although metrics based upon species composition may be useful in assessing the overall condition of the aquatic biota, priority will likely be given to restoration of lakes in the Sierra Nevada and southern Cascades to support populations of amphibian species, such as the Sierra Nevada and southern mountain yellow-legged frogs and the Cascades frog. Recent research has shown that successful restorations will likely depend on the elimination of introduced fish, whether or not *Bd* is present and how virulent it is if it is present, and habitat conditions that help frogs to withstand the effects of climate change and disease.

Management Implications

- Removal of introduced fish is an important strategy to help priority amphibians species withstand combined stressors including climate change and disease.
- Chytridiomycosis continues to be a major conservation concern for frogs in mountain lake ecosystems. Researchers are working to develop treatment options to help infected populations.

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7.0 Terrestrial Wildlife

A wide range of terrestrial wildlife species inhabit the synthesis area. This section of the report focuses on three of those species: two forest carnivores, the fisher (*Martes pennanti*) and the Pacific marten (*Martes caurina*)—recently split from the American marten (*Martes americana*)—and one raptor, the California spotted owl (*Strix occidentalis occidentalis*). The Forest Carnivores chapter (7.1) describes the ecology and context of fisher and marten, summarizing population trends, identifying threats, and highlighting science-based implications for the management of their habitats. The California Spotted Owl chapter (7.2) describes the ecological context, population trends, and habitat needs of this top predator, highlighting recent findings on the effects of forest management, wildfire, and other important ecological stressors. Several aquatic species of concern to management, including native trouts and amphibians, are addressed in the chapters of the Water Resources and Aquatic Ecosystems section (6.1 and 6.5).

The species in this section are not the only ones that could be affected by management, but they have been priorities for conservation in the Sierra Nevada because of ecological and social factors. They are important values at risk because of their roles as predators, relative scarcity, declining populations over past decades, and association with large trees, dead trees, and other characteristics found in old forests. These three species have been featured in previous synthesis reports for the Sierra Nevada (PSW General Technical Reports 220 and 237). They use large areas and have special habitat requirements and designations, which complicate landscape-scale management and restoration in the Sierra Nevada and southern Cascades. They have also been designated as Forest Service Sensitive Species (and Pacific fisher has been identified as “warranted” for listing under the Endangered Species Act). Therefore, they are likely to be a focus of further monitoring and analysis under the “fine filter” approach outlined in the planning rule to address species of concern. Many other species, including wolverine (*Gulo gul*o) and the Sierra Nevada red fox (*Vulpes vulpes necator*), might be prioritized based on similar criteria. A much broader analysis would be needed to systematically review species that may be appropriate as fine filter indicators. From a resilience perspective, and especially in light of climate change, it is important to consider how future fire severity and size may affect biodiversity, especially species that may be positively or negatively associated with high severity burn patches of different sizes (see Integrative Approaches chapter (1.1) and Post-wildfire Management chapter (4.3)).

The three species featured in this section provide useful focal points for a discussion of strategies to promote long-term, large-scale socioecological resilience in the synthesis area. Losses of biodiversity, declines of predators, and other changes to trophic webs are an important threat to ecological resilience (see Integrative Approaches chapter (1.1)). However, the Synopsis of Emergent Approaches (1.2) cautions that emphasizing needs of a few species can pose challenges to long-term, landscape-scale restoration efforts for a broader range of values, including overall native biodiversity. Consequently, the synthesis emphasizes the importance of weaving habitat requirements of these three species into a broader landscape and long-term perspective for forest planning.

7.1 The Forest Carnivores: Fisher and Marten

William J. Zielinski

Contents

Executive Summary

Martens and fishers, as predators, perform important functions to sustain the integrity of ecosystems. Both species occur primarily in mature forest environments that are characterized by dense canopy, large diameter trees, a diverse understory community, and abundant standing and downed dead trees. Martens occur in the upper montane forests, where the threat of wildfire is less, and fishers predominately occur in the lower montane forests, where the threat of severe wildfire is much greater. Both species use habitat at multiple scales: the resting/denning site, the stand, the home range, and the landscape. Thus, management may benefit from considering these components when evaluating the effects of treatment activities. Their diets are relatively diverse and their prey occurs in a variety of habitats within their home ranges. It appears that the heterogeneous conditions that are predicted to occur with the restoration of fire as a disturbance process may produce habitat for martens, fishers, and their diverse sources of food. New science, however, will be needed to test this assumption. Mechanical thinning may mimic some aspects of the disturbance ideally caused by fire, but this alternative would appear to be more justified—based on departures from fire return intervals—in the lower montane forests where fishers occur than in the upper montane forests where martens occur. During the course of mechanical treatments, however, attention should be paid to restoring or maintaining the distributions of large trees, conifers, and—for the fisher—black oaks, as well as sufficient understory habitat for both species. Management to reduce fire risk, or to restore ecological resilience to fire, may be consistent with the maintenance of landscapes capable of supporting fishers, as long as sufficient resting/denning structures are retained and the composition and configuration of the residual landscape is compatible with home range requirements. New scientific tools have been developed in the last 10 years to help evaluate the effects of proposed treatments on habitat features for fishers, in particular, but these have not yet been used in a coordinated manner to evaluate effects at multiple scales (resting/denning, home range, and landscape). Similar tools need to be developed to evaluate marten habitat. This review identifies the threats to each species in the Sierra Nevada and outlines the science available to assist managers in dealing with these threats. Our knowledge base is far from complete, however, which is why monitoring fisher and marten populations and their habitats should be a centerpiece of their management. Monitoring, together with new ideas about adaptive management (especially for the effects of treatments on fishers and their habitat), will be the key to ensuring that implementation of an ecosystem management scheme for the forests of the Sierra Nevada will benefit long-terms goals for marten and fisher conservation.

The Important Role of Predators in Ecosystems

Predators have important effects on the structure of biological communities, primarily because the act of killing and eating other species transfers energy and nutrients through the ecosystem. However, they also have important indirect effects on community structure because their consumption of prey—and their simple presence—affect the distribution of herbivores, which, in turn, affect the abundance and distribution of plants. These “trophic cascades” have been most clearly demonstrated by research on the effects of the return of wolves to the Yellowstone ecosystem (Ripple and Beschta 2004). The retention of predators in an ecosystem is, therefore, integral to the maintenance of biodiversity and, hence, the resilience and proper functioning of that system (Finke and Denno 2004, Finke and Snyder 2010). In addition, when larger predators are absent, intermediate-sized (meso) predators can increase to the point that they have destabilizing effects on their prey, a phenomenon referred to as “mesopredator release” (Crooks and Soulé 1999, Roemer et al. 2009). The “top down” control of community structure by predators, which has been demonstrated on every continent and ocean, is primarily why researchers work to understand the ecology of montane mammalian predators and to collect information that will help ensure their persistence in the face of environmental change. California has already lost some key mammalian predators (e.g., grizzly bear [*Ursus arctos*], gray wolf [*Canis lupus*]) and some species are so rare that they no longer play effective ecological roles (i.e., wolverine [*Gulo gulo*], Sierra Nevada red fox [*Vulpes vulpes necator*]). Thus, the disproportionately important functional roles of mammalian predators are already reduced in montane forest ecosystems in California, and elsewhere in North America (Laliberte and Ripple 2004). Using research results from studies on carnivores to inform management will help ensure that additional predators, especially the two species of forest mesocarnivores, the fisher (*Martes pennanti*) and the Pacific marten (*M. caurina*), are not lost from our forest ecosystems and continue their important roles throughout their geographic ranges.

This review considers the relevant science necessary to assist in the management of the fisher and Pacific marten. These are not the only species that could be affected by management, but they are the species currently generating the majority of policy and conservation interest in the Sierra Nevada. A case could be made for the inclusion of the wolverine (*Gulo gulo*) and the Sierra Nevada red fox (*Vulpes vulpes necator*), species that have generated recent interest (Moriarty et al. 2011, Perrine et al. 2010), but given time and space limitations, this review is focused only on the fisher and the marten.

The Fisher and the Marten in Context

The fisher, a medium-sized member of the family Mustelidae, is the largest of the nine species in the genus *Martes* (Aubry et al. 2012). There is a single species of fisher and it occurs only in North America. Its dark brown, glossy fur often looks black. Fishers have white or cream patches on the chest and around the genitals, and the head and shoulders are often grizzled with gold or silver (Douglas and Strickland 1987). The conical shape of the tail, thicker near the body and tapering to a thinner tip, distinguishes the silhouette of the fisher from that of the Pacific marten, *M. caurina*; fishers also have relatively smaller ears than martens, and lack the marten’s yellow or orange gular or ventral patches. There is a single subspecies of fisher in California, the Pacific fisher (*M. pennanti pacifica*) (Figure 1).

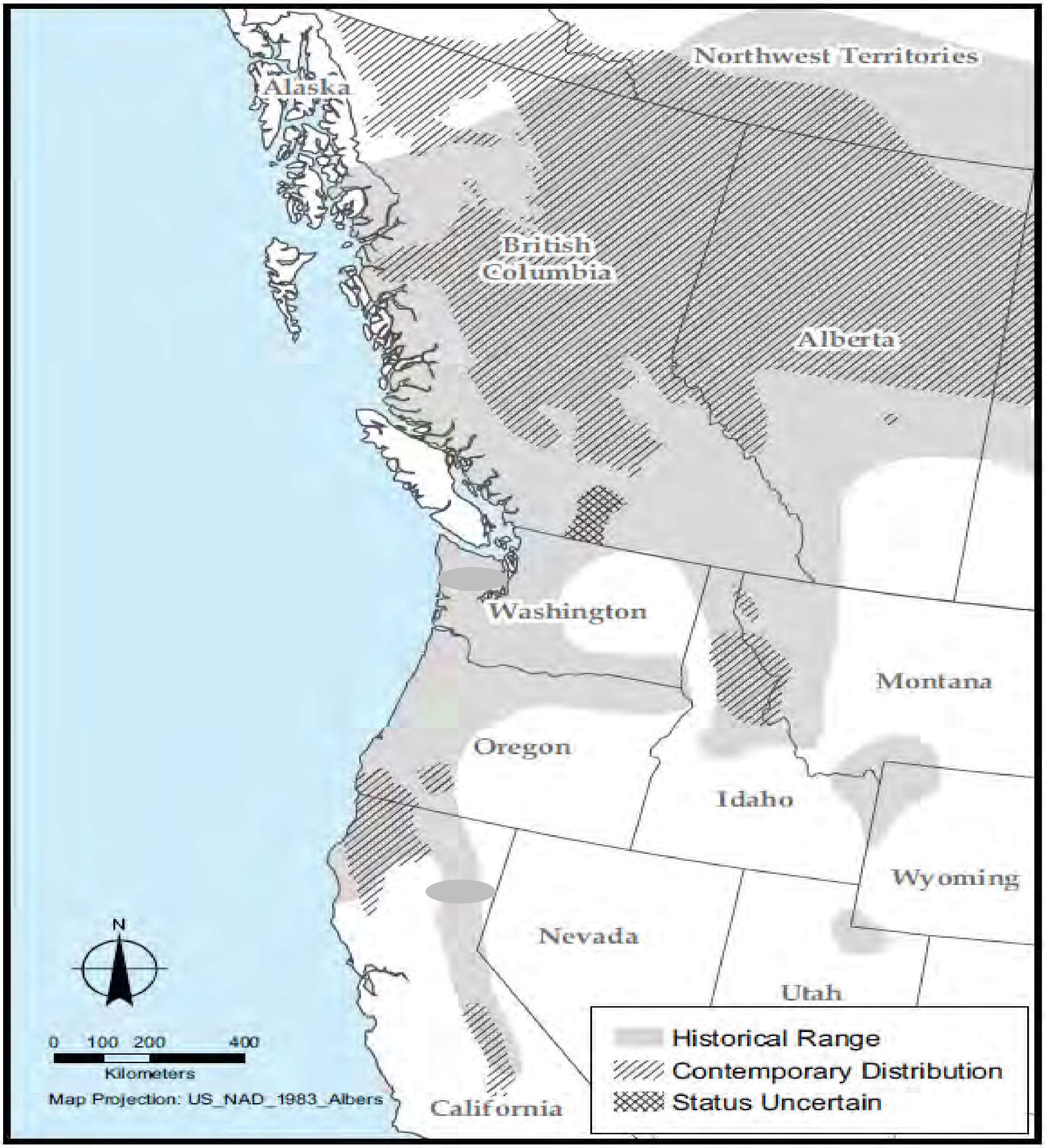

Figure 1: Fisher geographic range in western North American (adapted from Lofroth et al. 2010). Gray ellipses, in the Olympic Peninsula and in the northern Sierra, refer to the locations of recent reintroductions.

There are 14 subspecies of marten recognized by Hall (1981) (Figure 2). Recent genetic and morphological evidence has warranted splitting the American marten into two species: the American marten east of the Rocky Mountain crest (*Martes americana*) and the Pacific marten (*M. caurina*) west of the crest (Dawson and Cook 2012) (Figure 3). Two subspecies of the Pacific marten are recognized in California: the Sierra marten (*M. c. sierrae*) in the Sierra Nevada, Cascades, and Klamath/Trinity mountains, and the Humboldt marten (*M. c. humboldtensis*) in the redwood zone along the north coast. For the purposes of this review, I hereafter refer to all martens in California as the Pacific marten (*M. caurina*), with most of the focus on the Sierran subspecies (*M. c. sierrae*).

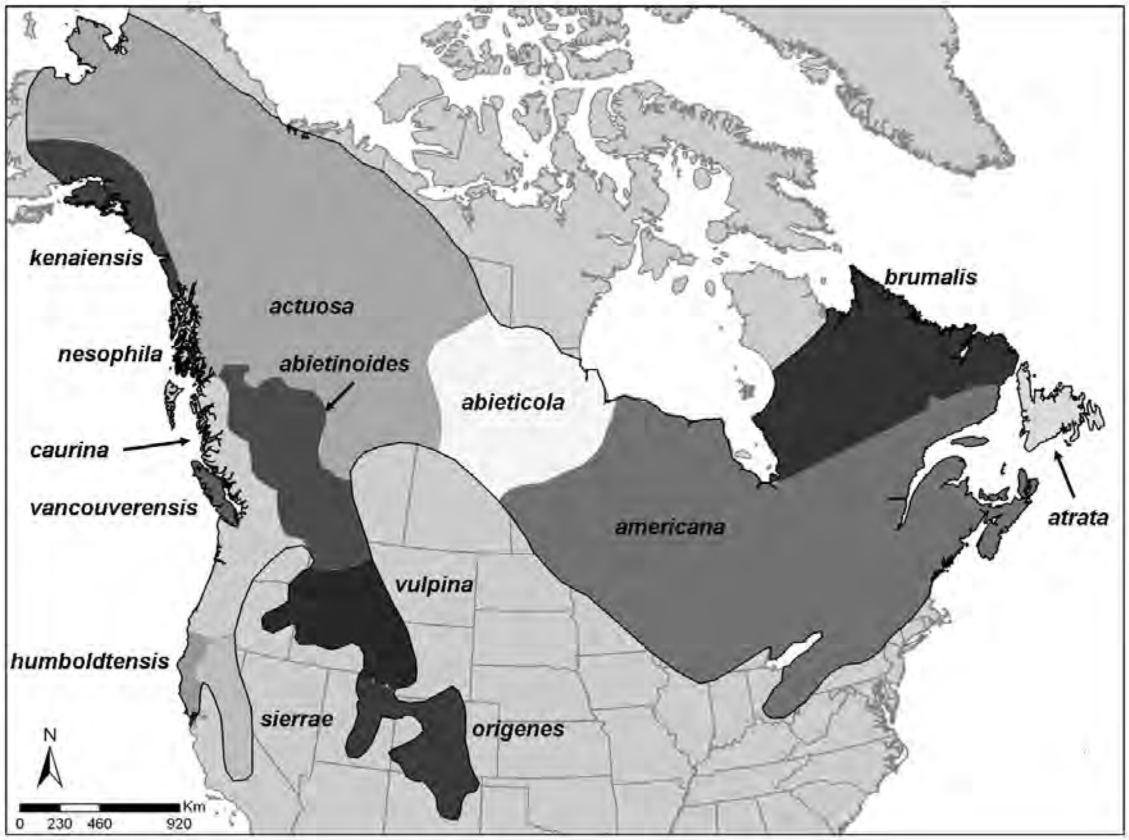

Figure 2: The subspecies of marten in North America (from Dawson and Cook 2012, Reprinted from Biology and Conservation of Martens, Sables and Fishers: A New Synthesis, edited by Keith B. Aubry, William J. Zielinski, Martin G. Raphael, Gilbert Proulx, and Steven W. Buskirk. Copyright © 2012 by Cornell University. Used by permission of the publisher, Cornell University Press.).

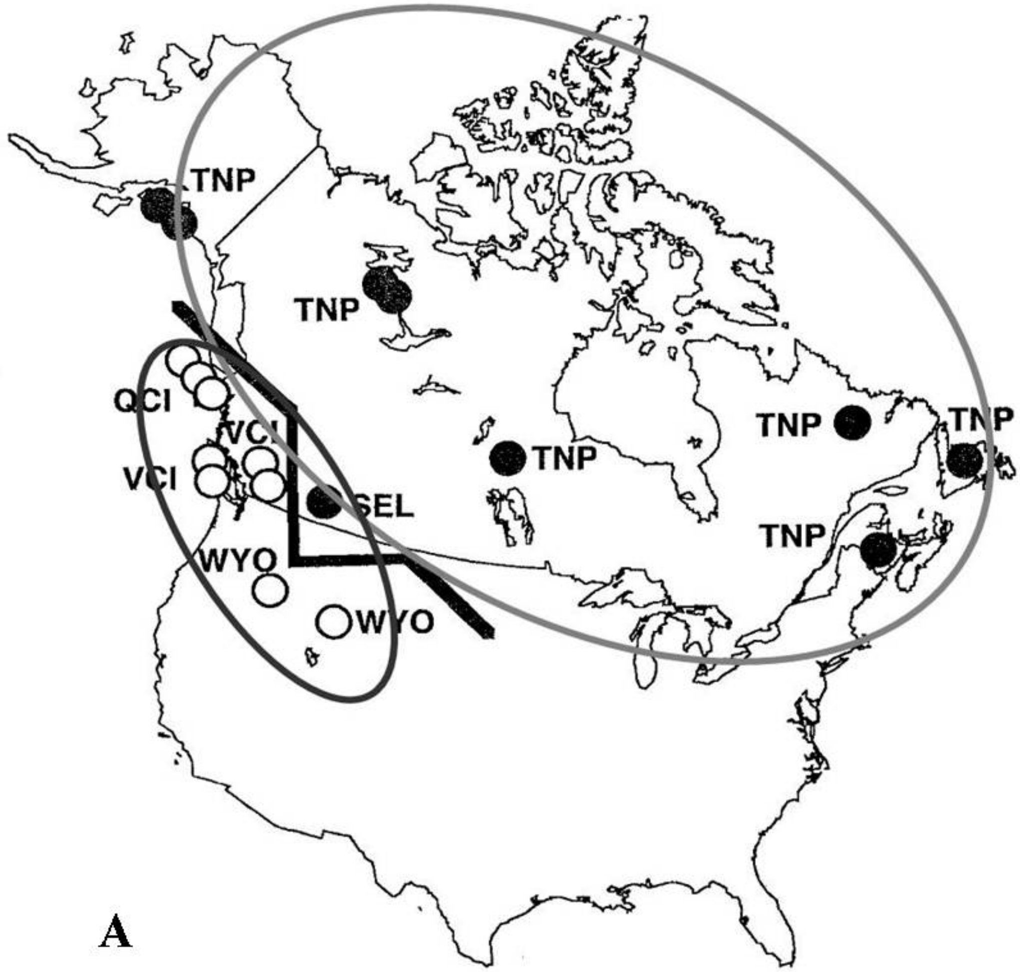

Figure 3: The geographic boundary between *M. americana* (dark circles enclosed by light grey ellipse) and *M.caurina* (open circles enclosed by dark gray ellipse) based on cytochrome *b* gene sequence data. Although not depicted, the martens in California are within the new species, the Pacific marten (*M. caurina*). Illustration from Dawson and Cook (2012), adapted from Carr and Hicks (1997). Reprinted from *Biology and Conservation of Martens, Sables and Fishers: A New Synthesis*, edited by Keith B. Aubry, William J. Zielinski, Martin G. Raphael, Gilbert Proulx, and Steven W. Buskirk. Copyright © 2012 by Cornell University. Used by permission of the publisher, Cornell University Press.

Marten Ecology

Martens are generally associated with late-successional conifer forests (Powell et al. 2003) characterized by an abundance of large dead and downed wood, and large, decadent live trees and snags (Buskirk and Ruggiero 1994), especially in boreal and montane forests of western North American (Thompson et al. 2012). The marten distribution overlaps the fisher distribution slightly in the Sierra Nevada but extends to much higher elevation (~4,500 – 10,500 ft) red fir and lodgepole pine forests. Martens are extremely sensitive to the loss and fragmentation of mature forest habitat and rarely occupy landscapes after >30 percent of the mature forest has been harvested (Bissonette et al. 1997, Chapin et al. 1998, Hargis et al. 1999, Potvin et al. 2000). Home ranges of Pacific martens in the Sierra Nevada average 740 – 1235 acres for males and 740 – 990 acres for females (Simon 1980, Spencer et al. 1983). The physical structure of forests appears to be more important to marten habitat quality than plant species composition (Buskirk and Powell 1994). Martens require abundant large trees and dead-wood structures to provide prey resources, resting structures, and escape cover to avoid predators. How these elements are provisioned over space and time in a manner that permits martens to persist is unknown. However, as discussed in the Integrative Approaches chapter (1.1)—and foreshadowed in North et al. (2009)—restoring forests to conditions where natural disturbances can affect vegetation structure and composition may likely provide sufficient habitat for martens, fishers, and other species that have evolved with periodic disturbance, especially low-intensity fire.

In the Sierra Nevada and Cascades of California, martens are associated with late-successional forests dominated by true fir (*Abies* spp.) and lodgepole pine forests and are most abundant where old-growth forest characteristics are abundant (Spencer et al. 1983, Ellis 1998). Riparian zones, especially near mature forests, are important foraging areas (Spencer et al. 1983, Zielinski et al. 1983, Martin 1987). Other than this, little is known about the effect of topography on the distribution of marten habitat or marten behavior. Resting sites used by Sierran martens differ with the season. Above-ground cavities in the largest diameter trees and snags are primarily used during the summer, whereas subnivean logs, snags, and stumps are typically selected for resting during the winter (Spencer 1987, Martin and Barrett 1991). Martens can inhabit younger or managed forests as long as some of the structural elements found in older forests remain, particularly those required for resting and denning (Baker 1992, Porter et al. 2005, Thompson et al. 2012). On the Lassen National Forest, male martens preferentially used open shelterwood stands during the summer, when chipmunks and ground squirrels were available in these relatively open areas; however, females showed strong year-round selection for old-growth stands (uncut, large-tree stands with tree cover > 69 percent; Ellis 1998). The size of openings that martens will cross in the Sierra Nevada or Cascades is currently under study (K. Moriarty, pers. comm.). However, in the Rocky Mountains, the average width of clear cuts (openings) crossed by martens was 460 ft; this distance is significantly less than the average width of clear cut openings that martens encountered but did not cross (average = 1,050 ft) (Heinemeyer 2002). Moreover, martens were more likely to cross larger openings (max distance = 600 ft.) that had some structures in them (i.e., isolated trees, snags, logs) than smaller openings (average distance = 160 ft) that had no structures (Heinemeyer 2002). Cushman et al. (2011) reported that snow-tracked martens in Wyoming strongly avoided openings and did not venture more than 55 ft from a forest edge.

Martens are dietary generalists, although their diet changes with seasonal prey availability, and during particular seasons they may specialize on a few specific prey species (Zielinski et al. 1983, Martin 1994). The diet is dominated by mammals, but birds, insects, and fruits are seasonally important (Martin 1994). The diet of the marten in the Sierra changes with season, as does the time of day that martens search for particular prey (Zielinski et al. 1983, Martin 1987). Winter prey is primarily Douglas squirrel (*Tamiasciurus douglasii*), snowshoe hare, voles (*Microtus* sp.), and flying squirrels (*Glaucomys sabrinus*). In the summer, the diet switches to include ground-dwelling sciurids, and voles continue to be important prey (Zielinski et al. 1983). Several of the key prey species reach their highest densities in forest stands with old-growth structural features (e.g., red-backed vole (*Clethrionomys californicus*), Hayes and Cross 1987, Zabel and Waters 1992; flying squirrel (*Glaucomys sabrinus*), Waters and Zabel 1995; Douglas squirrel, Carey 1991).

Pacific martens typically occur in forested regions that receive considerable snowfall, and they are well-adapted to these conditions. They have relatively low foot loading (i.e., high foot surface area to body mass ratio), which allow them to move relatively easily over deep, soft snow, and they are adept at using subnivean environments for foraging and resting. This gives martens a competitive advantage over larger carnivores that may otherwise compete with or prey on martens, such as bobcats, coyotes, and fishers, whose distributions are limited by deep, soft snow (Krohn et al. 1997, Krohn et al. 2004).

Fisher Ecology

In western North America, fishers are associated with late-successional conifer or mixed-conifer-hardwood forests characterized by an abundance of dead and downed wood, dense canopy, and large trees (Buskirk and Powell 1994, Zielinski et al. 2004a, Purcell et al. 2009, Lofroth et al. 2010, Raley et al. 2012). Fishers occur in a variety of low and mid-elevation forests (primarily the ponderosa pine and mixed-conifer types) where canopy is moderate to dense, but the vegetation types comprising the home range can be heterogeneous (Lofroth et al. 2010, Thompson et al. 2011). In the Sierra Nevada, fishers occur primarily in mixed-conifer and ponderosa pine forests from 3,500-7,000 ft., elevations that do not typically receive deep and persistent snow, which is thought to restrict their movements (Krohn et al. 1995, 1997). Powell and Zielinski (1994) hypothesized that forest structure was more important than tree species for fisher habitat. Complex structure, including a diversity of tree sizes, snags, downed trees and limbs, and understory vegetation, provides den and rest sites and hiding cover for fishers, as well as habitat for their prey. Both inactive (resting and denning) and active (foraging) fishers are typically associated with complex forest structure (Lofroth et al. 2010, Zhao et al. 2012).

Fishers forage in a manner that suggests that they use habitat at four scales: the resting site, the stand, the home range, and the landscape. Resting and denning (i.e., parturition and neonatal care) typically occur in trees, snags, and logs that are in the largest diameter classes (Lofroth et al. 2010, Raley et al. 2012, Aubry et al. 2012) and are either deformed or in some form of decay (Weir et al. 2012). For example, the average diameters at breast height (dbh) of conifer and hardwood rest trees in one study in the Sierra Nevada were 43 and 26 in, respectively (Zielinski et al. 2004a). In an innovative new study using LIDAR to characterize the vegetation structure surrounding den trees, Zhao et al. (2012) found that tree height and slope were important variables in classifying the area immediately surrounding denning trees. At scales larger than 65 ft., forest structure and complexity became more important. The variables identified using LIDAR were consistent with those identified from previous studies describing fisher resting structures.

The strong association of fishers with dense forest stands that contain a diversity of tree sizes complicates the ability of managers to achieve what seem like mutually exclusive goals: the reduction of stand densities and fuels, and the maintenance of fisher habitat. The basal area of small-diameter trees is an important predictor of fisher resting sites (Zielinski et al. 2004a). The smaller trees may provide the requisite canopy cover needed by fishers, as long as a suitably large resting structure (tree or snag) is also available (Purcell et al. 2009). Some researchers have speculated that the dense forest conditions that appear attractive to fishers today may be an artifact of past logging practices and fire suppression. These factors may have changed forest conditions from stands dominated by large trees and snags to dense stands with size class distributions that included more small-diameter trees (Scholl and Taylor 2010, Collins et al. 2011). Topography affects the distribution of dense forests and the effect of fire severity (North et al. 2009), as well as the distribution of fisher resting sites. Underwood et al. (2010) found that fisher activity locations were disproportionately found in lower topographic positions (i.e., canyons), as well as in southerly and northerly mid-slope positions.

As generalized predators, fishers prey on a variety of small and medium-sized mammals and birds, and they also feed on carrion (Powell 1993, Martin 1994). In California, reptiles and insects are also notable components of the diet (Zielinski et al. 1999, Golightly et al. 2006). Home range size appears to be a function of the abundance of food, in that fishers whose diet includes a significant component of relatively large (>400 g) food items (e.g., woodrat [*Neotoma* sp.] and western gray squirrel [*Sciurus griseus*]) have significantly smaller home ranges (Slauson and Zielinski, unpubl. data).

Predation is probably the predominant cause of death, and fishers are regularly killed by cougars (*Puma concolor*), coyotes (*Canis latrans*) and bobcats (*Lynx rufus*) (Lofroth et al. 2010). Fishers are also affected by viral and parasitic diseases, such as canine distemper, parvovirus, and toxoplasmosis, and they are victims of poison distributed to control rodents (Brown et al. 2006, Gabriel et al. 2012a, 2012b).

Population Status

Marten Populations

Martens were legally trapped for fur in California until 1954, and the earliest summary of the trapping records indicated that the marten was well distributed across its native range in the early 1900s (Grinnell et al. 1937). However, declining numbers due to intense trapping pressure during this period resulted in the prohibition of trapping. Before and during this period, California's primary forests were heavily harvested (McKelvey and Johnston 1992, Bolsinger and Waddell 1993, Franklin and Fites-Kaufmann 1996), adding to the pressure on marten populations.

Recent research has focused on the distributional dynamics of the marten in the Sierra Nevada and Cascades. Concern about the decreasing distribution of martens in some regions of the Pacific states has been voiced for decades (Dixon 1925, Kucera et al. 1995, Zielinski et al. 2001). Historical and contemporary distributions in the Sierra and Cascades were compared by contrasting the locations of animals trapped for their fur in the early 1900s (Grinnell et al. 1937) with surveys recently conducted (Zielinski et al. 2005) using noninvasive methods (track stations and cameras, Long et al. 2008). These surveys revealed changes in the distributions of a number of carnivore species, including the marten, fisher, wolverine, and Sierra Nevada red fox. Historically, the marten was reported to occur throughout the upper montane regions of the Cascades and northern Sierra Nevada, but survey results indicate that populations are now reduced in distribution and fragmented (Zielinski et al. 2005).

Change in habitat distribution in the Cascades and northern Sierra Nevada has also been demonstrated using predictive habitat modeling. The results of surveys were used to build landscape habitat suitability models by contrasting the environmental features at places where martens were detected with the features at places where they were not. This work confirmed that the available habitat for martens is isolated and has been reduced in area since the early 1900s (Kirk and Zielinski 2009). The model that best fit the data suggested that remaining marten populations are associated with sites with the largest amount of reproductive habitat (dense, old forest), the greatest number of nearby habitat patches, and nearby reserved land (land protected from timber harvest). The highest density of detections was located in the largest protected area in the study region: Lassen Volcanic National Park. This is in stark contrast to descriptions of historical distribution, which described martens as evenly distributed in the region during the early 1900s, including at lower elevation sites (Grinnell et al. 1937, Zielinski et al. 2005).

The loss of marten distribution seems clear, but what is less clear is the cause. This is difficult to ascertain with data from such a large region; what is necessary instead is a more focused effort to contrast areas that have maintained their marten populations compared to those that have not. Recent work in the Sagehen Experimental Forest (SEF), within the Sagehen Creek watershed on the Tahoe National Forest, may be helpful in this regard (Moriarty et al. 2011). The watershed has been the location for a number of studies on marten ecology in California, but the unique element of this body of work is that most of these studies also included a systematic survey of marten occurrence. Martens were assumed to be very common in the watershed in the 1970s and 1980s, but anecdotal observations have suggested that they subsequently became quite rare. This concern led to new surveys in 2007-2008, which detected no martens during summer surveys (June – September), despite the fact that martens were regularly detected during the summer in earlier surveys (Moriarty et al. 2011). A few marten detections occurred in the winter, but these were in a small western portion of the watershed. Marten detections in 2007–2008 were approximately 60 percent fewer than in surveys in the 1980s. Thus, at the scale of the Sagehen Creek watershed, the same phenomenon was observed that was described for the northern Sierra/southern Cascades as a whole (Zielinski et al. 2005).

The cause of the decrease in marten numbers at Sagehen is uncertain. However, GIS analysis comparing older vegetation maps from 1978 with maps from 2007 revealed a loss and fragmentation of important marten habitat (Moriarty et al. 2011). This included a decrease in habitat patch size, core habitat area, and total amount of marten habitat in the study area, as well as an increase in distance between important habitat patches. For example, the mean area of patches of reproductive habitat decreased from 56.6 ha to 44.5 ha, and the mean distance between these patches increased from 194 to 240 m over the almost 30 year period (Moriarty et al. 2011). Many of these changes occurred between 1983-1990, when 39 percent of the forest habitat in SEF experienced some form of timber harvest (i.e., regeneration, selection, hazard tree removal). The sensitivity of martens to forest fragmentation is well established (e.g., Bissonette et al. 1989, Hargis et al. 1999, Potvin et al. 2000). Given this, and the fact that the loss of martens at Sagehen coincided with the period of greatest harvest activity in the study area, the loss and fragmentation of habitat is the most likely explanation for the decline at SEF (Moriarty et al. 2011). Collectively, the evidence from studies at Sagehen, and from the larger Cascades and northern Sierra Nevada region, supports the conclusion that the distribution of martens in this region has decreased. On the contrary, however, evidence from surveys in the central and southern Sierra Nevada (Kucera et al. 1995, Zielinski et al. 2005) suggests that the marten population is well distributed.

Fisher Populations

Following European settlement of North America, fisher range contracted drastically, particularly in the southern regions, due to deforestation and trapping (Powell 1993). In California, Grinnell et al. (1937) described the original range of the fisher as including the northern Coast Range, Klamath Mountains, southern Cascades, and the entire western slope of the Sierra Nevada. The status of the fisher in California has been of concern for almost 100 years, beginning with Dixon (1925), who believed that the fisher was close to extinction in California and proposed that protective measures be taken. As a consequence, trapping of fishers in California was prohibited in 1946—considerably later than was suggested by Dixon (1925). Population decline and fragmentation have reduced genetic diversity in California, particularly in the southern Sierra (Drew et al. 2003; Wisely et al. 2004; Tucker et al., in press), and the two native fisher populations in California are geographically and genetically isolated (Zielinski et al. 1995; Wisely et al. 2004; Knaus et al. 2011; Tucker et al., in press). Due, in part, to the genetic and population data available at the time, the U.S. Fish and Wildlife Service determined that the West Coast Distinct Population Segment (as defined by the Endangered Species Act [ESA]) in California, Oregon, and Washington was "warranted but precluded" for listing under the ESA (U.S. Federal Register, April 8, 2004).

Survey data indicate that fishers currently occur in two widely separated regions of the state: the northwest, including the northern Coast Range and Klamath Province, and the southern Sierra Nevada (Zielinski et al. 1995, Aubry and Lewis 2003). This creates a gap in their distribution of approximately 250 miles in the northern Sierra and southern Cascades, which has previously been attributed to the historical effects of trapping and timber harvest (Zielinski et al. 1995, 2005). Contrary to the conclusions of Grinnell et al. (1937: 215)—and an earlier report suggesting fishers were trapped in the gap region in the early 1900s (Grinnell et al. 1930)—new genetic analysis suggests that the two populations in California were separated prior to European influence in the region (Knaus et al. 2011; Tucker et al., in press). The genetic work does not estimate the size of the purported historical gap in distribution; it may have been similar to the apparent gap identified in Grinnell et al. (1937:216) or even smaller (M. Schwartz, pers. comm.). Thus, this new genetic information does not necessarily mean that fishers did not once occupy most of the Sierra Nevada and southern Cascades. Depending on one's view of the size of the historical gap between fisher populations in the Sierra Nevada, the fisher currently occupies anywhere from 20-90 percent of its historic range in California. If the gap predates European influence and was as large then as it is today, the current range of fishers in California would be about 90 percent of the pre-European historical range. If the gap was as small as a few fisher home-ranges wide, then the current range may be no more than 20 percent of the historical range. In 2009, a small population of fishers was reintroduced to Butte and Tehama Counties (Facka and Powell 2010) within the presumed gap in the distribution. There has been successful reproduction each year (A. Facka, pers. comm.), but it is too soon to calculate the contribution these animals will make to fisher populations in California.

The distribution of the fisher population in the southern Sierra Nevada has been monitored since 2002, and there has been no change in the proportion of detection stations with a fisher detection (i.e., occupancy); the population appears stable (Zielinski et al. 2012). Based on habitat and population modeling, the size of the southern Sierra Nevada population has been estimated to be between125 and 250 adults (Spencer et al. 2011). This is consistent with an estimate extrapolated from fisher density calculated by Jordan (2007) from a mark-recapture study.

Threats to the Species and Implications for Management

Marten - Threats

Timber harvest, vegetation management, and wildfire

Timber harvest and fur-trapping are regarded as the primary causes of reductions in marten populations in the western United States (Buskirk and Ruggiero 1994). Commercial fur trapping is the most direct way that humans have affected marten populations, but California prohibited marten trapping in 1954. Current threats include the continuing effects of habitat loss and fragmentation from timber harvest, particularly clear-cutting, and vegetation management to reduce fuels. Wildfire is a serious threat, as well, especially if exacerbated by climate change; these issues are discussed in a separate section below, entitled “Climate Change Implications – Marten and Fisher.”

Given the rarity of clear-cutting on public lands in California, the largest potential direct threat from human activities is the effect of forest thinning. Abundant literature notes the sensitivity of martens to the effects of forest fragmentation (Bissonette et al. 1997, Chapin et al. 1998, Hargis et al. 1999, Potvin et al. 2000), but in these cases, the fragmentation is typically due to regeneration or clear-cut harvests. How thinning treatments fragment habitat is poorly known, but it is under study in the Cascades in California (K. Slauson, unpubl. data; K. Moriarty, unpubl. data). Although we have little local information on the effects of thinning on martens, Fuller and Harrison (2005) evaluated how partial harvests affect martens in Maine and summarized the few data on this subject that predated their study. Partial harvests (also referred to as “partial overstory removal” in the eastern US) leave residual forest cover in harvest blocks. In Fuller and Harrison’s (2005) study area, 52-59 percent of the basal area was removed in partial harvests. In these conditions, martens used the partial harvest stands primarily during the summer. When they were using partial harvest stands, their home ranges were larger, indicating poorer habitat quality in these areas. Partial harvested areas were avoided during the winter, presumably because they provided less overhead cover and protection from predators. How this work relates to predicting the effects of thinning in marten habitat in the Sierra Nevada is unclear, but the most conservative generalization would suggest that martens would associate with the most dense residual areas in thinned units and may also increase their home ranges, which may lead to decreased population density. The negative effects of thinning probably result from reducing overhead cover. Thinnings from below, which retain overstory cover, probably have the least impact on marten habitat, provided they retain sufficient ground cover. Downed woody debris provides important foraging habitat for martens. Andruskiw et al. (2008) found that physical complexity on or near the forest floor, which is typically provided by coarse woody debris, is directly related to predation success for martens; when this complexity is reduced by timber harvest (a combination of clear-cut and selection harvests with subsequent site preparation in their study area), predation success declines. Marten home ranges in uncut forests had 30 percent more coarse woody debris (> 10 cm diameter) from all decay classes combined than in cut forests (Andruskiw et al. 2008). Retaining sufficient understory vegetation and downed wood, which are necessary marten habitat elements, will be a challenge in applying fuels treatments that are meant to reduce the density of surface fuels.

Recent genetic work in Ontario finds that forests managed for commercial value (via clear-cutting) appear to be sufficiently connected to maintain gene flow, at least at the level of the province (Koen et al. 2012). This would appear to be the case in landscapes like those planned for the Sierra Nevada forests, where thinning is the dominant silvicultural treatment, because the impact on canopy is much less. Thus, it appears possible for gene flow to be maintained in commercial forests, even when forest fragmentation reduces the abundance or distribution of martens. Commercial harvest and thinning both occurred in the Sagehen Experimental Forest (Tahoe National Forest) in the last 30 years, and their cumulative effect on habitat loss was the most likely cause for the marten decline reported there (Moriarty et al. 2011). Similarly, shelterwood harvests in the red fir zone of the Swain Mountain Experimental Forest (Lassen National Forest) led to open conditions (i.e., percent canopy cover from largest size class trees = 10-19 percent) that were used less often by martens during the winter (Ellis 1998). Thus, clear-cutting, and partial cutting and thinning, have been reported to have negative effects on populations, though they may not necessarily have deleterious effects on gene flow.

Recreation

Recreation has the potential for significant impacts to marten populations, especially winter recreation that occurs in high-elevation montane forests or subalpine zones. The sound of engines from off-highway vehicles (OHVs)is presumed to be a disturbance, but in winter, the use of snowmobiles can also have indirect effects by compacting the snow, permitting access to marten areas by competing carnivores that would not typically be able to traverse deep snow (Buskirk et al. 2000). The only study to explore the effects of OHVs on martens in the Sierra Nevada found that marten occupancy at two study areas was unaffected by year-round OHV use (Zielinski et al. 2008). Martens were ubiquitous in both control and OHV use areas and there was no effect of use areas on probability of detection, nor were martens more nocturnal in the OHV use areas than the control areas. Moreover, females were not less common in the OHV use areas compared to the controls. However, martens were exposed to relatively low levels of disturbance overall, and most OHV use occurred at a time of day when martens were inactive (Zielinski et al. 2008).

Ski resorts are considered likely to affect marten populations because they remove and fragment high-elevation fir forest habitat. There are approximately 25 ski resorts in the Sierra Nevada, and nearly all occur within the range of the marten. The Lake Tahoe region includes approximately half of these resorts, constituting the highest density of resorts in the Sierra Nevada and one of the highest in North America. To create ski runs, chair lifts, and associated facilities, trees are removed, creating open areas and fragmenting forest. Martens typically avoid open areas that lack overhead cover or tree boles that provide vertical escape routes from predators (Drew 1995), are more susceptible to predation if they must cross such areas, and have been shown to avoid areas when >30 percent of mature forest is removed (Bissonette et al. 1997). Snow compaction from grooming alters surface consistency, making it easier for larger bodied carnivores (e.g., coyotes)—which, unlike martens, are not adapted for deep, soft snow—to expand their winter ranges and compete with or prey on martens (Buskirk et al. 2000, Bunnell et al. 2006). Skiers and staff are active during the majority of the day, and grooming and some skiing activity occur during the night. Thus, martens that are sensitive to these activities may not find time for important foraging activities. Ski resort effects are not limited to winter, as habitat fragmentation is a year-round effect and many resorts are developing summer recreational activities (e.g., hiking, mountain biking).

Kucera (2004) conducted the only intensive study of martens in a ski area in California. He captured 12 individuals at the Mammoth Mountain ski area, 10 of which were males, 1 was female, and 1 was of unknown sex, resulting in a highly skewed sex ratio. The single female raised two kits, but did not use developed areas and only used natural rest sites. Martens appeared to move away from the ski area and into unmanaged forest after winter. Kucera (2004) suggested that this fits a seasonal use pattern where martens occupy ski areas during winter when natural prey is least available and human-supplied food is most plentiful, then they move into unmanaged forests in spring. This migration would allow them to exploit artificial food sources during winter, but return to places where females maintain home ranges to breed in summer. Realizing that this study required confirmation and a larger sample, Slauson and Zielinski (unpubl. data) began a 4-year study in 2008 to evaluate the effects of ski area development and use on home range and demography of marten populations. Field work is completed and a final report is due in 2013.

Roads, predation, and other mortality

Roads represent a direct threat to martens (via road kill), as well as an indirect threat by facilitating an increase in the interactions between martens and their predators and competitors (Slauson et al. 2010). Martens are susceptible to predation by coyote (*Canis latrans*), red fox (*Vulpes vulpes*), bobcat (*Felis rufus*), and great horned owl (*Bubo virginianus*) (Thompson 1994, Lindström et al. 1995, Bull and Heater 2001). Marten populations in highly altered forest landscapes (i.e., dominated by landscapes fragmented by regeneration and partial harvests and roads) show higher rates of predation and lower annual survival rates than those in less-altered forest landscapes (Thompson 1994, Belant 2007, Bull and Heater 2001). The mechanism for these demographic effects is presumably linked to the risk of predation incurred by martens when they use stands with less cover or, for example, when fragmented habitat requires martens to use additional energy to travel through their home ranges using only the patches of residual stands. Use of rodenticides, particularly at illegal marijuana cultivation sites on public and private lands, is also a potential threat (Gabriel et al. 2012a), especially at the lower elevational extent of the marten's range (M. Gabriel, pers. comm).

Management Implications - Marten

It makes sense that treating forests to reduce the severity of fire be conducted in proportion to the expectation of catastrophic fires and priorities for restoration. Fortunately, fire return intervals in the true fir (*Abies* sp.) and subalpine zones, where martens are most common, are not as short as in the mid-elevation forest types (mean maximum fire return interval in red fir = 130 years; Van de Water and Safford 2011), so red fir forests are not a high priority for restoration treatments, especially given the backlog of treatments in more fire-prone forest types. Thus, it would appear that **there is less impetus for land managers to thin canopies and reduce surface fuels—both actions that potentially reduce habitat quality for martens—in true fir forests than in the mixed-conifer forests at lower elevations.** Recent summaries of forest activities in red fir, however, indicate a notable increase in the amount of area of red fir that has been commercially thinned in the last 8 years (J. Sherlock, pers. comm.). This trend should be monitored and evaluated for its consistency with restoration goals. **Although there may be a need for restoration treatments in the high-elevation forests, this should be initiated based on strategic fire planning that accounts for the habitat needs of martens at multiple scales** (i.e., landscape connectivity, home range quality, and the provision of microhabitat elements).

Long-term viability for martens will most likely require evaluating habitat connectivity and restoring it where is it is found to be lacking. This is challenging for a number of reasons: (1) there are few studies evaluating the viability of martens under alternative forest management regimes (e.g., Lacey and Clark 1993, Fuller and Harrison 2005, Carroll 2007), and none in California; and (2) fragmented marten habitat is at additional risk from a warming climate because it occurs near the upper elevational range of forests (Carroll 2007, Lawler et al. 2012, Purcell et al. 2012). Thus, **if maintaining adequate marten habitat is desired, plans could be made for connectivity of upper montane forest stands that may migrate to higher elevations in the future.** The California Essential Habitat Connectivity Project (Spencer et al. 2010) may be of some value in this regard, but it is a coarse-scale project of statewide scope. Guidelines for developing connectivity maps at finer resolution are available (e.g., Spencer et al. 2010, Beier et al. 2011), and some work of this nature has been conducted in the vicinity of Lassen National Forest to evaluate the effects of projects on connectivity (Kirk and Zielinski 2010). Furthermore, predicting marten habitat connectivity along the length of the Sierra Nevada and Cascades in California is underway and preliminary models are available (Spencer and Rustigian-Romsos 2012). Also noteworthy is the recent effort to reestablish fishers in the northern Sierra Nevada (Facka and Powell 2010); the implications of the growth of this new fisher population on martens in the California Cascades should be explored.

The Sierra Nevada Framework (USDA 2001, USDA 2004) specifies a marten monitoring program on Forest Service lands, but this has not been fully realized (P. Flebbe, pers. comm.). Periodically, over the 10 year duration of the fisher monitoring program (Zielinski and Mori 2001, Zielinski et al. 2012), some sample units at sufficiently high elevation have detected martens, but martens have not been the target of a species-specific program, nor have the data from the fisher monitoring program been analyzed for their value in monitoring change in occupancy of the Sierra marten population. **A marten-specific monitoring program would produce benefits to managers similar to the occupancy monitoring program for fishers in the southern Sierra.**

Fisher - Threats

Timber harvest, vegetation management, and wildfire

Fishers are sensitive to loss of late-successional habitat via timber harvest and vegetation management and to loss of habitat by uncharacteristically severe fire. Weir and Corbould (2010), studying fishers in British Columbia, found that a 5 percent increase in clear-cut logging (equivalent to 590 acres of a 11,800 acre study area, over 12 years) decreased the probability of home range occupancy by 50 percent. This is probably because fishers avoid establishing home ranges in areas with a high density of openings (Weir and Corbould 2010). Resting and denning structures are probably the most limiting habitat element (Powell and Zielinski 1994, Purcell et al. 2012). Because fishers move between rest sites on a daily basis, and reuse is low (Lofroth et al. 2010), suitable resting structures need to be numerous and well-distributed throughout home ranges. Fishers prefer to rest in shade-intolerant trees, such as black oaks and ponderosa pines (Purcell et al. 2009), which, due to selective harvest and fire suppression, are now less abundant than they were historically (McDonald 1990, Roy and Vankat 1999, Scholl and Taylor 2010, Collins et al. 2011). However, white fir, which is more abundant than historically, is also frequently used for resting sites in the Sierra (Zielinski et al. 2004a, Purcell et al. 2009). Thus, any management actions or disturbance factors (e.g., logging of large-diameter trees, high-severity fire) that further reduce the abundance of large conifers (> 30 inch dbh), particularly ponderosa pines, sugar pines and white fir, as well as black oaks, will negatively affect fishers. Therefore, a long-term strategy for the regeneration and growth of black oak and ponderosa pine (the two most shade-intolerant species that fishers use as resting sites) will probably be an important conservation action for fishers. This will require reducing the density of species that have benefited from fire suppression (e.g., incense cedar and white fir), as specified by North et al. (2009), especially trees that are in the smaller size classes (particularly < 20 inches dbh). Because fishers, martens, and other species, however, will rest in the cavities in the larger white fir, and we don't know the minimum number of cavities to maintain habitat for these species, a conservative approach would call for retaining white fir trees in the largest size classes.

Naney et al. (2012) summarized the significant threats to fishers for each bioregion within the fisher's range in the Pacific states and provinces. In the Sierra Nevada, the highest threats were determined to be severe wildfire and fire suppression activities, fuels reduction and timber harvest, and fragmentation. A tradeoff exists between the loss of habitat value that occurs when forests are thinned to reduce the severity of future fires and the loss of habitat that occurs when untreated stands are consumed by wildfire. Treatments to reduce fire severity can be beneficial if they do not reduce the density of important habitat elements, such as the largest size classes of trees, snags, and logs, or affect canopy cover on topographic positions where it is naturally dense, typically on north and east facing slopes (North et al. 2009). The definition of "large" is important, because managers frequently request threshold values. Subtracting one standard deviation from the mean dbh of fisher resting sites could be a reasonable, and conservative, threshold for the interpretation of "large" trees. Using this assumption, "large" live conifers and snags would be those that exceed 25 and 28 inch dbh, respectively (data from Purcell et al. 2009).

Simulation studies have revealed that carefully applied treatments (thinning from below and the treatment of surface fuels) within fisher habitat may be more effective at reducing the loss of habitat than when treatments are placed outside such habitat (Scheller et al. 2011). Unlike the California spotted owl (*Strix occidentalis occidentalis*), for which the response to fuel treatments has been explored both empirically and via modeling (e.g., Lehmkuhl et al. 2007, Roberts et al. 2011), there is no published work to evaluate the direct effects of fuel treatment (mechanical or prescribed fire) on fishers. A recent study, however, evaluated the effects of various treatment types on predicted values of fisher resting habitat (Truex and Zielinski, in press). The effects of actual treatments were compared by evaluating their effects on predicted fisher habitat based on fisher habitat models developed in the southern Sierra Nevada. The effects of mechanical thinning, prescribed fire, and the combination of mechanical thinning and prescribed fire were compared to controls at Blodgett Forest Research Station in the central Sierra Nevada. All treatments had significant short-term impacts on predicted fisher resting habitat quality, as well as on canopy closure, a key habitat element for fisher in California. Early (June) and late season (September and October) prescribed fire treatments were compared at Sequoia-Kings Canyon National Park. The late season burn treatment had a significant negative impact on modeled fisher habitat suitability when measured one year later (Truex and Zielinski, in press). Although predicted resting habitat suitability was significantly reduced by the treatments, there were no negative effects on predicted foraging habitat. Although the treatments that included mechanical methods had greater short-term reduction on modeled fisher resting habitat suitability than prescribed fire, these effects were mitigated by the fact that mechanical treatments could target or avoid individual trees. Hardwoods and all large trees and snags could more easily be avoided using mechanical means of treatment. Furthermore, even the use of fire could be controlled somewhat by raking debris from the base of particular trees that were viewed as important to protect. Thus, it appears that if care is taken to apply treatments with the goal of protecting large hardwoods and conifers, and there are funds to conduct the raking, the potential reduction in predicted habitat quality may be mitigated (Truex and Zielinski, in press). Long-term strategies that encourage the regeneration, and growth to large size, of black oaks, ponderosa pines, and white fir will also be an important conservation action (e.g., North et al. 2009). Some white fir may need to be removed to encourage the development of oaks and pines, but cavities in large white fir are used as resting sites by fishers and martens. The large white fir may need to be retained during the transition to forests that eventually produced pines and oaks of cavity-bearing size, particularly for fishers. However, martens use large white fir at an elevation above where black oaks and ponderosa pines occur, so the maintenance of an abundant supply of large white fir in the upper montane zone will be an important component of marten habitat management.

Studies on spotted owls suggest that the use of prescribed fire to reduce the density of small trees can be compatible with owl occupancy (Ager et al. 2007, Lehmkuhl et al. 2007, Roberts et al. 2011, Roloff et al. 2012). Some of this research predicts, via modeling, that fuels treatments on a relatively small proportion of the forest landscape result in significant decrease in the probability of owl habitat loss following wildfire (e.g., Ager et al. 2007). If this work also applies to fishers, it suggests that in fire-suppressed forests, a “no action” management option may involve greater risk to fishers than some form of treatment because these ecosystems have diverged from historical (and also more resilient) conditions (Thompson et al. 2011, Purcell et al. 2012). Management to reduce fire risk, or to restore ecological resilience to fire, may be consistent with the maintenance of landscapes capable of supporting fishers, as long as sufficient resting/denning structures are retained and the composition and configuration of the residual landscape is compatible with home range requirements (e.g., Thompson et al. 2011).

A recent meta-analysis of fisher resting habitat studies throughout the western U.S. and Canada provides some overarching conclusions about the features that distinguish resting sites from random forest sites (Aubry et al., submitted). This work includes the results of five studies in California and reinforces the conclusions of those independent studies, in terms of the importance of retaining large trees, snags, and logs and dense cover for fisher resting habitat. The authors found that resting sites differed from random points at all 8 study areas in the following respects: resting sites had steeper slopes, lower heat load indices, higher overhead cover, greater volume of logs ≥26 cm in mean diameter, greater basal area of large (51-100 cm dbh) conifers, greater basal area of large hardwoods, greater basal area of large snags, larger mean dbh of live conifers ≥10 cm dbh, and larger mean dbh of live hardwoods ≥10 cm dbh. Reductions in the values for these features should be considered threats to the availability of resting habitat for fishers in the Sierra Nevada, and elsewhere in the fisher's western range.

Because prescribed fire can pose a threat to fisher habitat, early season burns appear to be preferable to late season burns (Truex and Zielinski, in press). Early burns, which are timed to follow the fisher denning period in spring, will minimize the likelihood of disturbing denning female fishers. If conditions necessitate burning earlier than mid-May, efforts should be made to avoid treating areas that have a high density of structures likely to be used by females for denning (for reference to denning structures and denning habitat, see Lofroth et al. 2010; Thompson et al. 2010; Sweitzer and Barrett, unpubl. data.; W. Spencer, unpubl. data.; A. Facka, unpubl. data).

Roads, predation, and other mortality

Due to their delayed maturation and the fact that not all females reproduce each year, fisher population growth rates are relatively low (Lofroth et al. 2010). Thus, the recently reported high rates of predation on fishers (Thompson et al. 2010; Sweitzer and Barrett, pers. comm.), especially by bobcats (*Lynx rufus*) and mountain lions (*Felis concolor*), are of concern. Road-killed fishers are relatively common, even in national parks with relatively low posted speed limits (L. Chow, pers. comm.). Berg and Sweitzer (unpubl.) found that fishers were detected closer to roads than would be expected, particularly near high-use roads during the winter. A subcommittee of the interagency Southern Sierra Fisher Working Group is exploring ways to mitigate this source of mortality (K. Purcell, pers. comm.). Use of rodenticides, particularly at illegal marijuana cultivation sites on public lands, is also a growing threat (Gabriel et al. 2012a). Fishers are also affected by viral and parasitic diseases (Brown et al. 2006, Gabriel et al. 2012b), but the magnitude of the direct and indirect effects of these organisms on fisher populations is unknown.

Management Implications - Fisher

Recent research findings (summarized in Purcell et al. 2012) support the validity of previous recommendations to **focus habitat management for fishers in areas where, historically, fires would have burned less frequently, such as north and east-facing slopes, canyon bottoms, and riparian areas** (North et al. 2009). Resting sites are often found close to streams and on relatively steep slopes (Purcell et al. 2009, Zielinski et al. 2004a), and fisher telemetry locations include more observations in canyons and fewer observations on ridges (Underwood et al. 2010). These are landscape-scale recommendations, but as noted earlier in this review home range and stand-scale level recommendations frequently center on protecting large diameter hardwoods and conifers (but not specifying how many per acre are necessary) and maintaining canopy cover (e.g., a minimum of 56-61 percent canopy cover in stands, depending on the method of measurement; Purcell et al. 2009). Stand- and home range-scale management is often more problematic because areas used are not homogeneous, and no single threshold should be applied to all stands or home ranges. This is why a landscape-scale approach to management of forests in the Sierra Nevada would help ensure adequate fisher habitat. **Not all stands need to meet the minimum standard for occupancy, but for occupancy to occur in a home range-sized area there is a typical collection of composition and configuration attributes that are derived from stand-level information (e.g., Thompson et al. 2011). A process-based approach to generating heterogeneity in landscape condition, like that offered by North et al. (2009), will not only be superior to a stand-by-stand level approach, but may be the only approach possible given the logistical challenges of describing stand-level variables and the difficulty of recording and tracking the changes in stand-level conditions in large areas over space and time.**

The approach recommended in North et al. (2009) also encouraged the retention of oaks and pines, and stressed the importance of hardwoods, especially California black oaks (*Quercus kelloggii*). Black oaks require openings for regeneration and subsequent growth (McDonald 1990), suggesting that **the creation of small openings around mature productive trees would aid establishment of young trees needed to replace dying oaks. It would be best to balance this approach with retaining smaller trees around oaks with visible cavities that are currently suitable as resting or denning structures.** Most oaks used by fishers are live trees, although dead portions (e.g., broken limbs with access to cavities) of otherwise healthy trees are important (Zielinski et al. 2004a, Purcell et al. 2009).

Monitoring of fisher habitat and populations is an essential component of adaptive management. Fortunately, a number of research and management efforts have established the groundwork for these components. **There now exist empirical models that can be used to assess and monitor fisher habitat at the resting site, home range, and landscape scales for various locations in the Sierra Nevada**. The effects of forest practices on fisher resting habitat can be quantitatively evaluated with the development of a model for the southern Sierra Nevada that predicts resting habitat value from plot data (Zielinski et al. 2006, 2010). The model is specifically developed to use Forest Inventory and Analysis (FIA) data, but can use other types of plot data, as well. This model has also been integrated with the Forest Vegetation Simulator (FVS) to forecast future effects of proposed activities on fisher resting habitat (Zielinski et al. 2010). A number of regional landscape suitability models are also available (Davis et al. 2007, Spencer et al. 2011), and they can be used for assessment and monitoring of large-scale habitat distribution and connectivity. The California Essential Habitat Connectivity Project (Spencer et al. 2010) may be of some value in this regard, but it is a coarse-scale project of statewide scope. Guidelines are now available for the development of finer-scale connectivity maps (e.g., Spencer et al. 2010, Beier et al. 2011). Using these practices, a new effort is underway to model fisher habitat connectivity and to specify linkages in the central and southern Sierra Nevada (Spencer and Rustigian-Romsos 2012).

A specific research need identified in North et al. (2009) entailed examination of potential outcomes of proposed forest treatments based on modeling habitat in female fisher home ranges. This shortcoming has been partially addressed through the recent development of an analytical tool that predicts the relative impacts of management actions on fisher habitat in the vicinity of the Sierra National Forest (Thompson et al. 2011). This approach is a form of ecological risk management and is based on quantifying the range of variation in currently occupied female fisher home ranges. It assumes that if we manage landscapes to resemble those occupied home ranges, there is a high likelihood that the landscape will remain functional fisher habitat and minimize the risk of negative population impacts. Results in the Sierra National Forest study area indicate that female fishers use landscapes with relatively high proportions of large trees and snags, and where patches of high-quality habitat are connected in a heterogeneous mix of forest ages and conditions. Unfortunately, it is not known what size these patches must be nor how far apart they can be to assure that they become incorporated into a fisher home range. However, Thompson et al. (2011) specify the average values for female fisher home ranges in respect to a number of variables, including canopy closure, basal area, and a number of common indices of patch size and connectivity (see Table 2 in Thompson et al. 2011). Values for these variables suggest the importance of variation in canopy cover and tree size values among stands within home ranges. These values, however, apply only to the region where the model was developed, but the **results suggest that some level of management to reduce fire risk may be consistent with maintaining fisher habitat, as long as sufficient resting/denning structures are retained**. Currently in development is a decision tool that will allow managers to evaluate project areas for their suitability as female fisher home ranges and to adjust prescriptions accordingly (C. Thompson, pers. comm.). **It would be very useful if this tool were fully developed for the Sierra National Forest, and similar fisher home range habitat "templates" developed for other areas where sufficient data on the vegetation characteristics of home ranges have been collected by researchers.**

Based upon reviews of relevant science, members of the synthesis team concluded that **one of the most scientifically and economically defensible ways to protect biodiversity in the Sierra Nevada—including the habitat of martens and fishers—is to promote prescribed fire and**

managed wildfire for its beneficial ecological effects, including the capacity to make the forests of the Sierra Nevada more resilient to climate change. Fire is a disturbance that has historically influenced the vegetation structure and composition in the synthesis area and produced patterns to which the fauna and flora of the area have adapted. The Integrative Approaches chapter (1.1) and the Synopsis of Emergent Approaches (1.2) provide more details about this approach. **To minimize impact on individual fishers, prescribed fire and other treatments as needed should be dispersed over space and time.** Testing the hypotheses discussed under Information Gaps (below) in a rigorous assessment framework can lead to better guidelines for the spatial and temporal application of treatments.

Part of an assessment framework is a credible monitoring program. The Sierra Nevada Framework (2001, 2004) specified the development of a fisher monitoring program and a study plan was conceived (Zielinski and Mori 2001) that led to annual occupancy monitoring beginning in 2002. Analysis of the first 8 years of sampling data revealed that occupancy was stable over that period (Zielinski et al. 2012). This program is likely to continue (P. Flebbe, pers. comm.) and the results of **this population monitoring program could be reconciled with multi-scale habitat monitoring for a dual-monitoring approach (population and habitat)** in the future. This combined monitoring program will reassure us that the assumptions we are making about restoring resilient forest ecosystems with the use of prescribed fire and some mechanical treatment of surface and ladder fuels (see Integrative Approaches chapter (1.1)) will indeed produce the amount and distribution of habitat that favors the persistence of fishers and martens.

Finally, of great concern is the recent news that rodenticide, most likely associated with marijuana cultivation, may be a new source of morbidity and mortality in fishers (Gabriel et al. 2012a). **Reducing the application of rodenticides and remediating the environmental damages that have already occurred is an acute need.**

Climate Change Implications – Marten and Fisher

Lawler et al. (2012) investigated the potential direct and indirect effects of climate change on select species of the genus *Martes*. Climate change predictions suggest that the range of the Pacific marten in California will contract to the north and up in elevation over the coming century (Lawler et al. 2012). Furthermore, warming climate may favor the upward elevation expansion of fishers into areas currently occupied by martens (increasing potential competition), and marten habitat will become less common and more fragmented (Lawler et al. 2012). This is because the biggest predicted change in forests in California is the increase in mixed woodland and hardwood-dominated forest types and the reduction in conifer-dominated forest types (Lenihan et al. 2003). Because oaks—especially California black oaks—are a key component of fisher habitat, floristic changes may benefit fishers as long as temperature effects do not result in upward range shifts. Reductions in snowpack, as a result of climate change, could also favor fishers, since deep snow normally excludes fishers from marten habitat in winter (Krohn et al. 1997).

Climate change is also predicted to change fire regimes, increasing fire frequency, area, and intensity (e.g., Flannigan et al. 2000), and these changes are expected to result in loss of late-seral habitat important to both species (McKenzie et al. 2004). Decreases in the density of large conifer and hardwood trees and canopy cover are projected as fire severity increases (Lawler et al. 2012). As these factors are closely related to fisher rest site and home range use in the southern Sierra Nevada (Purcell et al. 2009, Zielinski et al. 2004b), the expectation is for an overall decrease in the availability of fisher habitat. The largest climate impacts to these species will probably occur at the southernmost portion of their ranges (the southern Sierra Nevada) (Lawler et al. 2012). The authors recommend protecting fisher habitat through targeted forest-fuel treatment, and applying more liberal fire management policies to naturally ignited fires during moderate weather conditions.

The change in marten distribution at the Sagehen Experimental Forest (Moriarty et al. 2011) appeared to occur rapidly due to the influence of traditional timber harvest methods during the mid to late 1900s. It occurred more rapidly than changes in the flora and fauna that affect marten populations might be expected to change due to climate. Yet the marten is a species that may not fare well given the predicted changes in vegetation distribution as a result of warming climates. The true fir forests where martens typically occur are predicted to diminish in area (Lenihan et al. 2003, Mortenson 2011), resulting in a poor prognosis for martens (Lawler et al. 2012). Climate change may also increase fire frequency and intensity in the upper montane zone, calling for increased levels of thinning treatments in this elevation, which may also diminish marten habitat.

Information Gaps

Purcell et al. (2012) summarized some of the gaps in knowledge that are preventing application of science to the management of fisher and marten habitat. They felt that there was a great deal of uncertainty around predicting impacts on marten and fisher habitat, particularly cumulative effects. This is largely because our knowledge of how habitat change influences survival and reproduction is limited, and because we do not yet understand how landscape heterogeneity affects these species at multiple scales. Managing for appropriate stand conditions in terms of density and abundance of large trees may be insufficient if the nature of the arrangement of these stands on the landscape is not also considered. If forests of the future will be more heterogeneous, and this heterogeneity will be a result of a strategic mixture of, for example, three types of patches: (1) high vegetation density, (2) low vegetation density resulting from thinning, and (3) openings—then it will be critical to understand how much of each should occur in landscapes managed for fishers and martens. In particular, managers need information about the necessary extent and connectivity of older forest patches, and the spatial heterogeneity and composition of the remaining landscape. Research is moving in this direction (e.g., Thompson et al. 2011), but we cannot yet provide these recommendations. Nor can we yet determine how fishers will respond to landscapes that have been managed to include active fire regimes, since there are too few of those landscapes available for study or they occur in atypical landscape settings.

Central to moving forward in treating forest fuels is the need to understand the tradeoff between the loss of habitat value that occurs during proactive fuels treatments (when surface fuels are reduced and/or canopies are thinned), and the loss of habitat that occurs when untreated stands are consumed by wildfire. Some simulation work has been done that indicates that the indirect negative effects of treatments are justified, at least in terms of modeled fisher habitat (Scheller et al. 2011). However, much more work needs to be done on this subject for martens and for fishers. Ongoing field studies on martens in the Cascades (Slauson and Zielinski, unpubl. obs.; Moriarty et al., unpubl. obs.), and on fishers in the Cascades and the Sierra Nevada (Thompson and Purcell, unpubl. obs.), are seeking to understand the direct effects of fuels treatments on these important wildlife species. Whether animals stay or relocate—and for how long—during management activities is important to know. More critical, however, is how to allocate treatments in space and time so that predicted fire intensity and the distribution of habitat are within acceptable ranges. There are examples of research that lead in this direction (e.g., Cushman and McGarigal 2007, Thompson et al. 2011), but this topic needs more urgent attention and results. Finally, it is important to note that our lack of knowledge about the effects of fuel treatments on fishers and marten extends to their prey. We know very little about the effects of management activities on important prey species and on foraging behavior.

Future work will need to explore the effect of understory management on fishers and martens. It is fairly well understood that treating surface fuels (shrubs and downed wood) and ladder fuels is necessary, and perhaps sufficient, to reduce the potential for high-intensity fires (Agee and Skinner 2005) and that this will have only modest effects on the habitat for some species associated with mature forests (Stephens et al. 2012). Focusing on ladder fuels without reducing overstory canopy cover has been considered a compromise that will achieve fuel reduction goals but still maintain habitat for mature forest associated species. However, these prescriptions have the potential to result in greater homogeneity because residual overstory cover is generally uniformly high, but vertical complexity is generally very low. Additionally, the high residual overstory cover may inhibit regeneration of shade-intolerant species (e.g., oaks and shrubs), which are important habitat elements for fishers. There may, in fact, be undue attention directed to diameter limits and the need to promote and protect large trees, when understory management may be a potentially greater source of conflict between achieving fire goals and fisher habitat goals. Lofroth et al. (2010) believed that management that reduced or removed understory vegetation may decrease prey availability, disrupt movements, and make fishers more vulnerable to predation. Furthermore, Naney et al. (2012: 39-40) noted that “a successful conservation strategy must…. recognize the importance of understory vegetation to support abundant prey populations and provide adequate fisher cover, and the contribution of diverse native vegetation to fisher habitat and the maintenance of resilient landscapes." It is necessary to determine what levels of surface fuels treatments are compatible with the retention of fisher and marten foraging habitat, and habitat for their prey. It is possible that the approach advocated by North et al. (2009), which would retain understory structural diversity on north- and east-facing slopes, may provide sufficient understory for fishers, martens, and their prey, but this remains an assumption to be tested. That we now have the ability to describe fisher habitat features using LIDAR (Zhao et al. 2012) is a technological breakthrough that will help characterize treatment effects on understory structure.

Our current understanding of fisher habitat use is also prejudiced by the fact that all fisher research has been conducted in forests where fires have been suppressed. This shortcoming means that we continue to assume that the places that fishers choose for home ranges and for resting sites are the same types of places that fishers would select if forest ecosystems were more heterogeneous, less dense in places, and fire was a dominant disturbance. Even when we possess this information, however, ecologists do not know how to influence vegetation dynamics to produce habitat that will replace the habitat lost to different stressors. Dense stands of preferred species, without the action of fire or thinning, will not guarantee the replacement of large conifers and oaks for use as resting sites. Maintaining habitat in riparian areas and on topographic positions that may have burned less frequently, and at times more severely—as advocated by North et al. (2009)—may help provide resting habitat elements, as well as connectivity, without significantly reducing the effectiveness of treatments designed to restore resilient forests. However, the resulting habitat connectivity in landscapes managed with these objectives is unknown. New analytical tools (e.g., Thompson et al. 2011, Zielinski et al. 2010) will help evaluate the merits of landscapes that develop under these new management regimes, but the tools may be inappropriate if they are modeled on forest conditions indicative of fire-suppressed forests.

A reasonable hypothesis to be tested in an adaptive management framework is whether fishers can tolerate fuels treatments up to the levels that may be needed to reduce risk of uncharacteristically severe wildfire at the landscape scale. Fishers have evolved with the effects of fire on their habitat, although it is uncertain how much disturbance via fire and fuels treatment they may tolerate. Researchers on the Sierra National Forest (C. Thompson and K. Purcell, unpubl. data), however, have surveyed multiple study areas equivalent to the average size of a female fisher home range (approximately 5 mi^2) and have found that in areas currently occupied by fishers, treatments have covered 2.2-16.9 percent of the land area over 3 year periods (a reasonable estimate of fisher generation time). Treatment, in this case, included a combination of prescribed fire, thinning, salvage logging, and other forms of timber harvest. Using round numbers, these data suggest that fishers may tolerate a disturbance (fire or surrogate treatment) over approximately 10 percent of a 5mi^2 area over a 3 year period (i.e., about 320 acres over a 3 year period over a 5mi^2 area). This aligns fairly well with the lower range of treatment area/year expected to reduce the likelihood of unacceptably large, high-intensity fires (10-25 percent of a landscape over a 5-10 year period) (Ager et al. 2007, Finney et al. 2007, Schmidt et al. 2008, Ager et al. 2010, Syphard et al. 2011). To be cautious, the area treated should account for the relative suitability of habitat patches as well as their contribution to fisher habitat conditions within the local area. For example, impacts to fishers may be too great if treatments target important patch types in a poor quality home-range area; this is in contrast to treatments that target less important habitat patches in a higher quality home range area. This approach remains a hypothesis to be tested, but the proposed rate and extent of disturbance in fisher habitat may permit the coexistence of fishers with a rate of application of fuels treatments that will also protect their habitat from loss due to high-intensity fire. Such a rate may be tolerated by fishers, especially since the fraction of the landscape needed to reduce wildfire risk may only partially coincide with occupied fisher areas. It is important to caution, however, that this proposal represents the desire to create a starting point for collecting new information that can evaluate its merit. This proposed guideline needs to be tested in an adaptive management framework.

We also still lack important information about specific life-history characteristics, such as reproductive site requirements for martens and fishers, including their requirements for den trees (parturition sites) and denning habitat at multiple spatial scales (Purcell et al. 2012). As suggested in North et al. (2009), one way to help ensure the retention of key forest structures would be to provide a list of attributes and representative photos of resting and denning structures for use by marking crews (see Lofroth et al. 2010 for descriptions of the specific types of structures used by fishers for resting and denning). It is one thing to identify these potential structures, but another to understand the effects of nearby disturbance, including prescribed fire, on the occupant(s)—especially if they are reproductive females. Some new work is being done at Kings River (Sierra National Forest) to understand how females with kits respond to nearby management activities (i.e., people, heat and smoke from prescribed fire), and preliminary results suggest that the work will be informative (C. Thompson, pers. comm.). However, much more work needs to be done to determine whether the administration of treatments near known dens, and in areas where den locations may not be known but will still be affected, will have tolerable effects on fisher behavior, physiology, and reproductive output.

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7.2 California Spotted Owl: Scientific Considerations for Forest Planning

John Keane

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Executive Summary

California spotted owls are top trophic-level avian predators associated with mature forests characterized by dense canopies and large trees, snags, and logs in the Sierra Nevada. Owls show the strongest associations with mature forest conditions for nesting and roosting and will forage in a broader range of vegetation types. California spotted owls use habitat at multiple scales ranging from: (1) patches and stands used for nesting and foraging; (2) home ranges that support a territorial pair of owls; and (3) landscapes necessary to support viable populations. Multi-scalar management is needed to address the habitat requirements of California spotted owls. Recent research reinforces the importance of mature forest to spotted owl occupancy, survival, and habitat quality. These studies have also demonstrated that some level of heterogeneity is also associated with high quality habitat for spotted owls. California spotted owls seem to be able to persist in landscapes that experience low- to moderate-severity fire, as well as some level of mixed-severity wildfire. Little information is available to evaluate how California spotted owls and their habitats are affected by mechanical treatments. New forest management concepts, based on recent research, have proposed focusing efforts on restoration of vegetation conditions at patch, stand, and landscape scales that would be more similar to the heterogeneous conditions that would occur under a wildfire-dominated natural disturbance regime. Although some degree of uncertainty exists, management approaches that seek to restore ecological resilience and the role of wildfire as the primary natural disturbance agent may produce landscapes capable of supporting viable populations of spotted owls, provided that mature forest conditions characterized by dense canopies and important large conifer and oak trees can be maintained. Current knowledge is incomplete regarding the conditions necessary to maintain high quality owl territories and viable populations under these new forest management concepts. Future forest management efforts will require carefully constructed monitoring and adaptive management components to reduce uncertainty and advance current knowledge about owl territory needs and population viability. In addition to habitat concerns, California spotted owls face a significant emerging threat due to the recent range expansion of barred owls into the Sierra Nevada. Barred owls appear to be competitively dominant over spotted owls and have the capacity to displace or replace spotted owls at sites where they co-occur. Future planning efforts may need to consider the potentially confounding effect of barred owls on California spotted owl habitat requirements, should barred owls continue to increase their distribution and abundance in the Sierra Nevada. This synthesis reviews the current knowledge and threats to California spotted owls in the Sierra Nevada reported since publication of the Sierra Nevada Forest Planning Amendment in 2001.

Introduction

California spotted owls (*Strix occidentalis occidentalis*) have been at the forefront of Sierra Nevada management and conservation debates for 25 years because of their strong habitat associations with commercially valuable large trees, snags, and late-successional forests. Initial concerns focused on the effects of timber harvest on large trees and late-successional habitat and potential risks to California spotted owl population viability. In recent years, the debate over Sierra Nevada forest management and California spotted owls has broadened with growing recognition that past management practices, specifically timber harvest and fire suppression, have fundamentally changed forest structure, composition, and function over the last 100 years. Removal of fire as the primary natural disturbance process, coupled with reductions in large trees and late-successional forests through timber harvest, has resulted in contemporary Sierra Nevada forests that are generally more homogenous at multiple spatial scales, have higher densities of shade-tolerant tree species and reduced numbers of large trees, and are at greater risk of high-severity wildfire compared to their historical counterparts. Additionally, the extent and severity of wildfire has increased in the Sierra Nevada and across the western United States as a result of climate change (Westerling et al. 2006, Miller et al. 2009). Eighty-five percent of known California spotted owl sites occur in moderate- or high-risk fire areas in the Sierra Nevada (U.S. Forest Service 2001), and though management and restoration of large trees and late-successional forest remains a primary objective of spotted owl management, there is uncertainty about the relative tradeoffs of actively managing forests to reduce wildfire risk and no-management approaches that leave habitat intact but don't address wildfire concerns. This issue is embedded within broader discussions about the most effective strategies for addressing long-term resilience and restoration of Sierra Nevada forests (North et al. 2009). Finally, range expansion of the barred owl *(Strix varia)* in the Sierra Nevada poses an increasing risk factor to California spotted owls.

Extensive scientific literature reviews on California spotted owl have been conducted as part of the California Spotted Owl Technical Review (Verner et al. 1992) and during development of the Sierra Nevada Forest Plan Amendment (U.S. Forest Service 2001). The goal of this chapter is to synthesize scientific information on the California spotted owl that has been reported since the SNFPA (2001). Though this section is focused on California spotted owl, relevant papers from the northern spotted owl (*S.o. caurina*) and Mexican spotted owl (*S.o. lucida*) subspecies are also included.

Ecological Context

As top-trophic level avian predators in Sierra Nevada forests, California spotted owls have several characteristics that are broadly associated with increased species vulnerability: they have large individual spatial requirements, low population densities, and they are habitat specialists. Spotted owls have high adult survival rates and low reproductive rates—life history characteristics associated with species that are long-lived with sporadic reproduction in response to variable environmental conditions. Although they are adaptations to variable environments, these life history characteristics also render spotted owl populations slow to recover from population declines. Spotted owl populations are regulated by territorial behavior in which owl pairs defend non-overlapping territories that include nesting and foraging habitat; they exhibit this behavior in response to resources and conditions, such as habitat, weather, prey, and competitors, that influence population dynamics. Maintaining viable populations of owls requires consideration and management of landscape vegetation conditions and dynamics, which determine the numbers and distribution of spotted owl pairs/territories that can be sustained across a landscape. Primary conservation and management concerns at the population-landscape scale include managing for a distribution of high quality territories that support high adult survival and high reproduction (= high quality) and facilitate successful dispersal and recruitment. Management and conservation efforts would benefit from considering the nested structure of spotted owl habitat associations at multiple scales, ranging from patches/stands used by individual owls for nesting, roosting, and foraging, to territories required to support a pair of owls, and up to landscapes necessary to support viable populations. These different spatial scales should all be considered in formulating strategies to promote forest restoration, resilience, and fire management, and they are consistent with current forest management proposals advocating a multi-scale perspective that encompasses a patch-scale focus to increase vegetation heterogeneity nested within a landscape-scale focus that considers topography, elevation, latitude, and natural disturbance regimes.

Population Distribution, Status, Trends, and Genetics

Historical and Current Distribution

Little is known about the historical distribution, abundance, and habitat associations of California spotted owls in the Sierra Nevada (Verner et al. 1992, Gutiérrez 1994). Early inventory and survey efforts are described in Verner et al. (1992). The first systematic efforts to survey California spotted owls occurred in 1973-1974, followed by broad efforts to inventory California spotted owls across the Sierra Nevada during the late 1980s and early 1990s. More recently, inventory and survey efforts have largely consisted of project-level surveys, generally conducted for 1-2 years in support of project implementation, and demographic study area-level surveys at four study sites where consistent annual monitoring has occurred over a much longer time period (Franklin et al. 2004, Blakesley et al. 2010).

Verner et al. (1992) found no evidence of range contractions or expansions of California spotted owls in the Sierra Nevada. However, they identified 11 areas of concern where potential gaps in distribution may become a concern due to fragmentation or bottlenecks in the distribution of owls or their habitat if the status of the owl in the Sierra Nevada were to deteriorate. No research has addressed the status of owls or their habitat in these areas of concern since the publication of CASPO in 1992. Anecdotal observations of California spotted owls suggest they are still widely distributed across their historical range in the Sierra Nevada.

Population Status and Trends

The most recent estimate of population size for California spotted owls in the Sierra Nevada reported 1865 owl sites, with 1399 sites on USDA Forest Service lands, 129 owl sites on USDI National Park lands, 314 sites on private lands, 14 sites on USDI Bureau of Land Management lands, eight on State of California lands, and one on Native American lands (U.S. Fish and Wildlife Service 2006). Demographic monitoring from four study areas from 1990-2011 provides the sole source of empirical data on the status of and trends in California spotted owl populations in the Sierra Nevada. Three of the demographics studies are conducted on National Forest Service lands (Lassen (LAS), Eldorado (ELD) and

Sierra (SIE) national forests), and the fourth study is located on National Park Service lands (Sequoia-Kings Canyon National Park (SKC). Two meta-analysis workshops have been conducted to analyze California spotted owl demographics and population trends across the four studies (Franklin et al. 2004, Blakesley et al. 2010). Blakesley et al. (2010) analyzed demographic data for the period 1990-2005 and estimated that the mean finite rate of population change (lambda) for each study area was 0.973 for the LAS (95 percent CI = 0.946-1.001), 1.007 for the ELD (95 percent CI = 0.952-1.066), 0.992 for the SIE (95 percent CI = 0.966-1.018), and 1.006 for the SKC (95 percent CI = 0.946-1.001). Ongoing research of recent population trends indicates increasing evidence for population declines on the three studies on National Forest Service lands and a stable/increasing population on the National Park Service study area, and it is providing new approaches for evaluating spotted owl population trends and interpreting the probability of population declines (Conner et al., in review; Tempel and Gutiérrez, in review). The factors driving these population trends are not known.

Population Genetics

Population genetics studies support the recognition of three sub-species of spotted owls based on both micro-satellites (Funk et al. 2008) and mitochondrial DNA(Barrowclough et al. 1999, Haig 2001, Haig et al. 2004, Barrowclough et al. 2005, Barrowclough et al. 2011). Early descriptions of spotted owl distribution based on limited numbers of specimens and records suggested that the range of northern and California spotted owls did not overlap (Grinnell and Miller 1944). More recent extensive survey and inventory efforts have documented a continuous distribution of spotted owls between the sub-species, and initial genetic analyses indicated the boundary between sub-species to be in northern California or southern Oregon (Barrowclough et al. 2005, Funk et al. 2008). Based on these initial genetic assessments, Gutiérrez and Barrowclough (2005) proposed the Pit River as a geographical boundary between the sub-species because it was located approximately halfway between populations known to consist largely of pure northern spotted owl haplotypes near Mt. Shasta and largely pure California spotted owl haplotypes near Mt. Lassen. However, recent research has refined understanding of genetic variation across this region and indicates that a hybrid zone exists across from approximately 94 km north of Mt. Shasta to southeast of Mt. Lassen, and that a more appropriate place to designate a boundary between northern and California spotted owls would be between the Pit River and Mt. Lassen (Barrowclough et al. 2011).

Habitat Associations

Patch/Stand Scale

Nest and roost habitat

Nest and roost site habitat requirements are the best-studied aspect of California spotted owl habitat associations. Spotted owls nest in cavities, on tops of broken trees, and on platforms located in older, large-diameter live conifers, oaks, and snags. Conifer nest trees average about 110 cm (45 inch) diameter at breast height (dbh) in the Sierra Nevada (Verner et al. 1992). Large conifers, oak trees, and snags are key habitat elements for California spotted owls, and large, downed logs resulting from these trees are important habitat elements for key prey species. Nests and roosts are typically located in stands that have ≥70 percent total canopy cover and contain one or several large trees of declining vigor and multiple canopy layers resulting from mixtures of different aged trees.

Foraging habitat

Recent telemetry studies of spotted owl foraging habitat use consistently indicate that owls use a broader range of vegetation conditions for foraging than they do for nesting and roosting (Ganey et al. 2003, Glenn et al. 2004, Irwin et al. 2007, Williams et al. 2011). California spotted owl foraging habitat use across all forest types and vegetation conditions remains a poorly understood aspect of their ecology. Use of a broader range of vegetation conditions for foraging is likely governed by the abundance and availability of important prey species. Northern flying squirrels (*Glaucomys sabrinus*) and dusky-footed woodrats (*Neotoma fuscipes*) dominate the biomass of owl diets in the Sierra Nevada, with deer mice (*Peromyscus maniculatus*), pocket gophers (*Thomomys spp*.), and other small mammals, birds, and insects also a component of the diet (Verner et al. 1992). Williams et al. (1992) synthesized the available ecological and habitat association information for key California spotted owl prey species. Given the importance of prey to spotted owls, much research has focused on small mammal ecology and habitat associations (Coppeto et al. 2006, Lehmkuhl et al. 2006, Innes et al. 2007, Meyer et al. 2007) and the effects of forest treatments on small mammals (Amacher et al. 2008). Results from studies of small mammal habitat associations demonstrate the species-specific importance of vegetation type, stand characteristics, and specific habitat elements (e.g., shrubs, downed logs, snags, truffles), and that habitat associations may vary in different parts of the Sierra Nevada. Fontaine and Kennedy (2012) and Stephens et al. (2012) provide recent reviews of fuels treatments effects on wildlife. A full review of the literature on habitat associations and forest management effects on key California spotted owl prey species is beyond the scope of this review. However, given the importance of prey to California spotted owls and other carnivores in the Sierra Nevada, a future synthesis of scientific information on small mammal habitat associations and effects of forest management on their populations and habitat is needed. This information may inform and tailor development of future forest management treatments and desired landscape conditions to different parts of the Sierra Nevada, as well as identify important information and research gaps.

Core Area/Home Range/Territory Scale

The size of California spotted owl home ranges is highly variable; it appears to vary with latitude and elevation, and likely in response to the availability and arrangement of vegetation types and dominant prey species (Gutiérrez et al. 1995). Across the Sierra Nevada, home range sizes are smallest in low-elevation, hardwood-dominated sites, where dusky-footed woodrats (*Neotoma fuscipes*) are the dominant prey species; intermediate in mixed-conifer forests; and largest in true fir forests of the northern Sierra Nevada, where northern flying squirrels (*Glaucomys sabrinus*) are the dominant prey species (Zabel et al. 1995). A number of studies have reported that the proportion of older forest is the best predictor of home range size, with smaller home ranges having higher proportions of older forest (Gutiérrez et al. 1995, Glenn et al. 2004). Recently, Williams et al. (2011) reported that the number of vegetation patches (a measure of habitat heterogeneity) is the best predictor of home range size for California spotted owls in the central Sierra Nevada, with larger home ranges associated with greater habitat heterogeneity. Whether owls were selecting home ranges for vegetation heterogeneity, or owls were simply using more heterogeneous home ranges that reflected habitat availability or increased travel distances associated with the lack of large areas of older forest within the study area, was uncertain (Williams et al. 2011).

Recent research includes observational studies that describe associations between habitat and spotted owl occurrence, occupancy, and demographic parameters (survival, reproduction, habitat fitness potential) at the core area and home range scales (Franklin et al. 2000, McComb et al. 2002, Irwin et al. 2004, Olson et al. 2004, Blakesley et al. 2005, Dugger et al. 2005, Lee and Irwin 2005, Seamans and Gutiérrez 2007a, Gaines et al. 2010, Kroll et al. 2010, Dugger et al. 2011). Although the size of the analysis areas varied across studies, habitat associations were generally assessed at similar spatial scales (core areas or home range scales) around spotted owl nests and roost or activity centers. Vegetation classifications and habitat definitions also varied across studies, but the studies generally defined spotted owl habitat as stands with large trees and high canopy cover (hereafter referred to as mature forest; see individual studies for specific definitions of habitat used in each).

Though recent studies are geographically and spatially varied, they share many key themes. Results consistently reinforce original findings of the strong association between spotted owls and mature forest habitat in core areas around nest sites. Modeling of habitat conditions with survival and occupancy shows that important habitat metrics at core area and home range scales include the total amounts of mature forest and/or the amounts of interior mature forest (defined as the amount of mature forest greater than some distance from an edge). Adult spotted owl survival is positively associated with the amount of mature forest (Franklin et al. 2000, Olson et al. 2004, Blakesley et al. 2005, Dugger et al. 2005, Dugger et al. 2011). Territory occupancy is also positively related to the amounts of mature forest at core area scales for both California and northern spotted owls, with higher colonization rates and lower extinction rates associated with territories with more mature forest (Blakesley et al. 2005, Seamans and Gutiérrez 2007a, Dugger et al. 2011). Dugger et al. (2011) reported that northern spotted owl extinction and colonization rates were negatively associated with the degree of fragmentation of mature forest across the larger home range.

Spotted owl reproduction exhibits high annual variation. Franklin et al. (2000) reported that 43 percent of annual variation in reproduction is explained by habitat covariates, however, most studies report little influence of habitat on variation in reproduction (Olson et al. 2004, Blakesley et al. 2005, Dugger et al. 2005). Weather may directly influence spotted owls, but it may also indirectly influence them by affecting the abundance or availability of prey. In general, annual variation in reproduction has been shown to be associated with weather (importance of specific weather metrics differs among studies), owl age/experience, reproduction in the previous year, and the presence of barred owls nearby(Seamans et al. 2001, Olson et al. 2004, Blakesley et al. 2005, Dugger et al. 2005, Kroll et al. 2010, Dugger et al. 2011, MacKenzie et al. 2012) .

Three studies have investigated territory habitat quality and its association with habitat amounts and patterns (Franklin et al. 2000, Olson et al. 2004, Dugger et al. 2005). For the purposes of these studies, habitat quality was defined as habitat fitness potential, an integrative metric that incorporates territory-specific estimates of survival and reproduction to generate a territory-specific estimate of lambda. In all three studies, habitat fitness potential values (indicating high quality territories) were highest for habitat conditions containing mature forest interspersed with a mix of other vegetation types.

Several consistent patterns emerge from recent research efforts devoted to understanding the factors that contribute to high quality owl habitat at core area and home range spatial scales (i.e., forest conditions that support viable territories). First, spotted owl occurrence and survival are significantly related to mature forest habitat. Second, higher owl survival and reproduction are also associated with areas that have a mix of different vegetation types and edge between mature forest and other vegetation types. Third, weather is an important factor influencing spotted owl demographics, particularly reproduction. Finally, spotted owl population dynamics are likely governed by both habitat and weather.

Landscape Scale

Spotted owls are a territorial species with each pair defending an exclusive territory, and they appear to exhibit an ideal despotic distribution where dominant individuals occupy the highest quality sites (Franklin et al. 2000, Zimmerman et al. 2003, Seamans and Gutiérrez 2006). In addition to habitat factors, the presence of other spotted owls in adjacent territories may also be a factor associated with the probability of a territory being selected (Seamans and Gutiérrez 2006). At the landscape scale, spotted owl territories tend to be more regularly spaced than randomly distributed. Little information is available at larger landscape scales on the relationship between landscape vegetation patterns and California spotted owl population density or number of territories, especially comparative information to assess how the number of territories varies across landscapes with different amounts and patterns of vegetation. Limited information suggests that the number of spotted owl territories may be related to landscape-scale habitat availability (Zabel et al. 2003). Additionally, creation of large gaps or large areas of low-quality habitat may affect dispersal of young and adult owls and successful colonization of unoccupied territories. Understanding of the relationships between landscape-scale vegetation condition and spotted owl territory density and dispersal behavior is a high priority for further research, as current forest fuels, restoration, and wildfire strategies are focused on larger landscape scales.

Effects of Forest Management and Wildfire

Forest Management

Two general approaches have been used to investigate the effects of forest management on spotted owls: (1) modeling, which is used to project the effects of forest treatments on spotted owls and their habitat; and (2) field-based studies, which measure the response of spotted owls to forest management effects. Simulation modeling suggests that landscape-scale fuels treatments on a small proportion of the landscape can minimize effects to owl habitat and reduce risk of habitat loss to wildfire (Ager et al. 2007, Lehmkuhl et al. 2007). Some treatments may also reduce fire risk within core areas with minimal effects on owl reproduction (Lee and Irwin 2005). Simulation modeling indicates that the long-term benefits of wildfire risk reduction may outweigh the short-term effects of treatments on spotted owl habitat (Roloff et al. 2012). Models have been developed to spatially integrate fuels treatments with protection of spotted owl habitat within landscape-scale restoration efforts (Prather et al. 2008, Gaines et al. 2010, Ager et al. 2012). Results from simulation modeling also suggest that fuels treatments can be effectively used to reduce wildfire risk and support restoration efforts while providing spotted owl habitat at home range and landscape scales.

Numerous observational studies have described spotted owl habitat associations (see review above), but very few studies have directly assessed the effects of fuels and forest treatments on spotted owls and their habitat. Researchers have long advocated for experimental studies to evaluate the effects of forest management on spotted owls (Noon 2002, Lee and Irwin 2005), and a lack of controlled experiments to test important hypotheses for forest management effects on spotted owls contributes to continuing controversy (Noon and Blakesley 2006). To date, no experiments using before-after-control-impact (BACI) designs have been conducted, with the exception of a study of forest treatment effects on owl foraging on the Eldorado National Forest (R. Gutiérrez, pers. comm., unpub data). These BACI studies are scientifically challenging to design, logistically difficult to implement, and expensive to conduct. Given the inherent variability in spotted owl populations, large numbers of individual owls or owl pairs in experimental and control groups may be necessary to have adequate statistical power to detect effects. Planning regulations, regulatory requirements (e.g., NEPA), and legal challenges make it logistically difficult to implement extensive treatments across space and time to meet rigorous scientific design requirements. In lieu of experimental studies, passive adaptive management approaches (sensu Kendall (2001)) have been used to investigate the effects of forest management on California spotted owls. Under a passive adaptive management framework, managers dictate the implementation of treatments in time and space governed by management priorities. Researchers attempt to establish a baseline and monitor changes in owl response using models to evaluate the evidence for treatment effects on observed responses. Inferences from these quasi-experimental, passive adaptive management approaches are weaker than those from BACI designs because observed responses may result from other uncontrolled factors. Under a BACI design, or active adaptive management (sensu Kendall (2001)), treatments and controls are implemented in space and time to meet rigorous experimental design criteria as governed by research priorities. Nevertheless, these passive adaptive management approaches may be the best option in situations where true experiments are not scientifically, logistically, or financially possible, or where political will may be lacking.

Seamans and Gutiérrez (2007a) reported that California spotted owl territories with more mature forest had higher probabilities of being colonized and lower probabilities of becoming unoccupied. Alteration of ≥ 20-ha of mature forest in these spotted owl territories may decrease the probability of colonization.

It is unclear whether breeding dispersal or other factors, such as lower survival, are associated with variability in the probability of a territory becoming unoccupied. Nor is it clear if the probability of a territory becoming unoccupied is related to the amount of mature forest within or among territories (Seamans and Gutiérrez 2007a).

Three ongoing studies may provide further insight into the effect of forest management on California spotted owls. Twenty years of demographic monitoring at four study sites distributed across the Sierra Nevada has provided an unparalleled long-term data set on owl occupancy and demographics. Efforts are underway to develop post-hoc annual vegetation maps for each study area that can facilitate a retrospective meta-analysis to assess habitat associations and investigate the effects of forest management on California spotted owl occupancy, survival, and reproduction across the four study areas. Results from the Plumas-Lassen Study and the Sierra Nevada Adaptive Management Project will also provide further modeling and monitoring of forest treatment effects on California spotted owls. These ongoing efforts will further understanding of forest treatment and wildfire effects on California spotted owls and their habitat in the Sierra Nevada.

Wildfire

Current information indicates that California spotted owls will occupy landscapes that experience low- to moderate-severity wildfire, as well as areas with mixed-severity wildfire that includes some proportion of high-severity fire. Bond et al. (2002) reported that first-year post-fire adult survival and site fidelity were similar at 11 territories that had experienced wildfire compared to unburned sites across the range of northern, Mexican, and California spotted owls. In contrast, Clark et al. (2011) reported lower adult owl survival 1 – 4 years post fire in eastern Oregon. However, their results were likely affected by past logging effects and post-fire salvage logging that resulted in low overall amounts of remaining suitable habitat after the wildfires. Jenness et al. (2004) reported no effects of mixed-severity wildfire on Mexican spotted owls at burned sites that experienced an average of 16 percent high-severity fire. However, Jenness et al. (2004) stated that the statistically non-significant higher occupancy and higher reproduction observed at unburned sites (31 sites) compared to burned sites (33 sites) were suggestive of a biologically significant effect. They recommended that their results, which indicate that wildfires do not affect spotted owls, should be interpreted cautiously because of concerns that limited sample sizes and high variability in both burn extent and severity across their burned sites may have limited ability (i.e., low statistical power) to detect biologically meaningful differences between burned and unburned sites. In Yosemite National Park, Roberts et al. (2011) estimated that spotted owls had similar occupancy and density between unburned (16 sites) and recently burned (16 sites) (<15 years since burn) montane forests that burned primarily at low to moderate fire severity. Lee et al. (2012) reported no difference in owl occupancy between unburned and burned territories from six fire areas in the Sierra Nevada. Further, Lee et al. (2012) concluded that the proportion of high-severity fire (an average of 32 percent of suitable vegetation burned within analysis areas) had no effect on post-fire occupancy, although the amount of high-severity fire was not included in models of occupancy, colonization, and extinction and was only qualitatively assessed relative to burned sites.

Little information is available on patch-scale habitat use in post-fire landscapes. Bond et al. (2009) reported that owls nested and roosted in unburned or low- to moderate-severity patches of forest, and, four years after fire, they foraged selectively in high-severity burn patches that were located within larger home ranges that generally burned at low to moderate severities. Patches of early-successional vegetation recovering from high-severity fire may provide access to early-successional associated prey, such as woodrats and gophers, within the mosaic of mixed fire severity landscapes. Additional information is needed on habitat use by spotted owls in post-fire landscapes. Further, it is important to know if the owls using the post-fire landscapes are the original occupants or whether the post-fire site was colonized by different owls to more fully understand the effects of wildfire on spotted owls.

Recent findings indicate that California spotted owls are able to persist in landscapes that experience low- to moderate-severity and mixed-severity wildfires. However, several key uncertainties remain regarding long-term occupancy and demographic performance of spotted owls at burned sites. Specifically, uncertainty exists regarding how increasing trends in the amounts and patch sizes of high-severity fire will affect California spotted owl occupancy, demographics, and habitat over longer time frames. Additionally, further information is needed on the effects of post-fire salvage logging on spotted owl habitat.

Additional Ecological Stressors

Barred Owls

Barred Owl (*Strix varia*) range expansion has posed a significant threat to the viability of the northern spotted owl (Gutiérrez et al. 2007). Barred owls are native to eastern North America but have expanded their range westward into Washington, Oregon, and northern California in the past 40 years and are now found throughout the entire range of the northern spotted owl. During initial colonization, barred owls may hybridize with spotted owls to produce hybrids (i.e., sparred owls). As barred owl numbers increase in a local area, they pair with their own species. Compared to spotted owls, barred owls are larger, are active both day and night, consume a broader diet, have smaller home ranges, and occur at higher population densities. These factors render them competitively dominant over spotted owls, and they have displaced or replaced northern spotted owls over many portions of their range. Inferences regarding barred owl effects on spotted owls must be tempered by the fact that studies to date are based on observational and correlational studies and require confirmation through experimental assessment of barred owl effects. Results to date suggest two key findings: first, barred owls have replaced or displaced northern spotted owls over large areas of their range through the hypothesized mechanism of interference competition (Dugger et al. 2011); and second, although little information is available on how forest management affects spotted-barred owl interactions, there is recent evidence suggesting habitat patterns can influence occupancy and colonization dynamics (Dugger et al. 2011, Yackulic et al. 2012).

Barred owls are an increasing risk factor for California spotted owls in the Sierra Nevada. Barred owls were first recorded within the range of the California spotted owl in 1989 on the Tahoe National Forest. Two sparred owls were reported in the Eldorado Demographic Study Area during 2003 – 2004 (Seamans et al. 2004), and one of these sparred owls is still present on the study area. Barred owls were first recorded in the southern Sierra Nevada in 2004 (Steger et al. 2006). Ongoing research has documented

73 records of barred or sparred owls in the Sierra Nevada to date, with the majority of records from the northern Sierra Nevada (Tahoe, Plumas, and Lassen National Forests). Of note, five new records of barred owls were documented in the Stanislaus and Sierra national forests in 2012, indicating further range expansion of barred owls in the southern Sierra Nevada.

Barred owl numbers are likely higher than documented in the Sierra Nevada, as there have been no systematic surveys for them to date. Rather, barred owls are recorded during spotted owl surveys or reported by the public. Spotted owl surveys are conducted annually within the four demographic study areas, however, outside of these study areas, spotted owl survey efforts are limited and sporadic over space and time in response to local project survey requirements. Thus, it is likely that additional barred owls are present in the Sierra Nevada given limited spotted owl survey work outside of the demographic study areas. Further, species-specific survey methods are required to account for differences in response behavior and imperfect detection between spotted and barred owls (Wiens et al. 2011).

Climate Change

Across their range, spotted owls exhibit population-specific demographic relationships with local weather and regional climates (Glenn et al. 2010, Glenn et al. 2011, Peery et al. 2012). Based solely on projections of climate change (i.e., not incorporating other factors such as habitat, etc.), this population-specific variation is anticipated to result in population-specific responses to future climate scenarios, which could range from little effect to potentially significant effects. These population-specific responses could result in high vulnerability. For California spotted owls, Seamans and Gutiérrez (2007b) reported that temperature and precipitation during incubation most affected reproductive output, and conditions in winter associated with the Southern Oscillation Index (SOI) most affected adult survival on the Eldorado National Forest. Weather variables explained a greater proportion of the variation in reproductive output than they did for survival. Further, these two weather variables were also included in the best models predicting annual population growth rate (Seamans and Gutiérrez 2007b). MacKenzie et al. (2012) found that SOI or other weather variables explained little variation in annual reproduction for this same population of owls. Unlike results for California spotted owls in southern California reported in Peery et al. (2012), subsequent analyses testing for effects of weather variables on demographic parameters showed no clear temporal associations for owls on the Eldorado National Forest in the Sierra Nevada (R. Gutiérrez, pers. comm.).

Future responses to climate change are likely to be governed by complex interactions of factors that directly affect spotted owls and their habitat, as well indirect factors that can affect habitat (e.g., insect pests, disease, increased fire risk, etc.). Carroll (2010) recommended that dynamic models that incorporate vegetation dynamics and effects of competitor species in addition to climate variables are needed for rigorous assessment of future climate change on spotted owls.

Disease and Contaminants

Disease can function as an important ecological limiting factor in wildlife populations, especially in the case of invasive diseases introduced into naïve populations that have not co-evolved mechanisms to cope with the risk. Little information exists on disease prevalence in California spotted owl populations, and no information exists regarding the effects of disease on individual fitness or population viability.

Blood parasite prevalence sampling for California spotted owls in the northern Sierra Nevada documented that 79 percent of individuals were positive for at least one infection, whereas 44 percent of individuals tested positive for multiple infections (Ishak et al. 2008). Gutiérrez (1989) reported 100 percent blood parasite infection rates across all three spotted owl sub-species, suggesting long-term adaptation to high parasitism rates.

West Nile Virus (WNV), a mosquito-borne flavivirus, was first detected in eastern North America in 1999 and spread rapidly across the continent. WNV was first detected in southern California in late 2003 and spread throughout California in late summer of 2004 (Reisen et al. 2004). WNV has been demonstrated to have high acute species-specific mortality rates in many raptor species (owls, hawks, and their relatives) (Gancz et al. 2004, Marra et al. 2004). None of the 141 individual California spotted owl blood samples collected from the southern (Sierra National Forest, Sequoia-Kings Canyon National Park) or northern (Plumas and Lassen National Forests) Sierra Nevada from 2004 – 2008 have tested positive for WNV antibodies, which would indicate exposure and survival (Hull et al. 2010). Adult, territorial California spotted owls have high annual survival (80 – 85 percent) that has been stable across years, and no evidence has been published from the four long-term demographic studies indicating changes in adult owl survival. Nevertheless, although no effects have been documented to date, future outbreaks of WNV may pose a risk to California spotted owls.

Environmental contaminants have not been identified as potential ecological stressors on California spotted owls. However, recent reports of high exposure rates of fisher (*Martes pennanti*) to rodenticides, likely associated with illegal marijuana cultivation, across the southern Sierra Nevada (Gabriel et al. 2012) may have implications for spotted owls and other forest carnivores, as they feed extensively on rodents.

Integration of Current Forest Management with California Spotted Owl Management and Conservation

Recent research suggests that fundamental changes in forest management may be required to promote resilience, given current forest conditions resulting from historical management practices, a changing climate, and an increased focus on wildfire, both as a threat to habitat and human values, as well as the primary natural disturbance agent that has historically shaped vegetation structure and function (North et al. 2009, Perry et al. 2011, Larson and Churchill 2012, North et al. 2012). North et al. (2009), North (2012), and North et al. (2012) have proposed a conceptual forest restoration framework that focuses management perspective across multiple spatial and temporal scales when identifying future desired conditions. Key operating concepts focus on fine-scale vegetation heterogeneity resulting from the primary role of fire, embedded within landscape vegetation patterns that are influenced by topography, elevation, latitude, and natural fire regimes over longer temporal scales and larger spatial scales. At the core of this framework is the hypothesis that forest resilience may best be realized by approaches that restore or mimic Sierra Nevada forest structure and function under a wildfire-dominated natural disturbance regime, with the goal of reintroducing fire as an important process to these systems where and when possible (North et al. 2009, North et al. 2012).

One key measure of success for the proposed forest restoration framework focused on forest resilience will be to sustain biodiversity and well-distributed, viable populations of sensitive, focal wildlife species, such as the California spotted owl, fisher, and marten. Similar to the multi-scale management considerations for forests, management of California spotted owls and their habitat has parallel considerations of spatial scales, ranging from patches containing specific habitat elements used for nesting and foraging(e.g.., large trees, downed logs) to landscapes capable of supporting high-quality territories and viable populations. For the past 20 years, California spotted owl management has been based on recommendations provided by the California Spotted Owl Technical Report (Verner et al. 1992).This strategy consisted of establishing protected activity centers (PACs) of approximately 300 acres for each owl site, and using forest treatments designed to maintain large trees within treatment units. Following the Sierra Nevada Forest Plan Amendment (U.S. Forest Service 2001, 2004), management was adjusted to include a 1000 acre home range core, where an additional 700 acres outside of the PAC is designated and managed as foraging habitat around core areas. Originally, the PAC concept was adopted as an interim strategy to reduce risk from timber harvest and protect the large trees and mature forest known to be important habitat around spotted owl core areas, as well as large trees throughout treated forest areas, until a more comprehensive forest management strategy could be developed. Little evaluation of the PAC strategy, and no evaluation of CASPO forest treatments, has been conducted, but recent work documents that PACs have been successful in protecting important nesting and roosting habitat and that owls use these PACs over very long periods of time (Berigan et al. 2012). However, it is uncertain if a reserve-based PAC strategy can effectively provide habitat to support a viable population of California spotted owls over the long-term in the wildfire-structured forests of the Sierra Nevada, given increasing trends in wildfire acres burned and amounts of high-severity wildfire associated with contemporary forest fuel loads and projected future climate change scenarios (Ager et al. 2007, Miller et al. 2009, Gaines et al. 2010, Ager et al. 2012, Roloff et al. 2012).

Recommendations proposed by North et al. (2009) focus on using topography, aspect, elevation, latitude, and desired vegetation conditions more consistent with patterns that would result under a natural disturbance regime structured by wildfire to guide management decisions. Management of denser forest habitat conditions would be targeted for topographic locations where wildfires would have burned less frequently or at lower severities, such as northerly aspects, canyon bottoms, and riparian areas. Conversely, south-facing slopes and ridge tops would be managed for more open vegetative conditions consistent with patterns that would be expected under a more natural fire regime. Nested within the overarching landscape scale, vegetation treatments would focus on generating patch-scale heterogeneity in forest structure and composition thought to be more consistent with conditions generated by a frequent, predominately low- to moderate-severity fire regime. Whether the management strategy proposed by North et al. (2009) can provide for viable populations of California owls is a hypothesis that requires field testing and validation.

No information is available on historical spotted owl distribution, numbers, or habitat associations under pre-European forest conditions in the Sierra Nevada. California spotted owls were present in these forests, but no information is available to evaluate how they interacted with forest landscapes that were generally less dense, more heterogeneous at multiple spatial scales, and dominated by large trees.

Further, there is no information on the population size and densities of California spotted owls that occurred under these historical conditions. Thus, there is no base of historical information to inform how California spotted owls might respond to future conditions that may be more similar to the pre-European forest conditions in the Sierra Nevada. However, results from recent research on spotted owl habitat associations provides a strong basis for identifying and managing for vegetation types (e.g., mature forest) and habitat elements (i.e., large trees, logs, and snags) important to spotted owls.

Recent empirical studies of spotted owl habitat associations consistently reinforce the importance of large trees and mature forest habitat at the stand, core area, home range, and landscape scales (Franklin et al. 2000, Olson et al. 2004, Dugger et al. 2005). Hence, management to protect and enhance large trees and mature forest habitat and their resilience to wildfire and climate change is an important foundational piece for a successful strategy to maintain viable populations of California spotted owls in the Sierra Nevada. However, recent studies have also indicated that vegetation heterogeneity is a component of high-quality habitat at core area and home range spatial scales, and that owls will forage in a broader range of vegetation conditions relative to nesting and roosting habitat. Simulation modeling studies also project that forest fuels and restoration treatments may be compatible with maintaining spotted owl habitat (Ager et al. 2007, Lehmkuhl et al. 2007, Ager et al. 2012, Roloff et al. 2012).

Considering that California spotted owls evolved within heterogeneous Sierra Nevada forests structured by wildfire as the primary disturbance agent, as well as results from recent empirical and modeling studies, it’s reasonable to expect that carefully crafted forest treatments can meet fuels and restoration objectives and provide habitat for California spotted owls. However, several caveats must be considered and uncertainty exists regarding how treatments will affect owl populations and their habitat. First, current understanding of California spotted owl habitat associations is based on studies conducted under contemporary forest conditions in the Sierra Nevada, which are shaped by timber harvest and fire suppression policies that have resulted in significant reductions in large trees and mature forest, and increases in forest homogeneity across stand and landscape spatial scales. It is uncertain if current habitat conditions are optimal for spotted owls and how owls may respond to future vegetation conditions that shift the landscape vegetation trajectory toward a condition more similar to patterns that would be expected under a natural disturbance regime largely driven by wildfire, but also influenced by other natural disturbances, such as insects, disease and wind. Second, uncertainty exists regarding what constitutes high-quality spotted owl habitat capable of maintaining territories and viable populations over time. Current ongoing research investigating California spotted owl demography-habitat associations will help address these knowledge gaps.

Current population status and declining population trends of spotted owls on NFS lands in the Sierra Nevada point to the need for a careful approach to management of California spotted owls and their habitat. Although the causes of the population declines are unknown, it is likely that historical and current vegetation management practices (primarily timber harvest and fire suppression) are a factor. Additionally, range expansion of the barred owl into the Sierra Nevada is a serious emerging threat to California spotted owl viability. Recent research indicates that comprehensive forest management strategies are required to address forest resilience and restoration in the Sierra Nevada, given current vegetation conditions, trends in wildfire, and projected climate change (North et al. 2009). Considerable scientific uncertainty exists regarding how California spotted owls and their habitat will be affected by the management direction proposed in North et al. (2009). Adaptive management and monitoring of California spotted owls and their habitat will be an important element of a management strategy to address forest resilience and restoration in the Sierra Nevada.

Adaptive Management, Monitoring, and Information Needs

A number of tools and assessments can be used to address current scientific uncertainty about how California spotted owls and their habitat will respond to the management direction proposed in North et al. (2009). Ongoing research will address some of the information needs, whereas other needs can be met by tailoring existing modeling tools and approaches that have been developed for other applications to specifically address California spotted owls in the Sierra Nevada. Key information needs are highlighted below.

1) Integrated conservation planning efforts are needed to synthesize existing and ongoing research efforts and develop adaptive management planning and assessment tools to inform forest management. Improved habitat models specific to California spotted owls in the Sierra Nevada are a core need to further understanding of owl-habitat associations and to develop adaptive management tools to assess how owls may respond to management scenarios. The current effort to conduct an integrated meta-analysis to relate over 20 years of California spotted owl demographic data to changes in habitat over 20 years across the four long-term demographic study areas in the Sierra Nevada will provide the most comprehensive assessment of owl habitat associations to date. Models from this effort can be further developed to function as adaptive management tools. Similarly, ongoing research developing patch-scale nesting and foraging models using forest inventory and analysis (FIA) data can be developed into adaptive management tools for assessing treatment effects.

2) Results from climate change assessments across the three sub-species of spotted owls indicate population-specific associations with weather factors and projected responses to future climate change (Glenn et al. 2010, Peery et al. 2012). Modeling is needed to project how California spotted owl populations in the Sierra Nevada may respond to future climate and habitat scenarios. The four ongoing demographic studies provide existing long-term data on owl populations across the Sierra Nevada that can inform this assessment.

3) Models at home range and landscape scales are needed to assess trade-offs between wildfire risk and treatment effects over short and long time periods specific to the Sierra Nevada (e.g. Lee and Irwin 2005, Scheller et al. 2011, Thompson et al. 2011, Ager et al. 2012, Roloff et al. 2012).

4) Improved information on vegetation status, structure, and condition is needed to facilitate development of habitat models and to assess the effects of treatments on California spotted owls and their habitat. Current, widely-available vegetation data are not consistent across the Sierra Nevada and vary among forests. At patch scales, existing vegetation data are not

adequate to describe the finer-scale heterogeneity that will result from the proposed new management direction. New vegetation information is required that describes finer-scale heterogeneity that can then be used to model and assess owl habitat and treatment effects. Efforts to develop vegetation information that is better able to capture fine-scale vertical and horizontal heterogeneity, such as LIDAR or WorldView2 imagery, are promising (Hyde et al. 2005, Hyde et al. 2006, García-Feced et al. 2011), yet as of now, they are not available and operational enough to be able to conduct patch- to landscape-scale analyses of owl habitat associations.

5) Efforts are needed to assess the distribution and status of barred owls across the Sierra Nevada. Barred owls pose an increasing risk to California spotted owls in the synthesis area and require conservation planning focus.

6) Additional research is needed on the effects of high-severity fire on California spotted owl occupancy, population dynamics, and habitat given increasing trends in the amounts and patch sizes of high-severity fire in the Sierra Nevada (Miller et al. 2009).

7) Little information is available on how prey affects California spotted owl foraging behavior and population dynamics. Better understanding of California spotted owl-prey associations across different vegetation types and elevations would be beneficial for tailoring treatments and desired landscape conditions across different regions of Sierra Nevada. Additionally, a synthesis of the literature on small mammal habitat associations in the Sierra Nevada, and comparative information from other western forests, would be valuable for identifying important forest types, stand structures characteristics, and habitat elements important to small mammals that could be used to inform future management.

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8.0 Air Quality

Andrzej Bytnerowicz, Mark Fenn, and Jonathan Long

Contents

Executive Summary

The major pollutants causing ecological harm in the Sierra Nevada are ozone, which can be toxic to plants, and nitrogen deposition, which can induce undesirable effects on terrestrial and aquatic ecosystems. Other airborne pollutants of concern include black carbon, particulate matter, pesticides, and heavy metals including mercury. Atmospheric pollutants that are delivered in wet and dry forms cause deposition of nitrogen to forests and other land areas. The highest potential for ozone to injure plants occurs on western, low-elevation slopes that have elevated daytime levels that coincide with the highest physiological activity of plants. However, recent evaluations of ozone injury in the Sierra Nevada are lacking. Ozone and nitrogen deposition interact with other environmental stressors, especially drought and climate change, to predispose forests to impacts of pests and diseases.

Impacts of air quality currently pose threats to public health and recreation along the western slopes of the southwestern Sierra Nevada, which experience frequent episodes of unhealthy air, as indicated by exceedances of ozone and particulate matter (PM) air quality standards. High levels and variation in day and nighttime ozone values can also occur at remote, high-elevation locations affected by pollution from distant areas; these locations can also have sufficient ozone precursors and meteorological conditions that favor localized photochemical ozone formation.

Emissions from wildfires and prescribed fires have potential to exceed air quality health standards, especially for particulate matter (PM2.5 and PM10). Because of the relatively low probability of wildfire in any given area, expected emissions from a regime of prescribed burning may often exceed those from wildfire. However, prescribed burning can be managed more easily to mitigate air quality impacts to people. Furthermore, the potential of prescribed fires to generate enough ozone to exceed federal or state air quality standards is limited because typically they are smaller, are less intense, and occur during periods of low potential for photochemical ozone formation. Better understanding of the impacts of wildland and prescribed fires on ambient ozone, nitrogenous pollutants, and nitrogen cycling would help to understand their potential effects on human health and the sustainability of forest ecosystems.

Figure 1: Smoke obscured visibility at Lake Tahoe on June 28, 2008. During that time, much of northern California was blanketed in smoke from large wildfires that reduced visibility, caused hazardous levels of air pollution including particulate matter, and forced cancellation of outdoor recreation events such as the Western States 100 mile endurance run. Photo by Jonathan Long.

Values at Risk from Air Pollution

Air pollution impacts a variety of ecosystem services, including supply of clean water, public health, regulation of greenhouse gases, recreational values, as well as growth and health of forests and biodiversity (Cisneros et al. 2010). However, quantifying impacts to ecosystem services will require integrative research at larger scales than the individual plants and forest stands that have been a focus of most research (Serengil et al. 2011). Several recent peer-reviewed publications address air pollution status and effects across the Sierra Nevada (Arbaugh and Bytnerowicz 2003, Fenn et al. 2003a, Fenn et al. 2010), and others focus specifically on the central Sierra Nevada (Hunsaker et al. 2007) and Sequoia and Kings Canyon National Parks (Bytnerowicz et al. 2002). Impacts of air quality currently pose threats to public health and recreation along the western slopes of the southwestern Sierra Nevada, which experience frequent episodes of unhealthy air, as indicated by exceedances of ozone and particulate matter (PM) air quality standards (Cisneros et al. 2010). In addition, impacts of other pollutants that may have significant biological effects, such as pesticides and mercury, are not very well characterized in the Sierra Nevada, but should also be taken into account.

Ozone

In many parts of the American West—especially the southwestern portions of the Sierra Nevada (Carrol et al. 2003, Grulke 2003)—increasing background levels of ozone have already approached thresholds of phytotoxicity. High levels of ozone have been measured in the California Central Valley (CCV) and southern Sierra Nevada since the early 1970s (Miller et al. 1972). These episodes are mainly caused by transport of polluted air masses from the highly polluted San Francisco Bay Area (SFBA) and the CCV. Polluted air masses from the SFBA move east into the Sacramento Valley (SV), where they circulate near Sacramento and move northwest along the Sierra Nevada western slopes. The polluted SFBA air masses also move southeast into the San Joaquin Valley (SJV), where they mix with the locally polluted air. Cool air masses descending from the Sierras at night create the Fresno eddies that circulate polluted air within the SJV along the Sierra slopes (Beaver 2008). These air currents, daytime eastward movement of air up the canyons into the Sierra crest, and long-range transport of air pollution from southern California affect air pollution distribution in the Sierra Nevada (California Surface Wind Climatology, 1984, Carle, 2006). Distribution of ozone concentrations in summer 1999 illustrates typical summer patterns in the Sierra Nevada (Figure 2; Frączek et al. 2003), which occur on 72 percent of the warm-season days (Carroll et al. 2003) (see Sidebar on Current Research at the end of this chapter regarding efforts to update these maps). These general patterns were confirmed in various recent studies; for example, elevated concentrations of ozone were reported in western parts of Sequoia and Kings National Park (Bytnerowicz et al. 2002), western and southern portions of Yosemite National Park (Burley and Ray 2007), and the western side of the Sequoia National Forest (Cisneros et al. 2010). Although ambient mean ozone concentrations show only a slight decline along a west-east Sierra transect along the wide San Joaquin River drainage (Cisneros et al. 2010), the highest phytotoxic ozone potential occurs on western, low-elevation slopes that have elevated daytime values that coincide with the highest physiological activity of plants. In locations that are close the CCV urban areas, nighttime ozone concentrations are much lower than daytime concentrations due to titration of ozone by nitric oxide (Bytnerowicz et al. 2002, Burley and Ray 2007).

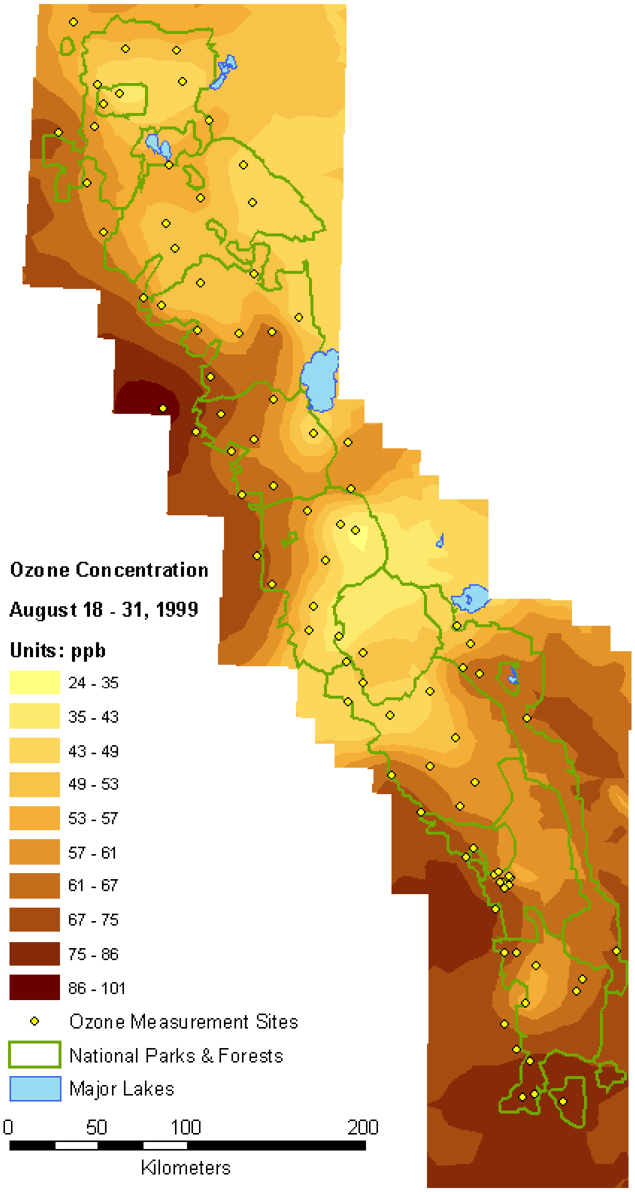

Figure 2: Distribution of ozone during the second part of August, with the intrusion of ozone into the Sierra Nevada from the California Central Valley to the west of the study area, as well as high concentrations of ozone in the southern part of the range and in the Owens Valley to the east of the study area (Frączek et al. 2003).

High diurnal ozone variation and elevated daytime values can also occur at remote, high-elevation locations affected by long-range transport of polluted air masses; these locations also have sufficient ozone precursors and meteorological conditions that favor local photochemical ozone formation (Bytnerowicz et al. 2013). Some high-elevation sites may experience elevated evening and nighttime concentrations due to transport of free-troposphere ozone (Burley and Ray 2007), whereas others may have low nighttime values when such transport does not occur and deposition to wet surfaces (meadows) takes place (Burley and Ray 2007, Bytnerowicz et al. 2013). Background ozone concentrations measured at remote Sierra Nevada locations are generally in agreement with those measured at high elevation (up to 4,250 m) in the White Mountains near the California – Nevada border of approximately 50-55 ppb as the summertime average (Burley and Bytnerowicz 2011) (Figure 2). This implies that any “remote” location in the Sierra Nevada may have background ozone close to 60 ppb, which is considered potentially phytotoxic (Fowler et al. 1999). Due to a steady increase of background ozone in the western United States (Brasseur et al. 2001), it may be a potential threat to sensitive vegetation in future years.

Figure 3: Air pollution monitoring site with active ozone instrument and passive samplers in the White Mountains during 2007. Research was conducted by collaborators from the USDA Forest Service Pacific Southwest Research Station, University of California, and the Saint Mary’s College. Photo credit Andrzej Bytnerowicz.

Ozone negatively impacts vegetation of the Sierra Nevada; effects on pines and other conifers were first reported east of Fresno in the western portions of Sequoia National Forest and Sequoia National Park (Miller and Millecan 1971). Permanent plots of forest health assessment (including effects of ozone) were established in 1974-1975 by the USDA Forest Service Forest Pest Management (FPM). Monitoring results showed that chlorotic mottle and premature needle senescence, well-known ozone injury symptoms, were common and widespread, especially among ponderosa (*Pinus ponderosa*) and Jeffrey (*P. jeffreyi*) pines. Such symptoms, although less pronounced, were also reported for a few other species (Williams et al. 1977). Between 1977 and 1987, a network of ozone injury evaluation plots was established throughout the Sierra Nevada. On that network, ozone injury was evaluated by the ozone injury index (OII) method developed by Miller et al. (1996), in which chlorotic mottle and needle retention are the basis of the assessment (Grulke 2003). Between 1977 and 1987, symptoms of ozone injury were found all over the Sierra Nevada on over 20 percent of the sampled ponderosa and Jeffrey pines. Severity of injury ranged from slight in the north to moderate/severe in the south, with the worst injury at elevations below 1,800 m. Injury decreased from the west to the east across the Sierras as distance from the source of photochemical smog (largely from the CCV) increased (Carroll et al. 2003). Highest injury to the surveyed pines was determined in Sequoia – Kings Canyon NP (39 percent of pines with chlorotic mottle), and Yosemite NP (29 percent of rated trees with injury symptoms). Ozone injury evaluation was repeated on a subset of the FPM plots in the Sierra and Sequoia national forests in 2000. That survey showed a major increase of trees with chlorotic mottle—from 21 percent of all trees in 1977 to 40 percent in 2000. On the southern Sierra Nevada plots, severe ozone injury resulted in 7 percent of mortality of trees over a period of 23 years (Carroll et al. 2003). Although ozone was a predisposing damaging factor in tree mortality, other factors, such as drought (exacerbated by climate change, densification of stands, and nitrogen deposition) and various species of bark beetles, such as western bark beetle (*Dendrocronus brevicomis*), Jeffrey pine beetle (*D. jeffreyi*), or mountain pine beetle (*D. ponderasae*), are the ultimate cause of tree mortality (Minnich and Padgett 2003, Fenn et al. 2003). It should be stressed that ozone phytotoxicity depends on the amount (dose) of ozone taken up by stomata and various abiotic and biotic factors (Matyssek et al. 2007). It has been shown for the ponderosa pine stands in the foothills that only 37 percent of total ozone deposition occurs in summer and that stomatal uptake accounts for less than half of that deposition (Goldstein at al. 2003). More recent evaluation of tree health in relation to ozone effects would help to better understand the condition of forests in the synthesis area.

Nitrogen Deposition

Forests on the western slopes of the Sierra Nevada receive substantial amounts of airborne nutritional nitrogen (N) that could have effects on nitrogen cycling, water quality, tree health, biodiversity, and sensitive indicator species, including lichens (Figure 4) (Fenn et al. 2010). Fenn et al. (2010) showed that overall N deposition ranges from about 2 to 20 ha^{-1} yr^{-1} in the Sierra Nevada, with the lowest levels in the northern region and the eastern side of the mountains, moderate levels (5-12 ha^{-1} yr^{-1}) in the central Sierra Nevada, and the highest levels of deposition (ranging from 15 to 20 kg ha^{-1} yr^{-1} or greater) occurring in the southwest part of the region (Figure 5). Concentrations of the nitrogen pollutants that are the main drivers of nitrogen dry deposition drop significantly in the Sierra Nevada as air masses move eastward (Bytnerowicz et al. 2002). Polluted air masses can move deep into the Sierra range up the canyons and valleys; for example, the San Joaquin River drainage functions as a corridor for transport of pollutants to the eastern side of the Sierras (Cisneros et al. 2010).

Figure 4: Sierra Nevada communities of tree-inhabiting lichens such as wolf lichen (*Letharia vulpina*) begin to change with atmospheric nitrogen deposition levels as low as 3 kg/ha/yr. Photo credit Mark Fenn.

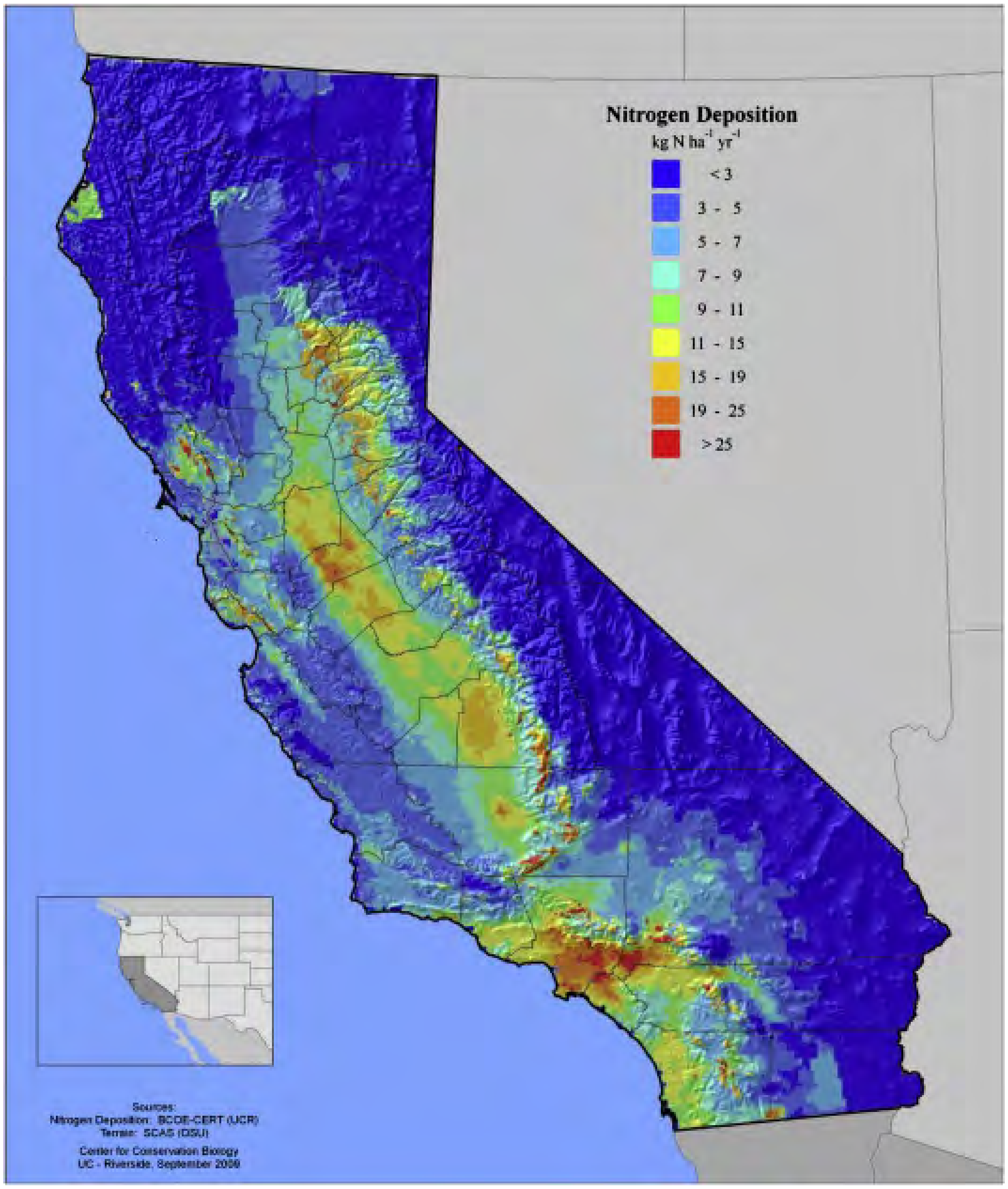

Figure 5: Map of total annual nitrogen (N) deposition in California based on simulations. Deposition inputs in the Sierra Nevada and other montane regions have been adjusted based on empirical deposition measurements (Fenn et al. 2010).

Nitrogen deposition can have a fertilizing effect on trees, reflected by increased above-ground growth and higher nitrogen tissue concentrations. Though fertilization has potential to enhance timber production, it poses a threat to forest composition, sustainability and function, as it alters nutrient cycles. Forecasting effects on forest composition is a challenge, but some have predicted that increases in soil nitrogen and ozone may reinforce shifts in forest composition associated with fire suppression by favoring firs over pines (Takemoto et al. 2001). Pines are generally more sensitive to ozone and excess N than firs and cedars (Fenn et al. 2003a, Grulke et al. 2009, Miller et al. 1983), and they are an important component of the mixed-conifer forest. Nitrogen enrichment can have other negative effects on biodiversity and ecological functions; for example, increased nitrogen can promote invasive grasses (Fenn et al. 2010), including cheatgrass (*Bromus tectorum*) (He et al. 2011), which can in turn have transformative effects on ecosystems by altering fire regimes, reducing carbon storage, and degrading forage quality (Bradley 2009).

Excess nitrogen deposition can also contaminate streams and ground water with nitrate, although throughout most of the Sierra Nevada, nitrogen appears to be well retained in the vegetation and soils. Fenn et al. (2010) identified “critical loads” of atmospheric nitrogen deposition below which sensitive elements of an ecosystem are not harmed. In Sierra Nevada mixed-conifer forests, they found elevated nitrate leaching in streams to be limited, with the most severe leaching losses less than 1 kg ha^{-1} yr^{-1} (Figure 6b). This is in contrast to epiphytic lichen-related critical loads, which are subject to widespread exceedances of nitrogen deposition throughout the range. Exceedances have also been noted for other vegetation types that occur in the Sierra Nevada, including pinyon-juniper, chaparral, and oak woodland. Furthermore, though nitrate leaching is limited, researchers have suggested that high-elevation lakes throughout the region may be experiencing eutrophication, which could result in increasingly severe ecological effects in the next several decades (Fenn et al. 2003b, Sickman et al. 2003). Accordingly, research is needed to evaluate the extent and impact of nitrogen deposition on high-elevation lake chemistry and biota in the Sierra Nevada.

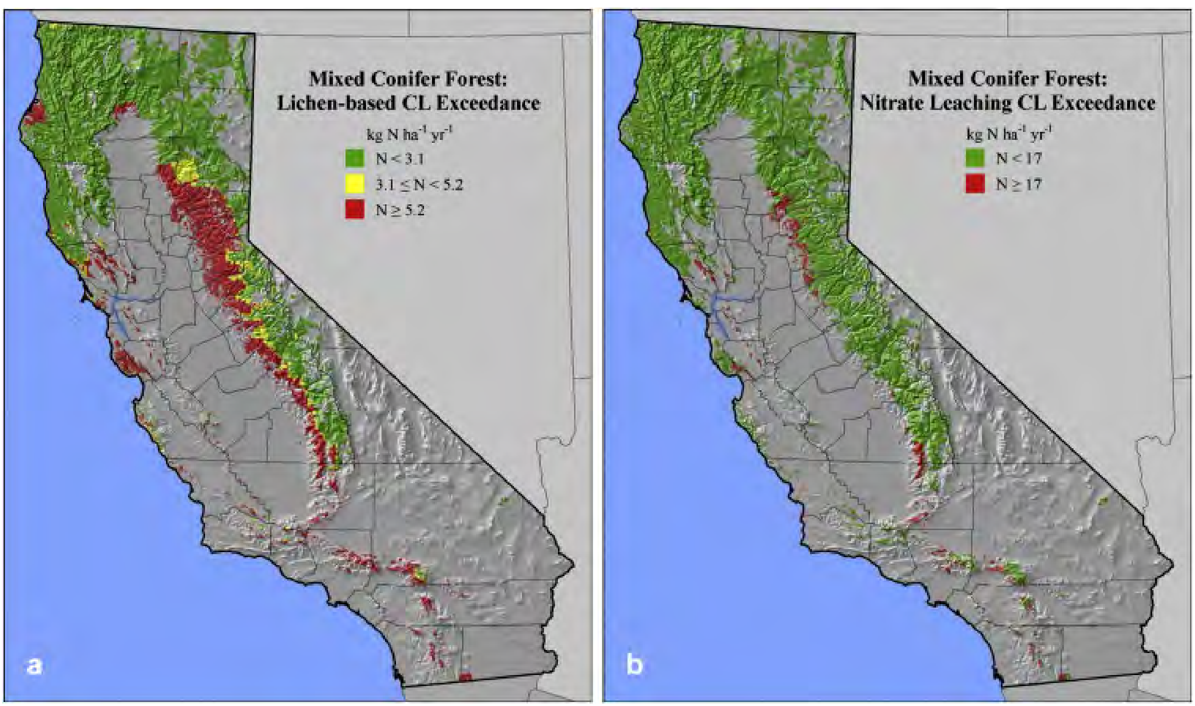

Figure 6: Critical load exceedance map for mixed-conifer forests based on (a) lichen community effects and (b) nitrate leaching (Fenn et al. 2010).

Interactive Effects of Ozone, Nitrogen Deposition, and Climate Change

A warmer climate in the western United States will affect the Sierra Nevada forests directly through soil moisture stress and indirectly through increased extent and severity of various disturbances. *Stress complexes*, a combination of biotic and abiotic stressors, will compromise vigor and, ultimately, the sustainability of forest ecosystems. Increased water deficit will accelerate normal stress complexes, which typically involve various combinations of long-term droughts, insects, and fire (McKenzie et al. 2009). In that general context, the combination of elevated ozone concentrations and nitrogen enrichment has already produced pronounced (and mostly negative) effects on California mixed-conifer forests ecosystems (Takemoto et al. 2001). These pollutants interact with other environmental stressors, especially drought, to predispose forests to impacts of pests and diseases. Through studies of nitrogen additions, researchers have found that nitrogen enrichment enhances mortality of ponderosa pines caused by bark beetles, as does ozone stress. Grulke (2009) highlights the San Bernardino Mountains as a case study in which multiple stressors, including ozone exposure, nitrogen deposition, and fire suppression, have predisposed forests to injury and mortality from bark beetles, drought, and fire. Though air pollution effects have been less severe in the Sierra Nevada than in the San Bernardino Mountains, chronic ozone exposure and nitrogen deposition are expected to become more prevalent, particularly in the southern part of the region (Takemoto et al. 2001, Fenn et al. 2003a). Furthermore, studies have shown synergistic effects of air pollution with other stressors; for example, in the southern Sierra Nevada, the negative effects of ozone on tree growth may be partially offset by nitrogen deposition, but the combined effects of ozone and chronic nitrogen deposition may lead to severe perturbation of tree physiology and ecosystem sustainability (Fenn et al. 2003a). Additionally, air pollutants may interact with climate change in complex ways that significantly differ from the sum of their separate effects (Bytnerowicz et al. 2007).

Figure 7: Symptoms of severe ozone injury in ponderosa pine foliage include chlorotic mottling, senescence or yellowing, and premature falling off of older needles. Photo by Mark Fenn.

Further research is needed to evaluate how nitrogen deposition and ozone affect carbon sequestration both aboveground and in the soil (Bytnerowicz et al. 2007). This information will be critical to climate change mitigation efforts in the region. Recent assessments suggest that many ecosystem and environmental responses to nitrogen deposition could lead to a net cooling effect, primarily as a result of enhanced carbon sequestration in woody biomass and increased haze and particles formed from nitrogen air pollution (Erisman et al. 2011), although there are many uncertainties in these evaluations. Many studies show that nitrogen enrichment and ozone exposure can lead to reduced carbon allocation belowground, resulting in greater carbon in aboveground detritus (e.g., Fenn et al. 2003a). Likewise, numerous studies confirm that long-term decomposition of litter slows down when nitrogen concentrations in litter are elevated; this may result in greater carbon storage in litter, especially during long fire-free periods (Whittinghill et al. 2012). However, when these polluted forests experience fire, more carbon may be released from burning litter.

Impacts of Other Pollutants

Other pollutants of concern include black carbon, particulate matter, pesticides, and heavy metals. Black carbon and dust particles pose a threat to water resources by promoting earlier melting of snowpack (Hadley et al. 2010). Although levels of methylmercury are relatively low in fish from Sierra Nevada lakes (Davis et al. 2009), mercury levels reported from sediments in Lake Tahoe are surprisingly high for alpine regions (Heyvaert et al. 2000). There is concern of long-range transport of semi-volatile organic compounds (SOCs), such as pesticides, polybrominated diphenyl ethers (PBDEs), polychlorinated biphenyls (PCBs), and polycyclic aromatic hydrocarbons (PAHs) in high-elevation aquatic and terrestrial ecosystems of the Sierra Nevada. Measurements have been performed within the Western Airborne Contaminants Assessment Project at the Emerald Lake and Pearl Lake area of Sequoia and Kings Canyon National Parks (SEKI), showing contamination of snowpack, lake sediments, vegetation, and fish. It was found that SEKI had the highest concentrations of current-use pesticides compared with other western national parks (Landers et al. 2008). Sources of these pollutants include wildfires, vehicles, urban and agricultural areas west of the Sierras, and, increasingly, long-distance transport from Asia (Heyvaert et al. 2000, Hadley et al. 2010). Wildfires have significant potential to mobilize heavy metals, including mercury, in ways that pose threats to human health (Goldammer et al. 2009) (see Post-wildfire Management chapter (4.3)). More research and monitoring of air, snow, vegetation, and lakes throughout the Sierra Nevada are needed to better understand spatial and temporal distribution of biologically important heavy metal and organic contaminants and their potential threats to ecosystems.

Forest Management Strategies to Address Pollutant Effects

In addition to direct pollutant load reduction, a prudent strategy to reduce the impacts of air pollution on forests would include treatments to reduce accumulated nitrogen in the forest by reducing stand stocking and fuel loads (Fenn et al. 2003a). To treat the problem of nitrogen saturation in highly polluted forests, such as the mixed-conifer forests of southern California, several papers have recommended the use of prescribed burning (Fenn et al. 2010, Gimeno et al. 2009). Although the conditions in the Sierra Nevada are generally less severe than in the mountains of southern California, frequent prescribed burning could help mitigate nitrogen inputs in forests experiencing elevated deposition. However, because prescribed fire has limited ability to reduce nitrogen in the mineral soil, Fenn et al. (2010) also suggest testing the potential of thinning to stimulate vegetation growth. Both thinning and prescribed fire can be used to proactively reduce the amount of plant matter available for combustion and reduce potential emissions of nitrogenous pollutants. However, long-term ecosystem protection and sustainability will ultimately depend on reductions in nitrogen deposition, and this is the only strategy that will protect epiphytic lichen communities. Measures to reduce nitrogen deposition through more stringent control of emissions caused by combustions of fossil fuels as well as those from the largely uncontrolled agricultural sector are needed. The critical load (CL) analyses and maps of CL exceedances are useful management tools for quantifying the severity of the pollution problem and identifying areas at risk from chronic nitrogen deposition. Also, there is a clear need for further decreases in ozone generation, and this can be accomplished through control of emissions of ozone precursors (nitrogen oxides and volatile organic compounds). Strict compliance with the federal and state ozone air pollution standards is needed. New measures, such as the federally imposed improved mileage of motor vehicles, could greatly help in reducing emissions of ozone precursors and lowering ambient ozone concentrations. If this is accomplished, future forests would be less stressed by direct phytotoxic ozone effects as well as secondary effects, such as increased susceptibility to drought and bark beetle attacks. Furthermore, the impacts of long-range mercury transport on Sierra Nevada aquatic ecosystems (especially high-elevation lakes) is still not well understood, and findings from the national Hg monitoring network administered by the National Atmospheric Deposition Program (NADP) should be evaluated and monitoring efforts intensified if needed. Additional monitoring efforts and research on potential impacts of long-range transport of pesticides and other potentially toxic organic compounds are needed to assess potential threats. If such threats are found, recommendations for stricter control of their use for agricultural production in the CCV should be made.

Fires, Smoke, and Air Quality

Prescribed fire and managed wildfire use entail a short-term impact to human communities to restore ecological processes and avoid the potential impacts of undesirably severe and poorly controlled wildfires. Fires release pollutants of concern, including fine particulate matter (PM 2.5), coarse particulate matter (PM 10), ammonia, carbon monoxide, carbon dioxide, nitrogen oxides, sulfur dioxide, and various volatile organic compounds (VOCs) (Urbanski et al. 2009, Cinseros et al. 2012). Use of prescribed fire as a management tool is constrained by state and federal air quality regulations for human health and visibility (Quinn-Davidson and Varner 2012), and potential smoke impacts to human populations (see chapter on Threats, Risks, and Health (9.3)).

A study of historical fire regimes and associated smoke emissions in California concluded that fires historically burned over extensive areas, and that smoke emissions were substantial, especially from the large areas of mixed-conifer forests that experienced frequent fire prior to the 19th century (Stephens et al. 2007). A long history of fire suppression has encouraged residents and visitors to the Sierra Nevada to expect exceptional visibility and smoke-free conditions during the summer and fall, but this may not be a realistic expectation for the area, especially under a changing climate projected to increase the likelihood of large, severe wildfires (e.g., Westerling et al. 2006). Many decades of altered fire regimes have also led to a large buildup of living and dead biomass in the understory and forest floor; in the Tahoe Basin, these accumulations represent a significant store of potential pollutants whether through high nutrient levels in runoff (Miller et al. 2010) or through emissions during combustion.

PSW researchers have worked with Region 5 Air Resources managers to study effects of wildland and prescribed fires on air quality in the context of state and national air quality standards. During severe fires, accumulated nitrogen in vegetation, litter, and surface soils may also be released as ammonia and nitrogen oxides (Urbanski et al. 2009), and these emissions could cause nitrogen deposition problems downwind of the fires (Goldammer et al. 2009). However, the potential of prescribed fires to generate enough ozone to exceed federal or state air quality standards is limited due to their low thermal intensity and geographic scale as well as their application during periods of low potential for photochemical ozone formation (Bytnerowicz et al. 2010). Better understanding of the impacts of wildland and prescribed fires on ambient ozone and nitrogenous pollutants is needed because of their potential effects on human health and the sustainability of forest ecosystems (Bytnerowicz et al. 2009). For instance, Preisler et al. (2010) detected a small but significant effect of wildfires on ambient ozone concentrations using the Blue Sky smoke dispersion model (O'Neil et al. 2009); however, these authors also pointed out serious weaknesses in monitoring and modeling approaches related to both wildland and prescribed fires. These are mainly related to the difficulty in distinguishing between fire-related ozone precursor emissions and emissions from non-fire anthropogenic sources, as well as complicated impacts of meteorology and complex mountain topography on ambient ozone concentrations (Preisler et al.2010).

Current Research: Air Quality in the Sierra Nevada

PSW researchers have obtained information on nitrogen air pollution and deposition in the Sierra Nevada using a variety of methods. They have generated maps of ozone air pollution for the entire Sierra and for the Lake Tahoe Basin, and they are working with UC Berkeley's Center for Forestry and the National Park Service to update maps of ozone distribution. They are also developing maps of critical levels for ozone, yielding a potential management tool for the Sierra Nevada, particularly the southern region. They are also developing maps of other pollutants (nitrogen oxides, ammonia, nitric acid, and sulfur dioxide) with data collected during 2006 – 2008 as part of research funded by the Joint Fire Sciences Program (JFSP). The southern Sierra Nevada has been a focus of air quality research, including a JFSP study that yielded several publications (both published and pending publication), including one on the ozone status of Devils Postpile National Monument in a low fire year (2007) and a high fire year (2008) (Bytnerowicz et al. 2013). In addition, in 2010, research was conducted on characterization of spatial and temporal distribution of ozone, its precursors, and nitrogen deposition in the Lake Tahoe Basin. In 2012, intensive study was conducted on ozone formation in the low- and high-elevation sites of the Lake Tahoe Basin. Results of these two studies will be published soon.

Effects of Management Strategies

A number of factors influence the amount and quality of emissions from burning, including fuel moisture, amount, and quality; these factors in turn are influenced heavily by weather and season. For example, burning material with higher moisture generally produces more carbon monoxide and ammonia, whereas burning drier fuels results in more complete combustion and greater release of smoke, carbon dioxide, and nitrogen oxides (Chen et al. 2010). The impacts of burning the forest in a prescribed burn are different from intense wildfire in important ways. First, intense wildfire often occurs in the summer under dry and windy conditions that facilitate smoke dispersal and lofting into the upper atmosphere (Cahill et al. 1996); however, dispersal depends upon local topography and weather conditions and is not assured. Even wildfires that occur under favorable ventilation conditions are still likely to cause emissions that exceed health and visibility standards (Gertler et al. 2010). Because wildfires have been relatively infrequent, their long-term average impact on respirable particular matter has often been relatively small (Cahill et al. 1996); however, a worsening of poor air quality days in the Lake Tahoe Basin has been linked to wildfires (Figure 1) (Green et al. 2012).

In contrast, managers can generally time and control prescribed burns to alter smoke production and transport in response to on a daily basis. Prescribed burns in many parts of the synthesis area, including the Lake Tahoe Basin, predominantly occur later in the fall when smoke tends to dissipate less readily.

As a result, models of prescribed burning under typical fall conditions indicate potential to violate air quality standards (Gertler et al. 2010) (see sidebar below). However, the modest amount of burning during the fall and winter, combined with protective measures to limit smoke, typically result in low contributions to PM10 particulate loading in inhabited areas (Cahill et al. 1996, Gertler et al. 2010).

A comparison of expected emissions from prescribed burning and wildfire would have to consider cumulative effects of prescribed burning. A recent modeling study found that prescribed burning would release less carbon dioxide than wildfire in frequent fire forest types of the Western U.S., but it assumed that the prescribed burning was so mild that it killed no trees and was conducted only once (Wiedinmyer and Hurteau 2010). These assumptions underestimate the severity and frequency of prescribed burning needed as a restorative practice in the synthesis area. The authors of that study indicated that cumulative prescribed fire emissions of carbon dioxide would likely be higher than wildfire emissions in cases where reestablishment of trees was relatively fast. However, treatments that prevent severe tree mortality due to wildfire would likely have an emissions benefit (Wiedinmyer and Hurteau 2010).

Sidebar: Comparisons of Wildfires and Prescribed Fire Effects in the Lake Tahoe Basin

The air quality effects of smoke under different scenarios have been compared in the Lake Tahoe Basin (LTB). The Lake Tahoe Air Model is a heuristic, cell-based predictive model of the LTB that was developed to analyze the effects of prescribed fires and wildfires on fine particle mass (PM 2.5) and visibility (Gertler et al. 2010). The model was used to compare impacts from a hypothesized regime of small, non-crowning wildfires burning 30 acres per day in the summer, a scenario intended to represent conditions prior to the mid-19th century. The results indicated that a regime of these “natural” wildfires would generate “spotty but persistent smoke in relatively low concentrations around the basin” that would not violate state and federal air quality standards, and have little impact on lake clarity (Gertler et al. 2010) (p. 71). The model analyses of prescribed burns of 50 and 100 ha in the fall season resulted in much higher smoke levels that violated state and federal standards for two to three days (Gertler et al. 2010) (pp. 66-67). The model was also used to examine effects from a moderately sized August wildfire (1500 ha acres); it predicted that smoke from such a fire would completely fill the basin with smoke and exceed air quality standards for four to five days (Cliff and Cahill 1999). These analyses supported the finding that severe wildfires in the Lake Tahoe Basin have greater potential than low-intensity prescribed burns to contribute to violations of air quality standards, obscure visibility across the lake, promote algal blooms, and reduce lake clarity (Gertler et al. 2010). Researchers have concluded that although some reduction in visibility is needed to accommodate increases in prescribed burning, “this would be offset by improved air quality from decreasing the fuel accumulation and resulting impacts of potential major wildfires that may occur” (Gertler et al. 2010: 72).

Research Needs

There is a need to integrate research and management planning to evaluate tradeoffs between wildfires and treatments that include prescribed burning. Sound management of forests, fuels, and air quality will require the scientific community to fill a number of research needs. An exhaustive list of those needs is included in a chapter written by Bytnerowicz et al. (2009). Among those, the most pertinent to this review chapter are: (1) better characterization of the spatial distribution of fuels as well as their physical and chemical properties; (2) improved weather forecasting of changing climate/atmospheric circulations at the local-to-regional scales; (3) more accurate empirical and statistical downscaling tools for assessing the impacts of climate change on fire behavior and emissions; (4) improved characterization of emissions of air pollutants and greenhouse gases during fire events; (5) detailed identification and chemical characterization of VOCs to develop markers (gaseous and aerosol tracers to distinguish smoke from prescribed vs. wildland fires); (6) real-time monitoring of ambient air quality during forest fires; (7) improved regional air quality models that include realistic wildland fire emissions; (8) fire behavior models coupled with meteorological and chemical models for improved understanding of pollution transport; (9) better understanding of ozone and nitrogen deposition effects, as well as interactions among various pollutants, drought, and pests on composition, structure, and function of forests and other ecosystems; and (10) models aimed at better understanding of the effects of air pollution and climate change on forests at the landscape scale.

Management Implications

- Emissions from wildfires and prescribed fires have potential to exceed air quality standards; however, prescribed burning can be managed more easily to mitigate air quality impacts to people. Additionally, because of the relatively low probability of wildfire in any given area, expected emissions from a regime of prescribed burning may often exceed those from wildfire.
- There are sound ecological reasons to promote greater tolerance and application of prescribed fires, and shifting smoke production from uncontrolled wildfires to managed fires can help reduce the overall impacts of burning. Acceptance of fire as a management tool will also require better large-scale monitoring of smoke emissions (including ground-level and remotely sensed) and development of models that are able to predict spatial and temporal distribution of toxic pollutants resulting from fires.
- A variety of tools are being employed and developed to allow better predictions and monitoring about burn activities, including the BlueSky smoke modeling framework, which provides real-time predictions of smoke impacts from prescribed and wildland fires; and the Fuel Characterization and Classification System (FCCS), First Order Fire Effects Model (FOFEM), and Consume for describing fuel loading and predicting emissions.
- These tools will help managers and the larger public to evaluate tradeoffs about how to reduce the debt of accumulated fuels and allow the return of a more natural fire regime where the impacts can be tolerated.

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9.0 Social/Economic/Cultural Components (Preface)

Patricia L. Winter, Jonathan Long, Susan Charnley

Introduction

The approach taken in the previous chapters of this synthesis relies on multiple disciplines from the ecological sciences to frame core aspects of a sustainable, resilient ecosystem. Approaching forest management in the Sierra Nevada in a manner that promotes socioecological resilience and sustains important forest values requires consideration of not only the ecological, but also the social, economic, cultural, and institutional, components of the ecosystem, using a systems approach (Higgins and Duane 2008). The term "socioecological system" has been widely used in scientific literature on resilience. Key ideas underpinning the concept of integrated socioecological systems are 1) there are interactions between biophysical and social factors, 2) there are linkages across spatial, temporal, and organizational scales, 3) the system regulates the flow and use of critical resources that are natural, socioeconomic, *and* cultural, and 4) the system is continuously adapting (Redman et al. 2004). In the following six chapters, we draw from published science that the authors felt was essential to informing an understanding of forest management for socioecological resilience in the Sierra Nevada synthesis area.

The first chapter of this section describes the social context of the synthesis area. Drawing from the extensive analysis of the Sierra Nevada Ecosystem Project Final Report (1996), the chapter explores the social complexities of the area. Recreation and tourism are used as a specific example of a triple bottom line approach to sustainability, which includes ecological, economic, and social considerations (Thomas 2012); these topics were chosen in large part because they are the subject of an established body of literature (see Bricker et al. 2010, Cottrell and Vaske 2006, Cottrell et al. 2007) and link to a global endeavor to understand and monitor sustainable recreation and tourism (see UNEP and UNWTO 2008).

The second chapter focuses on ecosystem services and how managers can use that concept to frame and describe concerns and tradeoffs as they relate to social, economic, and cultural values. This chapter also considers tensions between supply and demand for such services, especially in light of the population growth described in the first chapter.

The third chapter examines the connection between social and ecological health and well-being in the Sierra Nevada. It explores, from a sociocultural perspective, the ecosystem dynamics that are threats to and stressors on Sierra Nevada ecosystems—specifically, climate change, wildland fire, and invasive species. The chapter presents and discusses the complexities of decision making associated with effective management for resilience.

After consideration of broad regional issues in the first three chapters, the synthesis turns to the resilience of communities that lie within the synthesis area. The final three chapters examine how national forest managers can contribute to the socioeconomic resilience of communities in the Sierra Nevada to enhance overall socioecological resilience of the region.

One way to promote community resilience is to plan forest management in a manner that creates economic opportunities in local communities. This can be accomplished in a number of ways, including forest restoration and recreation, and commodity production—the subjects of the fourth and fifth chapters. The fourth chapter discusses strategies for job creation in forest communities through forest restoration and recreation on national forest lands, and the fifth chapter focuses on strategies for commodity production, including biomass, timber, non-timber forest products, and grazing, that support community residents who depend on these resources for their livelihoods.

Community resilience in the Sierra Nevada relies on local institutions and collaborative processes that promote adaptive management and contribute to overall socioecological resilience in the region. Thus, the final chapter in the section focuses on institutions, processes, and models for collaboration in national forest management that use an all-lands approach and incorporate traditional and local ecological knowledge. The importance of collaboration in the larger context of forest management, which is presented in the first chapter, loops back here to effective approaches for collaboration across scales, regions, and institutions throughout the state; these collaborative processes will continue to be an important influence on the success of managing for socioecological resilience in the Sierra Nevada synthesis area.

Though the chapters in this section address many topics, not all topics are addressed. Given the complexity of the social and ecological processes in the region, no one synthesis can cover all of the issues pertinent to assessments and plan revisions in Sierra Nevada national forests. The purpose of this synthesis is to provide a solid foundation of scientific information for these forthcoming assessments and planning efforts, which will necessarily involve more detailed, site-specific analyses and explorations.

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9.1 Broader Context for Social, Economic, and Cultural Components

Patricia L. Winter, Jonathan Long, Frank Lake

Contents

Introduction

This chapter lays out a context for the sociocultural and economic sections of the synthesis that follow. The discussion begins by describing the complexity of the synthesis area, including major social forces that influenced its development, and the present composition of counties and communities and how they have changed since the Sierra Nevada Ecosystem Project (SNEP) report was completed. Because amenity migration (i.e., movement to forested areas for their amenity values) has stimulated much of the growth in the synthesis area, this chapter briefly reviews specific impacts of this migration on social and ecological systems. Population increases within the synthesis area are expected to continue into the future, and influences from outside of the area will continue to have impacts on the Sierra Nevada. Of particular interest are continuing population and demographic changes, as well as additional social, cultural, economic, and political changes. Place meanings are briefly discussed in this chapter in terms of how they influence people's expectations for ecosystem services and their relationships to the synthesis area. Recreation and tourism are discussed because many people in the area develop place meanings through those activities. Management of recreation and tourism is discussed using a triple bottom line approach to sustainability, and finally, the chapter offers insights into managing for resilience.

Complexity Characterizes the Sierra Nevada Synthesis Area

The Past

Thousands of years of indigenous management, the influx of settlers during the gold rush, and, more recently, agricultural development and expansion of residential developments into foothill and forest communities, are some of the major human influences on the Sierra Nevada (Minnich and Padgett 2003). Transitions involving human settlements in the Sierra Nevada, including locations of settlements, physical movement of groups across a landscape (e.g., nomadic groups), and uses of the land (e.g., fishing, gathering, construction of structures), may be instructive in preparing approaches for mitigation and adaptation for the future (see for example Cornell et al. 2010). A study of 62 years of land use in the Lake Tahoe basin revealed that the most significant land conversions included an increase in developed land and decreases in forests, wetlands, and shrublands (Raumann and Cablk 2008). The authors attributed these changes to increased regional population and demand for recreation and tourism opportunities in the basin. The rate and extent of development in the Lake Tahoe basin has contributed to deterioration of environmental quality (Raumann and Cablk 2008). Impacts of development on environmental and social quality will remain a concern for the synthesis area.

A survey of workers and employers in Incline Village and Crystal Bay revealed fewer year-round residents and families with children, a near majority of workers in lower income sectors having commutes of 30 minutes or longer or living with many others to make housing affordable, and a toll on the local economy from lack of workforce housing. (Praxis Consulting Group 2009).

The Synthesis Area

The area of focus for the sociocultural sections of this report coincides with the region previously examined in the Sierra Nevada Ecosystem Project (SNEP). Stewart (1996) described that area as a 20 million acre zone with mixed federal, state, county, regional, local, and private ownership and management areas. The SNEP assessment included 180 community aggregations and covered 160 unique zip codes.

The synthesis area boundaries contain 24 counties in full or in part, though some of these were characterized as having a small minority of their county population within the SNEP region (Stewart 1996, see Table 2.1). Based on the 12 counties that fall primarily within the SNEP region, the 1990 regional population was 563,000 (Stewart 1996, see Table 2.3). At the time of the SNEP assessment, the majority of populations in the remaining counties fell outside of the region, thus county-level data were less instructive.

Population Change, 1990 to 2012

Population estimates for the 12 counties reported by Stewart (1996) show a dramatic 49.8 percent increase in population since the SNEP assessment (Table 1). However, regional variations within the assessment area are important to note. The overall increase in population is attributable primarily to increases in the north central region, whereas a decrease has been documented in the north region. It is also important to consider potential methodological differences in the two assessments.[1]

Table 1. Population changes between 1990 and 2012 in SNEP regions.

County-based Region	Counties	1990 Population	2012 Population	Percent Change
North	Plumas, Sierra	23,300	22,870	-1.8
North Central	Nevada, Placer, El Dorado	383,400	633,222	65.2
South Central	Amador,	126,600	153,510	21.3

[1] Methodological differences will also be of concern in the assessment process. as each community may have its own approach to arriving at or selecting and then interpreting current and projected population numbers. Each reporting and forecasting agency provides a methodological description as well as a statement of assumptions and potential sources of error. This approach may be prudent for upcoming assessments.

	Calaveras, Tuolumne, Mariposa			
East	Alpine, Mono, Inyo	29,700	33,949	14.3
Total		563,000	843,551	49.8

Note: The regions, counties, and 1990 populations are taken from the SNEP assessment (Stewart 1996). The 2012 comparison populations are calculated by the State of California, Department of Finance (2012; www.dof.ca.gov/research/demographic/reports/estimates/e-5/1011-20/view.php).

Between 1990 and 2000, the vast majority of population growth in the Sierra Nevada and Sierra Nevada foothills occurred in the wildland-urban interface and intermix (Hammer et al. 2007).

Amenity Migration

Population growth and settlement in the Sierra Nevada continues to be influenced by amenity migration and settlement of seasonal and year-round residents who are drawn to the area by its unique features (Loeffler and Steinicke 2007). Amenity migration in the Sierra has also been characterized by vertical expansion of human settlement (Loeffler and Steinicke 2006). For example, in the Lake Tahoe region, the upper regions of settlement have moved up in elevation to almost 2400 m, compared to the limit of about 2200 m 30 years ago. These increases in higher elevation year-round and seasonal residency cause new ecosystem impacts, some which may affect animal and plant species and supporting ecosystems in ways not seen in the past.

Urban development is affecting native ecosystems and biodiversity in the area, as demonstrated by Manley et al. (2009). Of particular concern are the lower level intrusions of development not typically considered in assessment of impacts on species. Manley et al. (2009) referred to this as a finer grained approach to analysis. They have shown, for example, that even small degrees of development have dramatically reduced the proportion of habitat that might be suitable for the California spotted owl (*Strix occidentalis*). Habitat fragmentation and impacts on wildlife as a consequence of amenity migration are further discussed in Haight and Gobster (2009).

Amenity migration is associated with shifts in local sociodemographics—for example, toward a younger, more affluent, more educated population (Loeffler and Steinicke 2007, Peterson et al. 2007). Housing values and overall costs of living have been shown to increase dramatically with ex-urban migration, sometimes outpacing the ability of long-standing residents to meet that increase (Loeffler and Steinicke 2007). Housing values have climbed above the housing affordability index in several parts of the synthesis area (Loeffler and Steinicke 2006). Many workers in the area are forced to commute due to expensive housing costs, and the most affected are Hispanic, Asian, and some younger workers (Loeffler and Steinicke 2006). This brings to light one environmental justice issue related to amenity migration—that of disparate impacts on less affluent residents in an area, as well as those who are employed in an area but cannot afford to live there.

One analysis of housing in Incline village and Crystal Bay suggested a median priced townhouse or condominium in 2009 required a median household income of $107,180. The median annual income in those areas in 2008 was $44,346, and in the entertainment, accommodation, and food services sector the median annual income was $30,389. Even households with two full-time wage earners would find it difficult to afford the median price accommodations (Praxis Consulting Group 2009).

Although opportunities to develop additional physical infrastructure are associated with increased economic capacity through amenity migration, demands on local social systems and resources are increased (Kruger et al. 2008b). Development of additional infrastructure was discussed in Duane (1996). The focus of management on private lands tends to shift as a result, from economic generation and family tradition to amenity and investment values (Ferranto et al. 2011), as well as to environmental protection (Jones et al. 2003). These changing private owner motivations and values require shifts in outreach and engagement (Ferranto et al. 2012), in part through the collaborative approaches presented in a later chapter (Collaboration (9.6)).

Protecting scenery, outdoor recreation opportunities, and environmental quality will likely continue to encourage amenity migration (Cordell et al. 2011). These efforts are more effective when partnered with a focus on maintaining community character and social fabric (Kruger et al. 2008b), while expecting that adaptation of a community and its character is likely. Amenity migration has both positive and negative impacts, and positive outcomes are reliant on local adaptive capacity to manage changes in both social and physical attributes of community (Krannich et al. 2006). This theme of capacity to change and adapt as part of resilience reflects the broader concept of resilience put forth in this synthesis. Inability of the system to adapt, whether it is physical or social, is thus viewed as a constraint to resilience. Whether all change is desirable cannot be determined here, and the quality of social fabric will remain of interest to residents well beyond the assessment and plan revision period.

Population Increases

Projections of population in 2050 for this same 12 county area anticipate an additional 48.5 percent increase above 2012 levels, for an estimated total of 1,252,735 (State of California, Department of Finance 2012). It is worth noting that not all counties are expected to have steady-state increases during this period; in fact, some areas are projected to have declines in population.

Counties and communities within the synthesis area will likely have their own projections and estimates to contribute to the plan revision process, similar to the assessment conducted by Struglia et al. (2003) for the southern region of the state. Thus, the projections offered here should not be viewed as definitive, but as demonstrative of what is anticipated by California's central planning and demographic resource. Projections have been the subject of debate and sometimes dispute (Struglia et al. 2003) because of their association with the allocation of resources from federal, state, and regional bodies, and because of the local responsibilities that may result from them. Social and economic assessments would benefit from considering these multiple and sometimes conflicting sources and their implications, where applicable to regional and forest plans. An approach that provides multiple perspectives mirrored after Struglia et al. (2003) may help represent these debates and foster a continuing collaborative and adaptive approach to management of the synthesis area.

Influences From Outside of the Synthesis Area

Increasing populations in metro areas surrounding the Sierra will have both indirect and direct impacts, including, for example, demand for water (indirect, see the boxed example), and recreation and tourism (direct). The Sierra Nevada contains features, species, and areas with heightened social value; these values present management concerns that extend well beyond local communities (see for example Kellert et al. 2000).

The LADWP (2010) outlines the significant value of water coming into the Los Angeles Aqueduct from the eastern and western watersheds of the Sierra Nevada and the regional benefit to southern California water supplies. A period of filling this demand at a cost to the Owens River and Mono Lake* ecosystems demonstrates the need to consider broader-scale impacts of managing water as a valued ecosystem service in the state (see for example Fitzhugh and Richter 2004).

*See Wiens et al. (1993) for a detailed ecological impact assessment from Mono Lake.

Human activities some distance away also impact the ecological quality and viability of the Sierra Nevada. For example, pesticide drift from the Central Valley in California is believed to be responsible for high environmental concentrations of pesticides in parts of the synthesis area. These same areas have been the zones with the greatest declines in amphibians (Fellers et al. 2007). These issues highlight the varying levels of scale that must be considered in managing for socioecological resilience (Engle 2011), and how larger scales of impact and interaction must be taken into account.

Influences at even larger scales have some relevance to natural resource management and decision making in the Sierra Nevada. For example, community well-being must be considered in the context of global economic trends, and the effects of local ecological systems and resource management must be distilled from broader social forces (see Davidson 2010).

The sociopolitical environment in California—which includes high levels of regional diversity, racial and ethnic diversity, political distrust, and a trend toward civic disengagement—portends greater rather than less difficulty in reaching public consensus on policy issues (Baldassare 2000). These trends are not constrained to California, and in some cases, they reflect a detachment, disconnection, and mistrust of anything 'governmental' by a segment of the public best characterized as angry or 'fed up' (Susskind and Field 1996). The kinds of collaborative approaches addressed in a later chapter (Collaboration (9.6)) should take into account a potential absence of trust or outright distrust in interaction with the public and stakeholder groups.

Based on larger socioeconomic trends across the United States, Cordell et al. (2004) laid out key implications for natural resources applicable to the Sierra Nevada, including a smaller and more fragmented rural land base (confirmed by patterns of land use reported by Ferranto et al. 2011), disproportionate pressures on public lands for recreation and raw materials, increased conflicts and competition for access, and less connection between people and the land.

Changes in ethnic composition within the regions surrounding the Sierra Nevada are worthy of note. In the Pacific Coast RPA (Resources Planning Act, see http://www.fs.fed.us/pl/rpa/summary.pdf) region (which includes California) between 1990 and 2008, there was an 80.4 percent increase in residents self-identifying as Latino or Hispanic, a 59.0 percent increase in those identifying as Asian or Pacific Islander, and an 8.9 percent increase in those identifying as African American (Cordell 2012). Science suggests that these changes can be met with adjustments in how the Forest Service works with and offers opportunities to the public. For example, some groups have stronger ties with the Forest Service and other managing agencies, whereas others may have little if any established relationships, or even a negative history of relationships. Services offered through existing communication and information approaches and more direct opportunities, such as those represented in recreation and tourism, might be a poor fit to these populations that are increasing in the region and surrounding areas. Planning for the Sierra Nevada may consider these cultural shifts and how they may be met through adjustments in local and regional services (see Roberts et al. 2009 for a discussion of some of these service adjustments). For example, communication may need to be through ethnic media or key contacts within communities (Winter et al. 2008), rather than through mainstream English-speaking media. Science suggests that messaging that is culturally sensitive and addresses issues that matter to the particular community of interest will be more effective (Roberts et al. 2009). Recreation and tourism opportunities may include more developed sites that accommodate extended family gatherings and support activities of interest to diverse groups (Roberts et al. 2009). Sensitivity to cultural differences in relationships to government, the land, and land management will aid effective management in this diverse region (see Cheng and Daniels 2003). Increased cultural diversity in California will continue to be reflected through immigration of Latinos and Asians into Sierra Nevada communities, thus increasing the importance of attending to cultural influences and values of long-standing and newly immigrated residents (Sturtevant and Donoghue 2008).

These dimensions of diversity add to the already diverse demographic, economic, and ethnic profile of Sierra Nevada communities. Both new and existing populations will challenge modes of outreach, engagement, and approaches to management. Particular attention will need to be paid attention to groups who may be underserved or underrepresented in opportunities to have their opinions heard, needs or interests represented in decisions about how places will be managed, and opportunities to use their public lands.

Place Meanings Link to a Diverse and Growing Population

Because locations and places have substantial variation in meanings and interests, discussions of place are characterized by significant complexity and diversity (Patterson and Williams 2005). Relationships to natural spaces (such as traditions of viewing oneself and nature as part of a whole, see Turner and Berkes 2006, Wiggins et al. 2012) may be embedded in culture, religion, personal experience (such as through recreation and tourism, see Wynveen et al. 2008); associated value sets (such as orientations toward the environment and nature, see de Groot and Steg 2010); familial experiences; narratives (through literature or shared documentation, such as photographs and images); and imaginations. Two parties or groups may express a particular value or attitude toward a place or location, and these may distinctly differ (e.g., sacred area versus lovely place to build a structure see McAvoy 2002); and will vary in response based on scale of place under consideration (Cheng and Daniels 2003).

These divergent views may also vary in strength of impression and importance, as well as how individuals will respond to changes in forest management. For example, individuals with direct vested interests in a place may have attitudes that are stronger than those whose interests may be equally satisfied by a comparable place (Wiggins et al. 2012).

Those whose connections or impressions of a place are intertwined with their sense of self are likely to hold much stronger attachments and may consider discussions of place as equal to discussions of self-determination and personal identity (Clayton and Myers 2009, Huntsinger et al. 2010, Knez 2005). Ranges of relationships, varying from contained or individualistic parts of association to those described as strong relationality, or embedded as the foundation of identity and existence (Wiggins et al. 2012). Management actions may be of significant concern when viewed as a threat to one’s self, or a personal attack (Cheng et al. 2003). Likewise, group identities may be attached to a particular place, where meanings and management preferences for areas are intertwined with social identity (Cheng et al. 2003, Huntsinger et al. 2010, Opotow and Brook 2003, Schneider and Winter 1998). Debates over place and attached meanings may then also be interpreted as discrimination against a particular group; for example, debates over impacts of grazing may be viewed as embedded in discrimination against ranchers and ranching as a way of life (Huntsinger et al. 2010), a fear of loss of community (Miller and Sinclair 2012), and a request of a majority to have a minority (ranchers) bear the burden of protection (Opotow and Brook 2003).

An approach that incorporates diverse place meanings may require collaborative stewardship of areas with symbolic or cultural significance for American Indians (McAvoy 2002, McAvoy et al. 2003). Tribal traditions and beliefs may require direct stewardship of lands to maintain place and culture. Taking this direct stewardship approach would allow for sometimes conflicting views and meanings of protected areas, including forest lands containing valued natural and cultural resources.

Place-specific attachment has been shown to differ from conceptual attachment, held by more technical ‘experts’ with knowledge of natural areas (Ryan 2005). These differing forms of attachment are associated with valuing different aspects or features of a place, and thus may be associated with differing preferences for management of that place and differences in how individuals and groups respond to change (Ryan 2005, Wagner and Gobster 2007, Yung et al. 2003).

As a result, generic discussions of landscape scale present a challenge, and it is clear that discussions of management are enriched when they also focus on specific landscapes in order to be able to consider social and cultural dimensions (see Asah et al. 2012, Cheng and Daniels 2003, Diamant et al. 2003, Williams 2006), including place meanings and personal and social identities (Cheng et al. 2003, Kruger et al. 2008a, Yung et al. 2003). Understanding meanings of place (e.g., places designated as sacred or otherwise valuable) is essential to discussions of socioecological resilience (Berkes and Turner 2006, Clayton and Myers 2009, McAvoy 2002). This is in contrast to the landscape-scale approach that is needed for addressing ecosystem threats, such as climate change. It is likely a nested approach to planning and management that will be most successful into the future (wherein the overall landscape is addressed and then places are nested within that—thus the whole is in fact likely to be greater than the sum of its parts).

Place-based approaches to planning represent one means of incorporating these various place meanings (Hibbard and Madsen 2003), and they provide the path to consider 'special places,' along with their divergent meanings (Schroeder 2002). However, it is important to note that proximity is not the sole determinant of meaning. Individuals and groups some distance away must also be considered. This can complicate deliberations over management direction when particular groups are not represented in planning approaches that gather input through more conventional mechanisms, such as through inviting public comments or holding public input meetings (Cheng et al. 2003). (For an extended examination of sense of place and implications for management, see Farnum et al. 2005 and Kruger et al. 2008b.)

The population of the Sierra Nevada represents a small portion of the statewide population, and it is thus a numerical minority centered in a highly valued socioecological and historical context. Statewide decisions or regional decisions to address majority interests may adversely impact human and non-human populations and ecosystems in the Sierra Nevada, sometimes in ways that put long-term sustainability at risk (Mittelbach and Wambem 2003). Competition for scarce ecosystem services and opportunities will remain a challenge for management of the forests in the synthesis area.

For many, recreation and tourism in the Sierra Nevada is a way to learn about the area's many features and to develop a connection to places within it. These connections may be instrumental in efforts to reduce demand on ecosystem services delivered far downstream, such as water drawn from the Sierra Nevada to be used in southern California, or the need to manage transportation in ways that reduces the transport of pollutants into the synthesis area.

Recreation and Tourism

Across the United States, nature-based outdoor recreation increased in total number of participants as well as in number of activity days between 2000 and 2009 (Cordell 2012). During that same period, site-based activities, including camping in developed sites and family gatherings, increased. An increased interest in nature was reflected in increased viewing and photographing of nature-subjects, especially wildflowers, trees, natural scenery, and wildlife and birds. Backcountry activities declined somewhat (e.g., horseback riding on trails and day hiking). While off-highway vehicle (OHV) use levels held steady, snowmobiling declined (see

Cordell 2012 for additional national trends). National participation levels in different types of activities varied among groups depending on gender, ethnicity and race, annual family income, place of residence, and residence status.

According to Roberts et al. (2009), cultural diversity will continue to increase in California, owing primarily to continuing increases within Latino and Asian populations, and this trend will have implications for outdoor recreation planning and management. A number of studies have revealed cultural variations among Latino and Asian populations, including recreation patterns and preferences for development, under-representation in some forested areas, and communication and information needs on and off site (Crano et al. 2008, Roberts et al. 2009, Winter et al. 2008).

Constraints to participation are sometimes common across groups (e.g., lack of time for recreation), whereas others are more likely to be reported among disadvantaged populations. Of particular importance in this synthesis are those barriers that can be changed by natural resource management agencies. Awareness of these barriers may lead to changes in communication approaches on and off site, site design and the types of opportunities presented, management interactions with visitors, signage, and increases in the presence and number of agency personnel that are from under-represented groups (for reviews, see Chavez 2012, Tierney et al. 1998, Winter 2007, Winter et al. 2004).

The tourist industry represents a major component of the Sierra Nevada economy, accounting for over $3.2 billion in 2000 (Mittelbach and Wambem 2003). Aside from national parks, the

Mammoth Lakes and Lake Tahoe basin are the most tourism-intensive areas of the Sierra Nevada (Loeffler and Steinicke 2006), accounting for 38 percent of jobs with direct ties to tourism and 74 percent of all jobs with indirect ties in the two areas, respectively.

California has a rich cultural and natural history, which can provide recreational and tourism opportunities. The heritage of California Indians can offer recreationists and tourists opportunities to learn about the cultures that shaped California's ecosystems (Evans 1986).

Roberts et al. (2009) also suggested that California's senior population, which is already the largest in California's history, will continue to grow and settle in foothill and rural counties. Experts also anticipate increases in tourism and second-home development, related in part to trends in the senior population.

Increases in visitation from culturally diverse populations recreating in the Sierra Nevada Forests are likely, as reported at some sites on the Giant Sequoia National Monument (Chavez and Olson 2010).

Among the shifts in land use associated with the aforementioned amenity migration are demographic differences in forms of recreation engagement that can cause conflicts between older and newer residents in an area. New recreation approaches may conflict with historical resource use and dependence in the region, and management would benefit from understanding and considering these conflicts (Mekbeb et al. 2009). Another recreation-related shift associated with amenity migration is greater development in wildlands, which could impact public access to outdoor recreation areas (Peterson et al. 2007).

Increases in numbers of users and certain types of use may also affect the ecosystem and the health of recreationists. Padgett et al. (2008) reported that OHV use contributed to erosion on unpaved roads and trails and increased transport of fine dust particles, and they argued that elevated dust concentrations may reach unhealthful limits for riders during periods of heavy trail and road usage. Managers of OHV use in California cited soil erosion and compaction as a primary ecological concern, as well (Chavez and Knap 2006). Damage to vegetation in areas with low rainfall was noted as a possible impact from fugitive dust (Padgett et al. 2008), and this is of particular concern in the Sierra Nevada, which has little precipitation for most of the year (Minnich and Padgett 2003).

The impacts of OHV use on wildlife have been the focus of some studies, though there is little published literature on the topic. Barton and Holmes (2007) identified impacts on breeding songbirds, and they reported greater nest desertion and abandonment in shrub nests < 100m from OHV trails than in nests > 100 m from trails. They suggested that OHV management would benefit from an understanding of the abundance and needs of nesting birds. Potential impacts on the marten are discussed in a separate chapter (see chapter on Forest Carnivores (7.1)). Potential impacts on lizard populations were reported by Tull and Brussard (2007).

Studies have also examined the economic/welfare impacts of restricting OHV access on public lands. Moving from open to limited access appears to have a minimal impact on consumer welfare, whereas moving from open to closed access is estimated to have large impacts (Jakus et al. 2010). Although open acreage is reportedly most highly valued by OHV users, restricting travel to existing roads and trails has minimal economic impact and may meet resource protection mandates (Jakus et al. 2010). However, in a statewide survey of managers with OHV management responsibilities, the vast majority of respondents had observed or received reports of use violations on closed roads or trails (Chavez and Knap 2006).

Studies have also examined the social and ecological impacts of horseback riding (Newsome et al. 2008). Off-trail and on-trail riding have both been associated with ecological damage. Varied biophysical impacts have been reported related to soil, surface water, and vegetation, and they have sometimes involved changes to trails (e.g., widening) and introduction of foreign materials (e.g., weeds) (Newsome et al. 2008, Price 1985). Many impacts vary by slope, elevation, vegetation, rainfall, and proximity to streams, and the extent of impacts therefore depends on specific site characteristics. Given the value of horseback riding as a recreation opportunity, there is a need for improved experimental research that more carefully assesses impacts, as well as evaluations of the efficacy and applicability of different management approaches (Newsome et al. 2008).

Additional studies examining impacts of recreational activities have focused on direct impacts from downhill ski use on vegetation during the winter (Price 1985); recreational use impacts on vegetation, slopes, and trails in summer (Price 1985); and overall reviews of impacts (Cole 2004). Mode of development also affects impact; for example, a comparison of ski slope development showed greater ecological impacts from cleared runs compared with those that were graded (Burt and Rice 2009).

Colby and Smith-Incer (2005) examined visitor values and economic impacts of riparian habitat preservation on the Kern River Preserve. They surveyed visitors to the preserve to explore willingness to pay for preservation. They reported an annual average of $467,000 to $616,000 per year in willingness to pay for preservation based on average payments and visitation levels. Furthermore, they found that visitor expenditures in the Kern Valley represent $1.3 million in local business activity. Respondents indicated that failure to maintain and preserve the ecosystem would likely result in decisions to not visit the area at all, or to significantly reduce the number and length of their visits. These values highlight the preserve's importance to the local economy, and the fact that local economic benefit is dependent on continued protection and preservation. These findings are also helpful in demonstrating benefits of protection, and they offer an alternative perspective from those focused on the value of addressing increased demand for surface flow and ground water (though these demand issues are not directly applicable to the Kern River Preserve at present).

The Triple Bottom Line and Socioecological Resilience

Approaches to the triple bottom line (ecological, social, and economic components of sustainability) have been applied to discussions of recreation and tourism management. Deliberations surrounding whether or not a recreation opportunity will be provided should, according to this body of literature, consider each component of the triple bottom line (Bricker et al. 2010, Cottrell and Vaske 2006, Cottrell et al. 2007). For example, social dimensions of sustainability include benefits to recreationists from being able to engage in desired activities, the ability of the area to sustain a particular level of use (carrying capacity), and the fit between various uses proposed for an area (where social conflicts would come into consideration). Cultural dimensions include things like historical traditions and uses, preservation of the culture of a community that might be affected by recreation and tourism, and protection of cultural resources. Economic dimensions include benefits to local communities from outside visitation of surrounding areas, as well as costs of managing and regulating impacts on the community itself (for example, through increased traffic on local roads). It also includes the economic capacity of the Forest Service and other managing agencies to maintain personnel and physical settings where recreation and tourism occur. Ecological sustainability incorporates several considerations, including the impacts on an ecosystem from various uses and ways to manage those impacts that consider the other components of sustainability, as well as the ability to take an approach that examines feedback between social and ecological systems to adaptively manage resources and opportunities over time.[2]

> Recent adoption of the Global Sustainable Tourism Council's Criteria in Wyoming Parks is a specific application of sustainability metrics. Multiple stakeholders worked in tandem to establish and commit to criteria for sustainability. This type of approach might be considered for the synthesis area.

Managing for Resilience

Recreation and tourism provide valuable examples of steps for increasing socioecological resilience (see chapter 9.0). Effects of development are in the hands of responsible agencies and surrounding human populations (Heckmann et al. 2008). A balance of values that

[2] The Global Sustainable Tourism Council (http://www.gstcouncil.org/) has supported development of criteria for sustainable tourism and facilitates certification efforts. Though the work is concentrated on recreation and tourism, the context is much larger.

incorporates the natural ecosystem as a space that is communal, meets basic human needs, and is central to development may aid environmental protection efforts (Kaplan and Austin 2004, Kaplan and Kaplan 2003). Organizational boundaries must be bridged to address issues of shared concern (Barbour and Kueppers 2012, Dietz et al. 2003), and institutional supports must be provided. Decisions have to move beyond steady states and across scales, and allow for adaptive learning and flexibility. Folke (2006) referred to this as adaptive governance.

Multiple approaches have been discussed for improving socioecological resilience. Schluter et al. (2011) offered a dynamic approach that represents feedbacks between social and ecological systems; they proposed that social changes impact ecological systems, then ecological systems further impact social systems (Berkes and Turner 2006). This 'codynamic approach' posits a coupling of both systems, and considers the resilience or agency of each system to adapt (Engle 2011).

Adaptive cycle functioning requires system-level awareness, and an ability to adapt factors that feed back into the next cycle (Folke 2006). The challenge of managing for resilience is compounded by increasing populations in areas already exhibiting high degrees of stress, or impending stresses, such as continuing population increases, increased demands for ecosystem services, and larger global threats such as climate change (further addressed in Sociocultural Perspectives on Threats, Risks, and Health).

Individual sections in this synthesis address uncertainty within and inter-relation among system components. These underscore the importance of clarity and full-system description in being able to identify and employ the broadest range of management options, and manage for ecosystem service scarcity and disruption (Patterson and Coelho 2009). As demonstrated in the next chapter (Ecosystem Services), addressing gaps and updating information about the importance and value of off-site uses of ecosystem services will be increasingly important in light of projections of future population growth and the continuing need to manage for resilience.

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9.2 Ecosystem Services

Trista Patterson

Contents

Executive Summary

Specific research quantifying ecosystem services has been limited in the synthesis area. Yet this approach may be increasingly important for planning, management, and partnerships in the future. This chapter reviews and synthesizes the concept of ecosystem services, focusing on information to assist planners and managers in framing and describing concerns and tradeoffs in social, ecological, and economic values. This chapter does not repeat information about specific ecosystem services that is found throughout the full synthesis document. This chapter provides examples of how the term "ecosystem services" may be used and understood in different ways by different people.

The Forest Service has a long history of managing and providing what are now called ecosystem services, beginning long before the term itself came into use. Many individuals in the agency are reporting applications of the concept, advances in quantification of service values, and some successes in engaging more diverse stakeholders and promoting interchange between management and research.

Although situational in nature, these examples illustrate breadth in the potential management application of the concept, and they are highlighted in callout boxes throughout this chapter.

Due to the cost of assessment and valuation efforts, it is likely that the team performing bioregional assessments will assess the condition and trend of most ecosystem services in general terms by selecting only a few to quantify and model. The information provided here may help inform which ecosystem services, data sets, and approaches should be emphasized during the assessment phase. Themes explored in this chapter are not prescriptive, but are intended to help identify information and expertise to help inform assessment of ecosystem services. The Frameworks for Adaptive Management section below reviews how Forest Service assessments to date have characterized relationships between elements of the ecosystem service system. Although certain relationships are highly quantified and familiar to managers, other relationships are of emerging importance and are less likely to have established quantified relationships. The Frameworks section underscores the importance of investments in data and efforts to understand relationships that are less well known—specifically, documenting factors affecting both supply *and demand* for ecosystem services. This information would be particularly important in describing the ability of the study area to provide future ecosystem services and to anticipate deficits or shortfalls. To address such shortfalls, this chapter highlights emerging options and broader arrays of management interventions and opportunities in agency operations planning. It also acknowledges that some challenges may require coordinated effort over time (i.e., new data sets, or new strategies to support both supply and demand challenges of ecosystem service issues).

Extensive detail in valuation methodology is beyond the scope of this document. Appropriate experts can be consulted when market or non-market services need to be valued either in dollar or other social terms. Rather, this chapter is intended to enable natural and social scientists from other disciplines to participate in meaningful discussion and deliberation over what kind of information regarding values might best inform management goals.

Overview and Chapter Organization

The first section of this chapter provides a summary of definitions, concepts, and uses of the ecosystem services concept by the US Forest Service. The second section provides a general framework for defining the scope of an ecosystem service assessment and characterizing the relationships among its various components. A particular emphasis is placed on relationships that reveal new or emerging management options for addressing ecosystem service deficits. The final section describes approaches in more detail, as well as methods for valuing ecosystem services.

Information on ecosystem services related to specific land covers, habitats, or species is covered in other chapters of this synthesis. The social science chapters characterize many of the dynamics of the local community, economy, and visitors, which all rely upon these ecosystem services to some extent for their well-being and resilience. The majority of data and information regarding ecosystem services for the study area is found in non-peer-reviewed sources, working papers, and mapping efforts. Although many peer-reviewed single-service studies exist, they are often approached from a diversity of perspectives, units of quantification, and spatial scales, rendering efforts to combine data and information somewhat unwieldy (Patterson and Coelho 2009). To date, the most comprehensive resource covering ecosystem services (though they are not termed as such) is the SNEP (1996) report to Congress, however, the various chapters of the SNEP report focus almost exclusively on elements of provision. As the Frameworks section of this chapter points out, a more systematic ecosystem service assessment will complement this supply-side information with quantified information about use of ecosystem service benefits on and off site. Information about ecosystem service supply and demand is needed to characterize ecosystem service scarcity and to articulate present and future value.

Importance of Ecosystem Services within the Synthesis Region

The Millennium Ecosystem Assessment, one of the most widely cited global assessments of ecosystem services, defines ecosystem services as the benefits people obtain from ecosystems. These benefits include provisioning, regulating, and cultural services that directly affect people, as well as the supporting services needed to maintain other services (MEA 2005). Ecosystem services provided by the Sierra Nevada contribute to the quality of life for millions of people, many living at great distance from the Sierra Nevada. A dramatic example is San Francisco's drinking water, which originates in Yosemite. More broadly, the Sierra Nevada snowpack provides nearly 65 percent of California's water supply (SNEP, Volume 1). The area produces over $2.2 billion worth of commodities and services annually in water resources, agricultural and timber products, ranching, and mining, and more than 50 million tourism and recreation visitor days annually (ibid).

Despite the many benefits they provide, many Sierra Nevada ecosystems, species, and their respective ecological processes are being negatively impacted by development trends, rising population, habitat fragmentation, and intensification of human activity. By 2040, almost 20 percent of Sierra Nevada private forests and rangelands could be affected by projected development (SNEP, Volume 1). These effects are of concern from an ecosystem services perspective, as they have resulted in diminished, interrupted, suspended, or redirected flows of ecosystem services. Primary concerns include forest disturbance events and trends, and phenomena such as climate change (Deal et al. 2010, McKenzie et al. 2004), erosion (Neary et al. 2009), invasives (Eiswerth et al. 2005, Zavaleta 2000), housing development (Stein et al. 2005), losses in species diversity and redundancy (Tilman 1997), and successional phases following timber extraction (Beier et al. 2009). Increasingly, studies are attempting to determine the economic impacts and tradeoffs of these losses before they occur (Sukhdev et al. 2010, Barbier 2007, Murdoch et al. 2007). As discussed later in this chapter, incentives to restore lost services, or to prevent losses before they occur, are becoming increasingly common in market-based approaches to private forest conservation.

Characterizing Ecosystem Services

Ecosystem services are generally described according to how they contribute directly and indirectly to human benefit (MEA 2005). Specifically, an introductory schema organizes goods and services according to whether they are provisioned (e.g., timber, drinking water, fuels, mushrooms, berries, venison, fish); regulate (e.g., carbon sequestration, erosion control, riparian forest cleaning, filtering and cooling streamside water); provide cultural services (such as recreation, spiritual enrichment, educational opportunities); or support the other services (biological diversity, nutrient cycling, etc.) (Figure 1).

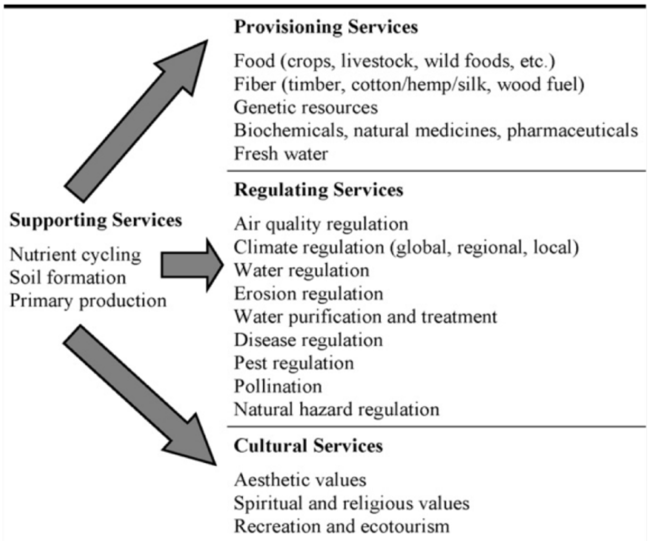

Figure 1: Broad categories of ecosystem services (adapted from MEA 2005, used with permission from Patterson and Coelho 2009)

Which Definition is Best, and for Which Purpose?

In part, an articulate depiction and accurate assessment of ecosystem services of the Sierra Nevada hinges on how the term 'ecosystem services' is used and approached. The ecosystem services literature is derived from the fields of ecology and economics (Ehrlich et al. 1977, Ehrlich and Ehrlich 1981, Krutilla 1967, SCEP 1970, Westman 1977), and has resulted in a particularly wide range of definitions (Patterson and Coelho 2009, Kline and Mazzotta 2012). In general, one can imagine a spectrum of increasing need for precise typology and definition to guide selection of terms and literature (Figure 2, adapted from Kline and Mazzotta 2012).

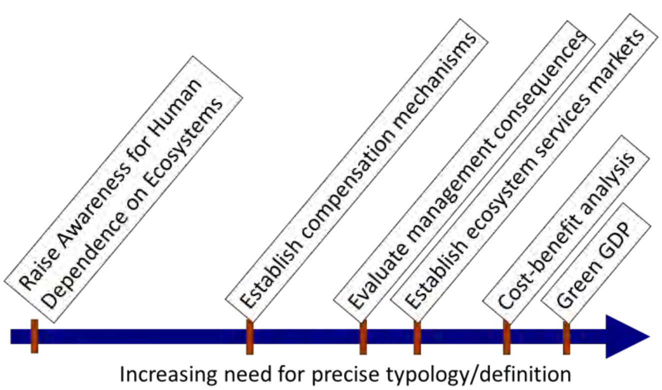

Figure 2: Specificity needs based on intended uses of the ecosystem service concept (adapted from Kline and Mazzotta 2012)

Two oft-cited works describe ecosystem services as the conditions and processes through which natural ecosystems, and the species that make them up, sustain and fulfill human life, thereby supporting quality of life on earth (Daily 1997, Costanza 1997). Forest Service projects designed to raise awareness for forested ecosystems and public investment tend to use similarly general language (MEA 2005, Daily 1997, Collins and Larry 2008). More specific definitions may be used to estimate replacement cost of lost ecosystem services, or to incorporate these benefits into conceptual framing of important social issues (Costanza et al. 1997, US EPA 2006). The narrowest definitions are needed to provide the criteria for specific accounting, tracking, and decision making (Boyd and Banzhaf 2007; see also reviews in de Groot et al. 2002, Costanza 2008, Fisher and Turner 2008, Kline and Mazzotta 2012).

Forest Service Use of the Ecosystem Service Concept

Ecosystem services, and their values and meaning to society, are important to consider as the Forest Service attempts to grow as a 'learning organization' (Apple 2000). To this end, the ecosystem service concept broadens the scope and the spatial and temporal scales of what scientists, managers, and public-private partnerships consider in forest management. An ecosystem services approach therefore relies on a mix of traditional and new performance measures that are important to society, based on the management targets from the activity site itself, and in conjunction with other measurable outcomes and influences experienced in the wider forest area.

The Forest Service has managed for ecosystem services since its establishment as an agency (McCleery and DeMaster 1999), but it currently uses the concept as a means to inform management decisions, increase potential funding of a broader suite of ecosystem services, and raise the visibility of the value of forests to the American public (Patterson and Coelho 2009, Collins and Larry 2008, Kline 2006, Smith et al. 2011). The sidebar below details uses of the ecosystem service concept within Forest Service management efforts, as adapted from a summary effort from the Deschutes National Forest (Smith et al. 2011).

Sidebar: Uses of the Ecosystem Service Concept within Forest Service Management[1]

1. ***Describing the value of forests***
   As management costs rise and timber revenues decline, visibility and awareness for the products of forest management, the value of generating shared value and values from public and private goods from forest systems help to improve wider understanding of what funded, sustainably managed forests provide.
2. ***Characterizing and evaluating trade-offs between different values, functions, goods, and services***
   Forest management activities (e.g., for timber, biomass, recreation, riparian enhancement) affect ecosystem services in different ways, and new tools are needed to describe and evaluate the benefits that result (a more complete account of the range of values, a better analysis of the

[1] Adapted from Smith et al. 2011

relationships between multiple values, or benefits of management activities that are relevant to particular stakeholders or potential partners).

3. ***Identifying ecosystem service decline and providing a wide range of potential mitigating or restorative options***
   Informed changes to forest policy, actions, and techniques can redress some declines. Meanwhile, planning, education, and public-private and federal-state-municipal partnerships can impact ecosystem service use- and conservation, reducing pressure on the resource and raising awareness for its value.
4. ***Providing a basis for consultation and collaboration with stakeholders by defining common objectives for forest stewardship***
   By clearly describing benefits, the ecosystem services approach offers a common language for forest owners and interest groups to describe and articulate management objectives.
5. ***Supporting the emergence of markets, products, and payments for ecosystem services***
   Many forest benefits, such as freshwater production, protection of topsoil, carbon sequestration, and preservation of biological and genetic diversity, as well as traditional commodities and services, such as timber, grazing, recreation and aesthetic beauty, cultural and educational benefits, can be supported through various mechanisms, which transfer payments to the lands producing those services.

Incorporating information about ecosystem service values into management planning is important because ecosystem service harvests, uses, and exchange often do not take place in markets. They might be collected by individuals, or shared amongst family and friends (e.g., game meat, subsistence salmon, mushroom picking, etc.). They may accrue to everyone publically as part of ecosystem function, and may not be particularly visible (e.g., carbon sequestration, water purification, etc.). Benefits may flow far from the landscapes where they are produced. Tracking indicators of ecosystem service supply and demand, and their status over time, is important because most common economic indicators (e.g., gross domestic product (GDP)) do not account for quantity or quality of natural capital stocks, or the value of many ecosystem services (Boyd and Banzhof 2006). The indicators used in many civic decisions often do not weigh the consequences of ecosystem service losses (Patterson and Coelho 2009, Boyd and Banzhoff 2006) until after those losses have already occurred.

Worldwide, national, and international policies are increasingly reporting on ecosystem services from public lands (EUSTAFOR and Patterson 2011). The concept is consistent with USDA's emphasis on collaborative approaches and outreach to increasingly diverse stakeholders. Consistent with the USDA 2012 planning rule, Forest Service integrated resource management must use the best available scientific information to guide management of NFS lands "so that they have the capacity to provide people and communities with ecosystem services and multiple uses that provide a range of social, economic, and ecological benefits for the present and into the future" (Miller et al., in press). Changes to NEPA requirements are still anticipated, and the extent to which approaches across different forests will be coordinated is not known (Miller et al., in review). However, some attempts at guidance documents have been made to guide management of NFS lands, and it is anticipated that future ecosystem service assessments will to be needed to support planners in their work to support social and economic sustainability (Miller et al., in press).

Frameworks for Adaptive Management

In 2005, the Millennium Ecosystem Assessment reported declines in more than two-thirds of the world's ecosystem service systems. Stemming ecosystem service declines will require more than simply quantifying provision of ecosystem services or willingness to pay for them, because even the healthiest ecosystem has upper limits to the rates at which it can provide ecosystem services in perpetuity (Patterson and Coelho 2008, Patterson and Coelho 2009). When consumption exceeds production of ecosystem services, management issues can arise quickly. Harvest/transport/waste systems related to ecosystem service consumption can negatively impact ecosystem service production systems (Patterson and Coelho 2009, Beier et al. 2009). Exceedances of certain thresholds can increase the probability and severity of ecological impairment, and can reduce system resilience to similar shocks over time (Folke et al. 2004). Reduced ecosystem service flows that result may limit management options for present and future generations. Thus, an important component of resilience in socioecological systems is the ability of management to keep a system within certain system boundaries (Chapin et al. 2009, Toman 1994, Toman et al. 1998).

One of the most important steps at the outset of any ecosystem service assessment is the declaration of an explicit framework, because it is within this construct that system boundaries can be defined, current status can be benchmarked, relations between system components can be examined for possible management or intervention options, and with quantification, progress can be tracked against overarching goals (Patterson and Coelho 2008). Just as ecosystem services may have various definitions, conceptual frameworks also vary widely (Boyd and Banzhaf 2006, Brown et al. 2007, deGroot et al. 2002, Fisher et al. 2009, Kline and Mazzotta 2012, Smith et al. 2011, Patterson and Coelho 2009). Guidance on framework selection specific to forest management is only beginning to emerge (Miller et al., in press).

An often overlooked step in declaring an assessment framework is evaluation of whether the system description contains all the necessary components and expresses the necessary relationships to address gaps between the present and desired state of the system (Patterson and Coelho 2008). Figure 3 reflects elements of the most common ecosystem service approaches, as summarized by Miller et al. (in press), in a review of existing frameworks for Forest Service management applications. The arrows between system components in Figure 3 reflect the relationships most frequently emphasized in ecosystem service study: between management decisions and forest resources/supply of ecosystem services, between stressors and resources/conditions, between resources/conditions and the supply of services, between supply of services and human use of them, and between the drivers for services and use of services. Each of these relationships lead to net benefits anticipated from the system.

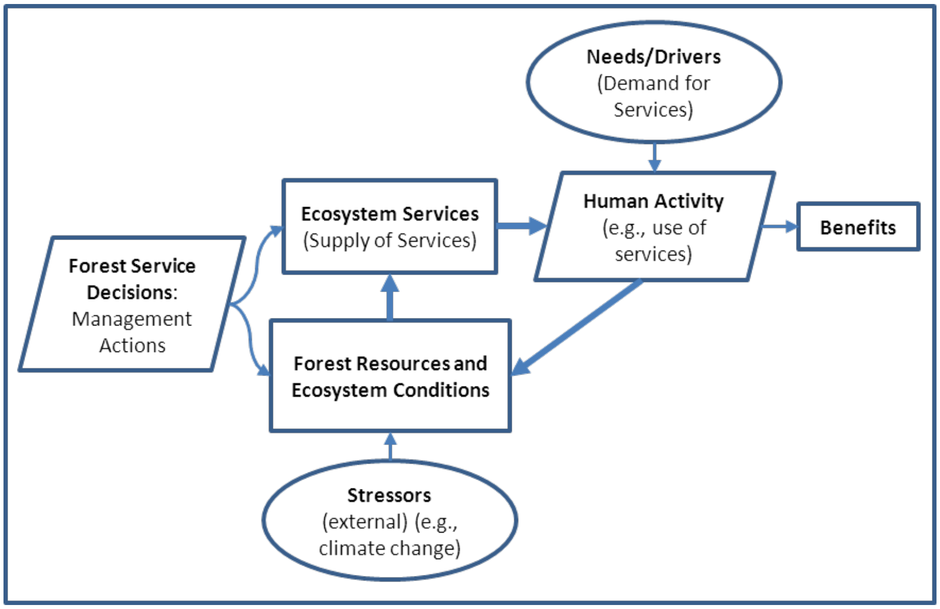

Figure 3: Ecosystem services as a framework for forest management (Miller et al., in press, as modified from Kline and Mazzotta 2012 and Patterson and Coelho 2009)

Long-standing resource management challenges can be deeply embedded within systems and structures that may push back on efforts to resolve them or reinforce their persistence (Folke et al. 2004). A systems approach to system intervention can assist in identifying intervention that will have more enduring, or systemic, impact and can thereby make best use available of scarce funds (Patterson and Coelho 2008). A systems assessment begins with articulation of important system components and agreement upon relationships and feedbacks among component parts. This step is particularly important, as system definition is often assumed as a tacit, rather than explicit, element of ecosystem service study (ibid), and lack of consensus in this regard leaves ecosystem service assessment subject to a few typical pitfalls—namely, failure to allocate sufficient resources to addressing data, informational, and relational gaps (Patterson and Coelho 2009).

As data and understanding of interdependency between people and ecosystem service systems become more commonplace, these connections and relationships will become easier to establish and communicate. For now, this chapter will point out three general relationships for which ecosystem service information has tended to be less available, but which are critical to establishing the feedback loops that can govern human use of ecosystem services within the bounds of ecological limits. Establishing these relationships is the first step to broadening management actions to include them. Each of these connections represents a whole range of possible management options to address ecosystem service system resilience. The more diverse the suite of management options, the broader the options to ensure effective and sustainable management of ecosystem service systems into the future.

The first connection involves identifying and quantifying ecological system capacities and limits (Rockström et al. 2011), in particular in cases where use of ecosystem services borders on 'over consumption' and impairs regenerative capacity or otherwise stresses the system (labeled A, Figure 4). A second connection involves examining ways in which effective management can inform 'best practice,' 'informed decisions,' and awareness of system vulnerabilities (labeled B, Figure 4). A third connection is using information about benefits resulting from forest management to feed back into those actions themselves (labeled C, Figure 4). This can serve to raise awareness for the benefits of the actions themselves, or to help anticipate system shocks (price or otherwise) when ecosystem service deficits arise and management for resulting ecosystem service losses is needed.

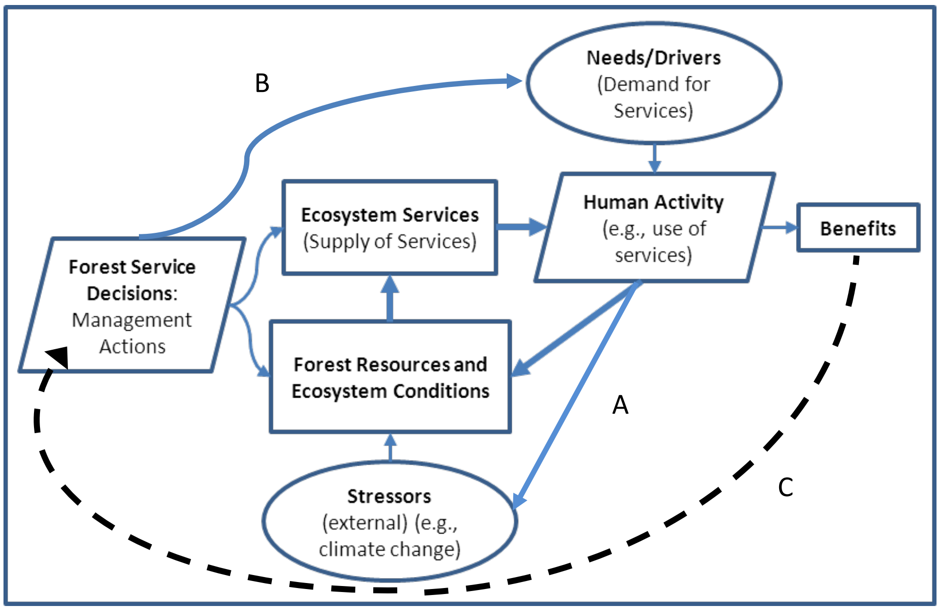

Figure 4: Ecosystem services as a framework for forest management (adapted from Miller et al., in press, as modified from Kline and Mazzotta 2012 and Patterson and Coelho 2009). Three 'challenge areas' are highlighted with labels A, B, C.

Addressing and Assessing Value

A clearly defined system is a necessary starting point for discussions of tradeoffs and value. Many disciplines across the social sciences are needed to fully articulate the many meanings of the word 'value.' It is often assumed (incorrectly) that the terms 'value' or 'valuation' are explicitly referring to the use of dollar figures as a common denominator, when in fact, a much broader interpretation is being implied. If and when it has already been decided that a dollar value is indeed useful, the discussion can then progress to whether the dollar value needed should be estimated by market or non-market terms, and in which form this information might be meaningful in a decision-making context.

Once conceptual and then quantified assessments of ecosystem services have been made, moving to application, evaluation of tradeoffs, and characterization of value becomes an increasingly specialized effort (Kline and Mazzotta 2012, Patterson and Coelho 2009). There are many ways to approach value, and the social tools employed to canvas and incorporate diverse user perspectives also affect ecosystem service emphasis (van den Belt et al. 1998, Spash 2008).

The selection of participants, and their awareness and perception of ecosystem services and their importance, can affect reported values. Studies relying on participant perception may not reflect important components of system complexity, or they may emphasize market over non-market contributions or reflect other equity and distributional predispositions both within and between generations (Brown et al. 2011). More recent approaches designed to control for this shortcoming may facilitate group deliberative techniques, wherein a group is assembled and facilitated with the goal of coming to common agreement on value (Howarth and Wilson 2006, Spash 2007). As with other techniques, a representative sample of the general public needs to be used for the deliberative process to yield values generalizable to the broader public (Brown et al. 1995). And, no matter which method is used, important equity issues remain.

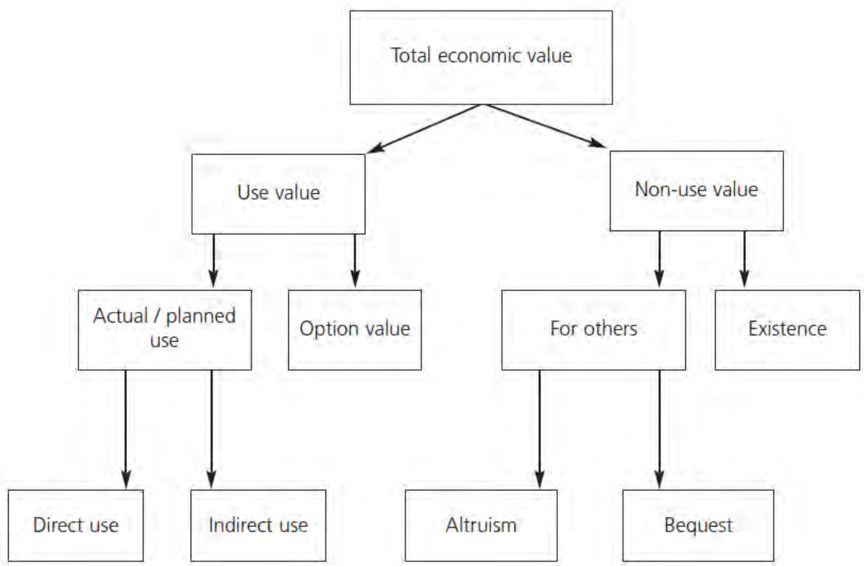

Figure 5: Components of total economic value of ecosystem services. Values are less tangible the further to the right (adapted from Miller et al. in press)

The total economic value (see Figure 5) is inclusive like the MEA model and is suited to the many ecosystem services provided by forests. Although many benefits from forests are tangible, and benefit people through direct use (such as timber products), other forest benefits are harder for people to identify in their daily lives, especially in any quantified form that would be easily associated with forest management actions. Some examples might be additional units of carbon sequestration, or additional quantities of water that may be 'embedded' in production of a final consumer product (e.g., Hoekstra and Hung 2005). Whether ecosystem service uses are categorized as direct or indirect, it is important to underscore that this changes the way they must be accounted for and the clarity and ease by which many consumers and citizens understand benefits. It does not change the fact that many benefits accrue to people indirectly. Biodiversity, described as an ecosystem service, is particularly challenging to address in this regard. Cautions abound in the literature, as each ecosystem service model is only as strong as its ability to describe these indirect connections and values (Brown et al. 2011). Although trends in values are beginning to emerge, meta analyses of existing values have suggested that prediction of a value based on previous studies remains uncertain, and the need for site-specific valuation efforts remains important (Woodward and Wui 2001).

Valuation Methodologies

Economic valuation exercises may provide a useful way to compare change in certain conditions resulting from a management action to a change in welfare experienced by a given set of individuals. This may be relatively straightforward in cases where the trade-offs are well defined, and where market prices exist for each element that the user considers of value. However, particularly in the public goods context, this is often not the case, and it adds a great deal of complexity to the task of evaluating tradeoffs among land use and land management objectives (NRC 2005).

Traditional approaches offer many techniques to elicit value. Values may be stated (Sugden 2005), revealed in preference studies (Bockstael and McConnell 2006), queried via willingness to pay (Carson and Mitchell 1993, Brouwer et al. 1999, Wilson and Carpenter 1999), estimated by travel cost method (Smith and Desvousges 1985), or transferred from other studies (Rosenberger and Loomis 2001), among other approaches.

Non-market estimation techniques for ecosystem service valuation have advanced a great deal in recent years (Freeman 2003, Loomis 2005). These techniques have included travel-cost methodology and contingent valuation (Loomis 1999, Loomis and Walsh 1997), among others, to estimate use of many ecosystem services. Although valuation efforts of ecosystem services have often focused on direct uses, passive uses are also of high value to users of public forest lands (ibid). Existence values, option values, and bequest values may be the highest economic values for certain protected areas (Loomis 1987, 1989), and also serve as important values to biodiversity and science, and education (Balmford 2002). Despite this awareness in general, pragmatic and specific decisions are still reliant on effective collaboration between ecologists and economists to ensure that the model is accurately reflecting the necessary level of ecosystem complexity. As Brown et al. reported (2011), the devil often lies in the details of these valuation exercises.

Quantifying ecosystem service changes that result from changes in ecosystem conditions, land use, and land management is a substantial challenge and therefore adds costs and, sometimes, barriers to ascertaining value (Kline and Mazzotta 2012). Benefit transfer techniques (taking an average value from existing valuation studies or using estimates from an existing study in a new one) (e.g., Rosenberger and Loomis 2001, Loomis and Rosenberger 2006) are therefore attractive, particularly where cost, method, and logistics on public lands are otherwise prohibitive (Iovanna and Griffiths 2006). However, numerous writers have pointed to shortcomings in these techniques (Ready and Navrud 2006, Spash and Vatn 2006), which require concerted efforts to overcome (Hoehan 2006, Loomis and Rosenberger 2006, Feather and Hellerstein 1997, and Smith et al. 2002, as summarized by Wainger and Mazzotta 2011).

Payments for Ecosystem Services

Increasingly, public institutions are relying on the emergence of market mechanisms to incentivize the provision of ecosystem services, especially to conserve forest as land cover (Collins and Larry 2008).

Indeed, addressing 'provision' of ecosystem services represents only a partial solution to rising ecosystem service deficits, and is addressed in the next section of this chapter.

A great deal of enthusiasm has been expressed for market-based approaches to ecosystem service provision. Over-extensions of market-based tools have led to pleas for a more 'rational exuberance' (see review in Kline et al. 2009). Markets are not a complete solution for the challenge of ecosystem service provision, because the vast majority of ecosystem services are not and will never be marketable. Certain characteristics of ecosystem services can determine whether a market-based tool may result in a useful and efficient way to incentivize production (Table 1). Yet even if these characteristics fit the ecosystem service issue at hand, distribution and equity issues may be left unaddressed, and this also entails management consideration.

Market efficiency assumes that that certain characteristics apply to the good or service at hand. In the most basic terms, each credit (or equivalent ecosystem service unit) must be able to be consumed as a private good (as opposed to collective consumption), and can be excluded from those who do not pay (Randall 1999) for markets to be efficient in their provision. Table 1 summarizes these characteristics.

Table 1: The public/private nature of goods. Markets are generally most effective when applied to the ecosystem services categorized in the lower right hand corner (adapted from Randall 1999)

	Low rivalry (collective consumption)	High rivalry (private consumption)
Difficult to exclude (unlimited access)	<u>Public goods</u> Scenic views, biodiversity clean air, carbon sequestration	<u>Common goods</u> Fresh water, fish stocks
Easier to exclude (limited access)	<u>Club goods</u> Private parks, car parks, recreation areas, ski areas	<u>Private goods</u> Timber, food, non-wood products

Increasingly, efforts are being made to move certain explicit and quantified ecosystem services from one quadrant to another, by modifying the excludability and rivalry characteristics of a well-defined 'proxy,' such as design of credits with which to track and trade in carbon sequestration (EUSTAFOR and Patterson 2011). The Forest Service, in partnership with other private and not-for-profit partners, has suggested that this may offer potential for more market-diversified product offerings (Collins and Larry 2008).

In market-based applications, additionality is a key concept that is often overlooked (Wunder 2005, 2007; Engel et al. 2008; Patterson and Coelho 2009). Additionality characterizes the extent (if any) to which the action, market, and payment increase the provision of the ecosystem services above and beyond that which would have been provided under a business-as-usual scenario. Payment systems may be initiated with seed funding, but in the absence of additionality, the credibility and longevity of ecosystem service market structures over time may be undermined (Wunder 2005, 2007).

Addressing Rising Demands for Ecosystem Services

Successfully addressing emerging deficits in ecosystem services requires stemming decline in ecosystem service production, as well as ensuring ecosystem service use is not wasteful or needlessly impactful to the systems that provide them (Patterson and Coelho 2008, Patterson and Coelho 2009, Beier et al. 2009). Management tradeoffs are often considered in planning because they produce different bundles of services (Maness 2007), and awareness for and interest in various ecosystem services changes over time. Assessment of tradeoffs over space and time will thus require explicit definition of the area and time-step being considered, and identification of beneficiaries both near and far (information that is often lacking). For this reason, forest management and valuation efforts often focus on the supply side of ecosystem service information. Yet public funds can be spent both on maintaining or increasing supply of ecosystem services as well as on preventing waste or conserving ecosystem service use. The latter options can also be highly cost effective in addressing situations where ecosystem services have become particularly scarce.

Although ecosystem service data describes in general terms human dependence on natural systems, this information is difficult to tie to the management organizations, municipalities, households, and individuals making decisions about ecosystem service use. Visualization efforts have attempted to raise awareness of where ecosystem services are produced (Naidoo and Ricketts 2006, Natural Capital Project 2010, Ricketts et al. 1999), but only a few studies have mapped ecosystem service uses or potential for disturbance (Beier et al. 2009). Synthetic indices (e.g., ecological footprints, carbon calculators, sustainability indicators, and sustainability report cards) are increasingly being used by states, cities, corporations, and individuals (Wackernagel et al. 1999, Wackernagel and Rees 1996, USFS Business Operations 2007, Patterson and Coelho 2009). Forest Service management, as a requirement of Executive Order 43514, has targeted reductions in each of seven ‘footprint’ areas since 2007 (USFS Business Operations 2007).

Federal entities, which often have large building footprints, fleets, and equipment portfolios, offer valuable opportunities for experimentation, innovation, and investment in conserving resources and reducing pressure on ecosystem service systems. The USDA Forest Service Sustainable Operations program works actively to daylight consumption trends of the agency, provide tools for cost-benefit analysis, and promote efficiency and behavior change. Decreasing the agency’s demand for resources and energy affects multiple ecosystem service systems and operating costs simultaneously. These business practices can include behavioral changes, such as turning off lights and computers when not in use; watering landscapes less frequently; recycling; using fuel efficient vehicles, energy efficient appliances, and electronics, and water aerators on faucets; minimizing packaging; and fixing water leaks—these are just a few of the items summarized by the Sustainable Operations program.

Conclusions and Directions for Future Research

This chapter has provided a review and synthesis of the ecosystem service concept, which will be important as planners and managers conduct assessments in the synthesis area. The chapter discusses different definitions of the ecosystem service concept, offering examples of its diverse applications. The

Frameworks section of this chapter highlights the current emphasis on the supply of ecosystem services, but it also describes three 'challenge areas' that can assist the agency in its aim to 'tell the story differently' and ultimately utilize a broader range of ecosystem service interventions to address deficits before they become acute.

The Sierra Nevada synthesis area has a unique opportunity to contribute a vivid and prominent case study to an emerging area of concern nationwide—specifically, quantifying and illustrating the dependence of urban areas, communities, and households on surrounding natural systems and the flows of ecosystem services that they produce. To date, this has been done in an abstract 'ecological footprint' approach, but the Sierra Nevada case illustrates an opportunity to be more explicit both in spatial and in ecosystem service terms. Habits, lifestyle, technology, social norms, and rules, incentives, and penalties all determine the rate at which humans collectively use ecosystem services. Urban areas are examples where this use is particularly concentrated, and this concept is acutely felt in California due to population expansion, land conversion, drought, and other factors that escalate demand for ecosystem services, or which interrupt their supply. These systems serve as valuable test cases that underscore the value of well-managed landscapes, and demonstrate the degree to which quality of life is dependent on flows of reliable ecosystem services. Unfortunately, the presence of tipping points for provision of ecosystem services is often not explicitly understood until after substantial economic, cultural, and social losses have occurred, and by then, the cost to replace those services is often prohibitive.

Thomas (2012) addressed the important distinction between strong and weak sustainability, and how it bears on the agency's ability to use GTR-220 to build on stakeholder collaboration and system sustainability. Future research is needed to articulate how methods and approaches from information management, systems analysis, and business and capital management strategies can help address the challenges described above in a cost-effective way. Increasingly, ecosystem service scarcities are spurring partnerships between public, private, non-governmental, and academic sectors (Smith et al. 2011). Forest Service Sustainable Operations provides several place-based, strategic, and quantified efforts in this regard, and these have been of great interest also to partnering organizations from municipal and non-governmental sectors. To date, however, these findings have been post-hoc, with few to no studies systematically comparing among options, testing hypotheses, or establishing baselines, experimental design, or statistical controls. These shortcomings can be overcome with some foresight, planning, and sharing of information, particularly between scientists familiar with experimental design, engineers familiar with the systems (water, electrical, fleet), and members of business operations who can reveal units and current and historical billing and prices for the ecosystem services currently consumed in agency operations. Systemic solutions that address declines in ecosystem services require a coordinated approach to energy and material inputs to the economy (and the resulting waste and emissions), and projections of these for future time periods (Folke et al. 2004, Rockstrom et al. 2009). The ecosystem service concept presents an opportunity for the agency to take a more diversified approach, and in doing so, it may offer an opportunity for experimentation with a broader, whole-systems strategy to support landscape and community, resilience and sustainability.

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9.3 Sociocultural Perspectives on Threats, Risks, and Health

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Introduction

This chapter examines the intersections of human and ecological health, and the anticipated impacts of ecosystem dynamics and threats in the Sierra Nevada. By following the chapter on ecosystem services with this discussion of linkages between ecological and human health, we hope to further illuminate the importance of managing for socioecological resilience using an adaptive approach. Although the selected dynamics and threats are discussed individually, common to all of them are varying degrees of uncertainty. For example, we can talk about impacts of climate change, yet because the anticipated changes are likely atypical and uncharacteristic of past patterns, readiness to identify, adapt to, and mitigate newly discovered yet unknown impacts—on social, cultural, and economic systems—will remain essential to effective management. Selected findings surrounding risk perception and risk management are presented. These are anchored with examples of how findings may improve risk management into the future.

How Sierra Nevada Ecosystem Health is Related to Public Health and Well-Being

Environmental and social health are intertwined, and relationships between the two have generated significant attention. "The overriding premise of the MEA[1] is that the condition of the environment makes a vast and essential contribution to the quality of life in general" (Williams 2006: 147). The connections between human health and forests hold great potential for improvement of well-being (Karjalainen et al. 2010), and understanding the linkages can greatly aid efforts to conserve and restore forests (Hernandez et al. 2012). By emphasizing the value of healthy ecosystems for social, cultural, and economic health, managers, researchers, and stakeholders alike have an opportunity to effectively frame why actions to protect an ecosystem are valuable investments.

In The Broader Context for Social, Economic, and Cultural Components chapter (9.1), we introduced the benefit of recreation and tourism in aiding the development of connections to place. There are, however, myriad benefits to managing for quality outdoor recreation experiences. Outdoor locations offer unique opportunities for engaging in active living through recreation and leisure, thus representing a benefit to physical and social health (Cronan et al. 2008, Gobster 2005, Rosenberger et al. 2005); a chance to develop connections to natural spaces, thus offering a place to develop bases for stewardship and caring that further protection of the physical environment and contribute to resilience (Clayton and Myers 2009, Crompton and Kasser 2009, Williams 2006, Winter and Chavez 2008, Zavaleta and Chapin 2010); a place to celebrate culture and family (see for example Anderson et al. 2000, Gunderson and Watson 2007); a place for restorative experiences (Kaplan 1995); and myriad other important benefits too numerous to list here (for reviews, see Clayton and Myers 2009, and specific to wilderness, Cordell et al. 2005).

Dialogue with stakeholders, including forest community residents, can help managers in the identification of valued ecosystem services. In addition, discussions of valued services can facilitate stakeholder recognition of benefits they may not be aware of or value (Asah et al. 2012). Given the projections of diversity of cultures and accompanying diversity of values that will continue to characterize visitors and residents in the Sierra Nevada and surrounding areas, engaging stakeholders in an ongoing and adaptive process for forest management practices and decision making is important.

Attachment to the natural environment, influenced by natural landscapes and views, presence of wildlife, and opportunities for outdoor recreation, has been demonstrated as a component of community attachment and well-being (Brehm et al. 2004). This is an especially important finding in the Sierra Nevada because of the strong influence of amenity migration and the independence from length of residency. Relative newcomers represent valuable contributors to community well-being. Long-term residents and newcomers alike find value in ecosystem quality and resilience for a variety of reasons, as may those who have no residency ties but have formed other connections to place (Kaltenborn and Williams 2002).

[1] MEA is an abbreviation for Millenium Ecosystem Assessment.

Rural communities in the Sierra have been through political, social, economic, and environmental transition. As these transitions have occurred, economic well-being in a number of Sierra Nevada communities has drawn increased attention. Though much attention has been paid to poverty in urban areas (owing in part to the large proportion of populations centered in urban locales), poverty in rural areas has received less attention. Examining rural communities in the Sierra Nevada synthesis area offers the opportunity to assess connections between poverty and well-being and linkages to ecological quality, short and long term. It is through identifying these linkages that we discover the layering of multiple threats that results in additive effects on Sierra Nevada community residents.

Evans et al. (2009) highlighted the unique experiences of rural youth living in poverty. These youths experience more day to day stressors than their middle-income counterparts. Evans et al. (2009) also reported that the greatest number of low-income American children and youth are white, and they are disproportionally represented in rural areas. Economic stressors in the Sierra Nevada suggest that some communities may be of particular concern. Evans and Rosenbaum (2008) documented generational impacts of poverty that are longstanding and affect cognitive and socioemotional processes, influencing life-long development and outcomes in adulthood. Evans and Kim (2010) connected multiple environmental and social risk exposures to socioeconomic status, highlighting the importance of understanding that poverty is typically linked to additional social risks as well as environmental risks. Evans (2006) provided linkages between childhood development and environmental quality, pointing to the importance of ecological health in proper development of future generations.

These findings are applicable to fostering socioecological resilience because environmental condition affects development, environmental quality and opportunity is linked to community and economic resilience, and poor conditions both environmentally and economically have costs that are demonstrated to have both immediate and long-term impacts on generations of youth. The cumulative risk assessment framework presented by deFur et al. (2007) is helpful in understanding issues of individual exposure to risk, impacts of cumulative risks upon the same individual, and individual and community resources available to respond to risk. Their framework echoes back to the theme of socioecological resilience, as it pairs human and ecological systems and the multiple risks each is exposed to as a way to conceptualize vulnerability and understand well-being. In a later chapter of this synthesis, we address community resilience and forms of risk as well as management tools to contribute to community resilience (chapter 9.4); however, it is useful to point out here that the current and future socioemotional well-being of human populations in the Sierra Nevada is directly linked to an array of influences, including resilience of the Sierra Nevada ecosystem.

The increasing cultural diversity within the synthesis area is an additional factor in risk management, as it clear that risk is perceived and acted upon in different ways by individuals and is sometimes influenced by culture (Earle and Cvetkovich 1999, Lindell and Perry 2004). Another significant factor is the need for environmental justice (Greenberg et al. 2012). Increasing cultural diversity requires consideration of differential exposure to risks and subsequent differential impacts of exposure, the ability of vulnerable communities to adapt to or mitigate risk, and effective approaches to working with communities in addressing risk as well as in communicating risk. Diversity requires the ability of managers to understand and take into account increasingly complex value sets, relationships to the synthesis area, and relationships to the managing agency.

Specific examples of the link between social and environmental health within the Sierra Nevada are represented in other chapters in this synthesis. The Air Quality chapter (8.0) presents a series of studies pointing to elevated ozone levels that exceed public health standards, thereby presenting a direct risk to health. This elevated risk has been identified in multiple locations in the Sierra Nevada, especially on the western slopes adjacent to the highly polluted California Central Valley. Poor air quality is of special concern in a number of ways. Year-round residents situated in the areas with documented high concentration levels, or frequently travelling to those areas in their local surroundings or for short distance day trips to nearby locations, are exposed to elevated ozone concentrations. Given the temperate climate during most months, and the natural amenities surrounding year-round residents, it is likely that residents spend a portion of their time outdoors and therefore have a greater exposure than if they were indoors most of the day. Sensitive populations that would be more adversely affected by poor air quality include the elderly, the very young, and those with respiratory conditions classified as chronic obstructive pulmonary disease[2]. Additional concerns for the recreating public are also worthy of note, as much of the Sierra Nevada is a prime recreation and tourism destination. Recreationists engaging in more physically exerting activities such as hiking or mountain biking would be of greater concern than those relaxing or enjoying more stationary activities. Those coming for shorter term visits with little time to acclimate to the higher altitude are likely at greater risk from degraded air quality. It may be prudent to warn the recreating public about the risks associated with increased ozone in the southern Sierra Nevada during the summer season (Cisneros et al. 2010). Discussions of air quality issues in montane forests offer additional insights into issues surrounding the public and risk associated with air pollution (see Winter 1999).

This chapter now turns to three specific ecosystem dynamics and threats, to further explore connections between social and ecological well-being in the Sierra Nevada.

Ecosystem Dynamics, Threats and Risks

Climate Change

Public perceptions surrounding climate change

A national study of American perceptions of climate change revealed that there is a high level of awareness of global climate change, a belief that it is real, and a significant concern. Nevertheless, the impacts are overall believed to affect distant peoples and lands, and to be of moderate severity (Leiserowitz 2005). This study also revealed segments of the population that believe that climate change is a fabricated hoax, alarmists with extreme perceptions of risk, and many in between (the majority). Thus, ideas about climate change vary among the public, as they do among politicians and environmentalists (Leiserowitz 2005). This makes the ability to manage for changes in climate more

[2] For a discussion of threats to human health from ozone see www.epa.gov/apti/ozonehealth/population.html.

challenging; without an understanding of risks, it will be more difficult to address changes needed at a larger scale, such as limiting or reducing the demand for and use of water from the Sierra Nevada if climate change leads to decreased ability to provide water for downstream uses.

Maibach et al. (2008) suggested that framing of messaging and outreach efforts must be tailored to address this diversity of viewpoints and values to have the best chance of being effective in changing behavior and policy to address climate change. The framing of climate change is essential to bridging perceptual divides and increasing understanding of climate change impacts, and generating support for the actions needed for mitigation and adaptation (Nisbet 2009).

Sidebar: Public Perceptions of Climate Change

A recent study in California demonstrates the ongoing complexities of public perceptions of climate change and its impacts. A majority or near majority of Californians are very concerned about possible impacts of global warming in the state, which include more severe wildfires (56 percent), increased air pollution (48 percent), and more severe droughts (45 percent) (Baldassare et al. 2011). A majority believe the effects of global warming have already begun, and it is necessary to take steps to counter the impacts right away. However, global warming is not among the top five most often mentioned environmental issues in California. Air pollution remains the top issue for Californians in the most recent statewide survey (27 percent), followed by water pollution (8 percent), water supply (8 percent), and energy (7 percent)(Baldassare et al. 2011).

Koger et al. (2011) suggested that framing climate change as a global environmental issue may make it distant or too removed from personal responsibility, thus preventing necessary actions to mitigate impacts. They suggested reframing climate change to focus on the immediacy and local nature of issues and impacts, and emphasizing behavioral control and actions that are problem focused. They also suggested emphasizing the public health issues involved, and the health benefits of preserving nature, thus providing a positive rather than negative framing for promoting action.

Including information about the potential social, demographic, and economic disruptions from climate change in addition to physical health impacts broadens the consideration of well-being and represents a wider range of values that might motivate support for mitigation measures and personal behavioral changes (Bain et al. 2012, McMichael et al. 2006).

Related research highlights a number of effective strategies for communicating about climate change; these include anchoring climate change discussions in ways that demonstrate impact to locales specific to the target audience and peoples viewed as similar to themselves, as well as stressing that impacts are expected sooner than later and are more certain than uncertain (see Spence et al. 2012). Likelihood and severity of localized impacts has also been suggested as important in the adoption of and investment in adaptation measures among agency decision makers (Syal et al. 2011). It is clear that uncertainty may be acceptable when the audience understands that uncertainties are part of a deeper understanding of complex mechanisms such as climate change (Rabinovich and Morton 2012). In this case, communicating the role of science, and revealing the complexities and uncertainties of impacts, would be just as important as relaying findings about climate change.[3]

Impacts of climate change

Expected impacts of climate change on tourism worldwide vary based on market segment and geographic region, but may include a decreased winter sports season, heat stress risks for tourists, risks of exposure to infectious diseases, increased competition for recreational opportunities and alternate uses dependent on water, loss of natural attractions and species in ecosystems, decreased access and compromised experiences from more frequent and larger wildfires, and changes in soils that may alter ecosystem impacts of uses (World Tourism Organization and United Nations Environment Programme 2008).

Research has identified many local impacts of climate change, including those presented here. Maurer (2007) outlined hydrologic impacts of climate change in the Sierra Nevada under two scenarios. With expected increases in temperature, he projected an increase in winter stream flow from increased precipitation, and decreasing late spring and summer flow associated with lessened snowpack at the end of winter. These anticipated shifts will not only have impacts on demands for water management (Maurer 2007), but they will also present challenges for California communities that are heavily dependent on water supply from the Sierra Nevada (i.e., they will broadly impact ecosystem services, see for example LADWP 2010, Morelli et al. 2011), and they will likely have impacts on spring and summer recreation and tourism, especially those activities that are water dependent.

Researchers have developed models to characterize the potential impacts of climate change in the Sierra Nevada, and these models may further aid planning and anticipation of impacts. Null et al. (2010) modeled sensitivity of various watersheds in the Sierra Nevada; they suggested that watersheds in the northern Sierra Nevada are most vulnerable to decreased mean annual flow, southern-central watersheds are most vulnerable to runoff timing changes, and the central portion is most sensitive to longer periods of low flow. They suggested that the Kern River watershed is the most resilient to climate impacts. The potential for flooding effects on downstream communities for the western Sierra Nevada has been studied by Das et al. (2011). Using down-scaled climate/precipitation models, their models predict larger-than-historical floods for both the northern Sierra Nevada and the southern Sierra Nevada, with increases in flood magnitude projected for the period 2051–2099. These projections highlight the importance of planning for increased flood events and developing contingency plans that provide approaches to mitigation and adaptation.

[3] A recent synthesis of the potential and ongoing contributions of psychology to address climate change impacts may be helpful to the reader of this report (see Swim et al. 2010).

Research and modeling efforts have resulted in some tangible management implications. For example, Mehta et al. (2011) recommended that climate change-induced hydrological change be included as a foreseeable future condition in planning and in FERC relicensing. Peterson et al. (2011) produced a guidebook on responding to climate change, and it may be useful in larger scale planning and adaptation. It is evident that climate change is incredibly complex, and requires working at a long time scale, large geographic scale, and across agency and institutional boundaries, and a willingness to accept a degree of uncertainty (Barbour and Kueppers 2012).

Climate change has also been reported as a contributing factor in larger, more severe wildfires, reviewed in the following section.

Wildland Fire

Public perceptions of wildland fire and risk

Sociocultural and economic aspects of wildland fire management have been an area of intense study for the last decade, stimulated by funding from the National Fire Plan that increased support for related work. A number of comprehensive reviews are available on this topic; here we review only some of them. Some of this research emerged from the risk management field, and may be instrumental in understanding management of other risks and threats. However, caution should be exercised in this

approach, as not all risks are viewed equally or are associated with the same sociocultural concerns.

People living in high fire risk zones tend to be unduly optimistic about the degree of risk involved, believing the risk of fire is less than what it actually is (Kumagai et al. 2004). The risk of wildland fires receives low levels of consideration when prospective residents are considering purchasing a home in a fireprone area, though once residency is established homeowners give some consideration to risk (Vogt 2008). In many cases, residents in fireprone communities have been found to take a number of risk reduction actions (Absher and Vaske 2007, McCaffrey 2006, Vogt 2008). Perceived risk is not the only influence in defensible space actions, as individuals need to be confident in their ability to perform the action (Martin et al. 2008), and they need to feel that the action will be effective in reducing risk (Martin et al. 2008).

Collaborative approaches to fire management and risk reduction tend to contribute to effective risk management. Successful approaches require addressing knowledge gaps between experts and laypersons in order to increase effectiveness of engagement efforts (Simons and Arvai 2004). A benefit of the collaborative process is the opportunity for the risk manager to learn stakeholder perspectives on the places of concern or the techniques involved, as well as to address gaps in knowledge they have given their perspective as experts (Slovic 2000). Fostering a mutual learning rather than instructive approach is characteristic of this mode of addressing management. However, building public understanding and agreement requires a long-term commitment (Olsen and Shindler 2010), and involves fostering and building trust and confidence among participants and the managing agencies (see Rivers et al. 2008, Winter et al. 2007).

Impacts of wildland fire

The complexities of fire management have increased in the Sierra Nevada and Sierra Nevada foothills, in part due to increased development in the WUI (Hammer et al. 2007). The importance of participating in local and regional land use planning efforts and discussions of fire risk has increased, as has the need for agencies to collaborate across boundaries, and with citizens and community groups.

Fire can evoke significant emotional distress and panic during a fire event (Simons and Arvai 2004), and lingering psychological impacts associated with a fire event and fire risk were shown to affect residents proximate to fireprone forest lands (Cvetkovich and Winter 2008). Fires that directly impact forest communities can alter community structure; however, engaging community members in collaborative approaches to recovery may be effective in restoring and healing impacts of the event (Burns et al. 2008). Smoke is one specific area of concern to individual health from wildfires. Fowler (2003) reviewed human health impacts of forest fires. She pointed to the importance of evaluating forest fires from the perspective of gains (improved social, cultural, economic, and political systems) as well as risks (for example, through short and longer term impacts on public health). Specific impacts of concern to vulnerable populations overlap those for air quality in general and include impaired visibility from smoke as well as health effects on young children, the elderly, and individuals with pre-existing conditions (Fowler 2003). Of additional health concern is the occupational exposure for firefighters (Fowler 2003). The literature on impacts to infrastructure from impaired air quality may be helpful in pointing to additional areas of consideration, for example the damage caused to exteriors of buildings from pollutants (see Winter 1999). Sandburg et al. (2002) examined the effects of fire on air quality and provide some analyses of impacts from fire and smoke pointing to the effects of damage to infrastructure and reduced highway safety. McCool et al. (2007) provide an extensive review of wildland fire impacts on communities at the individual, family, neighborhood, social group, and community scales, demonstrating the complexities of scale when applying social science to management of fire.

Large wildfires may impact soils, in turn affecting human health. The Post-wildfire Management chapter of this synthesis (4.3) provides evidence that large wildfires can increase the release of heavy metals in soils (see also Pereira and Ubeda 2010, Scherbov 2012), and that metals have likely built up over time through atmospheric deposition (see Air Quality chapter (8.0)) in the Sierra Nevada. If concentrations of heavy metals occur in riparian areas, effects may be increased further through movement in stream channels from the particular location to some distance away. Even if the metal concentration is stationary in a specific area, it still may present threats to human health. Studies examining heavy metal concentrations demonstrate the transfer into food supplies, particularly in areas where residents engage in outdoor gardening, or where residents and visitors collect edible forest plants and fungi (see for example Alm et al. 2008). Fire is only one of many ways that heavy metals may be introduced into the ecosystem and subsequently into the food supply (Sharma and Agrawal 2005), but it remains an important area of concern in the Sierra Nevada and other fire prone regions.[4]

Impacts on recreation and tourism

Fire has impacts on recreation and tourism that in turn may have economic impacts. For example, a fire in July of 2000 was associated with decreased economic activity and visitor expenditures when fire crews filled up local lodging and smoke lingered in the Kern River Valley for several weeks, impacting local scenery and air quality (Colby and Smith-Incer 2005). Studies suggest there are a number of economic costs of forest fires that are not typically considered (Dunn et al. 2005, Yoder and Blatner 2004), and when they are accounted for, investments to reduce fire risk and increase treatments may seem more financially prudent (Yoder and Blatner 2004).

Longer term effects of wildfires on recreation and tourism have also been examined. Wilderness visitation is affected by fire succession according to Englin et al. (2008), who reported that large wildfires are followed by an increase in the number of trips to an area, but over the longer term (40 to 50 years out), large areas burned by wildfires experience decreased demand. Further studies are needed to understand the dynamics underlying these patterns, but in the interim, these fire-caused shifts in demand may be important for planning purposes.

Loomis et al. (2001) reported variable effects of forest fires on recreation and tourism associated with the intensity of the fire and recreation use activity. Effects can vary, depending on impacts to the landscape and the activity in question; for example, hikers find obstructions less of an issue along a trail than do mountain bikers. Similar to Englin et al. (2008), Loomis et al. (2001) reported a decrease of use in some areas over time; however, this effect was for hikers. Recovery of an area was associated with increased mountain biking activity. Loomis et al. (2001) suggested practicing agency communications

[4] Mining is another likely source that has introduced heavy metals into the ecosystems in the synthesis area, addressed within the Watersheds and Stream Ecosystems chapter (6.1).

that allow user groups to understand fire impacts and make informed choices about where to go based on recency and type of fire. This approach might help mitigate economic losses associated with reduced tourism after a fire.

Other studies suggest minimal impact of fires on the overall experience of recreationists (Winter and Knap 2008) and tourists (Thapa et al. 2008). However, high fire danger conditions (Thapa et al. 2008), smoke from a nearby fire (Winter and Knap 2008), and health problems from smoke and ash (Thapa et al. 2008) are viewed as bothersome, and in some cases, these issues are of sufficient concern to inspire changes in travel plans. These findings about the impacts of fire are important, as they seem to point to more impact on the recreation visitor experience than anticipated by managers (see Bricker et al. 2008).

Social and institutional factors regarding smoke emissions

To dramatically reduce the legacy of fire suppression and associated fuel loading and restore the role of fire would require a sharp increase in the level of burning and emissions, which in turn would require increased political support (Stephens et al. 2007). Public land management agencies have an incentive to respond to short-term, local complaints about smoke while discounting hypothetical impacts from future wildfires. The fact that wildfires are often excluded from the regulatory constraints that apply to prescribed burns further diminishes the incentive to avoid wildfires through prescribed burning. Efforts to increase burning raise equity concerns by asking current residents and tourism-related businesses to bear a burden partially created by prior generations in order to mitigate impacts to future populations. Education, notification, and other outreach measures may help to diminish residents' concerns, but fundamentally, prescribed burning requires sacrifice on the part of present-day, local residents for the sake of a greater public good.

Invasive Species

The impact of invasive species can be extensive, resulting in economic losses, permanent ecological changes (such as the loss of native species), and effects on public health and well-being (Anderson et al. 2004). Emphasis on the impacts of invasive species, including pathogens or diseases, tends to focus on only a portion of ecosystem services (Charles and Dukes 2007). However, with an increasing need to clarify impacts of invasives to the public at large, and to weigh management options in terms of costs and benefits of management and prevention, a broader approach is suggested. This broader approach would incorporate impacts on 'regulating ecosystem services,' including ecosystem processes affecting air quality, climate, water, disease, and erosion. Charles and Dukes (2007) demonstrated the importance, for example, of considering the role of invasives in increasing fire risk, thus increasing concerns over degraded air quality and associated effects. Impacts to fire regimes of the Sierra Nevada can also occur from invasives (Brooks et al. 2004), thus affecting values or conditions of ecosystem goods and services. For example, the economic impact of weeds on wildlife-related recreation in the Sierra Nevada was recently estimated between $6 and $12 million per year (Eiswerth et al. 2005).

Emergent findings also encourage consideration of invasive impacts on cultural ecosystem services, including aesthetic value and tribal uses and access (Pfeiffer and Ortiz 2007). Finally, Charles and Dukes (2007) pointed out the need to consider impacts on supporting ecosystem services, such as longer term ecosystem dynamics (e.g., photosynthesis or soil nutrient cycling). The authors noted, however, the relatively low availability of completed work outlining impacts of invasives on regulating and supporting ecosystem services; this represents an important gap in the information necessary to fully assess and select appropriate management investments into the future.

Finnoff et al. (2005) pointed out the importance of examining a bioeconomic feedback loop in invasive species management, considering the expected benefits of adapting or controlling invasives versus lost benefits expected through inaction. An example for native versus exotic fish species demonstrates the complexity of recreational values held by the public. Some stakeholders, such as fishermen using national forests, may value more "pristine" lake, stream, or river fish communities compared with others who want the opportunity to "catch a fish" regardless of the species' origin or ecological function (Moyle et al. 2003). As a solution to the considerable expense of estimating impacts across multiple systems, Holmes and Smith (2008) suggested use of spatial benefit transfer methods, or using the economic values derived from one study site and applying them to another comparable site. For this approach to be effective, however, research needs to be conducted with sufficient replication of findings and the ability to identify what aspects of a novel setting or system may require adjustment in the calculation of benefits.

Management of invasives, both in remediation and prevention, requires an assessment of priorities and the weighing of perceived effectiveness (Randall 2010). It further requires a deliberative process to address the multiple and sometimes conflicting values, outlined as values structuring and probability modeling in formal decision analysis (Maguire 2004).

The management of invasives is especially difficult in areas with high land use diversity and increasing subdivision of lands between management agencies, referred to as "management mosaics" by Epanchin-Niell et al. (2010). Collective action is necessary across agency boundaries to effectively address control of invasives, and this is more difficult when there is a greater number of managing agencies involved. Here then is another example of how agencies will need to work across institutional boundaries to accomplish socioecological resilience.

Decision-making Science and Effects on Risk Management

Sustainability assessment tools and the indicators selected within them reflect the values of the evaluators who select the tools and indicators (Gasparatos 2010). In the Broader Context for Social, Economic, and Cultural Components chapter (9.1), sustainability surrounding recreation and tourism was discussed, including efforts to encourage global use of metrics for sustainability. How ecological, social, economic, cultural, and institutional sustainability are conceptualized and measured will reflect a set of values regarding what is important and should be focused on. The same can be said regarding the cut-offs for change in management direction, presented in the Synopsis of Emergent Approaches (1.2). In order to be meaningful and improve sustainability, selected indicators and feedback loops would consider impacts to affected stakeholders. Since stakeholders are unlikely to arrive at consensus on singular values, a mix of indicators and values may be necessary.

All considerations are not weighed equally in decisions regarding risk. For example, ample evidence suggests that gains tend to be discounted more than losses in environmental decisions (Hardisty and

Weber 2009), making planning for socioecological resilience and the reduction of long-term risks more challenging. In addition, short-term losses gather more attention than longer term ones, due in part to the belief that some change intervention will be possible in the future to mitigate longer term losses (Wilson et al. 2011). The focus on addressing and preventing short-term losses and risks further impedes the ability to address longer-term sustainability and resilience.

Furthermore, institutional, political, and social constraints impinge on managers' decisions and should be accounted for in modeling of socioecological resilience, supporting tools, and suggested applications (Dellasala et al. 2004, Horan et al. 2011, Quinn-Davidson and Varner 2012). For example, Williamson (2007) reported that US Forest Service district rangers cited a concern over lack of agency support (through limited budgets and the risk of personal liability) in decisions surrounding wildland fire use. Air quality regulations were also cited as an impediment. Areas of public concern, including smoke, risks to threatened and endangered species habitat, and resource damage were also cited as influencing decisions about fire use. Thus, recommended approaches need to incorporate contextual factors, not only in the recommendations offered for management, but also in the selected indicators for monitoring. Contextual factors need to be realistically examined in discussions of management of threats, and they need to include a feedback loop to account for changes over time.

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9.4. Strategies for Job Creation through Forest Management

Susan Charnley with a contribution from Patricia L. Winter

Contents

Executive Summary

Forest managers can contribute to community well-being by designing projects that accomplish forest management in ways that not only meet their ecological goals, but also create economic opportunities for nearby communities. A number of strategies for enhancing the economic benefits to communities of forest restoration work, infrastructure maintenance, and improvement projects, and managing for recreation and tourism, are provided in this chapter and are summarized near the end in a "Management Implications" box. These strategies include (1) making use of existing authorities and tools that enable managers to consider benefits to local communities when undertaking forest restoration; (2) being strategic about where and how projects are accomplished; (3) implementing projects that build on local community capacities and priorities; and (4) maintaining and developing recreation opportunities, infrastructure, and partnerships. If managers consider how to create these benefits when planning projects, they may increase the overall socioeconomic benefits of national forest management and help contribute to community resilience. Investing in communities can also benefit the health of forest ecosystems.

Introduction

The literature on community-based forestry in the United States suggests that healthy forest ecosystems and healthy forest communities are interdependent (Baker and Kusel 2003, Kusel and Adler 2003, Kelly and Bliss 2009). The focus of this chapter is on how national forest management may contribute to the socioeconomic health and resilience of forest communities in the Sierra Nevada through job creation associated with forest restoration, recreation and tourism, and infrastructure maintenance and improvement. Job creation associated with managing forest products, specifically timber, biomass, non-timber forest products, and forage for livestock, is the topic of Chapter 9.5. The chapter also draws attention to the ways in which investing in job creation through forest management may contribute to the health and resilience of forest ecosystems (Kelly and Bliss 2009). Forest communities are defined here as communities having social, cultural, or economic ties to nearby forest lands.

In the 1990s, forest restoration became the focus of federal forest management in order to restore watershed health, control invasive species, reduce fire hazard, enhance wildlife habitat, and improve forest health. Growing awareness of the importance of connecting people to nature, appreciation of and demand for the broad range of ecosystem services that federal forests provide, and the backlog of infrastructure maintenance and improvement projects on national forest lands have also come to inform management priorities. Thus, current economic opportunities for communities linked to federal forest management in the Sierra Nevada are most likely to be in the forest restoration sector, in recreation and tourism, in infrastructure maintenance and improvement projects (facilities, roads, trails), and from the production of timber, biomass, non-timber forest products, and livestock. Payment programs and emerging ecosystem services markets for federal lands (carbon, water quality, fish and wildlife habitat) could potentially accrue to outside organizations that would use these payments to fund needed restoration activities on national forests (Deal et al. 2012). However, these programs are still under development and do not yet constitute a source of jobs for forest community residents.

Contributing to community well-being by providing a broad range of economic opportunities for forest communities is consistent with current Forest Service direction from the U.S. Department of Agriculture (USDA) to generate jobs through recreation and natural resource conservation, restoration, and management in rural areas (USDA 2010), and from the Forest Service 2012 Planning Rule to contribute to social and economic sustainability, thereby supporting vibrant communities and rural job opportunities. The Forest Service is working to increase the pace of restoration on national forest lands and associated job creation (USDAFS 2012), but there are also other strategies that can be used to enhance job creation. What follows is a synthesis of the published literature about how forest managers may help create economic opportunities in local communities through forest restoration, recreation, tourism, and infrastructure maintenance and improvement projects to promote healthy communities and forest ecosystems.

The chapter begins with an overview of how understandings of the relationship between national forest management and forest community well-being have evolved since the mid-1900s. This overview is followed by a discussion of strategies for promoting job creation through forest management that could be adopted by Sierra Nevada national forest managers. It concludes by discussing how these strategies can contribute to the resilience of forest communities and ecosystems. The focus is on rural communities, because the majority of California counties in which Sierra Nevada national forest lands are concentrated are classified by the USDA's Economic Research Service as non-metropolitan.[1] Published literature on the links between forest management and community well-being from the Sierra Nevada is relatively scarce; findings from the wider literature are presented here that can be used to inform forest management in the Sierra Nevada.

Forest-Community Relations

Understanding of the relations between federal forest management and forest community well-being has changed over time. For much of the latter half of the 20th century, timber harvesting on national forests was thought to be an important contributor to economic stability in forest communities. This thinking gave way in the 1990s to the belief that multiple uses and values of national forests contributed to the well-being of forest communities and their capacity to adapt to change. More recent thinking embraces the idea of community resilience as an important component of overall socioecological resilience.

Community Stability

As reflected in the Sustained Yield Forest Management Act of 1944, from the 1940s through the 1980s, the dominant paradigm was one in which federal forest management was thought to be important in contributing to "community stability," defined in terms of stable timber industry employment

[1] For definitions and more information see http://www.ers.usda.gov/Data/RuralUrbanContinuumCodes/.

opportunities and income in forest communities (see papers in Le Master and Beuter 1989). Contributing to community stability through a policy of sustained yield timber harvesting to provide a non-declining, even flow of forest products and associated jobs and income was one goal of national forest management, though the importance of community stability as a management goal waxed and waned between the 1940s and 1980s (Le Master and Beuter 1989).

The notion that federal forest management alone can ensure community stability is flawed for several reasons (Charnley et al. 2008a, Nadeau et al. 2003, Power 2006, Sturtevant and Donoghue 2008). As Power (2006) notes, jobs in the forest products industry are not simply a function of timber supply; demand for wood fiber and wood products plays an important role in influencing harvest and production levels and associated jobs. In addition, changes in harvesting and wood processing technology increase productivity and reduce labor demands, displacing workers. The 1970s and 1980s saw many such changes in the wood products industry. Furthermore, trees harvested in one location do not always get processed in nearby communities. Federal managers must generally sell to the highest bidder, who may not be local. And mills typically obtain logs from a variety of sources, including private forest lands over which federal managers have no control (Power 2006). Finally, a number of variables influence social and economic conditions in forest communities; federal forest management is only one of these variables (Charnley et al. 2008a, Nadeau et al. 2003). For all of these reasons, national forest managers cannot expect to ensure community economic stability through their management actions alone, though timber production on national forest lands continues to make an important contribution to community economies in some parts of the Sierra Nevada.

Community Well-being and Community Capacity

The 1990s saw a dramatic decline in timber production on national forest lands in the Pacific Northwest and in California stemming from concerns about the effects of federal timber harvesting on old growth forest ecosystems, watershed health, and threatened species, such as the northern and California spotted owls (Berck et al. 2003, Charnley 2006). As the Forest Service adopted ecosystem management as its new management paradigm, it grappled with how to create quality jobs in ecosystem management and restoration that would provide new economic opportunities for displaced timber workers and communities affected by this transition in forest management (Spencer 1999). The Jobs in the Woods program, associated with the Northwest Forest Plan and Northwest Economic Adjustment Initiative, was an early attempt to do this. At the same time, amenity migration to communities around national forests was influencing the economic opportunities and social values associated with national forest management. Thus, the 1990s gave rise to new understandings of community-forest relations that acknowledged the diverse contributions federal forests make to “community well-being.” Community well-being studies recognized that (1) well-being in forest communities was based on more than just jobs and income, and included other quality of life attributes, such as health, safety, political participation, social equity, and access to social services; and (2) national forests can contribute to community well-being in multiple ways that include both the commodity (e.g., timber, grazing, minerals, non-timber forest products) and amenity (e.g., outdoor recreation, scenic beauty, clean air and water, open space, forests and mountains) values associated with them (Kusel 2001, Nadeau et al. 2003, Sturtevant & Donoghue 2008).

In the context of these shifts in forest management and rural community dynamics, community capacity—defined as the ability of community residents to respond to internal and external stresses,

create and take advantage of opportunities, and meet the needs of residents (Kusel 2001)—was found to be critical to well-being in forest communities. Community capacity, in turn, is a function of a community's physical, financial, human, cultural, and social capitals (see Kusel 2001 for definitions), or, put another way, of its foundational assets (e.g., physical infrastructure, natural resources, and other attributes of a community) and mobilizing assets (e.g., civic and organizational infrastructure, social processes and interactions) (Donoghue and Sturtevant 2007). Building on these concepts, Beckley et al. (2008) defined community capacity as the collective ability of a community to combine various forms of capital within particular institutional and relational contexts to produce desired results or outcomes.

Community Resilience

In the 2000s, concerns over the impacts of wildland fire and climate change on forests and forest communities have prompted social scientists studying these communities to think in terms of social vulnerability, adaptive capacity, and "community resilience" (e.g., Daniel et al. 2007, Lynn et al. 2011). In general, rural communities in the U.S. tend to be more vulnerable to climate change than urban communities because of their demographic characteristics, available occupations, lower earning rates, greater incidence of poverty, and higher level of dependence on government transfer payments (Lal et al. 2011). In California, people residing in the wildland-urban interface (WUI) are also especially vulnerable to fire (Sugihara et al. 2006). Climate change and fire risk make the concept of community resilience relevant because of its focus on a community's ability to cope with disturbance and change.

The concept of community resilience also applies in the context of socioeconomic stressors and change, as the impacts of reduced federal timber harvesting on forest-dependent communities illustrated in the 1990s. If local or regional economies are based on a single extractive industry, they are more vulnerable to changes in conditions that support that industry—such as market fluctuations, changes in technology, resource depletion, or changes in management policy—making them less resilient than if the economy is diversified (Chapin et al. 2009). “Resilience thinking” at the community level is not well developed, however (Berkes & Ross 2012).

The notion of resilience as applied to social systems has been criticized because of its use in the biological sciences to refer to the ability of a system to respond to stress and shocks in order to maintain function, implying stability and a return to equilibrium following disturbance (Folke 2006). Its applicability to social systems has been questioned because social and ecological systems do not necessarily exhibit the same properties or behave in the same ways (Davidson 2010). More recent thinking about resilience characterizes it as the capacity of socioecological systems to cope with, adapt to, and shape change; to persist and develop in the face of change or disturbance while retaining their basic function and structure; and to innovate and transform into new, more desirable configurations in response to disturbance (Folke 2006, Walker and Salt 2006). A formulation by Magis (2010) defines community resilience as “the existence, development, and engagement of community resources by community members to thrive in an environment characterized by change, uncertainty, unpredictability, and surprise” (Magis 2010: 402). Following Magis (2010), Folke (2006), and Walker and Salt (2006), community resilience is defined here as the ability of a community to successfully cope with, adapt to, and shape change and still retain its basic function and structure. Community capacity influences resilience in that communities having the capacity to recover from, and implement change in response to, stress and disturbance have greater resilience (Berkes and Ross 2012, Folke et al. 2010). It is challenging, however, to identify critical thresholds beyond which social systems will lose their resilience and break down (Davidson 2010). Because resilience within socioecological systems is multi-scalar and inter-connected, community resilience can enhance the overall resilience of a socioecological system operating at other (e.g., landscape) scales (Berkes & Ross 2012).

Job Creation through Forest Management

Given that rural communities in the Sierra Nevada, like rural communities elsewhere, are continually subject to social, economic, and ecological change, their ability to take advantage of job opportunities associated with national forests and their management can help strengthen their resilience. Creating economic opportunities in forest communities promotes and sustains a more diverse employment base there; leaves future opportunities for participating in forest-based livelihoods open; encourages innovation to find new ways of investing in communities; and helps communities adapt to change—all features that contribute to resilience (Walker and Salt 2006). It also maintains a local workforce that has the capacity to carry out forest management work that is needed to improve and restore ecological integrity and resilience in forest ecosystems (Kelly and Bliss 2009).

This section begins by discussing job creation associated with forest restoration. Ecological restoration not only improves the ecological integrity of forest ecosystems; it also has a number of socioeconomic benefits that go beyond its potential for creating jobs in nearby communities (Aronson et al. 2010). From an agency perspective, ecological restoration offers an opportunity to communicate positive messages, values, and activities to the public while addressing ecosystem threats (Egan et al. 2011, Gobster 2005). From a community standpoint, participating in restoration enables people to renew their connections and relationships to the land. Participation in ecological restoration can also be an empowering and positive experience, because participants take personal action to address problems, and discover successful solutions together with managing agencies (Westphal 2003). Further, by engaging in restoration with others, collective identities that form around improving ecosystems and caring for the land can be developed and supported (Clayton and Myers 2009). However, the success of ecological restoration efforts depends in part on the degree of trust that develops between the agencies managing the land, other stakeholder organizations involved, and the public (Winter and Cvetkovich 2010), as well as the perceived benefit and contribution to the community (Marcus et al. 2011).

Make Use of Existing Authorities and Tools

Between 1994 and 2004, there were at least six regional or national legislative and administrative directives that gave the Forest Service authority to consider benefits to local communities when undertaking forest restoration work (Moseley and Toth 2004). These included: (1) the Jobs in the Woods program of the 1990s (applicable in northern California counties affected by the Northwest Forest Plan); (2) the Secure Rural Schools and Community Self- Determination Act of 2000, which made it possible to establish local Resource Advisory Committees that could use Act funding to pay for forest restoration work benefitting federal lands, creating local jobs as a result; (3) the ten-year stewardship contracting authority approved by Congress in the fiscal year 2003 appropriations bill; (4) the National Fire Plan of 2000; (5) the Healthy Forests Restoration Act of 2003 (HFRA) (Moseley and Toth 2004, Steelman and DuMond 2009); and (6) the Tribal Forest Protection Act of 2004. Since 2005, the American Recovery and Reinvestment Act of 2009 (ARRA), and Title IV of the Omnibus Public Land Management Act of 2009 on forest landscape restoration, which established the Collaborative Forest Landscape Restoration Program, can be added to this list. Several of these directives were initiated in response to declines in federal timber harvesting, acknowledging the impacts of these declines on jobs in forest communities, and the shift to forest restoration as a potential new source of local jobs. The strategies for contributing to job creation through forest restoration using the authorities considered below are based on published, peer-reviewed literature.[2]

The National Fire Plan and best-value contracting

Under the National Fire Plan, Congress gave the Forest Service authority to direct fire hazard reduction work to local contractors and businesses, creating an opportunity for them to hire and train local workers (Moseley and Toth 2004). With the shift in agency management focus from timber production to forest restoration, the Forest Service has made less use of timber sale contracts for accomplishing work on the ground, and increased its use of procurement contracts. Procurement contracts are a mechanism for purchasing goods and services from private businesses. The Forest Service can use "best-

[2] Further ideas and guidance on how forest managers may facilitate job creation through forest restoration can be found at http://ewp.uoregon.edu/sites/ewp.uoregon.edu/files/WCF_JobCreation_QG.pdf.

value contracting" criteria—selecting contractors that provide the best value to the government rather than those who offer the lowest bid—as a tool for directing work to local communities by asking contractors how they would create economic opportunities in local communities if awarded a Forest Service procurement contract (Moseley and Toth 2004). The use of National Fire Plan authorities to target local contractors and businesses for jobs in fire management is a strategy that could be used by Sierra Nevada national forests when undertaking fuels reduction and fire suppression work. Doing so would have the added benefit of providing training and work experience that could help communities build their capacity to undertake such work.

Tribal Forest Protection Act

A survey of 31 of the 42 federally recognized tribes in Oregon, Washington, and Idaho found that tribes had a strong interest in taking advantage of jobs in fire management, including working on wildland fire suppression crews and undertaking hazardous fuels reduction work (Rasmussen et al. 2007). Developing projects with tribes using the 2004 Tribal Forest Protection Act authorities is one potential avenue for creating jobs for tribal members in fuels reduction and post-fire rehabilitation activities. The Act allows tribes to propose fire mitigation and environmental restoration activities on national forest lands adjacent to or bordering tribal trust lands in order to protect them from fire, insects, disease, and other threats (USDA FS 2005, cited in Burns et al. 2011, Congressional Research Service 2004). The Forest Service may enter into contracts or agreements with tribes for this purpose. Lands owned and controlled by California Indians in the Sierra Nevada are small and dispersed (Figure 1), creating potential for exploring use of these authorities for collaborative fire management and ecosystem restoration projects. Forest Service Region 5 is encouraging the development of contracts or agreements with tribes to reduce environmental threats in areas of mutual interest.[3]

[3] http://www.fs.fed.us/r5/tribalrelations/tfp.php

Figure 1: California tribal lands and reservations (source = Environmental Protection Agency, http://www.epa.gov/region9/air/maps/ca_tribe.html)

Tribes face several obstacles that limit their capacity to engage in fire management work, however (Rasmussen et al. 2007). These obstacles include the seasonality of the work, training required for employees and contractors, the cost of investing in the equipment necessary for undertaking the work, lack of financial capital with which to start businesses, and supportive tribal leadership to help form partnerships with public agencies (Rasmussen et al. 2007). Differences in communication and operating styles, and Forest Service bureaucratic processes—such as contracting and reporting requirements, timelines, and business plans—can also create barriers (Charnley et al. 2007). To the extent that the Forest Service can assist tribes in addressing some of these obstacles, it can help build the capacity of tribal communities to engage in fire management.

Stewardship contracting

Congress authorized a series of pilot stewardship contracting projects as part of the fiscal year (FY) 1998 appropriations, and gave the Forest Service stewardship contracting authority until FY 2013 in the FY 2003 appropriations bill. Stewardship contracting is a set of authorities that were designed to foster integrated forest restoration and local community benefit (Moseley et al. n.d.). It does the latter in a number of ways: (1) through the "goods for services" authority, which allows the Forest Service to combine the sale of timber and the purchase of services into a single contract, and use the value of timber sold for restoration purposes to pay for services acquired, creating a new source of funding for forest restoration; (2) by requiring the use of best value contracting (most timber sale instruments call for the lowest bid); (3) by allowing the Forest Service to enter into ten-year contracts (as opposed to five years, the limit for traditional service contracts); (4) by allowing the Forest Service to enter into stewardship agreements with nonprofit organizations and other government entities to perform restoration activities; and (5) by calling for collaboration in the development and implementation of stewardship projects. Although stewardship contracting can be a beneficial tool, it may not be appropriate or useful on every national forest (Moseley et al., in review).

The non-peer-reviewed literature that has been generated in association with required governmental reviews (GAO 2008) and monitoring (PIC 2011) of stewardship contracts points to many successes, both environmental and social. Existing peer-reviewed literature concurs that stewardship contracting can be an effective administrative tool for enhancing the social and economic benefits to local communities associated with national forest management (Donoghue et al. 2010, Hausbeck 2007, Kerkvliet 2010). The Eldorado National Forest is one of the national forests that has spearheaded the use of stewardship contracting nationwide (Forest Service stewardship contracting data), and to date, it is one of the top users of stewardship contracts in the National Forest System (Moseley et al., in review). Much can be learned from the Eldorado example by other Sierra Nevada forests interested in using this tool.

Administrative tools that enhance local community benefit

Different administrative tools for accomplishing forest restoration have different implications for local community benefit. Stewardship contracts and best-value contracting have already been discussed. Agreements are useful for targeting specific local recipients that the Forest Service would like to develop working relationships with, direct economic benefits to, and invest in capacity building with because they do not have to be awarded competitively. Charnley et al. (2011) provide a number of examples—both fire- and non-fire-related—in which national forest managers have used agreements to successfully target work to local groups to help build their capacity and provide local workers with job opportunities while accomplishing forest restoration on national forest lands.

Agreements and stewardship contracts are not only useful administrative tools for creating local jobs; they are also mechanisms that can make it more cost-effective for the Forest Service to accomplish mission-related work. Agreements are instruments that require a cost share by the partner, and therefore help leverage external resources to fund project work. Stewardship contracts make it possible to retain receipts from the sale of timber and use any excess income to pay for additional restoration work. Acquisition management staff could be better integrated into project planning activities as a means of helping forest managers determine how to accomplish their work in the most efficient way while enhancing local job opportunities through strategic use of the administrative tools available to them.

Invest Strategically

Target project work to communities in need

One method of increasing local jobs is to geographically target projects to communities in need. Low-capacity communities, high-poverty and unemployment communities, and communities having underserved populations are examples of places where project investments could potentially make a difference in helping communities gain access to increased economic opportunities. The Forest Service used this strategy in implementing ARRA projects. These projects were targeted to counties that had experienced high impacts associated with the economic recession, under the rationale that these were the most important places to create jobs (Charnley et al. 2011). The agency did this by developing economic distress rankings for every county in the U.S., on the basis of four unemployment indicators from the U.S. Census. Counties were ranked on a scale of 1 to 10, with 10 signifying the highest economic distress (Figure 2). Capital improvement and maintenance projects were funded on the basis of the economic distress ranking of the county in which they were located, with the vast majority of projects going to counties that ranked between 7 and 10. Wildland fire management projects were funded on the basis of a different index that weighted county economic distress ranking at 50 percent, insect and disease hazard at 25 percent, and wildfire hazard at 25 percent (Charnley et al. 2011).

Economic distress rankings are one method of targeting project work to create jobs in forest communities that have high need. They are not necessarily the best method; there may be other socioeconomic criteria that are more appropriate for strategically funding projects in communities. Another consideration is the ability of the community to respond and take advantage of job opportunities provided by the agency. Where they lack this capacity, workforce training programs can be one effective means of helping communities build their capacity to engage in forest restoration (Nielsen-Pincus and Moseley 2012).

Forest Service social scientists are currently developing methods for undertaking climate change social vulnerability assessments, which may be useful for helping the Forest Service strategically invest in communities to contribute to their resilience. In the context of fire, social scientists have developed indicators of social vulnerability and adaptive capacity that can also be useful for evaluating how to allocate agency resources to communities (e.g., Ojerio et al. 2011, Paveglia et al. 2009).

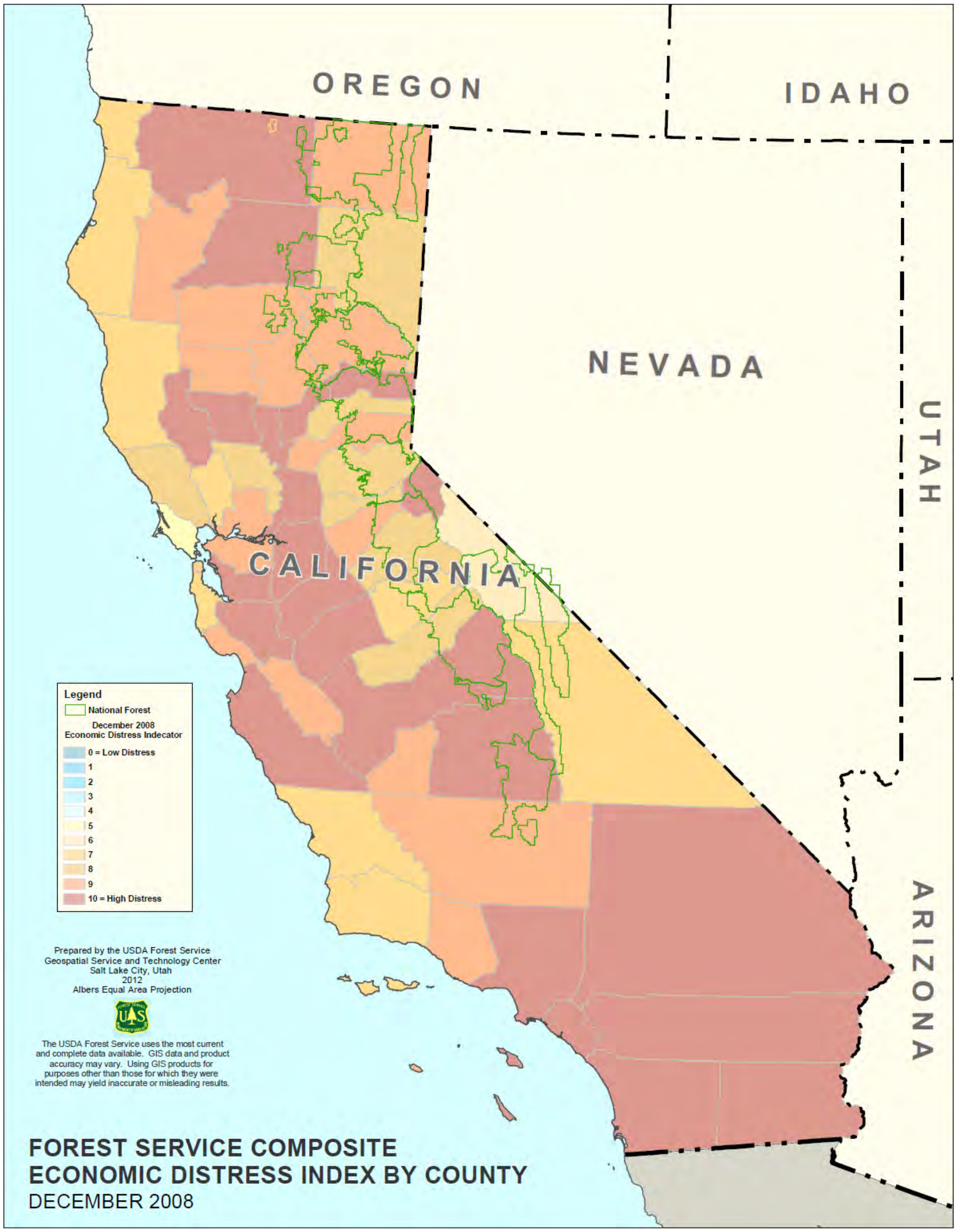

Figure 2: Economic distress rankings of California counties, 2008 (Sierra Nevada national forests are shown in green).

Structure work in ways that are accessible to the local community

Another strategy for promoting local job creation is to structure forest restoration work in a way that is accessible to local communities and can benefit multiple recipients. This strategy entails breaking down work into appropriate sizes and types. One example is road maintenance work. Many national forests consider roadside brush removal as one component of road maintenance, and therefore include it in larger road maintenance contracts. In contrast, the Six Rivers National Forest in northern California separates roadside brush removal from other types of road maintenance work, making it possible for small operators with less diversified equipment to bid on the projects (Charnley 2011).

Another example comes from the Rogue-Siskiyou National Forest in southern Oregon, which received over $30 million in American Recovery and Reinvestment Act funding for hazardous fuels reduction, much of which was labor-intensive because it was located on steep terrain and entailed hand thinning, pruning, piling, and pile burning (Davis and Moseley 2011). In the four-county area that contains Rogue-Siskiyou National Forest lands, there are over 20 local businesses that engage in forestry support work. These businesses range in size and experience, having from just a few employees to roughly 200 employees. There are also several non-governmental organizations that have natural resource crews in the region. To provide job opportunities for this diverse array of local businesses, the Rogue-Siskiyou broke the hazardous fuels reduction work up into 53 contracts and 7 agreements. Contracts ranged in size from $100,000 to $1 million. Agreements were used to target specific recipients that the Rogue-Siskiyou wanted to assist, such as youth job corps programs. The agreements and contracts were sorted into different sets of activities and into work at different scales to enable a number of different businesses to compete for them (Davis and Moseley 2011). Implementing projects in a way that breaks the work into different sizes and types and uses different funding mechanisms spreads the benefits by taking advantage of a range of skills and capacities in local communities. This strategy can be scaled to the availability of funding for project work; it does not rely on a large infusion of funding, as happened in this case under ARRA.

Assess the relative merits of labor- versus equipment-intensive work

The shift from timber production to forest restoration on national forest lands has brought about an associated shift from labor-intensive to equipment-intensive work (Moseley and Reyes 2008). Labor-intensive work has traditionally been associated with intensive timber management in which crews perform tasks such as small tree thinning with chainsaws and tree planting. Restoration work such as road maintenance and decommissioning tends to be accomplished with equipment. Labor-intensive work creates more jobs than equipment-intensive work; however, job quality is typically better with equipment-intensive work, and equipment-intensive work is more likely to go to local contractors due to the cost of hauling equipment long distances (Moseley and Reyes 2008).

Sometimes Forest Service decision makers have choices about whether to accomplish specific management tasks in labor- versus equipment-intensive ways. Despite the general shift mentioned above, there are many opportunities for labor-intensive work associated with forest restoration. In the context of wildland fire management, restorative understory burning is typically accomplished by fire suppression crews who are employed seasonally by the Forest Service in the spring or fall, when not fighting fires (Moseley and Toth 2004). Mechanical fuels treatments can be accomplished by hand crews or with equipment. Labor-intensive work is more common when fuels reduction occurs on steep slopes, entails thinning of small-diameter trees with no commercial tree removal, or involves tree planting in rehabilitation efforts (Moseley and Toth 2004). Labor-intensive work is also common in habitat improvement and watershed restoration projects (Nielsen-Pincus and Moseley 2012). Brush removal along forest roads can also be accomplished either mechanically or by hand (Charnley 2011). In these cases, decision makers can choose to accomplish work in a manner that creates more jobs, assuming that doing so is cost effective and meets their management objectives.

When deciding how to accomplish restoration work, it is important for decision makers to be aware of the relative merits and drawbacks associated with labor- versus equipment-intensive work. Labor-intensive work creates more jobs than equipment-intensive work, which is important in forest communities having high unemployment. It also creates opportunities for workers who would not otherwise have access to jobs on national forests because they lack the financial capital to invest in equipment. During the economic recession of 2007-2009, one way that the Forest Service used ARRA funds to create jobs in communities experiencing economic distress was by choosing to carry out work in labor-intensive ways (Charnley 2011). However, researchers have found that labor-intensive jobs in the forestry services sector often go to distant workers, are relatively low paying and create less total local economic impact than other jobs, may entail poor working conditions and worker abuse, can be dangerous, and are seasonal (Moseley 2006, Moseley and Reyes 2007, Nielsen-Pincus and Moseley 2012, Sarathy 2012). In contrast, equipment-intensive work tends to be better paid and is more often carried out by local businesses (Moseley and Reyes 2008), though it, too, is typically seasonal.

Forest decision makers who are aware of these patterns can make an effort to overcome them by targeting local workers, ensuring that contracting and labor laws are enforced so that workers are paid the required wages, and promoting fair and safe working conditions. Another strategy is the direct hire of workers using Forest Service 1039 employment authorities, especially in places where there are few forestry support businesses (Jakes 2011). Even when jobs—be they labor- or equipment-intensive—are short-term or seasonal in nature, they can have many benefits beyond short-term job creation. These include providing employees with training, skills, and experience for future jobs; improving employee access to the federal job network; improving employee physical and mental health; building teamwork and safety skills; and building awareness of nature, national forests, and resource management issues among local residents (Charnley et al. 2012).

Implement Projects that Build on Local Community Capacities and Priorities

Collaboratively design projects

A number of researchers have found that when the Forest Service works collaboratively with local communities to develop forest restoration projects that build on local community infrastructure, resources, values, culture, and collaborative relationships and address local needs and priorities, it can be especially effective in creating local community benefits and contributing to community resilience (Abrams 2011, Burns et al. 2011, Charnley et al. 2012, Hardigg 2011). It is not always easy to collaborate, given declines in agency staffing and resources, and there can be challenges in the process. Nevertheless, when opportunities exist to develop projects collaboratively and align them with community needs and capacity, they are more likely to create local community benefits.

Hire local-level decision makers who make local community benefit a priority

Individual decision makers at the ground level are responsible for implementing policies handed down from above, and exercise discretion in doing so (DeLeon and DeLeon 2002, May and Winter 2007, Vinzant and Crothers 1998). They make decisions about whether and how to implement policies based on direction from above, as well as their own interpretations, values, experience, and local circumstances. These findings imply that if contributing to forest community well-being and resilience is a priority for the management of Sierra Nevada national forests, then hiring forest- and district-level decision makers who make this priority a goal in project development and implementation can make a difference. Decision makers who have a thorough knowledge of local social and economic conditions will also be better positioned to make decisions that draw on existing capacity in a community, help build local capacities that need to be developed, and direct resources accordingly. As Charnley et al. (2012) found in the case of ARRA projects, Forest Service employees on the ground developed a number of strategies for increasing the socioeconomic benefits of projects to local communities, innovating and exhibiting leadership in the process. Individual employees make a difference, and those who are committed to enhancing community well-being and resilience can make choices to implement project work in ways that are likely to do so.

Invest in Recreation Infrastructure, Opportunities, and Partnerships

Recreation and tourism opportunities

Amenity migration, or "counterurbanization," has been high in the foothills of the central Sierra Nevada (Duane 1999), and in the high altitude regions (over 1800 m) of the Lake Tahoe basin (Loeffler and

Steinicke 2006, 2007; Raumann and Cablk 2008) and Mammoth Lakes (Loeffler and Steinicke 2006, 2007), in particular. Amenity migration has led to dramatic transformations in rural communities in many places where it has occurred as traditional land uses, economic activities, and social relations transition from those associated with the extractive industries to those associated with amenity values (for a review, see Gosnell and Abrams 2011). Some social scientists have argued that natural amenity values can be drivers of economic development in rural communities near federal lands, because rural communities having desirable physical and social environments attract tourists, new residents, and new businesses, which increases the financial and human capital of communities and create jobs, thereby stimulating local economic development (Charnley et al. 2008b). As a result, “jobs follow people” (Goodstein 1999, Nelson 1999, Vias 1999). National forests are important in this regard because of the natural amenities they provide, including recreation opportunities, scenic beauty, open space, clean air and water, and desirable environmental features, such as mountains, water bodies, and forests.

Recreation and tourism have brought new economic opportunities to many communities that were formerly timber dependent (Charnley et al. 2008a, 2008b). In places experiencing high levels of recreation and tourism, local economies may be extremely dependent on these activities. For example, an estimated 38 percent of all jobs in Mammoth Lakes and the Lake Tahoe Basin are directly tied to tourism, and 74 percent of all jobs, and 68 percent of all wage payments, are indirectly tied to tourism (Loeffler and Steinicke 2006). Forest Service managers may contribute to recreation and tourism-related development opportunities in forest communities through job creation associated with road, trail, and facilities maintenance and improvement projects. Trails and facilities projects in particular may present opportunities for hiring youth through job corps programs such as the California Conservation Corps. Working on such projects provides youth an opportunity to spend time in the woods, build job skills, learn about and connect with the Forest Service, and prepare for future jobs (Charnley 2011). Managers may also contribute to community development by maintaining and developing recreation opportunities and infrastructure on national forest lands and in local communities that attract visitors, who in turn spend money locally, supporting local businesses (e.g., Burns et al. 2011, Sturtevant et al. 2011). This assumes that local businesses exist or will develop to

take advantage of economic opportunities associated with Forest Service investments in recreation and tourism. If they do not, additional assistance may be needed so that communities can capture these benefits.

Although amenity migration can contribute to local economies, it also has drawbacks. Jobs created in association with amenity migration, recreation, and tourism are often in the services sector (English et al. 2000, Shumway & Otterstrom 2001). Although some services jobs pay well (Holmes and Hecox 2004), many jobs associated with natural amenities, recreation, and tourism are seasonal and low wage (McKean et al. 2005). Even if people living in high-growth amenity and recreation counties have higher incomes, these may be offset by higher costs of living (English et al. 2000, Hunter et al. 2005). In the Lake Tahoe Basin and Mammoth Lakes regions of the Sierra Nevada, for example, the influx of affluent amenity migrants has driven up the price of housing, making most homes beyond the reach of people employed locally (Loeffler and Steinicke 2006, 2007). Furthermore, new residential development in rural areas of the West brings with it substantial costs associated with providing community services and social infrastructure (e.g., roads, sewage treatment, schools, fire protection) (Gosnell and Abrams 2011). And new residents bring with them different sets of values that may clash with those of long-term residents, making collaboration associated with natural resource management more challenging (Walker and Hurley 2004). Nevertheless, recreation and tourism are an important component of many rural economies in the Sierra Nevada (Duane 1999, Stewart 1996). Supporting them is one way of helping to diversify the community economic benefits associated with national forest management.

Recreation partnerships

The Forest Service is increasingly accomplishing recreation management through partnerships that build relations with local groups and leverage the resources needed to maintain recreation opportunities and facilities in the face of declining agency budgets (Seekamp and Cerveny 2010). Seekamp et al. (2011) identified 35 common types of recreation partners with which the Forest Service works. Although volunteerism is common, many partners have a financial relationship with the Forest Service, providing the agency with revenue for projects or, conversely, making a living from federal lands; these partners include outfitters, guides, concessionaires, and contractors. Recreation partnerships contribute to both forest community and forest ecosystem health. On the community side, they provide jobs, job skills, organizational capacity building, and stronger collaborative relations; on the forest side, they support stewardship and conservation activities, and help build a conservation ethic among members of the public (Seekamp and Cerveny 2010, Seekamp et al. 2011). Thus, investing in recreation partnerships is another strategy for creating economic opportunities in the Sierra Nevada.

Management Implications: Strategies for Improving Job Creation through Forest Management

Make use of existing authorities and tools

- Use National Fire Plan authority to direct fuels management work to local contractors and businesses using best-value contracting; ask contractors how they would create economic opportunities in local communities if awarded a Forest Service procurement contract
- Use 2004 Tribal Forest Protection Act authorities to collaboratively develop fire mitigation and environmental restoration projects with tribes, and to enter into contracts or agreements with tribes to reduce environmental threats on national forests bordering Indian trust lands in areas of mutual interest
- Increase use of stewardship contracts and stewardship agreements
- Make use of agreements (which can be awarded non-competitively) to target work to specific local recipients in order to develop working relationships with them, provide local workers with jobs, and build their capacity to accomplish work on national forests
- Integrate acquisition management staff into project planning activities to help determine how work can be accomplished in a way that enhances local economic opportunities through strategic use of the available administrative tools

Invest strategically

- Geographically target project work on national forest lands near communities in need, where they can make a difference in contributing to local economies through job creation
- Implement projects in a way that breaks the work into different sizes and types, and uses different funding mechanisms, to spread the benefits by taking advantage of the range of skills and capacities present among local businesses, NGOs, and other workers
- Assess the costs and benefits of accomplishing project work in a labor-intensive versus an equipment-intensive manner
- Promote fair and safe working conditions for forest workers by ensuring that labor and safety laws are enforced

Implement projects that build on local community capacities and priorities • Work collaboratively with local communities to develop projects that build on local community infrastructure, resources, values, culture, and collaborative relationships, and address local needs and priorities • Hire local-level decision makers who are committed to creating local community benefits through forest management work by making choices to develop and implement projects in ways that do so
Invest in recreation infrastructure, opportunities, and partnerships • Maintain and develop recreation opportunities and infrastructure on national forests and in local communities to create jobs and attract visitors who support local businesses • Invest in recreation partnerships with guides, concessionaires, and contractors

Conclusions

This chapter has examined ways that managers can facilitate job creation associated with national forest management in forest communities to contribute to community well-being. Its goal is to encourage managers to consider how to create local community benefits with other project objectives when planning and carrying out forest management work. Doing so can enhance socioecological resilience. Indicators of resilience include social and economic diversity, new business and employment opportunities, community infrastructure, innovation, connections between people and places, and keeping options open for the future (Chapin et al. 2009, Berkes and Ross 2012, Magis 2010, Walker and Salt 2006). It follows that developing diverse economic opportunities from national forests (including jobs associated with the production of forest products, see chapter 9.5) increases resilience.

Developing and implementing forest management work in a manner that promotes local community benefits may sometimes require making tradeoffs between promoting socioeconomic goals and meeting other agency objectives and requirements (Charnley et al. 2012). Nevertheless, the long-term benefits of investing in local communities, helping them build their resilience, and increasing their capacity to engage in forest management work may outweigh the short-term tradeoffs associated with making community considerations of secondary importance in accomplishing projects. This is because job creation through forest management also benefits national forests. For example, jobs in forest restoration help maintain the local workforce and business capacity needed to perform restoration work on federal, private, and tribal forest lands, making it more feasible to achieve landscape-scale forest restoration goals across ownerships (Charnley et al. 2011). Recreation projects that improve trail design and construction, replace ineffective waste facilities, and provide developed access to lakes and streams help reduce the natural resource impacts of forest recreation by reducing erosion, protecting water quality, and contributing to the control of invasive species. Recreation projects that enhance the visitor experience can also help build public support for national forests and foster values associated with forest stewardship among visitors (Charnley et al. 2011). Thus, doing more to prioritize the social and economic benefits associated with forest management work in the future can ultimately be good for rural communities and national forests.

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9.5 Managing Forest Products for Community Benefit

Susan Charnley with a contribution from Jonathan Long

Contents

Executive Summary

Forest products from national forest lands remain important to local communities in some parts of the Sierra Nevada synthesis area. Managing national forests for the sustainable production of timber, biomass, non-timber forest products, and forage for livestock will help sustain forest-based livelihoods in parts of the region where they are socially and economically important, thereby contributing to social and economic sustainability and supporting community resilience. This chapter provides context for understanding the social and economic dimensions of timber production, biomass utilization, non-timber forest product harvesting, and grazing on Sierra Nevada national forests and associated management issues. The chapter also points out ways in which managing forest products for community benefit can also benefit forest and rangeland ecosystems. At the end of each section, there is a "Management Implications" box that summarizes strategies forest managers could pursue to help maintain California's wood products industry, increase biomass utilization, and support non-timber forest product harvesting and grazing on Sierra Nevada national forests.

Introduction

This chapter examines timber production, biomass removal, non-timber forest product (NTFP) harvesting, and grazing on national forest lands, and synthesizes the scientific literature that addresses how these activities can be supported on Sierra Nevada national forests in order to help sustain the livelihoods of community residents who participate in them. Mining is not addressed because it is no longer considered to be a significant economic activity in the Sierra Nevada (Duane 1999, Stewart 1999), and because of a lack of recently published literature about mining in Sierra Nevada communities. Recreation and tourism are addressed in Chapter 9.1 (Broader Context for Social, Economic, and Cultural Components).

Traditional forms of commodity production (e.g., timber production, grazing, and mining) from national forests in the Sierra Nevada are no longer as prominent as they were in the past (SNEP 1996, Duane 1999). Nevertheless, timber production and grazing remain locally important. Stewart (1996) found that recreation, timber, and agriculture were the employment sectors most dependent on Sierra Nevada ecosystems, and that the natural resources from these ecosystems generating the highest revenues were water, timber, livestock, and other agricultural products (in that order).

The Sierra Nevada Ecosystem Project identified six distinct social and economic subregions in the Sierra Nevada (Doak and Kusel 1996, Stewart 1996). An analysis by Duane (1999) also identified six distinct subregions of the Sierra Nevada based on social criteria. Although the subregional boundaries differ slightly, their overall characterizations are consistent (Duane 1999). Timber production is most prevalent in the northern Sierra Nevada counties; grazing is found mainly in the eastern Sierra Nevada and in the oak woodland ecosystems of the western Sierra Nevada; agriculture occurs largely on the west side, in the central and southern portions of the synthesis area; and recreation and tourism dominate the economies of the greater Lake Tahoe Basin and the eastern side of the Sierra Nevada. Nevertheless, many communities and counties in the Sierra Nevada subregions have mixed economies, as characterized by Doak and Kusel (1996) and Duane (1999). Some are still more natural resource dependent (timber, grazing); some have economies based largely on natural amenity values; and some are close to large urban areas that provide diverse economic opportunities. In addition, many counties contain communities that are highly variable in terms of socioeconomic well-being (Doak and Kusel 1996). Thus, the relevance of the forest products management strategies discussed in this chapter will vary by place across the region, depending upon the nature of forest-community relations in particular locations.

The production of timber, biomass, NTFPs, and forage from national forest lands is not just about management to contribute to community socioeconomic well-being. Current national forest management policy calls for approaches that accomplish ecological restoration goals and produce forest products to benefit local communities and economies (USDA 2010, USFS 2007). Such approaches can contribute to socioecological well-being and resilience in a number of ways: (1) by supporting community residents who maintain forest-based livelihoods in rural areas where alternative job opportunities are limited; (2) by helping to produce goods valued by society; (3) by maintaining the workforce and physical infrastructure needed to accomplish forest restoration on federal lands; and (4) by helping to conserve the biodiversity and ecosystem integrity of working forests and rangelands on the private and tribal lands that are ecologically and socioeconomically interdependent with federal lands (Charnley et al., in press).

This chapter focuses first on timber production and the wood products industry. It then moves on to address biomass removal and utilization, NTFPs, and grazing. The chapter concludes by summarizing key findings.

Timber Production and the Wood Products Industry

Trends in Harvesting, Employment, and the Industry

A detailed account of conditions and trends in California's wood products industry can be found in Morgan et al. (2004, 2012), upon which the following discussion is based. California has been among the top softwood lumber-producing states in the U.S. since the 1940s. The wood products industry in California is influenced by a number of variables, including national and international economic conditions, markets, technology, public policy and regulations, and available timber inventories. National forests have been an important source of timber for California's wood products industry since the 1960s. Although a severe recession and weak markets caused a drop in timber production and related employment in the early 1980s, this dip was followed by a recovery that lasted through the end of the 1980s. Since the early 1990s, the availability of timber—particularly from federal lands—has been a major factor influencing California's wood products industry. Timber harvests from national forests declined because of policy and legal constraints on harvesting related to the protection of old growth forests and threatened and endangered species, restrictions on harvesting in unroaded areas, and timber sale appeals and litigation. At the same time, state regulations caused timber harvests from state and private lands to decrease. In the 2000s, timber harvest on California national forests has been driven more by restoration goals (e.g., hazardous fuels reduction) than by timber production goals (Christensen et al. 2008). An economic recession in the early 2000s, declines in housing construction since 2006, and increased imports of lumber from Canada following expiration of the Canadian softwood lumber agreement in 2001 have caused the price of wood products to be low for much of the 2000s. Market conditions combined with other factors, such as increasing fuel prices and reduced timber availability, caused a further decline in California's wood products industry during the first decade of the 2000s (Morgan et al. 2012).

Trends in California's timber harvests are reflected in Figure 9.5.1. The total volume of timber harvested in California in 1988 was 4.84 billion board feet, and in 2010, it was 1.29 billion board feet—73 percent below what it was in 1988 and 74 percent below what it was in 1972. The volume of timber harvested from Sierra Nevada national forests was 1.29 billion board feet in 1988, and 183.8 million board feet in 2010, 86 percent lower than it was in 1988. As Figure 9.5.1 indicates, the decline in total timber harvest in California since 1990 has largely been due to reductions in timber production on national forest lands, though harvests from private lands also dropped for reasons explained above. In response to these trends, California mills have become increasingly reliant on out-of-state and Canadian sources of timber to meet their supply needs (Morgan et al. 2012). Imports have constituted an estimated six percent of the annual volume of timber processed in California in recent years (Morgan et al. 2012).

The number of primary wood processing facilities in California has also been declining, a trend ongoing since 1968 (Table 9.5.1). Reduced timber availability was the primary driver of sawmill closures between 1988 and 2006 (Morgan et al. 2012). Other factors contributing to sawmill closures over time have been technological advances leading to increased processing efficiency, market conditions, and the shift to harvesting smaller logs. Between the late 1980s and 2000, California milling capacity dropped by almost

60 percent; since 2000, it has continued to drop as mills have closed (Christensen et al. 2008, Morgan et al. 2004). As a result, California's capacity to process sawtimber went from 6 billion board feet in 1988 to below 1.8 billion board feet by 2009 (Morgan et al. 2012). In 2006, there remained 12 sawmills, 2 medium-density fiberboard and particleboard mills, and no veneer mills in counties within the Sierra Nevada synthesis area (Morgan et al. 2012). Figure 9.5.2 shows the distribution of mills of all types in California as of 2006.

Declining mill capacity has important implications for the ability of federal and private forest owners to produce timber. Mills provide a market for timber; fewer mills mean less competition and lower stumpage prices; and the further the haul distance from the harvest site to the processing facility, the higher the transportation costs and less economical the timber sale. Greater haul distances also mean an increase in fossil fuel consumption, increasing carbon emissions. Maintaining the remaining wood processing infrastructure in the Sierra Nevada synthesis area is an important strategy for supporting continued timber production from national forests to help accomplish ecological restoration goals and maintain jobs in the wood products industry.

Employment in California's forest products industries has fluctuated over time, and declined 33 percent between 1989 and 2010, from 112,500 jobs in 1989 to 75,100 jobs in 2010 (Figure 9.5.3). These trends have largely been due to fluctuations in the lumber and wood products sector, rather than in the paper and allied products sector. The decline in California's wood and paper products industry employment since 1989 can be attributed mainly to reduced timber harvest and availability, as well as increased mill efficiency and the recent economic downturn and housing decline (Morgan et al. 2012). The effects of declining forest products industry employment have been greatest in northern California counties, where the forest products industry is concentrated, including the northern Sierra Nevada counties of Lassen, Modoc, Plumas, and Sierra (Morgan et al. 2004).

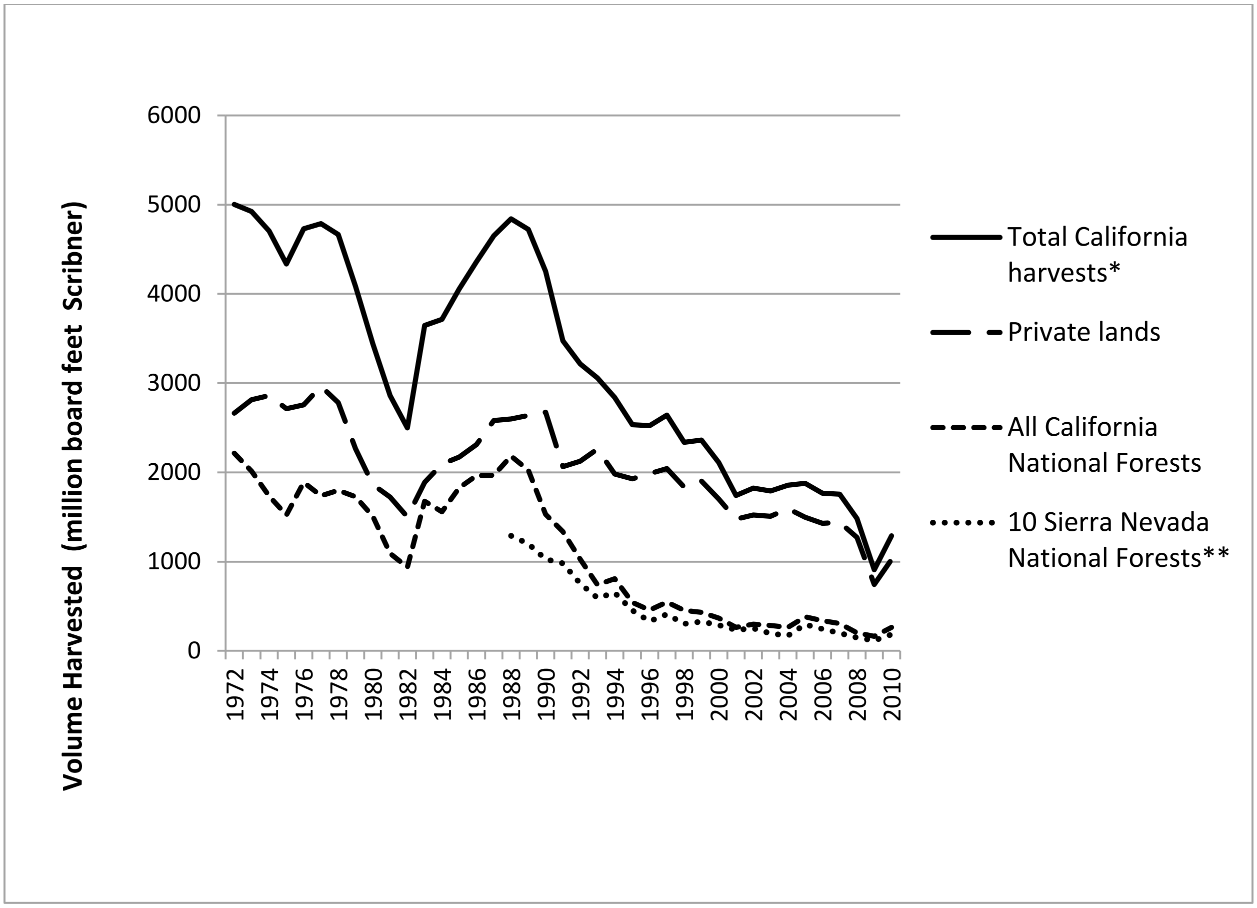

Figure 9.5.1. Volume of timber harvested from all lands, private forestlands, national forest lands, and 10 Sierra Nevada national forest units in California, 1972-2010. Source: Ruderman 1984, Warren 1989-2011

* Harvest data from state lands were missing for 2003-2010, and data from Bureau of Indian Affairs lands were missing for 2001-2010; they are not included in the totals for those years. Harvest data for Bureau of Land Management lands were <1 million board feet for 2001, 2003, and 2004, and are not included for those years.

** Modoc, Lassen, Plumas, Tahoe, Lake Tahoe Basin Management Unit, Eldorado, Stanislaus, Sierra, Inyo, and Sequoia. Data for Sierra Nevada forest harvests were unavailable prior to 1988 from the Warren and Ruderman reports.

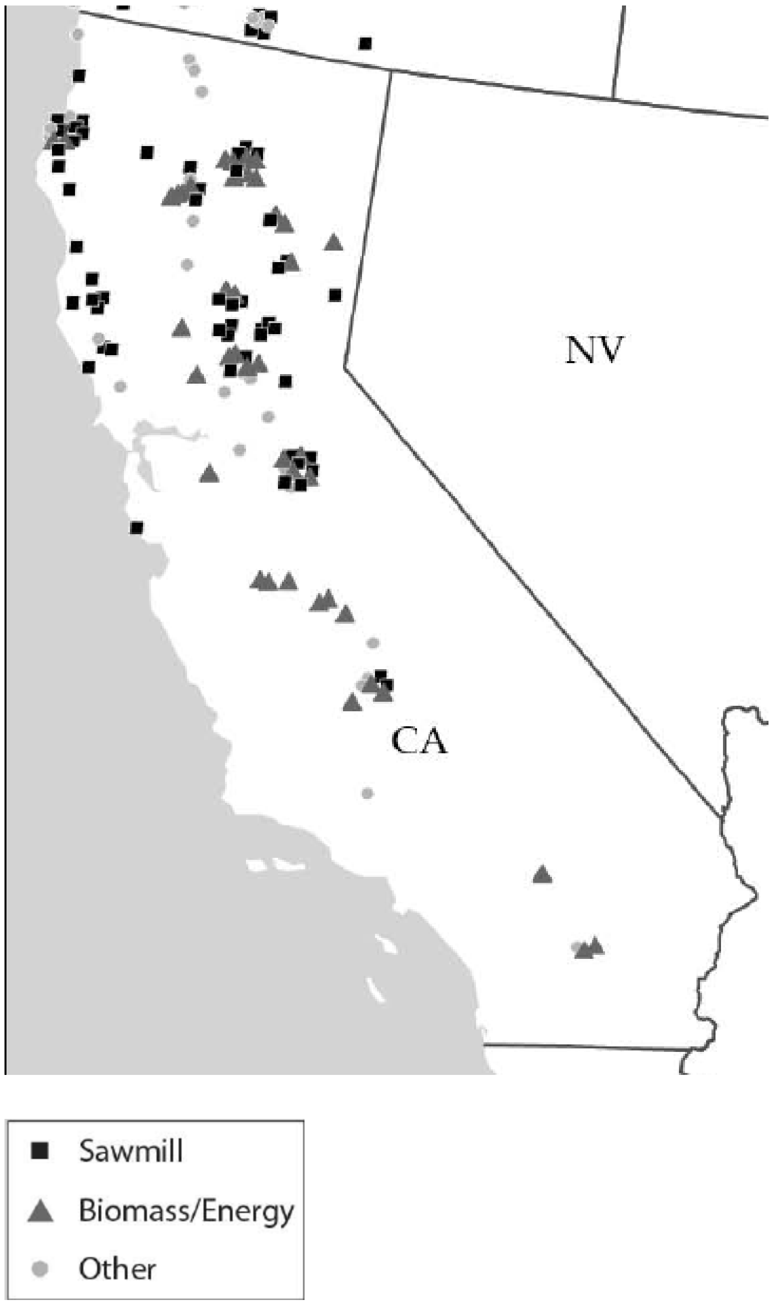

Figure 9.5.2. Mills in California, 2006. Data Source: University of Montana, Bureau of Business and Economic Research. Map Credits: Jean Daniels, Darin Jensen.

Table 9.5.1. Number of sawmills, veneer and plywood Mills, and pulp and board mills in California, 1968-2006.

Year	1968	1976	1985	1994	2006
Sawmills	216	142	89	53	33
Veneer & plywood mills	26	21	6	4	2
Pulp and board mills	17	7	11	12	4

Source: Morgan et al. 2012

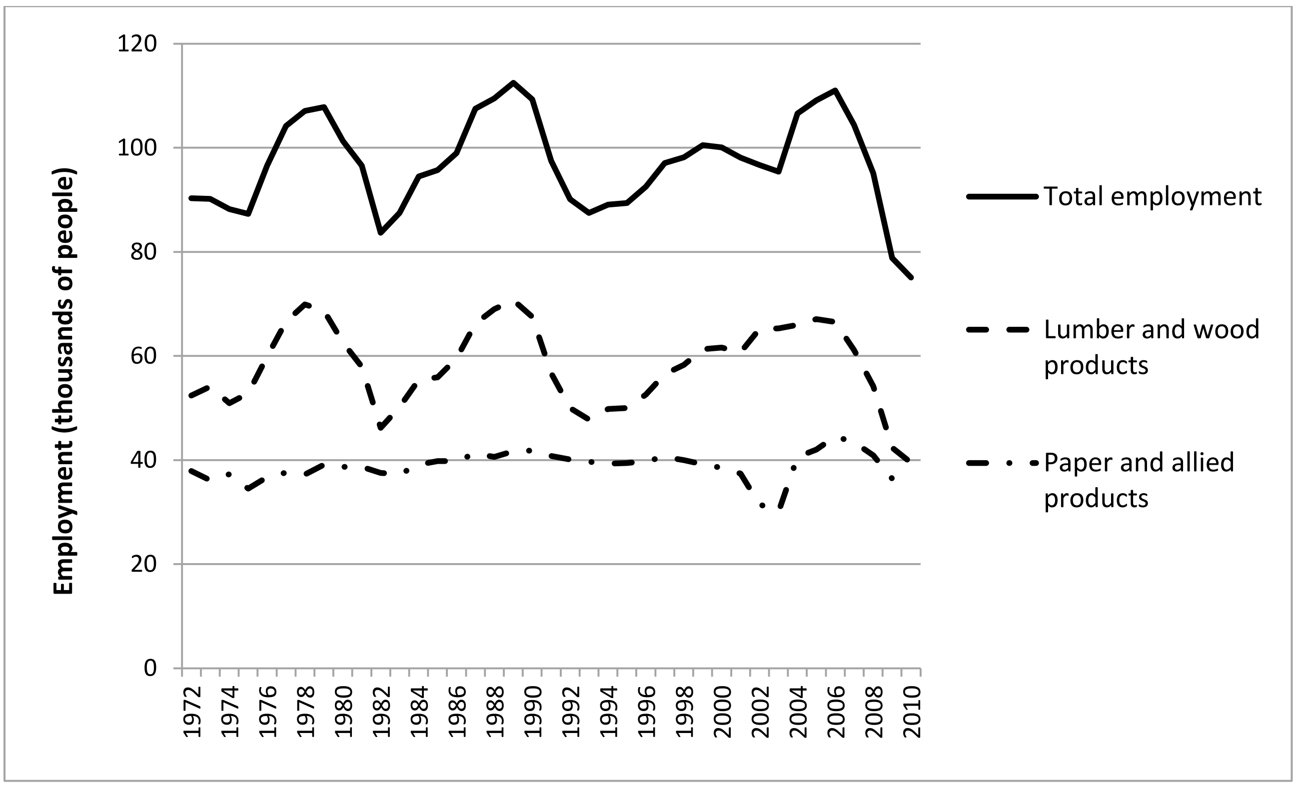

Figure 9.5.3. Employment in the forest products industries in California, 1972-2010. Source: Ruderman 1984; Warren 1992, 2002, 2011.

Wildland fire can also have a substantial economic impact on the timber industry and on wood products markets because it reduces the standing inventory of timber. Although socially controversial because of environmental concerns, salvage logging post-fire has a number of economic benefits for producers of damaged timber and consumers (Prestemon and Holmes 2004, Prestemon et al. 2006), and is socially acceptable among many residents of fire-prone and fire-affected communities (McCaffrey 2008; Ryan and Hamin 2008, 2009) (see Chapter 4.3, Post-Fire Management). Research from the Sierra Nevada community of Arnold, California, near the Stanislaus National Forest, found a strong level of support for post-fire restoration and rehabilitation activities, including salvage logging, on Forest Service lands among community members economically dependent on natural resources (Ryan and Hamin 2008, 2009). Reasons included the ability of salvage logging to provide local jobs, a supply of material for local industry, and income to fund post-fire restoration activities, another potential source of local jobs. Research on post-fire restoration and rehabilitation, and on salvage logging in particular, finds that salvage logging is likely to be more socially acceptable if it is done in ways that are appropriate to, and do not harm, the local ecology; if scientific research supports the approach used; and if the income from salvage logging is invested in local post-fire restoration or wildfire prevention activities around communities (Ryan and Hamin 2009). Extensive and consistent communication and outreach by the Forest Service during the process are also important (Ryan and Hamin 2008). In addition, planning and making decisions about how to approach salvage harvesting in advance of a wildfire in the context of overall forest restoration objectives at the landscape scale may help reduce debate about salvage operations following a fire (McCool et al. 2006).

Impacts of Reduced Federal Timber Harvesting on Communities

A number of social scientists have studied the impacts of reduced federal timber harvesting on forest communities in the Pacific Northwest, and how communities have been adapting to this change (e.g., Carroll 1995, Charnley et al. 2008a, Helvoigt et al. 2003, Kusel et al. 2000). Research on the impacts of reduced federal timber harvesting on Sierra Nevada communities is much less prevalent, and existing research has focused primarily on Plumas County, where the Quincy Library Group emerged. The community of Quincy is one of many places in California and the Pacific Northwest where the “timber wars” of the 1980s were fought, and there exist many published versions of this story (e.g., Bernard 2010, Bryan and Wondolleck 2003, Colburn 2002, Marston 2001) because it led to one of the first community-based, collaborative conservation initiatives associated with forestry in the western U.S. (see Chapter 9.6, Collaboration). As in many timber dependent communities and counties, decreases in timber harvests on the Plumas National Forest (which occupies roughly 75 percent of Plumas County [Bernard 2010]) led to the loss of logging jobs and mill closures with associated job losses in Quincy, home foreclosures, reduced payments in lieu of taxes to county governments to fund schools and roads, and declines in Forest Service budgets and staffing, making it harder to prepare timber sales and carry out treatments to reduce fire hazard and improve forest health (Bernard 2010, Colburn 2002). Changes in forest management policy that took place in the early 1990s have had different effects in different communities, depending on local characteristics and relations to national forest lands (Charnley et al. 2008a).

Quincy and Plumas County, like many communities and counties negatively impacted by the shift away from intensive timber production on national forests, have evolved over the past two decades. Jobs in agriculture and forestry are still important, though there are many fewer jobs associated with timber production alone, and there is a greater proportion of jobs in forest restoration. Recreation and tourism, long important in the area, have expanded to include golfing, wind surfing, high-end resort development, and shopping (Bernard 2010, Colburn 2002). New residents drawn by the county's natural amenity values have settled or bought second homes there, though the associated rise in real estate values has made it difficult for other residents to afford homes. Investment in watershed restoration and improvements in the Feather River watershed, an important source of water for California, has created jobs and had significant conservation outcomes; water from the Feather River watershed could be a source of greater local economic opportunity in the future. However, the economic recession that began in 2007 led to closure of Quincy's last sawmill in 2009, and a slump in real estate development (Bernard 2010).

Elsewhere in California and the Pacific Northwest, decreases in federal timber production had similar effects on forest communities. They have responded in a number of ways. Community capacity lost when workers who lost jobs in the forest products industry moved away has been gradually rebuilt in some communities where new residents drawn by recreation, natural amenities, and relatively low costs of living have moved in (Charnley et al. 2008a). Economic diversification has also occurred. Forest community residents have taken advantage of economic opportunities associated with recreation and tourism, agriculture, non-timber forest products, public and tribal administration, forest restoration, small-diameter wood manufacturing, and being located along major transportation corridors or close to regional centers (Charnley et al. 2008a). In California, some forest communities have turned to marijuana growing as an economic diversification strategy in response to declines in wood products industry employment, although this trend is much more prevalent in California's north coast range than in the Sierra Nevada (Leeper 1990). The emergence of community-based collaborative groups in forest communities in California—such as the Quincy Library Group—has been an important mechanism for innovation in seeking ways to link communities and forests to promote economic and ecological health associated with forest management (Donoghue and Sturtevant 2008). This topic is discussed further in Chapter 9.6 (Collaboration).

Tools

Some tools have been developed that can help assess how wood products obtained from national forests translate into jobs and income that benefit regional economies. These tools are useful for forest planning as well as monitoring. The Bureau of Economic Analysis has developed a Regional Input-Output Modeling System (RIMS II) that can help planners assess the regional economic impacts of planned projects by producing multipliers that estimate the total economic impact a project will have on a region. More about the model can be found at https://www.bea.gov/regional/pdf/rims/RIMSII_User_Guide.pdf. Regional Economics Models, Inc. (REMI) has developed another model called Policy Insight (PI+) that generates annual estimates of the total regional economic and demographic effects of policy initiatives, which can be used for forecasting.[1] Perhaps the most useful tool for forest managers is IMPLAN, developed by MIG, Inc.[2] IMPLAN can be used to model the economic impacts of management activities down to the zip code level. These models are not limited to assessing the economic effects of forest plans and proposed projects on the wood products industry; they also have application for assessing how the production of other forest products and recreation activities on national forests translate into jobs and income that benefit regional economies.

Future Prospects and Management Implications

Demand for wood products in California has been increasing and is predicted to continue to do so as a result of population growth (Christensen et al. 2008). The majority of wood products produced in the state are consumed there (Morgan et al. 2004, 2012). High demand for wood products, productive forests, and high quality timber in California mean that the wood products industry has the potential to remain viable there (Christensen et al. 2008). Maintaining the industry is important from the standpoint of both national forest management and jobs in forest communities. A 2002/2003 survey of California's primary wood products industry leaders asked them what issues they thought would affect the performance of their operations in the coming five years, in order of importance (Morgan et al. 2004). Energy costs, California regulations, and timber availability from private lands were at the top of the list. Timber availability from federal lands ranked number 10 on the list; most respondents no longer considered federal lands a reliable source of timber, basing their operations instead on timber harvested from private lands. Nevertheless, federal timber supplies were critical to the operations of some respondents, and for the future viability of their firms (Morgan et al. 2004).

These findings point to several strategies managers could pursue to support jobs in the wood products industry and keep mills operating to maintain wood products industry infrastructure so that forest owners (including the Forest Service) can accomplish forest restoration and hazardous fuels reduction. One is to provide a stable and predictable supply of wood from national forest lands. Another is to offer financial assistance to mills struggling to stay operational to help them invest in measures that improve their efficiency and competitiveness. The Forest Service used American Recovery and Reinvestment Act funds to do this during the economic recession of 2007-2009, with positive results (Sturtevant et al. 2011). A third strategy is to plan timber sales that are scaled in size to the capacity of local community operators, so that they can bid on them. The inability of small, local logging businesses to bid on big Forest Service timber sales when clear-cutting was a common practice was one source of controversy in Quincy that brought loggers to the table to search for alternative forest management approaches (Colburn 2002). Finally, post-fire rehabilitation and restoration activities, particularly salvage logging, can help reduce economic losses to the wood products industry following a wildfire. Planning for salvage operations in advance of a fire, using science to inform salvage operations to minimize environmental risk, investing revenue generated from salvage sales in post-fire restoration and fire risk reduction activities, and good communication can help salvage logging move forward so that its economic benefits are realized.

[1] For more information, go to http://www.remi.com/products/pi

[2] http://implan.com/V4/index.php?option=com_content&view=frontpage&Itemid=70

Management Implications: Maintaining California's Wood Products Industry

- Provide a stable and predictable supply of wood from national forest lands
- Offer financial assistance to mills struggling to stay operational to help them invest in measures that improve their efficiency and competitiveness
- Plan timber sales that are scaled in size to the capacity of local community operators so that they can bid on them
- When salvage logging is part of post-fire recovery plans, several strategies can make it more socially acceptable: planning for salvage operations in advance of a fire in the context of broader landscape-scale restoration; using science to inform salvage operations to minimize environmental risk; investing revenue generated from salvage sales in post-fire restoration and fire risk reduction activities; and good communication

Biomass Utilization

The U.S. Forest Service defines woody biomass as trees and woody plants—including limbs, tops, needles, leaves, and other woody parts—that grow in forests, woodlands, or rangelands and are the by-products of forest management.[3] Woody biomass typically has low monetary value and cannot be sold in traditional wood products markets. Nevertheless, it can potentially be converted into bioenergy (such as electricity, heat, gas, and biofuels) and be utilized for other bio-based products, such as solid wood products, composites, and paper and pulp. The development of biomass utilization opportunities has received much attention over the past decade because (1) biomass holds promise as a domestic source of renewable energy; (2) biomass utilization can help offset the cost of needed hazardous fuels reduction treatments on public lands; (3) it can contribute to economic development opportunities in forest communities (Aguilar and Garrett 2009; Morgan et al. 2011; Nechodom et al. 2008, 2011); and (4) biomass utilization prevents the onsite burning of piled material produced by ongoing fuels treatments on public lands, which emits greenhouse gases and reduces air quality (Daugherty and Fried 2007, Springsteen et al. 2011).

[3] http://www.fs.fed.us/woodybiomass/

Noting that treatment costs are a major constraint on the pace and scale of Forest Service fuels treatments in Sierra Nevada national forests (which are well below what is needed to mimic fuels reduction under historical fire regimes), North (2012) identified biomass utilization as one way of improving the economics of fuels treatments. Nielsen-Pincus et al. (in press) found that national forest

ranger districts that are close to sawmills and biomass facilities treated more overall hectares for hazardous fuels reduction, and more hectares in the WUI, than those further away, and that there was a threshold distance for this effect. Given its potential, why hasn't biomass utilization infrastructure developed more widely in association with federal land management in California and elsewhere in the West, and what can be done to support its development? These questions are the focus of this section. Figure 9.5.4 shows the location of biomass power plants in California as of 2011.

Economic Issues

A nationwide survey of Forest Service district rangers and biomass coordinators (Sundstrom et al. 2012) found that respondents from Region 5 (the Pacific Southwest Region) perceived the greatest barriers to biomass use to be economics and Forest Service capacity (e.g., declining agency budgets and staffing levels, lack of a guaranteed supply from federal lands, lack of staff expertise). Economic issues associated with developing viable biomass utilization opportunities include the supply of material, lack of industry infrastructure, harvest and transport costs, access to markets, and market trends.

In order for a business to be successful and attract investors, it must have an adequate and predictable supply of biomass, which is a concern in places where federal land is the main source of supply and inconsistent harvests have been a problem in the past (Becker et al. 2011, Hjerpe et al. 2009). The supply problem could be addressed by diversifying the source of raw material, and through the use of stewardship contracts, which can be awarded for up to ten years and provide a supply guarantee (Becker et al. 2011, Hjerpe et al. 2009, Nicholls et al. 2008). Factors contributing to inconsistent supply are lengthy NEPA processes and the threat of appeals and litigation, which slow down removal (Becker et al. 2011, Morgan et al. 2011); the ability to gain access to material (Becker and Viers 2007); requirements to conduct biomass inventories at the same level of detail as a traditional timber cruise, which is cost prohibitive; and lack of institutional support for biomass utilizationfor whatever reason (Morgan et al. 2011). Identifying and addressing institutional barriers, and disincentives to biomass utilization among employees, could help.

The presence of wood products industry infrastructure has been found to enhance the development or expansion of biomass utilization, which is difficult to develop as a stand-alone enterprise (Becker et al. 2011). Companies that use biomass often include sawmill residues produced as by-products from primary wood product manufacturing as an inexpensive part of their feedstock, making their operations more financially viable. The presence of timber industry infrastructure also helps maintain the capacity of the local workforce needed to carry out biomass harvesting and utilization (Becker et al. 2011). Furthermore, in places having a local market for sawlogs, harvesting timber as a component of hazardous fuels reduction treatments can help pay for the cost of biomass removal, making it economically feasible to treat larger areas for fire hazard reduction (Barbour et al. 2008, Skog et al. 2006). Furthermore, in some contexts, it may be necessary to remove sawlog-sized trees in intermediate or mid-canopy layers to reduce crown fire potential to acceptable levels (for an example from the synthesis area, see Schmidt et al. 2008). Lack of wood products industry infrastructure has been found to be a major barrier to forest restoration and associated biomass utilization in many parts of the west, though the reasons for this lack are variable (Becker et al. 2009a, Hjerpe et al. 2009). Supporting remaining wood products industry infrastructure in order to prevent its further loss can help provide opportunities for biomass removal and utilization.

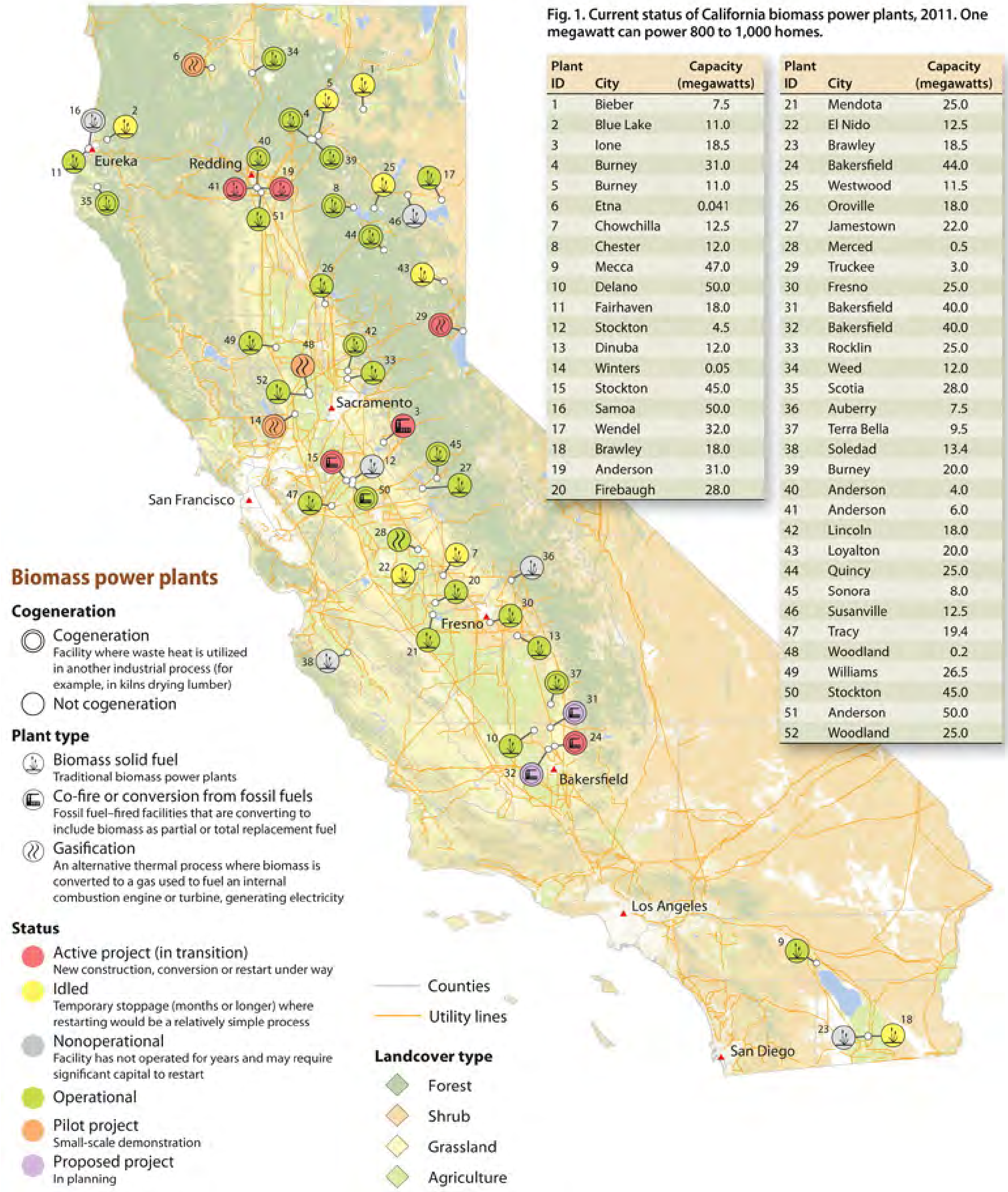

Fig. 1. Current status of California biomass power plants, 2011. One megawatt can power 800 to 1,000 homes.

Plant ID	City	Capacity (megawatts)	Plant ID	City	Capacity (megawatts)
1	Bieber	7.5	21	Mendota	25.0
2	Blue Lake	11.0	22	El Nido	12.5
3	Ione	18.5	23	Brawley	18.5
4	Burney	31.0	24	Bakersfield	44.0
5	Burney	11.0	25	Westwood	11.5
6	Etna	0.041	26	Oroville	18.0
7	Chowchilla	12.5	27	Jamestown	22.0
8	Chester	12.0	28	Merced	0.5
9	Mecca	47.0	29	Truckee	3.0
10	Delano	50.0	30	Fresno	25.0
11	Fairhaven	18.0	31	Bakersfield	40.0
12	Stockton	4.5	32	Bakersfield	40.0
13	Dinuba	12.0	33	Rocklin	25.0
14	Winters	0.05	34	Weed	12.0
15	Stockton	45.0	35	Scotia	28.0
16	Samoa	50.0	36	Auberry	7.5
17	Wendel	32.0	37	Terra Bella	9.5
18	Brawley	18.0	38	Soledad	13.4
19	Anderson	31.0	39	Burney	20.0
20	Firebaugh	28.0	40	Anderson	4.0
			41	Anderson	6.0
			42	Lincoln	18.0
			43	Loyalton	20.0
			44	Quincy	25.0
			45	Sonora	8.0
			46	Susanville	12.5
			47	Tracy	19.4
			48	Woodland	0.2
			49	Williams	26.5
			50	Stockton	45.0
			51	Anderson	50.0
			52	Woodland	25.0

Figure 9.5.4. Biomass power plants in California, 2011. Source: Mayhead and Tittmann 2012. Copyright 2012 Regents of University of California.

A number of authors have found that the cost of harvesting biomass, combined with the cost of transportation from the forest to the utilization facility, is an important factor limiting biomass use (Aguilar and Garrett 2009, Becker et al. 2009a, Pan et al. 2008). Becker et al. (2009a) used a financial model called the Harvest Cost-Revenue Estimator to estimate cost-to-revenue thresholds for different biomass harvesting scenarios under three different categories of policy options and applied it to southwestern ponderosa pine forests. The categories included policies to offset the cost of harvesting biomass, policies to reduce transportation costs through incentives or subsidies, and policies to stimulate favorable manufacturing and consumer markets for biomass and its products. They found that the cost of transporting biomass from the harvest site to the market outlet was the single greatest cost associated with biomass utilization, and that decreasing the proximity of markets to harvest sites was the only strategy that offset this cost in a meaningful way. Thus, locating processing facilities in close proximity to harvest areas in order to reduce transportation distances and associated costs is an important strategy. However, the nature of the processing infrastructure is also important; if the scale and type of processing infrastructure does not match the amount and size of hazardous fuels that need to be removed, this can be another barrier to utilization (Becker et al. 2009b).

Economic haul distance varies by place and depends upon the species and quality of the material (and therefore its value), ease of access to the site where harvesting occurs, as well as the presence of sawmills (Becker et al. 2011). Developing new harvest methods that are more cost-efficient can also help offset the cost of biomass use (Aguilar and Garrett 2009). For example, Skog et al. (2006) found that treatments would be cost-effective primarily on gentle slopes; treatment on steeper slopes requiring cable-yarding systems would require significant subsidies of either $300 or $600 per acre.

Some ways to address these limitations are to establish a network of decentralized processing facilities of an appropriate size and type closer to the source where biomass is removed (Aguilar and Garrett 2009; Nielsen-Pincus et al., in press); to develop utilization options that focus on higher value products; to bundle biomass removal with the removal of larger trees that produce higher value products (e.g., lumber) (Barbour et al. 2008); to develop transportation subsidies, which Oregon has done (Becker et al. 2011, Nicholls et al. 2008); and to implement financial incentives (e.g., cost shares and grant programs for facility development and equipment purchases, and tax incentives for facility development and harvesting and transporting biomass) (Sundstrom et al. 2012). Because biomass produced as a by-product of forest restoration tends to be of low value, strategies associated with national forest management are likely to focus on siting smaller processing facilities closer to public lands (Becker et al. 2011). Small and mid-sized facilities that focus on electricity generation, firewood, animal bedding, commercial heating, or combined heat and power systems may be more feasible than large processing facilities. This is because they tend to be less controversial and require a smaller supply of biomass to operate (making it easier to obtain in a reliable manner) (Becker et al. 2011). However, significant economies of scale favor construction of larger plants (or retrofitting of existing plants) to utilize diverse feed stocks (Nicholls et al. 2008). Daugherty and Fried (2007) found that in northern California and southern Oregon, unless small-capacity (< 15 MW) facilities are at least 90 percent as efficient as large facilities, they do not represent an economically viable alternative, because their lower efficiency offsets the reduced costs they may incur by gathering biomass from a smaller supply area (with a shorter average haul distance).

Biomass market conditions can change dramatically within the timeframes required for developing and implementing projects on national forests that include biomass removal (Morgan et al. 2011). Federal land managers involved with biomass removal from hazardous fuels reduction treatments suggested that placing individuals who are aware of biomass market conditions on NEPA-ID teams would help them plan economical projects (Morgan et al. 2011). Demand for bioenergy is contingent on energy markets, though plants with long-term power purchase agreements are sheltered from market volatility during the period of their agreement, assuming the agreement price is not tied to a floating market reference point.

California currently has more biomass power plants than any other state (Mayhead and Tittmann 2012), and its capacity to utilize biomass has been growing (Morgan et al. 2004); nevertheless, power derived from biomass currently contributes only about 2 percent of the state's electricity (Mayhead and Tittmann 2012). 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. Biomass is one eligible renewable energy source. However, the largest electrical utilities in California currently favor wind and solar sources of electricity, despite the fact that these sources do not provide a consistent baseload of power (unlike biomass) (Mayhead and Tittmann 2012).

Many existing biomass power plants have 30-year contracts with California's large investor-owned utilities, but these often pay low prices for the energy produced (though contracts vary), meaning some plants can no longer afford to run, and new contracts are not being developed (Mayhead and Tittmann 2012). Consequently, it has not been financially feasible to increase biomass capacity in California, with the possible exception of refurbishing and restarting nonoperational facilities or developing co-fire/conversion projects. Increasing the price paid for electricity generated from biomass is one way of overcoming these constraints and creating an incentive to expand biomass utilization capacity in California, whether through small-scale or larger-scale facilities (Mayhead and Tittmann 2012).

California's Senate Bill 1122, passed in September 2012, aims to address this problem by stimulating California's market for bioenergy from a distributed network of small renewable biomass projects. The law requires California's Public Utilities Commission, by June 2013, to direct the state's investor-owned utilities to collectively procure at least 250 MW of generating capacity from bioenergy projects that begin operating on or after that date.[4] Of this 250 MW, 50 MW must come from biomass produced through sustainable forest management in areas at risk for wildfire (the remainder will come from biogas sources). Eligible biomass facilities must have an effective capacity of no more than 3 MW, and must be interconnected with the electricity grid. This bill may alleviate some of the market barriers to developing biomass utilization in California.

Social Issues

One study focusing on the social acceptability of biomass utilization comes from Oregon, though the findings may be applicable to California (Stidham and Simon-Brown 2011). Based on interviews with people representing nine different stakeholder groups, the authors found a wide level of support for wood to energy projects, and that the main factor behind this support was a recognized need for forest restoration to improve forest conditions, which were viewed by many as being overstocked. However,

[4] http://leginfo.legislature.ca.gov/faces/billTextClient.xhtml;jsessionid=cd36e5138d18004eeb1fc4f367a0?bill_id=201120120SB1122

the social acceptability of fuels treatments and associated biomass utilization opportunities varied by forest type. Stakeholders were much more supportive of active management of lower elevation ponderosa pine forests than of upper elevation mixed-conifer forests, where the scientific evidence for an ecological need to reduce fuels was sparse. One finding is that science-based planning is an important mechanism for improving the social acceptability of biomass utilization projects. Science can demonstrate that forests have departed from their natural range of variability, and that restoration treatments are needed to bring them back into that range (Stidham and Simon-Brown 2011).

Even where scientific evidence attests to the need for forest restoration and stakeholders agree on this need, there can be social disagreement on the treatment types and specifications used to accomplish it. Restoration can mean different things to different people, with the removal of big trees and the intent to make economic use of restoration by-products controversial (Hjerpe et al. 2009). Lack of trust in agencies by some stakeholders can be another social barrier to developing biomass utilization opportunities (Stidham and Simon-Brown 2011). The concern is that agencies will overharvest in the name of restoration. Yet limiting the size and number of trees to be removed through restoration can reduce its effectiveness and make removal of small-diameter material and biomass even less economical. Developing fuels reduction and restoration activities, and biomass utilization projects and infrastructure, through collaborative processes that include stakeholders in planning, decision making, and partnerships to promote biomass use is one suggested approach for overcoming this social disagreement and lack of trust (Becker et al. 2011, Hjerpe et al. 2009, Stidham and Simon-Brown 2011, Sundstrom et al. 2012). Another suggested approach is to develop pilot demonstration projects in the places and forest types where activities would be located (Stidham and Simon-Brown 2011).

Tradeoffs

Although biomass utilization holds promise for contributing to the resilience of forest ecosystems and communities in the Sierra Nevada, it is important to note that it may involve tradeoffs. From an environmental standpoint, biomass utilization may encourage harvesting by whole-tree removal, which removes nutrients from the forest and poses a threat of nutrient depletion to coarse-textured, low-nutrient soils in particular (Raulund-Rasmussen et al. 2008). However, the California Forest Practice Rules may help to mitigate this risk (Evans et al. 2010). Some scientists predict that increasing harvests for the purpose of bioenergy in the Sierra Nevada may increase carbon emissions compared to "business as usual," despite the potential for reducing wildfire risk (Hudiburg et al. 2011). Other scientists question this prediction, as it depends on the parameters of the life cycle assessment being used (SAB 2012). Biomass removal may also threaten the long-term sustainability of forests if small-diameter trees are over harvested in response to high demand (Aguilar and Garrett 2009). From a social standpoint, communities may be concerned about traffic congestion and emissions associated with biomass facilities (Searcy et al. 2007).

Tools

A number of tools have been developed to help national forest managers assess the financial and economic dimensions of biomass removal during fuels treatments. They are summarized in Morgan et al. (2011) and described in Loeffler et al. (2010), with links for gaining access to them, a summary of data requirements, and key contacts provided. Tools that may be most relevant to forest managers in the Sierra Nevada are the Forest Service's Forest Inventory and Analysis program's BioSum model (Barbour et al. 2008, Daugherty and Fried 2007, Fried and Christensen 2004), which assesses how fuels reduction treatments and the siting of biomass-based energy facilities can be optimized to reduce fire hazard at the landscape scale;[5] the Forest Residue Trucking Simulator, which compares the relative costs associated with alternative methods of transporting biomass from the forest to a utilization facility;[6] the Fuel Reduction Cost Simulator, which estimates the cost of fuels reduction projects that entail tree removal for wood products or chips;[7] and the Forest Service's Southern Research Station's Moisture Content Converter, which helps managers estimate the dry mass of biomass that will be sold and processed from a treatment.[8]

Management Implications: Opportunities to Increase Biomass Utilization from National Forests

- Identify and address institutional barriers to producing a predictable supply of biomass from national forests
- Support establishment of appropriately-scaled and typed biomass utilization facilities close to harvest areas on national forests to reduce transportation distances and associated costs
- Develop biomass utilization options that focus on higher value products, and include merchantable trees in biomass removal projects to make them more economical
- Place individuals who are aware of biomass market conditions on NEPA-ID teams to help plan economical biomass removal projects
- Support science-based planning and engage in collaborative processes when developing projects and infrastructure to promote biomass use to improve their social acceptability
- Work to improve markets for biomass and maintain existing timber industry infrastructure because of its importance in making biomass utilization economically feasible

Non-timber Forest Products

Non-timber forest products (NTFPs) include a wide range of species and their parts, and can be grouped into five categories: (1) foods; (2) medicinal plants and fungi; (3) floral greens and horticultural stocks; (4) fiber and dye plants, lichens, and fungi; and (5) oils, resins, and other chemical extracts from plants, lichens, and fungi (McLain and Jones 2002). There is a rich literature documenting historical and more recent Native American uses of non-timber forest products in California, including the Sierra Nevada (e.g., Anderson 2005) (Wiegand 2002); there is much less literature available regarding present-day uses

[5] http://www.fs.fed.us/pnw/fia/biosum/
[6] http://www.srs.fs.usda.gov/forestops/biomass.php
[7] http://www.fs.fed.us/pnw/data/frcs/frcs.shtml
[8] http://www.srs.fs.usda.gov/forestops/biomass.php

by other groups. Reduced timber harvesting on national forests in the early 1990s and associated job loss in forest communities in the Sierra Nevada and elsewhere in northern California spurred interest in exploring the potential for commercial NTFP harvesting—especially of medicinal plants—as an alternative source of employment (Wiegand 2002). Most research on commercial NTFP harvesting has been carried out in the Pacific Northwest, however, leaving a research gap regarding the role of commercial NTFP harvesting in California and how it contributes to rural economies in forest communities. For most commercial harvesters, NTFPs provide a supplemental source of income (Jones and Lynch 2008).

Non-timber forest products harvested from Forest Service lands in the Sierra Nevada include wild food plants (e.g., mushrooms, fruits, ferns), medicinal plants, floral greens, seeds and cones, posts, poles, firewood, transplants, and Christmas trees (Richards 1996). Although NTFPs are not as abundant in the Sierra Nevada as they are in moister bioregions of California, such as the northern coastal areas, they are nevertheless relatively abundant compared with other bioregions in the state (Christensen et al. 2008). Most people harvest NTFPs for personal and subsistence uses, but commercial harvesting is also important. Nationwide, the annual retail value of commercial NTFP harvests from forest lands is estimated at $1.4 billion, with about 20 percent of the supply coming from Forest Service lands (Alexander et al. 2011). Not only do NTFPs have cultural importance and offer economic diversification opportunities in rural communities, but harvesters can also contribute to the sustainable management of NTFPs on national forest lands (Jones and Lynch 2008). They can do this, for example, by sharing the ecological knowledge and management practices they have developed through their harvest activities, and participating in NTFP research and monitoring efforts (Charnley et al. 2008b, Jones and Lynch 2008).

A number of authors have examined how national forest management can support economic diversification opportunities in forest communities through NTFP harvesting (e.g., Charnley et al. 2007, 2008b; Jones and Lynch 2008; Jones et al. 2002). Although these findings are based on research carried out in the Pacific Northwest, they are likely to be relevant to the management of NTFPs in the Sierra Nevada. They are summarized below.

Management Implications: Non-timber Forest Product Harvesting

- Engage in active management of commercially-valuable NTFPs to sustain or increase their diversity, productivity, and availability by integrating them into forest management activities
- Avoid the destruction of important gathering sites when planning timber sales and managing for fire
- Integrate commercial harvesters, buyers, and processers into forest management activities associated with NTFPs so that they can share their ecological knowledge and insights about these species, and information about harvesting activities, with land managers
- Enlist harvesters in inventorying NTFPs and in monitoring the impacts of forest management activities (e.g., timber harvest, grazing, fire management) and harvesting on NTFP species populations to support their management
- Adjust access fees and permit prices so that they do not undermine the financial feasibility of commercial harvesting
- Ensure reliable access to NTFPs, perhaps through forms of access such as zoning, stewardship contracts, or leases so that harvesters can engage in stewardship of harvest areas for an extended period of time
- Include harvesters in forest planning and decision-making processes

Grazing

Researchers studying grazing in California and elsewhere in the West have pointed out the important role of grazing on public lands for maintaining viable ranching operations (Gentner and Tanaka 2002, Huntsinger et al. 2010, Sulak and Huntsinger 2007). As of 2005, roughly 71,000 cattle used Forest Service rangelands in California under approximately 400 permits (Huntsinger et al. 2010). Some of the ecological considerations associated with grazing management on Sierra Nevada national forests, including potential benefits as an environmental management tool in California (e.g., Huntsinger et al. 2012), are addressed in Chapter 6.4 (Wet Meadows). Here, the focus is on the social dimensions of public lands grazing.

California ranchers often maintain livestock herds that are larger than their private lands can support because of the number of cattle needed to have a financially viable ranching enterprise (Sulak and Huntsinger 2007). This means they must lease public or other private lands for part of the year. Ranchers living in the western foothills of the central Sierra Nevada typically graze their animals in the foothills in the winter, and in montane meadows on Forest Service lands in the summer. Because summer range is relatively scarce and of high quality on national forest lands, its economic importance is high (Huntsinger et al. 2010). Research among grazing permittees using the Tahoe, Stanislaus, and Eldorado national forests found that on average, these ranchers used about 2.6 leases per year per operation, and that the public lands lease contributed an average of 41 percent of the income they earned from ranching (Sulak and Huntsinger 2007). The importance of public land leases on these forests led one-third of the permittees interviewed to state that if they lost the leases, they would probably sell all or part of their private ranch. Given the overall dependence of ranchers on leased rangelands, the public lands component is particularly important because private rangelands in California are rapidly being converted to more intensive land uses given high development pressures, with the rate of rangeland conversion to development increasing annually (Brunson and Huntsinger 2008, Sulak and Huntsinger 2007). This trend is leading to a shortage of leases on private lands. Thus, the stability of public lands grazing is critical for maintaining ranching operations in the Sierra Nevada and elsewhere in California. Not only does public land grazing enable ranchers to maintain ranching as a component of their livelihood strategies and their culture, it also contributes to the conservation of private rangelands and their associated ecological values by helping prevent the sale of private ranches by ranchers whose operations would fold without public leases (Brunson and Huntsinger 2008, Sulak and Huntsinger 2007).

Public lands play a critical role in providing a stable forage supply for livestock. However, there have been downward trends in authorized grazing and in the number of animal unit months grazed on Forest Service lands in the West over the past several decades (Huntsinger et al. 2010). Recent declines are attributed largely to drought. In addition, fire suppression has caused a buildup of woody vegetation on Forest Service lands, reducing forage productivity. Permittees feel uncertain about what the future productivity of their allotments will be because they have little control over how national forests are managed, and they perceive increasing restrictions and more costly and complicated management requirements. Maintaining stable leases and a stable forage supply through management actions, communicating with ranchers about grazing-related issues and problems, and involving permittees in management decisions by integrating their knowledge and recommendations can help sustain ranching in the Sierra Nevada, and the broader socioeconomic and conservation benefits that ranching brings to the area (Huntsinger et al. 2010, Sulak and Huntsinger 2007).

Management Implications: Grazing on National Forests

- Maintain stable leases and a stable forage supply for livestock on Forest Service allotments through management actions
- Communicate with grazing permittees about grazing-related issues and problems, and involve them in management decisions by considering their knowledge and recommendations

Conclusions

This chapter has sought to provide a social and economic context for understanding timber harvesting, biomass utilization, NTFP harvesting, and grazing in the Sierra Nevada synthesis area, as well as associated management issues. The sustainable management of timber, biomass, NTFPs, and forage from national forests in the Sierra Nevada can benefit nearby forest communities where these activities are important by contributing to both economic and social sustainability, consistent with the direction of the 2012 Forest Service Planning Rule. The chapter points out a number of strategies forest managers can take—grounded in the published social science literature—to support continued production of forest products from national forests in the Sierra Nevada in a manner that can benefit local communities. Doing so represents an investment in long-term, sustainable job creation and more diversified local economies (Charnley et al. 2012). Moreover, supporting local forest products industries can help the Forest Service meet its mission-related goals. For example, national forest timber sale programs support local processing infrastructure and maintain markets for sawlogs and small-diameter wood, helping the agency accomplish hazardous fuels reduction. They also produce timber sale receipts that can defray the costs of restoration projects. Plieninger et al. (2012) found that private landowners in California who maintain working forests and rangelands and engage in commercial timber and livestock production are much more active than purely residential owners in carrying out management practices related to biodiversity enhancement, soil and water protection, and improving "provisioning" ecosystem services associated with timber and livestock production. Yet these authors also found that the number of owners of working forests and rangelands in California is declining. To the extent that managing federal lands for productive uses helps maintain working forests and rangelands on private lands, managing forest products on national forest lands for community benefit can also have environmental benefits for forest and rangeland ecosystems across ownerships.

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9.6 Collaboration

Susan Charnley, Jonathan Long, and Frank Lake

Contents

Executive Summary

National forest management efforts have generally moved toward collaborative and participatory approaches at a variety of scales. This includes, at a larger scale, greater public participation in transparent and inclusive democratic processes, and, at a smaller scale, more engagement with local communities. Participatory approaches are especially important for an all-lands approach to managing forest ecosystems across ownership boundaries.

Despite the challenges (reviewed in this chapter), participatory approaches to forest management have potential to provide a number of benefits, including:

- yielding more information for decisions so that they can better meet the ecological and socioeconomic goals of forest management,
- sharing data, analysis, and other information more broadly within communities,
- reconciling the technical language and outlook of management agencies with the place-specific knowledge and perspective of communities,
- enhancing the legitimacy and acceptability of decisions among stakeholders,
- providing opportunities to redress under-representation in resource management,
- incorporating traditional and local ecological knowledge to enhance forest restoration and monitoring, and
- creating multi-stakeholders ownership of forest management processes, outcomes, and measures of success

A number of models for collaborative national forest management and knowledge integration are presented in this chapter, along with insights from the literature about how to develop successful collaborative efforts that may be useful in forest management and planning.

Introduction

The Forest Service 2012 Planning Rule calls for greater public participation in the planning process. It requires the Forest Service to work with interested members of the public, partners, tribes, affected private landowners, and other government agencies in each phase of this process (assessment, plan development, revision or amendment, and monitoring), using collaborative approaches where feasible and appropriate. The rule also proposes an “all-lands approach” to planning, putting national forest lands in the context of the larger landscapes in which they are situated in order to improve understanding of management issues that cross ownership boundaries, including fire, invasive species, water, and wildlife. In addition, the rule directs officials to request information about native knowledge, land ethics, cultural issues, and sacred and culturally significant sites from tribes as part of the tribal participation and consultation process in land management planning. Accordingly, this chapter focuses on processes and models for collaboration in national forest management using an all-lands approach and incorporating traditional and local ecological knowledge.

The chapter begins with a discussion of the all-lands approach to national forest management, the challenges managers may face in taking such an approach, and potential ways to address those challenges. It follows with a discussion of processes for collaboration in national forest management, and key characteristics that lead to success. The chapter provides several models of collaboration associated with national forest management, with examples from California that forest managers in the Sierra Nevada can consider in developing and engaging with collaborative processes. This is followed by a discussion of traditional and local ecological knowledge and models for integrating those forms of knowledge into collaborative forest management. The chapter concludes by focusing on the role of collaboration in adaptive management and monitoring.

All-Lands Approach to Forest Management: Opportunities and Challenges

Under the Forest Service Planning Rule, the all-lands approach proposes to "feature collaboration engaging the public early and often to build a common understanding of the roles, values and contributions of NFS lands within the broader landscape."[1] An all-lands approach to forest management is argued to be important for promoting the health and productivity of forest ecosystems, conserving biodiversity, and sustaining critical ecosystem services (Lindenmeyer and Franklin 2002). Forest restoration and fire management, like many environmental management activities, entail large-scale ecological processes and mixed land ownership patterns (Bergman and Bliss 2004, Cortner and Moote 1999). The Forest Service mission for sustainable forest management on public lands provides opportunities for integrating community values with hazardous fuels reduction, timber management, and restoration forestry on national forests. These activities also occur (to different degrees) on other land ownerships, with Forest Service management potentially affecting adjacent jurisdictions, and vice versa. An all-lands approach to forest management calls for cooperation and collaboration with other landowners, creating an opportunity for the Forest Service to build relationships with its neighbors and to promote broad-scale restoration. Yet managing across ownership boundaries remains challenging.

There is a proliferation of opportunities for cross-boundary collaboration to manage forested ecosystems for public benefits, but this proliferation poses a challenge for how to allocate resources among efforts. Some of these initiatives are being led directly by the Forest Service, such as the Collaborative Forest Landscape Restoration Program. Others are made possible under federal laws, such as the Tribal Forest Protection Act of 2004,[2] which authorizes the Forest Service to give special consideration to tribally proposed projects on agency lands bordering Indian trust lands. Still other opportunities are being created outside the Forest Service. For instance, the Pacific Forest and

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[1] http://planningrule.blogs.usda.gov/2010/07/16/all-lands-approach/
[2] http://www.fs.fed.us/forestmanagement/stewardship/tribal/documents/TribalForestProtectionAct2004.pdf

Watershed Lands Stewardship Council plans to transfer tens of thousands of acres of forested parcels throughout the synthesis area from private ownership by Pacific Gas & Electric Company to other entities, which may include local governments, tribes, CAL FIRE, and/or the Forest Service itself. In addition, private land trusts are acquiring land for conservation purposes, in some cases in collaboration with tribes (Middleton 2011). Many of these non-federal holdings are embedded within a larger matrix of Forest Service lands. Burgeoning opportunities to collaborate across boundaries and to acquire additional lands pose a challenge for agency staff who engage in these processes in light of existing workloads. Further developing the agency's institutional capacity to collaborate across boundaries may be an important strategy for building its capacity to collaborate in these and other opportunities, and for increasing socioecological resilience in the synthesis area.

Another challenge is to try to resolve mismatches of scale between ecological and social processes. Many chapters of this synthesis emphasize the importance of managing resources across boundaries at large landscape scales and over long time horizons. In addition, the problem solving for socioecological systems has to go beyond improved biophysical scientific understanding to attend to socioeconomic values, economic and political interests, policy incentives, and institutional structures (Cortner 2000, Pritchard and Sanderson 2002). Commonly, ecological processes operate at a different scale from the institutions responsible for managing them (Cumming et al. 2006). For example, in the Sierra Nevada, the divisions between federal, state, and local institutions responsible for managing fire-prone forests make it difficult to negotiate tradeoffs between the benefits and costs of managing fires within the "fireshed" (areas that fires are likely to burn across) versus the "smokeshed" (areas where smoke from such fires is likely to go). An advantage of collaborative processes is that they enable individuals and organizations to think at a regional scale, and act at whatever spatial scale is appropriate, often through nested efforts that address issues at different scales within the broader landscape (Kemmis and McKinny 2011).

These challenges require agencies like the Forest Service to innovate and evolve in ways that can be daunting and perhaps paradoxical, raising the question: how do we build a "nonbureaucratic bureaucracy" that makes the relationship between the agency and communities more workable, while increasing capacity to operate at multiple and dynamic scales (Pritchard and Sanderson 2002)? A general trend has been to move from systems dominated by expert bureaucracy toward expanded public participation to help balance competing interests. Another less common approach has been to move toward more decision making by communities about natural resources management (Pritchard and Sanderson 2002). Although there are no single solutions to governance challenges, an overall strategy is to cultivate flexible institutional arrangements that operate at different scales and can adjust and reorganize in response to changes in ecosystem conditions and associated management challenges (Cumming et al. 2006, Koontz and Thomas 2006, Margerum 2011, Pritchard and Sanderson 2002). The various models of collaboration provided in this chapter offer examples of these kinds of arrangements.

Cooperation entails working jointly with others to solve a problem or carry out an activity (Agranoff 2006). Cooperation can be formal or informal, occur on an occasional or regular basis, and take place inside, outside, or between organizations (Agranoff 2006). In the case of cross-boundary cooperation between federal agencies and nonindustrial private forest owners, Fischer and Charnley (2012) identified rural social organization (characterized by isolation and few opportunities for interaction), high rates of absentee land ownership, gulfs in values and goals relating to fire management, and fear of bureaucratic and regulatory burdens among family forest owners as barriers to cooperation for fire hazard reduction in eastern Oregon. Nevertheless, they found that roughly one-third of surveyed forest owners had cooperated with public agencies in the past to plan, pay for, and/or conduct practices that reduce hazardous fuels; and that owners expressed strong willingness to cooperate with agencies in the future. They also found that owners who perceived a risk of wildfire to their properties, and perceived neighboring public lands as contributing to that risk, were more likely to cooperate with agencies to reduce fire risk. These findings suggest that building a common understanding of fire risk across property boundaries and among landowners may increase the likelihood of their cooperation (Fischer and Charnley 2012). The authors identify several models of cooperation between family forest owners and federal agencies that could potentially be used to reduce fire risk across ownership boundaries, and that may be relevant for the Sierra Nevada synthesis area (see sidebar). Nevertheless, they note that the balance between the costs and benefits of cooperation in forest management must be favorable to Forest Service cooperators if they are to engage in it.

Sidebar: Models of cooperation between agencies and private nonindustrial forest owners

Informal	
Over the fence	Neighboring landowners observe each other's management practices and do something similar, encourage neighbors to do more, or undertake a management activity together
Wheel and spoke	A contractor or natural resource professional works with multiple landowners to help them learn from each other, address management problems, leverage resources, access services and markets, and address management concerns
Local group	A local "change agent" creates a forum in which landowners come together to discuss common management issues, promoting communication, learning, cooperation, and leadership
Formal	
Agency-led	A natural resource agency provides education and/or technical or financial support to help landowners interact around management issues, learn from each other, and implement activities
Collaborative group	Landowners commit to a process and product, are organized by a coordinator, and are guided by policy documents

Another study of cross-boundary cooperation in fire management from eastern Oregon (Bergmann and Bliss 2004) identified deterrents to collaboration that could also be operating in some communities in the synthesis area. These include (1) short tenures and high turnover of federal staff; (2) concerns about accountability of managers when rural people believe that their livelihoods are at risk; (3) strong ideological differences among stakeholders; (4) concern about administrative burdens and regulatory limitations imposed by NEPA and other federal environmental laws; (5) skepticism among environmental groups about local collaboratives; and (6) differential risks to landowners and managers due to scale. This last concern is reflected in the statement: “A prescribed fire that burns too hot and damages standing timber might have little impact on a national forest unit of which it is a small part. A similar fire on a private ranch might eliminate college funds and retirement savings and destroy family landmarks and special places” (Bergmann and Bliss 2004: 385).

Many of these deterrents may be beyond the ability of the Forest Service to control. Nevertheless, special roles, skills, and tools that could facilitate successful cross-boundary cooperation have been posited and include:

- dedicated boundary spanners with special skill sets and incentives to facilitate cross-boundary collaboration (Rickenbach et al. 2011);
- skilled, neutral party facilitators or mediators for collaborative groups (Bartlett 2012, Cheng and Mattor 2010);
- people who have cultural competencies in establishing and managing collaborative efforts, including respect for local knowledge, flexibility, humility, and understanding of the importance of long-term commitments (Fortmann and Ballard 2011);
- memorandums of understanding between the Forest Service and cooperators (Fischer and Charnley 2012).

Collaboration in National Forest Management

Collaboration can be defined as “an approach to solving complex environmental problems in which a diverse group of autonomous stakeholders deliberates to build consensus and develop networks for translating consensus into results” (Margerum 2011: 6). Consensus can range from a simple majority to unanimous agreement among stakeholders regarding a decision, but usually means reaching a decision that everyone can live with. The more complete the consensus, the more likely that stakeholders will support implementation of the decision reached (Margerum 2011). Community-based collaborative groups are local groups that come together at the community scale to address natural resource management issues associated with public lands and resources that affect the environmental and/or economic health of the community (Firehock 2011). These groups are composed of a diverse group of local

stakeholders who make decisions and recommendations to influence the management of public lands and resources, and take actions to implement them. Collaboration in national forest management often takes place through community-based collaborative groups.

The Quincy Library Group (QLG), based in Plumas County, California, was one of the first community-based collaborative groups in the western United States. It formed in the early 1990s in response to changing national forest management policy that aimed to protect the California spotted owl, but threatened the timber industry in the northern Sierra Nevada (see Chapter 9.5). The QLG's ultimate goal was to draft a plan for forest management that would sustain both the ecological and economic health of national forest lands and forest communities locally (Bernard and Young 1997). In 1993, the QLG produced their "Community Stability Proposal," which recommended a forest restoration program that would lead to "an all-age, multi-story, fire-resistant forest approximating pre-settlement conditions" (Bernard and Young 1997: 160). The QLG was unsuccessful in getting the Forest Service to adopt and implement their plan through administrative avenues, however (London et al. 2005). Thus, in 1997, California Representative Wally Herger (R) introduced a bill to Congress that would require the Forest Service to implement the Community Stability Proposal. The bill received wide support in both the House and the Senate, resulting in the Herger-Feinstein Quincy Library Group Forest Recovery Act (HFQLG) that was signed into law in 1998 (Marston 2001). The Act provided for a five-year pilot project to carry out select plans outlined in the Community Stability Proposal on roughly 1.5 million acres of the Plumas, Lassen, and Tahoe National Forests.[3]

The HFQLG Act has been subject to continuous lawsuits since the time of its passage over questions pertaining to protection for the California spotted owl, thinning methods used for hazardous fuels reduction, and proposed clearcuts (Bernard 2010, Marston 2001). These lawsuits have contributed to delays in implementing forest management projects under the Act, resulting in extensions in 2003 and again in 2008. Despite these delays, a number of forest restoration and fire hazard reduction projects have occurred, along with research to study the effects of these projects on wildlife, watershed health, and wildfire risk.[4] But the management plan has failed to provide long-term economic stability associated with forest-based jobs (Bernard 2010). These problems have been attributed to the failure of the QLG to represent the full range of community interests and stakeholders despite strong community support at the outset (Colburn 2002), to mixed support for the management plan among Forest Service administrators (London et al. 2005), and to strong opposition from many national environmental organizations who opposed the use of federal legislation to mandate adoption of a locally-developed management plan on national forest lands (Hibbard and Madsen 2003). The Act has provided opportunities for research to study the effects of projects on wildlife, watershed health, and wildfire risk.

Ingredients for Successful Collaborations

Community-based collaborative groups have sprung up all over the West since the 1990s to engage with national forest management issues (see Dukes 2011 for examples). Over time, extensive research has

[3] http://www.fs.fed.us/r5/hfqlg/news/2011/HFQLG%20Fact%20Sheet%202011.pdf
[4] http://www.fs.fed.us/r5/hfqlg/news/2011/HFQLG%20Fact%20Sheet%202011

been carried out to identify how collaborative institutions and processes can work best, whether in association with Forest Service lands or broader, multi-ownership landscapes. McDermott et al. (2011) group the features that lead to success into three broad categories. The first concerns external sources of support, which include involvement in and support from elected officials, agency leaders, and key decision-makers in the group; legal authority and supportive laws and policies that make it possible to accomplish the actions proposed; and community involvement. The second category pertains to access to resources: sufficient and stable funding, adequate staffing, and access to and exchange of information. The third category has to do with the capacity to act. This capacity includes effective leadership, trust among participants, and social capital (networks of social relations between people and groups that enable them to coordinate and cooperate for mutual benefit).

Cheng and Sturtevant (2012) proposed a more detailed framework for assessing the collaborative capacity of communities in the context of federal forest management. Their framework identifies six arenas of collaboration and associated capacities: organizing, learning, deciding, acting, evaluating, and legitimizing; they note that the broad capacities identified by McDermott et al. (2011) impact all six of these arenas. They suggest that the framework can be used to evaluate what capacities exist within local collaborative groups, and what capacities could be enhanced to target investments in building and sustaining these groups. They noted that since government resource management agencies were typically strongest in biophysical expertise, there were several cases where universities or nongovernmental organizations assisted with economic and social monitoring.

Harmony among stakeholders is not a key ingredient for success, but stakeholders want to be confident that working relationships will be productive before investing in collaboration (Bergmann and Bliss 2004). Perhaps counter-intuitively, solutions may become more attainable where there is a combination of conflict and cooperation between stakeholders (Scheffer et al. 2002). Even though command-and-control approaches commonly fail, the success of some decentralized collaborative networks has been associated with the incentive provided by having the threat of regulation as an alternative (Dasse 2002, Scholz and Wang 2006). Having a regulatory backstop may help to allay concerns that local collaborative groups may compromise national-scale priorities, which is not uncommon (Hibbard and Madsen 2003, Bergmann and Bliss 2004).

Another factor that may open windows of opportunity for collaborative approaches is the perception of an impending crisis, as described by Moir and Block (2001). During times of “crisis, breakdown, and reorganization”—which would include the aftermath of unusually large and severe wildfires—resilience theory suggests that moving beyond conventional decision support systems to decentralized, participatory, and collaborative approaches can help build adaptive capacity (Nelson et al. 2007, Walker et al. 2002).

Benefits of Collaboration

Several scientists have documented the social benefits of collaborative natural resource management. These include creating a sense of shared ownership over large and complex environmental problems (Bryan 2004); combining different forms of ecological knowledge and promoting better and shared understanding of natural resource management issues (Ballard et al. 2008a, Bryan 2004); integrating economic and social concerns together with ecological concerns so that they can be addressed together; enhancing opportunities to pool resources and assets in addressing resource management issues (Cheng and Sturtevant 2012); improving working relationships between agencies, members of the public, and other stakeholders; and increasing community understanding of and support for land management (Firehock 2011). Collaboration also builds community resilience (Goldstein 2012). It does this by facilitating the development of trust, leadership, and social networks; by building community capacity to work together to solve problems; by increasing knowledge, skills, and learning among participants; by deepening the connections between people and places to build a stronger sense of place; and through engaged governance (Berkes and Ross 2012, Walker and Salt 2006).

The environmental benefits of collaborative forest management are not well documented, however; it remains to be seen to what extent collaborative processes will improve environmental conditions (Koontz and Thomas 2006). Nevertheless, many groups have documented environmental accomplishments resulting from collaborative forest management—such as acres of forest restoration treatments, and education and policy changes—that are anticipated to positively affect environmental conditions over the longer term (Fernandez-Gimenez and Ballard 2011). And collaborative groups often engage in monitoring and evaluation, producing information that can be used to improve environmental management, with positive implications for the environment (Fernandez-Gimenez and Ballard 2011).

Lessons Learned from the Dinkey Creek Collaborative

Bartlett (2012) provides lessons learned from the collaborative process used for hazardous fuels reduction projects at Dinkey Creek on the Sierra National Forest that may be useful elsewhere in the synthesis area. The Dinkey Creek North and South project is a 3,000 acre project designed to restore diverse, healthy, and fire-resilient forest conditions while protecting California spotted owls and Pacific fishers (North and Rojas 2012). The project is located in an area having a long history of conflict and litigation because of concerns over project impacts on threatened wildlife species (Bartlett 2012). Successful collaboration at Dinkey Creek was based on a five-stage process: assessment, organization, education, negotiation, and implementation (see Bartlett 2012 for a description of these stages). Key elements that helped facilitate successful collaboration during this process include:

- bringing a broad range of participants to the table, which helped them understand each other's values;
- developing a common conceptual framework for management actions, including purpose and need and desired conditions over the long-term, which helped to align knowledge systems;
- involving scientists to provide technical expertise during group meetings;
- willingness and ability to move forward in the face of disagreement;
- conducting site visits during project development;
- engaging stakeholders in a timely way;
- taking actions to build trust, such as finding areas of conceptual agreement, designing projects to meet multiple objectives, and engaging stakeholders in project monitoring;
- testing project implementation methods when developing new approaches, and sharing them with the collaborative;
- project monitoring to demonstrate a commitment to learning from what worked and what didn't, and to adapt future management actions to improve forest conditions; and
- a willingness to be patient with the process.

Another critical ingredient for success was the use of a professional, impartial mediator to facilitate the collaborative process, though a mediator may not always be necessary. In this case, the mediator played an important role in organizing the collaborative process, helping build trust among participants, normalizing conflict and promoting problem-solving, managing timeframes, and helping the group reach outcomes (Bartlett 2012).

Models for Collaborative Forest Management

Cortner and Moote (1999) note that models for collaboration should be selected based upon the context of the challenge to be addressed. This section describes a number of models for implementing collaborative forest management taking an all-lands approach that could be fruitful for management efforts in the synthesis area. The models are summarized in the sidebar below and discussed in more detail in the following sections.

Fire Safe Councils

In 1993, the California Department of Forestry and Fire Protection established the California State Fire Safe Council, which became an independent, non-profit organization in 2002 (Everett and Fuller 2011). The mission of the Fire Safe Council is to help Californians mobilize to protect their homes, communities, and surrounding lands from wildfire. It does so by providing educational information to, and serving as a grants clearinghouse for, individual county and community-level fire safe councils (FSCs) that have formed across the state through local, grassroots efforts to address community-level wildfire risks (Everett and Fuller 2011). Local fire safe councils promote emergency preparedness, the creation of defensible space, and offer a forum in which community members can discuss their concerns about

forest health and wildfire safety (Sturtevant and McCaffrey 2006).

Research indicates that FSCs are effective community-based, collaborative organizations that help serve as a bridge between agencies and community members in fire hazard reduction efforts, and work to effectively define and address local priorities for wildfire mitigation (Everett and Fuller 2011, Sturtevant and McCaffrey 2006). They do this in multiple ways, ranging from education and outreach, to implementing fuels reduction projects on private lands, to creating defensible space around homes, to increasing fire preparedness and emergency response capacity, to leveraging local funds and volunteer hours that supplement federal grants for fuels reduction. Contributing to their success is the fact that FSCs operate at three scales (state, county, and community), which allows for the development of locally appropriate approaches to wildfire protection in the context of a broader support network that provides access to funding, technical assistance, and other resources (Sturtevant and McCaffrey 2006). Key challenges they face are sustaining community members' interest and participation in FSC activities, sufficient funding for fuels reduction projects and operations, and implementing fuels projects on private lands (Everett and Fuller 2011).

Everett and Fuller (2011) found that there is an important role for agencies like the Forest Service in helping support community- and county-level FSCs. This role includes (1) actively partnering with them to help support their activities; (2) developing memorandums of understanding (MOUs) between the agency and the councils to formally recognize a cooperative relationship and legitimize agency employee participation in their activities; (3) coordinating with the California FSC to make funding available through its clearinghouse to help streamline the application process; (4) recognizing their achievements; and (5) providing consistent engagement and support.

Sidebar: Models of Collaborative Forest Management using an All-Lands Approach

Model	Description
Fire Safe Councils	Community-based, collaborative groups that form to address wildfire risks on private lands in their communities through education and outreach, hazardous fuels reduction projects, creating defensible space around structures, and increasing fire preparedness and emergency response capacity.
Fire Learning Networks	Collaborative groups that form at the landscape level in fire-prone ecosystems, and that are connected to one another through regional and national networks, to develop and implement strategies for hazardous fuels reduction and restoring fire to forest ecosystems locally; and to share their knowledge, experiences, and best practices with other members of the network to encourage learning and innovation in fire management and ecological restoration.
Community Wildfire Protection Plans	Plans that communities create in collaboration with land management agencies and others that lay out a framework and strategy for managing wildfire risk on federal and nonfederal lands locally, identifying priority areas to receive hazardous fuels reduction treatments, and recommending types and methods of treatments.
Collaborative Forest Landscape Restoration Projects	Collaborative, science-based forest restoration projects that are developed in collaboration with local stakeholders, take place on Forest Service lands, and promote both ecological restoration and economic benefits for local communities. These projects are funded through the Collaborative Forest Landscape Restoration Program and facilitate development of restoration projects across ownerships, helping to leverage resources to support such projects.

Prescribed Fire Councils	Prescribed fire councils are inter-entity groups (including local, state, and federal agencies, tribes, non-governmental organizations, academic institutions, and private individuals) that facilitate collaboration among members who have an interest in applying prescribed fire.
Stewardship Contracting	An administrative tool for accomplishing forest restoration work that fosters collaboration in project development and implementation, makes it possible to exchange goods for services and to retain timber receipts on a national forest to spend on restoration activities, creates local community benefit, and can be used for cross-boundary restoration projects on Forest Service and BLM lands and private lands (under the Wyden Authority).
Wyden Authority Projects	Projects funded and implemented under the Watershed Restoration and Enhancement Agreement, which gives the Forest Service authority to enter into cooperative agreements with partners to undertake activities that protect, restore, and enhance resources on public or private lands if they benefit a watershed that contains federal lands and contribute to Forest Service management goals.
Participatory Action Research	A form of systematic inquiry involving collaboration among people affected by an issue—such as scientists, researchers, managers, community members, and resource users—so that they can share their knowledge and skills, generate new knowledge, jointly solve problems, educate, take action, and effect change.
Educational Outreach	Education and outreach programs that engage members of the public with science information about forest ecosystems are not a form of collaboration, but can lead to collaborative projects in which participants contribute to forest restoration.

Fire Learning Networks

The U.S. Fire Learning Network (FLN) was created by The Nature Conservancy, the Forest Service, and the Department of Interior land management agencies in 2001 to foster collaboration across organizations and administrative boundaries in developing landscape-scale ecological restoration plans for fire-prone ecosystems (Goldstein et al. 2010, Butler and Goldstein 2010). The FLN is one type of "conservation learning network," a community of people who organize around a core issue, have common objectives, and share their expertise, skills, methods, and techniques to solve problems (Goldstein et al. 2010). Conservation learning networks promote learning among members by fostering the spread of best practices based on lessons learned from members' experiences, and identifying barriers and solutions to problems. Fire learning networks can improve forest management decision making and increase the capacity of fire managers to manage fire and other landscape-scale ecological processes (Goldstein et al. 2010).

The national Fire Learning Network has three levels of organization: national staff, regional networks, and local landscapes—the majority of which are affiliated with a regional network. Between 2002 and 2011, 15 regional networks formed nationwide, encompassing 163 landscapes (not all of which are currently active) (TNC 2012). In California, one regional network is operative: the California Klamath-Siskiyou (encompassing the Trinity and West Klamath Mountains). There is also one "demonstration landscape" in the state (unaffiliated with a regional network): FireScape Monterey (focused on the Monterey District of the Los Padres National Forest) (TNC 2012).

The goals of fire learning networks are to develop strategies for reducing hazardous fuels and restoring fire to forest ecosystems in ways that are ecologically meaningful and socially acceptable, and to create local, regional, and national linkages between collaborative groups involved in these efforts to facilitate dissemination of knowledge and innovation throughout the network (Butler and Goldstein 2010). At the landscape level, diverse stakeholders that are involved in fire management collaborate to set ecological restoration goals, create fire restoration plans, identify priority treatment areas, and develop models and mapping tools that can be used to inform implementation of treatments. These activities occur through workshops, field trips, collaborative planning exercises, meetings, and web- and print-based communication. To date, fire learning networks have been effective in informing agency fire management plans, influencing where fuels reduction work takes place on national forest and private forest lands, guiding requests for federal funding to support treatments, and influencing policy (e.g., the Forest Landscape Restoration Act). By promoting the sharing of resources among participants and the dissemination of ideas, experiences, and lessons learned through the regional and national network, they are an effective institution for adaptive management and can contribute to socioecological resilience (Butler and Goldstein 2010). The Sierra Nevada is a region in which a fire learning network could be developed to address fire management issues.

Community Wildfire Protection Plans

The Healthy Forests Restoration Act of 2003 spurred the development of community wildfire protection plans (CWPPs). CWPPs are plans that communities create in collaboration with land management agencies and others that lay out a framework and strategy for managing wildfire risk on federal and nonfederal lands locally (Jakes et al. 2012). CWPPs identify priority areas to receive hazardous fuels reduction treatments and recommend types and methods of treatments. They are developed through a collaborative, multi-stakeholder-driven process that produces plans appropriate to local social and ecological circumstances, and at a scale that makes it possible to take action to reduce wildfire risk and enhance the resilience of forest ecosystems (Jakes et al. 2011). See Jakes et al. (2012) for a guide to best management practices for creating a CWPP. Developing CWPPs not only helps communities address fire risk locally, but it also helps community members build their social networks, enhance learning, and build community capacity—all of which foster community resilience (Jakes et al. 2007).

Example of Success: Fire Safe Councils

One example of success for community wildfire preparedness in partnership with the Forest Service is from Grizzly Flats, near the Eldorado National Forest. "The Fire Safe Council secured more federal grants to support residents' efforts to reduce fire hazards, turning their homes into models of wildfire safety and inspiring neighbors to take similar steps. They also aligned their efforts with Forest Service work on nearby public land so the projects would complement and strengthen each other" (Jakes et al. 2012: 10).

Federal forest managers can support the CWPP process by (1) participating as partners in development of CWPPS, providing leadership if needed; (2) providing data, information, and expertise; (3) providing funding to support development of CWPPs; (4) facilitating network building between stakeholders; (5) helping lower-capacity communities mobilize to take action; (6) working with communities to set fuels treatment and fire mitigation priorities; and (7) considering plan priorities and recommendations in implementing fuels treatments (Fleeger and Becker 2010, Jakes et al. 2007).

Collaborative Forest Landscape Restoration (CFLR) Projects

Title IV of the Omnibus Public Land Management Act of 2009 on Forest Landscape Restoration established a fund and a program to support collaborative, science-based forest restoration projects (called CFLR projects) in priority landscapes on Forest Service lands that encourage social, economic, and ecological sustainability.[5] Although the fund can only be used on National Forest System lands, project proposals can be for a landscape that includes other federal, tribal, state, or private lands. Thus, it can facilitate development of restoration projects across ownership boundaries, and help leverage resources to undertake such projects. To be eligible for funding, projects must be developed collaboratively and provide economic benefits to local communities.

Bartlett (2012) and North and Rojas (2012) provide detailed descriptions of a forest restoration project that took place in the Dinkey Creek area of the Sierra National Forest and was developed and implemented through a successful collaborative process. Bartlett (2012) highlights elements of the collaborative process that were key to project success (described in the preceding section) and led to the Dinkey Collaborative Forest Landscape Restoration Project, one of the original CFLR projects selected for funding in 2010 following passage of the Act. The project includes 130,000 acres of the Sierra National Forest and 24,000 acres of private land.[6]

A small number of social scientists are currently undertaking research on CFLRs to assess their strengths and weaknesses as models for collaborative forest landscape restoration. To our knowledge, no peer-reviewed publications resulting from this research have yet been published.

Prescribed Fire Councils

Prescribed fire councils are inter-entity groups (including local, state, and federal agencies, tribes, non-governmental organizations, academic institutions, and private individuals) that facilitate collaboration among members who have an interest in using prescribed fire (Quinn-Davidson and Varner 2012, Costanza and Moody 2011). In California, these councils are new or just beginning to form so that they can increase the application of prescribed fire in a responsible manner, and work to overcome constraints to its use. They serve as forums for disseminating knowledge and keeping people who undertake prescribed burns current with information about new research and technological advances, and can also be used to inform members about training opportunities and local fire issues (Wade et. al 2006). The recently-formed Northern California Prescribed Fire Council seeks to connect interested

[5] http://www.fs.fed.us/restoration/CFLR/documents/titleIV.pdf
[6] http://www.fs.fed.us/restoration/CFLR/documents/2010Proposals/Region5/Sierra/Sierra_NF_CFLRP_Proposal.pdf

persons and groups and enable discussion about possible barriers to prescribed fire application in the region, where its use is highly constrained (Quinn-Davidson and Varner 2012). For more information, see www.norcalrxfirecouncil.org.

Stewardship Contracting

As described in Chapter 9.4 of this synthesis (Strategies for Job Creation through Forest Management), stewardship contracting is an administrative tool for accomplishing community-based forest restoration work that fosters collaboration in project development and implementation. This collaboration can take many forms. In some cases, local collaborative groups form or, if they already exist, they morph as stewardship groups to develop projects that contribute to both forest restoration and local economic development. The White Mountain Stewardship Project on the Apache-Sitgreaves National Forest in Arizona is one example of a landscape-scale collaborative restoration effort taking place through the use of a ten-year stewardship contract. Although it has been extremely successful in building social agreement around forest restoration activities in the region, increasing community capacity to engage in forest restoration, and accomplishing hazardous fuels reduction treatments, it has fallen short of its goals with regard to the latter because of a shortage of federal funding to plan, administer, and implement projects (Abrams 2011). The use of stewardship contracting and utilization of restoration by-products have helped cover the cost of fuels treatments, but not completely; a funding gap remains that has been challenging to fill in the context of dwindling federal funding for forest management (Abrams 2011).

Stewardship contracting authorities apply to the Forest Service and Bureau of Land Management; thus, stewardship projects using these authorities typically take place on Forest Service and BLM lands. Stewardship contracting can be used to achieve forest restoration across the administrative boundaries of these two agencies to achieve broader landscape-scale restoration goals, as in the case of the Weaverville Community Forest in Trinity County, CA (Frost, in press). Stewardship contracting authorities can also be used together with other authorities (such as the Watershed Restoration and Enhancement Agreement, and the Tribal Forest Protection Act) to develop forest restoration projects across federal and private or federal and tribal boundaries.

Watershed Restoration and Enhancement Agreement (Wyden) Authority

The Watershed Restoration and Enhancement Agreement (Wyden) Authority, which became permanent in 2011, gives the Forest Service the ability to enter into cooperative agreements with partners in order to undertake activities that protect, restore, and enhance habitat and other resources on public or private lands, including activities that reduce risk from natural disasters that threaten public safety, if they benefit the resources within a watershed and contribute to Forest Service goals and objectives.[7] Under the Wyden Authority, federal funding can be used to implement projects and carry out activities on private lands within watersheds that include Forest Service lands in order to achieve watershed restoration goals. This authority makes it possible to collaboratively plan projects across ownership boundaries to achieve common management objectives that improve watershed health.

[7] https://www.cfda.gov/?s=program&mode=form&tab=step1&id=73c38aa3683fc789cedce7aa16f1df53

Participatory Action Research

Participatory action research (PAR) is a form of systematic inquiry that entails collaboration among people who are affected by an issue being studied so that they can educate, take action, and effect change (Ballard and Belsky 2010). It emphasizes joint problem solving and reflection by collaborative groups that can include scientists, academic researchers, managers, community members, and natural resource users who share their site-specific knowledge, skills, and expertise in solving natural resource management problems (Everett 2001). Models of PAR to address natural resource management on Forest Service lands can be found from the Shasta-Trinity National Forest in northern California (Everett 2001) and the Olympic National Forest in Washington (Ballard and Belsky 2010, Ballard and Huntsinger 2006). Examples of participatory action research on tribal lands come from the Olympic Peninsula (Ballard et al. 2008b) and Arizona (Long et al. 2008). Because of its emphasis on environmental learning, Ballard and Belsky (2010) argue that participatory action research can promote socioecological resilience in forests and forest communities. A substantive body of research provides guidance for how to conduct participatory action research, and reflects on its challenges and benefits (Fortmann 2008, Wilmsen et al. 2008). Forest Service managers could encourage and benefit from participatory action research by (1) inviting people affected by an issue—such as scientists, managers, community members, and forest users—to share their knowledge; (2) treating that knowledge with respect and considering it in decision making; and (3) actively engaging them as colleagues in scientific inquiries and experiments designed to promote more sustainable forest management.

Educational Outreach to Promote Collaboration

One means of engaging local community members in collaborative efforts on national forest lands is through educational outreach. The Sagehen Experimental Forest, located near Truckee in the Sierra Nevada, provides an excellent example of this approach. At Sagehen, school children, university students, and community members participate in education and outreach programs related to watershed restoration and hydrologic systems (Cerveny and Charnley, in press). For example, sixteen hectares (40 acres) were committed to the local school district for science programs. A partnership between the University of California, Berkeley and local elementary schools, as well as a summer speaker series that engages the public in science, has also been established. Sagehen staff members collaborate with the Truckee River Watershed Council on watershed restoration projects. And the Sagehen website has links to a Fish-Cam, news blogs, and podcasts about ongoing research. Each fall, 500-600 community members work together on a variety of watershed restoration projects on the Sagehen (Cerveny and Charnley, in press). Thus, active outreach and education programs, and an emphasis on citizen science, can lead to collaborative projects and build support for collaborative forest restoration.

Local and Traditional Ecological Knowledge

Every society and culture has knowledge systems that guide their interactions with their environment, including utilization of resources. Local ecological knowledge (LEK) is defined as “knowledge, practices, and beliefs regarding ecological relationships that are gained through extensive personal observation of and interaction with local ecosystems, and shared among local resource users” (Charnley et al. 2008: 2).

Traditional ecological knowledge (TEK) is defined by Berkes et al. (2000: 1252) as “a cumulative body of knowledge, practice, and belief, evolving by adaptive processes and handed down through generations by cultural transmission, about the relationship of living beings (including humans) with one another and with their environment.” Tribal traditional ecological knowledge is intergenerational knowledge derived from long-term relationships with places, but it is also dynamic, adapting to conditions of resources and ecosystems (Berkes et al. 2000). Native Americans view many aspects of the natural environment as vitally important to perpetuation of tribal cultures, economies, and societies, and the special relationship between the federal government and tribes provides opportunities and responsibilities to cooperatively protect and restore those values. In 2006, the Forest Service adopted an interagency policy to support traditional gathering of culturally important plants to promote ecosystem health using traditional management practices through collaborative relationships with tribes, tribal communities, tribal organizations, and native traditional practitioners.

Relevance of TEK/LEK to Desired Conditions and Evaluating Restoration

Traditional ecological knowledge (TEK) and local ecological knowledge (LEK) can facilitate understanding of the objectives, location, frequency, seasonality, and other characteristics of practices by indigenous people and more recent settlers who have influenced ecological characteristics across the landscape. Uses of these forms of knowledge for forest biodiversity conservation in the Pacific Northwest are discussed in detail in a report by Charnley et al. (2007, 2008). Baselines are often founded upon conditions prior to Euro-American settlement, so an understanding of past uses and management can provide information valuable in restoring ecosystems (Charnley et al. 2008). An important theme throughout this synthesis is the importance of reestablishing reference fire regimes, and in many areas, indigenous burning practices were an important part of those reference conditions (Van de Water and Safford 2011). Therefore, traditional burning practices are important to consider in formulating strategies to restore fire regimes and the numerous species that depend on fire and whose abundance and quality likely suffers due to the legacy of widespread fire suppression (see Chapter 4.2, Fire and Tribal Cultural Resources).

Traditional and local ecological knowledge may be used to complement and refine monitoring efforts to understand changes in culturally important resources, especially those that are harvested, and their broader environments. As an example, Shebitz et al. (2008) described how TEK practitioners identified beargrass as a culturally important plant undergoing declines due to changes in fire regimes and the impacts of commercial harvest, and they applied their knowledge in restoration projects. In collaboration with agency managers and/or researchers, tribal practitioners who have TEK of species, habitats, or ecological processes could help improve monitoring, restoration, and conservation activities. This interaction could be particularly valuable in understanding responses to climate change by considering traditional knowledge of phenology (Nabhan 2010). Collaborations between managers, researchers, and tribal practitioners holding TEK can also suggest appropriate metrics for evaluating socioecological resilience, which might include the quality and quantity of acorns, basketry materials, or other key resources derived from “cultural keystone species” (Garibaldi and Turner 2004) that support community health and livelihoods.

Engagement with TEK/LEK Holders and Practitioners

The Sierra Nevada is the aboriginal territory of dozens of Indian tribes and other Indian communities (Reynolds 1996). Because of the unique status of Indian tribes as sovereign entities, their special government-to-government relationship with the federal government, and the federal trust responsibility, Indian tribes are distinct from all other stakeholders (Getches et al. 2011).[8] The Leadership Intent document regarding ecological restoration policy in Region 5 notes that collaborations with regard to TEK are particularly important. Tribal communities within the Sierra Nevada present distinctive opportunities for mutually beneficial partnerships to restore ecologically and culturally significant resources and to promote socioecological resilience (Reynolds 1996). Culturally appropriate communications and procedures for information management are important to maintain trust, respect, and productive relationships between the agency and tribes.

Efforts to engage TEK and LEK in forest management are more likely to be successful when the knowledge holders are directly engaged as active partners in pursuit of mutual goals. Charnley et al. (2008) noted that engaging local forest users in 'joint forest management' will aid in the practical application of these forms of knowledge. An example of this type of partnership is a collaborative forest restoration project involving the Maidu community and the Plumas and Lassen National Forests (Charnley et al. 2008, Donoghue et al. 2010). As demonstrated by this example, possible tools to facilitate partnerships may include stewardship contracts or other agreements that allow tribes to have sustained access to resources for an extended period in order to engage in long-term ongoing management (Charnley et al. 2008). The models for Collaborative Forest Management described in the previous section of this chapter can be extended to include tribes and tribal traditional ecological knowledge. Additional examples of collaborations between the Forest Service and tribes are included in Chapter 4.2, Fire and Tribal Cultural Resources.

Efforts to incorporate TEK and LEK into forest plan revision will be easier where local collaborations are already underway and can overcome many of the challenges to sharing information in productive ways. In suggesting strategies to incorporate TEK into environmental plans, Usher (2000) explains that treating TEK as a dataset may decontextualize the information and is likely to be viewed as disrespectful. He recommends using complementary methods, including interviews, reports, and direct statements at public hearings, to include information at different stages of the assessment and planning process. Similarly, Raymond et al. (2010) emphasize the importance of integrating TEK into management as a cyclical process for solving problems rather than as a product. These findings reinforce the importance of successful collaborations, which help to overcome communication challenges by developing shared understandings of key terms as well as the different decision-making processes of the TEK/LEK holders and the Forest Service. A series of case studies on the role of TEK in tribal-federal collaborations reported by Donoghue et al. (2010) highlighted a variety of approaches and some of the benefits that can be achieved through tribal-federal collaborations when the parties shared in project implementation and the transfer of knowledge was ongoing throughout the process.

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[8] http://www.fs.fed.us/people/tribal/trib-1.pdf

Filling Gaps in Knowledge

Charnley et al. (2008) noted that present models and examples for integrating TEK and LEK into forest management focus mainly on Native Americans. More detail is needed about the degree of integration of TEK and LEK held by forest workers, non-native harvesters of non-timber forest product (NTFPs), ranchers, and other forest users into management, as well as information about variables that are barriers to or facilitate successful knowledge integration. Although Donoghue et al. (2010) started to fill this gap, additional research would address the diversity of communities and issues in socioecological restoration in the Sierra Nevada.

However, the first priority research area regarding Native American land use practices identified by Anderson and Moratto (1996) in the Sierra Nevada Ecosystem Project (SNEP) report to Congress was whether Native American uses of fire and other forms of vegetation management should be reintroduced. Additional participatory research partnerships in this vein would help answer important conservation questions, including expected effects of traditional light burns as well as more severe wildfires on valued resources. The Forest Service and many indigenous groups are likely to have mutual interests in restoration using fire for a number of plants valued for their cultural and ecological significance, and several examples in Chapter 4.2, Fire and Tribal Cultural Resources, show that progress is underway on national forests in the Sierra Nevada and surrounding regions.

Research is needed to go beyond describing ecological knowledge systems to understanding how TEK and LEK are implemented and what the associated ecological outcomes are in order to determine their potential contributions to biodiversity conservation (Charnley et al. 2008). When establishing partnerships intended to share information to address complex socio-cultural and environmental issues, it is important to consider how adaptive learning will be perpetuated over the long term. Turner and Berkes (2006) highlighted the need to practice incremental learning and knowledge dissemination. Promoting systems to track partnerships and their outcomes throughout the region would provide data to evaluate success of those efforts and would facilitate social learning about incorporating TEK and LEK into management strategies.

Collaboration in Monitoring and Adaptive Management

Adaptive management is broadly characterized as learning through management, with adjustments made as understanding improves (Williams 2011). Adaptive management is commonly conceived as a structured approach that involves cycles of planning, action, monitoring, and evaluation. Adaptive management is often described along a continuum from passive to active, with the more active formulations involving management interventions implemented as experiments (Williams 2011). A core characteristic of adaptive management systems is a design that facilitates responses based upon previously tested policies and accumulated knowledge, and that promotes social learning as a way to respond to novel challenges (Berkes and Folke 2002).

Components of adaptive management systems, such as modeling and stakeholder collaboration, can facilitate learning and adaptive responses; however, feedback processes are particularly critical for facilitating effective responses to and learning from surprises (Berkes and Folke 2002). These processes may include formal monitoring of quantifiable indicators, such as counts of species, as well as more qualitative and integrated socioecological indicators that are embedded in traditional and local ecological knowledge systems, including the accumulated knowledge of long-time agency employees, harvesters and other forest resource users, and local residents (Berkes and Folke 2002). These approaches may be complementary, since systems based upon traditional or local ecological knowledge may be well attuned to recognizing perturbations that portend major shifts in system function (Berkes and Folke 2002). As an example from the Sierra Nevada, the invasion of Asian clam into various locations in Lake Tahoe was detected both by researchers conducting routine near-shore monitoring and by citizens who recognized the clams as unusual and alerted specialists.

Critics have noted that initiatives labeled as adaptive management often do not address underlying problems, and that despite the rhetoric around the concept, it has rarely been implemented on the ground in the context of forest management (Stankey et al. 2003). Costs are often steep if active adaptive management, with the research that it entails, is the goal. For the Forest Service, the annual appropriations model severely constrains the ability to sustain major projects. An important demonstration project in the region is the ongoing Sierra Nevada Adaptive Management Project (SNAMP), a regionally-based, well-funded endeavor to practice project implementation through the collaborative study of forest land management by researchers, agency personnel, and stakeholders.[9] The Integrative Approaches chapter (1.1) points out that this and similar research projects provide valuable opportunities to advance learning, but they have not sustained sufficient funding and support to evaluate long-term ecological responses.

There may be numerous barriers, including funding and bureaucratic resistance, to transitioning from relatively short-term projects to long-term and larger adaptive management systems. Pritchard and Sanderson (2002) suggest that when adaptive management is adopted by bureaucracies, there are strong tendencies to revert back to more conventional technocratic approaches. Barriers to adaptive management within the Forest Service include dwindling resources, growing workloads for staff, lack of leadership, and institutional and regulatory constraints on innovation (Stankey et al. 2003). Nadasdy (2007) noted that many current management frameworks pay insufficient attention to the social and political dimensions of who the winners and losers are under different management approaches; these frameworks may winnow consideration of baselines and approaches based upon present political factors, rather than long-term sustainability.

Another critique of adaptive management is that monitoring is often not done well enough and for long enough periods to evaluate important and potentially surprising effects of management (Moir and Block 2001). Because management systems are typically scaled to the immediate future, they may not be well suited for dealing with slower, long-term ecosystem responses and surprises (Moir and Block 2001), both which may be expected under climate change. As a result, combinations of different types of monitoring and even some research applications may be needed to evaluate impacts and outcomes across different scales. The challenge of developing science capacity is even more important when trying to address complex, long-term changes in ecological systems; a key knowledge gap is to identify likely

[9] http://vtm.berkeley.edu/

thresholds that should be the target of monitoring, even if they have not been encountered (Walker et al. 2002), and the appropriate response if monitoring suggests that a threshold has been reached (Moir and Block 2001).

Collaborative, multiparty monitoring of selective key indicators has been recommended as an approach to deal with long-term ecological changes and contrasting views on forest management (Bliss et al. 2001, Moir and Block 2001). Multi-party monitoring entails community members or groups of interested stakeholders who organize to monitor forest resources or forest management activities and their social or ecological effects (e.g., Bliss et al. 2001, CFRP 2005, Lynch et al. 2004). It is also a way to allow verification of Forest Service findings and build confidence in Forest Service management. There are several examples of multi-party monitoring for national forest management.

Some community-based organizations have taken the initiative to conduct monitoring that evaluates the links between management actions on nearby federal forests and socioeconomic trends in their communities and counties (Danks 2009). These types of monitoring programs help communities, agencies, and others understand the social and economic effects of forest management practices. Elsewhere, stakeholder groups have become involved in monitoring collaborative forest restoration projects on national forests. They are determining how well the projects are achieving their social and ecological goals, and if restoration forestry techniques and plans should be altered to improve results (e.g., CFRP 2005). Another example of participatory monitoring comes from Lynch et al. (2004), who describe how non-timber forest product (NTFP) harvesters monitor the amount and distribution of NTFP species on federal forest lands and harvester impacts on them. Several organizations have developed handbooks to guide the participatory monitoring process (e.g., Davis-Case 1998, Moseley and Wilson 2002, Pilz et al. 2006, USDA FS 2005).[10]

Participatory monitoring initiatives face many of the same fundamental challenges of time, funding, and staffing as does agency monitoring. They also face added challenges in obtaining broad-based and sustained community participation for long-term monitoring and in securing technical assistance and science capacity to ensure data validity and credibility (Fernandez-Gimenez et al. 2008). Emerging technologies and accompanying paradigm shifts are aiding development of capacity to facilitate these efforts (Newman et al. 2012).

Although collaborative approaches have been considered a means of reducing the high costs of monitoring required for certain regulatory approaches (Dasse 2002) and a means to facilitate community participation, case-control comparisons of costs and benefits of collaborative versus conventional agency monitoring are needed (Fernandez-Gimenez et al. 2008). Therefore, although scientists studying resilience have suggested important elements of robust adaptive management systems, it would be difficult to quantify the benefits of incorporating them, especially given the short amount of time that has passed since more modern systems of adaptive management have been established.

[10] For more resources relating to monitoring socioeconomic indicators in the context of restoration on Forest Service lands, go to http://ewp.uoregon.edu/sites/ewp.uoregon.edu/files/WP_36.pdf.

Despite these potential problems, studies have documented that collaborative monitoring can yield social benefits, such as improved relationships and trust that build social capital to make collaborative natural resource management more successful (Fernandez-Gimenez et al. 2005, Fernandez-Gimenez et al. 2008, Kusel et al. 2000). It also leads to shared understandings of ecosystems and increased ecological knowledge among participants, social learning, community building, greater adaptive capacity, communication of monitoring results, and to some degree, adaptive management (Cheng and Sturtevant 2012, Fernandez-Gimenez et al. 2008). Increasing attention is also being given to various "citizen science" projects and other forms of public participation as opportunities to conduct monitoring and research, especially at large spatial scales, and to better engage the public (Dickinson et al. 2012).

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Appendix: Recent Topical Syntheses

Many of the recent topical synthesis reports cited in this report are listed below. Readers are urged to review the reference sections of each individual chapter for a more complete list.

Forest Ecology and Climate Change

Busse, Matt D.; Hubbert, Ken D.; Moghaddas, Emily Y. 20xx. Fuel reduction practices and their effects on soil quality. Gen. Tech. Rep. PSW-GTR-xxx. Albany, CA: U.S. Department of Agriculture, Forest Service, Pacific Southwest Research Station. XXX p.

Elliot, William J.; Miller, Ina Sue; Audin, Lisa. Eds. 2010. Cumulative watershed effects of fuel management in the western United States. Gen. Tech. Rep. RMRS-GTR-231. Fort Collins, CO: U.S. Department of Agriculture, Forest Service, Rocky Mountain Research Station. 299 p.

Joyce, L.A., G.M. Blate, J.S. Littell, S.G. McNulty, C.I. Millar, S.C. Moser, R.P. Neilson, K. O'Halloran, D.L. Peterson. 2008. Adaptation options for climate-sensitive ecosystems and resources: National Forests. Chapter 3 in Preliminary review of adaptation options for climate-sensitive ecosystems and resources. Synthesis and Assessment Product 4.4, U.S. Climate Change Science Program. 126 p.

Kliejunas, John T.; Geils, Brian W.; Glaeser, Jessie Micales; Goheen, Ellen Michaels; Hennon, Paul; Kim, Mee-Sook; Kope, Harry; Stone, Jeff; Sturrock, Rona; Frankel, Susan J. 2009. Review of literature on climate change and forest diseases of western North America. Gen. Tech. Rep. PSW-GTR-225. Albany, CA: U.S. Department of Agriculture, Forest Service, Pacific Southwest Research Station. 54 p.

Kliejunas, John T. 2011. A risk assessment of climate change and the impact of forest diseases on forest ecosystems in the Western United States and Canada. Gen. Tech. Rep. PSW-GTR-236. Albany, CA: U.S. Department of Agriculture, Forest Service, Pacific Southwest Research Station. 70 p.

K**napp, Eric E.; Estes, Becky L.; Skinner, Carl N. 2009.** Ecological effects of prescribed fire season: a literature review and synthesis for managers. Knapp, Eric E.; Estes, Becky L.; Skinner, Carl N. 2009. Ecological effects of prescribed fire season: a literature review and synthesis for managers. Gen. Tech. Rep. PSW-GTR-224. Albany, CA: U.S. Department of Agriculture, Forest Service, Pacific Southwest Research Station. 80 p.

Luce, Charles; Morgan, Penny; Dwire, Kathleen; Isaak, Daniel; Holden, Zachary; Rieman, Bruce 2012. Climate change, forests, fire, water, and fish: Building resilient landscapes, streams, and managers. Gen. Tech. Rep. RMRS-GTR-290. Fort Collins, CO: U.S. Department of Agriculture, Forest Service, Rocky Mountain Research Station. 207 p.

McDonald, Philip M.; Fiddler, Gary O. 2010. Twenty-five years of managing vegetation in conifer plantations in northern and central California: results, application, principles, and challenges. Gen. Tech. Rep. PSW-GTR-231. Albany, CA: U.S. Department of Agriculture, Forest Service, Pacific Southwest Research Station. 87 p.

McIver, J. D., S. L. Stephens, J. K. Agee, J. Barbour, R. E. J. Boerner, C. B. Edminster, K. L. Erickson, K. L. Farris, C. J. Fettig, C. E. Fiedler, S. Haase, S. C. Hart, J. E. Keeley, E. E. Knapp, J. F. Lehmkuhl, J. J. Moghaddas, W. Otrosina, K. W. Outcalt, D. W. Schwilk, C. N. Skinner, T. A. Waldrop, C. P. Weatherspoon, D. A. Yaussy, A. Youngblood, and S. Zack. 2012. Ecological effects of alternative fuel reduction treatments: principal findings of the national Fire and Fire Surrogates study (FFS). International Journal of Wildland Fire. On line: http://dx.doi.org/10.1071/WF11130

North, Malcolm; Stine, Peter; O'Hara, Kevin; Zielinski, William; Stephens, Scott 2009. An ecosystem management strategy for Sierran mixed-conifer forests. Gen. Tech. Rep. PSW-GTR-220. Albany, CA: U.S. Department of Agriculture, Forest Service, Pacific Southwest Research Station. 49 p.

North, Malcolm, ed. 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.

Peterson, D.L.; Agee, J.K.; Aplet, G.H.; Dykstra, D.P.; Graham, R.T.; Lehmkuhl, J.F.; Piliod, D.S.; Potts, D.F.; Powers, R.F.; Stuart, J.D. 2009. Effects of timber harvest following wildfire in western North America. General Technical Report PNW-GTR-776. USDA Forest Service, Pacific Northwest Research Station, Portland, OR. 51 p.

Reinhardt, E.; Keane, R.; Calkin, D.; Cohen, J. 2008. Objectives and considerations for wildland fuel treatment in forested ecosystems of the interior western United States. Forest Ecology and Management. 256(12): 1997-2006.

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Post-wildfire Management

Foltz, Randy B.; Robichaud, Peter R.; Rhee, Hakjun 2008. A synthesis of postfire road treatments for BAER teams: methods, treatment effectiveness, and decisionmaking tools for rehabilitation. Gen. Tech. Rep. RMRS-GTR-228 Fort Collins, CO: U.S. Department of Agriculture, Forest Service, Rocky Mountain Research Station. 152 p.

Rangeland Management

Briske, D.D., editor. 2011. Conservation Benefits of Rangeland Practices: Assessment, Recommendations, and Knowledge Gaps. United States Department of Agriculture, Natural Resources Conservation Service.

Meadows

Chambers, Jeanne C.; Miller, Jerry R., eds. 2011. Geomorphology, hydrology, and ecology of Great Basin meadow complexes - implications for management and restoration. Gen. Tech. Rep. RMRS-GTR-258. Fort Collins, CO: U.S. Department of Agriculture, Forest Service, Rocky Mountain Research Station. 125 p.

Ramstead, K.M.; Allen, J.A.; Springer, A.E. 2012. Have wet meadow restoration projects in the Southwestern U.S. been effective in restoring geomorphology, hydrology, soils, and plant species composition? Environmental Evidence. 1(11): 16.

Great Basin Ecosystems

Chambers, Jeanne C.; Devoe, Nora; Evenden, Angela, eds. 2008. Collaborative management and research in the Great Basin - examining the issues and developing a framework for action. Gen. Tech. Rep. RMRS-GTR-204. Fort Collins, CO: U.S. Department of Agriculture, Forest Service, Rocky Mountain Research Station. 66 p.

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