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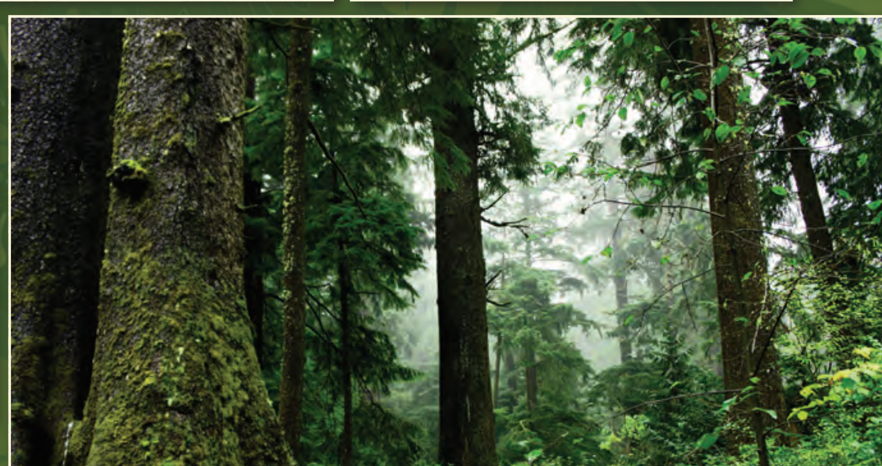
Synthesis of Science to Inform Land Management Within the Northwest Forest Plan Area

Volume 1



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Cover photos: Upper left: prescribed burn operations on the Wallowa-Whitman National Forest, Oregon; photo by Wallowa-Whitman National Forest. Upper right: a northern spotted owl in the McKenzie River Basin in Oregon; photo by John and Karen Hollingsworth, U.S. Fish and Wildlife Service. Lower right: old-growth forest, Oswald West State Park, Oregon; photo by David Patte, U.S. Fish and Wildlife Service. Lower left: marbled murrelet; photo by Kim Nelson, Oregon State University.

Synthesis of Science to Inform Land Management Within the Northwest Forest Plan Area

Volume 1

Thomas A. Spies, Peter A. Stine, Rebecca Gravenmier,
Jonathan W. Long, and Matthew J. Reilly, Technical Coordinators

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Abstract

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The 1994 Northwest Forest Plan (NWFP) was developed to resolve debates over old-growth forests, endangered species, and timber production on federal forests in the range of the northern spotted owl. This three-volume science synthesis, which consists of 12 chapters that address various ecological and social concerns, is intended to inform forest plan revision and forest management within the NWFP area. Land managers with the U.S. Forest Service provided questions that helped guide preparation of the synthesis, which builds on the 10-, 15-, and 20-year NWFP monitoring reports and synthesizes the vast body of relevant scientific literature that has accumulated in the 24 years since the NWFP was initiated. It identifies scientific findings, lessons learned, and uncertainties and also evaluates competing science and provides considerations for management.

This synthesis finds that the NWFP has protected dense old-growth forests and maintained habitat for northern spotted owls, marbled murrelets, aquatic organisms, and other species despite losses from wildfire and low levels of timber harvest on federal lands. Even with reductions in the loss of older forests, northern spotted owl populations continue to decline. Moreover, a number of other goals have not been met, including producing a sustainable supply of timber, decommissioning roads, biodiversity monitoring, significant levels of restoration of riparian and dry forests, and adaptation and learning through adaptive management.

New conservation concerns have arisen, including a major threat to spotted owl populations from expanding populations of the nonnative barred owl, effects of fire suppression on forest succession, fire behavior in dry forests, and lack of development of diverse early-seral vegetation as a result of fire suppression in drier parts of moist forests. Climate change and invasive species have emerged as threats to native biodiversity, and expansion of the wildland-urban interface has limited the ability of managers to restore fire to fire-dependent ecosystems.

The policy, social, and ecological contexts for the NWFP have changed since it was implemented. The contribution of federal lands continues to be essential to the conservation and recovery of fish listed under the Endangered Species Act and northern spotted owl and marbled murrelet populations. Conservation on federal lands alone, however, is likely insufficient to reach the goals of the NWFP or the newer goals of the 2012 planning rule, which emphasizes managing for ecosystem goals (e.g. ecological resilience) and a few species of concern, rather than the population viability of hundreds of individual species.

The social and economic basis of many traditionally forest-dependent communities has changed in 24 years, and many are now focused on amenity values. The capacities of human communities and federal agencies, collaboration among stakeholders, the interdependence of restoration and the timber economy, and the role of amenity- or recreation-based communities and ecosystem services are important considerations in managing for ecological resilience, biodiversity conservation, and social and economic sustainability.

A growing body of scientific evidence supports the importance of active management or restoration inside and outside reserves to promote biodiversity and ecological resilience. Active management to promote heterogeneity of vegetation conditions is important to sustaining tribal ecocultural resources. Declines in agency capacity, lack of markets for small-diameter wood, lack of wood processing infrastructure in some areas, and lack of social agreement have limited the amount of active management for restoration on federal lands. All management choices involve social and ecological tradeoffs related to the goals of the NWFP. Collaboration, risk management, adaptive management, and monitoring are considered the best ways to deal with complex social and ecological systems with futures that are difficult to predict and affect through policy and land management actions.

Keywords: Northwest Forest Plan, science, management, restoration, northern spotted owl, marbled murrelet, climate change, socioeconomic, environmental justice.

Preface

In 2015, regional foresters in the Pacific Northwest and Pacific Southwest Regions of the USDA Forest Service requested that the Pacific Northwest and Pacific Southwest Research Stations prepare a science synthesis to inform revision of existing forest plans under the 2012 planning rule in the area of the Northwest Forest Plan (NWFP, or Plan). Managers provided an initial list of hundreds of questions to the science team, which reduced to them to 73 questions deemed most feasible for addressing through a study of current scientific literature. The stations assembled a team of 50 scientists with expertise in biological, ecological, and socioeconomic disciplines. At the suggestion of stakeholders, a literature reference database was placed online so the public could submit additional scientific literature for consideration. By spring 2016, writing was underway on 12 chapters that covered ecological and social sciences.

The draft synthesis, which was ready for peer and public review by fall 2016, went through a special review process because it was classified as "highly influential science" in accordance with the Office of Management and Budget's 2004 "Final Information Quality Bulletin for Peer Review." The synthesis was classified as such because it fit the category of a scientific assessment that is novel, controversial, or precedent-setting, or has significant interagency interest. Per the bulletin, the two research stations commissioned an independent entity, the Ecological Society of America (ESA), to manage the peer-review process, including the selection of peer reviewers.

The bulletin also stipulates that such an assessment be made available to the public through a public meeting to enable the public to bring scientific issues to the attention of peer reviewers. Accordingly, a public forum was held in Portland, Oregon, in December 2016. For those who could not travel to Portland, the forum was accessible via live Web stream, and multiple national forests within the NWFP area hosted remote viewing. Written comments on the draft synthesis were collected for 2 months. This generated 130 public comments, totaling 890 pages, which were given to the peer reviewers for consideration in their review, as they deemed appropriate. The OMB guidelines further direct that the peer-review process be transparent by making available to the public the ESA's written guidance to the reviewers, the peer reviewer's names, the peer review reports, and the responses of the authors to the peer reviewer comments—all of which are available at https://www.fs.fed.us/pnw/research/science-synthesis/index.shtml.

The peer reviewer comments, which were received in spring 2017 and informed by public input, resulted in substantive revisions to chapters of the synthesis. The result is this three-volume general technical report (an executive summary of the synthesis is available as a separate report). This document is intended to support upcoming management planning on all public lands in the Plan area, but is expected to serve primarily lands managed by the U.S. Forest Service. We hope it will be a valuable reference for managers and others who seek to understand the scientific basis and possible tradeoffs associated with forest plan revision and management decisions. The synthesis also provides an extensive list of published sources where readers can find further information.

We understand that the term "synthesis" can have many different meanings. For our purposes, it represents a compilation and interpretation of relevant scientific findings that pertain to key issues related to the NWFP that were identified by managers and by the authors of the document. Such a compilation not only summarizes science by topic areas but also interprets that science in light of management goals, characterizes competing science, and makes connections across scientific areas, addressing multilayered and interacting ecological and socioeconomic issues. In a few cases, simple analyses of existing data were conducted and methods were provided to reviewers.

The synthesis builds upon the 10-, 15-, and 20-year NWFP monitoring reports, and authors considered well over 4,000 peer-reviewed publications based on their knowledge as well as publications submitted by the public and others suggested by peer reviewers. For some of the questions posed by land managers, there was ample scientific research from the Plan area. For many of the questions, however, little research existed that was specific to the area. In such cases, studies from other regions or current scientific theory were used to address the questions to the extent possible. In many cases, major scientific uncertainties were found; these are highlighted by the authors.

The synthesis chapters characterize the state of the science but they do not develop management alternatives, analyze management tradeoffs, or offer recommendations as to what managers should do. The synthesis does identify ideas, facts, and relationships that managers may want to consider as they develop plans and make management decisions about particular issues. The final chapter attempts to integrate significant cross-cutting issues, e.g., ecological and socioeconomic interdependencies, compatibility of different management goals, and tradeoffs associated with different restoration actions. All the chapters identify where more research is needed to fill critical information gaps.

We would like to acknowledge the peer reviewers who considered hundreds of public comments as part of the process of reviewing our lengthy draft manuscripts. We also thank the many contributors to the development of the synthesis in draft and final form, including those who provided editing, layout, database, and other support services.

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Koosah Falls, Willamette National Forest.
Photo by USDA Forest Service.

Chapter 1: Introduction

Peter A. Stine and Thomas A. Spies[1]

Background and Purpose of This Science Synthesis

We live in an era of information. Although this brings many benefits to society, it creates challenges for those responsible for understanding and applying new and older information to their day-to-day work. How does one keep up with the volume of relevant information that is published daily?

People who manage the 24 million ac (9.7 million ha) of public land within the area of the Northwest Forest Plan (NWFP, or Plan) depend on sound scientific knowledge about ecological systems and about how they function and how they respond to change. The Plan area stretches from Washington's northern border to a significant portion of northern California, encompassing diverse geography, ecological systems, and human communities. The authors of the NWFP understood that scientific knowledge would be critical to the efficacy of the plan, both in preparation of plan guidance and in learning how affected forests and communities (i.e., socio-ecological systems) would change over time, with and without active management. Current direction to national forests that are undertaking forest plan revisions also specifically calls for sound scientific information to guide plan preparation and to make selected changes to how forests might be managed in the future. Land managers responsible for updating forest plans find it challenging to remain current with all the new scientific knowledge. For a geographic region as large, diverse, and complex as the Plan area, this presents one of the greatest challenges to plan preparation and execution.

The majority of public lands within the NWFP area are managed by the U.S. Forest Service. This includes roughly 19.2 million ac (7.68 million ha) on 17 national forests (the Deschutes, Fremont-Winema, Gifford Pinchot, Klamath, Lassen, Mendocino, Modoc, Mount Baker–Snoqualmie, Mount Hood, Okanogan-Wenatchee, Olympic, Rogue River–Siskiyou, Shasta-Trinity, Siuslaw, Six Rivers, Umpqua, and Willamette National Forests). There are also roughly 2.5 million ac (1 million ha) of U.S. Department of the Interior Bureau of Land Management (BLM) lands and roughly 2.3 million ac (0 .92 million ha) of National Park Service lands within the Plan area. This synthesis is intended to support upcoming management work on all public lands, but is expected to serve primarily Forest Service lands and their impending forest plan revisions. In 2016, the BLM revised its resource management plans for its lands in western Oregon. Although the BLM and Forest Service are using distinct and separate planning processes to revise land use plans within the Plan area, the two agencies share common goals for long-term monitoring of the impacts of the implementation of their land use plans.

To help meet the challenge of forest plan revision, this science synthesis provides a comprehensive overview of the full body of relevant science accumulated in the 24 years since the NWFP was initiated. The synthesis was developed at the behest of the Pacific Southwest and Pacific Northwest Regions (Forest Service Regions 5 and 6). To accomplish this task, the Pacific Northwest (PNW) Research Station and the Pacific Southwest (PSW) Research Station assembled a team of scientists who are experts in a variety of biological, ecological, and socioeconomic disciplines.

The term "synthesis" can have many different meanings. For our purposes, it is a compilation of relevant scientific findings that pertain to key issues around the NWFP. Such a compilation not only summarizes science by topic areas but also makes connections across scientific themes and addresses multilayered and interacting natural and socioeconomic resource issues. This report has been prepared to assist land managers in updating existing forest management plans and on-the-ground projects. Our hope is that it will serve as a reference that provides a condensed and integrated understanding of the current state of knowledge regarding the NWFP, as well as an extensive list of published sources, where readers can find further information.

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This synthesis is not a bibliography or an interpretation of all available science; and is not intended to direct management through recommendations or analysis of management alternatives. In contrast, the charge given to the scientists who served as members of the Forest Ecosystem Management Assessment Team (FEMAT) under the original NWFP (FEMAT 1993) requested that scientists assess the science and use their expert knowledge to develop a set of plan alternatives and corresponding management recommendations. President Bill Clinton selected and adapted one of these plan alternatives, which formed the basis of the standards and guides for the NWFP. This science synthesis provides a summary and interpretation of relevant science findings to support subsequent planning efforts under Forest Service regulations.

Our approach largely follows the role of “science arbiters,” one of the four roles that scientists can play in policy arenas (Pielke 2003). Science arbiters answer questions from managers from a scientific perspective (e.g., What are the ecological differences between dry forests and moist forests, or what is known about the ecological effects of different restoration strategies?). But they do not develop or evaluate policy alternatives. We do not play an alternative role of “honest brokers of policy alternatives” who develop a wide range of policy alternatives and characterize their possible consequences using scientific findings and expert opinion. That was the role that the scientists in FEMAT played. Although this synthesis does not develop plan alternatives or evaluate them, it does characterize what is known about the ecological effects of various management practices (e.g., salvage logging or prescribed fire), and it identifies ecological and socioeconomic tradeoffs associated with different management goals (e.g., ecosystem integrity vs. single species) and practices. We also characterize how well the NWFP has met some of its original goals by using information from the monitoring programs and peer-reviewed published sources.

The synthesis builds upon the 10-, 15-, and 20-year NWFP monitoring reports and it considered well over 4,000 peer-reviewed publications. The authors of individual chapters have extensive knowledge of the scientific literature, and much of what was reviewed comes from their knowledge of the most relevant work. As part of this review process, we also established a Web portal to enable members of the public to offer appropriate literature that they wanted to ensure would be included in the review. We provided a comprehensive summary of the scientific literature that we considered salient to the key issues to be addressed by land managers as they begin considering forest plan revision.

The breadth of topics and number of scientific papers that could be covered in this synthesis is enormous. At the direction of Regions 5 and 6, we focused on topics that had a direct bearing on activities that resulted from the NWFP and subsequent forest plan revision. Focal topics were distinguished from a large set of management questions identified by Forest Service management staff in the two regions. The core author team worked with Forest Service managers to condense the initial set of questions to 73 (see app. 1). The final list was established by removing questions that were outside the scope of this effort (including those that could not be addressed by published scientific information or were not relevant to the NWFP), then identifying only those topics that could be addressed by reviewing the evidence contained in the scientific literature (i.e., at least some scientific information exists that would enable some insight on the question). The final questions were grouped into four main categories (Vegetation/Forest Management, Terrestrial Species/Habitat Management, Aquatic/Riparian Management, and Social/Economic, including Timber Production), which formed the basis for the organization of the synthesis. Lead authors used these questions to build chapter outlines and provide useful information to support subsequent management planning efforts.

The authors of the chapters address the management questions using a range of approaches. In some cases, there is ample scientific evidence from the Plan area to address the questions; however, in many cases, few research studies exist from the NWFP area. In such cases, studies from other regions or current scientific theory are used to address the questions to the extent possible. In many cases, major uncertainties are identified, while in others much uncertainty remains. The following chapters provide comprehensive reviews of the relevant scientific literature within their topic areas, but the authors do not evaluate tradeoffs among different resource management and planning objectives. Chapter 12, however, addresses the most significant integration issues as well as potential tradeoffs to identify where additional evaluation or more monitoring/research will be necessary in subsequent assessments and planning efforts to resolve potential or existing conflicts.

Northwest Forest Plan History and Context

The NWFP is rooted in the environmental history of the region and followed a series of ecological and socioeconomic triggers in the 1980s and early 1990s (Johnson and Swanson 2009). Historically, the ecosystems of this region have been influenced by many tribes of native people for millennia (see chapter 11). More than two centuries ago, their civilizations and stewardship of the ecosystems of the region were greatly affected by visitors and settlers from the Eastern United States or from European countries, and the United States gradually seized or acquired lands from tribes, converting much of the forested area into farmlands, industrial timberlands, and other new land uses. By the beginning of the 20th century, large tracts of forest lands in the Western United States were put into "forest reserves" and managed by the U.S. Forest Service to protect watersheds and ensure a continuous supply of timber. The initial reserve era gave way to the era of sustained-yield forestry to support economic growth (Steen 2004). These practices continued into the 1970s, when three significant federal laws were passed: the National Environmental Policy Act (NEPA) of 1970, the Endangered Species Act (ESA) of 1973, and the National Forest Management Act (NFMA) of 1976. Collectively, these laws engendered an era of increasing environmental awareness and concern. During the next two decades, the stage was set for conflict between timber-focused policies and the emerging public concern over the environmental impacts of forest management practices in the Northwest. By 1990, conservation of biodiversity had ascended to become a new priority for federal forests, and numerous organizations stepped in to initiate litigation, which ultimately led to establishment of the NWFP in 1994 (Johnson and Swanson 2009).

The NWFP was a product of many social and ecological drivers, but the focal point of the deliberations was the protection of the old-forest ecosystems that provide habitat for northern spotted owls (*Strix occidentalis caurina*). The Plan also addressed the needs of the marbled murrelet (*Brachyrampus marmoratus*), anadromous fish, and other species associated with older forests, as well as stressing the importance of sustaining rural communities and economies through continued timber harvest (Charnley 2006). There are many alternative views and definitions of "old growth" (chapter 3) (Haynes et al. 2006). For the sake of simplicity, we use only the term "old-growth forests" in this introduction.

The 1980s were part of a transformative period for the Pacific Northwest and northern California (Johnson and Swanson 2009). For many years, timber harvest was extensive across the region, and concerns about the effects that the logging of old growth had on wildlife and riparian areas grew steadily into the early 1990s. The 1990 listing of the northern spotted owl as a threatened species precipitated numerous legal challenges regarding the cumulative impacts of federal timber management in the Pacific Northwest and northern California. When a federal court issued an injunction in 1991 on all timber sales on federal lands within the range of the northern spotted owl, the political and environmental landscape shifted substantially. The ensuing political crisis set the stage for the emergence of the NWFP.

These dramatic events and emerging science precipitated federal government engagement, up to and including the White House, to seek a workable solution. Over the next 2 years, beginning in earnest with the Northwest Forest Summit in 1993, the federal government forged a plan. The extensive involvement of the White House and principal land management agencies (i.e., the Forest Service and BLM) led to the 1994 adoption of the NWFP by the Clinton Administration (Pipkin 1998).

The Forest Ecosystem Management Assessment Team

President Clinton established three interagency working groups to build a foundation for what would ultimately become the NWFP. One of these groups was FEMAT, a team of scientists, resource managers, and technicians from many different universities and public agencies, charged with identifying management alternatives that could attain the greatest economic and social contribution from forests, while meeting all applicable laws and regulations (FEMAT 1993). Specifically, FEMAT was asked to consider and develop conservation approaches, restoration actions, and adaptive management strategies to meet the following biological diversity goals: (1) habitat for the northern spotted owl and marbled murrelet, (2) habitat for other species associated with old growth, (3) spawning and rearing habitat for anadromous fish, and (4) maintenance of a connected old-growth forest reserve system on federal lands.

FEMAT issued an extensive report (FEMAT 1993) that analyzed the ecological, social, and economic implications of 10 management options for the federal forests within the range of the northern spotted owl. The team used expert opinion to assess biophysical processes and disturbances, community capacity, and economic factors, and it estimated tradeoffs and risk to species associated with different levels of protection for biodiversity and timber production. This was, and may still be, the most extensive regional forest biodiversity and management assessment of its kind. Many of today's persistent policy challenges were raised and considered 24 years ago in this report. The FEMAT report identified risk and uncertainties associated with the different conservation and management issues and recognized that monitoring and adaptive management would be needed to maintain a long-term, scientifically based and adaptive plan. This synthesis summarizes published research, monitoring and knowledge of plan implementation over the past 24 years, providing a current scientific foundation for forest planning.

Principal Elements of the NWFP

Conservation and management of old-growth forests are central to the NWFP and the past 24 years of its implementation. As readers consider the various chapters in this synthesis, they will see that old-growth forests have both an ecological and a social dimension. These dimensions can be linked, but also can emerge in quite different contexts. We address and discuss these facets in the following chapters.

The principal tasks of the NWFP were to conserve and restore habitats for animals and plant species associated with old-growth forests and maintain and restore habitat for anadromous fish within the confines of existing laws and regulations (e.g., NFMA and ESA). Management of the affected 24 million ac (9.7 million ha) of land was altered significantly to meet these new biological diversity goals. At the time, relatively little was known about most species associated with late-successional and old-growth forests, and this is still the case. Although the biology and ecology of the northern spotted owl were relatively well understood, there were many gaps in our understanding of this long-lifespan species with a low reproductive rate. The major shift in federal forest management was part of a larger global trend toward increasing protection for the forest biodiversity through a process called "ecosystem management" (Grumbine 1994). As Chuck Meslow, then leader of the Oregon Cooperative Wildlife Research Unit at Oregon State University, explained, the NWFP originated at a time when many scientists were beginning to advocate for a more ecological approach to managing remaining old-growth forests (FEMAT 1993).

The intent of ecosystem management, as it was initially envisioned at the time, was to sustain ecosystems by maintaining (1) viable populations of native species, (2) native ecosystem types, and (3) evolutionary and ecological processes over long time horizons (Grumbine 1994). In doing so, it was posited that such a management regime would accommodate human use and occupancy within the capacities of ecosystems. The NWFP changed federal management by giving priority to ecological sustainability; the team was directed to plan for social and economic values **after** meeting ecological objectives. The hope was that the Plan could find common ground through the right balance of biodiversity and timber management objectives (Charnley 2006).

The NWFP evolved out of three preceding efforts in the early 1990s to find a solution to the conflicts over federal forest management (Thomas et al. 2005): (1) a conservation strategy for the northern spotted owl (Thomas et al. 1990), (2) "Gang of Four" report on alternatives for management of Pacific Northwest late-successional forests for multiple species (Johnson 1997, Johnson et al. 1991), and (3) the Scientific Analysis Team (known as the SAT) report, which conducted a scientific analysis that added riparian protection and more species to the assessment. (Thomas et al. 1993). These efforts laid the foundation for much of the NWFP. FEMAT, established by the president, used this and other sources of information to develop options that would (1) consider human and economic dimensions of the problem; (2) protect the long-term health of forests, wildlife, and waterways; (3) be scientifically sound, ecologically credible, and legally responsible; (4) produce a predictable and sustainable level of timber sales and nontimber resources that would not degrade the environment; and (5) emphasize collaboration among the federal agencies responsible for management of these lands (Thomas et al. 2005).

FEMAT developed 10 options for the president and agency heads to consider. They selected option 9, which was based on both ecosystem- and species-level conservation and restoration strategies. This option was subsequently modified to meet viability requirements under NFMA during the final environmental impact statement process, and the final plan was set forth in the record of decision (ROD), with the following key elements:

- Adoption of a yet-to-be-defined **ecosystem management approach**
- **Seven land allocations** (see fig. 1-1) to address key conservation/management concerns, including:
    - Congressionally reserved areas (7.3 million ac/2.95 million ha)
    - New late-successional reserves (7.4 million ac/2.99 million ha)
    - New adaptive management areas (1.5 million ac/607 000 ha)
    - New managed late-successional areas
    - Administratively withdrawn areas
    - New riparian reserves (2.6 million ac/1 million ha)
    - Matrix (for ecologically sensitive timber production) (nearly 4 million ac/1.6 million ha)
- An emphasis on **effective consultation with more than 70 federally recognized tribes** to avert conflicts with American Indian trust resources on public lands and exercise of tribal treaty rights.
- **Standards and guidelines** that provided detailed requirements describing how land managers would treat forest lands within the range of the northern spotted owl.
- A **new monitoring program** consisting of implementation monitoring (are the standards and guidelines being followed?) and effectiveness monitoring (is the plan having the desired effect?).
- **"Survey and manage" measures to provide for other late-successional species** that may not be covered under the conservation strategies for the spotted owl and marbled murrelet, and for aquatic ecosystems and old-growth forests.

Reserves are a key component of the terrestrial and aquatic components of the NWFP and are discussed at length in chapters 3, 4, 5, 7, and 12. Reserves were intended to provide immediate and wide-ranging benefits for target species (e.g., spotted owls) and target ecosystems (old-growth forests, streams). Reserves were carefully delineated across the Plan area with the intention of improving ecological conditions for key Plan elements such as spotted owls or anadromous fish. We use monitoring results to evaluate how those conditions have changed and how well the underlying goals of the Plan have been met.

The ROD for the NWFP amended the planning documents for 19 national forests.[2] It is important to recognize that, over the past 24 years, implementation of the Plan across the entire area has varied from location to location. This can be attributed to geography and variation in how planning standards and guidelines have been interpreted by different forests, districts, and personnel over time. This is inevitable given the challenges of implementing a complex land management plan across a broad and diverse geography. The monitoring data we used to evaluate

[2] The Northwest Forest Plan area currently includes 17 national forests; in 2000, the Okanogan and Wenatchee National Forests administratively merged as the Okanogan-Wenatchee National Forest, and in 2002 the Fremont and Winema National Forests administratively merged as the Fremont-Winema National Forest. The Plan area also includes five Bureau of Land Management districts and one resource area (formerly six districts and one resource area), with extensive standards and guidelines that comprised a comprehensive ecosystem management strategy.

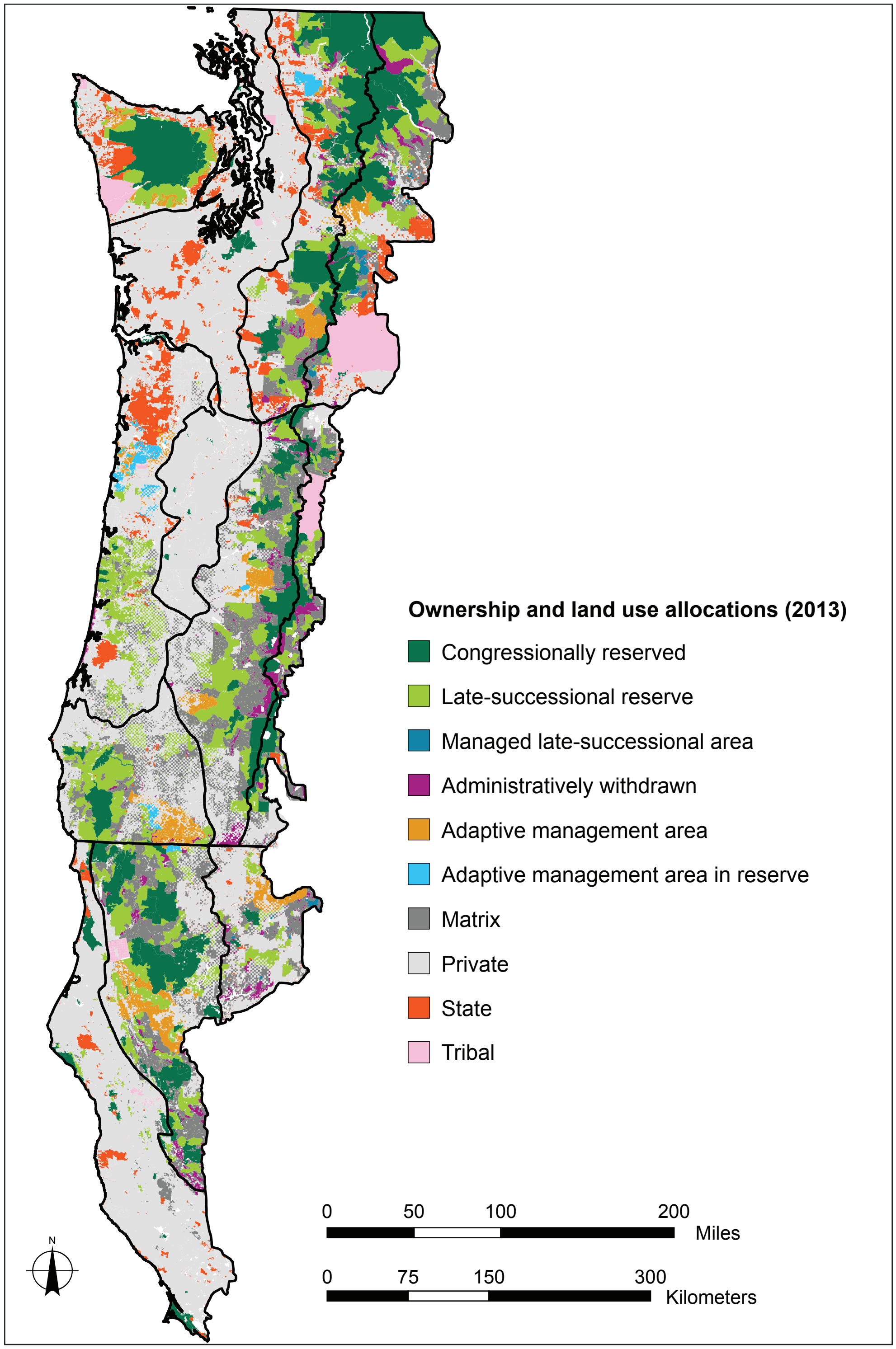

Figure 1-1—Land allocation categories and original 12 physiographic provinces (outlined in black) for the Northwest Forest Plan area. Note that "matrix" includes riparian reserves and other unmapped buffers (e.g., Survey and Manage).

the NWFP are regional in scale and may not capture variability in Plan effects. In addition, unlike the effectiveness monitoring program, the implementation monitoring program has not been continued, making it difficult in some cases to determine what has actually occurred. The limits of the monitoring programs mean that some of our characterization of the Plan may not be correct.

Decisionmakers considered monitoring to be an essential component of the selected alternative. Monitoring was intended to provide information to determine if standards and guidelines were being followed (implementation monitoring) and to verify if they were achieving desired results (effectiveness monitoring). In addition, a third type of monitoring, validation monitoring, was identified as a way to determine if underlying assumptions of the Plan were sound (this monitoring program was never formally established). The monitoring plan was subsequently cited by U.S. District Court Judge William Dwyer in his ruling upholding the Plan after challenges from the timber industry. The judge ruled that monitoring was a key element of the Plan and was essential to its success. Information obtained through monitoring, together with new research and experience gained through implementation, would provide the basis for adapting the Plan in the future (USDA 1994).

History of Reporting on the Research and Monitoring Within the NWFP Area

The NWFP involved the scientific community, through research and monitoring, in ways and to lengths not used before in Forest Service planning and management. The NWFP was driven, in large part, by a requirement to meet certain standards under the ESA and the viability clause of the NFMA, as well as by changes in land management related to three other federal laws (Thomas et al. 2006). These circumstances quickly triggered the need to engage scientists from the beginning, to provide both the planning and implementation process with robust, reliable scientific information.

The record of decision included the requirement of a detailed monitoring plan to ensure that management actions meet the prescribed standards and guidelines, and that actions complied with applicable laws and policies. Information obtained through monitoring, together with new research and information from adaptive management areas and studies, were intended to provide a basis for changes to the Plan, including changes to the standards and guidelines. Although a formal validation monitoring program was never established, research activities were conducted to help testing of hypotheses related to NWFP goals.

10-, 15-, and 20-Year Monitoring Reports

The NWFP was designed to include an adaptive management approach to enable “learning from doing.” The record of decision called for gathering information through an extensive monitoring effort, together with targeted new research and other new sources of information, to provide a basis for adaptive management and updating the selected alternative with new scientific knowledge. This set lofty aspirations for the scientific rigor of the Plan; however, there has been little adaptive management work done (i.e., actual designed experiments to test management strategies and assumptions in designated AMAs) since the Plan was initiated.

Monitoring was designed for data collection at multiple scales, ranging from site-specific projects to the regional-scale planning area, to allow localized information to be compiled and considered in a regional context. Many but not all of the data sources used in the 20-year reports were initially developed and used for the 10- and 15-year monitoring reports. During each 5-year monitoring cycle, previously used data sources are updated to incorporate new research findings and other information, or to correct errors or previous misconceptions. So, to the extent possible, results are comparable between the two major reporting periods, but caution is suggested when examining topics that relate findings from one time period to the next because of minor analytical or reporting differences between monitoring reports.

Monitoring results have been evaluated and reported in 1- and 5-year intervals since the inception of the NWFP. The first comprehensive analysis of 10 years of NWFP monitoring data was published in a series of general technical reports (GTRs) summarizing what had been learned over that time. This was an important first step in adaptive management. The 10-year report synthesized the status and trends of five major elements of the plan: old-growth forests, old-growth forest species at risk, aquatic systems, socio-economics, and adaptive management (Haynes et al. 2006). It also synthesized the new science that resulted from 10 years of research related to the Plan. At this time, the cadre of researchers and managers also addressed four additional interconnected questions:

1. Has the NWFP resulted in changes that are consistent with objectives identified by President Clinton?
2. Are major assumptions behind the Plan still valid?
3. Have we advanced learning through monitoring and adaptive management?
4. Does the Plan provide robust direction for the future (Haynes et al. 2006)?

Based on the first 10 years of data collection, findings were ambiguous and conclusions hard to reach—perhaps unsurprisingly for a plan that was expected to take 100 years to achieve its goals. It was clear that the complexity of ecosystem interactions and the effects of new drivers (e.g., encroachment of barred owls, climate change, and changes in social values) were far greater than had been envisioned 10 years earlier. Nonetheless, insights into ecosystem response began to emerge, including circumstances and ecological interactions not contemplated at the time the Plan began. Rapp (2008) provided some highlights of the first decade of monitoring and research as follows:

- Nearly all existing old-growth forest on federal land was protected from timber harvest (although 100-percent protection was not part of the original plan).
- Old-growth forest on federal land had an estimated net increase of roughly 1.2 million ac (~480 000 ha), increasing from 7.87 million ac (3.15 million ha) to 9.12 million ac (3.65 million ha) in the first 10 years as a result of accretion by growth.
- Despite protection of northern spotted owl habitat on federal land, spotted owl populations declined at a greater rate than expected in the northern half of their range, likely because of barred owl competition, and losses of habitat to wildfires.
- Watershed condition improved slightly because of reduced harvest in riparian areas, tree growth, and increased emphasis on restoration.
- Federal timber harvest in the NWFP area was only 54 percent of the level set by the Plan's goals.
- In spite of mitigation measures, most local communities near federal lands suffered significant job losses and other adverse effects.
- State, federal, and tribal governments worked together on forest management issues more effectively than in the past.
- Increased collaboration with communities changed how the agencies get work done.

Recently, reports analyzing a full 20 years of monitoring data under the NWFP were released by the Regional Interagency Executive Committee and published as GTRs (Davis et al. 2015, 2016; Falxa and Raphael 2016; Grinspoon et al. 2016; Miller et al. 2017). These reports summarize the latest periodic monitoring data gathered since 1994, with a focus on the past 5 years. Some of the key findings contained in these new reports include:

- Overall late-successional and old-growth habitat area has decreased 3 percent on federal lands, with the biggest losses resulting from wildfires. However, this rate of loss was in line with expectations outlined in the FEMAT report during the design of option 9.
- Nesting habitat of the marbled murrelet showed a net decrease of about 2 percent on federal lands and 27 percent on nonfederal lands.
- In Washington, there was an annual rate of decline of 4.6 percent in the population of marbled murrelets between 2001 and 2013; a cumulative decline over 10 years of 37.6 percent. Populations had no detectable trends in Oregon and California.
- The forest types suitable for nesting and roosting for northern spotted owls on federal lands decreased by 1.5 percent since inception of the NWFP. Forest succession is resulting in habitat recruitment that has compensated for losses resulting from wildfire, timber harvest, and insects and disease. However, suitable habitat (i.e., the full range of conditions necessary for a species to survive, persist, and reproduce) has declined more because of the influx of barred owls into forests with otherwise suitable forest vegetation throughout much of the range of

spotted owls. Recent northern spotted owl research indicates that populations are declining throughout the range of the subspecies, and that annual rates of decline are accelerating in many areas. Dugger et al. (2016) observed strong evidence that barred owls negatively affected spotted owl populations, primarily by decreasing apparent survival and increasing local territory extinction rates. The amount of suitable owl habitat, local weather, and regional climatic patterns also appear to be related to demographic parameters, including survival, occupancy (via colonization rate), recruitment, and, to a lesser extent, fecundity (Dugger et al. 2016).

- The attributes of watershed conditions (in-channel physical habitat, macroinvertebrates, and water temperature) showed slight improvements, but uncertainties in the trends of overall conditions remain. Upslope and riparian areas showed moderate, broad-scale improvements in vegetation structure and larger score increases from road decommissioning in a number of watersheds. In the regional average, these increases were largely offset by declines in scores because of fires, particularly on congressionally reserved lands.
- Timber volume harvested has fluctuated over the past 20 years. The volume of timber offered has been on a general upward trend since 2000, with volume offered in 2012 at about 80 percent of probable sale quantity (PSQ) identified in the NWFP (based on revisions to the original PSQ of 1.1 billion board feet, as stated in the ROD, to a PSQ in 2012 of about 805 million board feet).
- Rural communities are not all alike, forest management policies affect different communities differently, and the social and economic bases of many traditionally forest-dependent communities changed in the years since the start of the NWFP.
- Federal-tribal relations are more effective and meaningful when there is common understanding of consultation, tribal rights, federal trust responsibilities, and compatibility of tribal and federal land management.

Scope and Approach of This Science Synthesis

The PNW Research Station partnered with the PSW Research Station to prepare this synthesis, which was initiated at the request of Forest Service land managers. The two station directors guided this effort, and the day-to-day activities were led by Thomas Spies and Peter Stine. Other core team members included Matthew Reilly, Jonathan Long, and Becky Gravenmier. The core team, in consultation with the station directors, identified a group of experienced, knowledgeable scientists to serve as lead chapter authors. This put the responsibility for each chapter in one place and ensured that we would draw upon highly qualified sources.

The public has expressed interest in this synthesis, given the importance of the NWFP in the management of Northwest forests and its influence on forest management approaches around the world. During listening sessions held in spring 2015 to gather feedback from the public about forest plan revisions, attendees provided suggestions relevant to the development and publication of this science synthesis. We heard many participants express a desire for continuous communication about the science, more access to scientific information, and participation in a greater variety of information-sharing venues. A number of steps were taken to enhance public input into this process, including a Web portal for submitting literature for consideration in the synthesis, and a public forum to accept oral and written public input to the peer review team.

Rationale for Topics Covered

Questions from managers guided the focus of the synthesis. The set of 73 management questions were grouped into the following major headings:

- **Vegetation conditions**, including forest management/climate change/ecological disturbance effects on old growth and other vegetation types.
- **Terrestrial species**, including habitat management for the northern spotted owl; marbled murrelet; and other plant, plant-ally, invertebrate, and vertebrate species, and conservation of the biodiversity associated with old-growth forests.

- **Aquatic/riparian management**, including aquatic and riparian species and ecosystems.
- **Socioeconomic well-being**, including timber production, collaborator and stakeholder attitudes, and tribal values and resources.
- **Integrated topics:** themes that cross over between chapters or separate management activities.

This synthesis is organized into 12 chapters, in three volumes, that include an introduction, 10 chapters addressing the primary topics of concern, and a final "integration" chapter that ties together what has been learned and reported in the various chapters and conveys how this synthesized knowledge bears on vital forest management activities. Each chapter provides a summary of the relevant scientific literature, lessons learned over the past 20 years, and the relevance of these findings to management. The synthesis does not provide management recommendations, nor does it conduct assessments of likely outcomes of different approaches to plan revisions.

Sources of Information Considered

This science synthesis considered science published by peer-reviewed scientific or professional journals, or reviewed through an agency-sponsored, third-party process that meets the general criteria for competent and credible peer review. This process collected material from many sources, including an extensive body of original research and monitoring activities). In addition, academic theses, government reports, symposium proceedings, and the like may have been used to support certain topics that were not adequately covered in the peer-reviewed literature. Most of the literature considered was compiled by the authors based on their experience with the subject matter. In some cases, especially in chapter 3 ("Old Growth, Disturbance, Forest Succession, and Management in the Area of the Northwest Forest Plan"**),** some simple analyses of existing data were conducted to illustrate key ideas. Through a Web portal developed specifically for this purpose, we also provided opportunities for the public to suggest literature sources that we may not have already considered. A "Science Synthesis Literature Database" (https://www.fs.fed.us/pnw/research/science-synthesis/literature-database.shtml) for the NWFP area lists all publications reviewed in this report, including many recommended by the public.

Dealing With Scientific Uncertainty

There is always some degree of uncertainty embedded in scientific findings, especially related to our understanding of large and complex socio-ecological systems. The scientific literature in the fields covered by this synthesis does not necessarily address specific questions that land managers posed. Accordingly, chapter authors selected from a wider range of published research in an effort to reduce this uncertainty. To do so, we made judgments based on scientific consensus about how the findings of different scientific reports related to management questions, what the uncertainties are within published reports, and what the uncertainties are related to our interpretation of multiple reports. We report what is known about these topics with high confidence whenever possible, and describe what issues remain uncertain.

In the FEMAT report, an expert evaluation process was used to address gaps in the scientific literature, as well as limits to our understanding, to better estimate the likely outcomes and risks to biodiversity associated with different conservation and management options and practices. FEMAT convened panels of scientific experts to rate the probabilities of viability outcomes for components of the Plan (such as northern spotted owls and aquatic functions) for the different Plan options. Although the FEMAT results and recommendations represented a consensus of scientific knowledge at the time, they contained considerable uncertainties, thus monitoring and adaptive management were regarded as being critical to the Plan's scientific basis. This synthesis does not rely on an expert judgment process to fill large information gaps related to management questions or Plan trends. For example, we do not rate the probability of the long-term viability of the northern spotted owl in light of threats from barred owls or climate change. Although we use expert knowledge to interpret existing science, we avoid speculation about outcomes related to management effects, climate change, or other drivers or threats for which there is no published science. In this sense, the synthesis is more limited in scope than FEMAT was in the interface between science and policy. The process of assessing Plan alternatives, developing revisions to the standards and guidelines, or choosing actions in the face of uncertainties will be handled by federal land managers in subsequent steps of the upcoming planning precess. We report what is known to apprise managers of the best available scientific information and allow them to apply that information to their management concerns.

Role of Peer Review in This Document

Unlike FEMAT, the science synthesis has been subject to external peer review and revision based on those reviews. The Office of Management and Budget (OMB) explained the importance of peer review in its *Information Quality Bulletin for Peer Review*[3] as follows:

> Peer review is one of the important procedures used to ensure that the quality of published information meets the standards of the scientific and technical community. It is a form of deliberation involving an exchange of judgments about the appropriateness of methods and the strength of the author's inferences. Peer review involves the review of a draft product for quality by specialists in the field who were not involved in producing the draft.

The OMB guidelines require that influential scientific information developed by a federal agency be subjected to formal, independent, external peer review to ensure its objectivity. Scientific knowledge is cumulative, building upon previous findings; therefore, safeguarding this trust is essential. Peer-reviewed science does not guarantee that what is presented is true or factual, because new information may overturn, refute, or refine previous findings. Peer-reviewed science is also not necessarily definitive because of the limitations of knowledge, current perspectives, and available studies. However, peer review is the standard within the scientific community for determining which findings meet and exceed adequate thresholds of scientific scrutiny. For these reasons, this science synthesis focused on material that has been peer reviewed and published in print or online.

Peer-reviewed published literature, however, is limited for some topics. For example, some social, economic, health, cultural, or highly specialized ecological topics tend to have less coverage in the peer-reviewed literature. To address such gaps, authors were given latitude to incorporate relevant scientific information from academic theses and other research subjected to some form of committee review. In some cases, analyses were done using existing data and with data sources identified and methods of analysis provided. For example, in chapter 3, we developed a new classification and map of NWFP fire regimes by synthesizing existing data on climate, lightning ignitions, potential vegetation types, and fire-history studies. In contrast, forest management strategies and plans such as the NWFP are generally not peer reviewed or based only on peer-reviewed information. National forest managers consider a host of other sources of information to inform their plan revisions and involve the public in forest plan development.

In general, the authors focused on peer-reviewed research that occurred in the synthesis area or in forest ecosystems with highly similar ecological or social conditions. Ecological and social research is always context-specific, thus we attempted to guard against use of overgeneralizations applied to areas apart from where the research was conducted. This can be especially true of the ecologically and socially diverse region of the NWFP. Scientific studies are often published with caveats about their spatial and temporal scale. However, many basic ecological processes are universal, thus we can apply some findings to other locations. Obviously, basic research cannot be conducted everywhere, so it is important to make prudent application of scientific findings from a given location to other areas. To address this challenge, the synthesis notes the extent and limitations of available information, especially by highlighting various research gaps.

This science synthesis has been identified as a "highly influential scientific assessment," in accordance with the OMB's 2004 peer-review bulletin (see footnote 3), which means that the information contained therein could have a large impact on the public or private sector, or be of

[3] https://www.gpo.gov/fdsys/granule/FR-2005-01-14/05-769.

significant interest to multiple agencies, or be controversial. For this report, we have employed an external peer-review process that includes multiple reviewers with relevant expertise and experience assigned to each of the chapters, and three reviewers who reviewed the entire document. The review was managed by the Ecological Society of America, which selected the review team from scientists with extensive experience and strong credentials, and managed the review process independently.

The peer-review team, led by the Ecological Society of America's director of scientific programs, Clifford Duke, was given basic instructions for conducting peer review in accordance with OMB direction for peer review of highly influential scientific assessments developed by federal agencies (USOMB 2002). Peer-review comments were delivered to the author team in March 2017, and authors used them to develop the final document. Authors also prepared reconciliation documents for each chapter explaining how all comments were used.

The NWFP Area

The establishment and implementation of the NWFP was unprecedented in many ways. Its geographic scope, breadth of topic areas, and long-term investment in monitoring and research all combined to set a new standard for large-scale land management.

The NWFP area covers 24 million ac (9.7 million ha) of federally managed land, extending from the Mendocino National Forest and Ukiah District of the BLM near the coast of northern California to the northern boundaries of the Mount Baker–Snoqualmie and Okanogan-Wenatchee National Forests on the Canadian border. The area spans almost 10 degrees of latitude and ranges from coastal rain forest landscapes to dry east-side pine forests. This expansive and diverse footprint created significant challenges for establishing management guidance and the scientific foundation needed to support it. By recognizing and embracing the variability of this landscape, NWFP managers intended for management efforts to be more nuanced and thus more effective at addressing particular features in any given area.

Ecogeographic Variability of NWFP Area

Efforts to classify and partition the natural world into component parts have been directed at many different levels of biological or ecological organization, from genes and species to communities and ecosystems (Grossman et al. 1998). The NWFP area spans many biological community and ecosystem types and disturbance regimes, and the Plan goals include conservation strategies that focus on ecosystems as well as individual species. It is vital that the application of scientific findings within the Plan area recognize this broad geographic and ecological diversity. This concern is addressed in several chapters in which ecogeographic variation is central to careful treatment of management challenges (e.g., chapter 2 on climate, chapter 3 on old-growth forest, and chapter 5 on northern spotted owls).

Climate, geology, disturbance, and topography all play important roles in controlling forest community patterns at regional scales in the Pacific Northwest (Barbour et al. 2007, Franklin and Dyrness 1973, Ohmann and Spies 1998). The relationships among environment, the biota, and disturbance differ across the region, making it precarious to extrapolate findings from one ecoregion to another. Kennedy et al. (2012) highlighted the importance of understanding the finer grain patterns of forest ecosystems within the NWFP area and their response to disturbances. This understanding is critical for delivering effective management insights across the many, sometimes subtly different, forest conditions distributed within the Plan area. The authors made a concerted effort to address this subject, as in chapter 12, "Integrating Ecological and Social Science to Inform Land Management in the Area of the Northwest Forest Plan."

The NWFP area was originally partitioned into 12 physiographic provinces (see fig. 1-1) based on recognized landscape subdivisions exhibiting different physical and environmental features (Thomas et al. 1993). The resulting breakdown of provinces reflected the regional distribution of major forest types (and state boundaries for management purposes).

A number of qualitative approaches to classifying geographic variation have been used, including Ecoregions of the United States (Bailey 2009) and the Holdridge life zones, as discussed in Lugo et al. (1999). Quantitative ecoregionalization approaches are also available (e.g., Hargrove and Hoffman 2004, Hessburg et al. 2000), but are less often adopted by land managers because of the long-standing habit of using the more qualitative schemes. It is noteworthy that the quantitative schemes show highly intuitive, spatially disjunct patterns of ecoregions, which are largely absent in the qualitative approaches, suggesting that early delineations of ecoregional boundaries are inadequate. The various qualitative methods for identifying ecological regions use macroclimatic conditions (climate unaffected by landform), and prevailing plant formations as the means for classification (Bailey 2009).

Vegetation classifications are a critical part of regional ecological characterizations. Vegetation can be classified based on successional potential (e.g., the late-successional vegetation that would develop in the absence of disturbance for a particular environment), or on current vegetation structure and composition. Both types of vegetation classifications are needed. The two Forest Service regions use different vegetation classification schemes (Region 6 uses potential vegetation, and Region 5 uses actual or current vegetation [cover types]) (chapter 3), which makes it challenging to conduct a seamless ecological assessment across the entire Plan area. For this synthesis, we used the Region 6 potential vegetation classification and developed a crosswalk for linking the two types of classifications.

We also now have access to ecological delineations that are more data-driven, using data models based on machine learning. An example is the habitat modeling developed for the northern spotted owl and contained within the recent recovery plan for this taxon (USFWS 2011). The effort, aimed at partitioning habitat in the range of the spotted owl (essentially the same as the NWFP area), used machine learning via MaxEnt (Phillips et al. 2006) to predict relative existing habitat suitability. Results of this data-driven effort provide a delineation of 11 "modeling" regions as oppossed to the 12 ecoregions originally described for the NWFP area. It is unclear how accurate these habitat suitability models are for predicting actual habitat suitability of different vegetation conditions for northern spotted owls. Barred owls, a significant component of current northern spotted owl habitat through much of its range, drastically complicate our ability to assess habitat suitability. Further work will be needed to understand spotted owl response in the different habitat regions delineated by this modeling work.

Regardless of how this large Plan area is dissected, it is increasingly clear from recent scientific work that geography matters. The diversity of the NWFP landscape is both stark and subtle. We draw more specific attention to this issue throughout the following chapters.

Other Syntheses Reports Relevant to the NWFP Area

The effectiveness of the NWFP was originally evaluated through a set of reports produced 10 years after its initiation (Haynes et al. 2006). This set included a series of status and trends reports, a synthesis of all regional monitoring and research results, a report on interagency information management, and a summary report. Although some existing science was synthesized in the 2006 report, it was not a comprehensive characterization of the literature and did not address a special set of questions posed by managers. Updated monitoring reports were produced in 2009 and 2015 that evaluated the first 15 and 20 years of monitoring data developed under the NWFP (Davis et al. 2015, and others). Each of these monitoring reports included key summaries of the results for each monitoring module, methods, and a set of recommendations for monitoring into the future. These monitoring reports did not include a broader evaluation of the scientific literature.

Other efforts have been made in recent years to consolidate relevant scientific information within the Plan area. Notably, the Forest Service published *The Ecology and Management of Moist Mixed-Conifer Forests in Eastern Oregon and Washington: a Synthesis of the Relevant Biophysical Science and Implications for Future Land Management* (Stine et al. 2014). This synthesis overlapped with the NWFP area along the east Cascades of both Oregon and Washington and addressed some similar land management issues.

Role of Science in Supporting Land Management

This synthesis will inform the development of revised land and resource management plans for 17 national forests by synthesizing relevant information on key topics and management questions across the NWFP area. The synthesis will directly support land managers' ability to make decisions grounded in the best available science, and will provide managers with the needed foundation for assessments as required under the 2012 planning rule (USDA FS 2012).

Context of the NWFP and Forest Plan Revision Under the New Planning Rule

The 2012 National Forest System Land Management Planning Rule brought forth a wide range of changes to the forest planning process through the most collaborative rulemaking effort in agency history. The agency's goal was to implement an adaptive land management planning process that was inclusive, efficient, collaborative, and science-based, and that would promote healthy, resilient, diverse, and productive national forests and grasslands. This new rule is currently being used by national forests to revise forest plans that, in many cases, are 30 or more years old.

The 2012 planning rule, like the 1982 planning rule, sets a broader goal framework and direction for the NWFP revision. The National Forest Management Act requires the Forest Service to "provide for a diversity of plant and animal communities…to meet overall-multiple-use objectives" (Schulz et al. 2013). The 1982 rule required that this regulation be met by "maintaining viable populations of existing native and desired nonnative species in the planning area." As a result, the 1994 NWFP emphasized viability of all species as a goal. This requirement imposed an administrative burden on the agency and proved quite difficult to accomplish and provided controversial results. (Schultz et al. 2013). Consequently, the 2012 rule does not use viability of all species as a basis for conservation of biological diversity, but instead directs that maintenance of species be met through "coarse filter" (ecosystem) approaches that maintain ecological integrity, ecological functions, and habitat connectivity. The 2012 rule acknowledges that ecosystem-scale strategies do not necessarily provide for all species, and that a few species may require special attention as "species of special concern." We do not make recommendations on how to revise the NWFP, given the changes in planning rule direction since the Plan was developed. However, the NWFP contained specific objectives pertaining to conservation strategies for both ecosystems (coarse filter) and particular species (fine filter) and how these were intended to meet biological diversity goals. In several places in this synthesis, we discuss the published scientific findings that convey the advantages and shortcomings of employing these different conservation tactics.

Another change in the 2012 planning rule, compared to the 1982 rule, is its emphasis on using planning that is adaptive, as well as to more fully base Forest Service land management on scientific findings. The rule acknowledges that the body of science that can inform land management planning in such areas as conservation biology and ecology has advanced considerably since the 1982 planning rule was drafted. The new 2012 rule thus calls for planning to include three phases: assessment, plan development/amendment/revision, and monitoring (fig. 1-2). The assessment phase prepares the staff on a national forest for subsequent efforts to consider a full range of options for plan revision, including evaluation of existing information about relevant ecological, economic, and social conditions, trends, and sustainability, and their relationship to the land management plan within the context of the broader landscape. Assessment, including landscape assessments and other supporting science, can include local or traditional sources of information in addition to peer-reviewed science. This framework is intended to support an integrated approach to the management of resources and uses, incorporates the landscape-scale context for management, and ideally will help the Forest Service adapt to changing conditions, while improving management based on new information and monitoring.

The assessment process is conducted and managed by a responsible official, usually the forest supervisor, who has the discretion to determine the scope, scale, and timing of an assessment. Importantly, this synthesis is intended to be available to responsible officials in time to support their plan revision process. It also will support subsequent monitoring efforts, which are also required under the new planning rule. Monitoring information is intended to enable planners to change plan components or other content as needed.

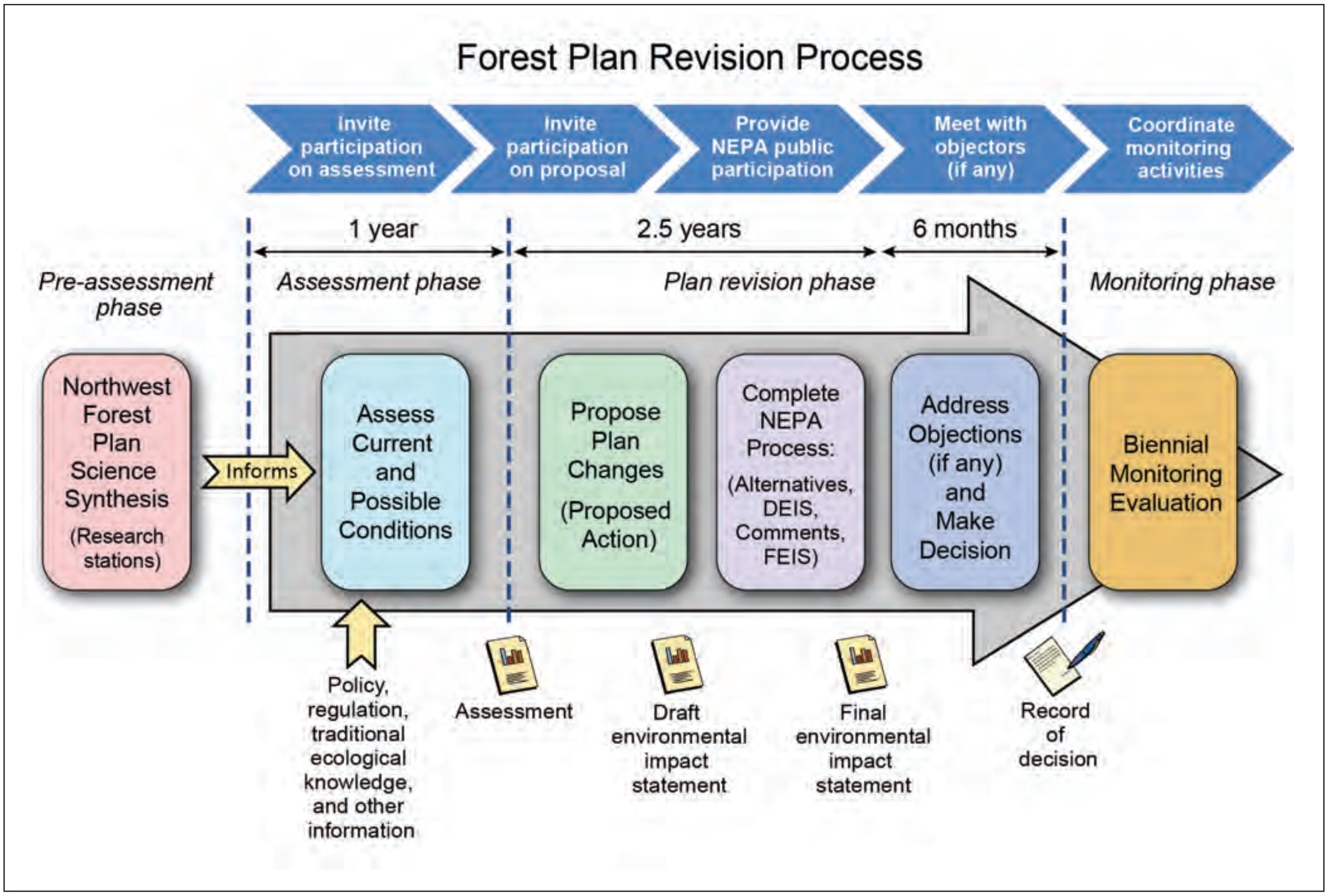

Figure 1-2—The science synthesis is part of the preassessment phase in forest plan revision and will inform the assessment phase of the planning process. NEPA = National Environmental Policy Act.

Given the pivotal role of science in the new planning rule, and the breadth and complexity of potential decisions in the NWFP area, development of this science synthesis was deemed essential to the entire plan revision process. The 17 national forests within the NWFP's footprint are expected to revise their land and resource management plans in the near future under the guidance of the new rule. The regional foresters in Regions 5 and 6 have been charged with following the new rule's detailed requirements, including the enhanced role of science in forest plan revisions. The new rule requires that:

> [the] responsible official shall determine what information is the most accurate, reliable, and relevant to the issues being considered. The responsible official shall document how the best available scientific information was used to inform the assessment, the plan decision, and the monitoring program as required in §§ 219.6(a) (3) and 219.14(a) (4). Such documentation must: Identify what information was determined to be the best available scientific information, explain the basis for that determination, and explain how the information was applied to the issues considered.

Accordingly, the Regions 5 and 6 regional foresters have asked that this science synthesis provide a thorough, up-to-date review of the relevant scientific literature pertaining to key resource management topics within the NWFP area.

Emergent Issues

Much has changed in the arenas of land management and science in the past 20-plus years. New issues have arisen that those designing or implementing the NWFP did not face at its inception. Going forward, some of these issues are particularly relevant to the fate of land management decisions within the NWFP area. The major considerations are summarized here briefly and amplified in subsequent chapters, particularly chapter 12, which explores various crosscutting themes and important implications for future forest plan revision.

Changing climate—
We devote an entire chapter (chapter 2) to the significance of climate change and the many ramifications it has on environmental conditions and on options that land managers have to achieve natural resource objectives. This issue has precipitated many shifts in conservation science and land management. Today, land managers are confronting difficult challenges and an uncertain future as they endeavor to mitigate climate effects through innovative management of forested landscapes. This development will continue to have a major impact on land management decisions throughout the NWFP area. Chapter 2 of this report is intended to lay a foundation for more indepth discussions of the realized and potential impacts of climate change on the other topics discussed in this synthesis. Although some core issues related to climate change are considered in chapter 2, additional chapters more specifically characterize climate change effects and concerns.

Single-species and multispecies conservation strategies—
The NWFP revolved around a select number of species at risk within the overall Plan area. Conservation of the northern spotted owl and the marbled murrelet were principal objectives for the Plan, and much NWFP management direction revolved around their species-specific needs. Additional focus was placed on conservation of aquatic ecosystems that support the many taxa of anadromous fish throughout the planning area. These include 15 species of salmon and steelhead formally listed as threatened, and one listed as endangered, since the Plan was initiated.

Although these particular taxa remain a vitally important focus in the Plan area, there has been much discussion and contemplation in the scientific literature about land management strategies aimed at single species, as reflected in changes in the 2012 planning rule described above. Management strategies aimed at individual endangered species may not always be in alignment with strategies to conserve ecosystem function. There is no single path to resolve this dilemma; it is a matter of much scientific debate and a subject we explore in more detail in chapter 12.

Successional and disturbance dynamics—
Succession, disturbance, and other ecosystem processes create a wide array of structural and compositional conditions within any given vegetation type. A primary focus of the NWFP was to manage for the continued existence of "old-growth forests" and their associated species. Succession and disturbance are continuously operating to shape forests, both independently and in concert. These topics are addressed in great detail in chapter 3.

The concept of ecological succession has been considered by ecologists for almost 200 years. More recently, however, the specific role of periodic disturbances (e.g., fire, windstorms, flooding) has been recognized as a critical element in shaping forests and promoting biological diversity by maintaining a variety of seral stages on landscapes. Disturbance ecology, especially fire ecology and the historical and contemporary role of fire within the NWFP area, has emerged in the past 30 years as a foundational science around which ecosystem management can be based. In many dry forests, simple models of successional change that were developed for moist forests do not apply because frequent fire regulated vegetation change in dry forests. Even within wetter forest areas, the effects of different historical disturbances, including fire, are important to consider in the conservation of important values (see chapters 3 and 11). This means that strategies to conserve and restore biological diversity across the diverse NWFP area may differ strongly between forest types, especially between dry and moist forests. After 150 years of Euro-American land use, the effects of anthropogenic disturbances, both obvious and subtle, have altered forest ecosystems and plant and animal communities. Knowledge of human influences on disturbance regimes is fundamental to sustaining biological diversity and ecosystem resilience.

Historical range of variability—

In the early developmental stages of the NWFP, the concept of historical range of variability (HRV) and its use in ecosystem management was just emerging in the scientific literature for the Pacific Northwest (Cissel et al. 1994). In the original discussions, this concept was useful for developing management goals for ecosystems that were based on inherent dynamics and processes rather than static structure targets. Although HRV is not explicitly referenced in the 2012 planning rule, the idea is addressed in directives for the rule in terms of "natural range of variability," which is essentially equivalent (Wiens et al. 2002). The rule does require forest plans "… to maintain or restore the ecological integrity of terrestrial and aquatic ecosystems and watersheds in the plan area," where ecological integrity depends in part on the functioning of natural disturbance regimes, which typically occur within some natural range of variation for a given climatic period. This is especially relevant in considering the significant role of fire in many different forest types throughout the NWFP area. For example, managing for ecological integrity in forest types subject to moderate- to high-frequency fire is quite different than in forest types where fire occurs infrequently. The complexity of land management becomes more apparent as we consider not just a simple dichotomy of wet and dry forests, but instead a spectrum of precipitation and fire regimes as well as the importance of fine-scale heterogeneity.

Research on changing climates has also emerged in the past 20 years, with a profound impact on our view of the HRV and its implications for management. We now face new scientific challenges in the restoration of degraded ecosystems, while managing for ecosystem resilience to climate change during the "Anthropocene," a proposed term for the geological and ecological epoch in which human activity has been the dominant influence on landscapes, invasive species, and climate change. These new impacts make maintaining some historical ecological patterns and processes difficult or impossible to reestablish (Corlett 2015). In chapters 3, 4, and 12, we assess this dilemma by describing scientific findings about the resilience of a variety of forest types to climate change, and consider what the implications are for maximizing suitable habitat for northern spotted owls. The notion of HRV and its potential consequences on other topics is also considered in other chapters.

Invasion of the barred owl and use of the term "habitat"—

The term "habitat" is widely used in natural resources publications and popular literature to describe the environmental area inhabited by a particular species of plant or animal. However, the many variations on the precise meaning of this term can lead to confusion. In common usage, "habitat" typically focuses primarily on the forest cover type chosen to depict the age and structure of a forest, or, more generally, the vegetation type that typifies the structure and composition of vegetation preferred by a given species. We note this because such definitions of habitat typically miss features believed to be important in conveying the full array of conditions suitable for a species. In particular, we identify the influence of an array of ecological factors, especially the role of nonnative species. Their impact has prompted much discussion as to what people generally consider to be habitat for any given indigenous species. In this report, we define habitat as follows:

> An area with the environmental conditions and resources (e.g., vegetation structure, food/prey, water, etc.) necessary for individuals of that species to survive and reproduce.

This definition specifically intends to draw attention to the phrase "environmental conditions," which includes potential effects of competitors or predators, including those that may be nonnative species. Clearly, competition between spotted owls and invasive barred owls represents a profound impact on the suitability of habitat for spotted owls.

Landscape ecology and management—

For many decades, forest management was conducted at the stand scale. The stand was traditionally an operational unit used by forest managers to target local forest management objectives, largely around local timber production goals. However, social and scientific trends over the past 25 years have led to broader scale silvicultural objectives and appreciation of more complex forest structures and nested scales for understanding forest dynamics.

Landscape ecology has emerged as a discipline that embraces the inherent spatial variation in landscapes, expressed at a variety of scales. We now more thoroughly appreciate the relationship between pattern and process in landscapes; the relationship of human activity to landscape pattern, process, and change; and the effects of scale and disturbance on the landscape. Above all, we now understand and intentionally incorporate the biophysical and societal causes and consequences of landscape heterogeneity as part of a landscape management philosophy. Several chapters in this report give consideration to the emergence of a landscape point of view.

Changes in agency capacity and workforce—
Federal agency budgets, number of employees, and number of field offices in the NWFP area have dropped substantially since the Plan was implemented, in large part because of shrinking timber programs and related budget allocations. These reductions have been most pronounced in Forest Service Region 6, and least pronounced on BLM lands. Declines in budgets and staffing have decreased the capacity of agencies to accomplish forest management goals, including forest restoration. Community-based organizations, local business partners, environmental and recreation organizations, and other groups have helped fill critical gaps by raising money and providing labor to accomplish forest management goals on federal lands in the face of declining agency capacity. But communities must have means to play this role. Title II funding from the Secure Rural Schools and Community Self-Determination Act has also played a vital role in helping pay for ecosystem management and forest restoration work on federal forests. However, the future of this law is uncertain given that this law expired in 2015 and it requires Congressional reauthorization. Thus, the issue of how to accomplish ecosystem management and forest restoration amidst reductions in agency capacity will continue to be a challenge.

Changes in wood processing infrastructure—
Wood processing infrastructure in Plan-area communities began declining in the 1980s. This decline has continued into the 2000s because of reduced demand for wood products from the Pacific Northwest, and in the supply available from federal forests, as well as because of changes in wood processing technology. Supply and demand of wood products is also influenced by a complex set of international market forces. Local supply is affected by changes in timber management resulting from policies and regulations that constrain available volume. Supply available to local markets is also significantly affected by international timber markets, which are entirely independent of federal forest policy. However, a decline in locally provided supply has had a profound impact on the local timber-processing industry, and its capacity to maintain its infrastructure.

This current lack of infrastructure makes the sale of timber, small-diameter wood, and biomass less economical, owing to longer haul distances and reduced demand for wood products, factors that reduce stumpage prices. Not only does this create a financial barrier to accomplishing forest management goals on federal forests; it also poses financial challenges for private forest owners who face declining markets for their wood products. For mills to stay in business, or for investments in new infrastructure development to occur, a reliable supply of raw material is needed. Private lands may be unable to increase wood product production and still ensure sustainable harvest levels. Thus federal lands have an important role to play in providing a sustainable supply of wood products to keep existing wood processing infrastructure operating, and to expand it if desired through new investments. To date, federal forests in the NWFP area have not met the goal of ensuring a predictable supply of timber, nor have they met the probable sale quantity established by the Plan. This topic is treated in detail in chapter 8.

Evolving public values and public policies around natural resources—
Social scientists and policy analysts studying environmental values and attitudes in the United States documented a shift away from the predominantly commodity-oriented view of forest management, common prior to the 1980s, to a more mixed or balanced perspective that includes commodity and noncommodity uses. This shift in public values followed a series of policies initiated in the 1960s that placed greater attention on protection of wildlife, wilderness, air, and

water, as well as a desire for improved relationships with tribal governments, to name a few concerns.

Longitudinal studies conducted both on a national scale and in subregions of the United States indicate a gradual shift in public attitudes. Since the 1990s, attitudes about public lands have shifted from a sole focus on economic values, outputs, and commodities toward a greater diversity of values that includes noneconomic values, especially protection of ecosystems and aesthetic values. Sometimes this transition is described as a shift from an exclusively anthropocentric perspective to a balance of anthropocentric and biocentric perspectives. Residents of the NWFP area echoed this national trend.

In reflection of this value shift, the Forest Service was one of the first public land management agencies to adopt an ecosystem management approach in the 1990s, one that aimed to conserve ecological services and restore resources while meeting the needs of current and future generations. In more recent years, public recognition of the dual focus of producing goods and services while protecting resources has gained ground, and the challenges in achieving this balance in a complex ecological system appear to be more widely understood.

Ecosystem services—

The concept of ecosystem services was originally characterized by economist E.F. Schumacher as "natural capital" in 1973. Only recently has the concept become widely recognized as relevant to land and resource management. The 2005 Millennium Ecosystem Assessment (MEA 2005) provided a simple definition of ecosystem services as "the benefits people obtain from ecosystems." Historically, management efforts focused on the provision of such resources as water and timber. Currently, policy and management efforts have increased the appreciation and importance of the full suite of services derived from ecosystems, including nonprovisioning services such as spiritual and cultural heritage values. Our understanding of the full scope of ecosystem services and attendant societal values associated with Northwest forests is still emerging. Our aptitude for quantifying these values, particularly in monetary terms, will continue to evolve as methods improve.

Attitudes toward land management agencies—

Public lands management is an important element of public discourse in the national environmental policy arena. Some recent issues have been controversial in the public eye. The number of appeals and litigation of forest decisions provides clear evidence that social views about forest management are often polarized. Effective public engagement can help provide accessible processes for public deliberation. Studies have shown that public dissatisfaction with opportunities to participate has led to more appeals of agency decisions, and that participants desire public processes that are more collaborative.

An important factor shaping natural resource management outcomes is the degree of trust between land management agencies and the public. A lack of public trust in government is cited as a primary barrier in natural resource planning (see chapter 9) that potentially can lead to litigation or noncompliance, and, ultimately, to managerial impasse. Furthermore, trust has been shown to be correlated with social acceptability of forest management actions, although the actual causes of social acceptability are likely far more nuanced. There are two basic kinds of trust: institutional trust (trust in agencies to represent and serve the public), and interpersonal trust (trust cultivated based on personal relationships). When social trust is improved, there is greater support for land management policies. The assumption held by many is that trust can be built (and conflict reduced) through fair participation processes or transparent decision-making. Trust building occurs when stakeholders engage in meaningful dialogue in a context of shared power and high levels of substantive knowledge. Collaborative processes represent opportunities to build iterative experiences and develop relationships among multilateral stakeholders and between stakeholders and public land management agencies. Examples of how collaborations between the Forest Service and tribal governments and communities are facilitating cross-boundary management and pursuit of integrated social and ecological objectives are featured in chapter 11. These examples illustrate how local units and communities are working to fulfill the many goals for public lands management as reflected in the NWFP and the new planning rule, as well as the many challenges in that pursuit.

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Appendix: Priority Management Questions to Guide the Northwest Forest Plan Science Synthesis As Defined by Pacific Northwest and Pacific Southwest Forest and Regional Staff and Edited by the Science Synthesis Team

The Northwest Forest Plan (NWFP) science synthesis was constructed based on a set of questions submitted by Forest Service land managers. The questions addressed concerns that developed from 24 years of experience in implementing the Plan, as well as new issues that have emerged since the Plan was initiated. The Science Synthesis Team reviewed an initial list of 190 questions submitted by Forest Service land managers and suggested additional questions that they believed were relevant and could be addressed in the synthesis. The team then removed redundant questions and grouped others to arrive at the final list of 73 questions delineated below. This list is sorted into four general topical areas that are covered in one or more of the 12 synthesis chapters. Based on available information, the synthesis attempted to fully or partially address all the questions. Although the chapters do not necessarily address these questions directly, they were organized to be consistent with the scientific understanding of the issues that these questions address. In each chapter, the management considerations section endeavored to more directly link the science to management issues related to these questions. To the extent possible, the synthesis addressed how the science differs by physiographic province, vegetation type, and disturbance regime.

Priority Questions

Vegetation/forest management/climate change/ecological disturbance (old-growth and other vegetation types)—

1. What is the latest science on active management, including "ecological forestry," to protect and restore late-successional forests and maintain ecological diversity?
2. How do the effects differ by treatment (mechanical and prescribed fire) in terms of key ecosystem components (structure, composition, connectivity, and function)? What are the associated costs and commodity outputs?
3. What is the latest science on the dynamic landscape approach versus a fixed reserve system in terms of providing sustainable amounts and adequate distribution and connectivity of late-successional forest across the landscape?
4. How does each approach allow us to adapt in response to large-scale disturbances?
5. What is the relationship between amount and configuration of old growth and potential to sustain a variety of disturbance regimes and late-successional-dependent species?
6. How might management and conditions on other ownerships affect the above relationship with the understanding that old growth is likely to persist only on federal lands?
7. What is the latest science on treatments in stands greater than 80 years of age when the objective is to accelerate the development of late-successional habitat?
8. Similarly, what is the latest science on limiting harvest of large trees (usually >21 inches diameter at breast height when conducting restoration activities?
9. What are the latest estimates for historical/natural range of variation (HRV/NRV)? What is the proportional mix of seral stages and special habitats (e.g., hardwoods, meadows, etc.)?
10. What are estimates of patch and gap size, connectivity, disturbance (fire, insect and disease, drought), habitat, and within-patch heterogeneity?
11. What are important differences between "dry forests" vs. "wet forests" and how can these distinctions be used to prioritize restoration activities?
12. What does the latest science tell us about the concept about using HRV/NRV to inform ecological restoration, in terms of the mix of structural conditions, species composition, patch size, etc.? Does HRV/NRV help inform landscape-level patch dynamics and within-stand heterogeneity?
13. What are the effects, if any, on invasive species on old-growth forests and succession following disturbance?

14. What is the competing science on restoration of Pacific Northwest forest systems? For example, we need to have an upfront discussion of differing viewpoints in the science on the need for restoration of late-successional/old growth (LSOG) in dry forests.
15. What is the relationship between retention of dead wood, including dead and damaged trees, and potential for disturbance in dry forests with a frequent fire regime?
16. How does dead wood affect our ability to maintain LSOG?
17. What is the relationship between retention of green trees in harvest units and ecological diversity and species viability?
18. What is the relationship between green tree retention potential and insect and disease epidemics (especially dwarf mistletoe) in post-harvest or post-wildfire situations?
19. How does each approach allow us to adapt in response to large-scale disturbances?
20. How do green tree retention effects differ by physiographic province and vegetation type?
21. What is the latest science on the connectivity of late-successional and other key habitats (fixed corridors versus landscape permeability)?
22. What does the current body of science suggest about postfire recovery options, including the social license and economics associated with salvage?
23. What are the ecological features associated with early-successional vegetation, and what is the role of early-successional vegetation in ecosystem function and biodiversity?
24. What are the potential conservation and restoration needs related to early-successional vegetation?
25. What are our most vulnerable ecosystems, species, and resources due to climate change?
26. What are the key adaptation strategies that could mitigate these vulnerabilities?
27. What different management strategies might be needed for forests and terrestrial and aquatic ecosystems?
28. How do we deal with uncertainty in our restoration efforts, models, and predictions?
29. What are the anticipated changes in climate within the NWFP area, and what are the potential impacts to disturbance processes (insect, disease pattern, drought, fire, etc.), vegetation, species habitats, aquatic ecosystems, and the provision of goods and services (timber, values, etc.) within the area?
30. What resources and components of a regional planning framework require analysis and consideration at the regional scale?

Terrestrial species/habitat management (northern spotted owl, marbled murrelet, other species associated with older forests)—

1. What is the latest science surrounding the effects of various treatments (silviculture, fuels) and wildfire on LSOG and plantations and what are the effects on terrestrial wildlife species, with particular attention on northern spotted owl (NSO), barred owl (BAOW), marbled murrelet (MAMU), and survey and manage (S&M) species?
2. How or do these species use these treated habitats post-treatment, and are there ways to modify treatment to benefit these terrestrial species?
3. How do these treated habitats compare to untreated habitat in terms of habitat use and reproductive success?
4. How does use of treated and untreated areas compare to use of postfire habitats, including salvage?
5. How do the risks of fire compare in treated and untreated habitats, and are the impacts of treatments by the risk of habitat loss due to fire?
6. What is the latest science on the interaction of barred owls and spotted owls and the impact to recovery of the spotted owl?
7. What is the relationship of fires to barred owl encroachment?
8. What is the current scientific understanding about the rarity of survey and manage species, and how effective are the management recommendations for habitat buffers in retaining these species across treated landscapes?

9. Is forest management under the NWFP providing habitat for rare and uncommon species as planned?
10. Are rare and uncommon species maintaining populations under NWFP management?
11. Have we accumulated enough information to change status of these species? Are there species originally ranked as having low potential for persistence that are now of less concern, particularly with the reduction in harvest levels of old growth we've seen under the NWFP?
12. Has the Interagency Special Status/Sensitive Species (ISSSP) program benefitted these species?
13. What is the effect of prescribed fire and wildfire on rare and uncommon species (S&M)?
14. Are known site buffers as effective as landscape scale habitat management in ensuring species persistence, dispersal and habitat connectivity?
15. Does the current S&M species list truly represent currently rare species with population persistence questions dependent upon LSOG habitat?
16. Does the current NSO critical habitat better represent late-successional forest and provide for a higher level of assurance of persistence for NSO, MAMU, and S&M species when compared to the current NWFP late-successional reserve (LSR) network?
17. Is there a difference in persistence in treated vs. untreated LSRs or LSOG habitat in the face of wildfire, insects and disease, and climate change?
18. What role and importance are riparian reserves and various buffer widths as terrestrial species (including mollusks) habitat, including dispersal and connectivity, and how does riparian reserve management impact the terrestrial species that utilize them?
19. How can we manage a riparian area for the variety of habitats needed?
20. What is the status of other species of concern (not included as survey and manage species) within the footprint of the NWFP?
21. What is the effect of pesticide use associated with cannabis cultivation or species viability (i.e. fisher)?
22. How can we manage for viable populations of snag-dependent species when snags are not present long-term on the landscape?
23. How can we identify important biological refugia? What are they and where are they?

Aquatic/riparian management (aquatic and riparian species and ecosystems)—

1. What is the current thinking/science on riparian thinning/management? Has it produced the desired results, including contributions toward recovery of listed fish species, impaired waters, and reduction of fire risk?
2. What are the effects of common silvicultural treatments/prescriptions with respect to Aquatic Conservation Strategy (ACS) goals and objectives (especially riparian microclimate and stream temperature, wood recruitment, diversity in riparian species structure and composition, fish populations, terrestrial processes)?
3. What are the effects of not managing previously harvested stands in riparian reserves (RRs)? What is the risk of severe wildfire in untreated riparian corridors, and do/how do various types of treatment reduce this risk?
4. What does the current science indicate regarding the value of woody material in second-growth riparian reserves? When and where should the creation of large wood be a purpose and need driving silvicultural treatment in riparian reserves?
5. What does the current science indicate about the role of vegetation management in affecting ground water flows and temperatures, and how do those changes affect surface water?
6. Does current science indicate that the ACS is needed to achieve Plan goals of maintaining and restoring the ecological health of watersheds and aquatic ecosystems on public lands?
7. Are all components (riparian reserves, key watersheds, watershed restoration, watershed analysis, ACS objectives, standards and guidelines, monitoring and evaluation) necessary to achieve these goals?

8. Does the current science indicate that refinements to the ACS may be needed to increase its efficacy?
9. Does ACS provide appropriate levels of connectivity or does it need to be refined?
10. What are the effects of interbasin water transfers and water diversions?
11. What does the current science indicate about where in the NWFP area the greatest potential for conflicts exist over water supply and demand for additional storage based on the current water supply and demand situation, projected changes in supply due to climate change, and projected changes in demand due to climate change and population growth.
12. How well have RRs met their intended objectives?
13. Does current science support or refine Forest Ecosystem Management Assessment Team (FEMAT) conclusions regarding the role and function of RRs? If so, how?
14. What have we learned since FEMAT that should be incorporated into RR designation and management in plan revisions?
15. What is the latest science on the effectiveness of treatments within riparian reserves, and implementation of varying riparian reserve widths?
16. Is the type, scope and scale of watershed restoration that has occurred over the life of the NWFP consistent with FEMAT and Plan assumptions?
17. How effective are instream restoration treatments (e.g., large woody debris [LWD] augmentation, channel reconstruction) in achieving ACS objectives at multiple spatial and temporal scales? Fish passage restoration? Road decommissioning and improvements? Riparian restoration treatments (e.g., reforestation, thinning, gaps)?
18. What does the current science indicate about potential short-term impacts to aquatic and riparian ecosystems when managing for long-term restoration of aquatic and ecosystem processes and functions (e.g., short-term stream temperature increases to achieve long-term large wood recruitment and normal disturbance processes)?
19. What are the consequences of the current road management regime on water and aquatic resources? Consider (a) the status and trends in the size of the road system on NFS and other federal lands, (b) the amount of the current system that poses a high risk to aquatic resource, and (c) the amount of the system that is being maintained or improved.

Social/economic (including timber production) (socioeconomic well-being, timber harvest; collaboration and stakeholder attitudes; tribal values and resources)—

1. What does social science tell us about how stakeholders' attitudes, beliefs, and values (ABV) have changed over the past 20 years, and how those ABV are associated with resource management (including recreational experience, resource use or protection)?
2. How have stakeholders' relationships to landscapes and natural resources changed in the Northwest Forest Plan area?
3. What value do people place on cultural ecosystem services from public lands, including outdoor recreation?
4. What are the general conditions of and influences upon values of special concern to tribes (including first foods such as salmon, elk, huckleberry, camass root) in the NWFP area?
5. What management strategies does science suggest would enhance these values of special concern to tribes?
6. What does the body of science indicate are important factors contributing to successful collaboration in forest management?
7. Where are our most successful examples of such collaboration?
8. What are the most important factors in successful collaboration?
9. What strategies are suggested by science for engaging communities in forest plan revision in the NWFP area?
10. What are implications for forest management from trends in the size and socioeconomic status of low-income, minority, and tribal populations (i.e., environmental justice populations) in the NWFP area?

11. Are these populations growing?
12. What are the drivers of change related to socioeconomic well-being in rural communities?
13. What are the implications for forest management of trends in socioeconomic well-being in rural communities in the NWFP area?
14. How does the body of science inform sustainable recreation and social interest in valuing place (as required under the 2012 planning rule)?
15. What does the science infer about the contribution of outdoor recreation across the region to social and economic sustainability?
16. What are the trends in outdoor recreation use and visitor satisfaction on public lands?
17. What are the drivers for change related to recreation?
18. What are the implications for forest management of changes in land use and ownership in the past 20 years?

Other Topics to Be Considered in the Integration Section of the Synthesis (Pulled From Region 5 and Region 6 Long List)

1. Influence of illegal marijuana cultivation on federal lands on resources (this was noted under terrestrial biological resources question #15, but effects on resources other than fisher will also be considered).
2. Effects of invasive species on forest succession and habitats (this topic is noted under vegetation question #10 in the context of old growth)
3. Salvage logging
4. Conservation of nonfederally listed species (noted under terrestrial biology question #5)


Prescribed burn operations on the Wallowa-Whitman National Forest, Oregon.
Photo by USDA Forest Service.

Chapter 2: Climate, Disturbance, and Vulnerability to Vegetation Change in the Northwest Forest Plan Area

Matthew J. Reilly, Thomas A. Spies, Jeremy Littell, Ramona Butz, and John B. Kim[1]

Introduction

Climate change is expected to alter the composition, structure, and function of forested ecosystems in the United States (Vose et al. 2012). Increases in atmospheric concentrations of greenhouse gases (e.g., carbon dioxide [CO_2]) and temperature, as well as altered precipitation and disturbance regimes (e.g., fire, insects, pathogens, and windstorms), are expected to have profound effects on biodiversity, socioeconomics, and the delivery of ecosystem services within the Northwest Forest Plan (NWFP, or Plan) area over the next century (Dale et al. 2001, Franklin et al. 1991). The ecological interactions and diversity of biophysical settings in the region are complex. The effects of climate change on ecological processes will occur through a variety of mechanisms at a range of spatial scales and levels of biological organization, ranging from the physiological responses of individual plants to the composition and structure of stands and landscapes (Peterson et al. 2014a). Understanding and incorporating how climate change projections and the potential ecological effects and uncertainties differ within the region (e.g., Deser et al. 2012) is essential for developing adaptation and mitigation strategies.

Climate change has the potential to affect all ecological and socioeconomic components of the NWFP, as well as other objectives for federal forest managers in this region. However, climate change is only one factor that managers must consider when addressing conservation and other goals for the NWFP region. The overarching goal of this chapter is to lay a general foundation of current knowledge and understanding of climate change for the subsequent chapters in this synthesis report, and not to analyze and report the projected effects of climate change on all the different components of the Plan in detail. The chapters that follow address the role of climate change in the context of their particular topics (e.g., northern spotted owls, aquatic ecosystems). This chapter focuses on the following topics:

- Regional climate setting, including an introduction to the major vegetation zones and disturbance regimes of the region (see chapter 3 for a more detailed discussion of disturbance regimes)
- Climate history of the region from the Holocene through the 20th century
- Overview of climate modeling approaches and limitations
- Projected changes in climate and how these vary across the region
- Mechanisms of vegetation change and potential climate change vulnerabilities
- Projected effects on vegetation at regional scales
- Uncertainties associated with models and knowledge of climate change effects
- Management considerations and strategies for adaptation and climate change mitigation goals. (See chapters 3 and 12 for a more complete discussion of management options)

This chapter does not address broader issues of NWFP ecological and socioeconomic goals in the context of climate change. These topics are covered in chapter 12, in which climate change is considered along with other factors (e.g., nonnative species, ecosystem vs. species approaches to conservation, and tradeoffs) in a discussion of the science underlying the goals of the NWFP and the 2012 planning rule. This chapter is also guided by questions from managers, as follows:

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Guiding Questions

This chapter addresses the following:

1. How did climate and vegetation change from the early Holocene to the late 20th century, and how did these changes vary across the NWFP area?
2. What are recent trends in climate change and how do they vary geographically across the NWFP area?
3. What are the major tools for projecting climate change and what are the associated uncertainties and limitations?
4. What changes in climate are projected for the NWFP area and how do these projections differ across the region?
5. What are the implications of recent and projected climate trends for vegetation change?
6. What are the mechanisms of vegetation change associated with climate change?
7. Which ecosystems and species are most vulnerable to climate change?
8. What are the key adaptation strategies that could reduce vulnerability to climate change?

Background and Setting

The NWFP area covers approximately 24.4 million ac (9.9 million ha) and includes multiple physiographic provinces across Washington, Oregon, and northern California (fig. 2-1). These physiographic provinces encompass a variety of disturbance regimes (see chapter 3 for more discussion and information) as well as a broad range of environmental and climatic gradients (fig. 2-2). Climate is cooler and wetter toward the north in the coastal and inland mountains, but transitions to a more Mediterranean climate with warmer, drier summers and greater interannual variability to the south (fig. 2-3). Most precipitation in the region falls during the winter months, often as snow at higher elevations. The Olympic Peninsula, Western Lowlands, and Coast Range are located in the western portion of the region. These receive the greatest annual precipitation and often experience a summer fog layer along the coast that can partially moderate summer moisture stress. The crest of the Cascade Range extends from northern Washington to northern California, bisecting much of the region and creating a steep gradient in precipitation from west to east. The western Cascades encompass a wide range of elevations, temperatures, and precipitation, which generally decreases toward the south. The eastern Cascades extend in a narrow band from Washington to the California border and are generally much drier than the western Cascades and most of the NWFP area. The Klamath Mountains, in southwest Oregon and northwest California, represent the most climatically and geologically diverse province in the area, with a strong west-to-east gradient in precipitation and summer moisture stress. The Willamette Valley makes up a relatively small portion of the NWFP area and is predominantly nonforested.

The broad range of environmental and climatic gradients is reflected in the distribution of several potential vegetation zones across the region (figs. 2-1, 2-2, and 2-3) (Simpson 2013) (https://www.ecoshare.info/category/gis-data-vegzones). Potential vegetation zones represent climax vegetation types that would eventually develop in the absence of disturbance; therefore, existing or current vegetation varies often within zones depending on seral stage (i.e., successional stage or stage of structural development) and time since disturbance. For example, the most abundant vegetation zone in the NWFP area, western hemlock (*Tsuga heterophylla*), is currently dominated by Douglas-fir (*Pseudostuga menziesii*). Vegetation zones provide an ecological framework for discussing climate and vegetation change across broad geographic extents (chapter 3). Vegetation zones have overlapping species pools but consist of unique plant community assemblages, as well as similar but internally variable biophysical conditions and historical disturbance regimes that differ geographically (Winthers et al. 2005; chapter 3). Vegetation zones have characteristic pathways of structural development that differ in complexity and reflect regional gradients in productivity as well as historical and contemporary disturbance regimes (Reilly and Spies 2015).

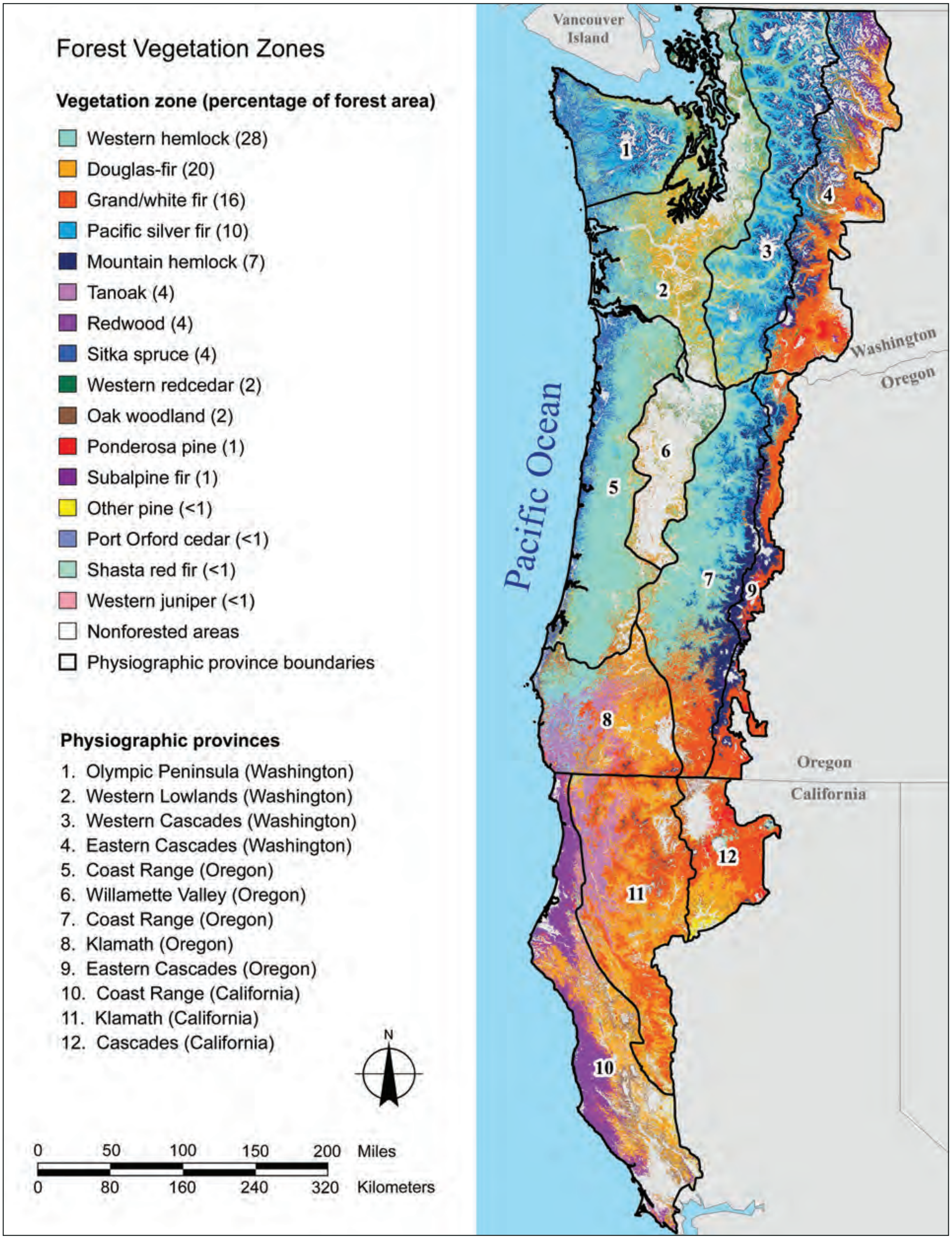

Figure 2-1—Geographic distribution of potential vegetation zones (Simpson 2013) and physiographic provinces within the Northwest Forest Plan area.

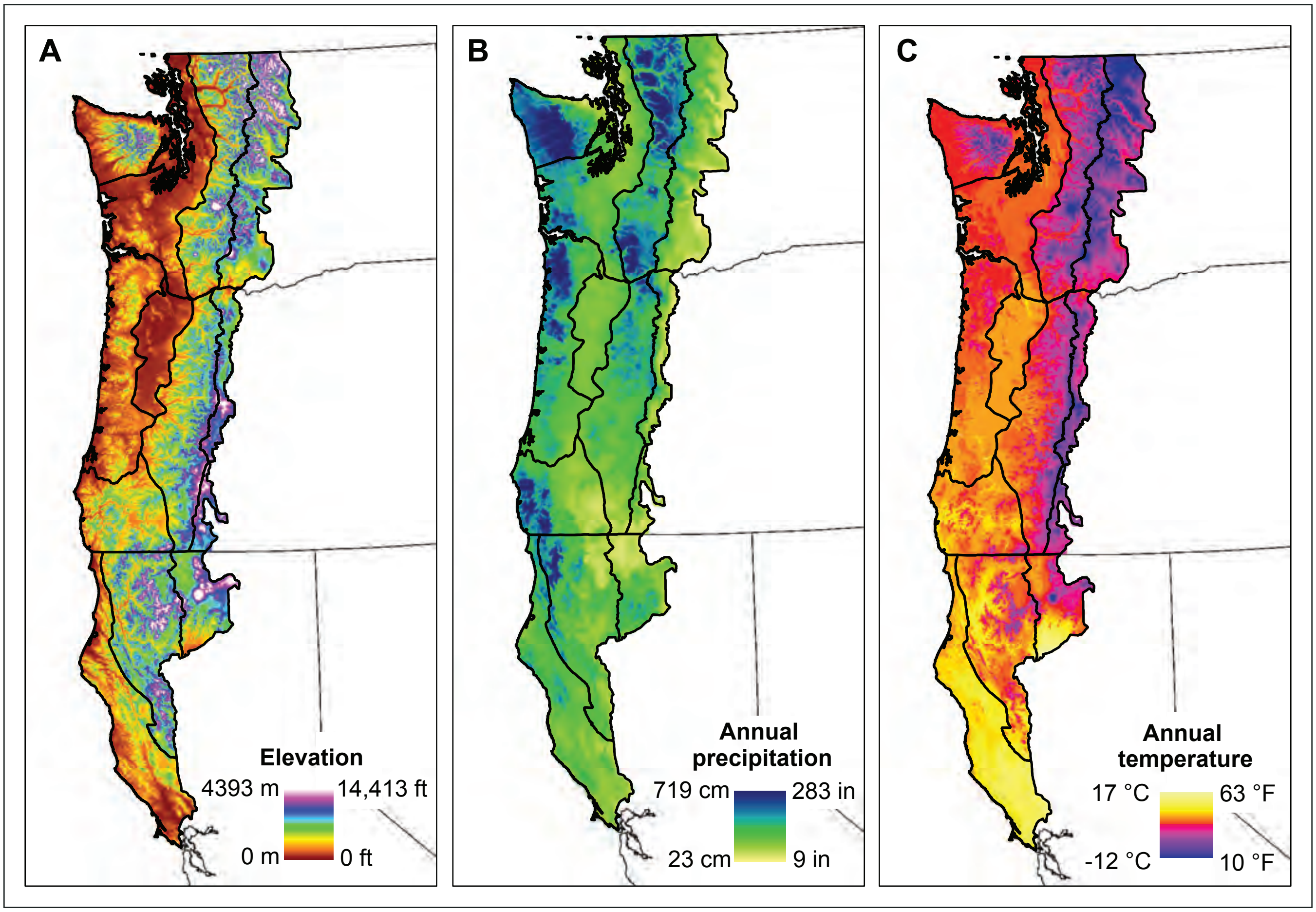

Figure 2-2—Maps of (A) elevation, (B) annual precipitation, and (C) annual temperature in the Northwest Forest Plan area. Temperature and precipitation are derived from 30 arc-second (~800 m) PRISM (parameter-elevation regressions on independent slopes model) (Daly et al. 2008) grids averaged from 1971 to 2000, and were obtained from the Landscape Ecology, Modeling, Mapping and Analysis group at Oregon State University. Darker lines outline physiographic provinces shown in figure 2-1; lighter black lines show state boundaries.

The major vegetation zones (figs. 2-1 and 2-4) of the region generally correspond to those presented by Franklin and Dyrness (1973) and were broken into moist and dry forests in the NWFP (chapter 3). This characterization is overly simplistic, as annual precipitation in any given zone varies geographically. Moist vegetation zones make up about 60 percent of the region, and are primarily located in coastal areas and west of the Cascade crest. These include Sitka spruce (*Picea sitchensis*), redwood (*Sequoia sempervirens*), tanoak (*Lithocarpus densiflorus*), western hemlock, western redcedar (*Thuja plicata*), Pacific silver fir (*Abies amabilis*), and mountain hemlock (*Tsuga mertensiana*). Dry forest vegetation zones are located east of the Cascade crest, and also comprise a large portion of inland areas in southwest Oregon and northwest California. They include western juniper (*Juniperus occidentalis*), ponderosa pine (*Pinus ponderosa*), Douglas-fir, grand fir (*Abies grandis*) and white fir (*Abies concolor*), and subalpine forests dominated by subalpine fir (*Abies lasiocarpa*), Engelmann spruce (*Picea engelmannii*), and whitebark pine (*Pinus albicaulis*). A more detailed and comprehensive characterization of plant communities in individual vegetation zones can be found in Franklin and Dyrness (1973).

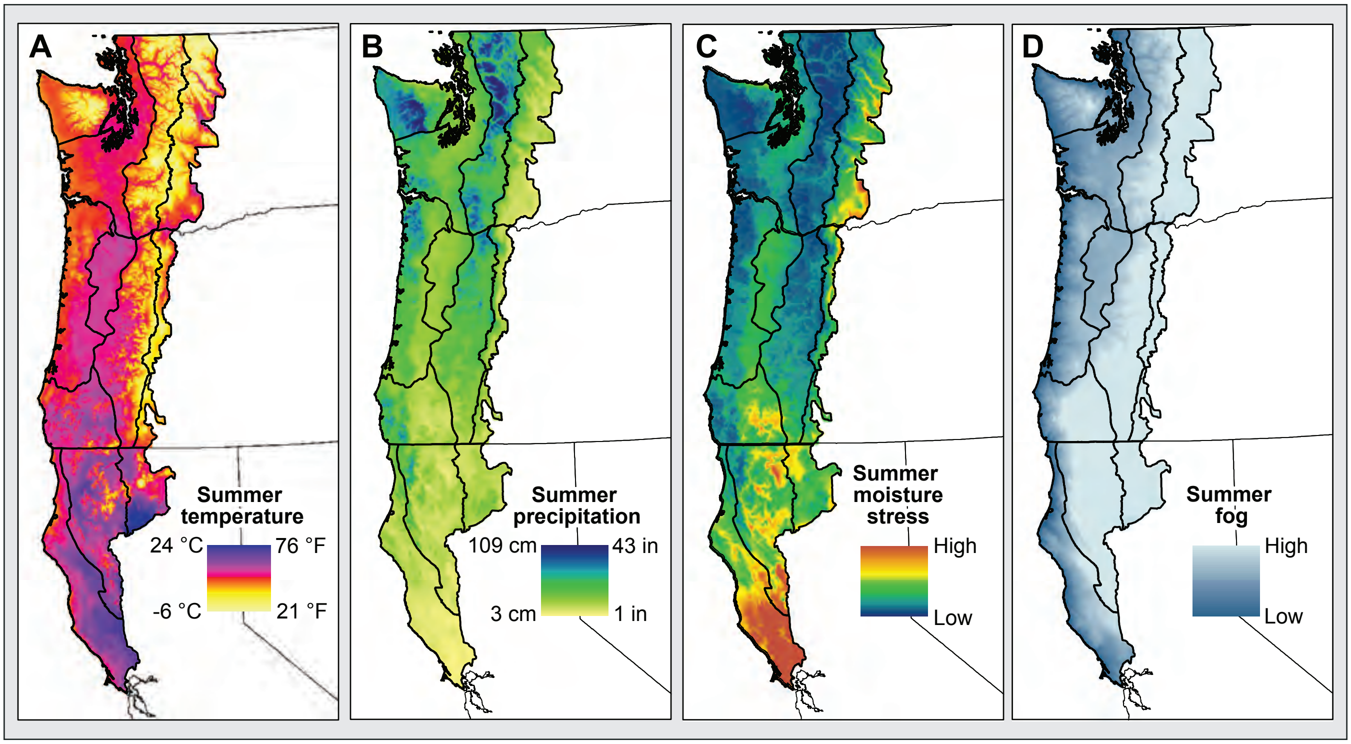

Figure 2-3—Maps of (A) mean summer temperature, (B) total summer precipitation, (C) summer moisture stress, and (D) summer fog in the Northwest Forest Plan area. Temperature and precipitation are derived from 30 arc-second (~800 m) PRISM (parameter-elevation regressions on independent slopes model) (Daly et al. 2008) grids averaged from 1971 to 2000, and were obtained from the Landscape Ecology, Modeling, Mapping and Analysis group at Oregon State University. Summer moisture stress was calculated by dividing summer temperature by summer precipitation for May through September. Summer fog is a proxy based on the optimal path length from coastline representing the easiest path of fog movement given topography and terrain blockage (Daly et al. 2008). Darker lines outline physiographic provinces shown in figure 2-1; lighter black lines show state boundaries.

More information on geographic variability and current vegetation in Oregon and Washington is available at Ecoshare (https://www.ecoshare.info/publications) and is discussed further in chapters 1, 3, and 12. Appendix 2-1 provides a crosswalk for linking equivalent vegetation types between the Simpson (2013) vegetation zones and existing vegetation in northern California based on Regional Dominance 1 in the Pacific Southwest Region (Region 5) CALVEG database. This crosswalk provides a means of interpreting the Simpson vegetation zones in terms of existing vegetation in California. More details on the CALVEG database are available at https://www.fs.usda.gov/detail/r5/landmanagement/resourcemanagement/?cid=stelprdb5347192.

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Figure 2-4—Examples of forests from several vegetation zones illustrating the broad range of environmental and biophysical settings in the Northwest Forest Plan area: (A) western hemlock, (B) redwood, (C) mountain hemlock, (D) subalpine fir, (E) grand fir/white fir, and (F) ponderosa pine.

Key Findings

Past Climate Change in the Northwest Forest Plan Area

The climate and vegetation of the NWFP area went through continuous change over the past 11,700 years during the Holocene. During this time, complex interactions between a fluctuating climate and fire drove vegetation change at millennial scales (Bartlein et al. 1998, Marlon et al. 2009, Walsh et al. 2015, Whitlock 1992, Whitlock et al. 2008). Species responded individualistically to changes in climate, sometimes forming assemblages that lack contemporary analogs (Whitlock et al. 2003). Species ranges expanded and contracted over time, with some species persisting in refugia where local conditions allowed persistence in regions where climate was generally unsuitable (Gavin et al. 2014). Refugia likely provided an important role in the persistence of populations through the numerous climatic transitions that occurred in the region since the last glacial maximum (Bennett and Provan 2008, Hampe and Jump 2011).

Knowledge of vegetation changes during the Holocene is particularly rich in the NWFP area, and a number of paleoecological studies document change across the region. The Holocene is commonly divided into different periods that can be distinguished by climate and fire activity. We follow the divisions of Walsh et al. (2015) in a recent review, though other studies use different dates to delineate periods, and the timing of changes in climate and vegetation differ across the NWFP area (Whitlock et al. 2003).

Paleoecological studies use charcoal and pollen found in sediment cores from lakes, as proxies for past climatic conditions, and to reconstruct changes in vegetation composition over time (Whitlock et al. 2003). These studies are limited in terms of their spatial and temporal precision, but offer important historical context and insight on climate and vegetation change by broadening our understanding of the historical range of variability at millennial time scales.

The early Holocene—approximately 12,000 to 8,000 years before present (BP)—was a time of rapid vegetation change, with assemblages that include current subalpine and lower elevation species that lack modern analogs (Whitlock 1992). Increased summer insolation during this period led to higher summer temperatures and drier conditions than the present, while lower winter insolation led to cooler and wetter winters, likely amplifying seasonality and summer drought compared to present-day climate (Bartlein et al. 1998, Whitlock et al. 2001). Fire activity was relatively low at the beginning of the early Holocene, but increased and remained high until approximately 8,000 years BP (Briles et al. 2005, Walsh et al. 2015). Nonforested areas and open woodlands were replaced by forests as glaciers receded early in this period, and xerophytic species increased at many low-elevation sites across western Oregon and Washington as summers warmed (Walsh et al. 2015).

As the climate warmed during the early Holocene, species responded individualistically and became distributed along elevational and latitudinal gradients (Whitlock et al. 2003). Douglas-fir, red alder (*Alnus rubra*), and oak (*Quercus* spp.) replaced spruce and pine at lower elevations in the Coast Range and western Cascades (Cwynar 1987, Grigg and Whitlock 1998, Long et al. 1998, Sea and Whitlock 1995, Walsh et al. 2008). On the Olympic Peninsula, herbaceous tundra was replaced by subalpine fir (Gavin et al. 2001). Mid-elevations of the eastern Cascades of Oregon were dominated by open pine (*Pinus* spp.) forests, initially with an understory of *Artemesia*, which likely transitioned into a closed-forest environment with a greater abundance of *Abies* spp. Mid-elevations of the Klamath Mountains in Oregon and California were dominated by open woodlands composed of *Pinus* spp., *Quercus* spp., and incense cedar (*Calocedrus decurrens*) (Briles et al. 2005, Daniels et al. 2005, Mohr et al. 2000).

Cooler, wetter conditions were associated with decreasing summer isolation during the middle of the Holocene (~8,000 to 4,000 years BP) (Bartlein et al. 1998). During this time, fire activity decreased (Briles et al. 2005, Walsh et al. 2015), and modern species assemblages were formed in some parts of the region (Whitlock et al. 1992). Redcedar and western hemlock increased during this period across low- and middle-elevation forests of the Coast Range, the Cascade Mountains, and the Puget Trough (Cwynar

1987, Prichard et al. 2009, Walsh et al. 2008). Species composition shifted toward silver fir, mountain hemlock, and Alaska yellow-cedar (*Callitropsis nootkatensis*) on the Olympic Peninsula (Gavin et al. 2001). In the Klamath Mountains, expansion of *Pinus* spp., *Cupressaceae*, and *Abies* spp. also indicated cooler, wetter conditions during this period (Briles et al. 2005, Daniels et al. 2005, Mohr et al. 2000). With the exception of lower elevations, fire activity increased again approximately 5,500 years BP (Walsh et al. 2015).

Fire activity continued to increase during most of the late Holocene (~4,000 years BP to present) despite evidence that this period remained cool and moist (Bartlein et al. 1998, Walsh et al. 2015). There is little evidence in the pollen record to suggest major changes in the composition of vegetation assemblages across most of Oregon and Washington during this time (Walsh et al. 2008, 2015; Whitlock 1992). Modern forest assemblages in the Douglas-fir and white fir zones established approximately 2,000 years ago in the Klamath Mountains, where fire activity also increased during this time despite cool and moist conditions (Briles et al. 2005, 2008; Daniels et al. 2005; Mohr et al. 2000). Climate and fire fluctuated during the past 1,000 years. The warmest temperatures occurred during the Medieval Climate Anomaly (MCA) (900–1250 CE) and the coldest temperatures during the Little Ice Age (LIA) (1450–1850 CE) (Steinman et al. 2012). Precipitation also varied during this time, but there is less consensus about this in the literature. Cook et al. (2004) argued that a period of drought occurred during the MCA, but more recent evidence suggests a wet MCA and dry LIA (Steinman et al. 2014). Fire frequency increased during the MCA in the Klamath Mountains (Daniels et al. 2005, Mohr et al. 2000) as well as the rest of the region in Oregon and Washington (Walsh et al. 2015). Many of the currently existing old-growth forests in moist vegetation zones established at this time (chapter 3).

Climate fluctuations associated with surface temperatures in the Pacific Ocean also became more apparent over the past 1,000 years (Nelson et al. 2011). Warming and cooling of sea surface temperatures in the equatorial Pacific Ocean, referred to as the El Niño Southern Oscillation (ENSO), result in periodic (2 to 7 years) anomalies that affect regional air temperature and precipitation. During the El Niño phase, winter and spring conditions are generally warmer and drier than average (McCabe and Dettinger 1999). During the opposite La Niña phase, winter and spring are generally wetter and cooler, leading to a deeper than average snowpack (Gershunov et al. 1999). The Pacific Decadal Oscillation (PDO) is defined by fluctuations in sea surface temperature in the Pacific Ocean and has longer characteristic periodicity of 20 to 30 years (Mantua et al. 1997), although the PDO is not consistent over time at these frequencies (McAfee 2014) and has exhibited variable regime transitions in the pre-instrumental period (Gedalof and Smith 2001). Newman et al. (2016) pointed out that the PDO is not an independent phenomenon, but a combination of multiple processes that include ENSO. The relationship between ENSO and PDO is weaker in northern California where the respective controls of ENSO and PDO on climate are less predictable (Wise 2010).

Fire History

Regional drought driven by teleconnections with sea surface temperature anomalies (e.g., PDO, ENSO) resulted in synchronous occurrence of fires in the NWFP area (Hessl et al. 2004, Trouet et al. 2006, Weisberg and Swanson 2003, Wright and Agee 2004), as well as elsewhere in the Pacific Northwest and other regions of the Western United States (Heyerdahl et al. 2008, Kitzberger et al. 2007, Schoennagal et al. 2005). Several fire history studies document fire frequency over the past 400 years (table 2-1). Historical fire regimes differed among individual vegetation zones as well as geographically within vegetation zones (see chapter 3 for an indepth discussion). Fire was generally infrequent in most moist vegetation zones but ranged from about 50 years to >200 years, with synchronous, regional fire episodes occurring across the region from the 1400s to the mid 1600s, and again from the early 1800s to approximately 1925 (Weisberg and Swanson 2003). Fire was far more frequent in dry vegetation zones, where return intervals were shorter, generally ranging from 10 to 50 years until the late 19th and early 20th century.

Table 2-1—Fire history studies in the Northwest Forest Plan area by vegetation zone

Vegetation zone	Study	Extent (time period)	Method	Frequency/ return interval	Rotation	Low/moderate/ high	High-severity patch size
		Hectares		*Years*	*Years*	*Percent*	*Hectares*
Redwood:							
	Stuart 1987	300 (1898–1940)	Scars	7.8	—	—	—
	Finney and Martin 1989	~600 (1300–1860)	Scars	10.1	—	—	—
	Brown and Swetnam 1994	<1000 (171–1962)	Scars	9.9	—	—	—
	Brown et al. 1999	Unknown	Age, scars	7–13	—	—	—
	Brown and Baxter 2003	20 316 (1550–1930)	Scars	6–20	—	—	—
	Stephens and Fry 2005	~1000 (1615–1884)	Scars	12	—	—	—
Western hemlock:							
	Means 1982	Unknown	Scars	100	—	—	
	Fahnestock and Agee 1983	Western Washington (pre-1934)	Age class from historical survey records	—	598	—	—
	Stewart 1986	<1 (~1200–1982)	Age, scars	50[a]	—	—	—
	Yamaguchi 1986	Unknown (post-1480)	Age, scars	40–150	—	—	—
	Teensma 1987	11 000 (1482–1952)	Age, scars	114	78	—	—
	Agee et al. 1990	3500 (1573–1985)	Age, scars	137	—	—	—
	Morrison and Swanson 1990	1940 (1150–1985)	Age, scars	96	95	0–86/0–60/ 0–100	<110 ha
	Garza 1995	3540 (pre-1910)	Age, scars	93–158	134	24–41/9–23/ 25–54	
	Impara 1997	~140 000 (1478–1909)	Age, scars	85	271	—	—

Table 2-1—Fire history studies in the Northwest Forest Plan area by vegetation zone (continued)

Vegetation zone	Study	Extent (time period)	Method	Frequency/ return interval	Rotation	Low/moderate/ high	High-severity patch size
		Hectares		*Years*	*Years*	*Percent*	*Hectares*
	Wetzl and Fonda 2000	2500 (1400–1985)	Age, growth release	21.3[b]	—	—	—
	Agee and Krusemark 2001	26 000 (pre-1900)	Age, live residual structure from air photos	—	296	7–9/18–31/ 62–90	—
	Robbins 1995	~1562 km^2 (1700–1990)	Age, scars	49 (2–191)	—	—	—
	Olson and Agee 2005	~7000 (1650–1900)	Age, scars	2–167	—	—	—
	Weisberg 2009	14 504 (1550–1849)	Age, scars	—	162	—	—
	Wendel and Zabowski 2010	1873 (1568–2007)	Age, scars	127	140	—	—
Silver fir:							
	Hemstrom and Franklin 1982	~53 000 (1200–1850)	Age	—	465	—	—
	Fahnestock and Agee 1983	Western Washington (pre-1934)	Age class from historical survey records	—	834	—	—
	Agee et al. 1990	3500 (1573–1985)	Age, scars	108–137	—	—	—
	Morrison and Swanson 1990	1940 (1150–1985)	Age, scars	239	149	0–80/0–78/ 0–100	<50
	Garza 1995	3540 (pre-1910)	Age, scars	154–246	—	24–57/ 20–22/45–50	—
Mountain hemlock:							
	Dickman and Cook 1989	18 000 (post-1400)	Age	—	—	—	>3200
	Fahnestock and Agee 1983	Western Washington (pre-1934)	Age class from historical survey records	—	598	—	—

Table 2-1—Fire history studies in the Northwest Forest Plan area by vegetation zone (continued)

Vegetation zone	Study	Extent (time period)	Method	Frequency/ return interval	Rotation	Low/moderate/ high	High-severity patch size
		Hectares		*Years*	*Years*	*Percent*	*Hectares*
	Agee et al. 1990	3500 (1573–1985)	Age, scars	137	—	—	—
Subalpine:							
	Fahenstock and Agee 1983	Western Washington (pre-1934)	Age class from historical survey records	—	800	—	—
	Agee et al. 1990	3500 (1573–1985)	Age, scars	109	—	—	—
Douglas-fir and grand fir/white fir:							
	Leiberg 1903	Southern Oregon (~1900)	Historical land survey	—	—		~14 000
	Weaver 1959	Unknown	Scars	47	—	—	—
	Agee et al. 1990	3500 (1573–1985)	Age, scars	52–93	—	—	—
	Agee 1991	197 (1760–1930)	Age, scars	16	37–64	—	—
	Bork 1984	~100 (pre-1900)	Scars	8	—	—	~400
	Wills and Stuart 1994	~20 (1745–1849)	Age, scars	10.3–17.3	—	—	—
	Taylor and Skinner 1998	1570 (1627–1849)	Age, scars	14.5	19	59/27/14	—
	Van Norman 1998	45 000 (1480–1996)	Age, scars	123	—	—	—
	Brown et al. 1999	2000 (1820–1945)	Age, scars	7.7–13	—	—	—
	Everett et al. 2000	3240–12 757 (~1700–1860)	Scars	6.6–7	11–12.2	—	2.4–40
	Stuart and Salazar 2000	~120 (1614–1944)	Age, scars	27 (12–161)	—	—	—

Table 2-1—Fire history studies in the Northwest Forest Plan area by vegetation zone (continued)

Vegetation zone	Study	Extent (time period)	Method	Frequency/ return interval	Rotation	Low/moderate/ high	High-severity patch size
		Hectares		*Years*	*Years*	*Percent*	*Hectares*
	Taylor and Skinner 2003	2325 (pre-1905)	Age, scars	11.5–16.5	19	—	—
	Wright and Agee 2004	~30 000 (1562–1995)	Scars	19–24	—	—	10–100
	Hessburg et al. 2007	~72 000 (~1930)	Historical aerial photos	—	—	18/58/24	~10 000
	Baker 2012	140 400 (~1770–1880	Live structure from historical inventory	—	496[c]	18/59/23	—
Ponderosa pine:							
	Weaver 1959	Unknown	Scars	11–16	—	—	—
	Soeriaatmadja 1966	(1500–5000) Unknown	Scars	3–36	—	—	—
	West 1969	Unknown	Age	—	—	—	<0.26
	Bork 1984	~100 (pre-1900)	Scars	4–7	—	—	—
	Morrow 1985	2 (pre-1900)	Age	—	—	—	<0.35
	Hessburg et al. 2007	~106 000 (1930–1940)	Live structure from historical aerial photos	—	—	30/58/12	—
	Baker 2012	123 500 (~1770–1880)	Live structure from historical inventory	—	705[c]	40/44/16	—

[a] Stewart noted 15 fires over a 750-year period.
[b] Estimated at a 200-ha scale.
[c] Rotation for high severity only.
Note: Most fire history studies are based on fire scars or identification of cohorts of trees with similar establishment dates. Fire frequency or return interval are the most commonly reported metrics of fire activity in fire history studies. Another metric related to fire frequency is fire rotation, or the time it takes to burn an area equal to the size of the area of interest. Relatively few studies report fire severity.

20th-Century Climate Change in the Northwest Forest Plan Area

Increases in temperature and precipitation across the NWFP area during the 20th century exceeded average global increases and vary across the region as well as among seasons (Abatzoglou et al. 2014b, Mote 2003). Most of the research examining 20th-century climate in the Plan area has been aggregated to the scale of individual states (i.e., California, Oregon, and Washington), or summarized for the entire Western United States, and there is less work that focuses specifically on the Plan area. There is evidence supporting both strong human-caused climate change (Abatzoglou et al. 2014a, 2014b) and temperature increases associated with ocean/atmospheric variability (Johnstone and Mantua 2014a, 2014b). However, Abatzoglou et al. (2014a) demonstrated that natural factors alone cannot explain warming in the region.

Average annual temperature in western Oregon and Washington increased by 1.6 °F (0.91 °C) during the 20th century, with the greatest increase of 3.3 °F (1.83 °C) occurring during winter (Abatzoglou et al. 2014b, Mote 2003). Likewise, precipitation during the same period also increased by 13 percent, with the greatest increase of 37 percent during spring (Abatzoglou et al. 2014b, Mote 2003). California also experienced accelerated warming since 1970 (Cordero et al. 2011) and recently experienced the hottest, driest period (2012 to 2014) in the observational record (Mann and Gleick 2015). This same period also includes the lowest precipitation in recorded history (Diffenbaugh et al. 2015) and potentially in the past 1,200 years (Griffin and Anchukaitis 2014). In northwestern California, Rapacciuolo et al. (2014) estimated that mean temperature increased by 0.3 °F (0.18 °C). The same study estimated that minimum temperature increased by 0.9 °F (0.47 °C) and maximum temperature decreased by 0.4 °F (0.24 °C) during the 20th century, although these trends were calculated using temporal differencing rather than traditional slope-based trends, and do not necessarily account for differences in the density of weather stations used in the study (Rapacciuolo et al. 2014). Twentieth-century trends in precipitation differed across northern California with evidence of overall increases (Killam et al. 2014) as well as slight decreases in some parts of the NWFP area (Rapacciuolo et al. 2014).

Climate trends across the region are similar to those reported from studies across the Western United States. These studies indicate changes in several characteristics of weather relevant to forest and vegetation dynamics. Spring (March to May) temperature increased approximately 1.8 °F (1 °C) from 1950 to 1998 (Cayan et al. 2001) and snowpack declined during the latter half of the 20th century (Knowles 2015, Mote et al. 2005). Increases in winter temperature are linked with decreases in snowpack (Mote 2006) and earlier snowmelt, which have altered streamflow timing (Hamlet et al. 2005; Jung and Chang 2011; Stewart et al. 2004, 2005). Decreases in the proportion of annual precipitation falling as snow (Klos et al. 2014), the amount of water contained in spring snowpack (i.e., the depth of water if the snow were to melt) (Hamlet et al. 2005), and increased evapotranspiration from longer growing seasons increased soil water deficits since the 1970s (Abatzoglou et al. 2014b). A longer freeze-free season, an increase in the temperature of the coldest night of the year, and increased potential evapotranspiration during the growing season also occurred during this period (Abatzoglou et al. 2014b). Fog frequency along the coast of northern California declined by 33 percent during the 20th century (Johnstone and Dawson 2010), as has low summertime cloudiness (Schwartz et al. 2014). Most recently, northern California experienced a dramatic shift with extreme drought conditions from 2012 to 2016 followed by extreme precipitation events and severe flooding (Wang et al. 2017). Remote-sensing studies indicate that most vegetation zones accross the NWFP area have already experienced moisture stress associated with drought and high temperatures during the early 21st century across the entire NWFP area (Asner et al. 2016, Cohen et al. 2016, Mildrexler et al. 2016).

Projecting Climate Change for the 21st Century

Atmosphere-ocean general circulation models (GCMs) are the primary tools for projecting future climate scenarios (e.g., IPCC 2014). GCMs incorporate interactions among several important components of the Earth's climate system, including atmosphere, land, ice, and ocean to simulate past and future climate at relatively coarse spatial scales (~0.25 to 14 mi^2 [~0.65 to 36.3 km^2]) based on different scenarios of increasing greenhouse gas concentrations in the atmosphere. Because of differences in model formulation and sensitivity to forcing from physical influences on the atmosphere (e.g., greenhouse gases), GCM projections using the same initial conditions and emissions scenario differ (Lynn et al. 2009), as do projections from the same GCM owing to natural climate variability within a region (Deser et al. 2014).

An ensemble of projections (combinations of projections from multiple GCMs) is commonly used in climate change studies to capture the range and patterns of variability among projections. Ensemble averages appear to provide the best estimates of observed climate (Pierce et al. 2009, Rupp et al. 2013). The range of projections in an ensemble also provides a measure of the amount of uncertainty, which increases as projections extend farther into the future (Tebaldi and Knutti 2007). Uncertainty in climate change projections can be attributed to three main factors: (1) climate change-scenario uncertainty, (2) model-response uncertainty, and (3) natural variability in climate (Hawkins and Sutton 2009). For a given climate change scenario, uncertainty in the warming estimates arises from differences in GCM formulation and parameterization. Natural climate variability presents the greatest uncertainty in the near to mid term for projecting climate change for the first half of the 21st century (Hawkins and Sutton 2009) and poses a major challenge for analyzing and communicating climate change variability within a region (Deser et al. 2012).

For its fifth and most recent assessment (AR5), the Intergovernmental Panel on Climate Change published a set of future scenarios that describe estimated trajectories of greenhouse gas concentrations. These scenarios are called representative concentration pathways (RCP), and each scenario is named after the increase in radiative forcing relative to preindustrial levels. Each pathway is the result of plausible future trends in human population growth, economic and technological development, and energy systems, as well as social beliefs and values that affect human behaviors influencing emissions and climate warming (van Vuuren et al. 2011). Climate change scenarios (e.g., climate changes that are likely given a specific RCP) are considered to be plausible and do not have probability distributions associated with them (Collins et al. 2014). Current rates of greenhouse gas emissions have exceeded previously anticipated concentrations, thus there is currently insufficient information to rule out any scenario (Manning et al. 2010, van Vuuren et al. 2010). All scenarios project increases in global mean temperatures, but there is a large range among the scenarios bracketing the low and high ends of potential greenhouse gas concentrations. Under the RCP 2.6 scenario, which represents strong mitigation action, global mean temperatures are projected to increase by 2.9 °F ± 0.7 °F (1.6 °C ± 0.4 °C) by the end of the century, while under RCP 8.5, the no-mitigation, high-growth scenario, the degree of warming is projected to be 7.7 °F ± 1.3 °F (4.3 °C ± 0.7 °C) (Collins et al. 2014). Changes in global precipitation are projected to increase 0.5 to 4 percent/°C under RCP 2.6 and by 1 to 3 percent/°C under other scenarios (Collins et al. 2014).

Many relevant studies, especially in northern California, use an earlier generation of climate change scenarios published in the Special Report on Emissions Scenarios (Nakicenovic and Swart 2000). In this set of scenarios, the A2 scenario represents a very heterogeneous world with continuously increasing global population. The B1 scenario represents a convergent world in which population peaks mid-century, then declines, transitioning to resource-efficient technologies. The B2 scenario is intermediate between A2 and B1, with population growth lower than the A2 and a less rapid transition to resource-efficient technologies.

21st-Century Climate Change Projections for the Northwest Forest Plan Area

Analysis of GCM projections for Oregon and Washington (Mote et al. 2014) and northern California (Cayan et al. 2008, 2016; Garfin et al. 2014) depict a future with significant warming by the end of the 21st century, although the magnitude of warming varies at finer scales across the region. In Oregon and Washington, Dalton et al. (2013) projected increases in annual average temperature of 4.3 °F (2.4 °C) and 5.8 °F (3.2 °C) by the middle of the century (2041 to 2070) under RCP 4.5 and RCP 8.5 scenarios, respectively. By the end of the century (2070 to 2099), average annual temperature is projected to warm by 5.9 °F (3.3 °C) to 17.5 °F (9.7 °C), depending on the scenario (Mote et al. 2014). Warming is projected to occur across all seasons, with the greatest temperature increases occurring during summer months (Dalton et al. 2013).

Projected changes in precipitation are more uncertain in Oregon and Washington. Some models project a 10 percent decrease in annual precipitation by the end of the century (2070 to 2099) while others project as much as an 18 percent increase in precipitation (Mote et al. 2014). GCMs generally project wetter winters and drier summers (Dalton et al. 2013). Under the A2 and B2 scenarios, no-analog temperature conditions are projected by 2100 across much of the western Cascades and Klamath Mountains compared with those occurring in the recent past (Saxon et al. 2005). Under RCP 8.5, most of Oregon and Washington are projected to depart from their historical climate regime by 2050, when the mean annual temperature of a given location will exceed the 20th-century range of variability (Kerns et al. 2016).

In northern California, under the mitigation-oriented B1 scenario, annual temperature is projected to increase by 2.7 °F (1.5 °C) by 2100, and, under the high-growth A2 scenario, the increase is projected to be 8.1 °F (4.5 °C) (Cayan et al. 2008). Simulations depict drier futures under the B1 and A2 scenarios, with total annual precipitation decreasing by 18 percent in the more extreme A2 scenario (Cayan et al. 2008). Increases in temperature are projected for all seasons across northern California, with the greatest increases occurring during summer months (Cayan et al. 2008). Projected decreases in summer precipitation range from 4 to 68 percent, whereas projected changes in precipitation during winter months range from a 9 percent decrease to a 4 percent increase. More recent projections of increases in winter precipitation using the RCP 8.5 scenario show a high degree of agreement among models (Neelin et al. 2013). Interannual variability is expected to increase with the occurrence of greater wet and dry extremes during the wet season (October to March) (Berg and Hall 2015). Most of northern California is projected to depart from its 20th-century climate by the year 2040 (Kerns et al. 2016). The projected future climate in the Klamath Mountains represents conditions of temperature and precipitation not experienced in the recent past by 2100 under the A2 and B2 scenarios (Saxon et al. 2005). Temperature is projected to depart the 20th-century range of variability between 2046 and 2065 under the A2 scenario (Klausmeyer et al. 2011).

Implications of Observed Climate Trends for Water Balance Deficit and Vegetation Change

Changes in the magnitude and seasonality of temperature and precipitation patterns will most likely affect vegetation by altering the availability of water in the soil. Cumulatively, these are expected to be experienced ecologically through hotter periods of drought and greater deficits in water balance. Water-balance deficit for vegetation is defined as the difference between potential evaporation and actual evapotranspiration (Stephenson 1998). Ecologically, the water-balance deficit equates to the difference between the atmospheric demand for water from vegetation and the amount of water that is actually available to use. Even if precipitation remains similar to 20th-century levels, projected increases in temperatures could reduce the amount of soil moisture available for plants.

Projections for changes in water-balance deficits differ among models (Littell et al. 2016) and across the region (fig. 2-5). The majority of the region is projected to experience an increased summer (June, July, August, and September) water-balance deficit during the middle part of the 21st century. The eastern Cascades, Klamath Mountains, and southern portion of the western Cascades in Oregon will likely experience the greatest increases in water-balance deficit, as well as the southeastern portion of the Oregon Coast Range and the northern portion of the California Coast Range. The least amount of change is projected in the northern portions of the Coast Range along the Pacific Ocean. Higher elevations of the Olympic Peninsula and the northern portion of the western Cascades in Washington are projected to experience less summer water-balance deficit in the future.

Although trends in average temperature and precipitation provide some context for vegetation change in the future, individual weather events are also expected to be important drivers of future dynamics (Jentsch et al. 2007). Climate extremes (e.g., acute drought) related to changes in the variability of temperature and precipitation may have disproportionate effects on vegetation and result in rapid vegetation change (e.g., Allen and Breshears 1998). Increased frequency and intensity of heat waves and extreme temperatures are predicted across North America by the end of the 21st century (Meehl and Tebaldi 2004). Prolonged heat waves (Bell et al. 2004), as well as dry daytime and humid nighttime heat waves, are projected in northern California (Gershunov and Guirguis 2012). Models project increases in the number of both dry days and very heavy precipitation days during the wet season in northern California (October to March) (Berg and Hall 2015). This is consistent with an intensified water cycle characterized by shifts from extreme drought to years with anomalously high precipitation (Wang et al. 2017). Increases in peak flow magnitudes also suggest greater potential for flooding in portions of inland northern California (Das et al. 2013), where floods may be more frequent and severe (Dettinger 2011, Salathé et al. 2014). Heavy precipitation events from warming and shifts in seasonal precipitation patterns may also increase flooding in most of Oregon and Washington (Tohver et al. 2014) and the northern California Coast Range (Kim 2005). Rain-on-snow events may also be more common given warmer winter and spring temperatures, which are also projected to alter the timing of seasonal streamflow (Elsner et al. 2010). The availability of regional climate model outputs provides the climatic basis for better simulating physically consistent extremes relevant to forests processes (e.g., McKenzie et al. 2014, for fires), but these outputs are also subject to the constraints of GCMs used as boundary conditions.

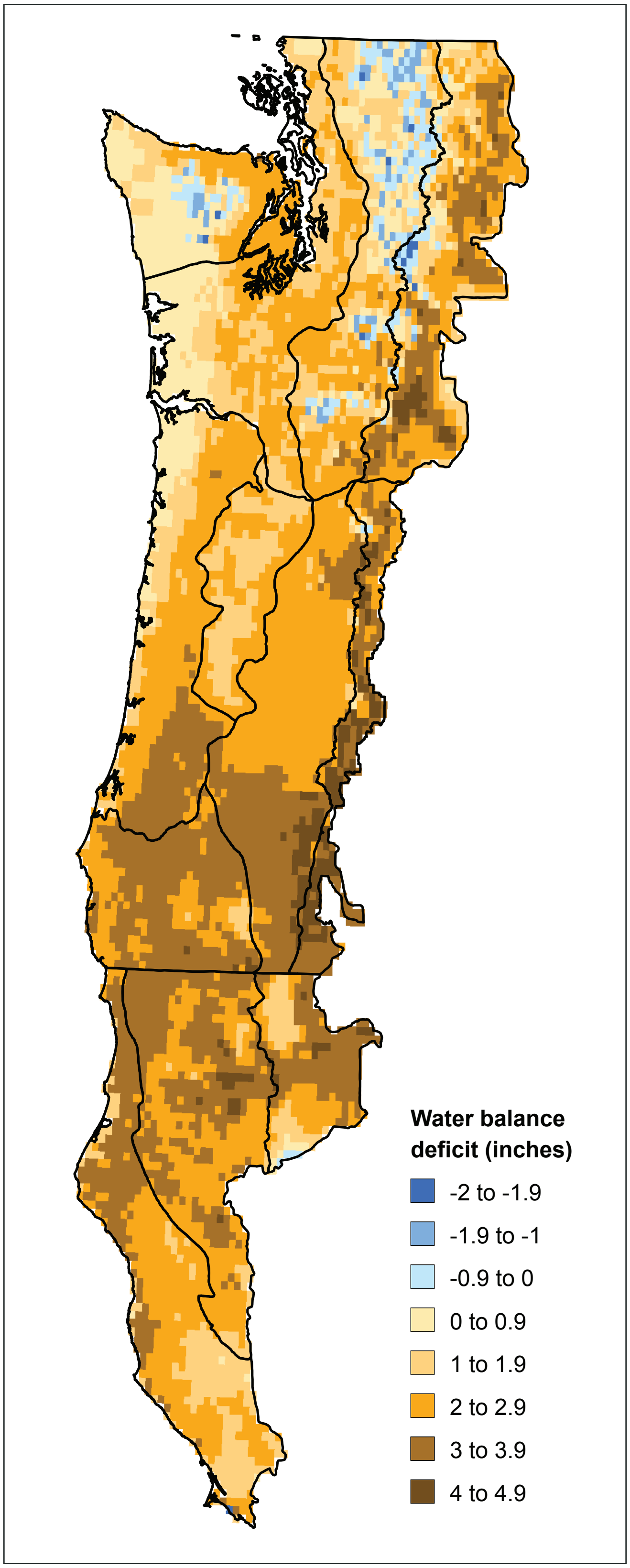

Figure 2-5—Projected changes in summer (June, July, August, and September) water-balance deficit across the Northwest Forest Plan area for 2030–2059 from a composite of the 10 best general circulation model projections based on the CMIP3/AR4 scenarios following Littell et al. (2016). Higher water-balance deficit (browns) means decreased water available for plant uptake. Change is compared to the water-balance deficit from 1916 to 2006. Map boundaries correspond with the physiographic provinces in figure 2-1.

Considering the coarse resolution of climate projections (~0.25 to 14 mi^2 [~0.65 to 36.3 km^2]), it is important to recognize the potential for landscape-scale variability in future climate and vegetation change. Differences in vegetation structure and topography can drive fine-scale variation in temperature extremes, with differences in maximum and minimum temperatures of similar magnitude to those projected at a broader scale in different climate change scenarios (Suggitt et al. 2011). Spatial variability in bedrock geology also has the potential to mediate seasonal changes in groundwater availability associated with increased temperature (Tague et al. 2008). Complex topography and cold air pooling may decouple climate conditions in mountain valleys from the surrounding landscape (fig. 2-6) (Daly et al. 2009), and snow may persist later in the season in canopy gaps and topographic depressions (Ford et al. 2013). Temperature is generally lower and soil moisture higher in interior late-successional forests than in clearcuts or edges (Chen et al. 1993), and denser canopies can attenuate warming by providing shade to the forest floor (De Frenne et al. 2013). Recent findings also indicate that dense, old-growth forests in moist vegetation zones of the region have the potential to provide cooling effects at local scales (Frey et al. 2016). Thus, the actual changes in future climate experienced by an organism may differ depending on their tolerances or habitat preference.

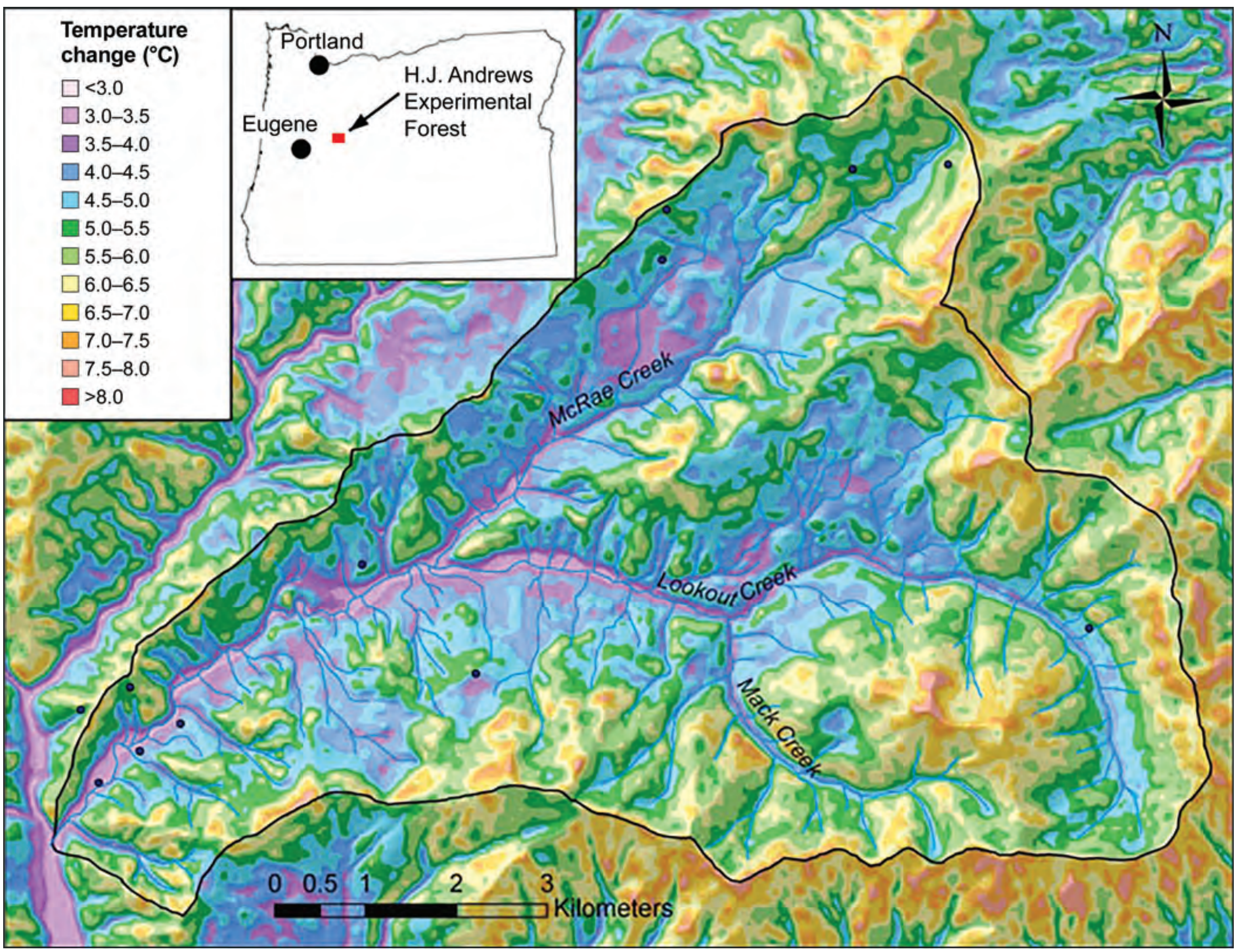

Figure 2-6—Projected changes in maximum December temperatures in response to a 2.5 °C regional temperature increase and changes in atmospheric circulation patterns in the western Cascade Range of Oregon. Source: Daly et al. (2010).

The potential for relatively stable climatic conditions at finer scales in some landscapes (e.g., topographically complex, mountainous terrain) suggests an important role for climatic refugia to contribute to the future persistence of some species (Noss 2001). Despite the conceptual appeal and historical importance of refugia, identification of refugia has proven difficult and has been largely descriptive, and refugia are likely to be species- and process-specific (Keppel et al. 2012). Refugia will most likely be found in topographically complex landscapes where microclimates differ because of differences in aspect, shading and insolation, and cold-air drainages (Dobrowski 2011). These areas may provide potential for species persistence through unfavorable climatic conditions, as well as sources for future recolonization provided that suitable conditions return in the future.

Mechanisms of Vegetation Change

Climate change is expected to alter vegetation through direct effects (e.g., from CO_2 and climate on vegetation processes) and indirect effects (e.g., from disturbance processes). The direct effects of climate change and increasing CO_2 on vegetation are expected to be expressed through changes in mortality, growth, and reproduction, all of which may be sensitive to altered phenology and biotic interactions within and among species (Peterson et al. 2014a). The indirect effects of climate change are expected to be expressed through increases in the frequency, severity, and extent of disturbances, particularly drought, fire, insects, and pathogens. These have the potential for rapid ecological change at landscape scales, and are predicted to be a greater driver of ecological change than direct effects (Dale et al. 2001, Littell et al. 2010). The relative importance of these drivers, however, is likely to vary geographically across the region among species, seral stages, physiographic provinces, and disturbance regimes. Species are expected to respond individualistically to future changes in climate as they have in the past (Whitlock 1992).

Direct effects of climate change: demographic responses— Tree mortality from higher temperatures and drought stress has already occurred in many forests of the Western United States, and is expected to increase in the 21st century (Allen et al. 2010, 2015). Warmer temperatures and increased frequency and duration of droughts projected for the NWFP area are likely to increase climate-induced physiological stress on plants (Adams et al. 2009). Drought-related stress can lead to two separate, but not mutually exclusive, mechanisms of tree mortality including hydraulic failure (irreversible desiccation and collapse of water transport structures) and carbon starvation (McDowell et al. 2008). Although there has been much recent work on the physiological mechanisms associated with tree mortality, a greater understanding of these mechanisms is needed to assess vulnerability among species and enhance our ability to predict mortality (Hartmann et al. 2015). Furthermore, a better understanding of the ecological consequences of mortality in terms of community-level change (i.e., structure and composition) and ecosystem function is needed (Anderegg et al. 2012).

Mortality rates in old-growth forests in the Plan area have increased above most published rates (>1 percent/year) since the mid 1970s (van Mantgem et al. 2009). A regional study on mortality rates on Forest Service lands in Oregon and Washington corroborated the occurrence of elevated mortality rates in old-growth forests across all vegetation zones from the mid 1990s to mid 2000s during regionwide drought (Reilly and Spies 2016). However, Acker et al. (2015) found that mortality rates in old-growth forests on National Park Service lands (Olympic National Park, North Cascades National Park) in western Washington were lower than those reported by van Mantgem et al. (2009) and Reilly and Spies (2016). Lower mortality rates could be due to geographic variation not represented in van Mantgem et al. (2009) and Reilly and Spies (2016), but may also be indicative of decreasing stress-related mortality following a period of elevated mortality. Consistent with this idea, Cohen et al. (2016) found that remotely sensed forest decline peaked in the mid 2000s during the warmest decade in the past 100 years (Abatzoglou et al. 2014b), then decreased.

Increasing tree mortality rates have been documented in young stands of other regions, and some researchers suggest that they may be more vulnerable to changes in climate than old-growth stands (Luo and Chen 2013). However, Reilly and Spies (2016) found that mortality rates in early- and mid-seral stages from the mid 1990s to mid 2000s were lower than rates in young forests in the western hemlock and silver fir zones of the western Cascades (Larson et al. 2015, Lutz and Halpern 2006). With the exception of old-growth forests, in which increased mortality led to cumulative losses in basal area and density (van Mantgem et al. 2009), there is generally poor understanding of the effects of recent mortality on stand structure and composition, as well as how these effects differ around the region.

The potential response of tree growth to climate change differs substantially among species depending on the factors that limit growth such as water and length of growing season (Littell et al. 2010, Peterson and Peterson 2001). Growth in Douglas-fir is predicted to decrease under climate change where it currently is water limited (Restaino et al. 2016), but growth may increase where Douglas-fir is limited by growing-season length or lower than optimal temperatures (Albright and Peterson 2013; Creutzburg et al. 2017; Littell et al. 2008, 2010). In species of high-elevation forests where growth is limited by temperature and growing-season length (e.g., subalpine fir, mountain hemlock), growth increased during the 20th century because of warmer winter temperatures and longer growing seasons (McKenzie et al. 2001, Nakawatase and Peterson 2006, Peterson et al. 2002). Warmer winters and earlier snowmelt may also increase potential for drought and water stress in higher elevation forests, especially toward the southern portion of their distribution in southern Oregon and northern California. However, these effects are not yet well documented or understood, and increased growth is expected to continue in the future (Albright and Peterson 2013). The effects of projected climate change on ponderosa pine is uncertain as wetter fall seasons may increase growth while drier summers decrease growth (Kusnierczyk and Ettl 2002). These effects may differ across the landscape as ponderosa pine and western juniper may be more sensitive to drought at lower elevations (Knutson and Pyke 2008). The response of these species also depends on the potential for CO_2 to enhance growth by increasing water-use efficiency (Soule and Knapp 2006). However, some evidence suggests that any benefits of CO_2 fertilization will be outweighed in the future as the climate warms and water becomes a more limiting factor (Gedalof and Berg 2010, Restaino et al. 2016). Increased levels of CO_2 also have the potential to accelerate maturation and increase seed production (LaDeau and Clark 2001, 2006), but little information is available on the effects of climate change on reproduction in species of the region.

The ability of a species to respond to changes in climate (e.g., earlier warming and drying) with shifts in phenology will be an important factor in determining responses to projected climate change. Altered seasonality may affect growth and reproduction in some plant species. A major concern in the NWFP area associated with warmer winters and earlier springs is the requirement for many species (e.g., Douglas-fir, western hemlock, *Pinus* spp., *Abies* spp.) to experience chilling for the emergence of new leaves, or budburst (Harrington and Gould 2015). Douglas-fir may experience earlier budburst in some portions of its range because of warming, but reduced chilling may cause later budburst in the southern portion of its range (Harrington and Gould 2015). Earlier growth in northern and higher elevation portions of Douglas-fir's range may lead to earlier growth initiation, but reduced chilling in the southern and lower elevation portions of its range are likely to lead to delayed growth initiation (Ford et al. 2016).

Climate change may also affect interactions among and within species in complex ways, but the effects are currently poorly understood. However, several recent studies from higher elevation moist forests in the silver fir vegetation zone of Washington provide some insights. For example, the negative effect of competition on growth is likely to be greater for saplings than for adults, and climate change may have less effect on closed-canopy forests at lower elevations than at higher elevations (Ettinger and HilleRisLambers 2013). Individual growth is likely to increase most in lower density stands as trees may show little response to climate at higher density (Ford et al. 2017). Little is known about the effects of climate change on positive species interactions (e.g., facilitation), though they are known to be important in stressful subalpine environments elsewhere in the Western United States (Callaway et al. 2002), and are thought to play a role in early stand development in dry and cold vegetation zones (e.g., ponderosa pine, subalpine, mountain hemlock) in the NWFP area (Reilly and Spies 2015).

Indirect effects of climate change: disturbance—
The indirect effects of climate change will likely be expressed through increases in the frequency, severity, and extent of disturbance, and are predicted to be the primary mechanisms of ecological change in the future (Dale et al. 2001, Littell et al. 2010). Disturbances include discrete events that alter the structure and function of ecosystems (Pickett and White 1985), but may also include prolonged droughts or multi-year epidemics of pathogens and insects. Disturbance agents are commonly characterized as biotic (e.g., pathogens, insects) or abiotic (e.g., fire, wind, volcanoes), and differ considerably in terms of their prevalence and severity (i.e., tree mortality) across the region and among vegetation zones (Reilly and Spies 2016) (chapter 3). There is great concern that interactions among climate change, forests, and disturbance regimes may result in disturbance effects outside of the natural range of variation (Dale et al. 2000).

Of particular concern are multiple, successive, or compound disturbances (e.g., Paine et al. 1998). Interactions among multiple disturbances may result in multiplicative effects on the structure and function of ecosystems that differ from the cumulative effects of both individual disturbances. The effects of compound disturbances are difficult to predict, but may amplify disturbance severity, cause changes between ecological states (e.g., forest to nonforest transitions), and decrease forest resilience (Buma 2015). However, despite growing recognition and interest in interactions among disturbances, the effects of compound disturbances remain poorly characterized and difficult to predict (Buma 2015, Seidl et al. 2017).

Biotic disturbances—
Biotic disturbances (e.g., insects and pathogens) elevate stand-scale mortality above what are considered normal "background mortality rates" associated with competition and stand development, but may also erupt into epidemic outbreaks that result in high levels of tree mortality (e.g., Raffa et al. 2008). Insects and pathogens do not always result in immediate tree mortality. However, the resulting decline in tree growth and vigor (Hansen and Goheen 2000, Marias et al. 2014) may initiate a long process of mortality (Manion 1981), making trees less resistant to wind disturbance and predisposing them to stem breakage (Larson and Franklin 2010). Although mortality rates associated with insects are generally much lower than those associated with fire in this region (Reilly and Spies 2016), insects resulted in greater loss of live carbon (Berner et al. 2017) and greater canopy mortality (Hicke et al. 2016) than fire in recent years at the regional scale.

Native insects and pathogen activity is expected to increase as trees experience more stress associated with growing-season drought; however, the implications and magnitude of their effects are likely to be variable and differ geographically as well as among species (Chmura et al. 2011, Kolb et al. 2016a, Sturrock et al. 2011). In addition to affecting host species, climate change will also affect population dynamics and geographic distributions of pathogen and insect species. Pathogen activity is likely to increase in areas where they typically infect drought-stressed host species, while the effects of climate change on pathogens that proliferate under moist conditions may be more variable and difficult to predict (Sturrock et al. 2011). Warmer winters and hotter droughts are expected to enable insects to move into previously unsuitable habitat (Bentz et al. 2010, 2016), and some regions in the Western United States experienced what are considered unprecedented outbreaks of insects in the past few decades (e.g., Raffa et al. 2008). Drought and insects may also interact to further stress trees and predispose them to mortality, but these dynamics are complex and are just beginning to be understood (Anderegg et al. 2015).

Native pathogens play a prominent but variable role in the disturbance regimes of both moist and dry vegetation zones of the region (Goheen and Willhite 2006, Hansen and Goheen 2000) (see Shaw et al. 2009 and chapter 3 for more information on insects and pathogens). Most native pathogens affect small, localized areas at low levels of tree mortality, but are pervasive and generally widespread across the region (Reilly and Spies 2016). Pathogens often initiate forest canopy gaps and can accelerate successional dynamics in old-growth Douglas-fir-dominated forests of the western hemlock vegetation zone (Holah et al. 1997). Laminated root rot (*Phellinus sulphurascens*) (formerly *weirii*) affects Douglas-fir, true firs (*Abies* spp.), and mountain hemlock. Armillaria (*Armillaria ostoyae*) affects Douglas-fir, hemlocks (*Tsuga* spp.), pines (*Pinus* spp.), and

Engelmann spruce. Annosus root disease (*Heterobasidion annosum*) affects firs, pines, hemlocks, and Engelmann spruce. Black stain root disease (*Leptographium wageneri*) affects Douglas-fir and ponderosa pine. Several other types of pathogens are also present, including rusts (*Cronartium* spp.) and mistletoes (*Arceuthobium* spp., *Phoradenron* spp.).

In the Coast Range, Swiss needle cast (*Phaeocryptopus gaeumannii*) is a disease specific to Douglas-fir that has increased since the early 1990s (Hansen et al. 2000b). Ritóková et al. (2016) found that the area affected by Swiss needle cast more than tripled between 1996 and 2015, with growth reductions of 23 percent in the Oregon Coast Range. Swiss needle cast is predicted to increase in the Oregon Coast Range in response to warmer and wetter conditions in the future (Stone et al. 2008), although an increase in drought conditions may inhibit spread of the disease (Rosso and Hansen 2003). High-density Douglas-fir plantations near the coast, where Sitka spruce and western hemlock were historically dominant, are thought to be particularly vulnerable to Swiss needle cast (Black et al. 2010, Hansen et al. 2000, Manter et al. 2003, Rosso and Hansen 2003). An extensive list of research studies of Swiss needle cast is available at http://sncc.forestry.oregonstate.edu/publications.

Several species of insects, including bark beetles and defoliators, are also native to the NWFP area. Insects are more prevalent in drier vegetation zones and affected large areas east of the Cascade Range in recent decades (Hicke et al. 2016, Meigs et al. 2015). In Oregon and Washington, recent mountain pine beetle outbreaks were positively associated with warmer winter temperatures and negatively associated with drought stress and precipitation in the current and previous year of outbreak (Preisler et al. 2012). Mountain pine beetle has the potential to cause extensive mortality in lodgepole pine (*Pinus contorta*) and also affect other species of pines, including ponderosa pine, sugar pine (*Pinus lambertiana*), western white pine (*Pinus monticola*), and whitebark pine. Defoliating insects are also common, and though they often do not result in mortality, they may reduce growth and make trees more susceptible to other insect infestations. Several species of pine are susceptible to outbreaks of pandora moth (*Coloradia pandora*), and ponderosa pine is also susceptible to pine butterfly (*Neophasia menapia*). Spruce budworm (*Choristoneura occidentalis*) is a major concern east of the Cascade Range and affects Douglas-fir and true firs. Williams and Liebhold (1995) projected decreases in the area defoliated by spruce budworm with increased temperature alone, but the area increased with increases in temperature and precipitation. Douglas-fir is also susceptible to Douglas-fir beetle (*Dendroctonus pseudotsugae*), which operates on small patches of trees, especially after blowdown from wind events (Powers et al. 1999).

Several nonnative pathogens and insects are of particular concern in the NWFP area. White pine blister rust (*Cronartium ribicola*) is a major threat to whitebark pine (Goheen et al. 2002, Ward et al. 2006) as well as both western white pine and sugar pine (Goheen and Goheen 2014). Decline of Pacific madrone (*Arbutus menziesii*) related to multiple fungal diseases has been reported over the past 30 years, with larger older trees experiencing the most mortality (Elliott et al. 2002). Balsam woolly adelgid (*Adelges piceae*) has affected subalpine fir and especially grand fir at lower elevations west of the Cascades (Mitchell and Buffam 2001). In southwest Oregon and northwest California, sudden oak death (caused by *Phytophthora ramorum*) has the potential to spread through air, water, and infected plant material (Peterson et al. 2014b, Rizzo and Garbelloto 2003) and may affect tanoak, various species of oak (e.g., California black oak [*Quercus kelloggii*]), other hardwood species (e.g., Pacific madrone and bigleaf maple [*Acer macrophyllum*]), and several species of shrubs (e.g., *Rhododendron* spp.) (see chapter 3). Warmer, wetter winters intensify risk of infection (Haas et al. 2015), and the area affected by sudden oak death is predicted to increase tenfold by the 2030s under projected warmer and wetter conditions (Meentemeyer et al. 2011). Sudden oak death is also associated with increased fire severity on soils in northwest California (Metz et al. 2011). Port Orford cedar (*Chamaecyparis lawsoniana*) is susceptible to a lethal, nonnative root pathogen (*Phytophthora lateralis*) that can be spread over long distances via organic matter carried on boots, vehicles, and animal hooves, and by water (Hansen et al. 2000a, Jules et al. 2002). Recent work suggests that despite rapid initial spread and colonization of *Phytophthora lateralis*, the rate of spread has slowed greatly since 2000 (Jules et al. 2014).

Abiotic Disturbances

Abiotic agents of disturbance in the NWFP area include windstorms, fire, volcanic eruptions, landslides, and avalanches. These disturbances result in much higher levels of tree mortality than biotic disturbances, and are the primary natural agents of stand-replacing disturbance (Reilly and Spies 2016). Abiotic disturbances can create forest gaps and patches of mortality that range in size depending on the disturbance agent (Spies and Franklin 1989). Smaller gaps created by abiotic disturbances may increase stand and landscape heterogeneity, while large, infrequent disturbances may have effects on landscape composition and structure that may persist for centuries (Foster et al. 1998) and are qualitatively different from smaller disturbances (Romme et al. 1998). More details on abiotic agents of disturbance can be found in chapter 3.

Windstorms arising from extratropical cyclones off the Pacific Ocean have the potential to produce hurricane-force winds and extensive damage to forested ecosystems, and large storms affected parts of the NWFP area several times in recorded history (Mass and Dotson 2010). These events are generally characterized by southwesterly winds and occur during the winter when soils are saturated. Coastal areas, particularly the Coast Range in Oregon and Washington, as well as the Olympic Peninsula, were subject to multiple synoptic winds events during the 20th century. Some of these storms also affected inland areas and caused substantial tree mortality in portions of the western Cascades, particularly near the Columbia River Gorge (Sinton and Jones 2002). The most intense of these events, the Columbus Day Storm of 1962 (Lynott and Cramer 1966), killed approximately 11 million board feet of timber in Oregon and Washington (Teensma et al. 1991). High-wind events are positively associated with neutral to warm PDO conditions, and their influence has shifted northward over the past 120 years (Knapp and Hadley 2012), but we are currently unaware of any published literature including future projections of the frequency or intensity of windstorms in the region.

Fire played an important role in the historical dynamics of the region (Agee 1993), but a long period of fire exclusion reduced fire activity during the mid-20th century (Littell et al. 2009). However, increases in the frequency and extent of fire across the Western United States since the mid-1980s have been attributed to longer fire seasons associated with earlier snowmelt and warmer spring and summer temperatures (Jolly et al. 2015, Westerling et al. 2006) as well as drought (Gedalof et al. 2005, Littell et al. 2009). A recent study also linked increasing fire activity to human-driven climate change, which is contributing to a more conducive fire environment by increasing fuel aridity (Abatzoglou and Williams 2016). Annual area burned has increased since the mid 1980s (Miller et al. 2012, Reilly et al. 2017). However, recent fire activity differs substantially depending on spatial scale and geographic location across the region (Davis et al. 2015, Reilly et al. 2017), and there is growing consensus that the region experienced less fire than would be expected under historical conditions (Marlon et al. 2012, Miller et al. 2012, Parks et al. 2015, Reilly et al. 2017).

The effects of recent fires have been extremely variable across the region, with most recent fire activity occurring in the Klamath Mountains, eastern Cascades, and western Cascades of Oregon (fig. 2-7). The annual area burned increased in most vegetation zones since the mid-1980s, but dry vegetation zones, including ponderosa pine, Douglas-fir, and grand fir/white fir, experienced less fire than they would have during presettlement times because of fire suppression (Miller et al. 2012, Reilly et al. 2017) (see chapter 3 for more discussion). Mean and maximum fire size from 1910 to 2008 increased in northwest California (Miller et al. 2012). Cold and moist vegetation zones (silver fir, mountain hemlock, and subalpine zones, but with the exception of western hemlock) experienced the greatest proportions of high-severity in recent fires, and most of the area burned in the previously mentioned dry vegetation zones has been at low and moderate severity (Miller et al. 2012, Reilly and Spies 2016, Reilly et al. 2017, Whittier and Gray 2016). Fire severity has been related to climate and drought at broad spatial scales since the mid 1980s (Abatzoglou et al. 2017, Keyser and Westerling 2017, Reilly et al. 2017). Although the area burned has increased in all major vegetation zones during this time, there is little evidence that the proportion burning at high severity has increased across the region (Law and Waring 2015, Miller et al. 2012, Reilly et al.

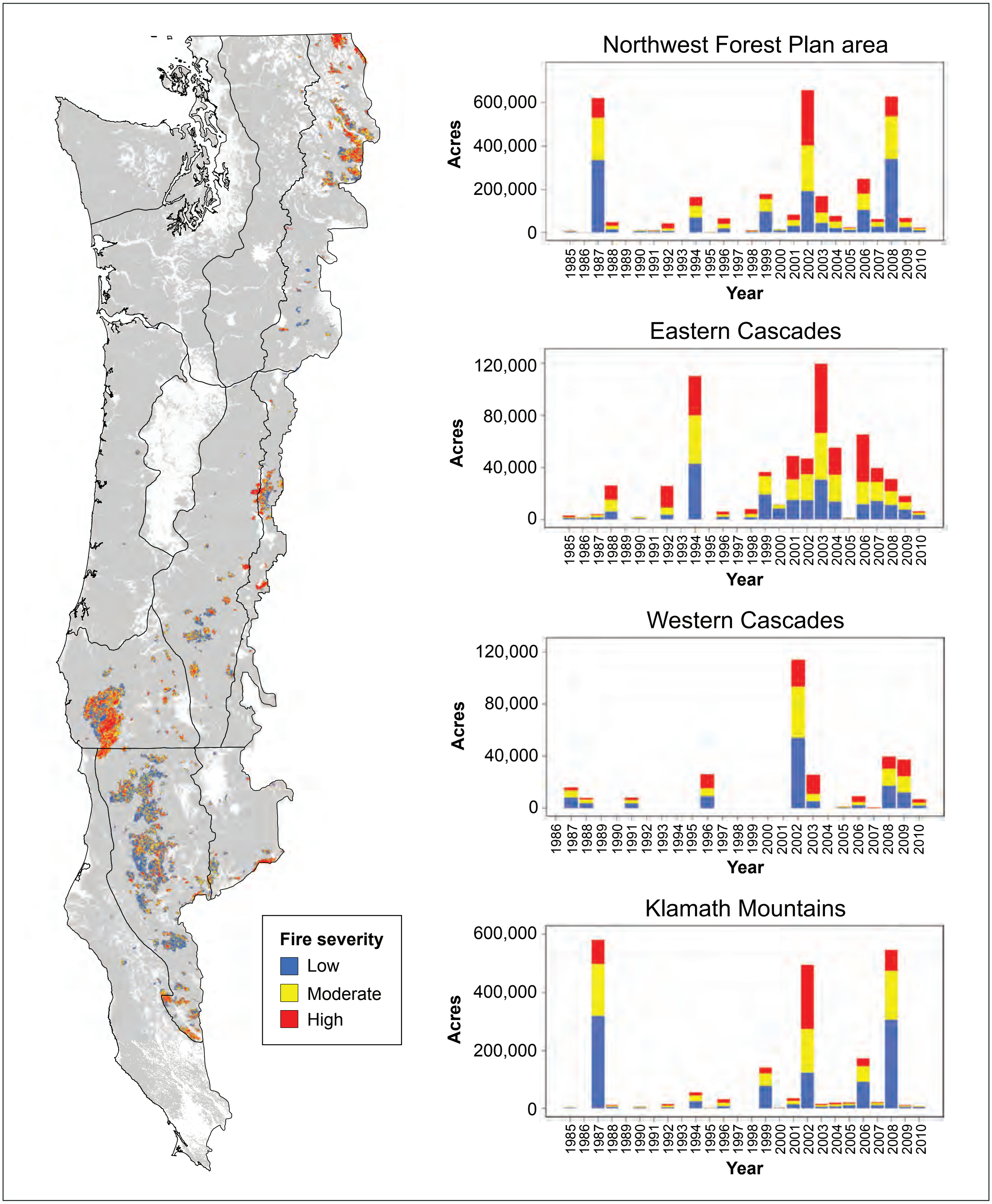

Figure 2-7—Geographic patterns of burn severity from 1985 to 2010 in the Northwest Forest Plan area. Burn severity is derived from the relativized version of the difference in the normalized burn ratio and is based on the percentage of basal area mortality as follows: low (<25 percent), moderate (25 to 75 percent), and high (>75 percent) (Reilly et al. 2017). Map boundaries correspond with the physiographic provinces in figure 2-1.

2017). Although they found no increase in the proportion of high-severity fire, Reilly et al. (2017) found that increases in high-severity patch size during this time were associated with more area burned during drought years in all major vegetation zones.

Despite concern that insect outbreaks may exacerbate fire effects by altering fuel structure (Hicke et al. 2012), there is a growing body of literature within the region and across the Western United States indicating that the two disturbances are not positively linked (Hart et al. 2015, Meigs et al. 2015), and that prefire insect activity does not make fires more severe (Agne et al. 2016, Meigs et al. 2016, Reilly and Spies 2016). These findings are also consistent with several other studies in other regions of the Western United States (Black et al. 2013, Bond et al. 2009, Donato et al. 2013, Harvey et al. 2013, Simard et al. 2011).

Hessl (2011) outlined a framework proposing three major pathways through which future fire activity may respond to climate change. Most studies to date have assumed that the major pathway to change will be based on alteration of fuel conditions as the relationships among weather, fuel moisture, and fire activity are well established. Fewer studies have focused on changes in the second pathway, alteration of fuel amount, though this may be of particular concern given its relation with severity. The least is known about the third pathway, changes in sources of ignition. This pathway will be subject to changes in lightning frequency as well as changes in human ignitions and fire-suppression efforts.

A number of studies using different techniques project increases in a variety of metrics of fire activity (i.e., area burned, fire size, fire severity, fire interval) during the 21st century, although projections differ considerably across the NWFP area (table 2-2). Most studies report coarse-scale projections (i.e., individual states), and few include details at geographic variablity within study areas (i.e., east vs. west). Stavros et al. (2014) found that the probability of very large fires will increase based on climate projections for Oregon and Washington, but increases will be minor in northern California. McKenzie et al. (2004) used statistical models and found that an increase in temperature of 3.6 °F (2 °C) will increase fire extent by 1.4 to 5 times for many Western states, including Oregon, Washington, and California. Using a similar statistical approach, Littell et al. (2010) found that area burned is likely to increase by 2 to 3 times across Washington by the end of the 2040s. They also found that area burned in the western Cascades of Washington is expected to increase by more than eight times, but on average will still affect only a small extent (9,100 ac) of the ecoregion by the 2080s. Liu et al. (2013) projected increases in fire potential associated with warming and drought from 2014 to 2070. Turner et al. (2015) projected an increase in area burned by 3 to 9 times in a portion of the central western Cascades of Oregon. Krawchuk et al. (2009) also predicted increases in fire probability in the western Cascades. Barr et al. (2010) projected an increase in annual fire extent of 11 to 22 percent in the Klamath River basin by 2100. Davis et al. (2017) projected increases in fire suitability across multiple provinces in Oregon and Washington during the 21st century (under RCP 4.5 and 8.5, respectively), including the Klamath Mountains (18 to 48–58 percent), the western Cascades (1 to 13–18 percent), and the eastern Cascades (11 to 40–45 percent). Although projections differ geographically, all studies predict increased fire activity during the 21st century.

There are few statistical predictions for moist maritime forests (i.e., Sitka spruce, redwood, western hemlock) because there has been very little area burned near the coast in the past several decades (Littell et al. 2010). Davis et al. (2017) found no increase is fire suitability in the Puget Trough and only minor increases (<1 to 2 percent) in the Coast Range. Creutzburg et al. (2017) projected very little increase in area burned by 2100 compared to the period from 1959 to 2009 in the Oregon Coast Range. Fried et al. (2004) suggested a decrease of 8 percent in area burned by fires along the north coast of California over the 21st century under continued fire-suppression efforts. Liu et al. (2013), however, predicted an increase in fire potential (measured as Keetch-Byram Drought Index) from 2.5 to 5 times owing to changes in fire weather in coastal forests by 2070. Westerling et al. (2011) projected 300 percent increases in area burned in northwest California. Krawchuk et al. (2009) projected little change in fire potential in coastal forests, but increased potential across the rest of the region. Rogers et

Table 2-2—Projections for future fire activity in the Northwest Forest Plan area from published studies

Study	Method	Geographic extent	Scenario	Time period	Projected change from current	Suppression effects	Variable
					Percent		
Stavros et al. 2014	Statistical	Oregon, Washington, northern California	RCP 4.5, RCP 8.5	2031–2060	+	No	Very large fire occurrence[a]
McKenzie et al. 2004	Statistical	Oregon, Washington, northern California	A2, B2	2070–2100	+	No	Area burned
Littell et al. 2010	Statistical	Washington	A1B	2020–2080	+200 to 300	No	Area burned
Turner et al. 2015	Process	Willamette Valley, Oregon	RCP 4.5, RCP 8.5	2100	+300 to 900	No	Area burned
Krawchuk et al. 2009	Statistical	Global	A2, B1	2070–2090	+	No	Fire probability[b]
Fried et al. 2004	Statistical	Northern California	2× CO_2	N/A[c]	-8	Yes	Area burned
Spracklen et al. 2009	Statistical	Oregon, Washington, northern California	A1B	2050	+78	No	Area burned
Barr et al. 2010	Process	Klamath Basin, Oregon, and northern California	A2	2075–2085	+11 to 22	No	Area burned
Liu et al. 2013	Statistical	Continental United States	A2	2041–2070	No	No	Fire potential[d]
Westerling et al. 2011	Statistical	Northern California	A2	2085	+100	No	Area burned
Rogers et al. 2011	Process	Oregon, Washington	A2	2070–2099	+76 to 310/29–41	Yes	Area burned/burn severity[e]
Sheehan et al. 2015	Process	Oregon, Washington	RCP 4.5, RCP 8.5	2071–2099	-82 to 14	Yes	Mean fire interval
Creutzburg et al. 2017	Statistical	Oregon	RCP 8.5	2100	Negligible	Yes	Area burned
Parks et al. 2016	Statistical	Western United States	RCP 8.5	2040–2069	No change to decrease	No	Fire severity[f]
Davis et al. 2017	Statistical	Oregon, Washington		2071–2100	No change to increase	No	Suitability for large wildfires[g]

Note: Most studies project area burned or other variables associated with increased area burned (fire suitability, large fire occurrence), and there are relatively few projections for fire severity.
[a] Very large fires are defined as those >50,000 ac.
[b] Fire probability is the probability of fire occurrence.
[c] This study does not project to an explicit time period in the future, but rather conditions based on 2× current CO^2 scenario.
[d] Fire potential is measured by the Keetch-Byram Drought Index y.
[e] Burn severity is based on combustion of biomass.
[f] Burn severity is based on a postfire composite burn index (CBI) based on changes in multiple strata, including soil and rock, litter and surface fuels, low herbs and shrubs, tall shrubs, and trees.
[g] Large wildfires are defined in this study as >40 ha.
N/A = not available.

al. (2011) used a mechanistic vegetation model (MC1) that integrates fire and suppression efforts, and found increases in area burned in Oregon and Washington from 76 to 310 percent by 2070 to 2099. Although this increase may seem high, it is important to note that the recent extent of fire in moist forest is very low, and a tripling of fire may still be a relatively small amount in absolute terms.

Although several studies have projected future increase in fire activity, far less work has been done on future fire severity. This component of fire regimes is less well studied and understood (Hessl 2011, Parks et al. 2016), potentially because of the complexities of incorporating feedbacks from fire and climate on fuel structure and arrangement at stand and landscape scales. Previous fires have the potential to inhibit the spread of subsequent fires occurring within a limited time window (Parks et al. 2014), and increased area burned in the future may provide a feedback related to decreased fuel availability. Rogers et al. (2011) used a process model (MC1) and suggested increases in burn severity of 29 to 41 percent that related to increases in productivity and biomass during non-summer months. However, a recent study incorporating changes in vegetation type, fuel load, and fire frequency predicted either no change or potential reductions in fire severity across the entire NWFP area for 2040–2069 under the most extreme climate change scenario (RCP 8.5) (Parks et al. 2016). The authors attributed decreases in fire severity to greater water deficits, decreased productivity, and less available fuel.

The wide range of projections of climate change effects on fire within the NWFP area are likely the result of several factors. These factors include differences in emissions scenarios, spatial and temporal scale, model structure (e.g., statistical vs. process), and variability in how models project precipitation. In addition, McKenzie and Littell (2017) showed that differences in climate-fire relationships among physiographic provinces are likely to be substantial, and further analysis is required to put differences in methodological and regional future projections of fire into context. At coarser regional scales, dynamical and statistical approaches to projecting future fire activity may agree, but the mechanisms operating at more local scales require careful interpretation.

Cumulative effects of climate change on tree species distributions and range shifts—

The cumulative effects of changes in mortality, growth, and recruitment will ultimately be manifest in shifts in species distributions and ranges. These effects will also depend on the size and degree of connectivity within populations. Range expansion occurs through migration and colonization at the outer limits, or "leading edge," of a species' distribution where climate is becoming more favorable. Range expansion at the leading edge is controlled by fecundity and dispersal (Thuiller et al. 2008). More vagile species that produce greater amounts of seeds and have a greater ability to disperse will have more potential to track climate change than those with poor dispersal ability. At the lower limits or "trailing edge" of a species' distribution where climate is becoming less favorable, range contraction and progressive isolation will occur through local extirpation. Range contraction is related to the ability of a species to persist in refugia that experience less change than the surrounding landscape. Individuals at the trailing edge may thus play an important role in the maintenance of genetic diversity for some species (Hampe and Petit 2005). Although local extirpation may occur throughout the range of species, small, isolated populations at the trailing edge may be particularly vulnerable as the climate changes rapidly (Davis and Shaw 2001).

It is likely that species that are more adapted to cold environments will be more sensitive to warming at their lower limits of elevation or latitude, while expansion of species adapted to warmer conditions is expected at upper range limits at high elevation or latitude (HilleRisLambers et al. 2015). Range limits may also be altered at the eastern limits of the range of some species as a result of increasing aridity. Warmer temperatures are likely to lead to range expansion at the leading edge for some species at the upper tree line, but not necessarily for species in closed-canopy forests at lower elevations (Ettinger and HilleRisLambers 2013, Ettinger et al. 2011). However, expansion at upper range limits may be limited by dispersal and low abundance of adult trees that produce seed (Kroiss and HilleRisLambers 2015). Warmer temperatures may increase germination and survival of seedlings provided adequate water, as well as increase sapling growth rates (Ettinger and HilleRisLambers 2013, Ettinger et al. 2011, HillesRisLambers et al. 2015), but many tree species are long lived and may exhibit lagged responses to climate change in terms of range shifts (Kroiss and HillesRisLambers 2015).

A common approach to detecting range shifts is comparing current distributions of mature trees and seedlings. Juveniles (and seedlings specifically) with limited root systems and smaller reserves of carbon are more vulnerable to mortality from drought and temperature extremes (Jackson et al. 2009). Monleon and Lintz (2015) provided evidence of range shifts for common tree species in California, Oregon, and Washington where the range of seedlings extended to temperatures 0.22 °F (0.12 °C) colder than that of adult trees, and seedlings were found at higher mean elevations and latitudes than mature trees for most species during the period from 2001 to 2010. Results also suggested that overall distributions of individual species remained relatively stable, but most species were more abundant toward the colder edge of their range and distributions changed the least at the warm end of their range. Some of the more common tree species with seedlings found at significantly colder temperatures included western redcedar, silver fir, western hemlock, grand fir, and mountain hemlock.

Thus far, individual tree species have shown differential responses to recent warming, and it is likely that tree species will respond differently to projected future changes in climate. Lintz et al. (2016) examined recent changes in basal area and density of 22 tree species on unburned Forest Service lands in Oregon and Washington from the mid 1990s to mid 2000s. Several species had stable populations in terms of density and basal area, including noble fir (*Abies procera*), western redcedar, western hemlock, ponderosa pine, and Douglas-fir. These findings are consistent with HilleRisLambers et al. (2015), who suggested that compositional change in the near term will be slow in higher elevation forests of the silver fir vegetation zone. The greatest levels of mortality in Lintz et al. (2016) occurred in western white pine, whitebark pine, Pacific madrone, subalpine fir, lodgepole pine, grand fir, Engelmann spruce, and western yew (*Taxus brevifolia*). Although this study suggested only slight mortality-related declines of Alaska yellow-cedar (*Callitropsis nootkatensis*), this species has experienced recent mortality across large areas in southeast Alaska associated with a warming climate (Krapek and Buma 2015).

Recent work from the Klamath Mountains and eastern Cascades in northern California suggests that multiple species, including red fir, Jeffrey pine, lodgepole pine, and white fir, experienced recent increases in mortality (Mortenson et al. 2015). Results from this study indicated that mortality rates for all species were generally higher in smaller size classes. Despite increases in the number of recently dead red fir associated with dwarf mistletoe and drought, the population structure of this species was stable.

Vegetation Models and Potential Future Vulnerability

Several climate change vulnerabilities have been identified either explicitly in the literature, or may be inferred based on knowledge of long-term vegetation change in the region, distribution and dynamics of current vegetation, and projected changes across the region. Increases in temperature, as well as altered precipitation and disturbance regimes, are expected to alter vegetation across the region (see "Summary of Vulnerabilities to Climate Change" on next page). Several types of simulation models are commonly used to predict vegetation responses to potential future climate scenarios, each with their own unique set of assumptions, strengths, and weaknesses (see Peterson et al. 2014a for a more indepth review). Models simplify the complexity of ecological processes by making assumptions that are ideally based on empirical measurements. However, because empirical data are often only available for a few species at a few geographic locations, models are most often based on applications of theory on how species interact and respond to environmental gradients. As a result, the best use of models may be for understanding variability in the magnitude of effects as opposed to predicting specific outcomes (Jackson et al. 2009, Littell et al. 2011). Some of the most common models used to project the effects of climate change can be generally characterized as species distribution models (SDM), dynamic global vegetation models (DGVM), and landscape models. These models have their own unique assumptions and relative strengths and weaknesses, which should be carefully considered when interpreting results.

Summary of Vulnerabilities to Climate Change

General vulnerabilities to climate change include increased wildfire and insect activity driven by drought and extreme weather events, ongoing and new invasions of nonnative species, and loss of some high-elevation species. Fragmented populations at range margins (e.g., Alaska yellow-cedar), as well as narrowly distributed species and species with poor dispersal, are vulnerable to declines from losses of climate-suitable habitat, especially in areas that lack topographic conditions that foster the potential for long-term persistence in relatively climate-stable refugia.

The greatest vulnerability to climate change exists in the drier and colder portions of the region in the eastern Cascades, southern portion of the western Cascades of Oregon, coastal and inland areas of the Klamath Mountains, and the California Coast Range. In dry vegetation zones of these regions, increases in area burned during drought conditions may result in larger patches of high-severity fire and drive landscape-scale change. In general, there is good model agreement that subalpine forests are likely to be reduced everywhere except in the northern portion of the eastern Cascades. Several tree species in both wet and dry vegetation zones are vulnerable to nonnative pathogens whose effects may be exacerbated by climate change. These include whitebark pine, subalpine fir, sugar pine, western white pine, Port Orford cedar, tanoak, and multiple species of oak. Old-growth forests may also be vulnerable to periods of elevated mortality rates associated with insects and pathogens during drought. Along the coast, decreases in summer fog may substantially reduce suitable climate for redwood and other coastal species that depend on it to mitigate summer drought.

Much of the coastal and inland area toward the central and northern part of the region show either less potential increase or decreases in water-balance deficit during the summer months. However, high-elevation areas may see reduced snowpacks with more precipitation falling as rain. Warmer, wetter conditions may also promote native and nonnative pathogen activity, especially Swiss needle cast on Douglas-fir near the coast. Some of these areas may be vulnerable to a continued northward shift of high-wind events, particularly near the coast in Washington. Although they have been rare in the past century, these areas have historically experienced large fires driven by synoptic warm, dry wind events from the east during drought conditions projected for the future.

Species distribution models are statistical models based on empirical observations of the relationship between a species occurrence and the observed range of environmental or bioclimatic conditions. SDMs are commonly used due to their simplicity, but generally do not represent ecological processes (e.g., biotic interactions, dispersal, adaptation) that constrain species distributions (Ibáñez et al. 2006), and are problematic when extrapolating to future climates that have no modern analogs (Bell and Schlaepfer 2016). Despite these limitations, SDMs provide a basic understanding of how suitable bioclimatic conditions constrain the current distribution of a species, as well as how this distribution might change under any number of different climate change scenarios.

DGVMs are a type of process model that predict ecosystem processes along with the distribution of specific biomes or plant function groups. These models (e.g., MC1) incorporate biogeography and ecophysiology of vegetation types (e.g., coniferous forests, grasslands, woodlands) as well as climate and disturbance to project broad-scale vegetation changes. Biogeochemistry models are also process models, but focus more specifically on carbon, water, and nutrient cycles and are often used to investigate the effects of climate change on productivity and carbon storage. Both types of models are capable of incorporating some of the important ecological processes affecting vegetation response to climate change (e.g., disturbance, CO_2, site water balance), but have generally been applied at broad regional scales with coarse spatial resolution.

Landscape models (e.g., LANDIS-II) (Scheller et al. 2007) generally focus explicitly on simulating processes (e.g., dispersal, growth, mortality) and can represent interactions among vegetation, disturbance, climate change, and management scenarios at a variety of different spatial and temporal scales. Landscapes are represented as gridded cells in which individual cohorts of trees compete for resources, grow, and die. Although some ecological processes are represented in landscape models, many processes that will be sensitive to climate change (e.g., CO_2 fertilization, phenology, biotic disturbances) are not incorporated in these or other models for projecting vegetation change.

Model projections—
DGVMs generally project persistence of cool, maritime forests in the western hemlock and Sitka spruce vegetation zones of the Coast Range in western Oregon and Washington (Creutzburg et al. 2017, Rogers et al. 2011, Shafer et al. 2015, Turner et al. 2015). SDMs project persistence of western redcedar, Sitka spruce, and western hemlock across 55 to 82 percent of their current distributions by 2080 (DellaSala et al. 2015). However, most species in lower elevation, moist vegetation zones are predicted to have less suitable climatic conditions than currently by the mid-21st century (Saxon et al. 2005). One DGVM-based study projected losses of conifer forest across much of the Coast Range in Oregon with increases in cool mixed forests under the RCP 4.5 scenario, and increases in warm mixed forests under the RCP 8.5 scenario (Sheehan et al. 2015). Although western redcedar is thought to be moderately vulnerable to climate change, bigleaf maple is considered to be one of the least vulnerable species in the region (Case et al. 2016). Consistent with a potential decrease in summer fog (Johnstone and Dawson 2010), DellaSala et al. (2015) projected a decrease in suitable climate for redwood of almost 25 percent by 2080.

SDMs project some of the greatest changes for the southern and southwestern part of the NWFP area, with less change in the north and in the western Cascades (Crookston et al. 2010; DellaSala et al. 2015; Hargrove and Hoffman 2004; McKenney et al. 2007, 2011; Rehfeldt et al. 2006). Using a DGVM, Turner et al. (2015) projected the dominant vegetation type in a portion of the central western Cascades of Oregon to remain forest by 2100, but that the forest would transition from evergreen needleleaf forest to a mixture of broadleaf and needleleaf growth forms. An SDM-based study by Latta et al. (2010) suggests annual growth increases of 2 to 7 percent in moist vegetation zones west of the Cascade Mountains depending on scenario. However, projections from mechanistic models differ, with some projecting moderate to extreme decreases owing to increases in fire activity (Rogers et al. 2011), and others projecting slight to small decreases in growth (Coops and Waring 2011b). Shafer et al. (2015) suggested that growth will decrease in the southwestern part of the region based on projections from a DGVM.

All types of models project that high-elevation forests will experience the greatest change within the region, with moderate to total reductions in suitable climate by the end of the 21st century (Crookston et al. 2010; Halofsky et al. 2013; Hargrove and Hoffman 2004; Mathys et al. 2016; McKenney et al. 2007, 2011; Rehfeldt et al. 2006; Shafer et al. 2015). Case et al. (2016) suggested that western white pine and whitebark pine have relatively high vulnerability to climate change, while noble fir and silver are moderately vulnerable. Mechanistic models project that suitable climate for subalpine fir will be available only in the northern Cascade Range (Coops and Waring 2011b, Rogers et al. 2011), although climate suitability may increase for mountain hemlock in Oregon (Coops and Waring 2011a). Two additional studies also using mechanistic models also predicted large decreases in the distribution of lodgepole pine by the 2100s (Coops and Waring 2011a, Mathys et al. 2016). SDMs project reduction of 15 to 39 percent by 2080 for several species occurring in high-elevation wet vegetation zones, including silver fir, grand fir, Alaska yellow-cedar, and mountain hemlock (DellaSala et al. 2015). In general, there is more model agreement for subalpine forests than for other vegetation zones, and most suggest that suitable climate is likely to be reduced everywhere except in the northern portion of the eastern Cascades.

Model projections for vegetation change in dry coniferous forests in the southern and eastern parts of the region show little agreement. Species distribution models suggest decreases in suitable climate for ponderosa pine, while some DGVMs project increases or only slight changes in temperate coniferous forests (Coops et al. 2005, Halofsky et al. 2013, Rogers et al. 2011, Sheehan et al. 2015) and others projected decreases (Coops and Waring 2011a). Halofsky et al. (2014) projected that while the area of dry mixed-conifer forest is expected to increase from 21 to 26 percent by 2100, the area of moist mixed-conifer forest is expected to decrease 36 to 60 percent in the grand fir/white fir vegetation zone of the central eastern Cascades. Shafer et al. (2015) projected expansion of woodland vegetation during the 21st century. Case et al. (2016) suggested that grand fir will only be moderately sensitive to climate change. Given the lack of agreement among model projections for vegetation change in dry coniferous forests, these results should be used cautiously in planning and management (Peterson et al. 2014a).

In northern California, the projected changes in most scenarios include losses of evergreen conifer forests and increases in mixed evergreen forest primarily because of increased fire activity (Lenihan et al. 2008). A mechanistic model projects that Douglas-fir will be stressed across almost all of northern California (Mathys et al. 2016). Increases are projected in the hardwood component, shrublands, and grasslands, particularly throughout the eastern and drier areas, while maritime evergreen needleleaf forests are expected to contract (DellaSala et al. 2015). Barr et al. (2010) projected that the upper Klamath River basin will support primarily grassland in place of sagebrush and juniper by 2100. In the lower Klamath River basin (California), conditions suitable for hardwood forests (oaks, tanoak, madrone, etc.) are projected to expand, while those suitable for conifer-dominated forests are projected to contract. Results from Kueppers et al. (2005) primarily suggest range expansion and persistence of currently existing populations of valley oak (*Quercus lobata*). Expansion and persistence of blue oak (*Q. douglasii*) is projected in the northern part of its range, but projections primarily suggest range contraction toward the southern portion of northern California.

Other Vulnerabilities

Invasions of nonnative plant species have the potential to alter vegetation dynamics, soil properties (Caldwell 2006, Slesak et al. 2016), and disturbance regimes (Brooks et al. 2004) (see also chapter 3). Most nonnative plant species were initially introduced for horticultural uses and erosion control, or as contaminated crop seed (Reichard and White 2001). Gray (2008) used a systematic inventory of forest health monitoring plots and found that more than 50 percent of plots in almost all physiographic provinces in the NWFP area had nonnative species present. Most common nonnative plants are associated with management (e.g., clearcuts, thinning), though there is potential for the spread of some nonnative, shade-tolerant shrubs in undisturbed forests (Gray 2005). There is also evidence from the region that roads facilitate the spread of nonnative plants (Parendes and Jones 2000, Rubenstein and Dechaine 2015). Little information is available on temporal trends in the abundance of nonnative plants, but increasing temperatures may favor exotic species, especially grasses in California (Sandel and Dangremond 2012). Warm, dry sites with increased topographic exposure may be particularly vulnerable to exotic species, especially annual grasses, following high-severity wildfire (Dodson and Root 2015). Gray et al. (2011) provided a field guide and prioritized list of nonnative plants along with range maps that cover the entire Plan area. More information on management of nonnative species is also available in Harrington and Reichard (2007).

Many species that depend on climate-sensitive habitats will also likely be sensitive to climate change (Case et al. 2015). Narrowly distributed species (e.g., rare and threatened, endemics) that specialize in uncommon or sparsely distributed habitats (e.g., serpentine soils, montane meadows) are expected to have difficulty responding to changing climatic conditions. Increases in Alaska yellow-cedar mortality in southeast Alaska associated with warmer climatic conditions and projections of future decreases in habitat suitability (DellaSala et al. 2015) suggest that this species may be particularly vulnerable to loss. Damschen et al. (2010) found decreases in the richness and cover of endemics on serpentine soils in southwest Oregon from the 1950s to early 2000s that were consistent with a warming climate. Harrison et al. (2010) found changes in forest herb communities in the Klamath Mountains of Oregon that were also consistent with expectations of a drier climate during the second half of the 20th century, including lower cover of species with northern affinities and greater compositional similarity to communities on southerly aspects. Loarie et al. (2008) projected decreases in the richness of endemic plant species by 2100 for those that cannot disperse, but potential increases if plants can disperse to suitable areas. If dryer growing season conditions accompany projected warming trends, cool, mesic topographic refugia are likely to become increasingly important for species persistence (Dobrowski 2011, Olson et al. 2012, van Mantgem and Sarr 2015). Montane wetlands may be especially at risk from reductions in water levels, shorter hydroperiods, and increased probability of drying out (Lee et al. 2015).

Adaptation to and Mitigation of Climate Change

Adaptation and mitigation are essential to strategic planning for the effects of climate change (Millar et al. 2007). Adaptation options include management actions at stand and landscape scales to reduce vulnerabilities to climate change. Mitigation includes efforts to increase carbon sequestration in forest ecosystems and provide new energy-efficient products and technologies for society. Halofsky and Peterson (2016) provided a summary of an extensive list of vulnerabilities and corresponding strategies and tactics that were identified and developed through a series of science-management partnerships across the northwestern United States (http://adaptationpartners.org/library.php). Strategies for adaptation and mitigation have been identified for forests in the Pacific Northwest, including drier forests of southwest Oregon (Halofsky and Peterson 2016; Halofsky et al. 2016, 2017). Here, we highlight general management actions that could promote adaptation to climate change. We summarize these options in table 2-3. For a broader discussion of conservation options (including reserves) in a period of climate and other landscape changes and their specific relevance to NWFP goals, see chapter 12.

Table 2-3—Summary of adaptation options for climate change vulnerabilities in the Northwest Forest Plan area

Vulnerability	Strategy	Tactics
Increased drought stress	Increase resilience	Thinning Favor drought-resistant species/genotypes
	Foster genetic and phenotypic diversity	Protect trees adapted to water stress Collect seed for future Maintain connectivity for natural species migration
Increasing area affected by fire, insects, and pathogens	Increase stand resilience	Thinning and prescribed fire Increase stand heterogeneity Favor fire-tolerant species
	Increase landscape resilience	Increase landscape heterogeneity Increase diversity of patch sizes Use topography to guide treatments
Loss of forest cover	Monitoring of change	Use existing data and add more where needed Planting/assisted migration Maintain connectivity for natural species migration
Exotic species	Increase control efforts	Early detection/rapid response/frequent inventory Interagency coordination

Source: Halofsky and Peterson 2016.

Adaptation—
Several adaptation options to reduce climate change vulnerability are available (table 2-3). These range from manipulation of stand and landscape structure to foster resistance and resilience to future disturbance, to protection of intact areas and climate change refugia that provide connectivity,and facilitate species migration to more favorable habitats. In the case of disturbance, managers may choose to take actions prior to and in anticipation of disturbance to reduce vulnerability, or after a disturbance to affect the ongoing process of recovery (Dale et al. 1998).

Manipulation of stand and landscape structure with management tools (i.e., thinning, prescribed fire) is thought to increase resistance and resilience to future vulnerabilities associated with drought and disturbance (e.g., fire, insects) in drier forests that may be subject to moisture stress and fire (Hessburg et al. 2015, Spies et al. 2010). Findings from dry forests in other regions support the use of thinning as an option to increase soil water availability, reduce growing-season moisture stress, and improve vigor in older trees (Bradford and Bell 2017; McDowell et al. 2003, 2006), but the NWFP area is lacking specific studies on this topic. Prescribed fire has also been found to increase resistance to drought in dry forests of the Sierra Nevada of California (van Mantgem et al. 2016). Thinning has effectively been used and reduced fire severity in dry Douglas-fir of Washington's eastern Cascades (Prichard et al. 2010), and other regions in the Western United States (Wimberly et al. 2009). Fuel treatment may be effective at reducing fire behavior and burn severity during moderate burning conditions; however, treatments may not be effective during large, weather-driven fires (Lydersen et al. 2014, Reinhardt et al. 2008).

A general principle for thinning to reduce fire severity at the stand scale includes maintaining older trees of fire-tolerant species, reducing understory density, and increasing height to live crowns (Agee and Skinner 2005). Given that these actions will likely increase surface fuels, thinning followed by prescribed fire may help reduce surface fuels. Landscape-scale treatments that restore structural heterogeneity in places where historical fire regimes have been interrupted are proposed as a way to reduce vulnerability to high-severity fire and extensive pathogen and insect outbreaks in the future (Hessburg et al. 2015). Topography can provide a physical template to consider when designing and implementing landscape-scale treatments (e.g., thinning on dry ridges). Increasing landscape heterogeneity is thought to impede the spread of contagious disturbances (e.g., fire, insects), but empirical evidence supporting this is currently lacking.

There is relatively little research on the use of thinning in moist forests as a climate change adaptation strategy. These forests were relatively dense historically. Thinning, specifically variable-density thinning, can help the growth and survival of the residual trees, as well as improve the adaptive capacity and ecological diversity of stands (Neill and Puettmann 2013) (see chapter 3). In drier parts of moist vegetation zones, where fire was more frequent, thinning and prescribed fire could be used to mimic low- and moderate-severity fire and promote landscape diversity, which in turn could promote landscape-scale resilience to climate change (chapter 3). The use of thinning in moist forests is generally focused on plantations and younger forests and would have to be balanced against landscape-level goals for maintaining high canopy cover in older forests, which can buffer climatic changes as described above (Frey et al. 2016).

Assisted migration of genotypes and species that are adapted to future climate scenarios may improve resilience of species that are not be able to migrate, but this option is controversial and poorly understood (Marris 2009). Coastal Douglas-fir populations in particular are considered genetically "maladapted" to future climates in Oregon and Washington (St. Clair and Howe 2007). Bansal et al. (2015) found that populations of Douglas-fir from cooler climates had greater resistance to drought than those from warmer climates, contrary to expectations. Populations from areas with relatively cool winters and dry summers were more tolerant to drought and cold and may be the best adapted to warmer future climate conditions (Bansal et al. 2016). There is little information available from other species from the NWFP area, though a study from Arizona found that ponderosa pine seedlings that originated from low-elevation, drier sites survived the longest during drought (Kolb et al. 2016b).

An alternative to assisted migration involves increasing connectivity by establishing large blocks of forest managed for biodiversity and resilience to climate change. Where forests are more fragmented by land use and past management, corridors can facilitate the flow of organisms through the matrix of unsuitable habitat (Krosby et al. 2010, Nuñez et al. 2013). Linking contemporary climates with future climate analogs is one approach to promote connectivity in the future and facilitate movement of species in the future (Littlefield et al. 2017). Vos et al. (2008) suggested the following to mitigate projected climate changes: (1) linking isolated habitats to nearby climate-proof reserves, (2) increasing colonization capacity of reserve networks that are projected to remain suitable in the future, and (3) optimizing reserve networks in which climate remains relatively stable (e.g., refugia). In the only biodiversity-climate resiliency study of the NWFP area, Carroll et al. (2010) found that reserves based on spotted owl conservation criteria overlapped areas of high localized-species richness, but poorly captured core areas of localized species' distributions. They found that resilience to climate change was improved when refugial areas were incorporated into the reserve design of the NWFP.

Protection of climate change refugia based on physiography, soils, and vegetation are a key part of climate change adaptation strategies (fig. 2-8), but identification of refugia has proven difficult (Keppel et al. 2012, Morelli et al. 2016). Most studies of refugia have been ad hoc or descriptive and primarily conceptual, and multiple lines of evidence using different approaches from across disciplines (e.g., SDMs, downscaled climate models, genetics) may be necessary to further understanding of refugia (Keppel et al. 2012). Refugia will most likely be found in topographically complex landscapes where microclimates vary from differences in aspect, shading and insolation, and cold-air drainages (Dobrowski 2011). McRae et al. (2016) mapped potential landscape resilience based on topoclimate diversity and regional connectivity for the Pacific Northwest and northern California. Many of the areas of highest resilience occurred in mountainous areas of federal

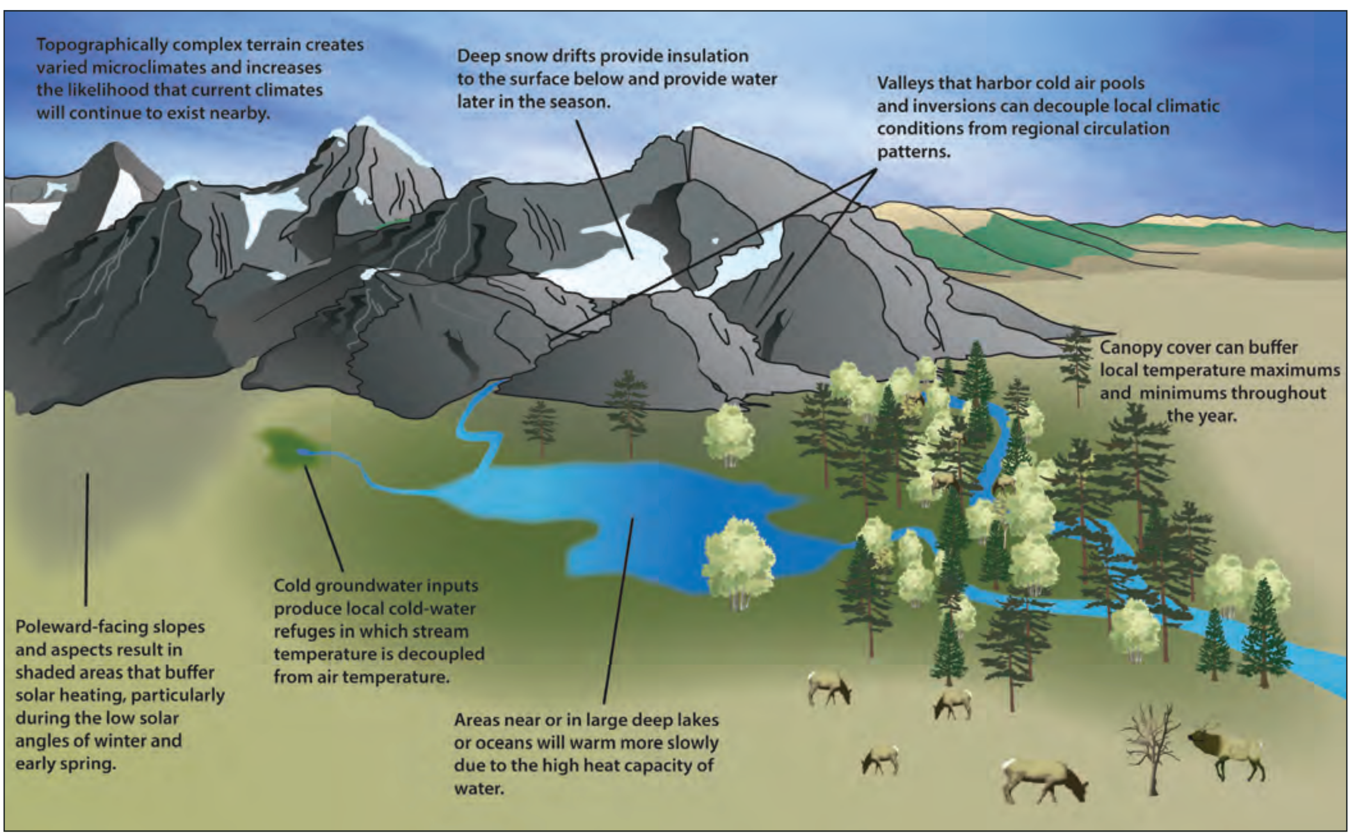

Figure 2-8—Examples of the physiographic and vegetation-based refugia that may experience reduced rates of climate change. Source: Morelli et al. (2016).

lands (e.g., Olympic Peninsula and the Klamath Mountains region). Morelli et al. (2016) presented a synthesis and review of literature pertaining to climate change refugia for climate adaptation. They provided a framework for identifying, mapping, and conserving climate change refugia to meet management objectives. This involves consideration of valued resources and vulnerabilities, identification of climate change refugia, and prioritization of refugial areas.

Increasing connectivity may be insufficient for those species that are unable to migrate as rapidly as the climate changes (Dobrowski et al. 2013). Connectivity considerations would likely need to be species-specific because each species experiences the same landscape in different ways (Betts et al. 2014). Refugia should also be large enough to support populations they are aimed at conserving (Stewart et al. 2010). Planning and monitoring are also essential for adaptation and can help identify microclimatic settings that may provide suitable refugia in the future, coordinate planning across jurisdictions and ownerships, and revise management goals and objectives to be consistent with the uncertainty that accompanies climate change (Spies et al. 2010). For a broader discussion of refugia and connectivity related to the reserve network of the NWFP, see chapters 3 and 12.

Mitigation—

Mitigation includes efforts to increase carbon sequestration in forest ecosystems and provide new energy-efficient products and technologies for society. Of these, we focus on the former, which has been proposed as a means of climate mitigation (Depro et al. 2008, Law and Harmon 2011, Ryan et al. 2010), and then discuss how management practices have the potential to affect carbon sequestration in the NWFP area.

Forests in the NWFP area have great potential to store large amounts of carbon in both live and dead biomass (Smithwick et al. 2002). Total carbon storage levels differ among physiographic provinces (fig. 2-9) as a result of

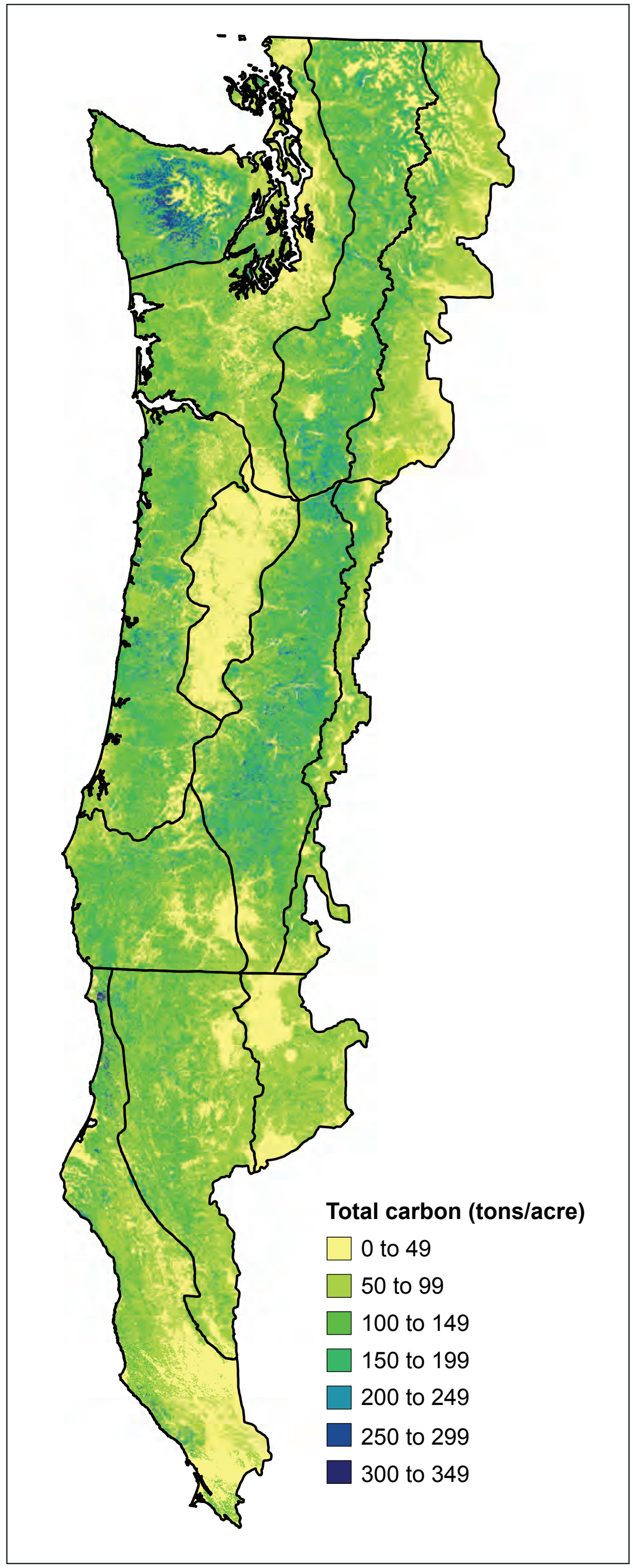

Figure 2-9—Total forest carbon density in the Northwest Forest Plan area (2000–2009). Carbon estimates are from Wilson et al. (2013). Map boundaries correspond with the physiographic provinces in figure 2-1.

productivity and disturbance (Law et al. 2004). Recent findings suggest that forests on Forest Service lands in Oregon and Washington currently store about 63 percent of their potential maximum carbon (Gray et al. 2016). At current rates, harvest and disturbance have little overall impact on carbon sequestration on federal lands in Oregon and Washington as a whole, but this differs at smaller scales among geographic areas (Gray and Whittier 2014). This is particularly true in areas in which dry forests have experienced substantial landscape change in recent fires. In the Oregon Coast Range, projected increases in productivity are associated with projections of increased carbon storage (Creutzburg et al. 2017), but gains could be offset by losses depending on harvest intensity (Creutzburg et al. 2016). Projections suggest future decreases in carbon storage from increases in fire activity in the eastern and western Cascade Range of Washington (Raymond and McKenzie 2012). In forests west of the Cascades where fire is less frequent, decreasing harvesting, increasing rotation age, and maintaining and increasing the extent of late-successional and old-growth forests are strategies to increase carbon storage toward theoretical maximum limits (Creutzburg et al. 2016, 2017; Hudiburg et al. 2009). Maintaining and increasing the area of dense old-growth forests with high biomass also has the potential to mitigate temperature changes in topographically complex mountainous environments (Frey et al. 2016).

Carbon stores in the more fire-prone drier eastern and southwestern part of the region are more unstable and less predictable owing to recent increases and future projections of increased fire activity (Restaino and Peterson 2013). Some studies from other regions in the Western United States (i.e., the Southwest and Sierra Nevada) suggest that thinning and fuel reduction can mitigate carbon loss from fire. Fuel reduction may reduce losses of carbon at stand levels compared with the consequences of high-severity wildfire burning in stands with high fuel loads (Finkral and Evans 2008; Hurteau and North 2009; Hurteau et al. 2008, 2011, 2016; North and Hurteau 2011; North et al. 2009, Stephens et al. 2009). However, because the probability of treated areas burning is generally low (Barnett et al. 2016), and most biomass is not consumed by fire, slight differences in losses resulting from combustion in fire compared with losses from fuel reduction are unlikely to make fuel reduction a viable mitigation strategy (Ager et al. 2010, Campbell et al. 2012, Kline et al. 2016, Mitchell et al. 2009, Restaino and Peterson 2013, Spies et al. 2017). As the amount of fire on the landscape increases, the difference in carbon sequestration between untreated and treated landscapes declines and the likelihood that thinning will pay off in respect to the overall carbon balance increases (Loudermilk et al. 2014).

Research Needs, Uncertainties, Information Gaps, and Limitations

Despite the accumulating scientific information that supports increased warming, considerable uncertainty surrounding the effects of climate change on precipitation, vegetation response, and disturbance remains a significant challenge to forest management (Halofsky and Peterson 2016, Millar et al. 2007). Many of these research needs are mentioned throughout this chapter, but we identify several specific information gaps here.

1. Future role of climate extremes and weather events as disturbances (e.g., heat waves, floods, windstorms).
2. Clarification of the effects of future changes in CO_2, temperature, and water deficit on growth and mortality, and how these effects differ geographically across the region within and among species and seral stages.
3. Effects of recent tree mortality on composition and structural development across seral stages in all vegetation zones.
4. Role of drought on future patterns of disturbance occurrence and severity (e.g., fire, insects, pathogens) in all vegetation zones.
5. Role of interactions among multiple disturbances (e.g., compound and linked disturbances, including insects and fire).
6. Effects of climate change on demographic processes related to migration (e.g., fecundity, dispersal) and how these differ among species in different vegetation zones.

7. Limited understanding of the role of biotic interactions (e.g., competition with and among species) on vegetation response to climate change.
8. Reducing uncertainty surrounding projections in the amount and geographic distribution of species in dry vegetation zones (e.g., ponderosa pine).
9. Response of high-elevation forests to increased summer drought.
10. Effects of thinning on resilience to drought in all vegetation zones.
11. Effects of increasing landscape heterogeneity from fuel treatments (e.g., thinning and prescribed fire) and recent wildfires on future fire and insect activity.
12. Phenotypic responses of individual species to drought and warmer winter temperatures.
13. The potential role and identification of climate and disturbance refugia in all vegetation zones.
14. Multiscale assessment (i.e., stand to landscape) of fuel treatment effects on carbon mitigation under increasing fire activity.
15. Potential of the current NWFP reserve network and management standards and guidelines to provide climate refugia, connectivity to facilitate migration of different species, and stand and landscape conditions that promote resilience to drought, fire, insects, pathogens, and nonnative species.

Conclusions and Management Considerations

Despite the uncertainty surrounding projections of future climate, disturbance and vegetation change, several key vulnerabilities have been identified and are supported by a large body of scientific evidence (see box on page 56). Most models agree and project that the region will experience warmer, drier summers and potentially warmer and wetter winters. Conditions are projected to exceed the 20th-century range of variability around the 2050s, particularly in the Klamath and southern Cascade Mountains. Potential impacts in lower elevation, moist vegetation zones (i.e., western hemlock) include decreased growth and productivity, especially where species are already water limited during the growing season. The greatest vulnerability to climate change is in higher elevation forests, specifically in the subalpine vegetation zone. These forests are likely to experience large decreases in area and may potentially be limited to refugia in the Northern Cascade Range (Mote et al. 2014). Although a great deal of uncertainty surrounds future vegetation change in dry forests, most models consistently agree on an increased role of fire in the 21st century, which is likely to include more area burned and larger patches of high-severity fire. However, most models do not project fire severity or include fire/climate/fuel feedbacks that could be used to project severity.

Projections for climate and vegetation change represent a range of outcomes that can be used to estimate the potential magnitude of effects across the region, but they do not predict specific outcomes. Recent scientific findings suggest several important management considerations for mitigation and adaptation in the face of ongoing climate change across the NWFP area. It is important to consider the potential variability in projections among physiographic provinces and even among landscapes and topographic settings **within** a physiographic province when planning management activities.

1. Considering a variety of approaches may be helpful when managing in the face of uncertainty. “Bet hedging” strategies and multiple courses of action may help to minimize risk and enable further learning. One strategy for dealing with this uncertainty in a planning context is to use scenarios and risk analysis (Acosta and Corral 2017, Bizikova and Krcmar 2015, Pasalodos-Tato et al. 2013) (see also chapter 12).

   Maintaining dense late-successional forests may help mitigate effects of climate change and have the potential to buffer warming at finer scales in moist vegetation zones where fires are infrequent. In addition to storing large amounts of carbon, late-successional forests may also provide refugia for species that depend on cooler, mesic habitats. In dry forest landscapes, maintaining large areas of dense, multilayered older forests would be inconsistent with a strategy for increasing resilience to drought and fire (chapter 3).

2. Landscape-scale treatments to reduce fuels with thinning, prescribed fire, and managed wildfire may promote heterogeneity in dry forests where historical fire regimes were interrupted during the 20th century. These activities can also reduce vulnerability to high-severity fire during moderate weather conditions, as well as to extensive pathogen and insect outbreaks. Topography can provide a physical template to consider when designing and implementing landscape-scale treatments (e.g., thinning on dry ridges and around sheltered refugia).

   Maintaining and increasing connectivity may facilitate migration of species experiencing unsuitable climatic conditions. However, connectivity needs are likely to differ among species, and generic connectivity measures may not be adequate for focal species. In situations in which species' climatic envelopes are changing more rapidly than species are migrating, assisted migration can promote genetic and phenotypic diversity and may help maintain forest cover, although the net benefits of this practice are uncertain and controversial in the scientific literature.

3. Monitoring of populations, species distributions, forest conditions, and disturbance are essential to inform management decisions and help prioritize objectives for adaptive management in response to changes. Most species are expected to respond individually to projected changes in climate and disturbance regimes, and future forest communities may not have contemporary analogs. Understanding the responses of an individual species and how they differ across its range can assist in developing strategies to promote species persistence and prioritize management efforts.

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Appendix: Crosswalk of Simpson (2013) Potential Vegetation Zones With Existing Vegetation From the Classification and Assessment With Landsat of Visible Ecological Groupings (CALVEG) System

Values indicate the percentage of the potential vegetation zone that falls into the CALVEG class. Existing vegetation comes from the Regional Dominance Type 1 field in the CALVEG database and indicates the primary, dominant vegetation alliance. The listed existing vegetation alliances comprise 95 percent of each potential vegetation zone in northern California. Current vegetation types with less than 2 percent cover in a potential vegetation zone are not shown. For information on CALVEG, see http://www.fs.usda.gov/detail/r5/landmanagement/resourcemanagement/?cid=stelprdb5347192.

Potential vegetation zone	CALVEG regional dominance 1
Western hemlock	Douglas-fir (40.3%), white fir (18.5%), Jeffrey pine (15.5%), tanoak (madrone) (9%), black oak (3.9%), ultramafic mixed conifer (3.7%), California bay (2.9%), red fir (2.4%)
Tanoak	Douglas-fir (40.3%), tanoak (madrone) (11.3%), Oregon white oak (6.2%), California bay (5%)
Shasta red fir	Red fir (33.2%), white fir (10.1%), Jeffrey pine (10.1%), barren (10%), mixed conifer–fir (8.1%), alpine grasses and forbs (5.1%), pinemat manzanita (5%), subalpine conifers (4.9%), upper montane mixed chaparral (2.9%), perennial grasses and forbs (2.1%)
Port Orford cedar	Douglas-fir (46.6%), ultramafic mixed conifer (24.8%), Douglas-fir–white fir (7.9%), tanoak (madrone) (2.9%), Douglas-fir–ponderosa pine (2.9%), mixed conifer–pine (2.2%), Oregon white oak (2%)
Other pine	Lower montane mixed chaparral (16.5%), gray pine (10.1%), chamise (8%), Oregon white oak (7.1%), interior mixed hardwood (6.6%), canyon live oak (5.6%), blue oak (5.6%), annual grasses and forbs (4.8%), Douglas-fir–ponderosa pine (4.4%), scrub oak (3.6%), Douglas-fir (3.5%), mixed conifer–pine (3.3%), Sargent cypress (3.2%), black oak (2.5%), knobcone pine (2.2%), ponderosa pine (2%)
Grand fir/white fir	Mixed pine conifer (27.1%), white fir (19%), Douglas-fir–white fir (14%), Douglas-fir (10.6%), Douglas-fir–ponderosa pine (6.3%), red fir (5.9%), mixed conifer–fir (2.5%), upper montane mixed chaparral (2%)
Douglas-fir	Douglas-fir (29.3%), Douglas-fir–ponderosa pine (13.3%), Oregon white oak (12.7%), mixed conifer–pine (7.8%), lower montane mixed chaparral (5.3%), canyon live oak (4.6%), black oak (4%), interior mixed hardwood (3.8%), ponderosa pine (3.2%), annual grasses and forbs (2%)
Juniper	Annual grasses and forbs (45.3%), mixed conifer–pine (17.2%), barren (8.3%), Douglas-fir–ponderosa pine (7%), upper montane mixed chaparral (4.3%), perennial grasses and forbs (2.9%), manzanita chaparral (2.8%), ponderosa pine–white fir (2.3%), Jeffrey pine (2%)

Map available from http://www.ecoshare.info/category/gis-data-vegzones.
Source: Simpson 2013.

Old-growth forest, Oswald West State Park, Oregon.
Photo by David Patte, U.S. Fish and Wildlife Service.

Chapter 3: Old Growth, Disturbance, Forest Succession, and Management in the Area of the Northwest Forest Plan

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Introduction

In this chapter, we examine the scientific basis of the assumptions, management strategies, and goals of the Northwest Forest Plan (NWFP, or Plan) relative to the ecology of old-growth forests, forest successional dynamics, and disturbance processes. Our emphasis is on "coarse-filter" approaches to conservation (i.e., those that are concerned with entire ecosystems, their species and habitats, and the processes that support them) (Hunter 1990, Noss 1990). The recently published 2012 planning rule has increased emphasis on land management rooted in ecological integrity and ecosystem processes, using coarse-filter approaches to conserve biological diversity (Schultz et al. 2013). Fine-filter approaches (e.g., species centric), which are also included in the 2012 planning rule, are discussed in other chapters. We synthesize new findings, characterize scientific disagreements, identify emerging issues (e.g., early-successional habitat and fire suppression effects) and discuss uncertainties and research needs. We also discuss the relevance of our findings for management. Climate change effects on vegetation and disturbance and possible responses (adaptation and mitigation) are addressed mainly in chapter 2 of this report. Although, our effort is primarily based on published literature, we bring in other sources where peer-reviewed literature is lacking, and we conduct some limited analyses using existing data. We are guided by the NWFP monitoring questions, those from federal managers and our reading of the past three decades of science.

Old-growth forests can be viewed through many ecological and social lenses (Kimmins 2003, Moore 2007, Spies and Duncan 2009, Spies and Franklin 1996). Socially, old growth has powerful spiritual values symbolizing wild nature left to its own devices (Kimmins 2003, Moore 2007), and many people value old growth for its own sake ("intrinsic" values, sensu Moore 2007). Old growth also has many "instrumental" or useful functions, including habitat for native plants or animals (e.g., the northern spotted owl [*Strix occidentalis caurina*]), carbon sequestration (Harmon et al. 1990), and other ecosystem services. No single viewpoint fully captures the nature of the old-growth issue as it relates to federal forest management. We focus here on ecological perspectives (Kimmins 2003, Oliver 2009, Ruggiero et al. 1991, Spies 2004, Spies and Franklin 1996), many of which are overlapping conceptually and in common parlance. Old growth is many things at the same time; for example, old growth is:

- An ecosystem "distinguished by old trees and related structural attributes. Old-growth encompasses the later stages of stand development that typically differ from earlier stages in a variety of characteristics including tree size, accumulation of large dead woody material, number of canopy layers, species composition and ecosystem function" (USDA FS 1989).
- An ecological state resulting from interactions among successional, disturbance, and ecosystem processes (e.g., nutrient and carbon cycles, microclimate).
- A biological condition defined in terms of life histories and demographics of forest plant species.
- A habitat for particular fauna, flora, and fungi.

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We define old-growth forests based on live and dead structure and tree species composition (see below). Old-growth forests in the NWFP area differ with age, forest type, environment, and disturbance regime (Reilly and Spies 2015, Spies and Franklin 1991). The variability and complexity of site conditions, forest succession, and disturbance processes make defining old-growth difficult or impossible under a single definition. Under the U.S. Department of Agriculture (USDA), Forest Service (USDA FS 1989) definition (above), the only features distinguishing old-growth from other forests, across all forest types, are the dominance or codominance of old, large, live and dead trees (multiple canopy layers are not necessarily a defining characteristic). For example, in fire-frequent historical forest types, old-growth forests have large old live and dead trees, but amounts of deadwood are low, canopies are generally open, and areas with multiple canopy layers are uncommon (Dunbar-Irwin and Safford 2016, Safford and Stevens 2016, Youngblood et al. 2004) (fig. 3-1).

In the NWFP, "older forests" were defined as "late-successional/old-growth" based largely on stand developmental and successional patterns of Douglas-fir/western hemlock (*Pseudotsuga menziesii*/*Tsuga heterophylla*) forests (Franklin et al. 2002) (fig. 3-2). This multilayered closed-canopy old growth (e.g., canopy cover >80 percent) was the focal point of old-growth conservation during the development of the NWFP, but as we shall argue, old growth is far more diverse than that and functions quite differently across the range of the northern spotted owl. "Older forests" in the original NWFP includes mature forests, 80 to 200 years of age—a pre-old-growth stage, known somewhat confusingly as "late-successional"[2] in the Plan), and old-growth forests. Old-growth has been defined in the NWFP and elsewhere as forests containing large and old, live and dead trees, a variety of sizes of other trees, and vertical and horizontal heterogeneity in tree clumps, gaps, and canopy layering (see O'Hara et al. 1996, Spies 2006, and Davis et al. 2015 for more discussion of old-growth or old-forest definitions). According to Spies and Franklin (1988), old-growth is part of a structural and compositional continuum of successional stages that varies by environment. According to O'Hara et al. (1996), speaking of frequently disturbed environments, old forest is a part of the successional continuum that varies by environment and disturbance processes, which have the ability to advance or retard succession.

To operationalize the successional continuum concept of old-forest development, Davis et al. (2015) created an old-growth structure index (OGSI) to characterize the degree of old-growth structure ("old-growthiness" calibrated by potential vegetation type) that occurs in a stand of any age or history, for use in mapping and monitoring in the Plan area. Two definitions for late successional/old growth were then created: OGSI 80 (structural conditions commonly found in forests that are 80 years and older) and OGSI 200 (structural conditions that are representative of forests containing trees that are more than 200 years of age). These classes roughly correspond to the definitions used by FEMAT, the Forest Ecosystem Management Assessment Team (FEMAT 1993), for mature trees (80 to 200 years old) (e.g., "late- successional" in the NWFP) and old growth (>200 years) but have the advantage of being structure based and calibrated to different potential vegetation types. Also, given that this is a continuous index, other age/development thresholds (e.g., 120 years) could be used for mapping and monitoring.

We note that the structure index and definitions used in the monitoring program are based on current forest conditions from forest inventory plots, which means that in fire-frequent dry zone forests, the structure and composition of old growth is a product of 100 years or more of fire exclusion and highly altered forest development processes. Inventory definitions for dry, old forests based on densities of large-diameter fire-tolerant trees have been developed for the eastern Washington Cascade Range (Franklin et al. 2007a). However, definitions and indices of dry, fire-dependent, old-growth forest structure at stand and landscape scales are still needed for the larger NWFP area (see below for further discussion).

[2] Most of the time in this document, we use the term "late successional" to refer to vegetation that is in the later stages of forest succession where age, height, and biomass are near maximum and shade-tolerant species are the primary understory or overstory tree species. This broad class would include old growth according to classic definitions in textbooks (Barnes et al. 1998).

Tom Iraci

Figure 3-1—Open, old-growth ponderosa pine stand maintained by low-severity fire in central Oregon.

Tom Iraci

Figure 3-2—Multilayered, old-growth Douglas-fir and western hemlock stand in the western Oregon Cascades.

Old growth has been the focal point for forest conservation and restoration on federal lands in the Pacific Northwest. However, the broad goals of forest biodiversity conservation would not be scientifically viable if they focused on only one stage of a dynamic system—all developmental phases and ecological processes must be considered (Spies 2004), including postdisturbance stages (fig. 3-3), nonforest vegetation, and younger forests that constitute the dynamic vegetation mosaics that are driven by disturbance and succession. These other stages and types contribute to biodiversity, and hence, are as important to any discussion of forest conservation or management for ecological integrity as is the discussion of old growth. Indeed, these other successional conditions become future old growth, so the successional dynamics of the entire landscape ought to be the broader focus of discussions. Consequently, our discussion includes these other stages of forest succession, in addition to old growth.

Thomas Spies

Figure 3-3—Early-successional vegetation 8 years after a high-severity fire in multilayered old growth in southwestern Oregon.

Guiding Questions

This chapter characterizes the current scientific understanding of old-growth forest conditions and dynamics and other successional stages in the NWFP area, especially as they apply to conservation and restoration of forest ecosystems and landscapes. We give special attention to composition and structure of trees (live and dead) as dominant components of forests but acknowledge that other characteristics are also important, including age (or time since disturbance) and composition, and structure of shrub, herb, and grass communities. Our focus is on the broad landscape, which inherently is a mosaic of vegetation conditions; questions related to conservation and restoration of animal species in terrestrial habitats and riparian and aquatic ecosystems and their habitats are dealt with in other chapters.

We address the following major questions in this chapter, though not directly given their breadth, complexity, and certain degree of overlap. See the conclusions section for bullet statements that are explicitly linked to these questions.

1. What are the structures, dynamics, and ecological histories of mature and old-growth forests in the NWFP area, and how do these features differ from those of other successional stages (e.g., early and mid successional)?
2. How do these characteristics differ by vegetation type, environment, physiographic province, and disturbance regime?
3. What is the scientific understanding about using historical ecology (e.g., historical disturbance regimes and natural range of variation [NRV]) to inform management, including restoration?
4. What are the principal threats to conserving and restoring the diversity of old-growth types and to other important successional stages (e.g., diverse early seral), and to processes leading to old growth?

5. What does the competing science say about needs for management, including restoration, especially in dry forests, where fire was historically frequent?
6. How do the ecological effects of treatments to restore old-growth composition and structure differ by stand condition, forest age, forest type, disturbance regime, physiographic province, and spatial scale?
7. What are the roles of successional diversity and dynamics, including early- and mid-seral vegetation, in forest conservation and restoration in the short and long term?
8. What is the current scientific understanding concerning application of reserves in dynamic landscapes?
9. How do recent trends of forests in the NWFP reserve network relate to both original NWFP goals, those of the 2012 planning rule, and climate change adaptation needs?
10. What is the current understanding of postwildfire management options and their effects?
11. What are the key uncertainties associated with vegetation under the NWFP, and how can they be dealt with?

We address these questions using an organization based on major forest regions, disturbance regimes, and potential and existing forest vegetation types.

Key Findings

Vegetation Patterns and Classification

Drivers of regional variation in vegetation—

Forest ecosystems of the vast NWFP region are ecologically diverse and complex and do not lend themselves to simple generalizations (fig. 3-4). In this synthesis, we account for some of that diversity by classifying ecosystems based on potential vegetation types at the zone or series level (Henderson et al. 1989, Lillybridge et al. 1995, Simpson 2007) in a manner similar to Küchler (1964, 1974). Potential vegetation types and disturbance regimes are somewhat correlated, although disturbance regimes can differ significantly within potential vegetation types (i.e., biological and physical environments) (Hessburg et al. 2007, Kellogg et al. 2007, Wright and Agee 2004,) and differences in potential vegetation types or forest composition do not necessarily mean differences in fire history (Taylor and Skinner 1998).

The major biophysical driving variables (aka "drivers") of structure, composition, and dynamics of old-growth forests (and forests in general) are climate, topography, soils, succession processes, and disturbance processes (Franklin and Dyrness 1973; Gavin et al. 2007; Hessburg et al. 2000a, 2015; O'Hara et al. 1996; Oliver and Larson 1990; Spies and Franklin 1996). In conjunction with landform and soil conditions, the geographic and historical variability of the regional climate set the stage for somewhat predictable biotic communities, pathways of forest development, levels of ecosystem productivity, and spatial patterns of disturbance regimes (Agee 1993, Gholz 1982, Hessburg et al. 2000a, Reilly and Spies 2015, Weisberg and Swanson 2003, Whitlock 1992). Climatic variation over time and space exerts a strong control over fire frequency (Agee 1993, Gavin et al. 2007, Walsh et al. 2015), and forest dynamics is a product of the self-organizing interactions of climate, topography, disturbance, and plant communities (Scholl and Taylor 2010). Forest succession is the process of change in tree, shrub, and herb species composition, and structure (size, density, and age structure) over time. Disturbances can advance, arrest, or retard succession either slowly and imperceptibly, rapidly and abruptly, steadily, or in other complex and poorly understood ways (O'Hara et al. 1996, Spies and Franklin 1996). In combination, forest succession and disturbance processes can produce a wide range of forest conditions within the NWFP area.

Classification of vegetation—

Ecological classifications of environment and succession are used to promote understanding and implementation of management objectives. One way that Oregon and Washington ecologists account for environmental differences in succession and in old-growth characteristics (Davis et al. 2015, Reilly and Spies 2015) is to use potential vegetation type (fig. 3-4).

Potential vegetation type is named for the native, late-successional (or "climax") plant community that would

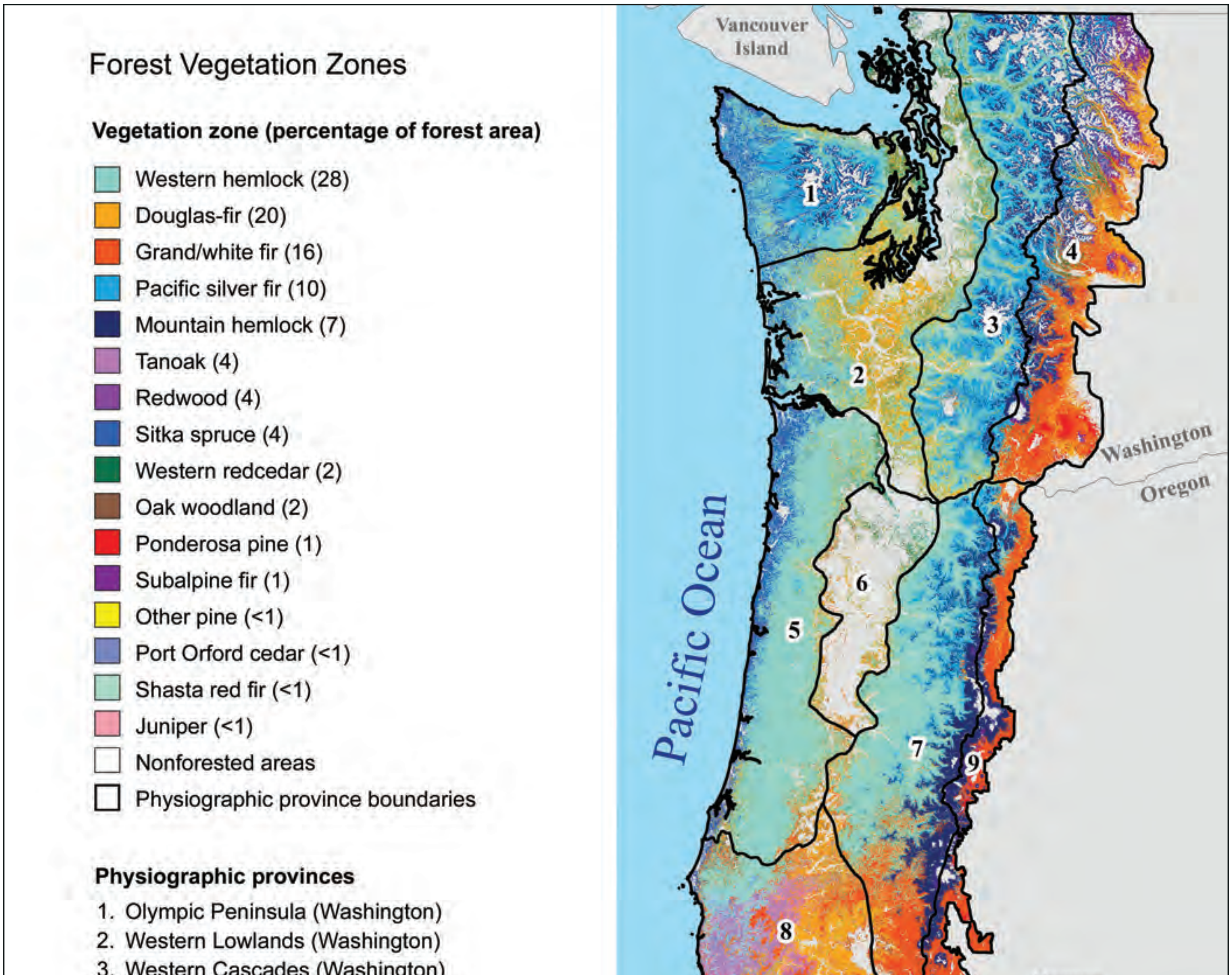


Figure 3-4—Geographic distribution of potential vegetation zones (aka vegetation types) (Simpson 2013) and physiographic provinces across the Northwest Forest Plan area.

occur on a site in the absence of disturbances (i.e., wildfire, bark beetle outbreaks, root disease, weather events), and reflects the biophysical environment (climate, topography, soils, productivity) and composition of overstory and understory species (Pfister and Arno 1980). Stages along the continuum within a potential vegetation type may be binned or categorized into distinct successional stages, which are mileposts for visualizing forest development subjectively given that no clear thresholds in development are known (Franklin et al. 2002, Hunter and White 1997, O'Hara et al. 1996, Oliver and Larson 1990, Reilly and Spies 2015, Spies and Franklin 1988). This classification is often required to enable large-landscape analyses, which cannot efficiently deal with developmental conditions treated as continuous variables.

Not all ecologists and managers use potential vegetation to stratify or map vegetation for management or research purposes. For example, managers in California do not use potential vegetation but use existing or "actual" vegetation cover type instead to classify their forests for management (CALVEG)[3] (http://www.fs.usda.gov/detail/r5/landmanagement/resourcemanagement/?cid=stelprdb5347192.) To help make our discussion more useful to managers in California, we provide a cross-walk table (app. 1) that links the Pacific Northwest Region (Region 6) potential vegetation types (see chapter 2, fig. 3-1) to Pacific Southwest Region (Region 5) existing vegetation classes. We also note, where appropriate, what the CALVEG classes might be for a given potential vegetation type. Most of our discussions in the text use estimated potential vegetation types for California and the rest of the Plan area based on a provisional map prepared by Michael Simpson (ecologist, Deschutes National Forest) (fig. 3-4).

[3] One reason given for doing this is that in California vegetation, historical fire frequencies were quite high and the time since fire exclusion has been too short (e.g., 100 years) to really know what the capacity (potential future vegetation) would have been in the absence of disturbance. For purposes of this document, we use potential vegetation types, because we have a classification and map of these that covers the entire NWFP area (e.g., Simpson 2013), and there is no existing vegetation classification and map for Oregon and Washington. The lack of consistent vegetation data layers between the two regions makes it challenging to apply the findings from one Forest Service region to another.

Moist and dry forests—

At a broad scale, forests of the NWFP area can be classified into moist forests (including the western hemlock, Sitka spruce [*Picea sitchensis*], coastal redwoods, Pacific silver fir [*Abies amabilis*], and mountain hemlock [*Tsuga mertensiana*] potential vegetation zones west of the crest of the Cascade Range in Oregon and Washington), and dry forests (mainly ponderosa pine [*Pinus ponderosa*], Douglas-fir, grand fir [*A. grandis*], and white fir [*A. concolor*] potential vegetation types) east of the Cascade Range and in southwestern Oregon and northern California (Franklin and Johnson 2012). We use this moist forest and dry forest classification to frame much of this chapter.

Disturbance Regimes

Fire regime classification—

For most forest types, fire was and continues to be the major landscape disturbance agent that resets succession or shifts its course to a new pathway (Reilly and Spies 2016). Other disturbance agents are important as well, including wind and biotic agents, but most disturbance regime classifications and maps focus on fire. We characterize the ecology of multiple disturbances for moist and dry forests in sections below. In this section, we focus on approaches to classifying historical fire regimes.

Most of our current understanding of historical fire regimes is based on frequency—empirical studies of severity proportions and spatial patterns at landscape scales are relatively few (Hessburg et al. 2007, Reilly et al. 2017). Fire disturbances occur along a continuum of frequency, severity (e.g., tree mortality), seasonality, spatial heterogeneity, and event sizes. While there is no single classification of disturbance regimes, they are often binned into regime types that are based on fire frequency and severity (Agee 1993, 2003). Average fire frequency interval classes of frequent (<25 years), moderately infrequent (25 to 100 years), infrequent (100 to 300 years), and very infrequent (>300 years) (Agee 1993) are often used, but other frequency classifications exist as well: e.g., ≤35, 35 to 200, and >200 years (Hann and Bunnell 2001, Hann et al. 2004, Rollins 2009, Schmidt et al. 2002).

A widely used classification of fire-severity regimes for vegetation uses three bins of basal area or canopy mortality:

low (<20 percent), mixed or moderate (20 to 70 percent), and high (>70 percent)[4] (Agee 1993, Hessburg et al. 2016, Perry et al. 2011) (fig. 3-5). Other classifications have been used, often with higher thresholds for canopy cover loss or mortality (e.g., 75 to 95 percent) (Miller et al. 2012, Reilly et al. 2017). The classification of Agee (1993) was initially developed for the stand or patch scale, but the metric has also been applied to larger regional areas (Agee 1993, Heinselman 1981, Reilly et al. 2017) or entire fire events, which can create confusion about the meaning of fire severity (Hessburg et al. 2016): Is it a fine-grained mix of severities, or coarse-grained mix of high and low severity, or both? Severity can also be characterized in terms of fire-induced changes to soils (i.e., soil burn severity); however, we focus on vegetative effects in this chapter. Soil burn severity is used in Burned Area Emergency Response analyses and is often confused with burn severity to vegetation (Safford et al. 2007).

[4] Note that while individual patches can exceed 70 percent mortality, fires typically have such high levels of mortality in only a small fraction of their total area. For example, the high-severity area of the 1988 Yellowstone fires was 56 percent (Turner et al. 1994), and the high-severity percentage of the 2002 Biscuit Fire in the Klamath of Oregon and California was 14 percent with an additional 23 percent at moderate severity based on a sample of inventory plots (Azuma et al. 2004).

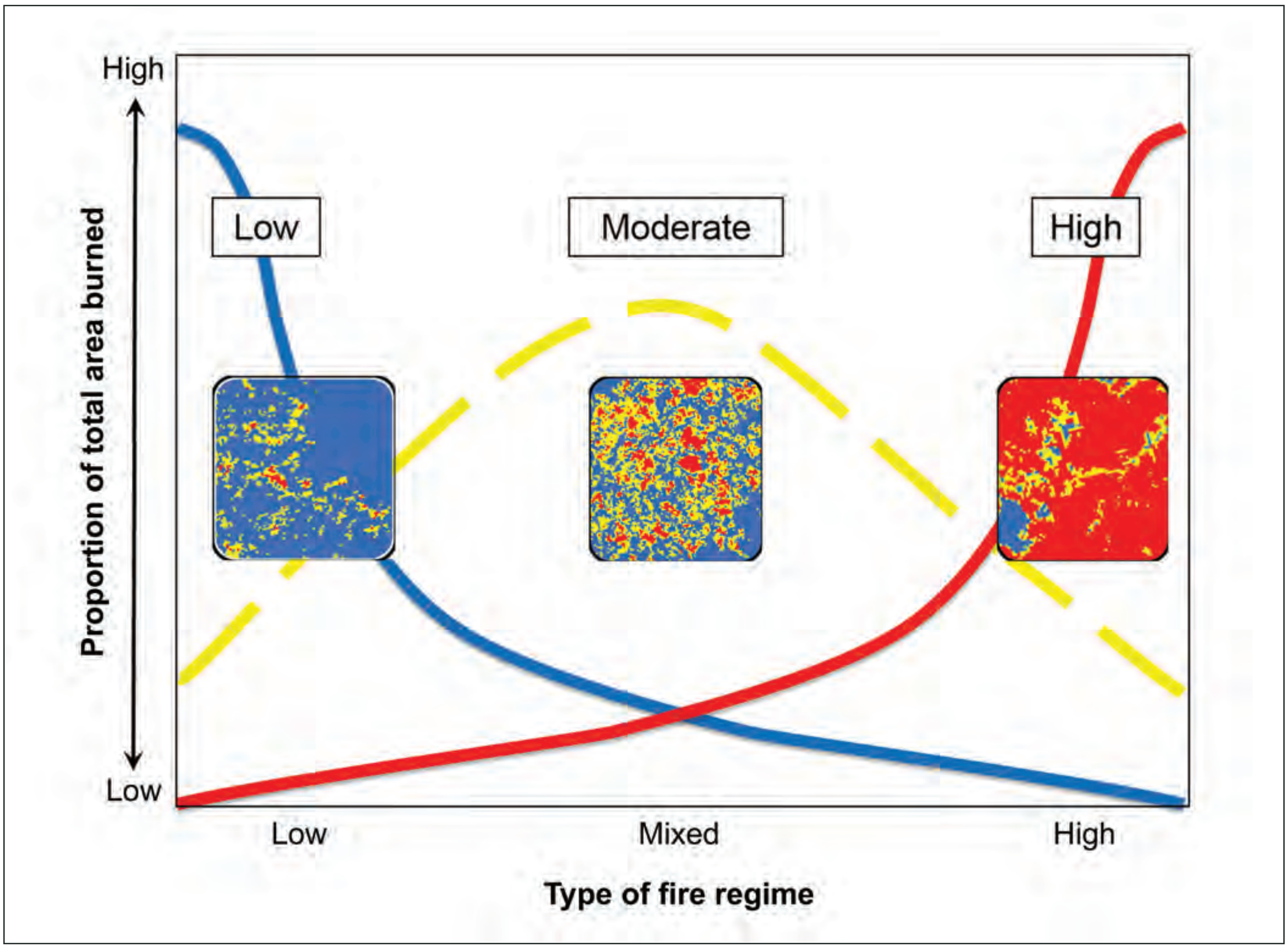


Figure 3-5—Conceptual diagram characterizing the proportions of low-, moderate-, and high-severity fires in three major fire regime classes. Inset panels represent idealized landscape dynamics associated with each regime based on proportions and size class distributions of patches at each of the three levels of severity. From Reilly et al. 2017, who modified it slightly from Agee (1993, 1998).

For management applications and regional planning, broad-scale regime classifications are typically used (Haugo et al. 2015), but fire history studies indicate that fire regimes can be relatively distinctive at topographic and landform scales (10° to 10^3 ac) (e.g., Taylor and Skinner 1998, Tepley et al. 2013). At landscape scales (ca. 10^3 to 10^6 ac), most fires occur as a mix of low, moderate, and high severity, driven by variation in topography, land forms, microclimate, surface and canopy fuels, soils, and vegetation, as we explore in later sections.

Combining fire regimes into broad average frequency and severity types is useful for regional planning (e.g., Rollins 2009, USDA and USDI 1994), but it oversimplifies variability that exists at finer scales, which is important for landscape planning and management. In general, simplifying fire into a few regime classes can obscure ecological diversity associated with fire effects (Hutto et al. 2016). Note that fire-severity proportions for any particular landscape or landform is often more restricted than implied by the broad ranges used to define broad regime classes. For example, for some landscapes in the very high frequency, low-severity regime (see below), the historical range of high-severity fire may be in the low end of the 0 to 20 percent[5] range used to define this class.

A new fire regime classification—
For national and regional planning and management purposes, managers often use the LANDFIRE (Rollins 2009) fire regime classification. Our review of recent science in the NWFP region suggests that the national-scale product oversimplifies the fire history within the NWFP area. Thus, we developed a new classification and map (table 3-1, fig. 3-6) by synthesizing existing data on climate, lightning, and potential vegetation types (see app. 2 for methods) and fire history studies (app. 3).

This classification and map are meant to be a rough guide for understanding and visualizing ecological variation at regional scales and for framing a discussion about forest conservation and restoration science in the NWFP area (figs. 3-4 and 3-5). They reflect current understanding of fire ecology and geographic variability in the region. This typology is different from that used in the record of decision (USDA and USDI 1994) and FEMAT (1993) documents, which divided the NWFP region into moist and dry physiographic provinces but did not characterize variability in regimes within them. The physiographic provinces explained much of the variation in the physical environment, but they contain considerable subregional variations in vegetation types and fire regimes that are important to understanding the ecology of the forests in NWFP area. The potential vegetation types differ in distributions of fire regimes that occur within them (fig. 3-7), and the distribution of potential vegetation types differs between fire regimes, though the differences are relatively small between regimes within the moist or dry forests (fig. 3-8). Almost all fires in these regimes have mixed-severity effects, but they typically differ in the proportion and distribution of the high-severity effects. The very frequent low-severity regime, for instance, contains some area in high-severity fire patches at the scale of acres to tens of acres. The recognition of a drier, more fire-frequent mixed-severity zone on the west side of the Cascade Range in Oregon (fig. 3-6) is based on a number of studies (Agee and Edmunds 1992; Dunn 2015; Impara 1997; Reilly and Spies 2016; Tepley et al. 2013; Weisberg 2004, 2009). This regime, which typically burns with mixed severity and includes medium to large patches of high-severity fire, was first identified by Agee (1993), based in part on the fire history work of Morrison and Swanson (1990) from the western Cascades in Oregon.

Our classification also recognizes that the California portion of the NWFP area cannot be simply divided into a moist (Coastal province) and dry (Klamath and Cascades provinces) province for understanding succession and disturbance regimes. In fact, that area has relatively little of the "moist" forest that is characterized by historically

[5] Odion et al. (2014) called for restricting definitions of historical low- and mixed-severity fires to regimes where crown fires and active or passive torching are generally absent. However, this classification would not be useful, as crown fires can occur in all fire regimes including low-severity regimes (Agee 1993), particularly when the regimes are intermixed, as they often are, where large landscape contain a range of topography, environmental, or vegetation conditions.

Table 3-1—Characteristics of major historical fire regimes used in this report and in figure 3-6

NWFP forest zone	Regime and landfire group	PVTs and cover types	Spatial characteristics
Moist	Infrequent (>200-year return intervals), stand replacing; LANDFIRE group V	PVT: wetter/colder parts of western hemlock, Pacific silver fir, mountain hemlock Cover types: Douglas-fir, western hemlock, Pacific silver fir, noble fir, mountain hemlock	Area dominated by large to very large patches (10^3 to 10^6) of high-severity fire; low- and moderate-severity fire also occurs. Small- to medium-size patches were most frequent.
	Moderately frequent to somewhat infrequent (50- to 200-year return intervals), mixed severity; LANDFIRE regime group III	PVTs: drier/warmer parts of western hemlock, Pacific silver fir and others Cover types: Douglas-fir, western hemlock, Pacific silver fir, noble fir	Mixed severity in space and time, typically including large (10^3 to 10^4 ac) patches of high-severity fire and areas of low- and moderate-severity fire. Small patches of high-severity would be common within lower severity areas.
Dry	Frequent (15- to 50-year return intervals) mixed severity; LANDFIRE regime group I and III	PVTs: Douglas-fir, grand fir, white fir, tanoak Cover types: Douglas-fir, white fir, red/noble fir, western white pine	Mixed-severity fire with medium to large (10^2- to 10^3-ac) patches of high-severity fire.
	Very frequent (5- to 25-year return intervals) low severity; LANDFIRE regime group I	PVTs: ponderosa pine, dry to moist grand fir, white fir Cover types: ponderosa pine, Douglas-fir, mixed pine, oak	Dominated by low-severity fire with fine-grained pattern (<10° to 10^2 ac) of high-severity fire effects; large patches of high-severity fire rare in forests except in earlier seral stages (e.g., shrub fields).

NWFP = Northwest Forest Plan, PVT = potential vegetation type/zone used in the Pacific Northwest Region. Cover type = current vegetation classification used in the Pacific Southwest Region. LANDFIRE regime groups follow Rollins (2009).

infrequent, high-severity fires. Rather, forests in the California Coastal province were dominated by frequent, mixed-severity regimes, while the eastern Klamath and California Cascades were dominated by historical regimes of very frequent, low-severity fire.

Historical maps of high-severity burned forest patches from Washington and Oregon (data not available from California) (Plummer et al. 1902, Thompson and Johnson 1900) provide an independent source of primary data to evaluate the regional regime map. These maps support the hypothesis that the largest patches and percentage of forest burned by high-severity fire occurred in the infrequent high-severity regime; whereas the smallest patches and lowest area of forest burned by high-severity fire occurred in the very frequent/low-severity regime (fig. 3-9).[6] The relatively high percentage of area burned in the infrequent fire regime may reflect elevated ignitions from Euro-American settlement activities, because lightning densities in these areas are low (fig. 3-10) and these forests are not typically fuel limited (Agee 1993). American Indian burning practices would have also been a historical component in some parts of the region, but the importance would have varied considerably among regimes (see chapter 11). For example, several studies (app. 3) have

[6] These early 20th century maps are our best snapshots of this time period but do not necessarily represent the range of variability in fire sizes that would occur in these regimes over time. This is especially true for the infrequent, high-severity regime where sample of historical fires is small and extremely large patches of fire may have occurred in past centuries.

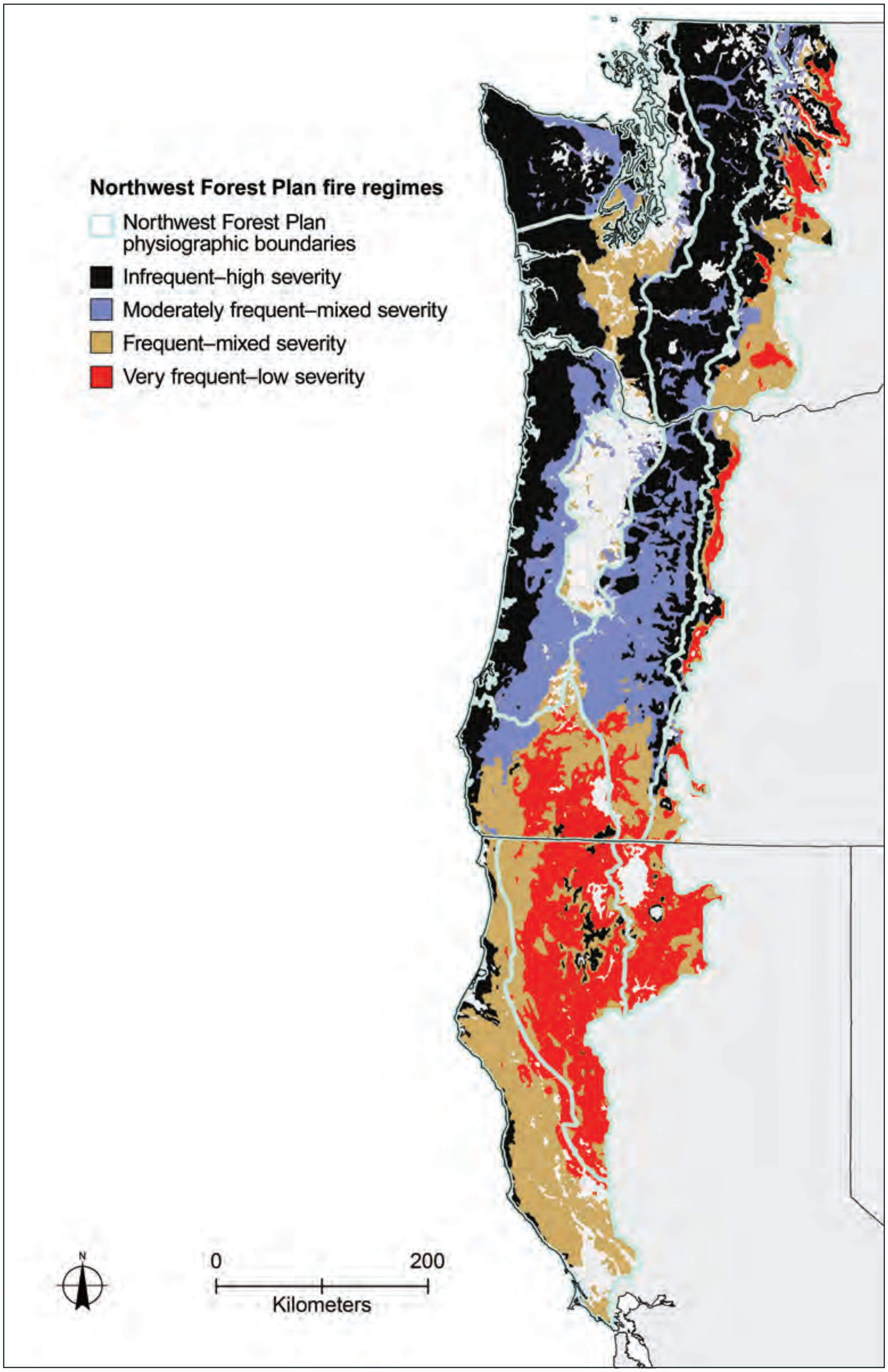


Figure 3-6—Generalized fire regimes for the Northwest Forest Plan (NWFP) area based on climate and lighting density. Fire frequency, particularly in coastal areas of California, may be underestimated because historical ignitions by American Indians are not included in the model. See table 3-1 for more information about the regimes and appendix 2 for methods. Moist forests are typically associated with the infrequent and moderately frequent regimes, while dry forests typically are associated with the frequent and very frequent regimes.

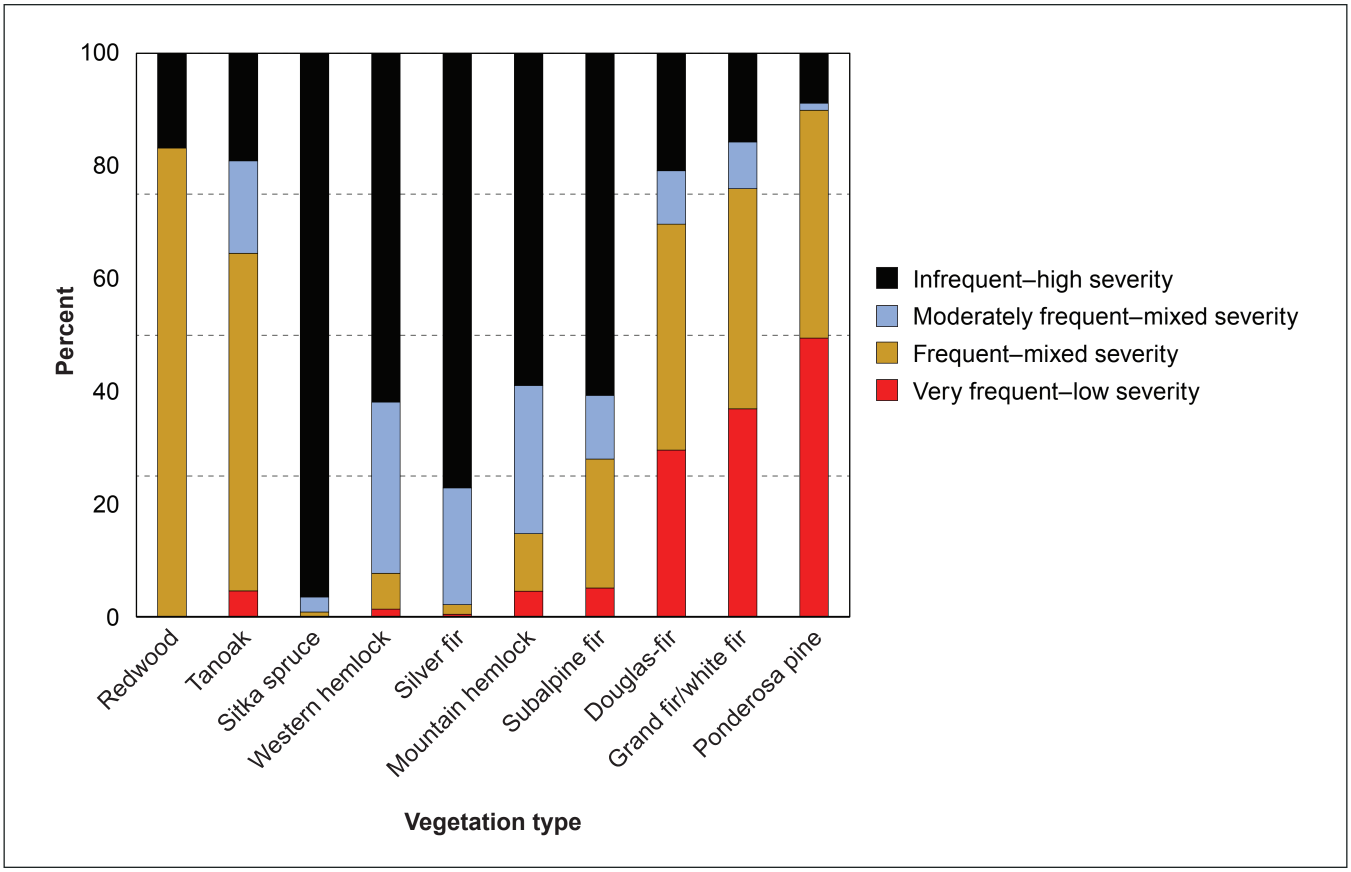


Figure 3-7—Percentage of major potential vegetation types (PVTs) in the four different fire regimes. Small percentages of a fire regime within a PVT may be a result of errors in the PVT maps, fire regime maps, or both.

noted that burning by American Indians likely caused fires to be very frequent (<29 years) (app. 3) in the redwood (*Sequoia sempervirens*) forests of northern California, although the map based upon climate and incidence of lightning classifies those areas as moderate frequency, mixed-severity fire regimes.

The lack of close correspondence of fire regime with major potential vegetation type or climate zone (figs. 3-4 and 3-6) indicates that vegetation type at the zone (series) level (at climax) and fire regime do not necessarily respond in the same way or at the same scale to variation in the environment (Kellogg et al. 2007) (see discussion of the regimes for more information). If disturbance regime variation within subregions and landscapes is not taken into account, efforts to retain or restore biological diversity based on historical fire regimes may not be effective or may have undesirable effects.

Disturbance regimes of moist forests—
Moist forests occur primarily west of the crest of the Cascades in Washington and Oregon, including the Coast Range forests, and on the west slope of the Cascades, they extend into high-elevation wet and cool forests (fig. 3-4). Potential vegetation types are dominated by western hemlock, Pacific silver fir, and mountain hemlock (fig. 3-8). Sources of stand-replacement disturbance in this region included fire, wind, and volcanic eruptions. Insects and diseases, especially root diseases, typically created finer grained disturbances such as canopy gaps (e.g., 0.1 ac [0.04 ha]) to several acres in size) (Dickman and Cook 1989, Spies et al. 1989). In California, moist forests with infrequent fire regimes are confined to relatively small areas along the coast and in some higher elevations.

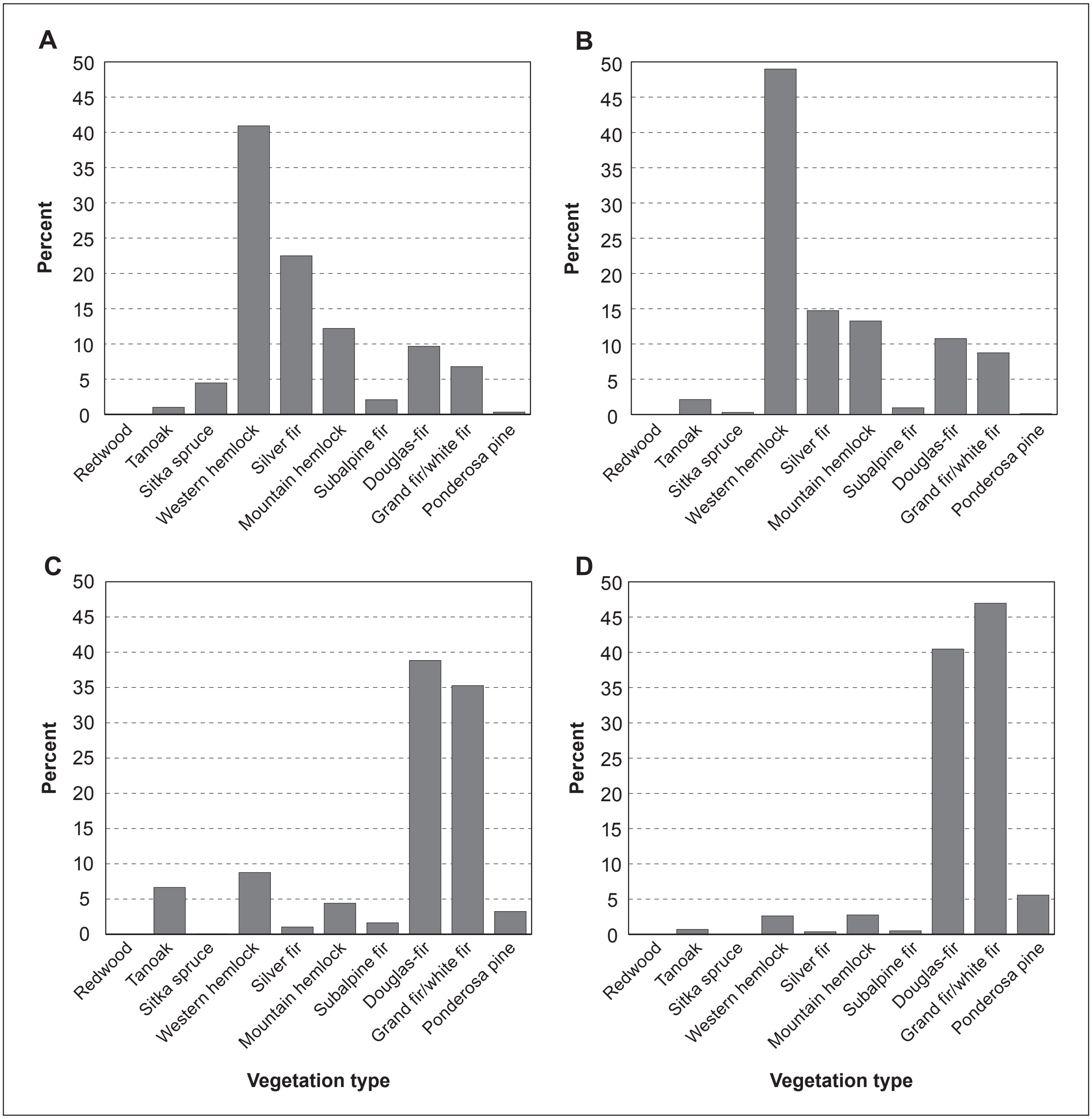
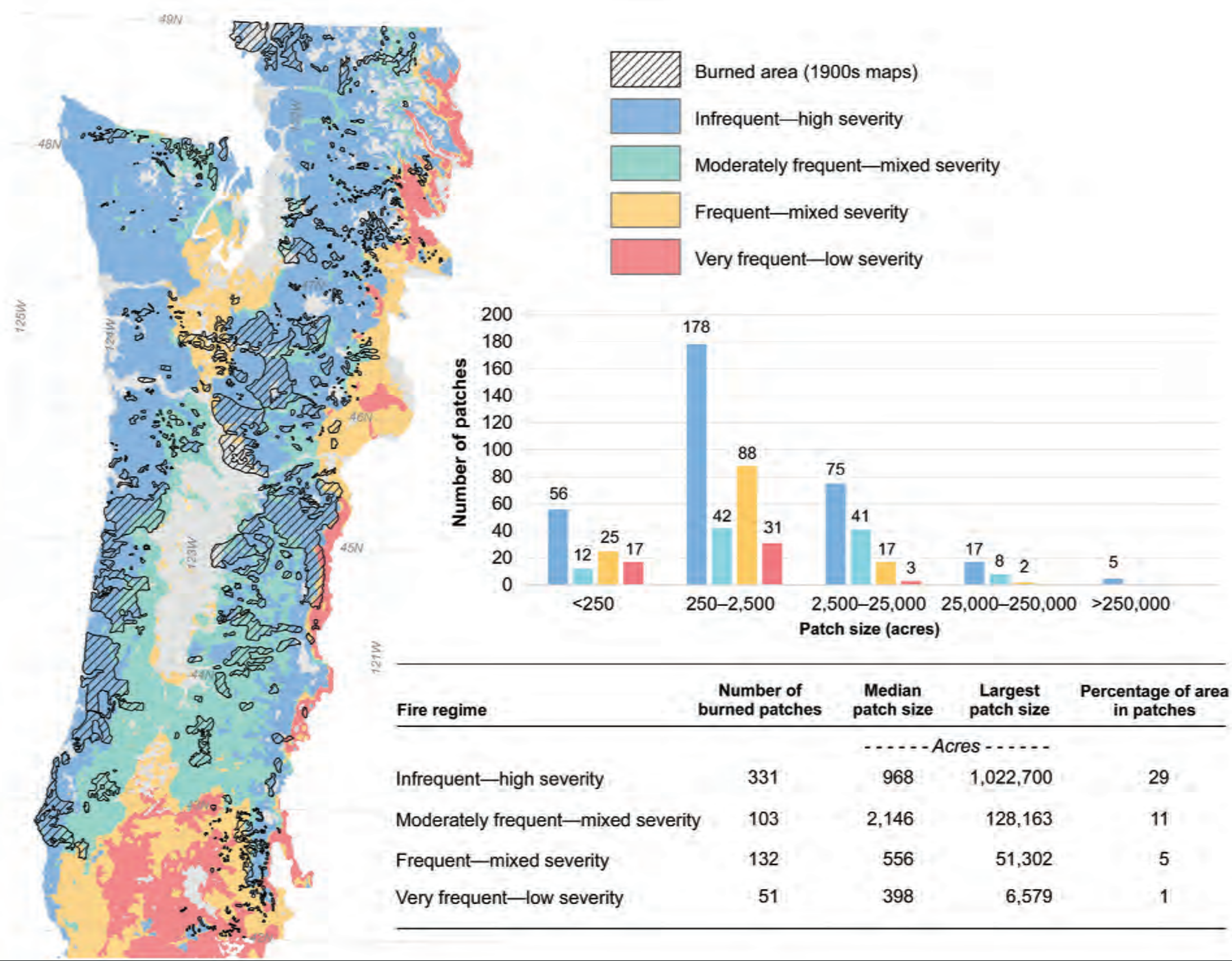


Figure 3-8—Distribution of major potential vegetation types (PVTs) within the (A) infrequent, high-severity regime; (B) moderately frequent, mixed-severity regimes of the moist forests; (C) frequent, mixed-severity regime; and (D) very frequent, low-severity regimes of the dry forests. Only major PVTs are shown. See appendix 1 for crosswalk to California vegetation types. Forests currently dominated by ponderosa pine would occur within the Douglas-fir, grand fir, and white fir PVTs.



Fire regime	Number of burned patches	Median patch size	Largest patch size	Percentage of area in patches
		- - - - - - Acres	- - - - - -	
Infrequent—high severity	331	968	1,022,700	29
Moderately frequent—mixed severity	103	2,146	128,163	11
Frequent—mixed severity	132	556	51,302	5
Very frequent—low severity	51	398	6,579	1

Figure 3-9— Historical (1900–1902) patterns of early-successional postfire patches (where “destruction of timber was nearly or quite complete…areas…with only a partial destruction are not here represented”) (Plummer 1902). Note how patches were fewer and smaller in the high-frequency/low-severity regime compared to the other regimes. Many of these fires would have been ignited by settlement and logging activities but would have burned before fire suppression was effective in most cases. Burned forest patches were digitized from Thompson and Johnson (1900) and Plummer et al. (1902). Data were not available for California.

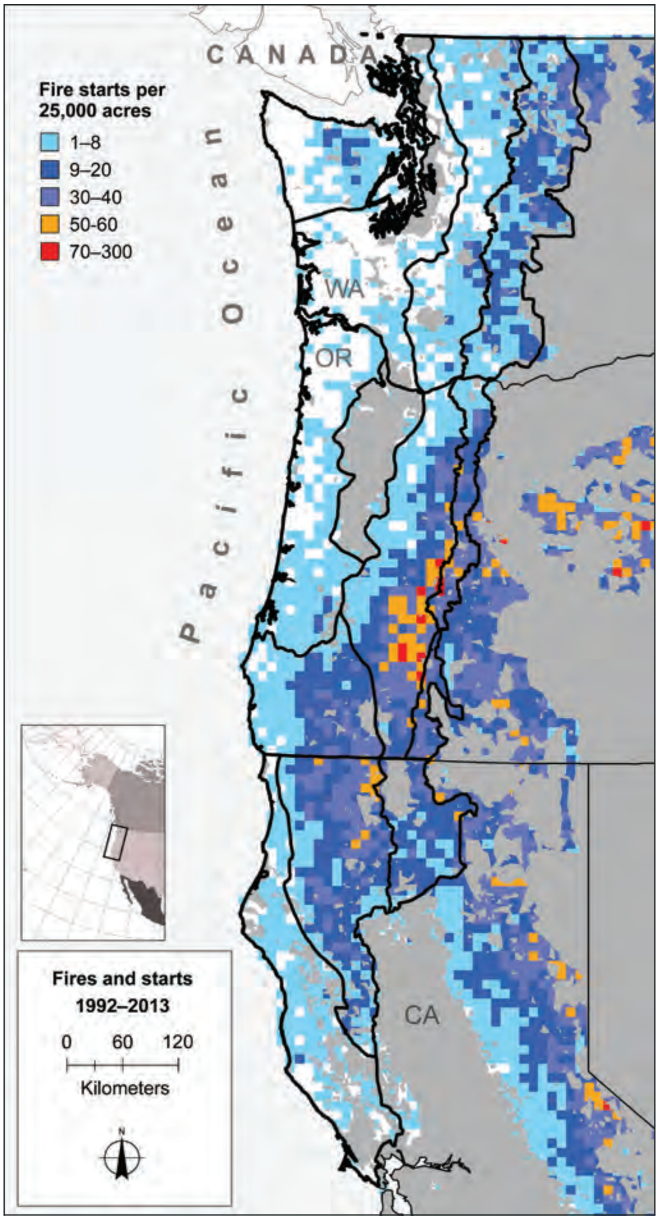


Figure 3-10—Density of lightning-ignited fires per 25,000 ac on forest lands in the Northwest Forest Plan area for the period 1992–2013. Black lines are physiographic provinces as delineated in figure 3-4.

Two major fire regimes can be recognized within moist forests: infrequent (>200-year return interval) and dominated by high severity; and moderately frequent to somewhat infrequent (50- to 200-year return interval) fire with mixed-severity patterns (table 3-1). The infrequent regime is characterized by relatively long fire-return intervals and dominance of high-severity fire in medium to very large patches. Historically, mean fire-free intervals averaged greater than 200 years with some areas not experiencing fire for more than 1,000 years (Agee 1998). Although most of the area in high-severity patches is contained within larger patches in this regime, individual fires could have high-severity (>70 percent mortality) patches ranging from quite small (1 to 25 ac [0.04 to 20 ha) to very large (>10^6 ac [~400 000 ha]) (Agee 1993, 1998). Given the historical infrequency of such fires and the tendency for high-severity fire to erase information about previous fires, there are few empirical studies based on actual fire occurrence (using fire scars), and most of our collective knowledge is derived from studies that used age-class data to reconstruct large-scale fire rotations (Hemstrom and Franklin 1982) and maps of historical fires (fig. 3-6). Climate variation at century scales controlled fire frequency and successional dynamics (Gavin et al. 2007, Long et al. 1998, Walsh et al. 2015). Fire frequency, for instance, was relatively high during the Medieval Warm climate anomaly about 1,000 years ago, but declined during the Little Ice Age between 1400 and 1850 BP. The low fire frequency in these systems was due to chronically high fuel moistures and infrequent lightning ignitions (Agee 1993) (fig. 3-10). Large high-severity fires would typically occur during unusually dry periods when synoptic weather patterns created strong hot and dry east or north winds (Agee 1993; Morrison and Swanson 1990; Weisberg 1998; Weisberg and Swanson 2001, 2003), but even those fires typically left patches with surviving live trees, which would contribute to regeneration and habitat diversity. As in other settings, the frequency-size distribution of fires followed a negative exponential distribution; i.e., the smallest fires were the most numerous, and the largest fires accounted for the majority of area burned (e.g., see Moritz et al. 2011).

Humans have played a role in fire occurrence in these forests. American Indian use of fire would have contributed to fire regimes, especially in drier regions and in local areas near Indian settlements in western valleys and coastal areas (Agee 1993, Walsh et al. 2015) (see chapter 11). We did not adjust the mapping of fire regimes for potential effects of Indian burning. Scientific opinions differ regarding the contribution of Indian burning to these forests over evolutionarily relevant time scales. Clearly, the contribution of such burning was locally important in many areas. Euro-American influence began around the time of settlement (early 1800s) and coincided with warming and progressively drier weather patterns as the Little Ice Age began winding down, potentially exacerbating fire activity (see Weisberg and Swanson 2003).

In the drier parts of the moist forest subregion, fires were more frequent and mixed in severity, although medium to large patches of high mortality were present (table 3-1). The moderately frequent to somewhat infrequent regime (Morrison and Swanson 1990, Van Norman 1998) occurred across a range of potential vegetation types (fig. 3-8), along the eastern slopes of the Olympic Mountains and Coast Ranges, and the interior valleys extending to the western slopes of the Cascades in Oregon (fig. 3-6). The climate there is warmer and drier than in the infrequent fire regime, and lightning ignitions are more frequent (fig. 3-10). Patches of high-severity fire could be highly variable and were probably somewhat smaller than in the infrequent high-severity regime (Morrison and Swanson 1990) (fig. 3-9). Mixed-severity fires likely affected many older forests (Weisberg 2004). For example, many of the existing old-growth trees in the southern western Cascades of Oregon and interior parts of the Coast Range in Oregon showed evidence of low-severity fire occurrence (fig. 3-11). Severe windstorms also played a role in forest dynamics

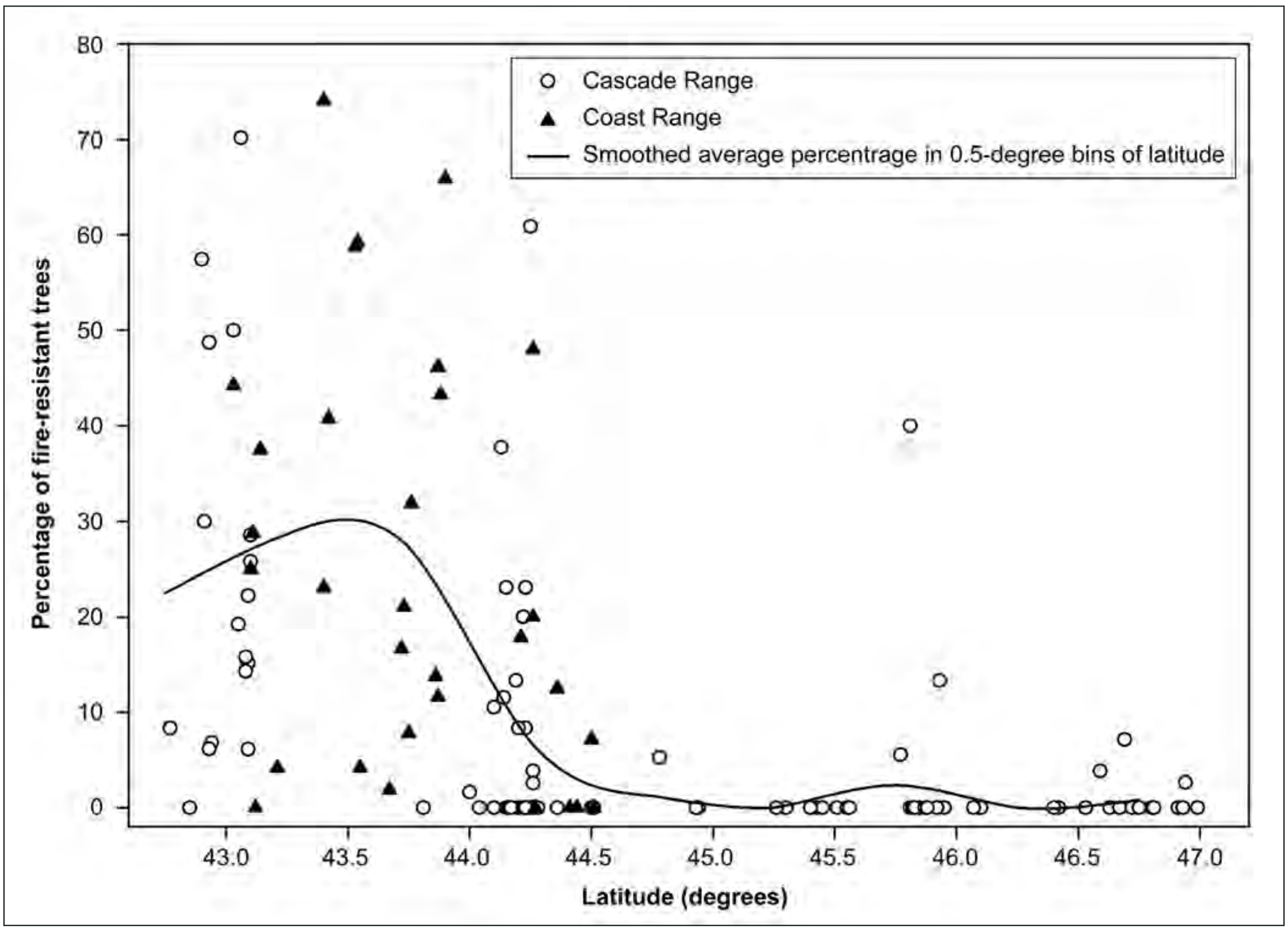


Figure 3-11—Percentage of fire-resistant mature and old trees with evidence of fire (scars or charred bark) in the western Cascades and Oregon Coast Range in relation to latitude. Line is smoothed running average in 0.5° bins. The increase in evidence of fire on tree boles around latitude 44.5° N in Oregon (about the latitude of Corvallis) indicates a shift from infrequent, high-severity to moderately frequent, mixed-severity fire regimes moving from north to south (right to left). Data source: Spies and Franklin (1991).

west of the Cascade crest (Knapp and Hadley 2012). Wind occasionally created large stand-replacement patches and frequently small gap disturbances across all forest types in the region. While the frequency of wind disturbance is greatest near the coast (Harcombe et al. 2004) and in the Columbia Gorge (Sinton et al. 2000), infrequent large regional-scale wind events, such as the 1805 "perfect storm" experienced by Lewis and Clark (Knapp and Hadley 2011), the 1962 Columbus Day windstorm (Lynott and Cramer 1966), and the 1981 Big Blow of November 14th can affect forests across the west side of Oregon and Washington. The 1962 storm may be the largest natural disturbance event in regional forest history, blowing down 11 billion board feet of timber across Washington and Oregon, in concentrations of over 80 ac/mi^2 (12.5 ha/km^2) in some areas (Teensma et al. 1991). The frequent occurrences of large windstorms in coastal areas control tree growth, forest structure, and successional patterns (Knapp and Hadley 2012). More frequently, windthrow disturbances are typically related to patterns of topographic exposure, which can concentrate windflow (Harcombe et al. 2004, Sinton et al. 2000, Wimberly and Spies 2001), root disease, or edges of older and younger patches of forests (Franklin and Forman 1987, Sinton et al. 2000) created by clearcutting or other stand-replacement disturbances.

Biotic disturbance agents play important roles in succession, and in ecosystem processes and patterns of moist forests (table 3-2). They also play important roles in producing dead and damaged trees that serve as wildlife habitat (Bull 2002). These agents primarily include root diseases and bark beetles, although foliage diseases, defoliators, heart rots, rust diseases, and dwarf mistletoes can also be quite important. Root disease fungi and related organisms cause root death, heart rot of large roots and tree butts, reduced tree productivity, top dieback, and tree mortality, while interacting with bark beetles or other mortality agents to influence gap dynamics and stand structure (Hansen and Goheen 2000, Lockman and Kearns 2016). *Phellinus sulphurescens* (syn *Poria weirii* or *P. weirii* in the older literature) clones are thought to occur on about 5 to 16 percent of the landscape in the moist forests (Lockman and Kearns 2016, Washington State Academy of Sciences 2013), for example. Root rot diseases are often called, "diseases of the site" in the sense that once established in a stand, the fungi can persist for decades on belowground wood depending on management or compositional changes (Hadfield et al. 1986, Shaw et al. 2009).

Foliage disease fungi can be major disturbance agents that influence competitive relationships and tree productivity potentially throughout a climatic region (Bednářová et al. 2013). However, foliage diseases in Pacific Northwest forests are best known in young plantation forests, and are poorly studied in natural, or especially, older forests (Shaw et al. 2011). Swiss needle cast, caused by the native fungus *Phaeocryptopus gaeumannii*, is currently causing an epidemic in managed Douglas-fir coastal forests of Oregon and Washington state, within about 35 mi (56.3 km) of the Pacific Ocean, reducing plantation productivity an average of 23 percent within a study area of the northwest Coast Range of Oregon (Maguire et al. 2002, 2011, Navarro and Norlander 2016, Ramsey et al. 2016, Ritóková et al. 2016). The disease is particularly associated with lower elevations of the infrequent–high-severity fire regime (fig. 3-6). The role of foliage diseases in the development of forest stands, and in particular, old-tree crown dynamics, remains elusive. It is generally thought that maintaining tree species diversity, canopy complexity, and adherence to site compatible seed zones reduces the threat of foliage diseases to forest health (Shaw et al. 2009).

Bark beetles and wood borers are diverse, but major disturbance from mortality is mostly associated with climatic events such as drought, ice/snow breakage, and windthrow (Furniss and Carolin 1977). Two particularly important species are the fir engraver (*Scolytus ventralis* (LeConte)) in true firs (Ferrell 1986) and the Douglas-fir beetle *(Dendroctonus pseudotsugae* (Hopkins) in Douglas-fir (Furniss and Kegley 2014). Mortality from both insects is associated with root diseases and drought, and, in the case of the Douglas-fir beetle, with windthrow events (Furniss 2014a, 2014b; Goheen and Willhite 2006). Typically, flareups of mortality from this beetle persist for a few years and then abruptly subside (Furniss and Carolyn 1977, Goheen and Willhite 2006).

Table 3-2—Major biotic disturbance groups, effect on trees, and ecological influences in forests of the Northwest Forest Plan area

Disturbance group[a]	Tree effects	Ecological influences
Root diseases	Major mortality agent	Alters stand composition/structure
	Growth reduction	Creates snags, down wood
	Root death	Wildlife cavities Creates ant/termite habitat Attracts bark beetle mass attack
	Root/butt heart trots	Increases surface fuels
Live tree decays	Wood volume reduction	Wildlife cavity creation
	Increased windsnap	Reduced carbon sequestration
		Creates ant/termite habitat
Foliage diseases	Reduce foliage retention	Less competitive in stands
	Reduced growth	Reduced carbon sequestration
	Carbon starvation	Alters stand composition/structure
Cankers and rusts	Branch, top, tree death	Reduced carbon sequestration
	Foliage loss	Reduce host species abundance
	Tree deformation	Wildlife habitat
Dwarf mistletoe	Growth reduction	Alters forest structure/composition
	Top, branch, and tree death	Encourages passive crown fire
	Branch and tree deformation	Wildlife habitat platforms
	Increased susceptibility to other agents	Influence with fire
Bark beetles	Major mortality agent	Alters composition/structure
	Patch attacks on bole	Increases forest fuels
	Top and branch death	Wildlife habitat
Defoliators	Growth loss	Alters composition/structure
	Top dieback	Reduces canopy density
	Mortality	Wildlife habitat impacts
Aphids, adelgids and scale insects	Growth loss	Alters forest structure
	Leaf, branch, and tree death	Reduced carbon sequestration
Terminal and branch insects and pitch moths	Tree leader death	
	Stunted growth	Forest structure
	Tree deformation	Reduced competitive ability

[a] Groups from Shaw et al. (2009).
Source: Furniss and Carolin 1977, Goheen and Willhite 2006, Scharpf 1993, Shaw et al. 2009, Wood et al. 2003.

Other important biotic agents include the hemlock dwarf mistletoe (*Arceuthobium tsugense* Rosendahl), which is the only known moist forest dwarf mistletoe, and can dramatically influence forest structure (Muir and Hennon 2007). The plant occurs localized in western hemlock-dominated forests, where it is estimated to infect 10.8 percent of the western hemlock trees in Oregon (Dunham 2008). Hemlock dwarf mistletoe has a strong connection to fire history (Shaw and Agne 2017); more frequent fires favor less mistletoe.

Disturbance regimes of dry forests—

This region includes the mid to lower elevations of the eastern Cascades from Washington to California, southwestern Oregon, in the Klamath region, and inland portions of the California Coast Range. It spans a range of dry forest potential and current vegetation types, including ponderosa pine, Douglas-fir, and white fir (figs. 3-4 and 3-6; table 3-1). Fire is the major stand-replacement disturbance in this region followed by outbreaks of major forest insects.

The more moist and productive part of this region experienced a frequent, mixed-severity regime with fire-return intervals of 15 to 50 years (Agee 1991, Agee et al. 1990b, Stuart and Salazar 2000, Taylor and Skinner 1998, Van Norman 1998, Whitlock et al. 2004, Wright and Agee 2004). Fire events contained medium to large patches of high-mortality and extensive areas of low- and moderate-severity fire. The 2002 Biscuit Fire is an example of such a fire (Halofsky et al. 2011, Thompson and Spies 2009) (fig. 3-12). The occurrence of mixed-severity fire even at short fire-return intervals (e.g., <25 years) probably reflects the higher moisture conditions and site productivities in parts of this regime in comparison to the very frequent, low-severity dominated regime in

Thomas Spies

Figure 3-12—Mosaic of high-severity burn patches in a portion of the 2002 Biscuit Fire in southwest Oregon in an area classified as historically supporting a frequent, mixed-severity fire regime (fig. 3-6). A large portion of the area with surviving tree canopies experienced low-severity surface fire.

California or the eastern Cascades. Patterns of mixed-severity patches were historically shaped by prevailing topographic features (Beaty and Taylor 2001; Hessburg et al. 2015, 2016; Taylor and Skinner 1998; Weatherspoon and Skinner 1995) with variable proportions of both surface and crown fires accounting in part for tree mortality in mixed-severity fire regimes (Perry et al. 2011, Stephens and Finney 2002).

The very frequent (<25 years) low-severity regime occurs in the driest forests[7] of the NWFP area in a variety of pine, dry Douglas-fir, dry grand or white fir, and oak potential and current vegetation types (figs. 3-4 and 3-6, table 3-1, app. 1). Historically, fires burned very frequently, with average fire intervals between 5 and 25 years (Bork 1984; Everett et al. 2000; Sensenig et al. 2013; Soeriaatmadja 1965; Taylor and Skinner 1998, 2003; Weaver 1959), although for many forests the range was much narrower. Overall, tree mortality from fire was low, with typically <20 percent of the trees killed in fires, and most high-severity effects occurring in very small patches (<1 ac [<0.40 ha]). Fire severity was primarily influenced by fine-scale patterns of surface fuels and topography (Churchill et al. 2013, Larson and Churchill 2012). Fuels were reduced frequently enough that active crown fire was infrequent. Frequent fires often created multicohort stands with low tree density and canopy cover (Hagmann et al. 2013, 2014; Sensenig et al. 2013). Larger patches (>250 ac [>101 ha]) of high severity could occur but were uncommon in most areas (Agee 1993, Rollins 2009; Skinner 1995; Taylor and Skinner 1998, 2003) and were linked to topography (Taylor and Skinner 1998, 2003). The forested landscape was dominated by open forests with islands of denser vegetation, including clumps of trees of various sizes (Churchill et al. 2013, Hessburg et al. 2007, Larson and Churchill 2012, Lydersen et al. 2013, Perry et al. 2011). Some scientists (e.g., Baker 2012) dispute the idea that these dry forests experienced a regime dominated by frequent, low-severity fire, and argue instead that they commonly experienced larger patches of high-severity fire (see section on alternative viewpoints below for more discussion of this).

Wind is not a major disturbance agent in drier forests of the region that are typically inland from coastal areas, and south of areas where the strongest windstorms occur. Coastal California is south of most of the mid-latitude cyclones that affect the Oregon and Washington coast (Lorimer et al. 2009). Coastal redwood forests experience winter storms and high winds, but effects appear to be limited to canopy damage and scattered blowdown of trees on high ridges (Hunter and Parker 1993, Lorimer et al. 2009). Drier ponderosa pine, Douglas-fir, and mixed-conifer forests experience scattered windthrow that creates canopy gaps and fine-scale pit and mound microtopography (Weaver 1943), but we are not aware of studies that document occurrence of larger patches of windthrow. Reilly and Spies (2016) report that between the 1990s and mid 2000s, wind was a very small component of all natural sources of mortality in dry forests of the Pacific Northwest. Agee (1994) reported similar results for the dry interior forests.

Major biotic disturbance agents in dry forests include several root diseases and host specialized dwarf mistletoes as chronic long-term stand influences that are associated with creating complexity in forest patches by killing and deforming trees, creating snags and gaps, and influencing fuels and fire (Goheen and Willhite 2006, Hadfield et al. 1986, Hawksworth and Wiens 1996, Lockman and Kearns 2016, Shaw and Agne 2017) (table 3-2). Major bark beetle and defoliator disturbances tend to be episodic, although individual old-tree death caused by bark beetles is chronic in some forests. Large outbreaks are more common in the eastern slope of the Cascades than in northern California, where tree species diversity, complex terrain, geological diversity, and contrasting site microenvironments may reduce the potential for widespread outbreaks. Heart rots, rust diseases, cankers, as well as foliage and tip diseases and insects may be locally significant, especially heart rots, which create cavities for wildlife (Bunnell 2013).

Root diseases are widespread in dry forests (Filip and Goheen 1984, Hadfield et al. 1986, Lockman and Kearns 2016), where they play an integral part in forest stand dynamics and canopy gap formation. In northwestern California, Hawkins and Henkel (2011) found that root diseases caused more mortality and gap formation in white

[7] In the Klamath and southern Cascades of California, these regimes occur where the climate is characterized by long warm/dry seasons but relatively high precipitation, which is concentrated in the winter months.

fir than Douglas-fir, which in the absence of fire, allowed Douglas-fir to better persist in forest stands. This is not always the case in the dry forests.

Dwarf mistletoes are host specialized parasitic seed plants that are a major influence on dry forest structure. Host-specialized mistletoes infest nearly all species, where they create structures such as witch's brooms, dead tops, dead branches, and fuel ladders (Hawksworth and Wiens 1996, Mathiasen and Marshall 1999, Shaw et al. 2004). A key ecological function of dwarf mistletoes is the creation of wildlife habitat structures via their large witch's brooms, which provide nesting and roosting platforms for a variety of forest birds and other small mammals (Shaw et al. 2004). Douglas-fir dwarf mistletoes can provide the majority of nesting sites for the spotted owl in dry interior forests (Buchanan et al. 1995, Forsman et al. 1984). Dwarf mistletoe distribution and abundance is related to fire history; with more regular fire there is less dwarf mistletoe because heavily infested trees are prone to torching or passive crown fire initiation (Shaw and Agne 2017). Although fire influences dwarf mistletoe, dwarf mistletoe also influences fire behavior by creating complex fuels structures, contributing to surface fuels, increasing ladder fuels, decreasing canopy base height, and increasing canopy bulk density.

Bark beetles are associated with most mortality events in dry forests, however, determining whether the beetles are to blame for individual tree mortality can be a challenge. Drought, dwarf mistletoe, root diseases, defoliators, and other biotic or abiotic factors can all predispose weakened trees to bark beetle mass attack. Bark beetle outbreaks can also be initiated by long-term drought events, and these outbreaks can last well over a decade. Bark beetles are also host specialized, and they influence forest stand structure and development by killing specific tree species. In the aftermath, tree mortality associated with beetle outbreaks can contribute significantly to forest fuels, but it can take more than a decade or two for the snags of the former forest structure to fall down and accumulate on the forest floor. Major bark beetle outbreaks typically occur in dry forests east of the Cascade crest where expansive stands of lodgepole pine (*Pinus contorta*) have been hit very hard by mountain pine beetle (MBP) (*Dendroctonus ponderosae*) (Gibson et al. 2009). Recent large bark beetle mortality events associated with periods of extended drought in the southern and central Sierra Nevada of California suggest that the potential for major climate change-driven outbreaks is ongoing and may result in species conversion in some areas (Moore et al. 2017). The interaction of fire with prior MPB events has become a significant research emphasis following large outbreaks throughout western North America. Following MPB mortality, canopy fuels decrease drastically within a few years, and depending on composition of the stand, surface fuels will significantly increase with time (Hicke et al. 2012).

Defoliators on the east side of the Cascade Range are a major disturbance agent in forest stands, with the western spruce budworm, Douglas-fir tussock moth (*Orgyia pseudotsugata*), pine butterfly (*Neophasia menapia)*, and Pandora moth (*Coloradia pandora*) potentially able to defoliate large regions (Furniss and Carolin 1977, Goheen and Willhite 2006). Outbreaks of the western spruce budworm (*Choristoneura occidentalis*) have not occurred in dry forests of California and southwestern Oregon (Brookes et al. 1987), although the Douglas-fir tussock moth may defoliate true firs, and the Pandora Moth may affect ponderosa pine (Brookes et al. 1978, Wood et al. 2003). Defoliators have the potential to shift composition of stands to nonhosts owing to reduced growth and mortality effects, as well as increased potential for bark beetle infestation in defoliated trees (Brookes et al. 1978, 1987). The interactions of fire with forest defoliators suggest a negative association of fire and defoliated stands (Meigs et al. 2015).

Forest Succession and Landscape Dynamics

Moist forests—

Succession—Our synthesis of this regime is primarily based on studies from Douglas-fir and western hemlock forests (i.e., the western hemlock potential vegetation type) (Franklin et al. 2002, Oliver and Larson 1996, Reilly and Spies 2015, Spies et al. 1988). Patterns of postfire and postwind stand-replacement succession for other potential vegetation types in this fire regime, which have received less study (e.g., mountain hemlock in Oregon and Washington, Pacific silver fir potential vegetation types) may have been generally similar, but they differ in a number of ways, including species composition, varied pathogen and insect associations, and slower rates of structural and compositional development. These potential vegetation types also likely have lower levels of total biomass relative to Douglas-fir/western hemlock forests in late-successional stages, owing to shorter and cooler growing seasons.

The archetypal or standard model of forest succession in this forest region and under these disturbance regimes has been characterized in many papers but is developed in greatest depth by Franklin et al. (2002), and most recently by Franklin and Johnson (2017) and Franklin et al. (2018) (fig. 3-13). Simply stated, after a stand-replacement disturbance such as wildfire or windstorm (1) considerable dead and live legacies of the disturbance remain for decades; (2) new shade-intolerant and tolerant plants and early-seral associated wildlife colonize a site; and (3) a dynamic mix of nonforest and forest plant species develops and persists until conifer canopy closure, which may take between 30 and 100 years. The forest then goes through a process of structural and compositional changes and stages driven by growth, competition, immigration of shade-tolerant species, and fine- to moderate-scale mortality events that create canopy gaps of various sizes (Bradshaw and Spies 1992, Spies et al. 1990). These canopy gaps can promote growth of shade-tolerant trees growing in the understories of densely shaded forests. This is not the only successional pathway that forests followed in this large and ecologically diverse region, but it is a common one, especially in wetter and northern parts of the western hemlock potential vegetation type in cover types characterized by Douglas-fir and western hemlock (Winter et al. 2002a, 2002b), and a lack of fire between stand-replacement events. We characterize this model of succession further below and describe its variations and other successional pathways that can occur.

Early post-stand-replacement fire vegetation in the western hemlock–Douglas-fir forests of the western hemlock zone typically occurred as heterogeneous mosaics of grasses, herbs and shrubs, and hardwoods often with high levels of dead snags and down wood, and high species richness (Donato et al. 2011, Reilly and Spies 2015, Swanson et al. 2011) (fig. 3-3). Species compositional change, which can be rapid over the first 20 years as a function of the relative importance of invading and residual plant species groups, differs with time, the availability of propagules, disturbance characteristics, and properties of the environment (Halpern 1988, 1989). Standing dead tree structure and decay states are also dynamic within western conifer forests during the first decade or two following fire (Russell et al. 2006). Studies of post-wildfire conifer forests in the Western United States indicate that wildlife use of early-seral vegetation following fire and logging can change rapidly with time-since disturbance, with some species appearing in the first few years before disappearing later and others increasing in abundance as snag conditions and plant species composition changes (Saab et al. 2007, Smucker et al. 2005). Gashwiler

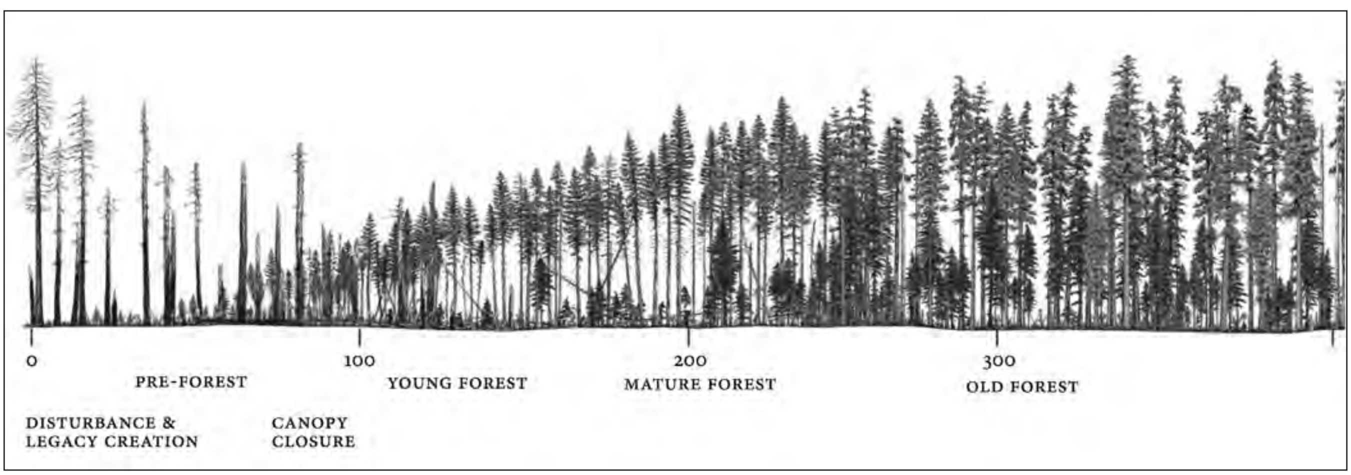


Figure 3-13—A common stand developmental pathway for a Douglas-fir and western hemlock forest following stand-replacement wildfire (from Franklin et al. 2018).

(1970) found that small mammal communities were quite dynamic in the first 10 years following clearcutting of an old-growth forest in the western Cascades of Oregon. The general pattern seems to be that while the "pre-forest" or early-seral stage can persist for many decades, the plant and animal communities are dynamic within that stage, and some species and communities are ephemeral.

Dead wood levels were especially high where prefire forests were late successional or old growth (Spies et al. 1988). Where fires burned early-successional and younger forest stand conditions, dead wood legacies were typically few and composed of smaller down logs (Nonaka et al. 2007, Spies et al. 1988). In contrast, where fires burned in forests containing large trees, levels of down wood were high, and individual pieces of large down wood may have persisted for several centuries while undergoing decomposition. Charcoal deposits from fires lasted in soil for up to one or more millennia (DeLuca and Aplet 2008).

Scientific and conservation interest in early-successional vegetation has increased in recent years as scientists learned about ecosystem responses to severe disturbance from studies of the eruption of Mount St. Helens (Dale et al. 2005) and high-severity wildfires that have occurred in the Western United States in recent decades (e.g., Donato et al. 2011; Hessburg et al. 1999a, 1999b; Hutto et al. 2016). Post-high-severity and mixed-severity disturbance ecosystems are generally understood to support unique biodiversity and ecosystem functions (Donato et al. 2011; Franklin et al. 2017; Hessburg et al. 2016; Swanson et al. 2011, 2014) relative to closed-canopy forests. This understanding is based largely on studies of clearcuts (e.g., Halpern 1988, Harr 1986) and volcanic eruptions (Dale et al. 2005) in the Northwest Forest Plan area, and few studies have been conducted in early-seral vegetation following wildfire or windstorms (e.g., Fontaine et al. 2009, Larson and Franklin 2005). Early-successional stages following natural disturbances are rich in biological legacies that include surviving organisms and organic matter such as dead trees. With tree canopies gone or greatly reduced, other life forms, including shrubs, grasses, and herbs often dominate the site, taking advantage of higher resource levels in light, water, and nutrients. These legacies clearly influence postdisturbance succession, stand development, and ecosystem function, though the variability in these relationships over time is not well understood. Variation in disturbance severity and predisturbance forest conditions has strong influence on legacy patterns, and subsequent forest succession that can persist for hundreds of years (Donato et al. 2011, Dunn and Bailey 2016, Spies et al. 1988). In sum, early-seral stages are important when managing for conservation of native biodiversity and resilience in forested ecosystems and landscapes.

Given new scientific perspectives on early-seral vegetation, some have proposed that new terminology be used to describe it. For example, Franklin et al. 2018 suggest that early-seral vegetation be termed "pre-forest" because trees are not the dominant life form, although they are often present as seedlings. They also suggested that the term "early-seral forest," which has been used to define this stage, is not correct because this stage is not forested and introduces a "tree-centric" bias to discussions about conservation and management (Franklin et al. 2018). Other terms that have been used to describe this stage include grass-forb, shrub-seedling, stand initiation, and cohort establishment. Terminology to describe successional stage, structural or developmental stage, or seral stage can be confusing and not interchangeable (Powell 2012). For example, some trees such as Douglas-fir and red alder are characterized as "early-seral" species (Franklin and Hemstrom 1981, Klinka et al. 1996), which can form early-seral stands or forests. The ambiguity of the terminology around postdisturbance changes in vegetation (including later successional stages) makes it important to define how terms are used (e.g., Powell 2012), and in the case of early-seral or pre-forest vegetation to clearly identify the ecological characteristics (life forms, species, structures) and functions (habitat, nutrient cycling, productivity) that reflect the underlying meaning and use of those terms.

The timing, composition, and structure (including cover thresholds) of tree canopy cover closure (e.g., canopy cover >70 percent (Yang et al. 2005) would have differed regionally by site conditions, disturbance characteristics, and seed source availability (Freund et al. 2014, Yang et al. 2005). Canopy closure may have occurred as early as 20 to 30 years following fire in moist productive sites, or where seed sources persisted in a canopy seed bank (Larson and Franklin 2005), but could have taken almost 100 years on other sites, after very large fires and with limited seed sources. These observations are based on studies of mature forests from the western Cascades (Freund et al. 2014). Tree establishment ended as the forest floor was covered by shrub and herbaceous vegetation, and tree canopies eventually closed (Freund et al. 2014, Tepley et al. 2014).

Not all stands or patches followed the same pathway to older forest structure. Multiple successional pathways would have occurred that varied in timing of composition and structural change over the first 100 to 200 years or longer (fig. 3-14) (Spies 2009). In riparian areas and moist coastal upland forests, shrubs and hardwood trees would often become established immediately after fire, limiting the establishment of conifer trees for many decades, and creating patches of hardwoods and shrubs with scattered conifers (Spies et al. 2002). Ultimately, those shorter lived hardwoods would die, leaving lower density conifer stands (or stands with variable-canopy dominance) with large dominant trees and well-developed crowns. For example, Spies and Franklin (1991) found that some 100-year-old stands of Douglas-fir and western hemlock that developed along with shrubs and hardwoods in the Oregon Coast Range had structural diversity that approached that of 400-year-old stands. Variability in seed sources, productivity, competition with shrubs and hardwoods, and partial stand replacement disturbances would have led to low-density relatively open younger forests where conifer canopy closure never occurs. These processes and pathways may actually be a faster route to complex older forest structure in some places than pathways that go through stages characterized by a higher density of conifers and conspecific competition (Donato et al. 2011, Tappiener et al. 1997).

Where closed-canopy forests developed, succession was driven by processes of growth, competition, understory development, maturation, and small- to moderate-size canopy disturbances from wind, insects, disease, fire, hydrologic, or geomorphic processes (Franklin et al. 2002). Somewhat arbitrarily, 80 years after conifer forest establishment has been used as the onset for "mature" (e.g., OGSI 80) Douglas-fir forests, and 150 to 200 years for the onset of multilayered old-growth forests (OGSI 200), depending on environment and disturbance history (Franklin et al. 2002, Spies and Franklin 1991). Eighty years was used as the threshold for late-successional/old growth in the NWFP (USDA FS 1994) because that is about the earliest time when such stands begin to resemble maturing forests in the moist forest (does not apply to the dry forest zone). Analyses of chronosequences indicate there is considerable variation in forest structure around these age breaks (Spies and Franklin 1991) (fig. 3-15) likely driven by multiple successional pathways, legacies, and time since disturbance. The stands (i.e., sample plots) in figure 3-1 would have followed individual development pathways, some pathways may be sigmoid shaped in the case of stands developing after a nonforest condition, other pathways may have been more U-shaped in the case of stands developing with significant live or dead legacies of the predisturbance old-growth forest (Spies and Franklin 1988).

The variability in structure with stand age indicates that at a regional scale, age or time alone is only a partial predictor of forest structure. The structural features of mature and old-growth forests would have included medium- to large-size (e.g., >40 inches) shade-intolerant tree species; smaller shade-tolerant trees of similar and lesser age in the mid to lower canopy layers; large standing and down dead tree boles; and horizontal and vertical structural heterogeneity of live and dead trees. Not all stands would have grown for centuries without stand-replacement fire—sometimes reburns within a few decades of a fire would occur consuming decayed dead wood and restarting succession (Donato et al. 2016, Gray and Franklin 1997, Nonaka 2003, Tepley et al. 2013).

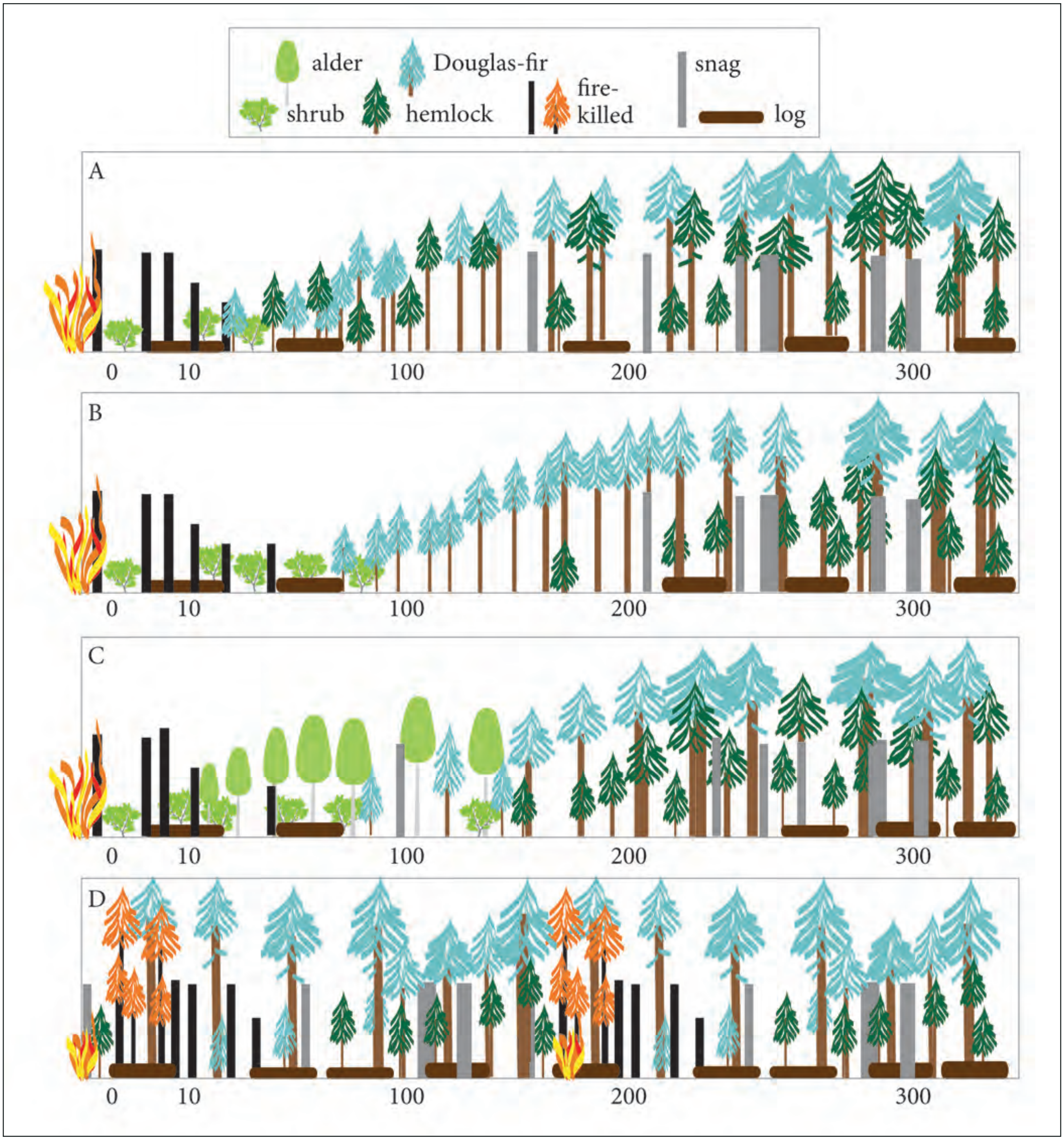


Figure 3-14—Multiple pathways of succession that could occur in the moist forests. Pathway A occurs when Douglas-fir canopy closure occurs within 50 years after a fire and western hemlock establishes early in succession. Pathway B occurs when the pre-forest shrub-dominated stage persists for many decades and hemlock is slow to establish. Pathway C occurs where shrubs and hardwood trees dominated early-successional development and reduced conifer densities so that conifer trees would not go through a self-thinning phase and large-diameter conifers and complex older forest structure would develop well before 200 years. Pathway D occurs where a partial stand-replacement fire occurs periodically in older forests and creates patches of dead trees, initiating new age cohorts of Douglas-fir or western hemlock trees beneath the surviving canopy and in openings created by the fire.

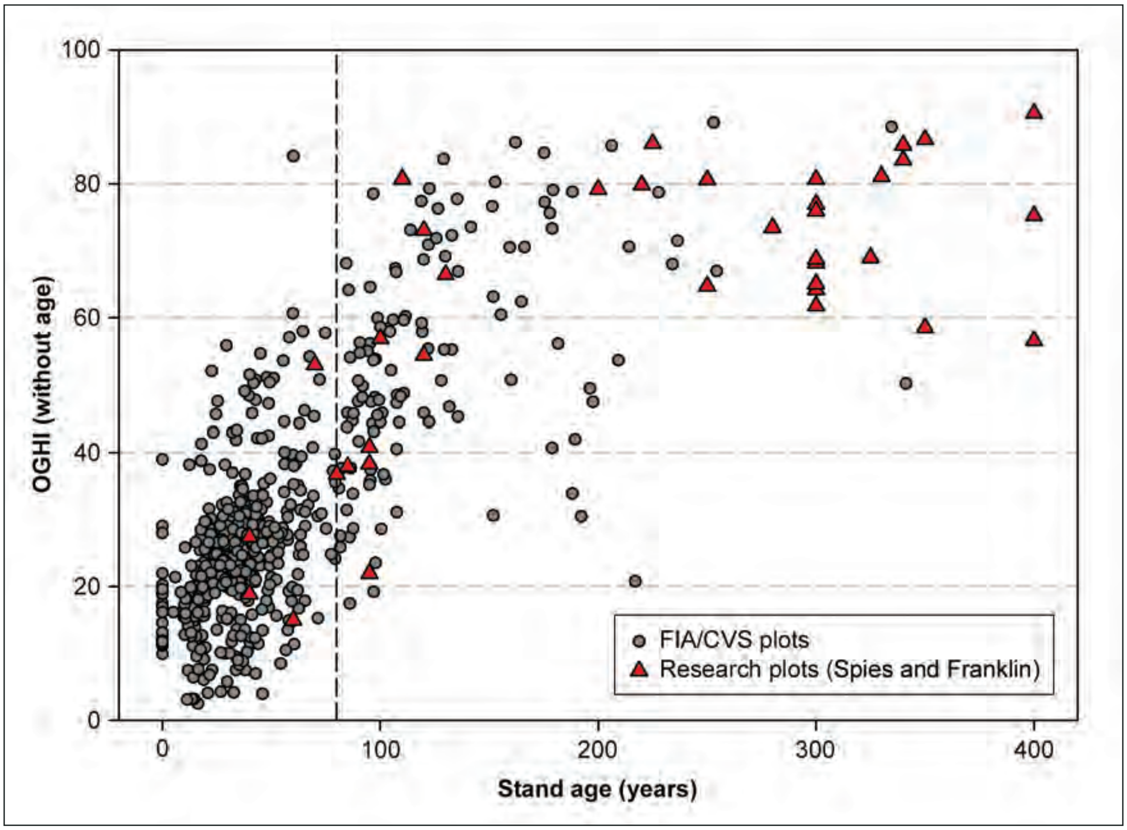


Figure 3-15—An old-growth forest habitat index (OGHI) (Franklin et al. 2005) in relation to stand age for forest inventory and research plots in the Oregon Coast Range. The index is based on number of large trees, large snags, volume of down woody debris, and tree size diversity, which is a surrogate for canopy layering. Age was not used to develop the index. The index is similar to the structure index used in Davis et al. 2015. FIA/CVS = Forest Inventory and Analysis/Continuous Vegetation Survey.

Successional and landscape dynamics in the drier, southern part of the western hemlock zone, where fire frequency was 50 to 200 years (fig. 3-4), would have included some of the same pathways as would have occurred in the infrequent fire regime, but with different frequencies of those pathways across landscapes. At the scale of large patches and small landscapes (e.g., 10^2 to 10^4 ac or ~40 to 4000 ha), these forests would have had more age, structural and compositional heterogeneity than equivalent areas for the moister parts of the region where an infrequent fire regime occurred (fig. 3-16). For example, reanalysis of data from Spies and Franklin (1991) from the old-growth forests in the southern western Cascades of Oregon indicated that stand ages (age of the oldest Douglas-firs in the stand) were younger (~270 years) and basal area, proportion of shade-tolerant trees, and density of large snags and volume of down wood were all much lower than in old-growth stands in the northern Cascades of Oregon and the Cascades of Washington (400 to 500 years), after controlling for topography and aspect. Ares et al. (2012) found that snag densities in older forests in western Oregon also varied by aspect, with lower densities on south-facing slopes and in the foothills of the Cascades, where fire frequencies are higher than in the Coast Range. The mature and old-growth stages probably have more age classes of Douglas-fir than in the infrequent, high-severity regime forests as a result of more frequent partial stand-replacement fire (Dunn 2015, Tepley et al. 2013) (figs. 3-16 and 3-17). For example, Tepley et al. (2013) found that 85 percent of the older forest in their central western Oregon Cascades study area (primarily western hemlock potential vegetation type with some areas of Douglas-fir potential vegetation type) experienced non-stand-replacing wildfire during its centuries-long development (fig. 3-14D). These fires killed a portion of the

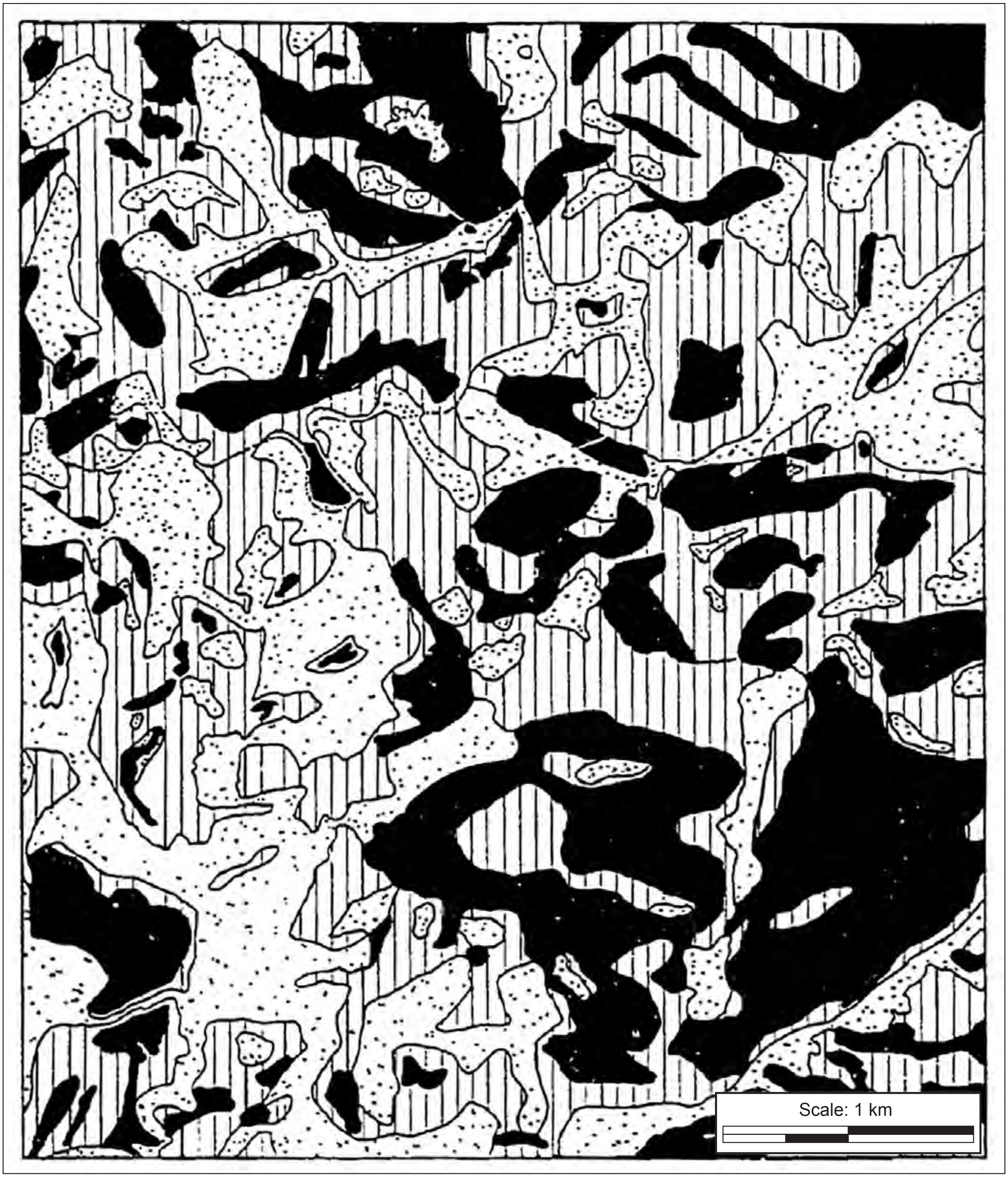


Figure 3-16—Mosaic of fire severity patches in a Douglas-fir and western hemlock landscape in the western Cascade Range of Oregon. Black = a high mortality area (>70 percent), vertical lines = moderate mortality (30 to 70 percent), and stippled = low mortality areas (<30 percent). From Morrison and Swanson 1990.

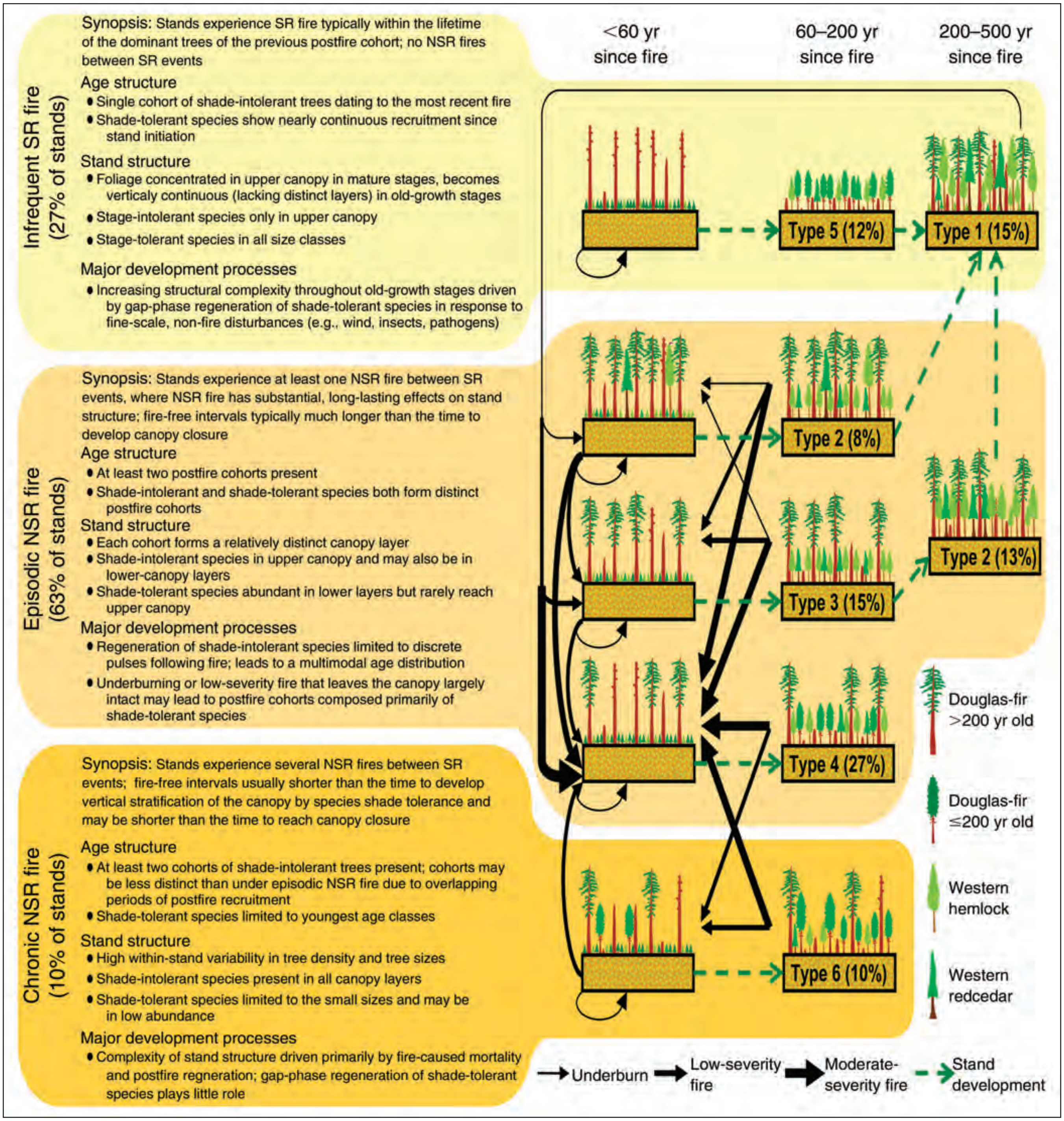


Figure 3-17—Conceptual model of stand-development pathways in Douglas-fir/western hemlock (current vegetation) forests in the moderately frequent, mixed-severity fire regime of the central western Cascade Range of Oregon. Dashed arrows represent stand development in the absence of fire, and solid arrows represent nonstand-replacing fire. Percentages indicate the percentage of the sample plots found in each structure type. SR = stand-replacing, NSR = non-stand-replacing. From Tepley et al. (2013).

overstory and established new cohorts of shade-tolerant or intolerant trees. Given the long time period that often occurred between fires, these landscapes of the infrequent and somewhat infrequent regimes would have typically been dominated by mature and old-growth forests.

***Historical landscape dynamics*—**Many of the current old-growth stands of the wetter portions of the moist forests date to around 400 to 500 years ago (Spies 1991), a period with widespread fire (Tepley 2010, Weisberg and Swanson 2003) associated with positive phase of the Pacific Decadal Oscillation, which produced warmer conditions and drought. This warm period with many fires was followed by the Little Ice Age when cooler temperatures caused a reduction in both lighting- and human-ignited fires (Walsh et al. 2015) that may have allowed stands that established during the warm period to develop into older, multistoried forests. Empirical estimates of the amount or variation in old-growth forests or of any successional stage that occurred prior to Euro-American settlement are not available from any historical studies. Maps from the early 1900s can be used to approximate the amount of old forest present in the mid-20th century, suggesting that about 50 percent of all forest lands in this regime were covered by older forest (defined then in terms of large dominant and codominant trees), but that number varied widely across landscapes and watersheds (Davis et al. 2015). However, it is not clear how earlier mapping criteria related to current definitions of old growth, and by the 1930s, significant areas of older forest had already been lost to land clearing for settlement and agriculture, logging, and human-set wildfires.

Empirical studies of fire frequency and severity can be used with statistical models and other simplifying assumptions to estimate the age-class distributions that might have been present in a historical landscape (Agee 1993, van Wagner 1978). For example, Fahnestock and Agee (1983) used historical maps and statistical models to estimate fire cycles in western Washington. They found the proportion of large trees to be 0.6 in Douglas-fir, 0.82 in western hemlock, and 0.87 in mountain hemlock forest cover types. Spies and Turner (1999) estimated that on average, 61 percent of a given landscape would be old growth (>150 years since stand-replacing fire) if fire frequencies were 300 years. They assumed a constant climate and fire frequency, equal flammability of successional stages, and high-severity fire—assumptions that are violated in real landscapes. For example, temperature and precipitation has varied considerably over the Holocene (past 11,700 years), including the past several thousand years when the current forest community assemblages developed (chapter 2). Susceptibility of successional stages often differ depending on fuel conditions and microclimate, and old forests can be less flammable than younger ones (Kitzberger et al. 2011).

Wimberly et al. (2000) used estimates of fire frequencies from lake cores in the Oregon Coast Range (Long et al. 1998) to estimate that fire rotation[8] varied from about 150 to 300 years during the past 3,000 years. They then used a spatial landscape simulation model to estimate that the mean amount of old-growth (>200 years) and late-successional forests (>80 years) (including old growth) could have varied from 39 to 55 percent and 66 to 76 percent, respectively, during the 3,000 years prior to Euro-American settlement. The model indicated that the minimum and maximum amount (i.e., the historical range of variation [HRV]) of old-growth and late-successional forest in the Coast Range during this period was 24 to 73 percent and 49 to 91 percent, respectively. The range of variation was also a function of the scale of observation, with larger ranges for smaller areas, e.g., at the scale of a NWFP late-successional reserve (LSR) (~100,000 ac [~40 470 ha]) the range of late-successional forest would have been 0 to 100 percent. These analyses suggest that older forest conditions would have dominated forests of the region, but large areas of dynamic early-seral vegetation and younger forest would occur episodically as evidenced by the large blocks of old-growth forest that would have originated after fire. LANDFIRE[9] (https://www.landfire.gov/NationalProductDescriptions24.php) estimated that the amount of "late

[8] Fire rotation refers to the time required to burn an area equal to a defined landscape area (e.g., 1,000 ac [404.7 ha]). The entire area may not burn during this period; instead, some sites may burn several times and others not at all, but the summed area is equal to the defined area. Fire rotation = fire cycle.

[9] LANDFIRE is an interagency geospatial data development program that used expert opinion to model historical amounts of vegetation stages for potential vegetation types based on published literature. The estimate of amounts of vegetation classes do not include historical ranges.

development" closed-canopy forest for the western hemlock zone was 70 percent, and the amount of open "early development" vegetation was 5 percent. Estimates of the HRV in successional stages are still needed for the NWFP area.

At the scale of regional landscapes or ecoregions, models suggest that early-successional patches occupied <20 percent of the area on average but may have reached as high as 30 percent over the span of several thousand years (Wimberly 2002). At the scale of LSRs, some watersheds may have been entirely composed of early-seral conditions after wildfires. Studies from Washington and southwest British Columbia (Dunwiddie 1986, Hallett et al. 2003) indicate that fire-return intervals were much longer in the northern part of this regime, so periods when early-successional conditions were abundant in these ecoregions were probably less than in the Oregon Coast Range. Moreover, the amount of fire and early-successional forest probably varied considerably over the past several thousand years in resonance with climatic variation.

The HRV in old-growth and other successional stages in the drier part of the western hemlock and other potential vegetation zones is less well known. It is also more difficult to estimate their abundance with statistical or simulation models given that many fires were non-stand replacing (Weisberg 2004) and resulted in multiaged patches and a large range of stand structures with a wide range of large live and dead tree densities, and tree species compositions (fig. 3-17). Estimates of historical amounts of old growth (i.e., areas of older trees with canopy layering) have been made from a few localities in the drier parts of the region. In the eastern part of the Oregon Coast Range, Wimberly (2002) estimated that the amount of this type of old growth over the 1,000 years prior to 1850 would have been less than 30 percent, where the fire-return interval was about 75 years, and many fires were non-stand-replacing (Impara 1997). The LANDFIRE estimates of these classes of historical amounts of "late" and early-development forest in drier parts of the western hemlock zone were 60 percent and 15 percent, respectively (https://www.landfire.gov/NationalProductDescriptions24.php). The amount of dense old growth without a history of non-stand-replacing wildfire, was probably less in these types, however, while the amount of other types with old trees would have been more common (Tepley et al. 2013) (fig. 3-17). The ecological functions and broader ecological significance of this diversity of old-growth forest conditions have not been studied, but Tepley et al. (2013) suggest this structural and composition diversity of older forests may have promoted resilience of large old-growth forest structures to disturbances and climate changes.

Dry forests—

As fire-return intervals decrease from over 200 years in the wetter forests to less than 25 years in the driest forests, the role of fire shifts from resetting succession and creating large patches of early-seral vegetation to regulating forest structure and dynamics altogether, and creating fine to mesoscale mosaics of different vegetation conditions, including early seral (fig. 3-18). At the shortest fire-return intervals, the simple model of succession and stand dynamics—i.e., a stand-replacement fire followed by long intervals of vegetation change without fire—no longer applies. In fact, the entire concept of succession and stand development toward multilayered old-forest structure in fire-dependent systems becomes problematic where fires are very frequent (O'Hara et al. 1996). A pathway of stand-replacement disturbance followed succession toward multilayered, closed old-growth forests still applies to some sites within the frequent, mixed-severity regime dry forests (Camp et al. 1997, Merschel et al. 2014), but not so much in the very frequent, low-severity regime where fire was more of an intrinsic ecological process than an external disturbance event. Forest structural stages (e.g., stem exclusion, old-forest multistrata, old-forest single stratum) can still be classified and identified in two dry forest fire regimes, but the structural conditions can be quite variable and complex, and pathways of change can be multidirectional owing to the interplay of fire severity, time since last disturbance, seed sources, and environmental heterogeneity (Reilly and Spies 2015). We discuss the two regimes separately below but recognize that for many landscapes and existing forest history studies, the two regimes may intermingle or have been lumped together.


Eric Knapp

Figure 3-18—Aerial photo of Beaver Creek Pinery showing spatial heterogeneity that can develop with frequent burning on a productive site in the southern Cascade Range of California.

Frequent, mixed-severity fire regimes—The potential vegetation types of the frequent, mixed-severity regime (15- to 50-year return interval) include Douglas-fir, grand fir, and white fir, and oak woodlands (fig. 3-4). The cover types of this regime include Douglas-fir, white fir, red/noble fir (*Abies procera*), and western white pine (*Pinus monticola*). Ponderosa pine can still be a component of some of these forests (Merschel et al. 2014). Forests of this type were characterized by multiaged cohorts of seral dominants and landscape mosaics created by medium to large patches of high-severity fire (fig. 3-12), but the landscapes were probably dominated by areas of moderate- to low-severity fire. In a Douglas-fir-dominated landscape of northern California, Taylor and Skinner (1998) found older stands with diverse age structure, but fire-return intervals were shorter (e.g., ~15 years), severities were lower, and large severe fires were uncommon compared to Douglas-fir forests of the western Cascades of central Oregon. Many of "mixed-severity" areas of the drier eastern part of northern California have been mapped in our classification into the very high frequency, low-severity regime (fig. 3-6). Stands with the most diverse age structure in the Taylor and Skinner (1998) study experienced the greatest number of fires, whereas stands with fewer age cohorts had experienced fewer fires. Those with the most diverse age structure were those most closely exhibiting late-successional structure. However, in landscapes where fires were mostly low severity, the age-class/fire association was unclear (Taylor and Skinner 2003).

Mixed-severity regimes in dry forests would likely result in higher diversity of plant and animal communities and patch (area that differs from its surroundings) heterogeneity compared to high-severity regimes or very frequent low-severity regimes (Hessburg et al. 2016, Perry et al. 2011). Areas of passive and active tree torching, mostly associated with clumps or groups of small understory trees with low limbs, would have created patches of tree mortality that would function as canopy gaps of various sizes in older forests. Subsequent fires, either by torching or girdling, would in turn thin these patches diminishing the even-aged group to a few individuals. Shade-intolerant tree regeneration would be more likely to establish in larger (e.g., >1 ac [0.04 ha]) high-severity patches. A prominent hardwood component was often associated with conditions emerging after mixed-severity fires. These hardwoods may play a pivotal role in continued mixed-severity fires (see discussion below).

The ecological importance of forests shaped by mixed-severity regimes (in both dry and moist forests) is widely recognized (DellaSala and Hanson 2015, Hessburg et al. 2016, Perry et al. 2011), but fine-scale studies that document how microclimate, wildlife, and fire respond to different expressions of vegetative heterogeneity, and different types of mixed-severity regimes have not been conducted. Our understanding of the mixed-severity regime in dry forests comes from patch- and landscape-scale reconstructions. That understanding is further complicated by lack of consistency in defining mixed-severity fire regimes across studies and lack of historical information about their spatial and temporal characteristics (app. 3). Several studies have characterized the spatial heterogeneity of patches dominated by this regime, especially for the eastern Cascades provinces (Hessburg et al. 1999a, 2000b, 2004, 2007; Perry et al. 2011).

The stand-development trajectories of high-severity patches could initially follow the pathway described by Franklin et al. (2002), but where shrubs or seed source limitations occurred, stand development might not proceed through the stem-exclusion closed-canopy stage. In addition, some elements of complex older forest structure (e.g., large-diameter trees and heterogeneous understories) might develop more rapidly than in the wetter forest types (Donato et al. 2011), which often have to develop following a relatively uniform and dense self-thinning phase. The trajectory of development of a low-density tree patch can be altered if the area is severely burned again before trees are mature (Coppoletta et al. 2015, Lauvaux et al. 2016, Tepley et al. 2017).

Topography would have been an important driver of the mosaic pattern. Ridges and south-facing aspects with more frequent fire would tend to support more open-canopy stands of multicohort shade-intolerant and fire-tolerant trees, while valley bottoms, benches, and more northerly aspects with less frequent fire would have tended to support more complexly structured closed-canopy, multilayered stands of shade-tolerant and fire-intolerant trees (Agee 1998, Hessburg et al. 2016, Tepley et al. 2013).

For the eastern Cascades of Washington, Agee (2003) used historical fire-return intervals and simple mathematical models to estimate range of variation in forest structure classes. This region would contain both the frequent mixed-severity and very frequent low-severity regimes (fig. 3-4). The proportion of medium to large trees (>15 in [40 cm]) in dry to moist forest vegetation types (ponderosa pine, Douglas-fir, grand fir warm and cool mesic), regardless of canopy cover, ranged from 38 to 64 percent of the landscape. Agee (2003) found that late-successional forest (containing shade-tolerant tree species and multilayered canopies) was not present in ponderosa pine, warm-dry and cool dry Douglas-fir, or warm grand fir forest types, and present in about 10 to 16 percent in the “cool-mesic grand-fir” type. The amount of early-successional vegetation in these potential vegetation types in this region ranged from 6 to 15 percent (Agee 2003, Hessburg et al. 2000b). Hessburg et al. (2007) used aerial photography from the 1930s to 1940s to estimate that old, multistoried forests ranged from less than 5 percent to about 20 percent or more of dry coniferous forest watersheds, while the area of multistoried late-successional forest ranged from 17 to 68 percent in mixed-severity-regime forests. The estimates of forest conditions from this period would have been affected by logging, fire exclusion, and fires associated with Euro-American settlement around the turn of the century (e.g., the widespread fires of 1910), but Hessburg et al. (2017) used methods that reduced the impact of these anthropogenic effects.

Several historical studies have estimated pre-Euro-American settlement amounts of older forest and other successional stages for the eastern Cascades of Oregon (Andrews and Cowlin 1940 as cited in Davis et al. 2015; Baker 2015b; Hagmann et al. 2013, 2014; Kennedy and Wimberly 2009). The estimates of the percentage of forests of the eastern Oregon Cascades (across all lands in the ponderosa pine to moist mixed-conifer potential vegetation types) with large old trees are 35 percent (Kennedy and Wimberly 2009); 76 percent (Baker 2015b); 42 to 76 percent (Hagman et al. 2013); and 91 percent (Hagmann et al. 2014). LANDFIRE estimated that "late development" (both open and closed-canopy classes) covered 55 to 65 percent of the dry ponderosa pine and mixed-conifer forest environments that occur in the eastern Cascades of Oregon and Washington. Using empirical reconstructions from early 20th century aerial photos from this area, Hessburg et al. (1999a, 2000) showed that more than 40 percent of the eastern Oregon Cascades area contained patches with medium and large-size old trees in the overstory. They also noted that given logging in the ponderosa zone during the early 20th century, which they documented via photointerpretation, that amount may have been 50 percent larger, i.e., 60 percent of the area with medium- and large-size trees in the overstory. The much lower numbers from the Kennedy and Wimberly (2009) modeling study may be a result of the assumptions about the frequency and severity of fire in this region, which is not well-known given the lack of fire history studies that were available at that time (Baker 2015b). The estimates of historical older forest structure among these studies are not strictly comparable because of use of different definitions, geographies, potential vegetation types, disturbance regimes, and methods and data sources. It is especially difficult to compare different studies because of the environmental heterogeneity of the region, including strong precipitation and topographic gradients. Also, some moist mixed-conifer forests in the eastern Cascades of Washington have high fire frequencies (<25 years), which can be similar to that of drier ponderosa pine forests (Wright and Agee 2004); that relationship would mean that the moist mixed-conifer potential vegetation type is not necessarily a good indicator of regimes with longer frequencies or higher fire severity. The frequent and very frequent fire regimes are spatially intermingled in many landscapes and are difficult to separate.

Most estimates of older forest described above are from landscape simulation studies and do not take into account canopy cover or forest density, with the exception of Hessburg et al. (2007), which is limited to the early and mid 20th century. The historical percentage of the eastern Cascades in denser older forest (e.g., areas that have not had fire for many decades, including areas that could potentially support northern spotted owls) has been estimated to be 9 percent (Kennedy and Wimberly 2009) and as much as 22 to 39 percent by Baker (2015b). Hagmann et al. (2014) estimated that areas of higher density forest (>185 trees per acre—"group 1") and grand fir trees were historically rare in dry and moist mixed-conifer forests of the northern eastern Oregon Cascades, which would have included mixed- and low-severity fire regimes. Perry et al. (2004) also found relatively little grand-fir in the central Oregon Cascades.

The fire regimes and forest dynamics of frequent mixed-severity regime forests in California have been described by Taylor and Halpern (1991), Taylor (1993, 2000), Taylor and Solem (2001), Bekker and Taylor (2001, 2010), and Skinner (2003) and summarized by Skinner and Taylor (2006). Although no direct estimates of HRV have been made, these studies show that fire-return intervals tend to be at the low end of the range for this regime. The frequent mixed-severity fire regime is characteristic of the upper montane forests of red fir/noble fir, western white pine, mountain hemlock, and lodgepole pine. These forests are typified by precipitation being predominantly snow with snowpacks often lasting into early summer contributing to a relatively short, yet mostly dry, fire season (Skinner and Taylor 2006). Higher productivity (e.g., more fuels) and greater sensitivity of the species to fire compared to the very frequent, low-severity fire regime may help drive occurrence of moderately large patches (hundreds to thousands of acres) of high-severity fire despite the high frequency of fire.

***Very frequent, low-severity fire regimes*—**The very frequent fire (<25-year interval), low-severity regime dry forests often occur in association with the forests of the infrequent, low-severity regime especially in the eastern Cascades and Klamath provinces in areas of topographic variability and strong climatic gradients (fig. 3-4). This fire regime would have been common in ponderosa pine, dry to moist mixed-conifer and oak woodlands vegetation types. The successional dynamics, structure, and composition of low-severity regime forests can be simplified into two pathways that lead to very different major types of old growth (Stine et al. 2014). In the first, a dominant low- or mixed-severity fire-dependent pathway maintained old-growth conditions (primarily old live and dead trees) in a shifting mosaic of open and moderately closed canopy patches (e.g., 20 to 60 percent canopy cover) (figs. 3-18 and 3-19).

A second, historically much less common pathway occurred where local climate and topoedaphic circumstances (e.g., rocky ridges) reduced wildfire frequency and led to development of patches of denser (60 to 90 percent canopy cover), multistory old-growth with shade-tolerant species (Agee 1993; Camp et al. 1997; Hessburg et al. 1999a, 1999b, 2000, 2007; Merschel et al. 2014; Sensenig et al. 2013). Levels of large standing and dead down wood would be much lower than in old-growth forest types in the other fire regimes (see Youngblood et al. 2004 for density estimates), owing to lower densities of large trees and frequent consumption of down wood (Safford and Stevens 2016, Skinner 2002). Despite the lower densities relative to denser old growth, large standing dead trees would have been present throughout though they would have been patchy and not found on every acre (Stephens and Fulé 2005). The pattern of seral stages within the forest matrix would be a fine-meso-scale mosaic of patches (<1 ac [<0.40 ha] to thousands of acres). The dominant pathway was maintained by high- to moderate-frequency, low- to mixed-severity fire (Baker 2012, Hessburg et al. 2007); scattered small- to medium-size patches with canopy tree mortality (individuals or small- to medium-size clumps) would have been present with medium and large fire-tolerant trees occurring in low to locally moderate densities (Churchill et al. 2013, Larson and Churchill 2012). For old-growth ponderosa pine in Oregon and California, canopy trees were not uniformly distributed and tended to occur in either clumps of up to 80 ft (24 m) in diameter (Youngblood et al. 2004) (figs. 3-17 and 3-18). These forests are sometimes characterized as being open, low-density forests, "park-like" stands (Agee 1993, Hessburg et al. 2015, Sensenig et al. 2013, Youngblood et al. 2004) (fig. 3-1). Bark beetles, which attack trees in small groups, may have interacted with fire in these forests to promote patchy regeneration of ponderosa pine. This would occur where beetle-killed patches of dead trees had accumulations of small branches and coarse woody debris that burned with high severity, killing rhizomatous grasses and promoting patchy regeneration of ponderosa pine regeneration in ash of the burned logs and sterilized mineral soil (Agee 1993).

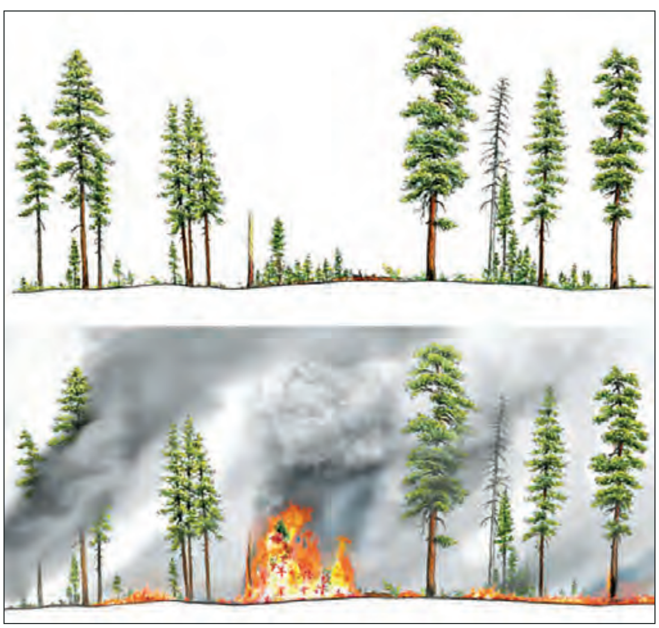


Figure 3-19—Hypothetical structural profile and typical historical fire behavior in a ponderosa pine forest of the eastern Cascade Range of Washington. From Van Pelt (2008).

The second successional pathway would lead to denser patches of pine and Douglas-fir or true fir regeneration, as mentioned above, often associated with variation in topography (steeper slopes and higher elevation), microclimate, and fire frequency that allowed trees to develop on moister microsites associated with north-facing lower slopes, concave areas, riparian areas, and wetter soils (Camp et al. 1997, Merschel et al. 2014). However, Baker (2012) did not find that concentrations of fir were associated with aspect or topography in an analysis of General Land Office (GLO) survey data from the eastern Oregon Cascades. Following low- to moderate-severity fire on these more moist sites, white fir or grand fir could establish in the understory and occasionally reach the canopy where bole diameters and bark thickness was sufficient to withstand surface fires. On some productive sites (e.g., benches), old-growth grand-fir or white-fir patches developed even while experiencing frequent surface fires that burned in from adjacent drier ponderosa pine and grassland sites (Hessburg et al. 1999, Taylor and Skinner 2003). The relative amount of open and denser older forests may have varied over time with climate. Many studies across the area support this characterization of forest structure and dynamics for this type in some portions of the region (Bisson et al. 2003; Hann et al. 1997; Hessburg et al. 1999a, 1999b, 1999c, 2000, 2003, 2005; Keane et al. 2002, 2009; Lehmkuhl et al. 1994). With fire exclusion, the dense late-successional and old-growth pathway (either with ponderosa pine, Douglas-fir, or *Abies* spp.) has become dominant (fig. 3-20). White fir and grand fir have widely expanded out of their historical environments and fire refugia into sites that were historically dominated by ponderosa pine (or sugar pine (*Pinus lambertiana*) in California) or pine mixes with Douglas-fir (Camp et al. 1997; Hagmann et al. 2017; Merschel et al. 2014; Taylor and Skinner 1998, 2003), or grassy woodlands often originally dominated by hardwoods (Skinner et al., in press). This expansion of shade-tolerant trees (which is discussed more below) has been widespread across a range of topographic settings and forest types, including drier mixed-conifer and ponderosa pine types (Hagmann et al. 2014; Hessburg et al. 1999a, 1999b, 2000a, 2003, 2005, 2015, 2016; Merschel et al. 2014; Stine et al. 2014).

Thomas Spies

Figure 3-20—Old-growth ponderosa pine in the eastern Cascade Range of Oregon with understory of grand fir that established in the early 1900s after fire exclusion.

***Woodlands, shrublands, and grasslands*—**A significant portion of some of the dry forest landscapes was occupied by patches of semistable, woodlands, shrublands, and grasslands (Hessburg et al. 2007) (figs. 3-21 and 3-22). These included oak, juniper, and pine woodlands that did not succeed to denser forest as a result of climate, soils, and frequent fire (Agee 1993, Franklin and Dyrness 1973, Hessburg and Agee 2003, Skinner et al. 2006). In many cases, a frequent grass- or shrub-driven fire cycle was responsible for maintaining low tree cover (Hessburg et al. 2016). These areas were so dominated by grasses over a geologically long timeframe that mollisols can be seen today as the characteristic soil type. Open stands and oak dominance were maintained by American Indians in many areas using fire to promote desired resources associated with such habitats (Anderson 2005, Skinner et al. 2006) (chapter 11). Figures 3-21 and 3-22 illustrate these landscapes, and although large fires in the early 1900s would have affected these patterns, many of the large fires would have occurred in grasslands and shrublands (that were historically maintained by frequent fire) as evidenced by the lack of snags and dead trees in the large nonforest patches in these photos. Interestingly, the concept of old growth (in a general sense of a vegetation type that persisted for very

Reino R. Sarlin

John F. Marshall

Figure 3-21—Photographs of the Mission Peak area on the Okanogan-Wenatchee National Forest in 1934 and 2010. The 1934 image illustrates the mosaic of closed forests, open forests, woodlands, and grasslands that would have characterized many landscapes with low- and mixed-severity fire regimes. Open areas typically lack snags that would be indicative of recent high-severity fire in forests. Landscapes in 1934 may have been influenced by settlement fires, logging, and fire exclusion.

long periods under natural processes) has also recently been applied to these nonforest vegetation types (Veldman et al. 2015) because they have distinct conservation values that arise as a result of being "ancient"[10] ecosystems with characteristic biotic and soil properties that have been lost owing to changes in fire regimes, grazing, and other land use changes.

[10] Grasslands have existed for millions of years, and some grasslands may take 100 to as much as 1,000 years to develop; and clonal grasses can live for over 500 years.

Oak woodlands dominated by California black oak (*Quercus kelloggi*) and Oregon white oak (*Q. garryana*) and other hardwoods were maintained in an open old-growth state by very frequent low-severity fire (Agee 1993, Cocking et al. 2012, Franklin and Dyrness 1973). These species can form large, old trees with high value because they produce mast or berries, as well as large cavities for wildlife. They often support a high diversity of understory plants, fungi, and associated wildlife of particular importance to tribes (see chapter 11). However, a lack of fire in many of these areas has permitted conifer

Klamath National Forest

Figure 3-22—View from Eddy Gulch Lookout in the Salmon River watershed of the Klamath National Forest in 1935 (top) and 1992. The 1935 image illustrates the mosaic of closed forests, open forests, shrub fields, woodlands, and grasslands that would have characterized many landscapes with low- and mixed-severity fire regimes. Open areas typically lack snags that would be indicative of high-severity fire in forests. Landscapes in 1935 may have been influenced by settlement fires, logging, and fire exclusion.

trees such as Douglas-fir to increase shade, accumulate conifer litter, and form ladder fuels, which consequently, render mature hardwoods more vulnerable to top-kill from fires. These trends are particularly evident in riparian forests of southwestern Oregon, where the shift in fire regime has led to reductions in both hardwoods and large trees (Messier et al. 2012).

Role of shrubs and hardwoods in Klamath-Siskiyou forest dynamics—The successional dynamics of low- and mixed-severity regime forests in the Klamath-Siskiyou region of Oregon and California are distinctive for the prominent role of shrubs and hardwoods in the vegetation community and their interaction with both fire and forest succession. In the northern and western part of this region, mixed-severity fire can lead to patchy old growth with tanoak (*Notholithocarpus densiflorus*) understories (as small trees) intermixed with Douglas-fir that either survives the lower intensity fire as a large tree or regenerates in patches of high-severity fire that kill the tanoak (Agee 1993). In other areas of this region, and extending into the southern Cascades and northern Sierra Nevada, dense stands of the shrub form of tanoak (*N. densiflorus* var. *echinoides*) can be found. These stands often do not burn well under less-than-severe conditions but will strongly sprout following severe fires even though the acorns are killed by fire.

Throughout the Klamath-Siskiyou region, shrub species resprout after fire and are also stimulated to germinate from seeds stored for long periods in soil seed banks following fires (Knapp et al. 2012b) with areas of higher severity fire leading to greater density of shrubs (Crotteau et al. 2013). Hardwoods (especially oaks, tanoak, and madrone (*Arbutus menziesii*) mixed in with the often more dominant conifers are often able to resprout following high-severity fires that kill the conifers (Cocking et al. 2012, 2014; Skinner et al. 2006). This adaptation facilitates the reestablishment of trees in severely burned forest areas at an early-seral stage. For conifer forests to again occupy these areas requires sufficient time between severe burns to allow conifer trees to reestablish and mature. Where severely burned areas are reburned before such conditions are achieved, shrubfields and hardwoods are likely to be maintained and can become a more permanent part of the landscape (Cocking et al. 2014, Coppoletta et al. 2015, Lauvaux et al. 2016). Several recent studies have documented how severely burned areas that are reburned within a few decades are likely to again burn severely (Coppoletta et al. 2015; Odion et al. 2004; Perry et al. 2011; Thompson and Spies 2010; Thompson et al. 2007, 2011). In other cases, hardwoods in mixed-wood forests may play an important role in protecting some of the coniferous forest cover from severe fire effects via their foliar moisture content (Agee 2002, Perry 1988, Perry et al. 2011, Raymond and Peterson 2005, Skinner 2006, Skinner and Chang 1996). Likewise, depending upon the forest community type, hardwood trees and shrubs may in fact facilitate conifer succession via mycorrhizal fungi shared by both hardwood and coniferous species (Horton et al. 1999).

In complex topography, such as that found in the Klamath-Siskiyou area, it is unlikely that disturbance regimes and seral stages randomly moved about the landscape. Rather, particular parts of the landscape were more prone to severe burns. Upper thirds of slopes, and especially south- and west-facing slopes, were prone to repeated severe burning that perpetuated shrub dominance (Jimerson and Jones 2003, Taylor and Skinner 1998, Weatherspoon and Skinner 1995). Shrubfields may be places where forests burned severely or places where fires have long maintained shrubfields (Baker 2012, 2014; Lauvaux et al. 2016; Nagel and Taylor 2005). In the latter case, these were not places that periodically contributed large wood and snags but reburns of shrubs, grasses, and occasional small conifers.

Alternative views of disturbance regimes of the dry forests—Some have argued that most ponderosa pine and mixed-conifer forests in the Western United States, including the area of the NWFP that we define as having had a very frequent, low-severity regime, have been mischaracterized. They contend that these forests are better characterized instead as having a more variable-severity fire regime, with significant components of mixed and high-severity fire as well (Baker 2012, Odion et al. 2014, Williams and Baker 2012). Hessburg et al. (2007) has also been cited in support of this argument (Baker 2012); however, the results of Hessburg et al. (2007) do not fully support the claims of Baker (2012); there are some key differences. The classification of high-severity fire from aerial photos in Hessburg et al. (2007) included areas with small trees, grasslands, shrublands, and sparse woodlands. These nonforest areas would have typically burned with high-severity given the low stature of their vegetation driven by a predominantly grass-fire cycle. When Hessburg et al. (2007) restricted their analysis to forest cover types, they found that less than 20 percent of each cover type was consistently affected by high-severity fires (fig. 3-23). For example, the dominating ponderosa pine and Douglas-fir cover types exhibited 13 and 18 percent high-severity fires across the study area, respectively. Similarly, when they restricted their analyses to forest structural classes (fig. 3-24), they found that no structural class experienced more than 17 percent high-severity fire across the study area. Furthermore, Baker (2012) uses Hessburg et al. 2007 to support his claim that "substantial" areas of high-severity fire occurred in ponderosa pine and dry mixed-conifer, but he cites Hessburg et al. (2007) data from Ecological Subregion 5 (ESR5), which is not a dry forest environment, but is classified as "moist and cold forest" type, with lesser amounts of dry forests. Hessburg et al. (2007) found considerable evidence of high-severity fire in their regional analysis of dry pine and mixed-conifer forest landscapes, but much of the high-severity fire was associated with grasslands and shrublands that where common in these landscapes in the past and were intermingled with forested patches. These vegetation types would typically burn with high severity. Figure 3-23 shows the proportion of forest structural classes affected by low-, mixed-, and high-severity fire in three ecoregions.

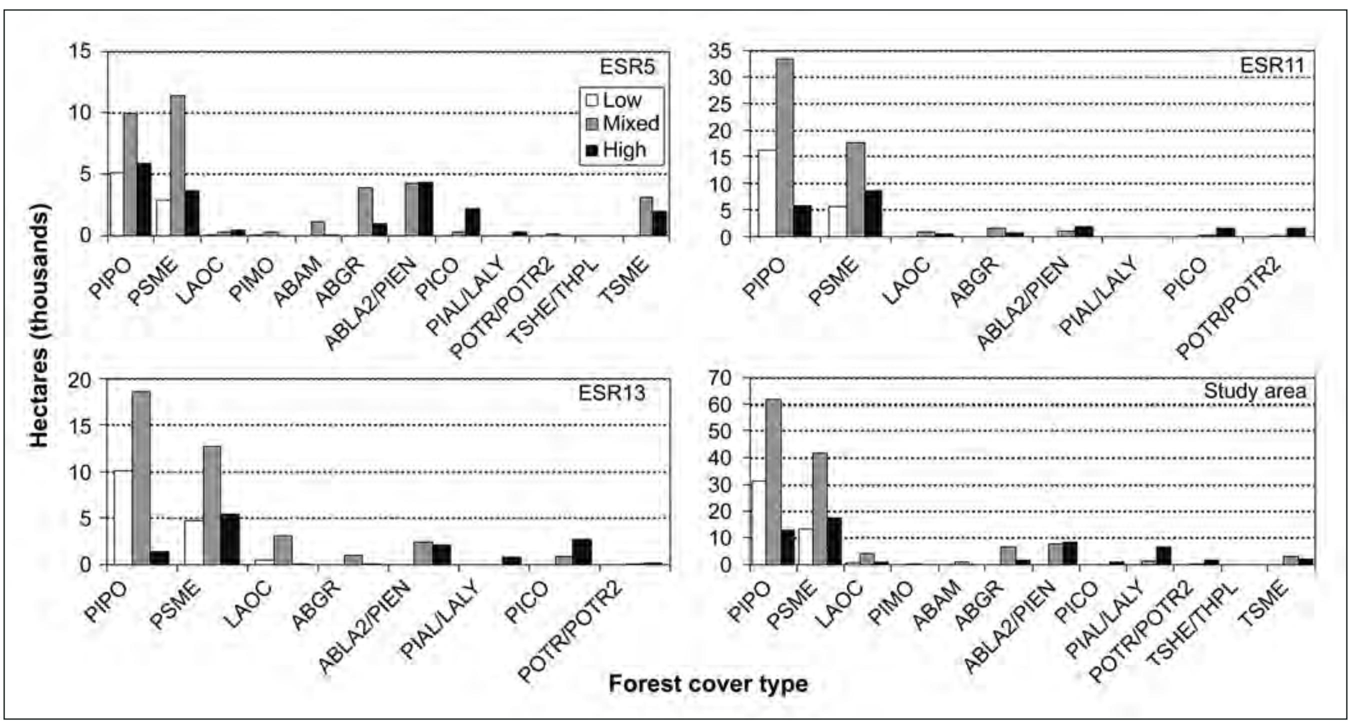


Figure 3-23—The proportions of premanagement-era total forest area (hectares) by forest cover type in low-, mixed-, and high-severity fire (corresponding with percentage of canopy mortality values of ≤20 percent, 20.1 to 69.9 percent, and ≥70 percent, respectively) of Ecological Subregions (ESRs) 5, 11, and 13. Cover type abbreviations are TSHE/THPL = western hemlock/western redcedar; PIMO = western white pine; POTR/POTR2 = *Populus* and *Salix* spp.; LAOC = western larch; TSME = mountain hemlock; PIAL/LALY = whitebark pine/subalpine larch; ABAM = Pacific silver fir; ABGR = grand fir; PICO = lodgepole pine; ABLA2/PIEN = subalpine fir/Engelmann spruce; PSME = Douglas-fir; PIPO = ponderosa pine. From Hessburg et al. (2007).

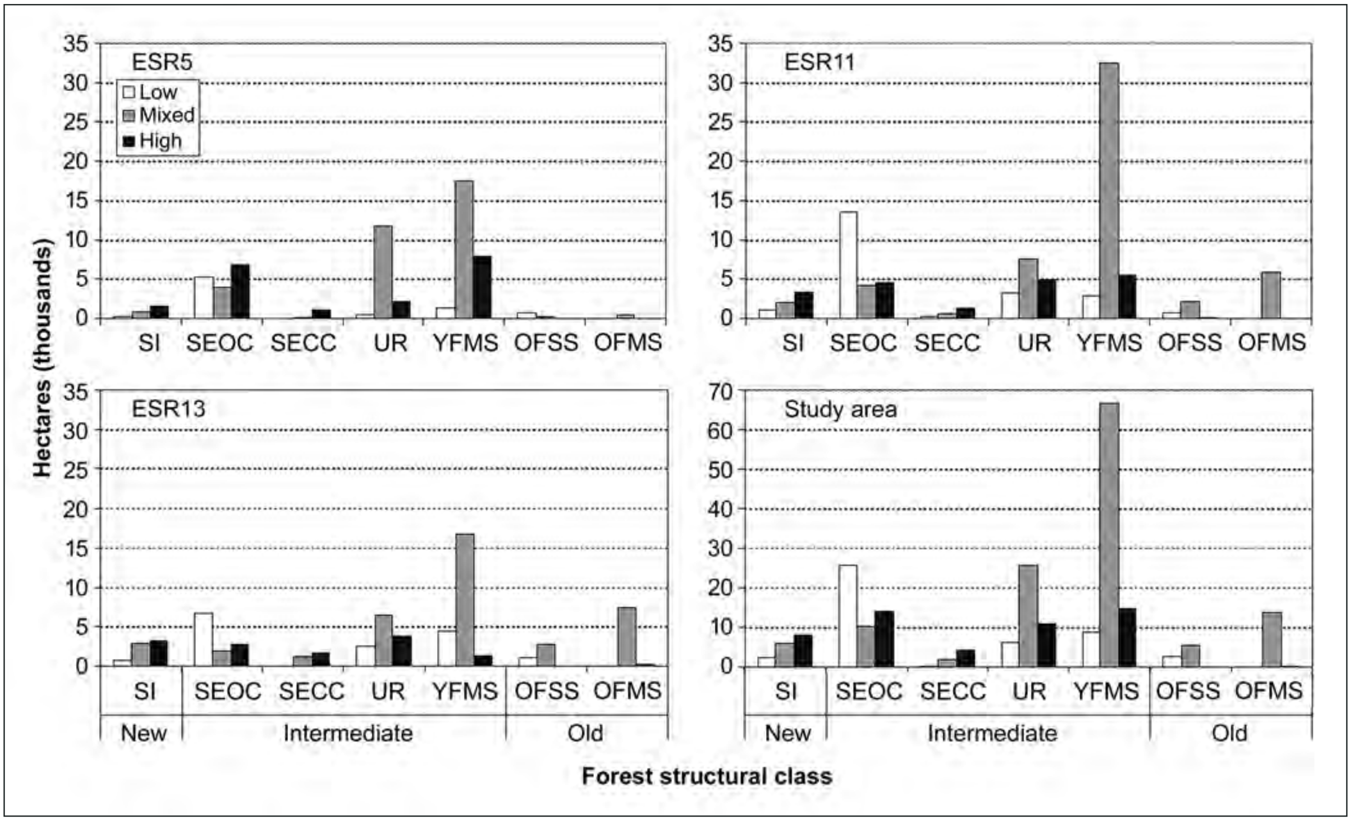


Figure 3-24—The proportions of the premanagement-era dry forest area (hectares) by forest structural class in low-, mixed-, and high-severity fire (corresponding with percentage of canopy mortality values of ≤20 percent, 20.1 to 69.9 percent, and ≥70 percent, respectively) of Ecological Subregions (ESRs) 5, 11, and 13. Structural class abbreviations are: SI = stand initiation, SEOC = open canopy stem exclusion, SECC = closed-canopy stem exclusion, UR = understory reinitiation, YFMS = young multistory forest, OFMS = old multistory forest, OFSS = old single-story forest. New, intermediate, and old designations are used to group structural classes into broad age groups. From Hessburg et al. (2007).

Williams and Baker (2012) and Baker (2012) use GLO survey data from the 1880s and 1890s on live tree sizes and species to infer historical stand densities and fire regimes from central Oregon. The evidence and methods used to support the claims that the historical role of high-severity fire in low-severity regimes has been underestimated has been the subject of several published critiques and counter arguments by both sides of the debate. In one critique, Fulé et al. (2013) point out three problems with using GLO survey data to infer disturbance history (e.g., Baker 2012): (1) the use of tree size distributions to reconstruct past fire severity and extent is not supported by empirical age-size relationships nor by local disturbance history studies; (2) the fire-severity classification based on the survey data is qualitatively and quantitatively different from most modern classification schemes, limiting the validity of comparisons to history; (3) their finding of ”surprising” heterogeneity within these stands does not actually differ substantially from other previous studies (some from ponderosa pine forests outside the NWFP area but still potentially relevant to dry forests in the NWFP area) that found areas and clumps of relatively high density in ponderosa pine and mixed-conifer forests (e.g., Brown and Cook 2006, Youngblood et al. 2004) (fig. 3-25). For example, the lower left corner (66 by 66 ft [20 by 20 m]) of the old-growth plot that Youngblood analyzed had 16 trees (equivalent to a density of upper canopy trees of about 160 trees per acre), while the upper right corner had one tree (an acre-scale density of 10 trees per acre).

Williams and Baker (2014) responded to that critique of Fulé et al. (2013) by arguing that the concerns are unfounded and based on misquoting their 2012 paper.

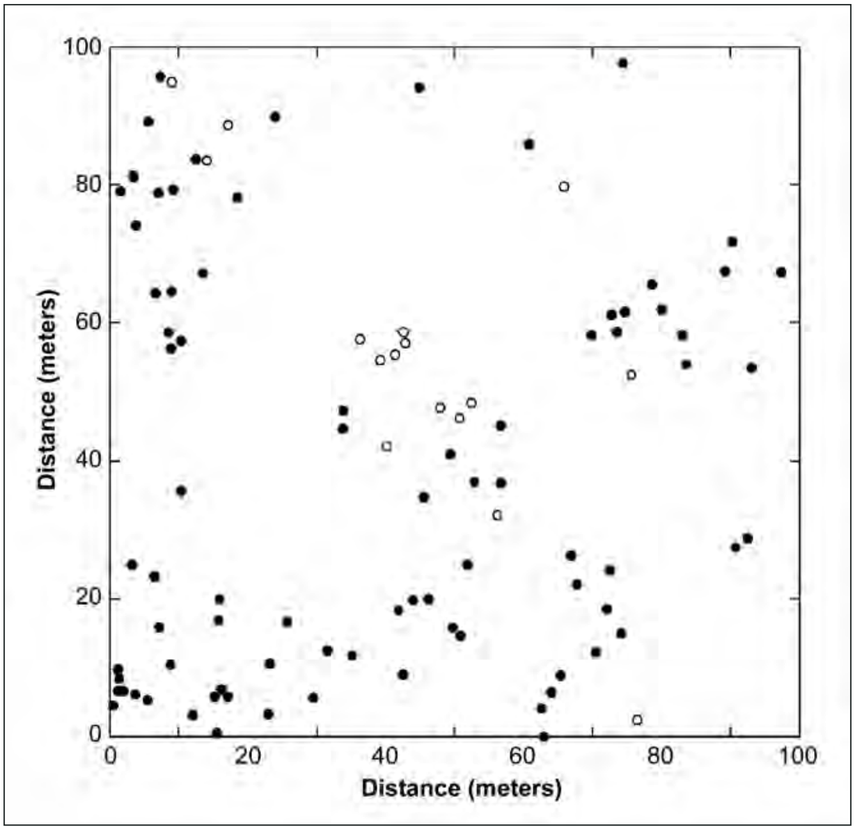


Figure 3-25—Spatial patterns of live (filled circle) and dead trees (open circle) in the upper canopy of an old-growth ponderosa pine forest in central Oregon. From Youngblood et al. (2004).

Williams and Baker (2012) used tree density and relative proportions of small and large trees to classify GLO data areas as either low- or high-severity fire. According to Baker (2012), 26 percent of pine and dry mixed-conifer forests in the eastern Oregon Cascades showed evidence of high-severity fire based in part on tree density. The findings of Baker (2012) depend on many assumptions, the most important being that the method for calculating tree density from GLO survey data (Williams and Baker 2011) produces an unbiased estimate. However, a recent paper by Levine et al. (2017) indicates that the method (Williams and Baker 2011) used by Baker (2012) overestimates tree density by a factor of 1.2 to 3.8. This finding could help explain why the estimates of historical tree densities that Baker has reported (mean of 100 trees per acre) are considerably higher than those reported from other studies, e.g., 62 trees per acre (Munger 1917) or 26 to 32 trees per acre (Hagmann et al. 2013, 2014). Other assumptions made by Baker (2012) could explain the higher densities relative to other studies including the assumption that his survey points represent dry environments and not wetter mixed-conifer sites that often occur in the eastern Cascades where topographic and precipitation gradients are strong, and produce high variability in forest structure, composition, and dynamics (Merschel et al. 2014).

Odion et al. (2014) have also argued for the occurrence of more high-severity fire in ponderosa pine and mixed-conifer forests of western North America using inferences from analysis of current tree-age data from unmanaged areas collected through the U.S. Forest Service Inventory and Analysis (FIA) program (Odion et al. 2014). Age data were analyzed and it was assumed that if stand-age diversity was low, then fire effects represented low- or mixed-severity regimes; if stand-age diversity was high, then the forest came from a mixed-severity regime with significant areas of high severity. However, a critique by FIA and other scientists argues that the assumptions, analysis and conclusions of this paper are invalid (Stevens et al. 2016). First, the FIA stand-age estimator underestimates the age range of trees in plots, and it routinely undersamples old trees, which would be relatively common in forests subject to low-severity fire regimes (see Merschel et al. 2014). Forests with a low-severity fire regime also continuously recruit new cohorts of regeneration, which would be poorly reflected in the stand-age estimator. Second, recruitment events are not necessarily related to high-severity fire occurrence as we have described above. Odion et al. (2016) responded to Stevens et al. (2016) and identified areas of "agreement and disagreement." Areas of agreement include high-severity fire was a component of forests in low-severity fire regimes, that tree recruitment occurs in the absence of fire, and FIA stand data may provide evidence of past high-severity fire. Areas of continued disagreement according to Odion et al. (2016) include deciding what threshold to use for mortality from high-severity fire, plot sizes needed to detect high-severity fire, use of diameter-age relationships for reconstructing basal area, and historical data sources that document high-severity fire in patches larger than 2,500 ac (~1000 ha). We disagree that their historical sources present many examples that document the occurrence of large patches of high-severity fire in forests with low-severity regimes. Historical maps we found from the early 1900s document three patches of high-severity fire larger than 2,500 ac (~1000 ha) in Oregon and Washington that account for 1 percent of the area of this regime (fig. 3-6). In addition, the so called large patch of high-severity fire in the "eastern Cascades" of Oregon that is cited in Dellasala and Hanson (2015: 30–31) from mapping of Leiberg (1903) as evidence of a 35,000-ac (~14 200-ha) patch of high-severity fire in ponderosa pine forests actually comes from a township in the western Cascades in an area of mixed-conifer forest, containing red fir and noble fir. This township and the boundaries of this fire straddle the infrequent high-severity regime and moderately frequent to somewhat infrequent mixed-severity regimes of our regime map (fig. 3-6).

These concerns about interpretation of forest history data notwithstanding, there is essentially no disagreement that very frequent, low-severity regime forests (e.g., ponderosa pine and mixed conifer) included occasional small- to medium-size (e.g., tens to hundreds of acres) patches of high-severity fire. In addition, the broader landscapes would have contained grasslands or shrublands maintained by high-severity fire (relative to that life form) (e.g., see Hessburg et al. 2007, Perry et al. 2011). Given that many larger landscapes (including forested areas and nonforest areas) are often a mosaic of environments that support both low- and high-severity fires, it would not be surprising to find landscapes where the amount of high-severity fire to forest and nonforest vegetation exceeded 20 percent (e.g., see historical landscape data in Hessburg et al. 1999a, 2000, 2007). However, over smaller areas or areas with less topographic variability and within environments that predominantly supported forests, the amount of high-severity fire in low-severity regime forests would be expected to be lower than 20 percent. For example, Hagmann et al. (2014) found that only 9 percent of forest survey transects in 123,500 ac (~50 000 ha) of mixed-conifer landscape in eastern Oregon showed potential evidence of high-severity fire based on absence of large trees.

In summary, we believe the preponderance of evidence supports the view that large patches of high-severity fire were not a major component of dry forests with very high frequency, low-severity forest fire regimes. However, they were an important component of the frequent, mixed-severity regime. Remember that these regimes exist along a continuum of environments that differ across regions and landscapes. This means that landscapes often do not fit neatly into one regime or another. These alternative views of the role of high-severity fire in low-severity fire regimes highlights that generalizations either for or against management interventions across a wide range of forest types and environments should be made with caution. Different definitions of severity, scales of observation, and types of evidence (e.g., maps, surveys, aerial photos, tree age and size distributions, etc.) make it difficult to compare across studies because these factors influence the scope of inferences that can be made. In addition, subregional and landscape-scale variation in ecosystems and interactions among climate, topography, soils, vegetation, and disturbance agents make it difficult to accurately extrapolate over to large areas. Efforts to infer process (e.g., disturbance history) from pattern (e.g., ages, sizes, or densities of trees, and patches of trees in maps and aerial photos), as is done in many of the fire history studies we cite, can also be fraught with some degree of uncertainty because similar patterns in biotic communities can arise from different processes (Cale et al. 1989). For example, much of the open forest reported by Baker could have been made up of aggrading meadows and shrublands that were much more common during the early 20th century (Hessburg et al. 2005, 2007). A lack of information on the presence of snags and dead wood limits any inference on fire severity in forests from studies based only on live trees (Reilly and Spies 2015, Reilly et al. 2017). Uncertainties about fire history are unlikely to be resolved given the limits of historical information (especially prior to Euro-American colonization) and the heterogeneity of ecosystems. In the end, the details of historical regimes (e.g., the level of high-severity fire in the past) may not be as important as what society wants and can have for their forests given changing climate, succession, and fire behavior (see chapter 12).

Effects of Fire Exclusion

Forest structure and composition—

***Dry forests*—**There is less debate in the literature about the effects of fire exclusion on forest structure and composition in dry forests where fire was historically frequent. Nationally, over 95 to 98 percent of all wildfires are suppressed while small during initial attack (i.e., 2 to 5 percent escape initial attack) with suppression in the NWFP area especially common in dry forests (fig. 3-10, table 3-3). Many of these fire starts would have resulted in larger fires that would have altered forest structure and fuel beds and created or maintained early- and mid-successional vegetation over much of the region in the ensuing century.

The recent trends in fire extent and severity in the NWFP area (chapter 2) suggest that fire has generally been less common in recent decades than would be expected under the historical fire regimes (Reilly et al. 2007) (table 3-4), especially given the occurrence of the warmest decade (~1995–2005) since the early 1900s (Abatzoglou et al. 2014) and the historical link between fire and temperature, and drought. The amount of fire (fire rotation) in the frequent and very frequent regimes (117 to 182 years for federal lands) has been considerably less than the historical range for these two dry forest regime classes (5 to 50 years) (table 3-4). For example, in the very frequent regime, most areas would have burned at least once (e.g., a fire rotation of less than 25 years), if not more, during 30 years, the length of the recent satellite record.

Forests have responded to the lack of fire in the two dry forest fire regimes through increases in density and changes in composition. It is well documented that the structure and composition of these forests have changed across the Western United States since Euro-American settlement (Hann et al. 1997; Hessburg and Agee 2003; Hessburg et al. 2005, 1999a, 1999b, 1999c, 2000; Lehmkuhl et al. 1994) as a result of fire exclusion. For example, forests are now typically several times denser in most locations than under native fire regimes (Camp 1999; Dolph et al. 1995; Hagmann et al. 2013, 2014; Merschel et al. 2014; Perry et al.

Table 3-3—Number of lightning fire starts[a] between 1992 and 2013 in summer months (June–September) on federal forest lands in the Northwest Forest Plan area[b]

Regime	Total fire starts	Number per 25,000 ac (10 117 ha)
Infrequent, high severity	4,271	12.2
Moderately frequent, mixed severity	2,350	13.4
Frequent, mixed severity	2,511	15.2
Very frequent, low severity	4,240	17.4

[a] Most of these would have been suppressed by fire crews.
[b] Sources of data: Bureau of Land Management Wildland Fire Management Information system; U.S. Fish and Wildlife Wildland Fire Information System; U.S. Forest Service fire statistics.

Table 3-4—Comparison of historical fire frequencies and rotations (in years) with recent (1985–2010) fire rotation estimates from satellite remote sensing for the Northwest Forest Plan area by fire regime class[a]

Historical regime class and fire frequencies in years	Range of frequencies from historical studies, all fires (number of studies)	Range of estimates of historical rotations, all fires (number of studies)	Recent rotations (all severities) for USFS lands/all ownerships	Recent rotation (high severity) for USFS lands/ all ownerships	Recent frequency (low severity) for USFS lands/all ownerships
Infrequent, high severity (200 to 1,000 years)	No data	296–834 (5)	758/1,525	1,628/3,326	3,056/6,069
Moderately frequent, mixed severity (50 to 200 years)	40–246 (19)	78–271 (6)	582/1,055	2,398/4,530	1,321/2,342
Frequent, mixed severity (15 to 50 years)	21–27 (2)	No data	110/276	333/851	305/761
Very frequent, low severity (5 to 25 years)	3– 36 (18)	11–64 (4)	111/143	690/852	218/286

[a] See appendix 3 for fire history data. Recent data from Reilly et al. (2017). USFS = U.S. Forest Service.

2004; Reilly and Spies 2015; Ritchie et al. 2008; Stephens et al. 2015; Youngblood et al. 2004), and composition has shifted toward shade-tolerant species. Baker (2012) did not agree with this characterization and described these forests of the late 1800s as historically "generally dense." However, the finding that his method overestimates tree density by 20 to 380 percent (Levine et al. 2017) suggests that forests were not generally dense as he claims, and data may be coming from a period in which shifts from a more frequent fire regime had already occurred as a result of various effects of Euro-American colonization (Fry and Stephens 2006, Norman and Taylor 2005, Skinner et al. 2009), including the loss of burning[11] by American Indians. Even if the overestimates of the Baker (2012) method are at the low end of the range of bias found by Levine et al. (2017), they are still lower than the least dense areas found in contemporary forests (Merschel et al. 2014, Reilly and Spies 2015). Baker (2012) estimated that the interquantile range (25th to 75th) for density in mixed conifer was 69 to 142 trees per ac (170 to 352 trees per ha), whereas the interquantile range in current forests was 298 to 586 trees per acre (736 to 1,447 trees per hectare) an increase of 67 to 75 percent. Consequently, the 2012 Baker paper cannot be used as evidence that forest density has not substantially increased since the 1900s—only that the increase may not be as large as some studies indicate.

A consequence of succession in these forests is that dense understories of shade-tolerant species can shade out pine regeneration and eventually provide abundant seed sources that compete with pine regeneration in lower fire severity postfire environments. Restoring the dominance of large fire-tolerant tree species in these forests is a key component of restoration strategies (Hessburg et al. 2016). The accumulated seed source of shade-tolerant species in these landscapes and large-landscape inertia has probably altered the successional probabilities following fire disturbances toward shade-tolerant pathways as Stine et al. (2014: 140) indicates:

[11] Note that American Indians were marshalled onto reservations by 1850, and with this came the loss of intentional burning that occurred near seasonal encampments and customary food production and gathering places (Stewart 2002).

> Landscapes exhibit varying degrees of inertia. The degree of change over the 20th century in forest structure, tree species composition, and disturbance regimes has given landscapes an inertia (which can be thought of also as ecological momentum or resistance to change) that will be difficult to alter through restoration-based management. For example, field observations suggest that after recent wildfires, instead of regenerating to ponderosa pine or western larch, some areas now quickly regenerate to Douglas-fir and white, grand, or subalpine fir, or lodgepole pine, despite intentional efforts (which often fail unless done well) to reestablish ponderosa pine or larch. The presence of abundant seed from shade-tolerant tree species (e.g., firs) provides this inertia. Likewise, high contagion of surface and canopy fuels creates large homogeneous patches that reinforce the occurrence of a higher than normal number of large and very large fires, and higher than normal fire severity.

This landscape-scale successional trend may be locally disrupted by large disturbances, but if the rate of disturbance is not high enough, or the disturbance does not kill the shade-tolerant species over large areas, the trend is likely to continue unless climatic changes alter the disturbance regime and the growth or survivorship of tree species.

***Moist forests*—**Fire suppression also appears to be having an effect on the amount of fire in the moist, west-side forest fire regimes (Agee 1993) (figs. 3-4 and 3-10). Over 6,600 lightning-started fires were recorded in this region over a recent 21-year period, and most of these would have been actively suppressed (table 3-3). Although the vast majority of these fires probably would not have turned into large high-severity or mixed-severity fires, a few probably would have. Before the era of fire suppression, a few of these starts likely smoldered for weeks as small fires or as burning snags until a dry east wind event occurred, when those fires could spread rapidly producing large patches of high-severity fire along with patches of moderate- to low-severity fire. Recent fire rotations for high-severity fires in the two west-side fire regimes also appear to be at the high end of the historical range for U.S. Forest Service lands (table 3-4) (Reilly et al. 2017). Historical fire occurrence in these regimes varied at centennial scales with climate and human population density (e.g., Weisberg and Swanson 2003). Thus, given the occurrence of warm, dry conditions during much of the contemporary fire period, a rotation exceeding the upper end of the range suggests we are currently experiencing much less fire than would have occurred historically under a similar climate.[12]

The effects of fire suppression in the moist, west-side forests are quite different than in the dry forests. Fire suppression in relatively productive forests with long-fire-return intervals has little effect on fuel accumulation at the stand level (Agee 1993). However, fire suppression would drastically reduce the amount of early- and mid-successional vegetation in the landscape and thereby, reduce landscape-scale heterogeneity in forest composition, structure, and patch sizes. Mixed-severity fires burning at rotations of 50 to 200 years would have created a mosaic of forest successional stages, including multicohort old-growth stands (figs. 14, 16, and 17) (Tepley et al. 2013).

Fire severity in dry forests—

Although weather is the primary controller of fire occurrence, size, and severity, in some cases, in the NWFP area (Littell et al. 2009, Reilly et al. 2017), local controls (e.g., topography and fuels) are also important (Cansler and McKenzie 2014). There is significant concern that accumulation of live and dead fuels in understories as a result of fire exclusion and suppression has increased the threat and occurrence of larger areas of high-severity fire (Hessburg et al. 2000, 2005; Miller and Urban 1999a, 1999b, 2000; Parsons 1978, Parsons and DeBenedetti 1979). This threat is thought to arise from two processes: (1) increased accumulations of surface and ladder fuels (shrubs, small trees, lower canopy base heights) that increase flame length and fireline intensity under extreme fire weather conditions, and risk of mortality, even in large fire-resistant canopy trees; and (2) higher spatial continuity of fuel beds that can lead to

[12] Note, however, that for the infrequent and moderately frequent regimes, the recent 25-year record is very short and does not necessarily indicate deviation from historical regimes where fires were relatively infrequent (e.g., 505 to 1,000 years). Note also the relatively small sample sizes of fire history studies.

more rapidly spreading and larger patches of high-severity fire (fig. 3-26). These changes in fire behavior as a result of fuel accumulation are supported by theory, simulation models of fire behavior, and empirical studies of differences in fire behavior between stands where fuels have been reduced by mechanical and prescribed fire and those that have not been treated (North et al. 2012, Ritchie et al. 2007, Safford et al. 2012b, Schmidt et al. 2008, Stephens 1998, Stephens and Moghaddas 2005, Stephens et al. 2009, Weatherspoon and Skinner 1995). Evaluation of changes in fire patch size distributions with those of pre-Euro-American settlement era fire regimes are problematic because we lack landscape-scale quantitative data on frequency-size distributions of fire-severity patches for most areas (Collins et al. 2006; Collins and Stephens 2010; cf. Perry et al. 2011; Reilly et al. 2007; Williams and Baker 2014) (app. 3).

Empirical evidence for increasing total area of fire, and increasing area of fire patch sizes in recent decades, exists from studies across the Western United States, which are relevant to the NWFP area (Cansler and McKenzie 2014, Littell et al. 2010, Miller et al. 2008, Odion et al. 2004, Reilly et al. 2017, Westerling et al. 2006). However, evidence for increased proportion of high-severity fire in recent decades is mixed. Lutz et al. (2009) found evidence for increasing proportion of high-severity fire in the Sierra and southern Cascades of California, but Miller et al. (2012) did not find evidence of increasing total proportion of high fire severity from northwest California between 1987 and 2008. Miller et al. (2012) did find the sizes of high-severity patches to be increasing along with the overall increasing size of fires. Baker (2015a) did not find evidence for increasing proportion of high-severity fire in recent years in a study of ponderosa pine and mixed-conifer forests of the Western United States. Reilly et al. (2017) found no increases in the proportion of area burned at any level of severity between 1985 and 2010 in the Pacific Northwest but did see increasingly severe fire effects (e.g., large patches of high-severity fire) related to drought and annual area burned. Cansler and McKenzie (2014) found significant positive relationships in the northern Washington Cascades between climate and fire size, and between fire size and the proportion of fire events found in high-severity fire patches. They also found that the spatial aggregation of high-severity area within fires was greater in ecoregions with more contiguous subalpine forests and less complex topography.

It also appears that while recent fire frequencies for all severity classes are below what would have been expected for all the historical fire regimes in the region, the proportion of high-severity fire in fire-frequent regimes may be somewhat higher than it would have been historically. However, note that the recent rotations of high-severity fire in dry forests are still very low (table 3-4). Reilly et al. (2017) found that the amount of recent high-severity fire (23 to 26 percent) in the ponderosa pine, grand-fir, white fir, and Douglas-fir potential vegetation types was higher than what would be expected for these types under historical fire regimes. Mallek et al. (2013) reported that the percentage of high-severity fire in mixed-conifer forest types of the Sierra Nevada and southern Cascades of California was 5 to 8 percent during the pre-Euro-American period but was 22 to 42 percent in dozens of fires between 1984 and 2009. Miller and Safford (2012) reported that larger recent fires in pine and mixed-conifer forests in the southern cascades of California experienced 33 percent high severity, which was probably higher than the historical amount of high-severity fire. However, Odion et al. (2004) found that fires in 1987 in remote areas of the California Klamath had relatively low percentages (12 percent) of high-severity fire (defined as 100 percent scorch or consumed) and the percentage of high-severity fire in the 2002 Biscuit Fire was only 14 percent (Azuma et al. 2004). The relatively low percentage of high-severity fire in 1987 may be a result of weather conditions that were not as extreme as those of more recent fires (Taylor and Skinner 1998, Weatherspoon and Skinner 1995). Although the forests of the Klamath may have been less affected by fire suppression than more accessible forests, fire-return intervals during the suppression period are still nearly 50 percent longer (21.5 vs. 14.5 years) than during the presettlement period (Taylor and Skinner 1998). As fire sizes increase with climate warming (Odion et al. 2004), patch sizes of high-severity fire may also increase (e.g., Miller et al. 2012, Reilly et al. 2017). Very large patches of high-severity fire that kill older, dense forests would not be characteristic of the very frequent low-severity regime (Taylor and Skinner 1998), and efforts

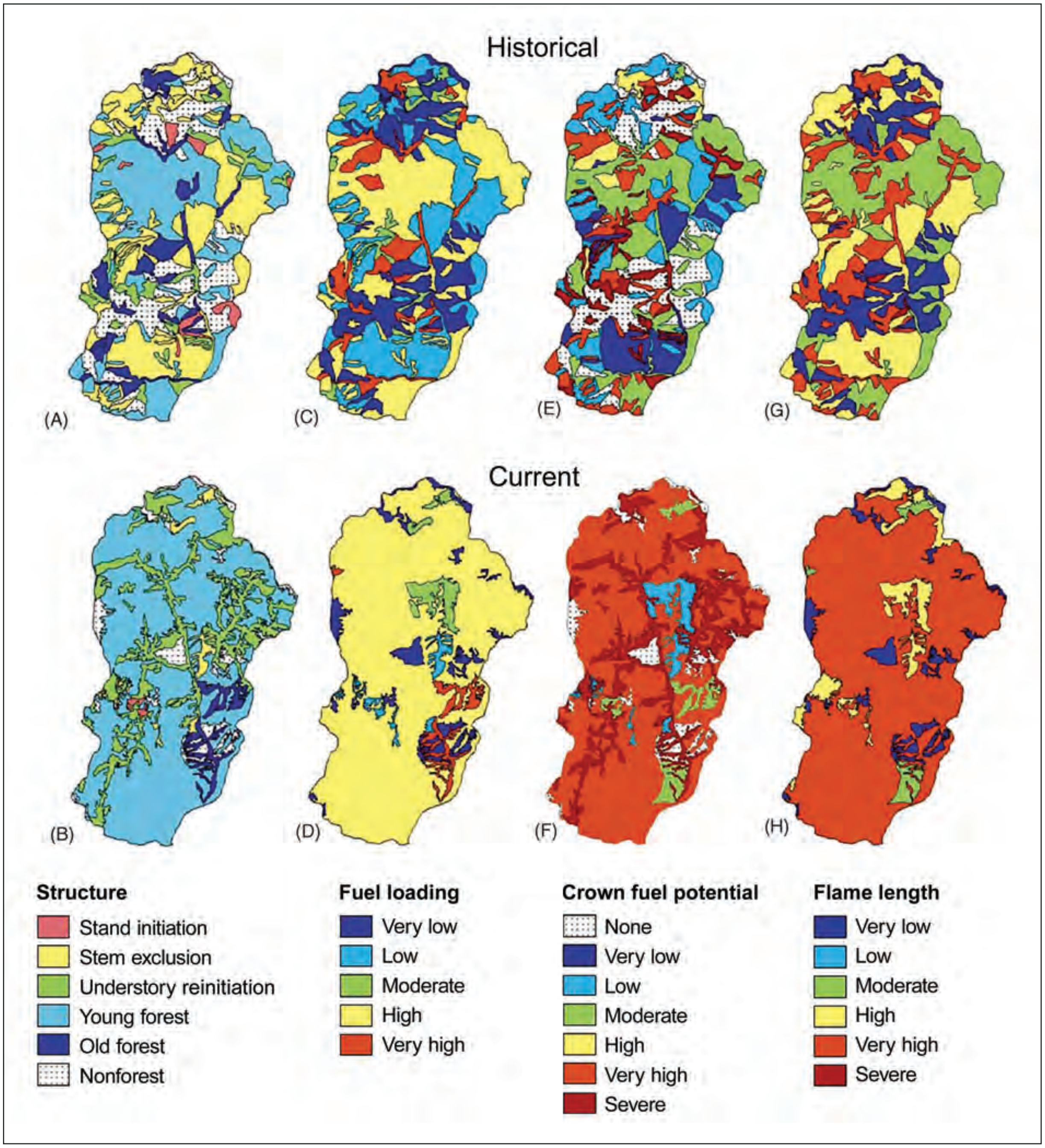


Figure 3-26—Reconstructed historical (1900s) and current (1990s) maps of dry forest subwatershed of the Lower Grand Ronde subbasin in the Blue Mountains province displaying historical and current structural classes (A and B), fuel loading (C and D), crown fire potential under average wildfire conditions (E and F), and flame length under average wildfire conditions (G and H), respectively. (From Hessburg et al. 2005). Although this is from a landscape outside of the Northwest Forest Plan (NWFP) area, similar changes have likely occurred in dry forests in many areas within the NWFP.

to restore frequent fire and reduce fuels in older and younger forests would contribute to maintaining the biodiversity (including spotted owls in the southern part of their range) that was adapted to a dynamic and heterogeneous mix of forest ages and structures.

Factors explaining variation in how fire-excluded forests burn when wildfire returns are not well understood. The observation that dry forests are experiencing less fire (excluding a direct effect of fire suppression), but more high-severity fire, or larger patches of high-severity fire than was true historically, is related to climate and fire suppression, but may also be due to shifts in vegetation-fire feedbacks. For example, it may be that with the absence of fire, coupled with succession to shade-tolerant and fire-intolerant species, is leading to forests that are less flammable under typical fire weather owing to a number of factors, including moister microclimate, denser stands that inhibit the free flow of wind, lower air and fuel temperatures owing to less direct sunlight, and more compact fuel beds (Engber et al. 2011, Estes et al. 2012, Kitzberger et al. 2011, Odion et al. 2004). For example, Weatherspoon et al. (1992) suggested that:

> ...success of initial attack on wildfires evidently is greater in areas of owl habitat within the Sierran mixed-conifer type. Countryman's (1955) description of fuel conditions within old-growth stands applies in large measure to fuel conditions within many mixed-conifer stands used by the California spotted owl. These stands are less flammable under most conditions, because the dense canopies maintain higher relative humidities within the stands and reduce heating and drying of surface fuels by solar radiation and wind. The reduction of wind velocity within closed stands discussed by Countryman is supported by wind reduction factors identified by Rothermel (1983) for stands with closed canopies. Windspeed at mid-flame height for fires burning in surface fuels is approximately one-tenth of the windspeed 20 ft (6.1 m) above the stand canopy.

However, they go on to say that:

> As fuels accumulate, however, fires that do escape initial attack—usually those burning under severe conditions—are increasingly likely to become large and damaging. Success in excluding fire from large areas that were once regulated by frequent, low- to moderate-severity fires has simply shifted the fire regime to one of long-interval, high-severity, stand-replacing fires… .

Some areas within the 2002 Biscuit Fire (which had relatively low total area of high-severity fire) could be an example of this shift in this regime, where moist multistoried older forests on north-facing slopes burned with high severity during the most extreme weather periods (hot dry east winds) of the fire (Thompson and Spies 2009).

Note that Countryman (1955) and Weatherspoon et al. (1992) never directly tested the hypothesis of higher humidity and fuel moisture in closed stands vs. more open stands. This was simply assumed to be so. Estes et al. (2012) measured an array of different sizes of fuels in closed, unthinned stands and open, thinned stands from spring snowmelt through fire season to the onset of fall rain/snow in the southern Cascades. They found moisture differences only in the early part of fire season (May–June). Moisture differences were gone by mid-season (July), and this carried through the remainder of the fire season. Further, the more open stands responded more quickly to the few rain events (thunderstorms) than did the closed stands. It appears that the long, dry summers of the Mediterranean climate areas in the southern parts of the NWFP area negate potential differences in moisture conditions because the closed stands catch up with the dry conditions of the open stands as the fire season progresses. Thus, the ability for crews to more readily catch fires in closed stands appears to be due to differences in exposure to sunlight creating higher air and fuel temperature and greater ease of windflow in the open stands.

Thinning can alter fire potential and microclimate. Higher windspeeds in thinned stands compared to unthinned stands may have contributed to the former burning with higher fireline intensity (Raymond and Peterson 2005) than the latter in the 2002 Biscuit Fire. Although most of the differences in fire effects in that study were attributed to higher fine fuel loading and lower moisture in the stands that had been thinned but were not underburned to reduce fine fuels. Bigelow and North (2012) noted that thinning and group selection can change microclimates of forests but they did not find that such changes had a large effect on fire behavior.

The interaction between vegetation and fire severity is also determined by foliar moisture of the herbaceous, shrub, and hardwood fuels. For example, in open dry forests subject to frequent fire, well-developed herbaceous layers can reduce flammability because moisture contents can remain high into September (Agee et al. 2002). In the Klamath Mountains and western Cascades, hardwood understories can significantly reduce fire intensity (Agee et al. 2002, Perry 1988, Perry et al. 2011, Skinner 2006, Skinner and Chang 1996). Some species of evergreen shrubs can also reduce flammability of forests landscapes under most weather conditions but provide dense flammable fuels under extreme fire weather conditions (Skinner and Weatherspoon 1996, Weatherspoon and Skinner 1995).

Weatherspoon and Skinner (1995) suggested that another reason for the differences between stands of larger, old trees and those of smaller young trees and plantations experiencing different levels of fire severity in the Klamath could be simply the susceptibility of trees of different sizes to damage by fire. Large trees, especially stands dominated by old Douglas-fir and ponderosa pine, would be more likely to survive fires than younger trees, especially small trees in plantations (Agee and Skinner 2005, Skinner et al. 2006). Although these multistoried stands have similar-size trees that succumb to the fires as do the young stands or plantations, the mortality is often hidden from satellite sensors by the surviving older, main canopy trees. Thus, the older stands become classified as experiencing mostly low-severity fire effects, while the others are classified as moderate- to high-severity fire effects even though fire intensity and sizes of trees actually killed could have been very similar (Weatherspoon and Skinner 1995). This is another example of the challenge of defining fire severity using single or simple metrics across variable vegetation types, and a potential source of confusion and debate (Reilly et al. 2017).

Use of Historical Ecology in Conservation and Restoration

As illustrated above, knowledge of the ecology of the period prior to Euro-American settlement and widespread changes in land use can be very useful in understanding these forests and can serve as a starting place for developing conservation and restoration plans and management practices for them (Allen et al. 2002; DellaSala et al. 2003; Demeo et al. 2012; Hessburg et al. 1999a, 1999b, 1999c, 2000, 2005; Keane et al. 2002, 2009; Landres et al. 1999; Morgan et al. 1994; Safford et al. 2012a; Swetnam et al. 1999). Knowledge of ecological history and knowledge of the historical range of variation (HRV) are not necessarily the same thing. General knowledge of ecological history may be more useful in management than a precise understanding of the range of variation in forest conditions (Hiers et al. 2016), which cannot be fully achieved for a number of ecological and social reasons. For example, while we may lack precise models or reconstructions of HRV for many landscapes in the region, we do have a reasonable foundation of historical knowledge for most areas. Ecological history reveals that forests were dynamic and best understood in terms of a HRV or its equivalent natural range of variation. The concept of HRV recognizes that habitats and ecosystems are dynamic in space and time, with historical ranges of behavior that are strongly constrained by the dominant climate, environment, and disturbances of an ecoregion. For the NWFP area, the HRV of forest structure among the four major fire regimes would have differed based on fire frequency and severity patterns and scale as described in the previous sections (fig. 3-27). Likewise, the HRV of forest structure would have differed across the major disturbance regimes based on whether small- to medium-size severity patches or high-severity patches were the major successional influence controlling patch dynamics.

Application of historical ecology HRV concepts and potential vegetation types in the Pacific Northwest and northern California must recognize the central role of climate variability in forest dynamics (Keane et al. 2009, Wiens et al. 2012, Wimberly et al. 2000). Temporal variation in climate drove the variability of historical fire regimes (Hessburg et al. 200b, 2004; Trouet et al. 2010), which are the product of interactions between forest composition and structure, fire weather, and ignitions. Variation in climate and fire regime was the driving force of the "range" in the HRV in forest structure and composition. For example, fire occurrences in many of the moist and cool forests of the region are "climate limited" (Briles et al. 2011, Colombaroli and Gavin 2010, Littell et al. 2009) or

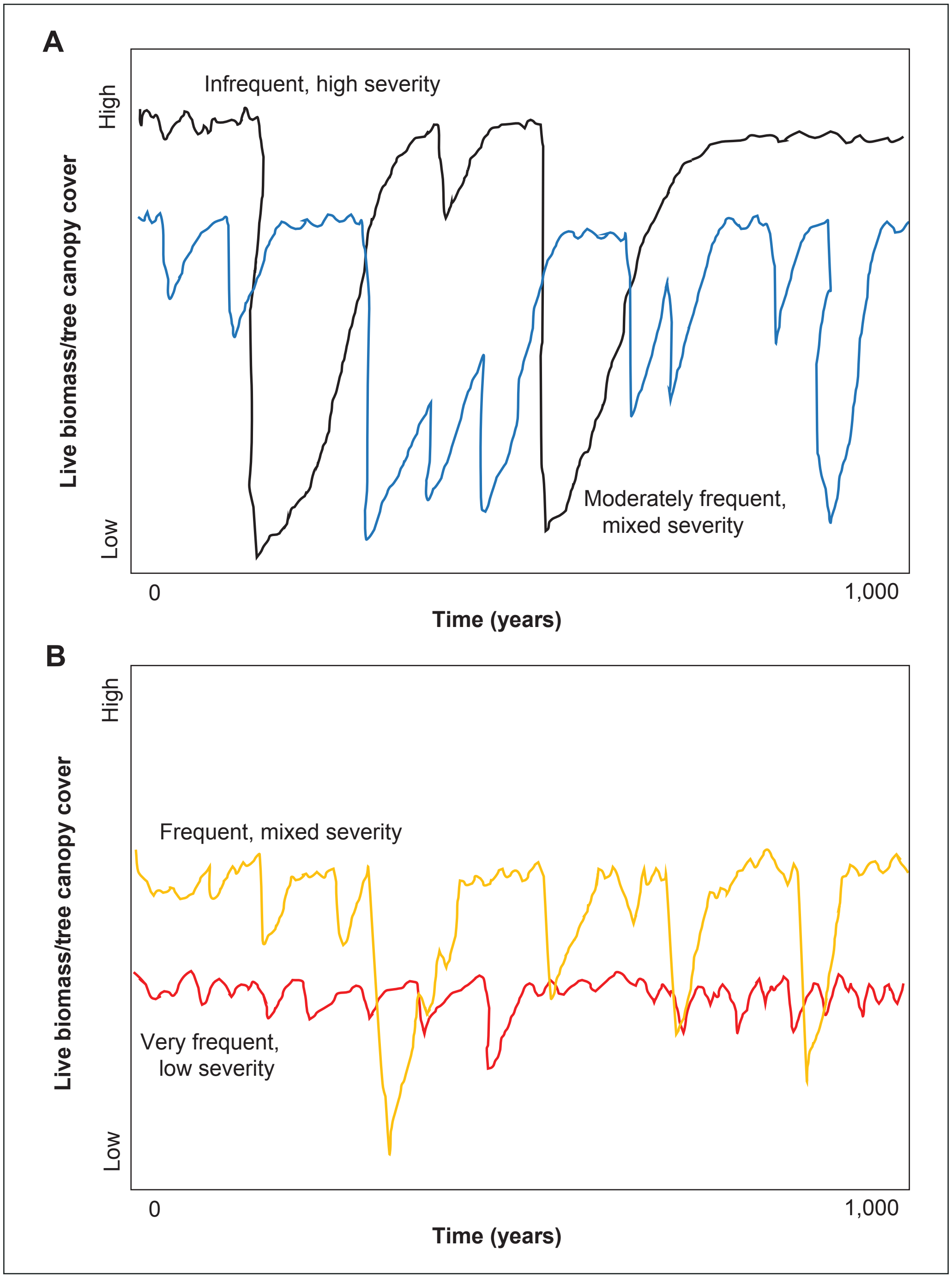


Figure 3-27—Hypothesized dynamics (historical range of variation) in live forest structure (biomass or cover) over a hypothetical 1,000-year period during the pre-Euro-American settlement period for an area of several thousand acres for (A) moist forest fire regimes and (B) dry forest fire regimes. Large declines in live biomass result from fire or wind; small declines result from fire, wind, insects, and disease.

"ignition limited" (sensu Agee 1993), but not fuel limited as the environments are typically productive enough to produce adequate fuels for burning within 10 to 15 years of a fire. If shrubs such as *Ceanothus* are present, they can act as a barrier to fire spread under less-than-extreme burning conditions (Briles et al. 2005, Mohr et al. 2000, Whitlock et al. 2004), or encourage rapid and intense fire spread under extreme fire weather conditions (Agee 1993, Moritz 2003, Schmidt et al. 2008).

Regionally, wildfire was episodic and could be synchronous in parts of the region especially in wetter climates of the high- and mixed-severity regimes (Weisberg and Swanson 2003). Although fires were frequent in the driest forest regions, variability in frequency existed, and climatically driven synchrony of widespread fire still exists even in the fire-frequent forests of the Western United States (Falk et al. 2011, Heyerdahl et al. 2008). Wildfire frequency was variable at decadal to millennial scales, i.e., it was nonstationary. According to Whitlock et al. (2008), who examined paleo fire history of forests of the Northwestern United States, "There is no stable fire regime on millennial time scales, because fire-episode frequency varies continuously as a consequence of long-term climate variations and their influence on vegetation." They go on to say, "Without supporting long-term paleoecologic data, short-sighted inferences about natural disturbance regimes and forest sensitivity are likely to be incorrect."[13] In other words, there were periods with relatively less frequent fire and other periods with relatively more frequent fire, creating a larger HRV if climate context is not taken into account. However, in drier parts of the region with more frequent fire, large-scale temporal variability and regional synchrony in fire was probably less than in regions with less frequent but larger fires (Hessburg et al. 2005; Heyerdahl et al. 2001, 2008; Kitzberger et al. 2006; Mohr et al. 2000; Morgan et al. 2008; Skinner et al., in press; Taylor et al. 2008; Trouet et al. 2010). Nevertheless, regionally extensive fire events associated with drought did occasionally occur in the eastern Cascades of Washington (Hessl et al. 2004).

Going forward, several authors have argued that given climate change, invasive species, and widespread landscape change, using historical conditions or ranges of variation as a narrow goal or target for conservation and restoration can be unrealistic, impossible, or even incongruent with conservation goals (Millar et al. 2007, Palmer et al. 2005). This is especially true if the goals include threatened and endangered species, such as the northern spotted owl in dry forests, whose habitat can be the product of human land use activities and altered disturbance regimes. However, it is self-evident that knowledge of historical forest dynamics is essential for conservation and restoration of native (historical) vegetative communities and associated wildlife species even under climate change. The challenge for application of the concept is to be aware of limitations and apply historical knowledge with caution. Hessburg et al. (2016) offer four caveats to using historical reference conditions as management guidelines:

- Mimicking historical conditions is not an end in itself, but is a means of accomplishing objectives (e.g., resilience to fire), and therefore appropriate only when it meets those objectives.
- The true value of historical information is in understanding how interacting fire and climate, and their variability through time and space, influenced ecological patterns of forest structure and successional conditions. This information can provide valuable direction for the complex process of ecological goal setting in management planning and implementation.
- Past conditions may not fully reflect future climate-vegetation-disturbance-topography linkages as a result of pervasive climate and

[13] Although paleoecological fire histories can give us a broader perspective on HRV, they are subject to methodological limitations. For example, fire history studies based on charcoal occurrence in sediment cores are subject to bias because charcoal production is partially determined by the nature of the fuels (e.g., herbaceous vs. woody). In the Klamath Mountains, the frequency of fire generally exceeded the resolution of the sediment cores, which was usually no finer than 30 years at best (Briles et al. 2005, Mohr et al. 2000, Whitlock et al. 2004). Further, over most of the Holocene, there was rarely a time when charcoal was not entering the lakes in the Klamath region. Rather than being an indicator of fire events, the amount of charcoal at different periods appeared to be more indicative of biological productivity. Charcoal varied by amount with the periods of light, flashy fuels characteristic of pine/oak woodlands represented by lower charcoal influx than in more productive periods characteristic of mixed-conifer forests (Mohr et al. 2000, Skinner et al. 2006, Whitlock et al. 2004).

land-use changes. Hence, one of the challenges may be deciding the degree to which past lessons are relevant to future management. Relevance will depend on goals, reasonable expectations of the future climate, and resources required to attaining those goals.

- Because regional landscapes are highly altered, restoration restricted to local landscapes is insufficient to address large-scale restoration needs.

Remember that we understand recent HRV (e.g., past 500 years) better than we understand HRV of the more distant past or what the range of variation will be in the future. Consequently, planning efforts based on ecological history or HRV will need to be flexible, adaptive, and periodically revised to keep up with new knowledge and changing ecosystems. To deal with the challenges of restoration or managing for resilience, Hobbs et al. (2014) recommended that landscape frameworks and assessments be used to identify where it is possible to retain or restore native biodiversity and where novel or "hybrid" (seminatural) ecosystems might be a management goal either because of human values (e.g., areas of dense forests for wildlife created by fire exclusion) or because of the impracticality or impossibility of returning those areas to their pre-Euro-American state or HRV (see chapter 12 for more discussion of this issue). We further discuss scientific understanding of approaches for dealing with these and other challenges of restoration or creating resilient forests in sections below.

Ecosystem Function

The preceding sections have emphasized forest structure, composition, and disturbance process, but ecosystems can also be characterized through their functions (ecological processes or activities), which also differ with successional stage and disturbance regime. Key functions include primary productivity and carbon fixation, nutrient cycling, hydrological functions, and habitat for biota (Franklin et al. 2018). We briefly review how these differ with succession here with a focus on productivity, carbon and nutrient cycling. For more information about hydrological functions and habitat, see chapters 6 and 7.

Old-growth forests are productive ecosystems, fixing a large amount of solar energy in what is termed gross primary production (Franklin and Spies 1991). Following major disturbances, ecosystem live biomass and net primary productivity (difference between carbon fixed through photosynthesis and lost to respiration) are relatively low (Bormann et al. 2015, Spies 1997), in contrast with later successional stages. As trees grow and canopies close, the rate of carbon sequestration and biomass accumulation becomes high. Biomass reaches its highest level in older forests, but net primary production declines toward zero because growth and mortality are roughly equal. While stand-level net primary productivity and carbon accumulation is low in older forests, the rate of biomass growth for individual trees continues to increase with tree size (Stephenson et al. 2014).

Carbon, which primarily resides in the wood and soils, is highest in old forests (Law and Waring 2015). Douglas-fir/western hemlock forests can continue to be a net sink for carbon for more than 500 years, thanks to the contribution of primary production of shade-tolerant understory trees (Harmon et al. 1990, 2004). Older moist forests of the NWFP area can attain higher stand (tree) carbon biomass than tropical or boreal forests (Law and Waring 2015). Young forests store less carbon but accumulate it at higher rates than old forests.

Recent large wildfires in coniferous forests of the region release carbon, but the total emitted carbon is less than previously thought, partly because most fires in the region have burned with mixed severity. For example, Campbell et al. (2007) found that only 1 to 3 percent of the carbon in trees larger than 3 inches (7.6 cm) was combusted in the 2002 Biscuit Fire (Campbell et al. 2007). Total carbon emitted from four fires in Oregon averaged 22 percent of prefire carbon for all pools (Meigs et al. 2009). As the biomass killed in fires slowly decomposes over decades to centuries, carbon is emitted to the atmosphere as carbon dioxide and other trace hydrocarbons. About half the carbon remaining after a fire stays in the soil for about 90 years; the other half persists for more than 1,000 years as charcoal (Deluca and Aplet 2008, Law and Waring 2015).

Forest management effects on carbon differ with management intensity, rotation length, and forest type. It is often thought that managing forests on a short rotation (e.g., 40 to 50 years) would provide the most effective long-term carbon sequestration, but longer rotations and selective or no harvest provides the most carbon sequestration (Harmon et al. 1990, Ryan et al. 2010). Forest management under the NWFP to promote older forests with a low level of timber harvest would provide for more carbon sequestration than more intensive management (Creutzburg et al. 2017, Kline et al. 2016).

In forests prone to frequent fires, the carbon and forest management picture is more complex, with some studies showing a positive benefit of forest fuel reduction on carbon sequestration and others showing a negative effect. Some modeling suggests that carbon stocks over the long term are best protected by fuel treatments that create relatively low-density stands dominated by large, fire-resistant trees (Hurteau and North 2009). Other studies (Ager et al. 2010, Loudermilk et al. 2016, Spies et al. 2017) found that active management reduced carbon stored in the forest landscape by 5 to 25 percent for at least several decades. The effect of management on carbon depends on how frequently management treatments encounter fire and reduce fire severity. When a fire encounters a recently treated area, less carbon is likely to be emitted than when it encounters an untreated forest of the same type. However, at a landscape scale, many treatments will not experience a fire and the management actions there will reduce carbon sequestration. The net effect at a landscape scale may be to reduce carbon sequestration unless those treatments are strategically placed and occur where fire is most likely to happen. Further, the more active the fire regime becomes under climate warming scenarios, the more important strategically placed fuels treatments (e.g., Finney et al. 2007, Schmidt et al. 2008) become in protecting carbon stores (Loudermilk et al. 2013, 2016).

Nutrient cycling varies with successional stages and forest region. Old-growth forests are highly retentive of nutrients, and sediment outputs from old-growth watersheds are typically low (Franklin and Spies 1991, Swanson et al. 1982). Many of the forests of the NWFP area are nitrogen limited, but several natural processes exist that capture nitrogen and make it available for vegetation growth. old-growth forests can support canopy lichens such as *Lobaria oregana, L. pulmonaria,* and others that fix nitrogen and then "leak" significant amounts of nitrogen to the ecosystem (Antoine 2004). Immediately following stand-replacement disturbance, rates of erosion and nutrient loss can be elevated until vegetation recovers (Ice et al. 2004). As plants establish and cover increases during early-successional and young forest stages, sediment losses return to predisturbance levels, and N_2-fixers such as *Ceanothus* spp. and hardwoods such as red alder (*Alnus rubra*) begin to increase organic matter and nutrient availability (Borman et al. 2015, Compton et al. 2003). While red alder can add available nitrogen to forest ecosystems, the high rates of nitrification can accelerate cation leaching and soil acidification relative to conifer-dominated stands (Compton et al. 2003). Shrubs and sprouting hardwood trees can also help reduce nutrient losses after wildfire in forests of southwestern Oregon. While the longer term benefits of early-seral plant communities to conifer tree growth are still not well understood (Bormann et al. 2015), it is generally understood that early-seral herbaceous, shrub, and hardwood tree communities can all play an important role in supporting forest nutrient cycling and productivity.

Restoration efforts in dry forests can also benefit soil fertility and productivity. Fire suppression can lead to increases in nitrogen pools in ecosystems, but the majority is bound in forms that are less available to plants (Ganzlin et al. 2016). Forest restoration treatments, including prescribed burning, can produce short-term pulses of nitrogen in forms that are available to plants. Thinning alone will not produce these nutrient benefits and is not an effective surrogate for fire in terms of nitrogen. Frequent prescribed fire that emulates historical fire frequency and severity is necessary to maintain rapid rates of nutrient cycling in these dry forest ecosystems. However, while the nutrient effects of fire may be ephemeral, benefits to other soil resources and processes such as available water and photosynthetic rates may be longer term (Ganzlin et al. 2016).

Conservation and Restoration Needs

In this section, we summarize the major conservation (e.g., protection of existing vegetation) and restoration (e.g., promotion of desired conditions) needs for moist and dry forests relative to the original goals of the NWFP and of the 2012 planning rule under which the NWFP currently operates (table 3-5).

Estimates of forest change for the NWFP region suggest that the need for conservation and restoration of the ecological integrity of old-growth forests and other successional stages of the region spans a wide range of the disturbance regimes and forest types. For example, Haugo et al. (2015), found that at least 40 percent of all coniferous forests in eastern Washington and eastern and southwestern Oregon are in need of management to restore wildfire, fuel, or forest structure conditions to be more consistent with the natural range of variation. After more than 125 years of land clearing, timber harvest, 20th century high-severity wildfire associated with early logging and land use, fire suppression and succession, the sum of mature and old-growth forest (OGSI 80) across all the fire regimes is 17.8 million ac (7.2 million ha), or ~ 39 percent of all public and private forest-capable lands in the Plan area (Davis et al. 2015). When only the oldest multilayered forests with trees >200 years old (OGSI 200) are considered, the current amount is ~7.6 million ac (3.1 million ha), or 17 percent of all public and private forest-capable lands. Of that 17 percent, more than 80 percent is on federal lands. It is difficult to estimate what percentage of the historical range of older forests this represents for several reasons, including lack of quantitative studies of HRV across the region, uncertainties in estimates of HRV, and the current definitions do not fully capture the diversity of older forest conditions, especially for older ponderosa pine and mixed-conifer forests of the low- and mixed-severity regimes. If we focus on trees older than 200 years (OGSI 200) in moist forests zones west of the Cascade crest, then the total remaining may represent 17 to 23 percent of the amount that was present on average before the mid-1800s. This assumes that at least 60 percent of these forests areas were covered by forests containing trees older than 200 years (FEMAT 1993, Wimberly 2002).

Moist forests—

In the moist forests zone, losses of older forest have resulted mainly from clearcutting for timber management (Spies et al. 1994). The decline in older forest has been sharp as indicated above. For example, the vegetation structure of northern spotted owl habitat (not necessarily the same as

Table 3-5—Summary of vegetation conservation and restoration needs for moist and dry forests of the Northwest Forest Plan (NWFP) region related to the ecological goals of the NWFP and the 2012 planning rule

Forest region	Conservation needs	Restoration needs
Moist forests	Protect existing older forests stands and large patches of older forests from logging and high-severity fire. These have been greatly reduced by timber management and other land uses.	Increase vegetation diversity in plantations and accelerate development of older forest structure and composition. Reduce fragmentation and increase connectivity of older forest patches. Create or promote early-seral vegetation where needed to provide seral stage and landscape diversity. Restore disturbance processes (e.g., fire) where feasible.
Dry forests	Protect existing large fire-tolerant trees in areas of dense and open forest. Manage and protect existing dense old-growth forest stands as necessary to meet late-successional species and ecosystem integrity needs.	Restore low- and mixed- severity fire as key ecological process. Increase areas of open old forests to promote resilience to fire and climate change and meet needs of species. Develop landscape-level strategies to create desired mosaics of open and dense old forest and to increase resilience and meet simultaneous needs of wildlife species and ecological integrity. Restore diversity to plantations, including tree species mixes.

old-growth forests) has declined by 20 to 52 percent across the different provinces between 1930 and 2002 (Lint et al. 2005). Many plantations on federal lands are 30 to 60 years old and average about 20 to 25 ac (8.1 to 10.1 ha) with some as large as 60 ac (24.3 ha) (Cohen et al. 2002). They were often planted primarily with Douglas-fir (or at most a total of one or two additional species) at an even spacing. Logging and site-preparation treatments to control competing or unwanted vegetation resulted in uniform stand density with lower levels of shrub and hardwood components, and fewer snags and down wood structures (Bailey and Tappeiner 1998, Spies and Cline 1988). A large percentage of federal forest land outside of wilderness areas is covered by such plantations—as much as 40 to 55 percent of some landscapes, including many late-successional reserves (LSRs) (fig. 3-28). In summary, management efforts to ensure high density and species uniformity were often so successful that conditions in these stands do not match the heterogeneity and growth trajectories of naturally regenerated postwildfire stands (Donato et al. 2011, Freund et al. 2014, Larson and Franklin 2005, Tappeiner et al. 1997, Tepley et al. 2014, Winter et al. 2002a) (fig. 3-14).

Other vegetation restoration needs for the moist forests zone relate to early-seral and other mid-successional stages that have been reduced by fire-suppression reforestation, timber stand improvement treatments that ensured full stocking, optimal sawtimber growing conditions, and control of unwanted vegetation (Agee 1993, Cole and Newton 1987, White and Newton 1989). Fire suppression in these infrequent-fire regimes has little impact on the risk of high-severity fire but it does reduce the amount of early-seral and vegetation diversity in a landscape. Numerous small- to mid-size fires would likely have served as barriers to fire spread where they created patches of deciduous shrubs and trees. The vegetation diversity created by these fires probably regulated the frequency-size distributions, especially of the larger fires. The amount of early-seral condition may have been relatively high (<30 percent) in these regimes during the late 1800s and early 1900s when the legacy of aboriginal burning was still evident (Robbins 1999) along with Euro-American-ignited fires from land clearing and logging (fig. 3-6). The amount and diversity of early-seral vegetation created by these fires would have been reduced where snags were cut down and large-scale planting efforts reduced the period of time before tree canopy closure. The patterns of early-seral patch size shapes, distribution, and structural heterogeneity created by logging and reforestation in the late 20th century are not representative patterns typically found under historical fire regimes (Nonaka and Spies 2005). The structure and composition of early-successional vegetation and young forests created by clearcut logging significantly differed from those of postwildfire conditions because intensive timber management removed all live and dead trees, and herbicides (in early years on federal lands), and planting of Douglas-fir seedlings reduced diversity of vegetation and shortened the nonforest period of succession. Moreover, harvest unit boundaries often followed land ownership boundaries on private lands, and older cutting units on federal lands (the most recent occurred in the early 1990s) represented small-size (25 to 40 ac [10.1 to 16.2 ha]), regularly shaped units with landscape patterns that differed from those created by fire.

Dry forests—

We have already described many of the changes that have occurred in the dry forests as a result of fire exclusion and logging. Analysis from the Interior Columbia Basin Ecosystem Management Project (Hann et al. 1997; Hessburg et al. 1999a, 2000) provides a picture of how the area of dense multilayered older forest has changed from historical to current (late 1990s) in dry forests of eastern Washington and Oregon (fig. 3-26) (table 3-6).

In another study, Lint (2005) estimated that the amount of dense older forest with grand fir and Douglas-fir that is suitable for spotted owls (we use this as an approximation of multilayered old growth, but it is not necessarily the same as dense old-growth forest structure) has actually increased by 16, 6, and 11 percent in the eastern Cascades of Washington, Oregon, and California Klamath Provinces, respectively, from 1930[14] to 2002. These data suggest that the historical fire regime in these provinces did not favor large areas of either late-successional, multilayered old forest or northern spotted owl habitat.

[14] Landscapes of the 1930s would have already been altered by logging, grazing, fire exclusion, and occurrence of fires associated with land use activities. Fire exclusion would have increased the amount of dense forest by 1930 (McNeil and Zobel 1980, Merschel et al. 2014).

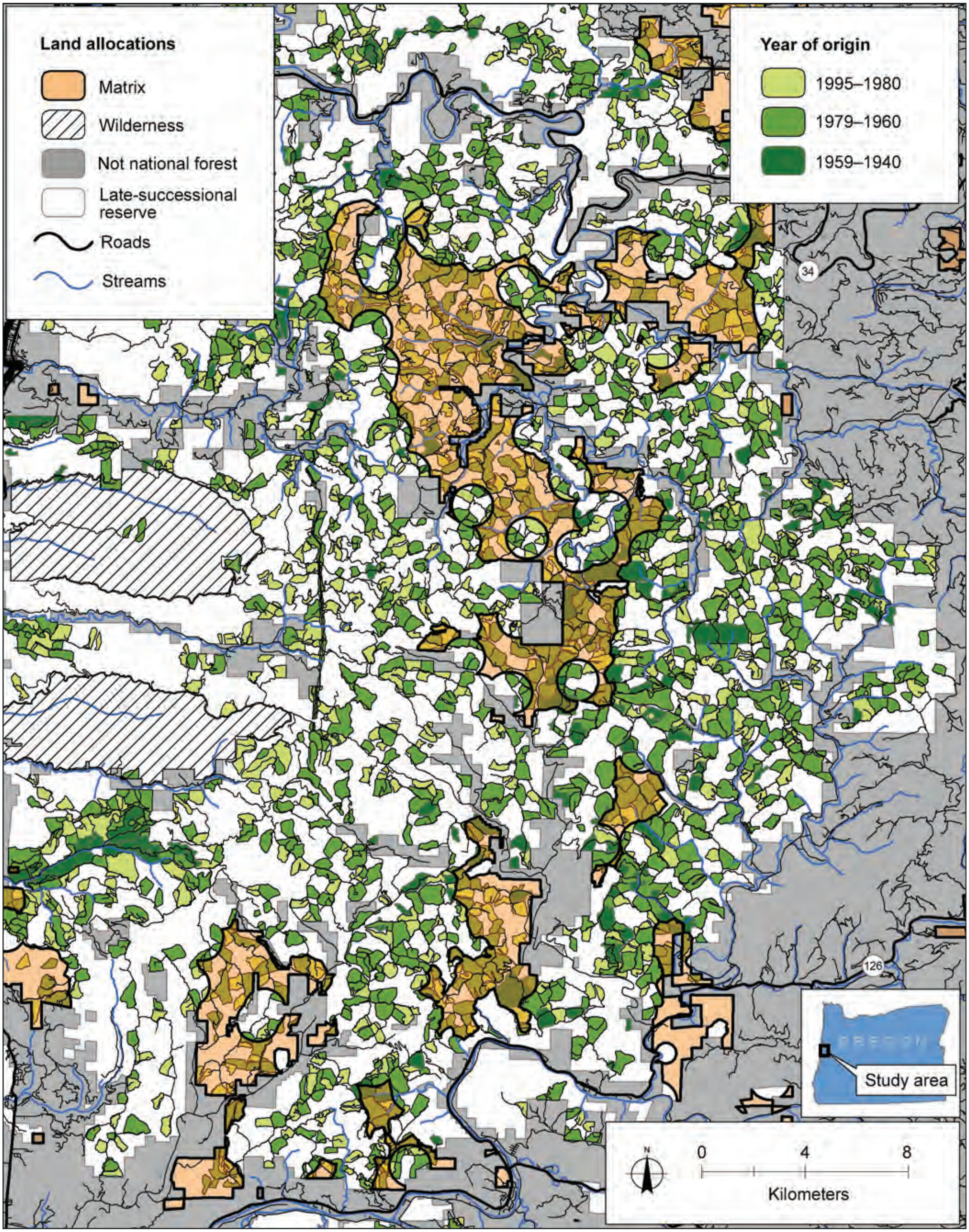


Figure 3-28—Plantations and the dates of their origin in a landscape containing late-successional reserve (in white), wilderness (striped), and matrix (orange) lands on the Siuslaw National Forest in coastal Oregon. From Stewart Johnston (retired), Siuslaw National Forest.

Table 3-6—Historical and 1990s percentages of total forest area in late-successional multistory forest in provinces of the Interior Columbia Basin Ecosystem Management Project

	Province		
Time period	Northern Cascades	Southern Cascades	Upper Klamath
Historical	7.0	0.7	4.8
Current	16.6	4.0	3.5

Sources: Hann et al. 1997; Hessburg et al. 1999a, 2000.

Changes in area of medium and large old trees have also occurred. Hessburg et al. (1999a) documented reductions in province area of forest patches with medium and large trees in the overstory (>40 percent canopy cover) in the interior Columbia River basin. In the Northern Cascades and Upper Klamath provinces, area of medium- and large-size trees in the overstory declined from 30 to 24.9 and from 28.9 to 25.3 percent, respectively. However, area of medium and large trees in the overstory significantly increased in the Southern Cascades province from 17.1 to 32.8 percent. They also show historical landscapes with significant areas of grassland, shrubland, woodland, and stand initiation forest conditions and young forests that had invaded meadows (figs. 3-21 and 3-22). These mid- to late-20th century increases in forest density are in addition to the substantial increases in stand density and shade-tolerant species that occurred between 1890 and 1930 as a result of fire exclusion (owing to grazing, logging, and eventually active fire suppression) and other factors (Merschel et al. 2014, Taylor and Skinner 2003). Currently, the percentage of relatively open, low-density (<80 trees per acre) forest with large old trees in mixed-conifer and Douglas-fir potential vegetation types is about 10 percent, while the area of dense forest (>584 trees per acre (1,442 trees per hectare)) with old trees covers about 35 to 42 percent of the potential vegetation types (Reilly and Spies 2015). These increases in shade-tolerant densities have made forests less resilient to fire as described above.

Increases in forest density are not the only conservation and restoration concerns in the dry forests. Loss of large, fire-resistant trees to logging and wildfire has also strongly affected forest ecosystem integrity, resilience, and wildlife habitat in both the very frequent low-severity and frequent mixed-severity fire regimes of the dry forest zone. For example, the density of large fire-tolerant tree species (e.g., ponderosa pine and Douglas-fir) has decreased substantially as a result of high-grade logging (selective removal of large mostly commercially valuable trees) (e.g., Hessburg et al. 1999a, 2000, 2003, 2005; Merschel et al. 2014) and clearcutting and plantation establishment. Hagmann et al. (2014) estimated that the area of forests dominated by large old trees has been reduced from 91 to 29 percent for dry and moist mixed-conifer in one landscape in the eastern Oregon Cascades. Increases in future development of large, old fire-intolerant trees may be limited as a result of forest densification and fire suppression. We could find no disagreement in the literature on the issue of restoration needs and concerns for large old conifers (e.g., Baker 2012, Stine et al. 2014). This issue is prominent in the eastern Cascades of Washington and Oregon and in California, where topography and proximity to settlement made these large valuable trees an easy target for logging (Hessburg and Agee 2003; Hessburg et al. 2005, 2015, 2016; Merschel et al. 2014; Richie 2005). Loss of large trees is less of an issue in more remote sites in rugged and difficult-to-access areas such as the less roaded areas of the Klamath Mountains.

Timber Management and Old-Growth Conservation

The NWFP strategy was based on the assumption that historical timber management approaches (e.g., removal of large or old early-seral and fire-tolerant trees) are not compatible with the full ecological functions of old-growth forests and other successional stages. Since FEMAT (1993), no scientific evidence has emerged that intensive timber production (e.g., clearcutting and short-rotation plantation forestry) and old-growth forest conservation are compatible at stand levels for any of these forest types and disturbance regimes.

Moist forests—

In moist forests zones, partial cutting, in the form of green tree retention harvest (see section below for more discussion of this method), patch cutting (creating gaps less than a few acres), or selection harvest methods may retain the habitats of some late-successional animal and plant species (Baker et al. 2016, Gustafsson et al. 2012, Halpern et al. 2012, Hansen et al. 1995a, Rosenvald and Lohmus 2008). It also retains some of the ecological functions of old growth, but could strongly affect dead wood amounts. The accompanying road and harvest systems would add additional impacts. Very long management rotations (e.g., more than 150 or 200 years) could in theory produce some of the habitat and ecosystem service benefits of older forests (Kline et al. 2016), but it would take at least a century to quantify these effects, and no long-term studies are currently underway.

One of the only operational plans to meet both older forest conservation goals and timber production in moist forests in the literature is the "structure-based management" approach proposed by the Oregon Department of Forestry for the state forests in the northern Oregon Coast Range (Bordelon et al. 2000). In this approach, management targets were sorted into five stand types, with the two oldest, "layered" and "older forest structure" intended to meet late-successional conservation goals. There are no reserves, and older forest conditions are met through long rotations. The areas in each stand type can differ over time, e.g., between 20 and 30 percent of older forest structure, as harvesting and succession shift age and structure classes over the landscape. Spies et al. (2007) and Johnson et al. (2007) used a landscape model to approximate this strategy. Modeling results suggest that, over time, this approach created a greater diversity of habitat benefits, including increases in older forest habitats and higher levels of wood compared to federal management under the NWFP. No formal field assessment of the ecological or economic implications of this approach has been attempted. At this stage, the Oregon Department of Forestry is under pressure from the counties to increase revenues and is in the process of modifying or abandoning the approach (http://www.nwtimberblog.blogspot.com/2013/11/board-of-forestry-seeks-better.html; http://www.northcoastcitizen.com/2016/12/officials-say-county-will-not-opt-out-of-class-action-lawsuit-over-timber-harvest/).

Other examples of management agency efforts to meet biodiversity and timber management goals exist for moist forests but have not been published or reviewed in the peer reviewed literature. The most prominent and well-developed approach for integrating timber management with old-growth forest conservation in moist forest zones may be the Washington Department of Natural Resources Habitat Conservation Plan for state trust lands (http://www.dnr.wa.gov/programs-and-services/forest-resources/habitat-conservation-state-trust-lands), which has been implemented across more than a million acres of state and private land with the goal of maintaining old-growth forest species and providing sustainable levels of timber production. It is based on maintaining a mosaic and network of patches of old-growth and mature forest structure for terrestrial and aquatic species.

Until more research is done, including field-based tests and monitoring, there is little debate that the best way to conserve and maximize old-growth values in moist forests is to exclude intensive timber management activities (e.g., clearcutting and plantation establishment) in old growth. This was the direction of the NWFP when it placed 80 percent of the remaining old-growth forest patches on federal lands into LSRs. The remaining 20 percent was placed into matrix lands—open to timber management, using innovative silviculture (e.g., ecological forestry) according to approved plans (USDA FS 1994) (fig. 3-29). The suggested management approach of the NWFP in the matrix lands, along with experiments in adaptive management areas, had they been implemented, would have enabled scientists and managers to learn about tradeoffs associated with managing for timber and ecosystem values at patch levels. As it stands now, we know relatively little about these tradeoffs because of a lack of implemented studies—the exceptions being the simulation studies of Cissel et al. (1999) and Spies et al. (2007) for moist forests.

Gary Rost

Figure 3-29—Example of a green tree retention unit created on Central Cascades Adaptive Management Area on the Willamette National Forest. The goal was to emulate stand structure created by a partial stand-replacement fire and produce timber.

Dry forests—

Clearcutting and plantation management are also not compatible with management for ecological integrity and resilience in dry forests (Franklin et al. 2013). However, other forms of management (table 3-5) may be needed to promote ecological integrity and resilience to climate change as characterized by the 2012 planning rule. Restoration thinning and prescribed fire in forests containing trees over 80 years would promote resistance and resilience to fire and climate change both within and outside LSRs. Some of these restoration activities could provide economically valuable wood products. Areas of dense old, multilayered forests and owl habitat can still be provided at landscape scales, but they would be more dynamic, shaped by fire and other natural disturbance agents. A holistic landscape-restoration strategy has been proposed for the 4-million-ac (~1.6-million-ha) Okanogan-Wenatchee National Forests. The plan seeks to use a variety of vegetation and fuels management techniques to reduce wildfire vulnerability across the landscape, including in areas adjacent to owl habitats in "critical habitat" (USDI 2012), and to restore fire regimes, to increase resilience to climate change. More research is needed in these dry dynamic landscapes to develop and evaluate approaches for achieving both ecosystem and focal species goals (see chapter 12).

Reserves in Dynamic Ecosystems

Concepts—

Protected areas or reserves are a well-established strategy for conserving biodiversity by limiting human activities (e.g., intensive timber management and development) that are incompatible with certain ecological objectives (Lindenmayer and Franklin 2002). However, the efficacy of reserves as the **sole** basis for conserving biodiversity has been challenged by a number of authors (e.g., Fischer et al. 2006, Lindenmayer and Franklin 2002). These challenges relate to several concerns: (1) biodiversity reserves cover only a small part of the Earth's land surface (e.g., <6 percent) (Fischer et al. 2006); (2) globally, the majority of reserves tend to be small in area (tens to <25,000 ac [~10 000 ha]) (Bengtsson et al. 2003), making them susceptible to impacts from large rare events (e.g., fire and wind) and influences (e.g., invasive species and human activities) from outside the reserves; and (3) most reserves are static and climate change may shift environments and species distributions to unreserved areas (Carroll et al. 2010).

A fundamental design recommendation for reserves is that they should be considerably larger than the largest disturbance patch size if they are to maintain habitat and populations of the most extinction-prone species (Pickett and Thompson 1978). This concept, which is known as "minimum dynamic area" requires knowledge of patch size distributions of infrequent disturbances that would be considered incompatible with conservation goals. Such knowledge is lacking for most disturbance regimes, especially under climate change, but it can be estimated using historical information and power laws (e.g., see Moritz et al. 2005).

The reserve design of the NWFP was a late-successional forest coarse-filter strategy that was based heavily on the needs of the northern spotted owl and leveraging existing reserves (e.g., wilderness) where appropriate. The reserve strategy attempted to mitigate the shortcomings of other reserve-based conservation approaches by increasing the proportion of reserves on federal lands to 80 percent (including congressional reserves, LSRs, riparian reserves, and administratively withdrawn areas). The congressional reserves and LSRs represented 28.1 percent (15.8 million ac or 6.4 million ha) of all public and private forest lands in the NWFP area, which made it one of largest reserve systems for any temperate forested ecoregion in the world. The individual LSRs under the NWFP are also relatively large. For example, 47 percent of the individual LSRs are larger than 25,000 ac (~10 000 ha), and three are larger than 250,000 ac (~100 000 ha) (fig. 3-30). Compared to the size of recent patches of high-severity fire (fig. 3-30), the sizes of the reserves are typically larger, although many (>120) LSRs are relatively small (e.g., <25,000 ac) and could be completely burned in a single fire event with large patches of high-severity fire (e.g., 25,000 ac).

The NWFP hypothesis was that a large network of reserves well-distributed across the region would be resilient to expected losses from wildfire over a period of 100 years. While losses were expected, there was no estimate of how much loss would be too much for the goals of the Plan. The reserve patch size and fire-size analysis indicated that, for the most part, the reserves have been large enough and numerous enough to absorb many recent large fires with limited loss of OGSI 80 or OGSI 200 forests in many but not all provinces. However, it must be remembered that recent historical fire history trends will not necessarily continue in the future. Given current trends, it is likely that one to several of the LSRs, especially the small ones, will experience significant losses of OGSI to large patches of high-severity fire over the next few decades. The infrequent fire regimes of the area have the potential to burn with very large fires, and it remains to be seen if the sizes and numbers of LSRs are sufficient to meet the goals of the Plan under climate change or other threats (e.g., invasive species).

The effectiveness of the NWFP regional reserve-matrix strategy in meeting ecological goals under current and future climate has received relatively little attention in scientific literature. The limited studies suggest that the existing network and standards and management guidelines of reserves, which spans a wide range of elevations and 10 degrees of latitude, will provide a good (but not necessarily optimal) foundation for meeting conservation goals in moist forest zones under a changing climate (Carroll et al. 2010, Spies et al. 2010). However, other than Carroll et al. (2010) and Carrol (2010), no quantitative studies of the NWFP reserve network or the regional plan as a whole have been conducted outside of efforts focused on conservation planning for the northern spotted owl (USDI 2012, USFWS 2008). In general, the science of regional conservation planning and assessment, including evaluation of reserve networks, has advanced considerably since the NWFP was implemented. For example, Margules and Pressey (2000) presented a systematic approach for evaluating reserve network plans and implementation and Virkkala et al. (2013) demonstrated a methodology to evaluate the viability of reserve networks for protecting biodiversity in the face of climate change in Finland. According to Carroll (2010), "Rigorous assessment of the implications of climate change for focal species requires development of dynamic vegetation models that incorporate effects of competitor species and altered disturbance regimes." In his assessment of the resiliency of the NWFP reserve network for multispecies conservation under climate change, Carroll (2010) did not address how wildfire might affect the conservation goals of the Plan, which is a significant concern. The development of regional-scale vegetation and species occurrence data and vegetation dynamics models, including spatial fire landscape models (e.g., Scheller et al. 2011, Spies et al. 2017), in recent years suggests that a more rigorous and comprehensive evaluation of the NWFP regional strategy would now be possible.

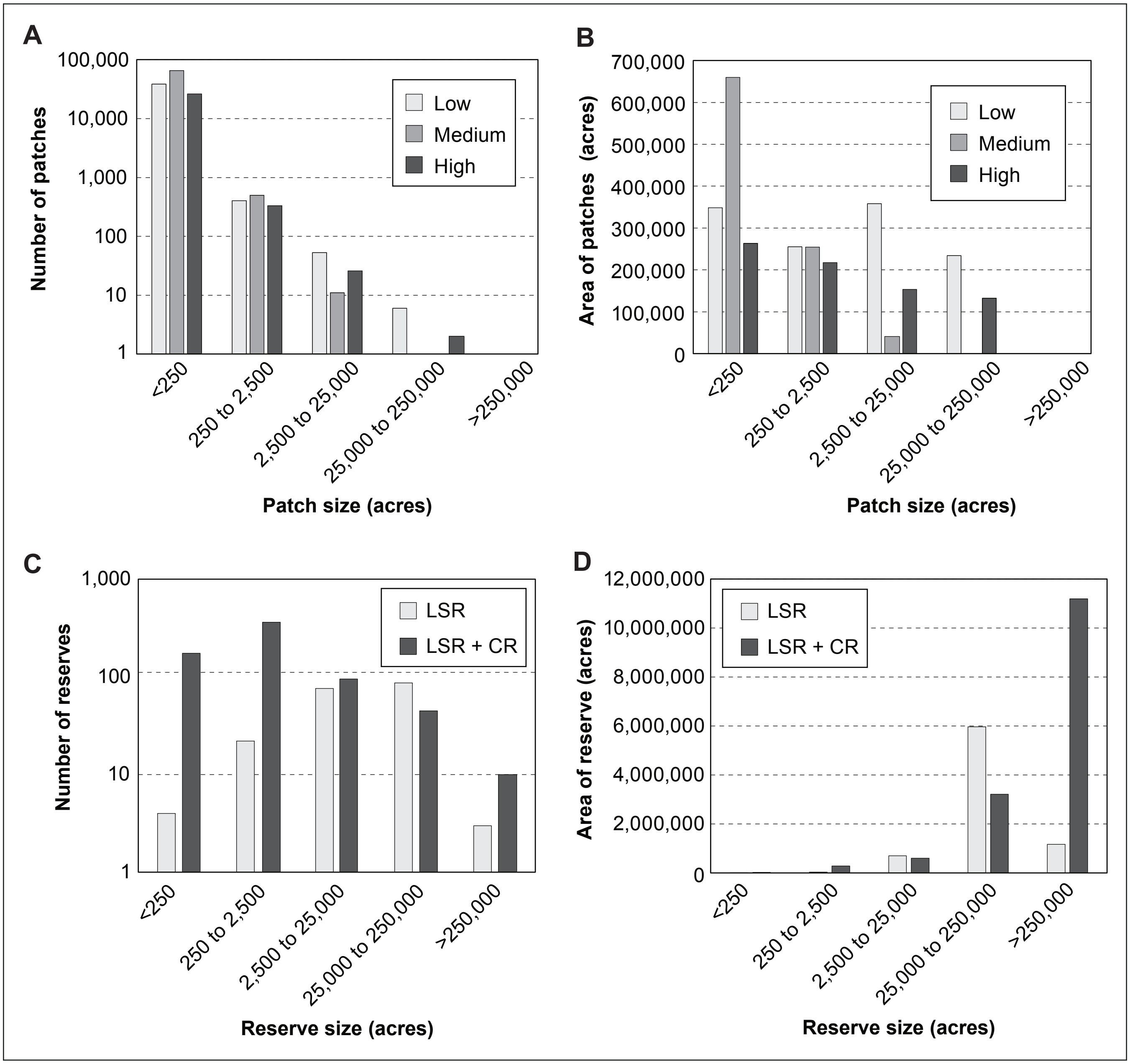


Figure 3-30—Patch distributions of recent fire (2000–2012) and sizes of Northwest Forest Plan (NWFP) reserves: (A) frequency distribution of number patches by fire severity and area class, (B) fire-severity patch sizes by area class, (C) frequency distribution of number of NWFP reserves (late-successonal reserves [LSRs] alone and LSRs plus congressionally designated reserves [CRs] by area, and (D) area of reserves by size class.

Reserves or protected areas are not necessarily areas where all human activities are excluded or are inconsistent with ecological conservation goals (Soule 1985). There are many types of protected areas with different degrees of human activity permitted (Spies 2006), including recreation areas, management allocations for degree and type of vegetation manipulation, invasive species removal areas, and fire management (prescribed fire or fire suppression) areas (Pressey et al. 2007). In most cases, including the NWFP standards and guidelines, biodiversity reserves permit and encourage restoration activities that further the species and ecosystem goals of the reserved area. For example, the

NWFP indicated that restoration activities within reserves were needed for both moist and dry forests (USDA FS 1994) in plantations in wetter and drier forests, and in older forests in fire-frequent regimes where forest structure and composition has been altered by fire exclusion and logging of older trees.

Wildfire and fire exclusion both pose serious challenges and dilemmas to managers seeking to conserve biodiversity using reserves or any other conservation approach (Driscoll et al. 2010, Fischer et al. 2006, Spies et al. 2012). This observation may seem contradictory or ironic, but it is the reality when conserving fire-prone forests in the Western United States. The multifaceted nature of wildfire makes it difficult to find a conservation and management "sweet spot." For example, fire is a vital and dynamic ecological process that maintains some communities, renews other communities, and increases plant growth and productivity (Ahlgren and Ahlgren 1960), but it also kills trees and destroys valued habitats, forest resources, and human infrastructure and lives (DellaSala and Hanson 2015). The assumption that reserves could conserve habitat for the northern spotted owl and other old-growth-associated species in dynamic ecosystems subject to fire, succession, and climate change was a major hypothesis of the NWFP. We examine this hypothesis below using data from the monitoring program (Davis et al. 2015) and new scientific knowledge.

Is the reserve system meeting the original goals of the Northwest Forest Plan?—

The reserve-matrix system was intended to protect and recover older forests in response to threats from logging and natural disturbances that destroy older dense forests. The general goal was to increase the amount of late-successional/old-growth forest in the reserves to recover toward levels that were present before extensive logging began on federal lands in the early 1950s. No specific targets for the future proportion of late successional/old growth in reserves were made in terms of HRV at the LSR scale, but the expectation was the amount of late successional/old growth in general on federal land would approach 60 percent over 100 years (Davis et al. 2015), including expected losses owing to wildfire. Dry zone forests were included in this rough estimate though the likelihood of achieving this goal was considered to be lower in dry forest zones than in moist forest zones (FEMAT 1993: fig. IV-3). It was expected that millions of acres of younger forests and plantations would eventually grow into an old-growth condition making up for any losses to wildfire or other disturbance agents. Between 1993 and 2012, disturbances, including wildfire and planned timber harvest, have reduced older forest (OGSI 80) area by 6.0 percent and OGSI 200 by 7.6 percent (Davis et al. 2015). Wildfire has accounted for the greatest reduction in older forest: annualized losses to wildfire were 0.22 percent and 0.28 percent for OGSI 80 and OGSI 200, respectively. In comparison, FEMAT (1993: IV-55) assumed that the annualized percentage of high-severity fire in reserves across all provinces would be about 0.25 percent over the first 50 years. At the scale of the entire NWFP, the losses from wildfire approximated expectations (Davis et al. 2015, FEMAT 1993) across the entire plan area (no projected losses were made by province), but losses from timber harvest were much less than planned.

The rates of change in OGSI 80 were not uniform across the physiographic provinces. Provinces with net declines that were higher than the regional averages are in order: Oregon Klamath (-9.9 percent), Oregon Western Cascades (-4.9 percent), and California Klamath (-4.1 percent).[15] Net change in OGSI 80 in eastern Oregon and eastern Washington Cascades, where wildfires have been relatively common (Davis et al. 2015), (table 3-6) were at or less than the regional average (e.g., -2.8 and -2.2 percent). While losses to fire and other disturbances get much attention, monitoring reveals that forest dynamics are also about succession, which will always at least partially offset losses: 757,900 ac (306 842 ha) of loss to disturbance appears to have been partially offset by 396,100 ac (160 364 ha) of gain from succession (Davis et al. 2015) (table 3-6). If losses from timber harvest are excluded (to highlight the role of natural disturbance agents), those losses (609,800 ac

[15] For OGSI 200, more physiographic provinces exceeded the regional average of -2.8 percent net change: Washington western lowlands = 7.0 percent; Oregon western Cascades = - 6.0 percent; Oregon Klamath = -10 percent; California Coast Range = -3.0 percent; California Klamath = -7.9 percent. table 3-8. From Davis et al. (2015).

[246 882 ha]) from all disturbance agents drop to 4.7 percent from 6.0 percent as gains from succession replaced about 65 percent of those over 20 years. Some provinces (e.g., Washington western Cascades, Oregon Coast Range, California Coast Range, and California Cascades) actually showed a net increase in OGSI 80 on federal lands (Davis et al. 2015) (table 3-6).

At the scale of individual LSRs, the range in net changes in OGSI 200 forests ranged widely (from -52 to >100 percent) (fig. 3-31) as would be expected for relatively small land areas. Most of the LSRs with the largest net changes are relatively small in area, with the exception of those in the Klamath regions of Oregon and California, where large patches of high-severity fire have occurred in the past 20 years. Three reserves in the eastern Cascades of Washington show relatively high rates of net loss, but all of these are relatively small reserves and the total net change in this province is about the regional average. The majority of the LSRs show little or no change. In general, large reserves have been more stable than smaller ones (fig. 3-32), which was why some of the largest reserves were drawn in fire-prone areas during FEMAT.

If rates of loss of dense old-growth were much higher, LSR function would be threatened because they were designed to be dominated by dense, complex older forests and serve as stepping stones for connectivity of old-forest species across the NWFP area. The loss of large areas of older forest in one or more of these reserves could challenge the connectivity design functions; however, no research has investigated the degree of change in the reserve network that might affect its overall function. At the recent rate of net change (-0.15 percent per year) (Davis et al. 2015) (table 3-6), the original matrix and reserve system appears sufficient to maintain areas of OGSI 80 at a regional scale, with greater declines (-0.23 percent per year) in the dry forests. This is especially so if it is assumed that the rate of ingrowth into denser older forest types will increase dramatically in coming decades as large areas of younger plantations and early 20th century wildfire-initiated stands begin to reach the age and structure where old-forests characteristics appear (Davis et al. 2015). However, the current trends may not hold given that fire activity is projected to increase across the NWFP area. With increasing drought fire sizes, including patches of high-severity, fire may increase (Reilly et al. 2017). Projections of the amount of increase in area or size of fires differ considerably across the NWFP area and among studies. For example, Stavros et al. (2014) found that the probability of very large fires will increase for Oregon and Washington, but increases would be minor in northern California. Littell et al. (2010) found that area burned is likely to increase by two to three times for Washington. Ager et al. (2017) modeled increases in fire and their effect on northern spotted owl habitat and fire regimes in the eastern Cascades of Oregon. They found that increases of two to three times in rates of wildfire would reduce spotted owl habitat by 25 to 40 percent within 30 years. They also found, however, that as fire increased, negative feedbacks on fire area and intensity occurred, suggesting that as fire increases, fuel limitations would affect future fire behavior. Most climate projection studies focus on area burned and not on severity and do not include fire feedbacks. Studies are needed to evaluate how climate change and fire might affect the LSR network conservation goals for different network configurations and management guidelines (e.g., levels and types of restoration).

While understanding annual rates of change in LSRs during the past 23 years is important to assessing Plan outcomes, it is also important to acknowledge that annual rates of disturbance or loss over short periods of time (e.g., 23 years) have limited value in the infrequent, high-severity regimes and across all regimes given climate change. Large fire or wind disturbances may be rare or episodic in infrequent regimes but can strongly control landscape dynamics and leave legacies that persist for centuries or longer (Foster et al. 1998, Spies and Franklin 1989). The real test of the reserve network can only be done over very long periods of time, and ultimately managers will have to be prepared for surprises and inevitable large events. Knowledge of trends and annual rates of change are useful but are of limited value for predicting the future in ecosystems, where fire, wind, volcanic eruptions, earthquakes, or invasive species can change forests rapidly over large areas.

The "losses" of late-successional/old-growth structure in reserves to fire may be a loss from the perspective of

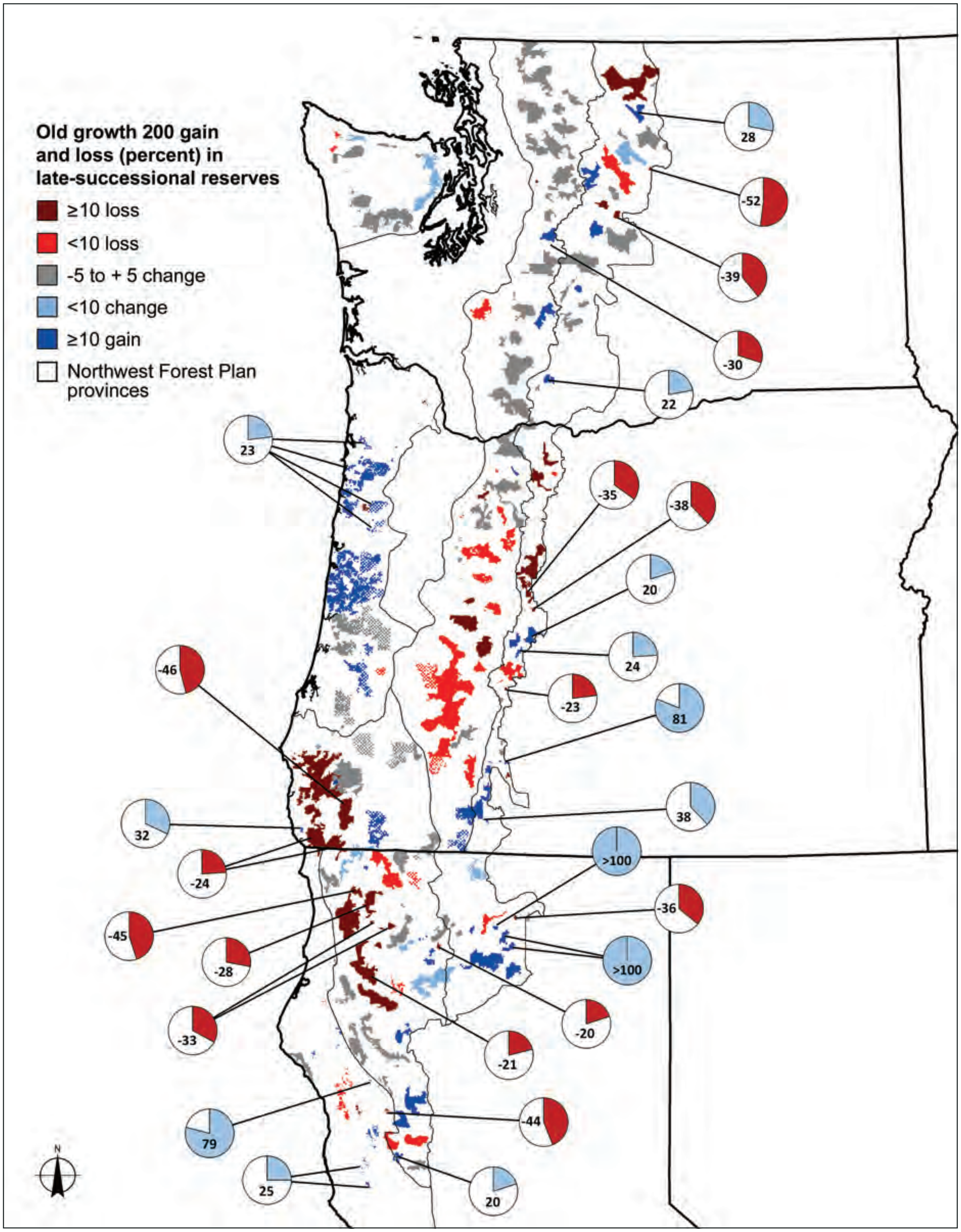


Figure 3-31—Map of 192 late-successional reserves (LSRs) in the Northwest Forest Plan area showing percentage of net change (gain or loss) in old-growth structure index (OGSI) 200 from 1993 to 2012. The LSRs are color coded by degree of gain (blue) or loss (red). The LSRs with little net change are shown in gray. Pie charts only show LSRs with greater than 20 percent net change (e.g., annualized rate of 1 percent), either gains or losses. Colored sections and numbers in pie charts indicate percentage of OGSI 200 in LSRs that was gained or lost. Percentages can exceed 100 percent where gains occur. Data based on Davis et al. 2015.

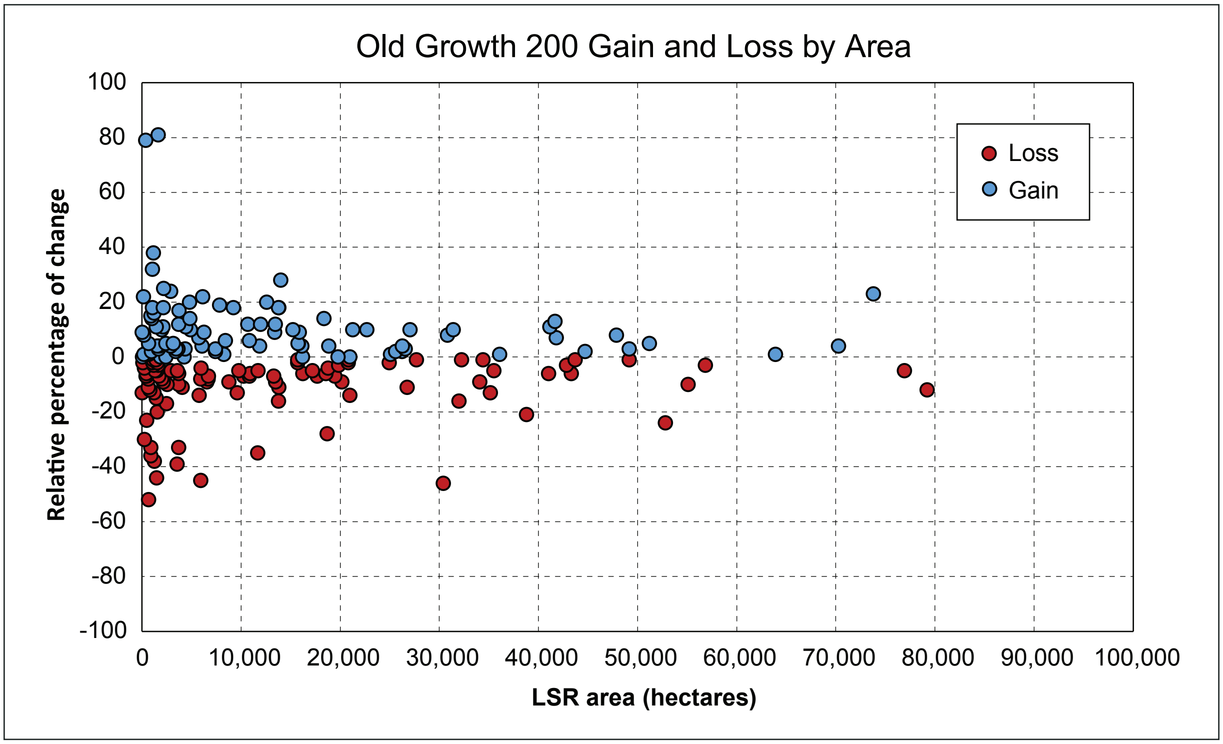


Figure 3-32—Relative change in old-growth strucuture index (OGSI) 200 in reserves between 1993 and 2013 in relation to late-successional reserve (LSR) size. Reserves smaller than 50,000 ac (20 224 ha) tended to show more change than larger reserves.

conservation of dense older forests, but they do not necessarily represent a loss from a broader biodiversity perspective (e.g., ecosystem integrity), especially where those fires burn at lower severities and thin out understories, leaving lower densities of fire-tolerant species. This is especially the case in dry forest landscapes, where open old growth and mosaics of old and early successional were characteristic. However, as mentioned above, the OGSI thresholds in frequent and very frequent fire regimes were based on plots from existing older forests that have been subject to fire exclusion and succession that would have increased stand density, layering, and amounts of shade-tolerant and fire-intolerant species. Hence, the reference conditions for older forests do not typically represent the older forest structure and composition types that developed under more frequent fire regimes. Large fires such as the 2002 Biscuit Fire often have less than 20 percent of their total area in high-severity patches and have large areas of historically moderate to low severity (Reilly et al. 2017, Thompson and Spies 2009). Lower and moderate-severity wildfire shifts stands from dense old forests to more open old forests (i.e., thins out understories but leaves many of the older fire-tolerant trees) that were characteristic of forest structure and composition under frequent fire regimes (Kane et al. 2013). However, monitoring and inventory definitions for these more open older forest types do not exist (Spies et al. 2006b, Taylor and Skinner 1998) and were not applied in the monitoring program.[16] Reilly and Spies (2015) classify forest structure in the NWFP area using existing inventory plots and identify conditions that may approximate the historical structure of more open old-growth forests. The lack of focus on open types of old growth was probably the result of the original emphasis of the NWFP on dense late-successional old-growth forest habitats of the western Cascades of Oregon and Washington which are associated with northern spotted owl and other species.

[16] The OGSI for pine types was based solely on density of large live trees, which may approximate historical amounts, but they do not include canopy cover and layering.

Concerns—

Although general trends revealed by monitoring at the regional scale appear consistent with NWFP goals and expectations, there are other less obvious trends that may be cause for concern in dry forests. First, in the Klamath Mountains and other regions, where chaparral and other shrub species are an important component of the vegetation, an increase in size and frequency of high-severity fire patches can lead to more extensive areas of early-seral or chaparral vegetation that can become a semipermanent landscape feature (Lauvaux et al. 2016, Tepley et al. 2017). It is not clear how much of this type of change would be desirable to meet ecological or social goals, and management may be needed to promote succession toward trees that are resistant to fire and climate change. On the other hand, Donato et al. (2011) suggested that low-density conifer regeneration in the presence of hardwoods and shrubs is an alternative successional pathway to promote early development of old, complex old-forest structure.

Very large patches of high-severity fire also occur in other low- and mixed-severity forest types in the NWFP area (Hessburg et al. 2016) with the possibility that recovery to forest is slowed or precluded as a result of lack of conifer seed rain (Dodson and Root 2013). This is especially in large reburn patches and may require planting to mitigate these effects (see restoration section below). The degree to which large patches of high-severity fire are slowing forest succession after recent large fires in the NWFP area is not known. On the other hand, relatively large patches of high-severity fire can result in areas of nonforest vegetation (e.g., grasslands and shrub lands) that were more common in the past than today in many dry forest landscapes (figs. 21, 22, and 26).

A second concern in dry forests is that older forests and landscapes in reserves and outside of reserves are slowly transitioning to conditions characterized by denser forests, more shade-tolerant species, buffered microclimate (less wind and shaded and cooler forest), and less flammable fuel beds. Thus, they become less likely to burn under low to moderate weather conditions and more likely to burn under high-severity conditions. Assuming continued fire suppression (Calkin et al. 2015, Stephens and Ruth 2005) and increased warming, the forests of the reserves in mixed- and low-severity regimes will continue to change in ways that do not support the historical dynamics of these forest types.

On balance, the science reveals that fire-dependent forests in LSRs are continuing to be squeezed into altered states and dynamics by two forces: (1) succession toward historically unprecedented structure, composition that affects biodiversity, landscape structure (e.g., larger more connected dense forest patches), and ecosystem function in absence of fire; and (2) a shift toward much less frequent but higher severity fire regimes as a result of fire exclusion, climate change, and changes in vegetation, including increased fuel loading and contagion. Losses of old growth and owl habitat to high-severity fire are the focus of the current monitoring reports and strategies, and succession toward dense forests with shade-tolerant species (e.g., owl habitat) is typically considered a positive outcome relative to the goals of the NWFP. However, within the dry forest zone and some drier parts of the moist forest zone, these types of forests are not a desirable outcome if the goal is ecological integrity based on frequent fire, open fire-resilient old growth, diverse successional conditions, and disturbance processes and landscape dynamics that maintain resilience and a full complement of native biodiversity. Landscape-scale research and strategies are needed to find options that provide for late-successional species while improving the overall resilience and functions of dry forests (Hessburg et al. 2016; Sollmann et al. 2016; Spies et al. 2006, 2017). Frameworks based on knowledge of ecological history or on NRVs or the HRV and departure from those references (Haugo et al. 2015) could be used to guide development and implementation of alternative approaches for dry forests to meet the goals of the NWFP and the 2012 planning rule. For more discussion of reserves and possible alternatives to static reserves, see chapter 12.

Connectivity and Fragmentation

Connectivity and fragmentation of mature and old-growth forests were important considerations in developing the NWFP (FEMAT 1993). The spatial pattern, size, and isolation of habitat patches of older forests can affect species richness, population dynamics, as well as the spread of fire and other disturbances. Davis et al. (2015) found that older forests on federal lands have become slightly more fragmented by disturbance over the period of the Plan. However, this analysis only takes into account late-successional and old-growth conditions and does not factor in changing connectivity relations over the remainder of the landscape, which may be the larger story. Consequently, it is not clear what the cumulative ecological effects (e.g., species richness, microclimate) of spatial pattern changes have been as a result of disturbance and succession over the past 20 years. It is now recognized that the ecological effects of spatial pattern of vegetation types and successional stages (e.g., edge effects, patch size effects, connectivity) differ with species and processes and are difficult to generalize about using a coarse-filter approach (Betts et al. 2014). Cushman et al. (2008) found that maps of existing forest cover types and successional stages in the Oregon Coast Range were not effective in estimating abundances of breeding birds and cautioned that maps based only on coarse vegetation classes may not provide a good metric of species abundance. If maps of vegetation types have limitations for conservation, then the analysis of spatial pattern is also likely to have limited value for predicting community or species outcomes. Fahrig (2013) has recently hypothesized that habitat amount is a better predictor of species richness than patch size and isolation for community-scale (i.e., coarse-filter) approaches to conservation. However, this does not mean that patch size, isolation, and connectivity are not important components of habitat at the scale of individual species (e.g., fine filter) or for key processes. The implication for the NWFP is that patch size and connectivity concerns are best dealt with at the individual-species scale (e.g., northern spotted owl, carnivores) or processes (e.g., fire spread through landscapes). The question of connectivity for late successional/old growth as a coarse-filter metric and even use of maps of late successional/old growth to represent “habitat” in general (e.g., concern of Cushman et al. 2008) is an area of uncertainty and needs research. See chapter 12 for more discussion of regional-scale issues.

Restoration Approaches

Here we address our scientific understanding of management actions that could be used to achieve goals for ecosystem restoration, especially those related to successional diversity and natural disturbance regime processes. We use a loose definition of restoration given that climate, landscape, and species changes make it from difficult to impossible or perhaps undesirable to really restore the structure, composition, and function of past ecosystems (Spies et al., chapter 12). Ecological restoration has been defined as *“the process of assisting the recovery of an ecosystem that has been degraded, damaged, or destroyed” (*http://www.ser.org/resources/resources-detail-view/ser-international-primer-on-ecological-restoration*).* Despite the limitations of restoration, management can promote resilience of ecosystems to fire or climate change or increase vegetation diversity that has been lost as a result of management actions such as timber management or fire suppression. Restoration may be able to promote some of the features of the pre-Euro-American period (e.g., dead wood, large fire-resistant trees, or multistoried old-growth habitats), but ecosystems may not have the same overall structure and function (or even fall within their historical ranges) as those of the pre-Euro-American period. We address these management actions by forest zone and disturbance regime, acknowledging that these ecological management approaches may be similar across regimes. Numerous authors have addressed restoration needs specified in the NWFP (Baker 2012; Franklin and Johnson 2012; Franklin et al. 2008, 2013; Haugo et al. 2015; Hessburg et al. 2016; North et al. 2009, 2012; Stephens et al. 2009; Stine et al. 2014). In general, these restoration needs are to restore disturbance processes (e.g., fire) and longer times for natural succession to operate without disturbance (Haugo et al. 2015) as young forests develop following logging (table 3-5).

Moist forests—

***Stand scales*—**Forest plantations are the primary focal point of restoration in these forests. Approaches to restoring old-growth forest conditions in plantations include:

- Passive management—increasing the amount of older forests by electing to simply allow younger postlogging forests to naturally progress, through growth and mortality to older life stages (Haugo et al. 2015).
- Active management—using variable-density thinning (restoration thinning) (Carey 2003, Churchill et al. 2013, Haugo et al. 2015, Muir et al. 2002) to increase structural and compositional diversity in unnaturally uniform plantations that reduced typical shrub and herb layers and accelerate development of future mature and old-forest structures (figs. 3-33 and 3-34).

Currently, the most common approaches are to allow younger stands to age and mature on their own and to use variable-density thinnings (i.e., restoration thinning) to increase habitat diversity within uniform plantations (especially 30- to 80-year-old stands, where thinning is typically profitable) and thus accelerate the development of older forest structure and composition (Carey 2003) (figs. 3-33 through 3-35). They can also be used to promote elk habitat, huckleberries, and other species associated with forest openings (chapter 11). While restoration thinning is a relatively new practice for ecological goals, the effects of standard thinning (Tappeiner et al. 2007) on tree growth and mortality in regular-spaced plantations are relatively well known. For example, growth-growing stock relationships for Douglas-fir suggest minor differences in stand volume growth over a range of residual densities (Marshall and Curtis 2002), which provides some flexibility in terms of thinning prescriptions (Dodson et al. 2012). However, extremely low residual densities and gap creation obviously lead to lower stand-level tree growth. However, where stand-level foliage biomass is concerned (which is important for tree growth and litter production), thinning can stimulate growth of foliage biomass on a branch and tree scale, which may not be a desirable outcome from a restoration perspective where reducing canopy fuels is a goal (Ritchie et al. 2013a). Decreases in stand growth owing to low tree numbers are partially offset by better growth of residual trees (Dodson et al. 2012), and by establishment and growth of regenerating trees.

Given the recency of restoration thinning practices and studies, our understanding of how this practice affects older forest development is based on only short-term results (typically less than 20 years) (Poage and Anderson 2007). To understand possible ecological effects, we extrapolate from the many studies of standard thinning operations, which suggest that such approaches would not produce many of the outcomes associated with old-growth forests (e.g., spatial heterogeneity, large dead trees, compositional diversity) in the short term (up to 50 years), other than larger diameter trees (Anderson and Ronnenberg 2013).

In contrast to standard thinning operations, restoration thinning includes preferentially retaining minority species and creating a wider range of density conditions from open gaps to unthinned patches of various sizes (Carey 2003, Davis et al. 2007, Neill and Puettmann 2013). This appears to be key to increasing the heterogeneity in thinned stands and accelerating development of late-successional elements (Anderson and Ronnenberg 2013, Cissel et al. 2006, Poage and Anderson 2007). Also, the initial responses to variable-density thinning treatments suggest that not all structural components and processes react in synchrony (Puettmann et al. 2016). For example, one study found that after a brief delay, likely due to increases in crown size (Ruzicka et al. 2014), restoration thinning led to an increase in average-tree-diameter growth. However, larger trees, which would likely become the dominant trees that are the major features of an old-growth stand, barely responded unless they were growing in extremely low densities, e.g., adjacent to gaps (Davis et al. 2007, Dodson et al. 2012). Also, diameter growth responded rather quickly within the first 5 years, while changes in other vegetation components were slower or delayed, such as in crown structures (Davis et al. 2007, Seidel et al. 2016) or bark furrows (Sheridan et al. 2013). That study also found that other vegetation components followed a counterproductive trend relative to late-successional/old-growth biodiversity goals. For example, the shrub layer was knocked down during harvesting operations and did not recover to

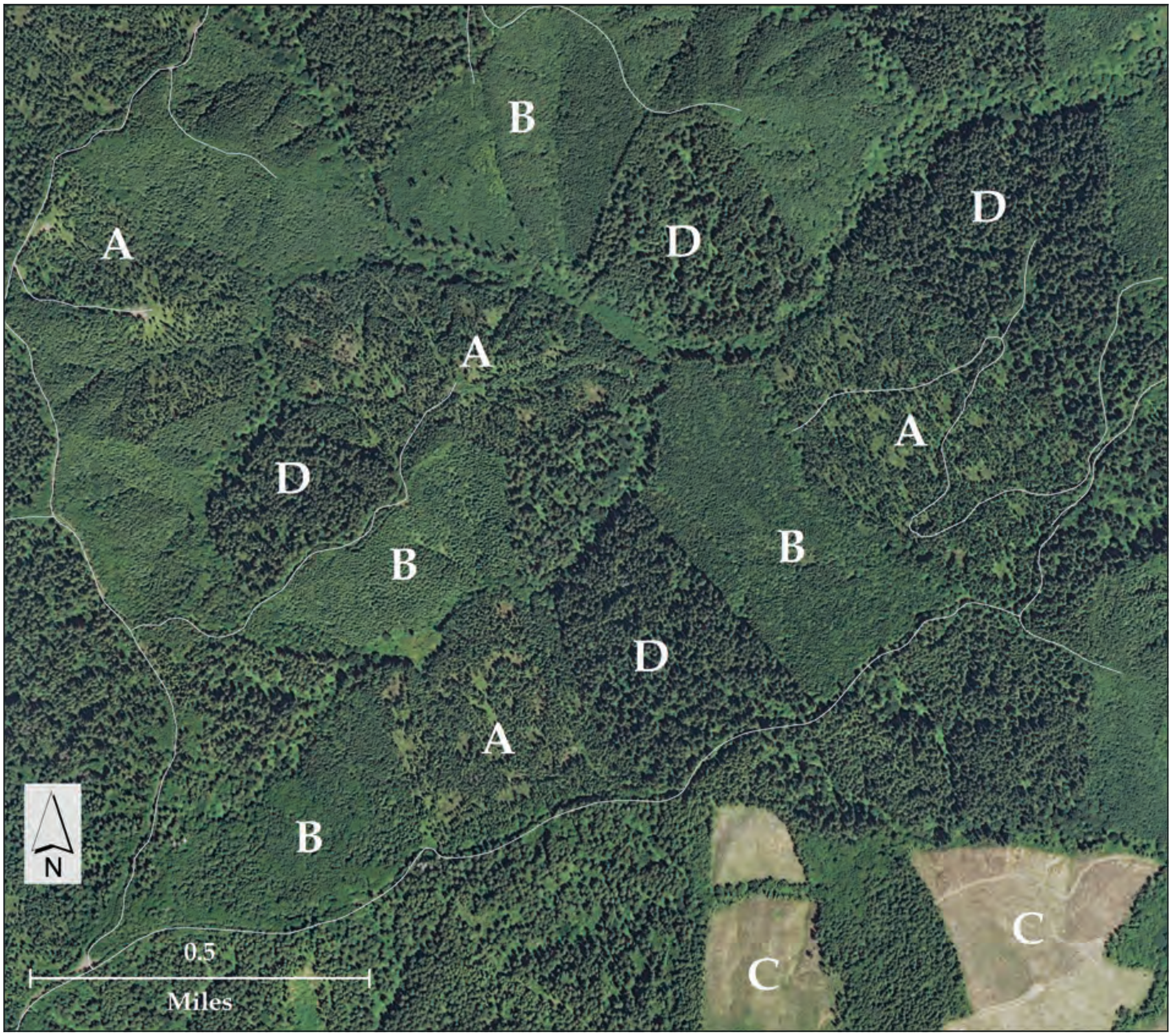


Figure 3-33—Aerial image from 2011 of management units and unmanaged stands in an area of late-successional and riparian reserves and matrix allocation on the Siuslaw National Forest, and private lands in the Oregon Coast Range: (A) plantations treated with variable density thinning, (B) uniform plantations that have not been thinned (these plantations are younger than those that have been treated), (C) recent clearcuts on private land, and (D) older naturally regenerated forests that have not been managed. Note areas of hardwood and shrub gaps in the older conifer forests that occur in root rot (*Phellinus sulphurascen*s pockets). Roads are indicated by white lines. From Oregon Explorer Natural Resources Digital Library.

preharvest levels within the first decade (Puettmann et al. 2013). Also, the understory vegetation composition shifted toward a higher component of early-successional species. This trend started to reverse within a decade (Ares et al. 2009, 2010) but was still detected 20 years after a precommerical thinning (Lindh and Muir 2004). Exotic species remained a minor component after restoration thinning and showed a similar trend of decline after a decade. With little postharvest mortality after thinning, snag recruitment was reduced 11 years after thinning (the time of the last measurement) (Dodson et al. 2012) and likely in the longer term as well (Garman et al. 2003, Pollock and Beechie 2014). This trend can be counteracted by creating snags (Lewis 1998); however, if this is done during restoration thinning, these snags would be smaller and shorter than in older stands. Alternatively, leaving untreated patches of high tree density ensured that competition-related mortality continued, allthough this led to snags at the

Sam Chan

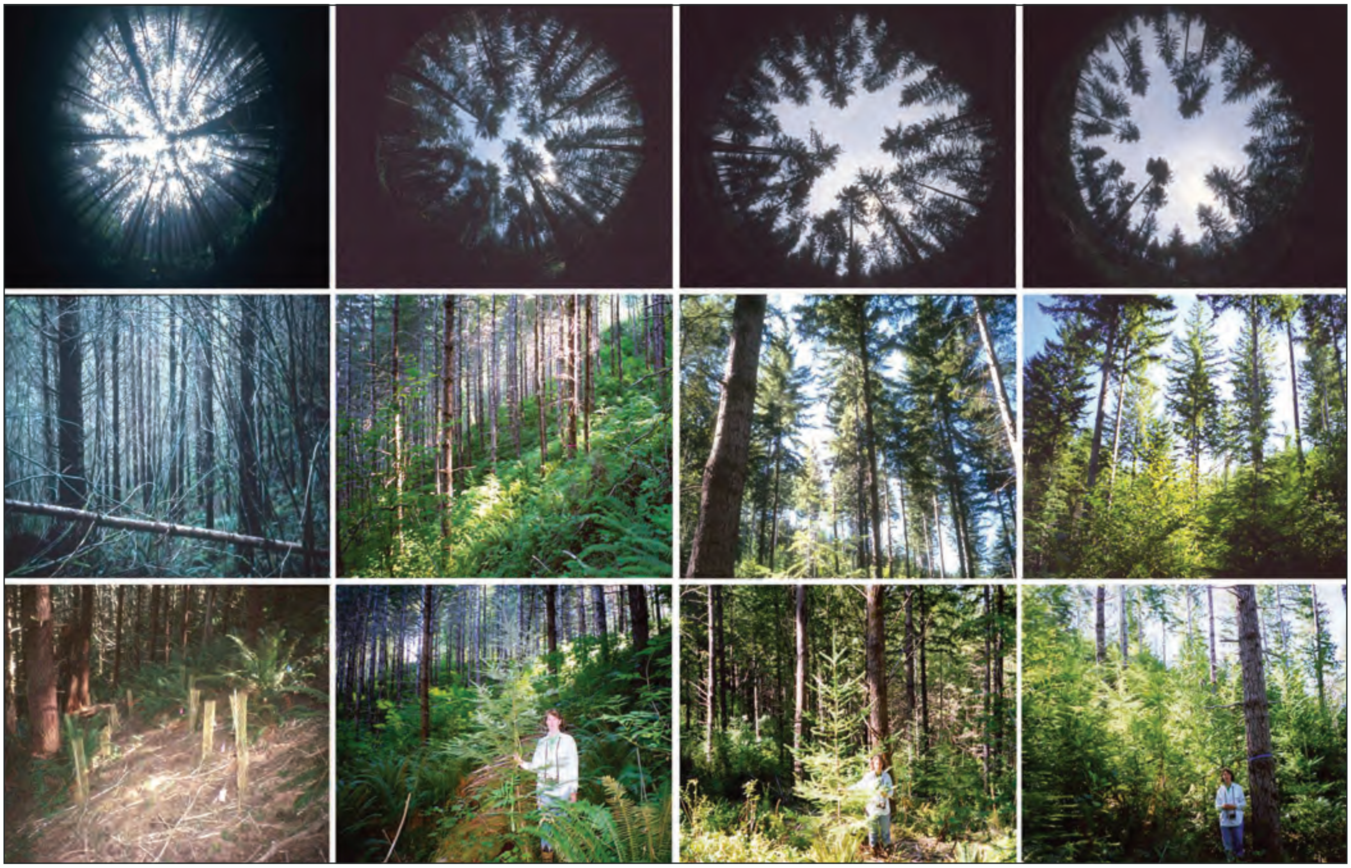
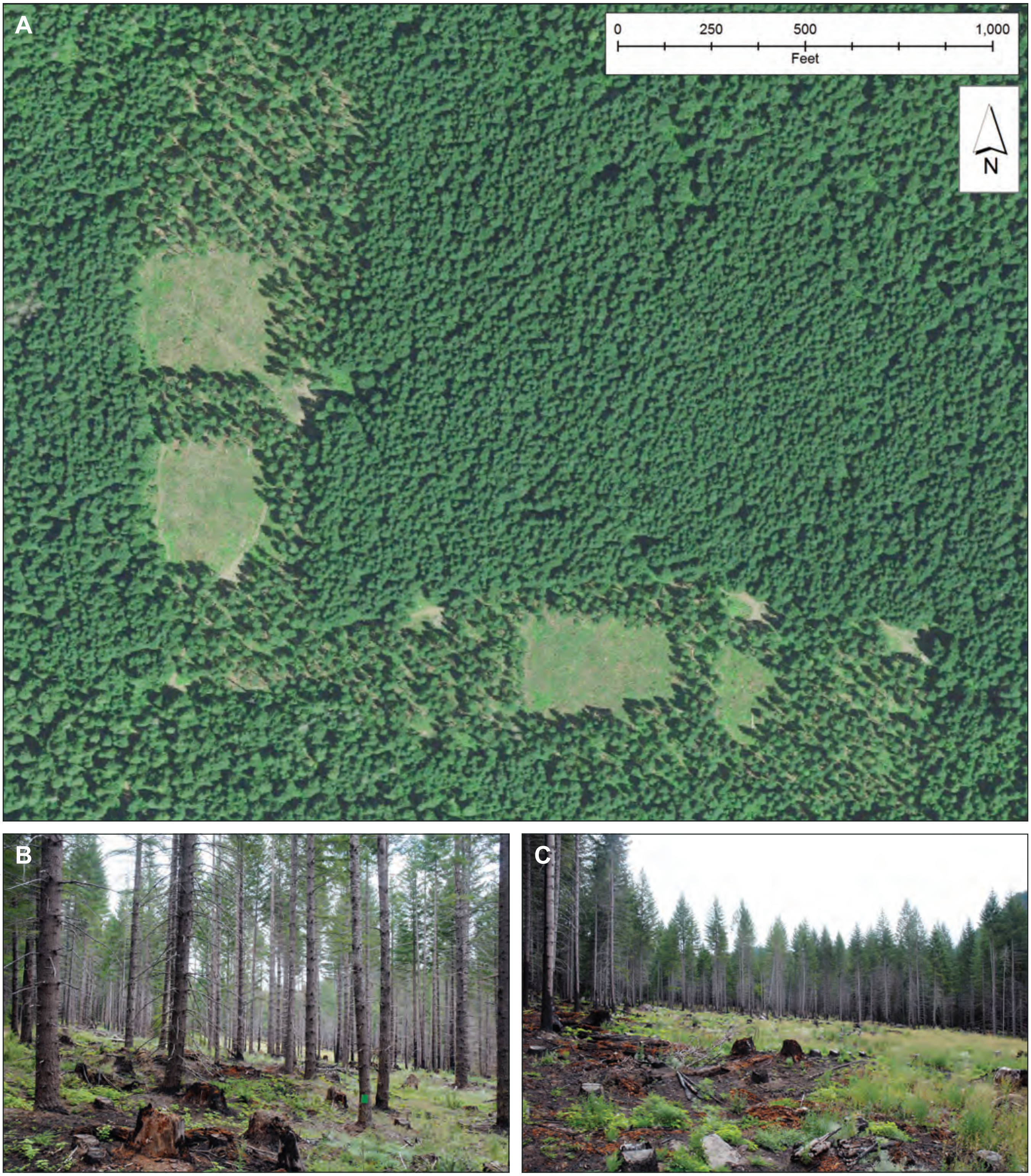


Figure 3-34—Canopy (fisheye) and understory photographs of unthinned and thinned 30 to 40 year old plantations of Douglas-fir on the Siuslaw National Forest. Densities of four stands from left to right: unthinned; 100 trees/acre; 60 trees/acre; and 30 trees/acre.

smaller end of the size distribution (Dodson et al. 2012). Tree regeneration typically increased right after restoration treatments (Dodson et al. 2014, Kuehne and Puettmann 2008, Urgenson et al. 2013), showing three general trends. First, while stand-level differences were obvious, studies showed very high spatial variability at small spatial scales. Second, seedling establishment increases after thinnings, but densities appeared to be similar, regardless of thinning intensities. Third, seedling and sapling growth differed by species and responded to higher degrees of overstory removal (e.g., Shatford et al. 2009).

The benefits of restoration thinning relate as much or more to increasing spatial heterogeneity as to reducing density per se, as high-density patches are not uncommon in natural stands. For example, Spies and Franklin (1991) reported that stand densities (trees >2 inches [5.1 cm] diameter at breast height) in young stands (40 to 79 years old) that regenerated naturally after wildfire in western Washington and Oregon averaged about 400 stems per acre (1,000 stems per hectare.). Some plantations 40 to 60 years old that regenerated naturally after logging (Curtis and Marshall 1986) or following clearcutting and planting can have similar densities, though plantations with much higher densities (e.g., 800 stems per acre [~2,000 stems per hectare) occur.[17] In some places, natural regeneration (e.g.,, western hemlock) will establish itself in Douglas-fir plantations (Puettmann, personal observation) leading to extremely high tree densities. While average tree density can be high in plantations, density differences do not explain all potential differences between natural young stands and plantations. The differences are also

[17] Pabst R. Personal communication. Senior faculty research assistant, College of Forestry, Oregon State University, Corvallis, OR 97331.



Google Earth™

Gary Rost

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Figure 3-35—Example of variable-density thinning from 2013, including skips and gaps (1 to 2 ac [0.40 to 0.80 ha]), in a 56-year-old plantation on the Willamette National Forest: (A) the pattern across the entire treatment area and the surrounding unthinned plantation, (B) a view from inside the thinned area, and (C) the view looking across the gap. The goal was "volume production, promotion of high-quality elk forage in the short term, while encouraging development of elk-optimal cover."

expressed in spatial variation in density and variability of tree age and size (Tappeiner et al. 1997). The age ranges and spatial heterogeneity of trees in naturally regenerated stands may lead to greater variability in canopy differentiation than would occur in plantations where trees are the same species, the same age, and are planted with uniform spacing (Oliver and Larson 1990). A combination of tall shrubs, hardwoods, or other vegetation would have occupied much of the open growing spaces (i.e., spaces not occupied by conifer regeneration in naturally regenerated stands). The short-term effects of variable-density thinning aimed at improving longer term structural and compositional diversity may be to fragment canopies and root systems and temporally reduce habitat quality for animal, plant, or fungal species keying in on canopy and root structure (Davis and Puettman 2009, Pilz et. al. 2006). This is an important issue requiring more research. Alternative ways of implementing thinning prescriptions (e.g., leaving larger unthinned areas or thinning very young stands) may actually improve conditions for lichens (Root et. al. 2010) and may help to mitigate some of the short-term negative effects of discontinuous forest canopies on canopy species (Wilson and Forsman 2013).

Empirical studies are critical, but evaluating long-term and landscape-level effects of variable-density thinnings requires landscape simulation models. Traditional growth and yield models provide fairly reliable information about tree growth for more or less evenly spaced, even-aged Douglas-fir plantations (Fairweather 2004). Most models assume the absence of disturbances, but ongoing efforts include a better representation of disturbance (e.g., insects and pathogens) on tree and stand growth (Crookston and Dixon 2005). Predictions for open or irregular-spaced conditions (Lord 2005) and growth of other species are less reliable or missing (Gould et al. 2011, Kuehne et al. 2015, Weiskittel et al. 2007). Similarly, there is a broad understanding and agreement about general trends, e.g., in understory vegetation, but specific dynamics cannot be modeled with high precision because they are based on interactions of initial conditions, species traits, local environmental conditions, and stochastic events (Ares et al. 2010, Burton et al. 2014), which may vary over time (Thomas et al. 1999) and space (Burton et al. 2014, Chen et al. 1992).

In the few modeling studies (Garman et al. 2003, Pollock and Beechie 2014), thinning promoted the development of large boles, vertical diversity, and tree-species diversity over 100+ years, compared to controls. At the same time, less dead wood was produced over many decades compared to no thinning, highlighting that at least some of the early trends found in the experimental studies (e.g., Dodson et al. 2012) may last longer. As mentioned above, the negative effects of thinning on deadwood production can be countered by creating snags (Lewis 1998) or leaving cut trees on the sites where they can immediately contribute to terrestrial and ecological functions (Huff and Bailey 2009, Walter et al. 2005).

Thinning has variable effects on wildlife and plant communities. In the short term, it can increase species diversity and abundance of some species, especially those associated with more open forest conditions (Ares et al. 2009, Berger et al. 2012). This can lead to increased flowering and seed productions, i.e., provision of food resources for selected insects, mammals, or songbirds (Neill and Puettmann 2013, Wender et al. 2004). The response of songbird populations showed similar trends (Hagar et al. 2004), but responses appear to vary by species and over time (Yegorova et al. 2013). Thinning may also attract avian predators that prey on marbled murrelet (*Brachyramphus marmoratus*) nests (chapter 5).

Although general stand-level trends from restoration thinning are mostly understood, uncertainties remain. For example, vegetation development for specific locations appears partially unpredictable for several reasons, including microclimatic conditions, initial variability in plantations, and stochastic events such as seed crops, disease, and windthrow (Dodson et al. 2012, Lutz and Halpern 2006). In addition, there are important effects of thinning on residual trees, such as harvesting damage to residual trees. Damage is typically higher the more wood is harvested and often concentrated near skid trails (Han and Kellogg 2000). Through careful layout and logging (e.g., Picchio et al. 2012) and avoidance of early summer harvests, damage can be reduced to levels that are not likely to affect future health of Douglas-fir stands (Bettinger and Kellogg 1993, Kizer et al. 2011). However, other species such as western hemlock may be more affected (Hunt and Krueger 1962). With proper logging layout, techniques, and timing (e.g., avoidance of wet soil conditions), the impact of thinning operations on soils should be limited to removal of humus and upper soil layers (Froehlich et al. 1981). However, these impacts that are concentrated near or in skid trails are only temporary as patches of exposed soils are reinvaded quickly.[18] In this context, harvesting operations that removed limbs and crowns before skidding (and in some cases limited maximum log length that could be skidded) not only scattered down wood throughout the stand, but led to lower soil damage, as well as lower damage to residual trees (K.J. Puettmann, personal observation).

In summary, ecosystem dynamics after restoration thinning are generally predictable, but specific responses can be highly variable owing to small-scale variability in environmental conditions and initial vegetation composition. In addition, other factors, such as weather patterns; seed availability; impacts of insects, diseases, and herbivores on seed or seedlings; as well as harvesting impacts as described above, suggest that restoration treatments are not likely to hit any specific target perfectly in terms of vegetation conditions and dynamics. Instead, restoration efforts may be better off acknowledging these inherent uncertainties by setting structural goals that allow for a range of conditions; e.g., between 10 and 30 percent of the restored area should have regeneration at a density from 50 to 500 trees per acre. Similarly, rather than locking in a spatial layout of prescriptions, any treatment prescription that can accommodate already existing variability within the homogenous stands that are to be restored will likely be more efficient at increasing heterogeneity in that stand (Puettmann et al. 2016). For example, a goal to provide more broadleaf shrubs and trees may be achieved more easily with prescriptions that protect existing patches of broadleafs during harvesting than by creating open conditions that facilitate their development (Davis et al. 2007). Similarly, the provision of snags may be more efficient if it accounts for the harvesting damage to residual trees. Finally, flexibility in restoration prescriptions and adequate monitoring is key to efficient and successful operations.

[18] Unpublished data. On file with: K.J. Puettmann, Oregon State University, Forest Ecosystems and Society, 301L Richardson Hall, Corvallis, OR 97331.

Landscape scale—Landscape-level effects of restoration thinning are not well-studied, and experimental studies are very difficult at this scale. In a simulation study, thinning in plantations on federal ownerships increased habitat for olive-sided flycatchers (*Contopus cooperi*) but had only a slight or no effect on total habitat for northern spotted owls and other associated late-successional species (Spies et al. 2007a). The lack of effects on habitat of owls and other late-successional species was probably due to several factors, including a relatively short simulation period (100 years) compared to the several hundred years needed for old growth to fully develop. Also, the thinning prescriptions were conservative, the number of thinned trees retained for dead wood recruitment was fairly low, and the proportion of landscape thinned in the first 10 years was limited to less than 8 percent of the entire federal landscape (Spies et al. 2007a). The scope of landscape-scale restoration benefits is also limited by the state and rate of succession in the population of plantations. While young plantations cover up to 30 percent of federal forest ownerships, not all of them have the structure (high density of small and relatively young conifers) that would benefit from restoration thinning. Also, even with increased resources, it likely will take decades to treat an area that is sufficiently large enough to have a major landscape-level impact, especially as some of the ecological benefits do not show up instantly but develop slowly over time. Lack of information about the structural and compositional conditions of plantations (and location amount of restoration treatments) as well as limited understanding of the importance of fragmentation and connectedness across the region limit our ability to assess restoration needs and potential at landscape scales.

A byproduct of any large-scale restoration program is the need to maintain or even increase infrastructure. Road systems and associated travel, which are needed for various management objectives, have also been shown to negatively affect terrestrial and aquatic biological diversity and ecosystem processes (Forman and Alexander 1998, Trombulak and Frissell 2000) by serving as travel corridors for invasive species (Parendes and Jones 2000), for example. Consequently, scientific reviews note that reducing roads through decommissioning is important for meeting many biodiversity goals (chapter 7) (Franklin and Johnson 2012, Trombulak and Frissell 2000).

The 80-year rule—Under the NWFP, harvesting for any goal, including thinning for old-growth restoration, is generally restricted in moist forests in LSRs to stands less than 80 years old (USDA and USDI 1994: c-13) (though some exceptions may occur). The NWFP allowed management in stands >80 years old in the matrix lands. This 80-year rule for LSRs is a one-size-fits-all approach that does not take into account that stand age is only a rough proxy for stand structure and development potential, both of which can differ greatly based on site conditions and history (Pabst et al. 2008, Reilly and Spies 2015) (fig. 3-15). That said, in general, treatments of stands >80 years old are not expected to result in substantial short- or medium-term shifts in developmental trajectories, as characterized by size and shape of trees and crowns, because trends established early in a tree's life are not easily reversed (Wilson and Oliver 2000). Understory vegetation would be more responsive. In that context, restoration thinning to promote development of complex older forest structure (e.g., large live and dead trees in stands >80 years old) of moist west-side forests is less likely to have large benefits for development of old-growth forests in the long term compared to younger forests, as many stands around age 80 begin to have some characteristics of older forests (Spies 1991, Spies and Franklin 1991) (fig. 3-15).

Our scientific understanding of the ecological effects of restoration thinning in older forests has not changed much since the early 1990s, as few empirical studies and modeling of management in older forests have been conducted (see Cissel et al. 1999 for a landscape-level modeling study). Removing larger trees could have negative impacts on the number of large live and dead trees, as trees over this age are often beginning to function as habitat for late-successional species in middle-aged stands; e.g., they develop bark characteristics that may act as microhabitat for a variety of species (Sheridan et al. 2013). However, the age, or better, the set of structural conditions (e.g., density, spatial pattern, size distribution) at which such negative impacts become important will differ with tree, stand, site, and landscape conditions, and such relationships have not been quantitatively tested. Research and adaptive management studies are needed to test and evaluate the alternative approaches and assess the relative benefits and tradeoffs of restoration thinning in forests >80 years old.

***Fire and early-successional vegetation*—**Possible activities relative to restoring or emulating the beneficial effects of wildfire in moist forests include creating early-seral forest and creating some of the effects of partial stand-replacement fire that were common in mixed-severity regimes of the drier part of this region. There is relatively little research and management experience with either of these activities. Managing wildfire to promote desirable fire effects may be increasingly feasible in the dry forests and remote areas of the wetter forests. However, relatively little is known about public perceptions of risk in moist forests and their willingness to tolerate wildfire in remote areas, but they do understand that any fire in moist forest is likely to be "catastrophic" (Hall and Slothower 2009). This leaves mechanical treatments and prescribed fire as the primary way to schedule and produce fire effects. The first problem in creating early-seral vegetation is determining where to create these habitats on a landscape that has already experienced a significant decline in old forests from clearcutting. Creating early-seral habitat from older forests is possible (Cissel et al. 1999, Hansen et al.1993) and would most closely mimic natural processes that have been disrupted; however, such treatments could also reduce habitat for at-risk, older forest species and have encountered public resistance (Franklin and Johnson 2012). Consequently, Franklin and Johnson (2012) suggested that forest plantations (<80 years old) be the primary focus of any efforts to create early-seral habitat. Heavy partial harvest (i.e., retention harvest), leaving dead trees and islands of live trees, and prescribed fire would constitute an approach to creating early-seral vegetation in plantations and create variable within- and between-stand patterns for late-seral development. Such efforts would be a compromise between how wildfires would have created such communities—they would lack large live and dead trees, might not have some of the same ecological effects of fire on soil surfaces and vegetation, and would not occur in very large patches—but they would restore some components and values of this ecosystem. Combining plantations into large groups would help address the patch size issue. A larger problem is how to determine how much of this vegetation should be created and how to schedule and distribute it in landscapes where wildfires could appear in any year and create thousands of acres of this vegetation type in a few days.

Moderately frequent mixed-severity fire regimes—Similarly, little published research exists on restoration in moderately frequent to somewhat infrequent, mixed-severity fire regimes, which occur in the drier parts of the moist forest zone (Tepley et al. 2013) (fig. 3-6). Managers have had some experience implementing treatments that attempt to emulate partial stand-replacement fire in older forests (fig. 3-29). Cissel et al. (1999) modeled stand and landscape management based on the mixed-severity fire regimes of the western Cascades of Oregon. They found that it produced more old-forest habitat and larger patches of older forests than would have occurred if the NWFP reserve-matrix strategy had been implemented as originally designed. However, it probably would have produced less older forest structure than if no timber harvests had occurred in the matrix and wildfire was suppressed. The broader ecological effects of mixed-severity fire in forests more than 80 years old have not been studied. One hypothesis is that some late-successional conditions (e.g., spatial heterogeneity, species cohort composition, diameter diversity and development of large-diameter trees) in the drier parts of the western hemlock and Pacific silver fir zones are no longer developing at the same rate because lower severity fire would have thinned the older stands, creating gaps, initiating new shade-tolerant cohorts, and accelerating growth of surviving canopy trees (Brown et al. 2013, Tepley et al. 2013, Weisberg 2004). In general, landscapes with more fire-severity diversity (“pyrodiversity”) (e.g., mixed-severity landscapes) are known to support more biodiversity (Kelly and Brotons 2017, Perry et al. 2011, Tingley et al. 2016). Landscapes with more vegetative diversity would likely affect the rate of wildfire spread and wildfires would create more heterogeneous vegetation. Research is needed to evaluate alternative approaches to restore successional diversity in this moist forest regime through mechanical treatments, prescribed fire, and wildfire.

Ecological forestry—The “ecological forestry” approach (Franklin and Johnson 2012, Seymour and Hunter 1999), which seeks to use knowledge of disturbance ecology and retention-based management to achieve ecological and commodity goals simultaneously, has been promoted as a restoration approach for meeting goals of the NWFP. It can be applied to both moist and dry forests and is, to some degree, a branding of a collection of management actions (including those already identified for moist and dry forests [table 3-5]) that can be applied to meet ecological and social goals. Ecological forestry encompasses restoration thinning in plantations, prescribed fire, and retention silviculture (focusing on what to retain rather than on what to remove) to create early-successional patches in plantations or older forests (e.g., >80 years old) where appropriate (figs. 3-35 and 3-36). The theory behind ecological forestry is supported by scientific understanding and rooted in established concepts in silviculture and ecology (Batavia and Nelson 2016; D’Amato et al. 2017; Franklin et al. 2007b, 2018; Seymour and Hunter 1999).

No published empirical research studies exist that evaluate long-term ecological and socioeconomic effects of ecological forestry in the NWFP area. However, several of its components, including retention silviculture and disturbance-based forest management, have been evaluated in the Pacific Northwest and other places with shorter term studies. For example, global studies (Baker et al. 2016, Gustafson et al. 2012) and work in the Pacific Northwest (Halpern et al. 2012, Hansen et al. 1995a, Urgenson et al. 2013) show that retention silviculture can provide habitat and “life boats” (i.e., refugia) for older forest species (Rosenwald and Lohmus 2008) within patches of early-successional vegetation. Cissel et al. (2002) simulated a landscape-scale design for a watershed in the western Cascades that contained many elements of Franklin and Johnson’s ecological forestry approach. They found that their approach produced better ecological outcomes than implementation of the current NWFP standards and guides; however, relatively little empirical research has been published on this issue in the NWFP area.

Batavia and Nelson (2016) recently criticized ecological forestry for its lack of a clear normative or ethical goal (e.g., conserve all species, or maximize timber production). They suggested that this deficiency will limit its practical application and subject it to the same social pitfalls as earlier and current management concepts or frameworks for finding solutions to balancing ecological and social objectives, such as "new forestry" (Franklin 1989), ecosystem management (Christensen et al. 1996, Grumbine 1994, Franklin 1997), or sustainable forestry (Lindenmayer and Franklin 1997). Different world views and values appear to present a major challenge to the implementation and acceptance of any of these approaches that attempt to achieve multiple goals from the same stands or locations. For example, DellaSala et al. (2013) criticized ecological forestry on federal lands as placing too much emphasis on timber production and not enough on protecting habitat for the northern spotted owl, especially given the threat posed by the barred owl (*Strix varia*). At the same time, Oregon county commissioners are seeking higher levels of timber production, especially from Bureau of Land Management lands, and complain that ecological forestry does not produce enough timber for local lumber mills (Hubbard 2015). Clearly, the social aspects of active management to restore or create desired ecological patterns and processes (in any of the disturbance regimes) and producing socioeconomic values are as important to consider as the biophysical aspects (see chapter 12 for more discussion of the tradeoffs and value issues).

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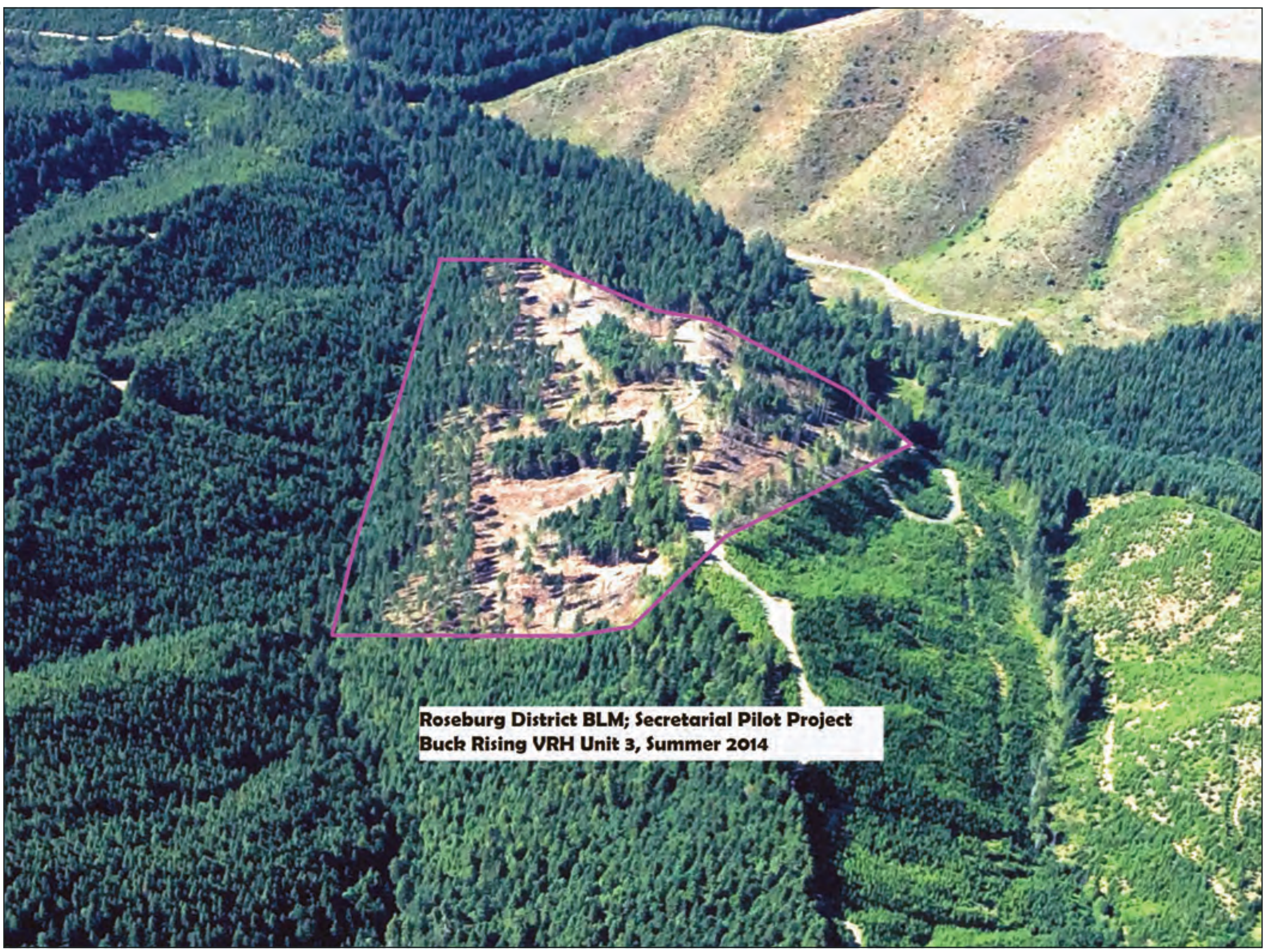

Figure 3-36—Management unit designed to create a mosaic of early habitat and leave trees, and produce wood from a young Douglas-fir forest on Bureau of Land Management (BLM) land in western Oregon. VRH = variable-retention generation harvest.

Dry forests with frequent, mixed-severity fire regimes—Restoration approaches in both fire regimes of the dry forests include mechanical treatments and use of fire in plantations and older forests to restore or create seral stages, surface fuel beds, forest density conditions, and spatial patterns of trees that are more resistant and resilient to fire and better adapted to warming climate. Restoration strategies for the frequent mixed-severity regime in the area of the NWFP have recently been summarized in Hessburg et al. (2016) who provide an indepth review. Restoration challenges are large in this regime because of the complexity of successional pathways and variable disturbance patterns. The management strategies outlined include:

- Restoring pyrodiversity at landscape levels through prescribed fire and managed wildfire.
- Creating and maintaining successional heterogeneity based on local disturbance regimes and the needs of late-successional forest species.
- Using topography to tailor restoration treatments across landscapes.
- Protecting and restoring large and old, early-seral fire-resistant trees.
- Restoring diversity to plantations.
- Creating and maintaining early-seral vegetation, including grasslands and shrublands.
- Mitigating threats from climate change, forest insects, and pathogens.

***Prescribed fire and wildfire*—**The literature on restoring forest fire regimes indicates that prescribed fires and wildfires managed under moderate conditions are vital components of ecological restoration. Thinning and other mechanical manipulations can achieve many structural and composition restoration goals. However, they cannot replace many important ecological processes and effects of fires, whether prescribed or wild (McIver et al. 2013). Fire, in particular, reduces surface fuels and coarse woody debris and can both increase and decrease snags and large-diameter logs depending on severity. Fire also affects soils (Certini 2005), insects (e.g., carabid beetle) (Niwa and Peck 2002), and other arthropod communities (Apigian et al. 2006). On the other hand, fires can also lead to increases of exotic plant species (Keeley 2000) and weaken high-value trees as well as attract bark beetles (Gibson and Negrón 2009). This may be viewed negatively in a narrow sense, but in a larger ecosystem context, such indirect impacts can feed a whole suite of ecosystems processes. For example, larger bark beetle populations can attract more woodpeckers that in turn spread more wood decaying fungi, thus providing more cavities, dead and down wood and associated habitat for a whole suite of species.

Prescribed fire is often implemented at least initially following variable-density thinning to reduce stand density. Here, thinning and prescribed fire can be implemented in denser stands with or without large fire-resistant trees. Such treatments can increase the range of microclimate and resource conditions (e.g., soil moisture, light) (Ma et al. 2010). For example, Dodson et al. (2008) found a neutral to positive treatment effect from thinning and prescribed fire on understory vegetation, while other studies showed a short-term decline followed by an increase (Abella and Springer 2015). The high variability of responses appear to reflect (among others) the variability in initial conditions and the scale of observation (Dodson and Peterson 2010), with areas of low understory richness benefiting most (Dodson et al. 2008). At the same time, such treatments would reduce the likelihood of very large patches of high-severity fires that are incompatible with ecosystem and habitat needs for many species (Harrod, et al. 2009, Hessburg et al. 2016, Knapp et al. 2012a).

Landscape-scale perspectives are needed to understand the potential effectiveness of fuel treatments in modifying fire behavior. Fuel treatments affecting a small area of landscape have a low probability of intersecting a fire, given the relatively low frequencies of fire in these dry forests under full fire suppression strategy (Rhodes and Baker (2008). To be effective, treatments need to be widespread enough to influence the current level of landscape inertia (see Stine et al. 2014), and then be allowed to interact more commonly with wildfire ignitions not influenced by suppression. Spies et al. (2017), using a landscape dynamics model, found that a doubling of rates of restoration in central Oregon, which is still a relatively small area compared to historical fire frequencies, led to only a small reduction in the mean occurrence of high-severity fire over a projected 50-year period. That study found that treatments were more effective in reducing high-severity fire years with more fire and that resilience of the entire landscape and the potential for high-severity fire was significantly lowered by higher rates of fuel treatment. Similar findings about the effectiveness of fuel treatments in altering fire outcomes have been reported by Loudermilk et al. (2013, 2014) for the relatively dry forests of the Lake Tahoe basin. Treatments to reduce density and surface fuels will need to be repeated at intervals that depend on the treatment intensity and productivity of the site (Collins et al. 2010). Given the widespread effect of fire exclusion, large areas will need to be treated (Hessburg 2016), which may be difficult for administrative and social reasons. Strategic spatial optimization of treatments can improve effectiveness per unit area treated (Finney et al. 2007), where prior commitments of land area to reserves or unique management allocations are minimal. Where major parts of the landscape are already committed to any management allocation that prevents optimal treatment allocation, spatial optimization efforts are essentially equivalent to random treatments (Finney et al. 2007)

Use of naturally ignited wildfires to achieve resource objectives is very important because, in most areas, current amounts of prescribed fire are too little to affect a sufficient area (North et al. 2012, 2015). Managing wildfire to promote ecological benefits is especially well suited for remote areas, with steep, complex topography, although it can become a more viable option in other landscapes when used in conjunction with prescribed fires, fuel reduction treatments, and footprints from past fires to create a patchwork that helps to contain the spread of natural ignitions to achieve desirable outcomes. Such fires will promote a high diversity of fire effects under moderate weather, including patches of low-, mixed-, and high-severity fires (Miller et al. 2012; Skinner et al., in press). Fire suppression and exclusion would also still be an important management tool, especially where dense older forest habitat conditions are desired, where landscapes may not yet be adapted for wildfire (e.g., contain many younger unthinned forests), or where human values are at risk from fire or smoke. Effectively managing wildfire depends on having moderate weather conditions that reduce the risk of high-severity fire effects (e.g., Estes et al. 2017). There are few published studies about restoring fire processes and structural diversity in older forests within the mixed-severity fire regimes in the NWFP area. However, examples exist from forests of the Sierra Nevada that are quite relevant to the dry forests of the NWFP area (Collins et al. 2006, 2008, 2010; North et al. 2009; North and Sherlock 2012; van Wagtendonk et al. 2012; Webster and Halpern 2010) and the Rocky Mountains (Holden et al. 2010; Larson et al. 2013; Parks et al. 2013, 2016). Among other things, these studies point out the importance of patch heterogeneity and topography as a driver in dry forest restoration.

Landscapes and resilience to climate change— Successional heterogeneity is a product of pyrodiversity and is fundamental to biodiversity and resilience of forests to climate change (Hessburg et al. 2016). This heterogeneity occurs across a range of spatial scales from tree clumps, patches and patch neighborhoods, to landscapes (Hessburg et al. 2015). Using variable-density thinning or varying prescribed fire treatments can promote heterogeneity at these fine scales (Churchill et al. 2013, Fry et al. 2014, Lyderson and North 2012). Developing landscape-scale prescriptions for use of thinning, prescribed fire, and managing wildfire can help promote landscape-scale heterogeneity. Landscape strategies are also important to maintaining and providing habitat for species that used dense, late-successional forests (Hessburg et al. 2015, 2016) or a mosaic of late- and early-successional forests (e.g., Franklin et al. 2000). Landscape-scale models and scenario analysis are needed to better understand tradeoffs associated with managing mixed-severity landscapes for a diversity of seral stages and biodiversity objectives (Lehmkuhl, et al. 2007, Roloff et al. 2005, Spies et al. 2017). Topography can provide a valuable template for implementing landscape strategies in mixed-severity regimes (Hessburg et al. 2016). Topography, whose patterns and effects differ regionally can be used to help set goals for seral stages and prioritize treatment locations (Lyderson and North 2012, Taylor and Skinner 2003).

Increasing resilience of forests to insects, pathogens, and drought can be accomplished through efforts described above related to managing for pyrodiversity, and successional diversity in a landscape context. Altering species composition can address a number of insect and disease concerns, including spruce beetle (*Dendroctonus rufipennis*), laminated root rot (*Phellinus sulphurescens*), and western spruce budworm (*Cholristoneura freemaii*) (Hessburg el. 2016). Thinning forests can lower the likelihood of mortality associated with mountain pine beetle (*D. ponderosae*) and western pine beetle (*D. brevomis*) (Fettig et al. 2007). Thinning can reduce dwarf mistletoe infestations. Strategies to increase resilience to climate include reducing surface and ladder fuels, reducing and maintaining lower tree densities, and restoring horizontal spatial heterogeneity in forest structure, including openings where early-seral species can establish (Churchill et al. 2013). Baker and Williams (2015) argued that efforts to remove most small trees may compromise resilience, because the presence of small trees can increase resilience to insect outbreaks, which can disproportionately affect large trees. They further argued that reducing stand density is not consistent with restoration of forests, because most dry forests were historically dense (based on their GLO survey, which overestimates tree densities as we discussed above). Allen et al. (2010) in a global review of drought-induced mortality found situations where mortality in forests increases with tree density as a result of increased competition, and situations where mortality was not related to density. Bradford and Bell (2017) examined thousands of forest inventory plots from the Southwestern United States and found that mortality during warm and dry conditions was related to basal area. Similarly, Guarin and Taylor (2005) found mortality associated with basal area and tree density in mixed-conifer forests of Yosemite. Both Allen et al. (2010) and Bradford and Bell (2017) suggested that thinning is one option for increasing resilience of forests to drought. Baker and Williams (2015) argued that forest resilience is a function of diverse sizes of trees and species, which is consistent with the literature that supports the idea that efforts to increase resilience should focus less on stand or landscape averages but focus on increasing heterogeneity and forest structure and composition at multiple scales (Hessburg 2016).

***Large, old, fire-resistant trees*—**The number of large, old, early-seral, and fire-resistant trees have been reduced in many areas as mentioned above. These keystone forest structures promote forest resilience to fire and climate change (Agee and Skinner 2005, Hessburg et al. 2016). Management actions for maintaining and promoting these trees include (1) identifying environments that support them; (2) protecting them from logging, crown fires, and drought stress; and (3) developing future cohorts through stand management practices (e.g., reducing stand densities and prescribed fire) that promote their regeneration, growth, and crown development.

***Plantations*—**Although plantations are a strong focus of restoration in the wetter forests, many thousands of acres of plantations also exist in dry forests landscapes that are in need of attention to promote resilience to fire and other threats. For example, precommercial thinning and prescribed burning can be used to reduce the near-term risk of loss of young, dense plantations to high-severity fire, while variable-density thinning can promote development of early-seral fire-resistant species where they are lacking in commercial-aged plantations (Stephens and Moghaddas 2005, Weatherspoon and Skinner 1995). Where desired species are lacking, planting may be needed (Hessburg et al. 2016). Where thinning is done, it will be important to treat surface fuels because logging slash will typically increase severe fire behavior in the residual stand (Huff et al. 1995, Raymond and Peterson 2005, Weatherspoon and Skinner 1995) unless trees are whole-tree yarded and slash piles are burned.

***Early-successional vegetation*—**To cover the full suite of landscape conditions found under natural conditions, restoration efforts in the mixed-severity regimes may also consider providing early-successional habitats (Haugo et al. 2015), as mentioned above (Hessburg et al. 2016). Collins et al. (2010) suggested that silviculture could be used to mimic stand-replacing fire patches in a portion of the mixed-severity fire regime landscape. Other restoration treatments in older forests would not be stand replacing but may be targeted to remove at least part of the vegetation that established after fire exclusion, thus improving growing conditions and vigor for dominant residual trees (Latham et al. 2002). We lack research that provides guidance on how to implement restoration for early-seral conditions at landscape scales given that wildfires will continue to create this vegetation type, but early-seral conditions may highly differ from those of historical conditions depending on the successional stage of the predisturbance forest. Collins et al. (2010) cautioned that silvicultural prescriptions may never achieve the complexity that freely burning fire can. Instead, allowing for more freely burning wildland fires would increase patch heterogeneity across landscapes and decrease potential for spread of very large high-intensity fires. However, cautions apply. Fires freely burning through dense layered stands produce very different fire effects in comparison to those where stands are open canopied and surface fuels are more limited (Miller and Urban 2000b).

Dry forests with very frequent, low-severity regimes— ***Management approaches*****—**The restoration needs and approaches (e.g., variable-density thinning, prescribed fire, and promotion of large fire-tolerant trees) in the very frequent, low-severity regime have many similarities to the frequent mixed-severity regime, but targets in terms of density, tree sizes and species, spatial patterns, and disturbance processes (e.g., frequent fire) are quite different. We emphasize some of the approaches that are unique to this fire regime. The overall needs for restoration in the very frequent, low-severity fire regime forests are larger given that fire suppression and widespread logging of large trees in many ecoregions has had a greater overall effect on forest structure and composition than in other dry zone forests; e.g., the larger number of fire cycles that have been missed owing to fire suppression.

Guidance for restoration of forests of this disturbance regime can be found in Franklin et al. (2008), North et al. (2009, 2012), Stephens et al. (2009), Franklin and Johnson (2012), Franklin et al. (2013), Stine et al. (2014), Haugo et al. (2015), and Hessburg et al. (2015, 2016). Strategies to restore old hardwood components of forests and woodlands are described for California black oak in Long et al. (2016), for Oregon white oak in Devine and Harrington (2006), and for riparian areas in southwestern Oregon in Messier et al. (2012). We summarize some of the recommendations from these publications below. A combination of harvesting and fire management is important to foster regeneration and development of large shade- and fire-tolerant canopy trees, associate understory and midstory vegetation, and to increase structural heterogeneity (e.g., areas of relatively open patches with large canopy trees). In forests that have become denser as a result of fire exclusion, the old-tree component is often diminished or absent. This is especially prominent in drier forest areas, likely owing competition from the higher number of younger, competing trees (Dolph et al. 1995, Ritchie et al. 2008). Restoration thinning that is aimed at improving growing conditions for the larger trees appears to reverse this process (Latham et al. 2002). Thinning stands for resilience to drought and fire will require very low densities, especially of small trees and shifting composition to fire- and drought-tolerant species (Churchill et al. 2013). Studies by Hagmann et al. (2013, 2014, 2017) provide snapshots of the structure of low-density pine forests in central Oregon. Where large trees are lacking, sufficient numbers of intermediate-size trees will be needed to produce future large trees (Ritchie 2005). Flexible tree size criteria for thinning are needed to remove relatively large shade- and fire-intolerant trees that have developed in the past century of fire exclusion. It will be important to treat fuels created by mechanical treatments to reduce the risk of high-severity fire. Thinning and fuel treatments and prescribed fire should seek to reintroduce spatial heterogeneity into stands and landscapes (Haugo et al. 2015, 2016). Prescribed fire should aim for low levels of canopy mortality (e.g., 5 to 10 percent) to promote snag recruitment and spatial heterogeneity.
In some cases, it may be necessary to plant drought- and fire-tolerant tree species. Topographic and soil patterns can provide a template for distributing treatments across landscapes (Hessburg et al. 2016, North et al. 2009). It will be important to consider understory plant communities in restoration plans (Franklin et al. 2013) as they have been severely degraded by grazing and are important for wildlife habitat, productivity, and providing fine fuels to promote the movement of low-severity surface fire through the landscape. For example, introducing prescribed fire after a long period of fire exclusion and accumulation of litter can lead to locally intense fires that still kill trees and rhizomatous grasses that are important for browse and form surface fuels that are needed to sustain relatively frequent surface fires. Other important considerations in restoration planning include developing efficient and effective marking guides that promote spatial heterogeneity (e.g., the individuals, clumps, and openings method) (Churchill et al. 2013, Franklin et al. 2013).

***Landscapes*—**Landscape-scale considerations are important for altering successional patterns, general resilience to drought and wildfire, and for providing habitat for wildlife species that depend on different types of habitat, including dense conditions that may not be resilient to fire. Where restoration actions such as thinning and prescribed fire are done, it will be important to treat large patches to reduce the likelihood that treated areas will be rapidly recolonized by shade-tolerant tree species and certain shade-intolerant trees (e.g., lodgepole pine) that seed-in from nearby untreated areas. The landscape inertia (e.g., mass effects) (Stine et al. 2014) created by large areas dominated by shade-tolerant tree species will be a major influence on the rate and potential for restoring successional dynamics in these landscapes. Patch types and sizes differ in their susceptibility to high-severity fires and considering their patterns and relative abundances in landscapes is critical for restoration planning in low-severity forests and in other fire regimes. The following patch types are listed from highest to lowest susceptibility to high-severity fire (Odion et al. 2004, Thompson and Spies 2009). Note that order is not necessarily the same as management priorities, which take multiple factors into account. Landscape context (e.g., edge effects, also can play a large role in determining fire severity (Weatherspoon and Skinner 1995):

- Young homogenous plantation vegetation without slash treatment greater than 10 years after logging or fire.
- Young naturally regenerated and shrubby vegetation greater than 10 years after fire.
- Dense uniform stands of young conifers with low crown base heights.
- Dense young to mature forests without large trees.
- Dense forests containing large fire-tolerant trees and fuel ladders.
- Relatively open forests with large fire-resistant trees and low fuel ladders.

This list does not account for deciduous and evergreen hardwoods that can make patches less flammable, under less than extreme burn conditions. The appropriate mix of these types and management actions can only be determined using multiscale (patch, landscape, ecoregion) approaches that integrate fire protection, fire restoration, and wildlife habitat goals (Hessburg et al. 2016, North et al. 2009). An overarching aim of restoration efforts could be to introduce more heterogeneity in fuel conditions at landscape levels with the goal to reduce the likelihood of rapidly spreading large fires that include large patches of high-severity fire. Such landscapes would have lower threats to large overstory fire-resistant trees that were once common and widely distributed across a large percentage of these forest landscapes (Baker 2015; Hagmann et al. 2013, 2014; Sensenig et al. 2013). A special concern with large fires that may burn as large high-severity patches is that they can remove habitat for the northern spotted owl and other late-successional species (Camp 1999, Camp et al. 1997). However, the effect on spotted owl habitat at landscape scales is a subject of uncertainty and active research (chapter 4).

Williams and Baker (2012) argued that restoration programs for ponderosa pine and dry mixed-conifer forests are "misdirected in that they are seeking to reduce all high-severity fire." Eliminating all high-severity fire patches from forests with predominantly low-severity or mixed-severity regimes would not be supported by our understanding of fire history and ecology in these systems. Instead, efforts to reduce the size of high-severity patches or the homogeneity of current fuel loads that lead to large high-intensity fires can be justified where knowledge of local landscape conditions and fire regimes indicates that such patches would not be characteristic of the landscape or would pose a threat to important social and ecological values.

Consideration should also be given in these regimes for promoting open woodlands (e.g., oaks), open shrublands, and meadows and grasslands that have been lost as a result of overgrazing, fire exclusion, succession to forest, and other land use changes (Hessburg and Agee 2003, Hessburg et al. 2005). However, because reintroduction of fire to these systems may increase exotic species or have other unintended effects, restoration actions need to be done thoughtfully (Perchemlides et al. 2008).

Invasive Plant Species and Pathogens

Nonnative invasive plants, insects, and disease can have major economic and ecological effects on forests (Lovett et al. 2016, Moser et al. 2009). While the problem of invasive plants and pathogens is most severe in the forests of the Northeastern United States, there are several species of plants and pathogens that are having or could have significant impacts on forests within the NWFP area (Brooks et al. 2016, Gray 2005, Lovett et al. 2016, Moser et al. 2009).

Invasive plant species often have early-successional life histories and are well adapted to colonizing disturbed areas. Examples of this type of invasive plant in this region include Scotch broom (*Cytisus scoparius*) and Himalayan blackberry (*Rubus armeniacus*), which can invade disturbed areas and oak savannas, altering soil nutrient conditions, limiting tree regeneration, and promoting growth of other nonnative species (Gray 2005, Shaben and Myers 2009). Management of these species requires an understanding of their ecology and does not lend itself to a one-size-fits all solution (D'Antonio and Meyerson 2002). Once tree canopy closure is attained, these species typically drop out of the ecosystem.

Although many invasive species invade disturbed, early-successional and open-canopy forests, closed-canopy forests, including old-growth forests, are not immune to invasive species (Martin et al. 2009). Invasion of forests by shade-tolerant species may just be slower but not necessarily less impactful in the long run than invasion of disturbed nonforest vegetation. Shade-tolerant invasive species of concern in this region include the perennial false brome (*Brachypodium sylvaticum*) and English holly (*Ilex aquifolium*). These species can outcompete native species, alter fire regimes, and possibly alter soil conditions where they occur within forests (Berger and Fischer 2016, Stokes et al. 2014, Taylor and Cruzan 2015). Management strategies for reducing spread of false brome, which is most likely to be found in lower elevation forests, include limiting disturbance within stands, cleaning clothes and equipment to reduce seed dispersal, and possibly promoting hardwoods, whose litter is less suitable for germination (Taylor and Cruzan 2015). False brome may increase flammability of forests, and short-interval fire may promote it; as climate warms, invasion of forests by false brome is expected to increase (Brooks et al. 2016).

Invasive pathogens with significant effects on forests of the NWFP area include white pine blister rust (*Cronartium ribicola*), Port Orford cedar root disease (*Phytophthora lateralis*), and sudden oak death (SOD) (*P. ramorum*) (see also chapter 11). Whitebark pine (*Pinus albicaulis*), a high-elevation species of the Cascades, is in decline throughout its range as a result of the combined effects of white pine blister rust and native bark beetles (Ellison et al. 2005). The loss of this species is having cascading effects on hydrology and other species.

Sudden oak death is of particular concern because it has caused extensive mortality of tanoak (*Notholithocarpus densiflorus*), coastal live oak (*Quercus agrifolia* var. *oxyadenia*), California black oak (*Q. kelloggii*), and several other oaks in coastal forests of northern California and southern Oregon. The pathogen also infects a number of other tree and shrub species, many of which have special cultural significance to tribes (see chapter 11). Management strategies for SOD have focused on preventing or reducing transmission through quarantines that limit commercial movement of wood and host plants, and stand-level treatments, including killing and removal of infected trees and host plants, especially California bay laurel (*Umbellularia californica*), through cutting, burning, or herbicide application (Rizzo et al. 2005, Swiecki and Bernhardt 2013). Moritz and Odion (2005) reported that infections in stands that had experienced fire since 1950 were extremely rare; they suggested that a lack of fire could contribute to infestation by increasing shading, stand density, and abundance of hosts. Meentemeyer et al. (2008) concluded that reductions in fire frequency have likely facilitated SOD by increasing woodland cover and continuity at the expense of grasslands and chaparral, and by increasing bay laurel and creating more shaded, cooler microclimates.

The loss of mature tanoaks and various oaks has significant impacts on forest ecosystems in the infested areas. In heavily infested areas in conducive environments, stands formerly dominated by tanoak have been converted to shrubfields (Cobb et al. 2017, Klein et al. 2013). Additionally, infested stands could form stands with multiaged structures, a higher proportion of redwood and a lack of tanoak, and large canopy gaps (Waring and O'Hara 2008). While such changes could enhance stand structural heterogeneity, they could also jeopardize valuable ecological services such as nut production and abundance of large tree cavities in hardwoods, which are important for fisher, owls, and other animals (Long et al. 2016). Other likely effects of the dieback include increased fuel loads, risk of high-severity burns, hazardous conditions for firefighters, increased soil erosion, and spread of invasive plants (Forrestel et al. 2015, Swiecki and Bernhardt 2013). Research in one burned landscape indicated that stands with recent SOD establishment may experience higher vegetation burn severity, while stands where dead trees have fallen may experience increased soil burn severity (Metz et al. 2011). Although high-severity fire in particular can reduce pathogen load, infected bay laurel plants that survive within such burns may infect the resprouting vegetation (Beh et al. 2012). The combination of severe fires and SOD infection may increase the likelihood of extirpating tanoak in redwood-dominated areas, because redwood generally outcompetes tanoak after fire (Ramage et al. 2010). Consequently, it is important for managers to consider landscape-scale strategies that could promote resilience to both the disease and other disturbance agents such as severe wildfire and drought. Evaluating restoration strategies through an adaptive management framework seems particularly important given the complex dynamics among vegetation, SOD and other diseases, and fire (Odion et al. 2010, Rizzo et al. 2005). Use of managed wildland fire, especially in stands that are not already heavily infested with SOD, may be particularly important as a means of promoting forest resilience. Meanwhile, infected stands may be a priority for silvicultural treatments to reduce the potential for severe crown fires (Kuljian and Varner 2010).

Postfire Salvage and Management

Ecological effects—

Postfire salvage logging is typically proposed as a means of recovering some of the lost economic value in dead or damaged trees. The ecological consequences of salvage logging are often considered negative from the perspective of soils, hydrology, postfire seedling establishment, and wildlife habitat resources, although species responses differ. Early scientific understanding of salvage logging after wildfire was hindered by a lack of studies with sufficient replication and controls (McIver and Starr 2001), but recent research offers a more complete understanding of some ecological effects of salvage logging (Long et al. 2014). Table 3-7 summarizes key findings from several reviews to help inform management decisions surrounding postfire salvage; research on this topic is developing as more large and severe fires occur in fire-excluded landscapes. We focus on effects of salvage logging (i.e., the removal of dead trees and those that are likely to die following wildfire) rather than a broad range of other postfire management activities. However, it is important to recognize that managers often avoid replanting in areas that have not been salvage logged for crew safety and for silvicultural reasons.

Immediate stand-level effects of fire are primarily related to intensity, duration, and corresponding severity, most commonly interpreted through some measure of tree mortality and combustion of surface fuels, including dead and down wood and organic matter stored in duff, litter, and soils. Fire can reduce live tree density and canopy cover and increases the density of standing dead trees (snags) and the future abundance of dead and down wood. Although enormous amounts of carbon stored in live and dead biomass may be lost to the atmospheric carbon pool in a large fire (Campbell et al. 2007), most is retained in biological legacies, including snags, dead and down wood, charcoal, and live remnant trees (Acker et al. 2013, Baird et al. 1999, Donato et al. 2013). This carbon pool is then slowly lost from the forest as the retained deadwood decomposes or is consumed in subsequent fires (Campbell et al. 2016b, Donato et al. 2016). These biological legacies play important ecological roles that differ from the enrichment of recovering vegetation to providing microhabitats, stabilizing soils, and moderating harsh environmental conditions on burned sites (Lindenmayer and Noss 2006, Lindenmayer 2004).

Salvage logging alters postfire vegetation structure by reducing the basal area and density of live and dead trees (McIver and Otmar 2007) and decreasing the persistence of remaining snags (Russell et al. 2006) and altering the microclimate of a site (Marañón-Jiménez et al. 2013). What's more, once a tree dies, it functions as a snag, down log(s), mulch, and charcoal in soils for a period that can far exceed the period spent as a live tree (DeLuca and

Aplet 2008), although those dynamics should vary widely based upon moisture and fire regimes. Cumulatively, these reductions result in decreases in live and dead biomass (Donato et al. 2013) and reduced soil carbon. However, the down dead wood would not likely have been able to decompose in frequent fire regimes before the onset of fire suppression (Skinner 2002). Studies have shown that as wood becomes more decayed, it is more likely to be consumed in subsequent fires (Knapp et al. 2005, Uzoh and Skinner 2009). Numerous studies document initial short-term decreases in natural regeneration following salvage (McIver and Starr 2001) for various reasons, including direct mortality from mechanical damage (Donato et al. 2006) as well as indirect effects of altered competitive interactions with shrubs and harsher microclimate (Marañón-Jiménez et al. 2013, Ritchie and Knapp 2014, Stuart et al. 1993). However, one study 10 years after salvage showed no difference in natural regeneration following a severe fire with different levels of salvage ranging from leaving everything to taking everything (Ritchie and Knapp 2014). Planting following salvage may be needed to mitigate any effects on regeneration or to establish tree species and genotypes that are better suited to climate warming or diseases. The effects of salvage logging versus no intervention on loading of fine fuels and coarse fuels and the effects of reburn are expected to differ considerably over time. If not followed by fuel treatment or accomplished through whole tree harvesting (Ritchie et al. 2013b), salvage logging can increase fine fuels to levels that support high-severity fire, which kills regeneration (Donato et al. 2006). There are few studies of the effects of salvage on subsequent wildfire, but Thompson et al. (2007) found higher reburn severity in stands that were salvaged and planted than in unmanaged stands. The Thompson et al. (2007) study hypothesized that salvage logging without sufficient treatment of the slash after logging and uniform

Table 3-7—Suggestions for ecologically based postfire management in terrestrial ecosystems from three major reviews

Recommendations	Karr et al. 2004	Beschta et al. 2004	Lindenmayer and Noss 2006
Promote natural recovery	✓	✓	
Retention of old, large trees and snags	✓	✓	✓
Protect soils against compaction and erosion	✓	✓	✓
Protect ecologically sensitive areas (e.g., reserves, roadless areas, steep slopes, fragile soils)	✓	✓	✓
Rehabilitation of roads and fire lines, avoid creation of new roads	✓	✓	
Limit reseeding and replanting	✓	✓	
Protect and restore watershed before fire	✓	✓	
Continue research, monitoring, and assessment of the effects of salvage treatments	✓		
Educate public on the natural role of wildfires, allow natural regimes	✓	✓	
Ban introduction of exotic species		✓	
Curtail livestock grazing		✓	
Low-intensity or no harvesting in unburned or partially burned patches		✓	✓
Limit removal of biological legacies from particular areas (e.g., burned old-growth stands)			✓
Ensure maintenance and creation of essential habitat elements for species of concern			✓

conifer plantations likely contributed to higher surface fuel loads after salvage and consequently to the higher reburn severity. More work is needed to evaluate the effects of salvage logging and adequate slash disposal on risk of high-severity fire. One study found that fine fuel loading following salvage returned to untreated levels after about 25 years (McIver and Ottmar 2007).

Salvage logging also reduces large fuel loads over time through removal of snags that would otherwise begin to fall and increase large dead wood on the ground as early as the first 10 years following fire, but typically over much longer periods (Dunn and Bailey 2015, McIver and Ottmar 2007, Peterson et al. 2015). One study showed that regardless of intensity of salvage logging, more than 80 percent of tree biomass left standing had transitioned to become surface fuel after 8 years (Ritchie et al. 2013b) with pines falling more rapidly than either white fir or incense cedar (*Calocedrus decurrens*) (Ritchie and Knapp 2014). Greater log biomass in unsalvaged stands resulted in higher surface temperatures during prescribed fire 20 to 30 years following wildfire (Monsanto and Agee 2008). Large areas of the Western United States have been burned by high-severity fire or killed by bark beetle outbreaks. The resulting dead fuels will become future surface fuels. Long-term research is needed to better understand the tradeoffs among postfire salvage logging and future surface fuels, and the ecological benefits of dead and down wood and future fire severity and community succession.

Salvage logging can affect ecosystem processes by altering microclimate and hydrology, increasing sediment production, and reducing soil nutrients and carbon sequestration in the forest. Removal of snags can affect microclimate by reducing shade (sometimes referred to as dead shade) and consequently reducing temperatures at night and increasing temperatures during the warming part of the day (Fontaine et al. 2010). Risk of accelerated erosion comes with ground disturbance during salvage logging (Wondzell 2001); however, there is a noticeable lack of studies from the Northwest on this issue. In one Western United States study, Wagenbrenner et al. (2015) found that salvage logging increased soil compaction, decreased soil water repellency, and slowed recovery of vegetation, but the degree of impact depended on the method of logging, local climate, and soils. Where a winter snowpack is typical, the potential for hydrological impacts is greatest where harvest operations occur outside of the winter months. Logging over snow and frozen ground could reduce the effects on soil and sediment (Poff 1989). Indeed, Peterson and Dodson (2016) found that postfire commercial logging on dry or frozen soils in northeastern Oregon displaced or compacted an average of 15 percent of the soil surface in commercial logging units and 19 percent of the soil surface in the fuel reduction logging units, yet they found no persistent impacts on understory vegetation 15 years following treatment. In a study from central Oregon, compaction following salvage logging decreased soil respiration and available nitrogen, while later subsoiling to alleviate compaction decreased available phosphorus (Jennings et al. 2011). In several studies of boreal forests, postfire removal of snags reduced soil carbon for several years (Bradford et al. 2012, Kishchuk et al. 2015, Poirier et al. 2014). In two studies from relatively dry Sierra Nevada forests, Johnson et al. (2005) and Powers et al. (2013) found that postfire salvage resulted in a substantial reduction in onsite carbon compared to fire alone, although the authors of both studies noted that their studies, as with many other studies, did not account for sequestration in the resulting wood products. Moreover, it is important to consider long-term carbon dynamics given future fires (Carlson et al. 2012), because planting treatments can potentially accelerate carbon storage in trees, and fuel reduction treatments can potentially reduce future tree mortality.

The impacts of salvage logging on biota are mostly associated with the removal of snags and deadwood, which are important habitat components for a variety of terrestrial and aquatic organisms. Salvaging has been reported to have negative effects for several species of cavity-nesting birds, such as black-backed woodpeckers (*Picoides borealis*), three-toed woodpeckers (*P. tridactylus*), and mountain bluebirds (*Siala currucoides*), (Hutto 2006, Hutto and Gallo 2006, Saab et al. 2007), but neutral or positive effects have been documented on a few species

(Peterson et al. 2009). In a recent study from the Sierra Nevada, White et al. (2015) suggested that it was important to retain some relatively dense stands of dead or dying trees (40 to 60 per acre) at the landscape scale, to promote snag-associated species such as black-backed woodpecker, mountain bluebird, and olive-sided flycatcher, rather than evenly thinning all stands and retaining smaller numbers of snags; they suggested further research would be needed to guide the extent and configuration of such treatments. Soil bacteria and fungi appear resilient to salvage (Jennings et al. 2011). Removal of snags and large coarse woody debris could adversely affect habitat for carnivores such as fisher (*Pekania pennanti*) and Pacific marten (*Martes caurina*), if the large dead wood would have otherwise persisted into closed-forest stages where the animals use large structures for den and rest sites (Bull et al. 2001).

Fire may have positive effects by contributing wood and coarse sediment for aquatic habitats (Benda et al. 2003, Reeves et al. 1995) that may be partially negated by removal of wood during salvage logging, especially when the large wood is removed from key source areas to streams. Many aquatic and riparian organisms are adapted to fire (Flitcroft et al. 2016, Reeves et al. 2006) so postfire management is typically not needed to support aquatic ecosystems. Hillslope processes and subsequent erosion after periodic fires are critical to aquatic habitat succession, and native fish populations can often rebound within a decade after a wildfire, especially when they can recolonize altered reaches from connected refugia (Bisson et al. 2003, Dunham et al. 2003, Rieman and Clayton 1997, Rieman et al. 1997).

Management of postfire environments—
The ecological effects of postfire salvage logging can differ depending on treatment, fire severity, and biophysical setting (Peterson et al. 2009). In general, research supports the conclusion that salvage logging does not benefit native species and terrestrial or aquatic ecosystems (Beschta et al. 2004, Karr et al. 2004); an exception might include, e.g., fire-suppressed forests with high densities of trees. Further long-term research on contemporary salvage practices would greatly enhance understanding of the circumstances under which salvage might be beneficial.

Peterson et al. (2015) and Hessburg et al. (2016) identified situations, including elevated long-term woody fuel loads, lack of seed sources, and potential for reburns that maintain undesirable shrubfields, in which postfire management might be used to meet ecological goals. These include (1) fuel reduction treatments that reduce long-term levels of large woody fuels (which may be elevated as shade-tolerant species increased under fire suppression and that may pose a risk to soil fertility were the area to reburn), (2) fuel treatments or planting trees to reduce potential for high-severity reburns and forest succession where potential for large semistable patches of shrubs is high and regeneration is lacking (Dodson and Root 2013), and (3) removing surface fuels that may impede establishment of trees. The effects of particular strategies may differ considerably with ecological conditions across the NWFP area. In some cases, shrub removal may be important for promoting native plant species richness (Bohlman et al. 2016) in subsequent decades. However, shrubs may also have important roles in increasing soil carbon and nutrients, especially nitrogen. For example, in a dry ponderosa pine site in central Oregon, Busse et al. (1996) found that shrub removal aided tree growth in the first two decades, but the effect then leveled off and shrub removal was associated with decreases in soil carbon and nitrogen 35 years later.

Tree replanting, which as mentioned above is often practically tied to postfire snag removal, may be an important strategy to consider in areas where natural regeneration is too low to meet objectives for a landscape in the time desired. One example of such low regeneration was reported for several fires in the northern Sierra Nevada (Collins and Roller 2013) bordering the NWFP area but that has similar species to the Klamath region. The authors of that study noted that several studies from mixed-conifer forests in the mixed-severity regime of the Klamath-Siskiyou Mountains (Donato et al. 2009, Shatford et al. 2007) had found generally abundant conifer regeneration in stand-replacing patches. Where sites reburn and high-severity patches are large, regeneration can be low (Tepley et al. 2017). Lower and less consistent moisture may also contribute to incidents of sparse conifer regeneration in regions predisposed to a frequent fire regime. Because promoting vegetation heterogeneity may reduce fire spread and burn severity (Thompson et al. 2007) and promote biodiversity, managers have experimented with more variable planting patterns (e.g., spacing and clustering) than have traditionally been used, but more research is needed to evaluate outcomes from such strategies.

Accumulation of large dead fuels can lead to severely burned soils if forests reburn. A study from the eastern Cascades of Oregon found that severely burned soils can have lower fertility and depleted microbial communities (Hebel et al. 2009). However, this study also found that several native plants appeared highly competitive in severely burned, low-resource soils; based upon a laboratory study component, they suggested that those native plants might be more competitive in those burned soils than invasive nonnative species. Relationships between plant diversity and fire severity are complex because they reflect variation in environment (especially precipitation and fire regime) and species composition (such as presence of invasive species). For example, DeSiervo et al. (2015) hypothesized that diversity would be promoted in fires that matched the reference fire regime, and they indeed found that native species richness was greater in areas of low to moderate vegetation burn severity of northern California (in a region of frequent fire), while areas burned at higher severity experienced more incursion by cheatgrass and other nonnative species. Similarly, Stevens et al. (2015) found that high burn severity shifted composition toward nonnative species and native species with southern-xeric affinity and away from native species with northern-temperate affinity.

Application of salvage logging in these contexts would need to consider overall effects of a wildfire on the larger affected landscape, and tradeoffs with other ecological and economic objectives. More research is needed to better understand the ecological effects of low to moderate levels of salvaging that may be done to recover economic value (Campbell et al. 2016a) from fire-killed trees.

Research Needs, Uncertainties, Information Gaps, and Limitations

While much as been learned about the ecology, conservation, and restoration of these forests, many knowledge gaps and uncertainties remain. We mention them throughout the document and summarize the major ones here:

1. While the range- and regional-scale patterns of disturbance regimes are known, much less is known about them at subregional and landscape scales. Our knowledge of the region is based on extrapolation from relatively few fire and forest history studies. Research is needed to help fill in the gaps in our knowledge especially as they relate to fire sizes, frequencies, and function in mixed-severity regimes of both the moist and dry forests.
2. We know much about the structure of old-growth forests from studies of contemporary older forests across all forest types but lack stand-structure definitions for use in monitoring and inventory related to old-growth forests that developed in the mixed- and low-severity fire regimes of moist and dry forests. Our current monitoring efforts (e.g., definitions and indices) use reference conditions for old growth that are based on forests that have been altered by fire exclusion and do not take into account structures associated with historical disturbance regimes. Research is needed to develop old-forest definitions and landscape-scale targets based on HRV, desired levels of resilience given fire, and future climate change or other considerations such as species habitat needs.
3. We lack information about the biodiversity and ecosystem functions of early-seral vegetation as well as frameworks for developing landscape-scale goals for these conditions given fire suppression. Mechanical treatments and prescribed fire can be used to approximate some of the ecological functions of diverse early-successional habitats. We also lack knowledge of what restoration actions (e.g., planting in post-wildfire environments) might be beneficial for longer term successional goals (e.g., recovery of conifer forest canopies).

4. The effects of fire suppression on forest biodiversity and ecosystem function in older forests are not well studied in much of the NWFP area. This is apart from knowledge of how succession has altered fire regimes and fire risk. Lack of fire in high-fire-frequency forests is altering plant community diversity, but more research is needed on the long-term ecosystem effects of increased stand density and shade-tolerant species in forests that were burned frequently to moderately frequently by low- to moderate-severity fire.
5. We lack a solid understanding of how drought, beetles, and disease are likely to affect forests given climate change and interactions with fire.
6. The ecological tradeoffs associated with variable-density thinning (i.e., restoration thinning) to restore or create ecological diversity in forest plantations are not well understood at stand or landscape scales and are known only from relatively short-term studies. Long-term research is needed to understand how ecosystems and the biota respond to these management actions and to learn more about the possible ecological costs and benefits of these actions in stands older than 80 years that might have undesirable densities or uniformity of trees. Similarly, long-term effects of postfire management warrant further study at large and long-term scales.
7. Given tradeoffs associated with restoration actions or inactions for different types of habitats and successional stages, research is needed to explore options for managing for a dynamic mosaic of vegetation and habitats at landscape scales under climate change. For example, how much do the pace, scale, and pattern of restoration activities at landscape scales affect fire severity and patterns of successional stages under a changing climate?
8. It will also be important to better understand the tradeoffs associated with use of both coarse- and fine-filter approaches to conservation. Dynamic landscape modeling is needed, and where feasible, landscape-scale experiments and demonstration areas will be important to advancing our understanding of this issue.

Conclusions and Management Considerations

Timber harvest, fire exclusion, fire suppression, and the loss of burning by American Indians have profoundly changed the moist and dry forests of the NWFP area. Although the motivation for the NWFP arose from clearcutting of old growth and loss of spotted owl habitat in moist forests, the dry zone forests, which occupy about 43 percent of the Plan area, have actually experienced more pervasive ecological changes as a result of human activity. Key changes in dry forests are loss of large, fire-resistant trees to logging, large departures in amounts and patterns of surface and canopy fuels, widespread shifts in proportions of seral stages, and changes in the patch sizes of those seral stages. These changes have affected all species and all processes; some in favorable ways (e.g., more habitat for dense forest species) and others in unfavorable ways (e.g., loss of open old-growth forests and ecological resilience to fire and drought). Changes in the moist forests are also significant, but they have been affected to a lesser and different degree by fire exclusion. Here, intensive timber harvest has been the primary impact on biodiversity by dramatically reducing the amount of dense old-growth forests and fragmenting habitats for species associated with these older forests. Fire exclusion in moist forests has had an important but different and less visible effect: the loss of diverse early-seral vegetation and associated reduction in landscape diversity.

The 2012 planning rule adds a new context for NWFP national forests that will undergo plan revision in the coming years: management for ecological integrity (ecosystem characteristics) and species conservation using coarse-filter approaches; fine-filter approaches are to be used for a limited number of species where coarse-filter approaches may not be sufficient. Coarse-filter approaches based on managing for ecological integrity (as opposed to coarse-filter approaches based on one vegetation type, i.e., dense old growth) would promote basic ecological processes, including major disturbances that regulate successional and fuel patterns (i.e., “habitat” for fire). Ecosystem-dynamics approaches are needed to rebuild more functional ecosystems, reduce threats to and possible listing of additional species, and provide a more ecologically viable approach to maintaining existing listed or sensitive species within the context of meeting other ecological and socioeconomic goals.

Management Considerations Summarized

- The 2012 planning rule sets a new context for ecosystem management under the NWFP: it focuses on ecological integrity based on maintaining and restoring disturbance and other ecological processes. Natural range of variation is a guide but not necessarily a target. This is a broader focus than the original coarse-filter approach of the NWFP, which focused primarily on one type of forest condition: dense, multilayered older forest.
- The goals and standards and guides for LSRs of the moist forests with infrequent fire are a relatively good match for managing for ecological integrity and resilience, especially in the face of climate change and invasive species.
- Focusing restoration (e.g., variable-density thinning) in LSRs in moist forests on plantations makes sense from a conservation perspective, and can provide jobs and economic returns. However, there will be tradeoffs with some ecological goals (e.g., amounts of dead wood) that may need mitigation.
- Fire suppression has had an effect on vegetation conditions in moist forests, especially in the drier part of the zone where fire was historically more frequent and mixed-severity effects more common. The effect is not the same as in dry forests. Fire exclusion in moist forests has reduced the amount of early-successional vegetation in the landscape, reduced diversity of structure in old-growth forests that were subject to partial stand-replacement fire, and thus reduced landscape-scale diversity. Managers may want to consider restoring fire or using fire surrogates to promote early-successional forests and landscape-scale diversity in plantations and forests more than 80 years old in the matrix. Managing for diverse early-seral stages would require a landscape-scale approach to ensure that old-growth goals are not compromised.
- The goals, standards, and guides for LSRs in dry forests are inconsistent with management for ecological integrity and resilience to climate change and fire. Dense late-successional older forests would have been historically uncommon in dry forests, and their current higher abundance is a function of fire exclusion and suppression. Fires have become much less frequent than historically, but, when they burn, they are more likely to include large patches of high-severity fires that kill fire-resistant older trees and alter landscape-scale patch patterns. In the absence of fire, the forest structure and composition are shifting toward denser forests and shade-tolerant species that are less resistant to fire and drought.
- Management actions that promote resilience in dry forest landscapes include reducing the continuity of surface and canopy fuels to reduce patch sizes and thus the extent of high-severity fires and using prescribed fire or managing wildfire for ecological benefits where appropriate. Landscape-level strategies are needed to provide for dense forest conditions as indicated by the NWFP in environments where they are more likely to persist in the face of fire and climate change.
- Alternative approaches to the LSR network and standards and guides may better meet both coarse- and fine-filter goals by incorporating the evolving understanding of the ecological dynamics of dry forests and threats from climate change and invasive species that apply to both moist and dry forests.

Our main findings and conclusions are listed below by general topic. We also indicate which of the following questions the conclusion applies to:

Guiding Questions

1. What are the structures, dynamics, and ecological histories of mature and old-growth forests in the NWFP area, and how do these features differ from those of other successional stages (e.g., early and mid successional)?
2. How do these characteristics differ by vegetation type, environment, physiographic province, and disturbance regime?
3. What is the scientific understanding about using historical ecology (e.g., historical disturbance regimes and natural range of variation) to inform management, including restoration?

4. What are the principal threats to conserving and restoring the diversity of old-growth types and to other important successional stages (e.g., diverse early seral), and to processes leading to old growth?
5. What does the competing science say about needs for management, including restoration, especially in dry forests, where fire was historically frequent?
6. How do the ecological effects of treatments to restore old-growth composition and structure differ by stand condition, forest age, forest type, disturbance regime, physiographic province, and spatial scale?
7. What are the roles of successional diversity and dynamics, including early- and mid-seral vegetation, in forest conservation and restoration in the short and long term?
8. What is the current scientific understanding concerning application of reserves in dynamic landscapes?
9. How do recent trends of forests in the NWFP reserve network relate to both original NWFP goals, those of the 2012 planning rule, and climate change adaptation needs?
10. What is the current understanding of postwildfire management options and their effects?

Ecology of Old-Growth and Other Vegetation Types (Questions 1 and 2)

1. Knowledge of historical disturbance regimes and successional dynamics is essential for conserving, restoring, and promoting resilience of old-growth forests and other successional stages to climate change, fire, and other disturbances.

    a. All seral stages contribute to maintaining native forest biodiversity, ecosystem function, and other ecosystem services. Moist forests and dry forests have fundamentally different disturbance regimes, developmental pathways, and ecological potentials.

    b. We developed a fire regime map (fig. 3-6) to provide a framework for planning and managing these diverse forests. Four major fire regimes are recognized, two in the moist forests and two for the dry forests.

    c. The major regimes of the moist forests are:

        i. Infrequent (greater than 200 years), high severity

        ii. Moderately frequent to somewhat infrequent to (50 to 200 years) mixed severity.

    d. The major regimes of the dry forests are:

        i. Frequent (15 to 50 years) mixed severity

        ii. Very frequent (5 to 25 years) low severity

    e. Of these four regimes, the two mixed-severity regimes are the most variable and complex. All severities of fire occur in all regimes, but the regimes differ in proportion and spatial pattern of high-severity fire.

2. Old-growth forest structural elements common to all forests of the region include relatively large and old live, decadent and dead trees, and spatial heterogeneity of forest structure and composition. Other characteristics such as multiple canopy layers, shade-tolerant associates, and large amounts of dead and down wood are not necessarily characteristic of all old-growth forest types under the historical disturbance regimes of the region. Large-tree elements can also be found in younger forests, and patches of early-seral vegetation that developed following high-severity disturbance in older forests.
3. Definitions of old growth that recognize old-growth structural features as a continuum across stands of various ages and disturbance histories are more ecologically realistic and useful for restoration planning than a definition that has only one threshold with the result that forests are either old growth or not.
4. Current definitions of old growth used in monitoring are based on current forest inventory plots. This means that definitions for dry forests, which have

been heavily influenced by fire exclusion, are not reflective of historical forest structure and composition that were typical of this environment. Better definitions or reference conditions that reflect the variety of old growth are needed for conservation and restoration of old-growth and landscape dynamics for dry forest types, as well as communities with significant hardwood components.

5. Older forests differ in tree density, spatial heterogeneity, and species composition between moist and dry forest zones and across their associated disturbance regimes. Dense, multilayered old forests were typical of infrequent/high-severity fire regimes in moist forests parts of the region, while relatively open forest of pine, Douglas-fir, and other conifers were typical of very frequent/low- and mixed-severity regimes in dry zone forests. Dense multilayered older forest in dry forest landscapes occurred in fire refugia such as topographic settings where fire was infrequent. Old-growth forest structure and composition were most diverse in the mixed-severity regime of the moist forests and the mixed-severity regime of the dry forests.
6. Early-seral and "pre-forest" vegetation was an important component of many landscapes. Early-seral vegetation that results from high- and mixed-severity disturbance provides distinctive biodiversity and ecosystem function. Grasses, herbs, shrubs, hardwoods, and legacy live and dead trees that develop during these stages can influence forest development, biotic communities, and ecosystem function for decades to centuries.
7. Landscape diversity also varied across the disturbance regimes. In the infrequent/high-severity regime of the moist forests, the dominant landscape pattern was medium to coarse grained with very small to very large patches of older forests of complex structure, patches of younger more homogeneous forests, and rare to common (depending on climate period) very large patches of early-successional vegetation. Patches of hardwoods and shrubs would have occurred along many streams. In the mixed-severity regime of the moist forests, the landscape would have been a relatively dynamic mosaic of well-connected and dispersed mature and older forests and differently aged and sized patches of younger forests and preforest vegetation forests, often containing remnant live and dead large trees.
8. The forest landscape of the frequent/mixed-severity regime of the dry forests would have been a complex mosaic of forest structural types that was very strongly controlled by frequent fire. In the very frequent fire regimes, the forested part of the landscape would have been a fine- to medium-grained mosaic of older trees and very small to small patches of early-successional conditions. The open nature of the forest combined with the fine grain of patches often led to blending of areas of old trees with understory vegetation (forbs, grasses, shrubs) otherwise typical of early-seral conditions. In steep, dissected topography (e.g., northwest California), the mosaic of forest conditions would have been more strongly expressed as a function of topography and fine-scale variability in disturbance regimes and successional pathways.

Value of Ecological History (Question 3)

1. Knowledge of ecological history is essential for conducting and guiding conservation and restoration. Using HRV in forest structure, composition, and landscape patterns can be a useful guide for conservation and restoration efforts. However, returning forests and landscapes to a narrowly defined state of historical conditions and dynamics will not be possible nor desirable in many landscapes given anthropogenic forest change (e.g., land ownership patterns and forest management) and climate change. Approximations of historical regimes and forest conditions or management for resilience to fire as a recurring ecological process and climate change will be a more realistic and sustainable goal for many areas.

Conservation and Restoration Needs (Questions 4 and 5)

1. While the restoration needs differ between the moist and dry forests, logging and plantation silviculture have affected forests in all of the regimes. In the moist forests, clearcutting and plantation establishment for timber production reduced the area of old-growth forests and fragmented the landscape across millions of acres of forest lands. Intensive timber management has reduced stand-level diversity, reduced dead wood and snag abundance, increased the amount of sharp edges, and increased road densities. Clearcutting and plantation establishment affected the drier forests as well, but a more pervasive effect may have been the partial harvest of old pines that significantly reduced the abundance of large, fire-resistant trees, leaving existing older forests with far fewer large live and dead trees than they would have had under natural disturbance regimes. Moreover, often the larger overstory trees are species (e.g., Douglas-fir, grand fir, or white fir) that are not as resistant to fire.
2. Fire exclusion effects are also present in all regimes but are significantly different between the dry and moist forest zones. In the dry forests, lack of fire has greatly increased tree density and reduced resilience to fire, drought, insects, and disease. Specifically, the area of multilayered, closed-canopy older forest has increased outside the historical range over the past 100 years despite logging and recent fires. Fire suppression has also had an effect in the moist forests, but there has generally been little impact on fuel accumulation (except where logging has occurred and slash has not been treated) and fire risk as these productive forests naturally have high fuel loads. Instead, the effects of fire suppression in moist forests have been to reduce the area of high-severity fire (relative to historical dynamics), and, consequently, the area of diverse early-successional vegetation. Thus, lack of fire in the moist, mixed-severity-regime forests has likely reduced landscape diversity.
3. Fire exclusion and succession toward shade-tolerant, fire-sensitive species may be leading to more fire-resistant older forest vegetation in some dry forests under a wider range of fire weather conditions. Forests in these areas are more shaded, dry out more slowly, have lower windspeeds, and have more compact fuel beds that are less able to carry fire than more open pine-dominated older forests. However, under extreme weather, these forests are less resistant and resilient because they are more likely to burn with high severity than historically, when forests were more open and contained less fuel. As climate changes, such extremes (e.g., drought and high winds) are expected to increase.

Competing Science Related to Need for Restoration (Question 5)

1. Some have argued that restoration is not needed because most ponderosa pine and dry mixed-conifer forests have been mischaracterized as simply having a low-severity fire regime. Instead, they contend that these forests were historically denser than most other studies indicate and are better characterized as having a more variable-severity fire regime, with significant components of mixed- and high-severity fire. Baker (2012) and others cite Hessburg et al. (2007) in support of their arguments; however, the results of Hessburg have been misinterpreted in these papers and do not fully support claims about the importance of high-severity fire in dry forests. In addition, recent research (Levine et al. 2017) indicates that the method used by Baker (2012) overestimates tree densities. We believe the preponderance of evidence supports the view that prior to Euro-American settlement, pine and dry mixed- and some moist mixed-conifer forests had relatively low tree densities and that large patches of high-severity fire were not common in dry forests

with very high frequency (<25 years) and low severities. However, larger patches of high-severity fire were an important component of dry forests (e.g., mixed conifer) with frequent (15 to 50 years) mixed-severity regimes.

Trends in Forests in the NWFP Reserve Network (Question 9)

1. At the scale of the NWFP area, losses of older forest owing to logging and wildfire over the 20 years of the NWFP have been relatively small and compensated for by significant gains from succession offsetting almost two-thirds of the losses from high-severity disturbance. However, dynamics differ geographically and with scale, and some areas, especially the Klamath region in Oregon and California, have had much higher net losses as a result of very large high-severity patches (mainly from a single large fire [Biscuit]). The NWFP reserve strategy, which focused on closed-canopy older forests is currently meeting many of the expectations of the NWFP, but it appears unlikely that this network will support the original conservation goals and new goals of the 2012 planning rule in dry forests under climate change. Threats include more frequent and larger patches of high-severity fire, which are promoted by high canopy fuel continuity and elevated surface fuel loads.

Reserve Approaches in Dynamic Landscapes (Questions 8 and 9)

1. Reserves are a valuable strategy for conserving biological diversity in the face of development and many extractive land uses. The literature indicates that goals and management guidelines for reserves need to be clearly defined. Management within reserves also may be needed in many cases to address past management effects or restore ecological processes and ecosystems that have been altered by past land use, including timber management, fire exclusion, and invasive species.
2. The options that were developed in FEMAT (1993) and set the foundation for the NWFP were based on the best available science at the time, but that science emphasized moist zone forest ecology and did not adequately deal with the substantially different ecology of forests and landscapes of the dry forest zone (Spies et al. 2006b). Although the LSRs are currently providing for late-successional/old-growth forest conservation, new science and increased understanding of fire regimes and climate change indicate that focusing only on dense older forest as the primary conservation goal across the entire NWFP area will likely have unintended negative consequences in terms of diversity of successional stages, resilience to fire and climate change, and biotic disturbance.
3. The current LSR standards, guidelines, and spatial patterns for dry forests do not appear to be consistent with emphasis on ecological integrity and other approaches for conserving biodiversity under the 2012 planning rule. In addition, threats from climate change and invasive species including the barred owl would appear to justify a reassessment of the reserve network in both dry and moist forests (see chapter 12). Development and evaluation and testing of new, highly integrated conservation approaches is encouraged to deal with changing knowledge, new perspectives on fire regimes, climate change, invasive species, and recognition of tradeoffs among biodiversity goals (e.g., coarse filter and fine filter) and between the ecological and social dimensions of forest ecosystem management (see chapter 12 for more information).

Restoration Approaches (Questions 6 and 7)

1. Restoration is more about creating landscapes for the future that are resilient to future fires and changes in climate and support native species than it is about recreating past conditions. We can use historical ecology at the community and landscape scales to understand how various patch- and landscape-level patterns will respond under these new conditions. Restoration strategies include:

a. Variable-density thinning in plantations to increase ecological heterogeneity and accelerate growth of large trees and tree crowns.
b. Variable-density thinning from below and prescribed fire in dense older forests in very frequent/low-severity and frequent/mixed-severity regimes to increase resilience of those forests to fire and climate change through restoring more diverse structures and compositions of older forests.
c. Careful use of prescribed fire and managing wildfires away from the wildland-urban interface in dry forests and mixed-severity regimes of moist forests to restore key ecological processes while protecting critical areas of dense, older forest conditions.
d. Creating diverse early-successional habitat where feasible given other ecological goals and social constraints. This could include partial cutting (retention silviculture) and prescribed fire (e.g., "ecological forestry") in plantations and perhaps in forests over 80 years old (which is allowed in the NWFP in the matrix of moist forests and within LSRs in dry forests) where this practice would be consistent with other landscape goals (e.g., resilience to fire and climate change, habitat for spotted owls, creating landscape-scale successional diversity).
e. Using landscape-level strategies based on disturbance regimes, topography, spatial pattern, and departure from desired historical conditions.

2. The scientific understanding of using 80 years as a threshold for restoration of stands within LSRs in moist forests has not improved much since the NWFP was established. The 80-year rule from the NWFP was based on expert opinion of stand development from data collected in natural forests of different ages. Eighty years is a one-size-fits-all threshold that does not recognize that stand age is only a rough proxy for stand structure and development potential, both of which can vary greatly based on site conditions and disturbance history. Depending on the structure and composition of stands, and landscape context and objectives, restoration treatments in forests over 80 years could promote old-growth characteristics or reduce them (e.g., reduce number of large dead trees). However, in general, and given a lack of new information, treatments of stands over 80 years in moist forests would still be expected to have less benefit for reaching old-growth structure than restoration in stands under 80.
3. There is no new ecological science that undercuts the guideline of using alternative silviculture to meet both wood production and ecological goals in stands over 80 years in the NWFP matrix of the moist forests. Studies of retention silviculture suggest that some biodiversity elements of older forests can be retained in stands managed for a combination of timber and structural and compositional diversity.
4. All management (including restoration activities and lack of activities) involve ecological tradeoffs:
   a. Commercial thinning can provide short-term early-seral habitat and accelerate the development of large live trees and habitat diversity for some species but may have a short-term impact on habitat quality for other late-successional species and can reduce amounts of deadwood in the future (although deadwood may be higher than the historical range owing to fire exclusion).
   b. Thinning and restoring fire to forests with a history of very frequent fire can increase resilience to wildfire and increase habitat for species that use more open older forests and are dependent on fire, but these actions can degrade habitat quality for species that use dense older forests, which may have developed owing to fire exclusion.
   c. Excluding fire from dry forests will increase surface and canopy fuel continuity and increase size of patches of high-severity fire when fires escape suppression and burn under extreme conditions.

d. Excluding fire and disturbance from dry forests will typically increase stand density and shift species composition toward late-successional species and species that use dense forests and lower the resilience of these forests to fire and drought.
e. Excluding fire from moist forests (especially in the drier parts of the moist forests) likely reduces landscape-scale vegetation diversity and the area of diverse early-successional forest and may increase the sizes of high-severity fire patches.
f. The effects of stand-level management actions may be different when examined at different spatial scales and time periods. Multiscale and multitemporal analysis can help reveal how management effects differ with spatial and temporal scale.
g. Tradeoffs among goals are particularly strong in managing road networks, because existing road networks can negatively affect some native species and ecosystem processes, but they also can support landscape restoration, fire management, and active management to support other ecological and socioeconomic goals.

Post-Wildfire Management (Question 10)

1. Salvage logging after wildfire does not typically generate ecological benefits for species and processes associated with patches of high-severity wildfire. However, in some cases (e.g., where fire exclusion has led to dense forests), post-wildfire management may be justified, including:
   a. Planting key tree species after wildfires in uncharacteristically large patches of high-severity fire that may otherwise be slow to regenerate where seed sources are lacking
   b. Thinning high-density post-wildfire regeneration as appropriate to increase heterogeneity and resilience to drought and wildfire.
   c. Salvaging postfire pole and small-sized trees that have grown in during the period of fire exclusion in dry zone forests, where these may constitute a significant fuel bed for reburns in the future, while retaining the medium, large, and very large trees as dead snags and down logs.
2. Actions can be taken to mitigate many of the potentially undesirable effects of salvage logging, particularly by retaining many areas that are not salvaged to ensure heterogeneity and availability of those distinctive postfire communities.

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Appendix 1: Crosswalk of Simpson (2013) Potential Vegetation Zones With Existing Vegetation From the Classification and Assessment With Landsat of Visible Ecological Grouping (CALVEG)

Table 3-8—Potential vegetation zones with existing vegetation from CALVEG[a]

Potential vegetation zone	CALVEG Regional Dominance 1
Western hemlock	Douglas-fir (40.3%), white fir (18.5%), Jeffrey pine (15.5%), tanoak (madrone) (9%), black oak (3.9%), ultra mafic mixed conifer (3.7%), California bay (2.9%), red fir (2.4%)
Tanoak	Douglas-fir (40.3%), tanoak (madrone) (11.3%), Oregon white oak (6.2%), California bay (5%)
Shasta red fir	Red fir (33.2%), white fir (10.1%), Jeffrey pine (10.1%), barren (10%), mixed conifer–fir (8.1%), alpine grasses and forbs (5.1%), pinemat manzanita (5%), subalpine conifers (4.9%), upper montane mixed chaparral (2.9%), perennial grasses and forbs (2.1%)
Port Orford cedar	Douglas-fir (46.6%), ultramafic mixed conifer (24.8%), Douglas-fir–white fir (7.9%), tanoak (madrone) (2.9%), Douglas-fir–ponderosa pine (2.9%), mixed conifer–pine (2.2%), Oregon white oak (2%)
Other pine	Lower montane mixed chaparral (16.5%), gray pine (10.1%), chamise (8%), Oregon white oak (7.1%), interior mixed hardwood (6.6%), canyon live oak (5.6%), blue oak (5.6%), annual grasses and forbs (4.8%), Douglas-fir–ponderosa pine (4.4%), scrub oak (3.6%), Douglas-fir (3.5%), mixed conifer–pine (3.3%), Sargent cypress (3.2%), black oak (2.5%), knobcone pine (2.2%), ponderosa pine (2%)
Grand fir/white fir	Mixed pine conifer (27.1%), white fir (19%), Douglas-fir–white fir (14%), Douglas-fir (10.6%), Douglas-fir–ponderosa pine (6.3%), red fir (5.9%), mixed conifer–fir (2.5%), upper montane mixed chaparral (2%)
Douglas-fir	Douglas-fir (29.3%), Douglas-fir–ponderosa pine (13.3%), Oregon white oak (12.7%), mixed conifer–pine (7.8%), lower montane mixed chaparral (5.3%), canyon live oak (4.6%), black oak (4%), interior mixed hardwood (3.8%), ponderosa pine (3.2%), annual grasses and forbs (2%).
Juniper	Annual grasses and forbs (45.3%), mixed conifer–pine (17.2%), barren (8.3%), Douglas-fir–ponderosa pine (7%), upper montane mixed chaparral (4.3%), perennial grasses and forbs (2.9%), manzanita chaparral (2.8%), ponderosa pine–white fir (2.3%), Jeffrey pine (2%)

[a] Percentages indicate the percentage of the potential vegetation zone that falls into the CALVEG class. Existing vegetation comes from the Regional Dominance Type 1 field in the CALVEG database and indicates the primary, dominant vegetation alliance. The listed existing vegetation alliances comprise 95 percent of each potential vegetation zone in northern California. Current vegetation types with less than 2 percent cover in a potential vegetation zone are not shown. For information on CALVEG, see: http://www.fs.usda.gov/detail/r5/landmanagement/resourcemanagement/?cid=stelprdb5347192 .

Appendix 2: Fire Regime Mapping Method

Wildfire studies in Pacific Northwest forests have shown strong correlations between fire occurrence and area burned with summer temperature and precipitation (Dalton et al. 2013, Littell et al. 2009, McKenzie et al. 2004). Accordingly, we used climate variables for temperature and precipitation that coincided with the regional fire season as covariates in this mapping method. Our climate data source was the parameter-elevation regressions on independent slopes model (PRISM) climate normal data (PRISM 2015) for the period 1971–2000. We included a third variable for density of lightning-ignited wildfires data from 1970 to 2002 (Brown et al. 2002) because fires in some regions may be limited by lack of ignitions during dry periods. Each mapping variable was classified into categories based on the equal divisions of the distributions in the forested areas. Thus, each class covered a relatively equal proportion of the forested landscape. Temperature was divided into five classes, and the other two variables were divided into three classes (table 3-9).

Potential vegetation zones (potential vegetation types) were summarized across all combinations of variable classes. Review of these data (e.g., temperature, precipitations, lightning ignition, density, and vegetation types) and expert opinion were used to assign each variable combination to one of four fire regimes: (1) infrequent (>200-year return interval) stand replacing; (2) somewhat infrequent to moderately frequent (50- to 200-year return interval), mixed severity; (3) frequent (15- to 50-year return interval), mixed severity; and (4) very frequent (5- to 25-year return interval), low severity (table 3-10). The final map product was filtered to remove pixel noise using a 3 by 3 majority filtering process.

Table 3-9—Variable map classification scheme based on quantile (by forested area) breaks

Rank	July–August mean monthly maximum temperature	May–September mean monthly precipitation	Lightning ignition density 1970–2002
	°C	*Millimeters*	*Ignitions/km²*
Very low	15–23	NA	NA
Low	23–25	6–32	<0.05
Moderate	25–27	32–54	0.05–1.2
High	27–30	54–189	>1.2
Very high	30–37	NA	NA

NA = not applicable.

1.1. Infrequent (>200-year return intervals) stand replacing (Landfire group V)
    a. Potential vegetation type (PVT): wetter/colder parts of western hemlock, Pacific silver fir, mountain hemlock. Cover types: Douglas-fir, western hemlock, Pacific silver fir, noble fir, mountain hemlock
    b. Area dominated by large to very large patches (10^3 to 10^6 ac) of high-severity fire, low and moderate severity also occur. Small- to medium-size patches were most frequent.

1.2. Moderately frequent to somewhat infrequent (50- to 200-year return intervals) mixed severity (Landfire regime group III)
    a. PVT: drier/warmer parts of western hemlock, Pacific silver fir and others. Cover types: Douglas-fir, western hemlock, Pacific silver fir, noble fir.
    b. Mixed severity in space and time, typically including large (10^3 to 10^4 ac) patches of high-severity fire and areas of low- and moderate-severity fire. Small patches of high severity would be frequent.

1. Dry forests, primarily east side of Washington and Oregon, southwest Oregon, northwest California
    1.1. Frequent (15- to 50-year return intervals), mixed severity (Landfire regime group I and III)
        a. PVT: Douglas-fir, grand fir, white fir, tanoak. Cover type: Douglas-fir, white fir, red/noble fir, western white pine
        b. Mixed-severity fire with medium to large (10^2 to 10^4 ac) patches of high-severity fire
    1.2. Very frequent (5- to 25-year return intervals) low severity (Landfire regime group I)
        a. PVT: ponderosa pine, dry to moist grand fir, white fir. Cover types: ponderosa pine, Douglas-fir, mixed pine, oak
        b. Dominated by low-severity fire with fine-grained pattern (<10° to 10^2 ac) of high-severity fire effects, large patches of high-severity fire rare in forests except in earlier seral stage (e.g., shrub fields).

Table 3-10—Temperature, precipitation, and lightning class levels (see table 3-9) of fire regimes and percentage of vegetation zones in that set of environmental classes

Regime	Temp.	Precip.	Lightning	PISI	THPL	TSHE	CHLA	LIDE	SESE	ABAM	TSME	ABLA	ABMAS	PSME	ABGRC	PIPO	PINUS	OAK
Infrequent—high severity	Very low	Low	Low	0	0	0	0	0	12	0	0	0	0	0	0	0	0	0
Infrequent—high severity	Very low	Low	High	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Infrequent—high severity	Very low	Moderate	Low	14	1	0	0	0	2	0	0	0	0	0	0	0	0	0
Infrequent—high severity	Very low	Moderate	Moderate	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Infrequent—high severity	Very low	Moderate	High	0	0	0	0	0	0	3	7	31	1	0	1	0	0	0
Infrequent—high severity	Very low	High	Low	32	0	2	0	0	0	2	0	0	0	0	0	0	0	0
Infrequent—high severity	Very low	High	Moderate	2	0	1	0	0	0	32	15	13	3	0	0	0	0	0
Infrequent—high severity	Very low	High	High	0	0	0	0	0	0	5	10	9	0	0	0	0	0	0
Infrequent—high severity	Low	Low	Low	0	3	0	0	0	8	0	0	0	0	0	0	0	0	0
Infrequent—high severity	Low	Low	High	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Infrequent—high severity	Low	Moderate	Low	4	41	4	0	0	2	0	0	0	0	1	0	0	0	0
Infrequent—high severity	Low	Moderate	Moderate	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Infrequent—high severity	Low	Moderate	High	0	0	0	0	0	0	2	18	13	6	1	6	0	0	0
Infrequent—high severity	Low	High	Low	37	18	23	0	1	0	2	0	0	0	1	0	0	0	0
Infrequent—high severity	Low	High	Moderate	9	0	7	1	2	0	30	4	0	0	0	0	0	0	0
Infrequent—high severity	Low	High	High	0	0	1	0	0	0	7	21	0	0	0	0	0	0	0
Infrequent—high severity	Moderate	Low	Low	0	0	0	0	0	1	0	0	0	0	0	0	0	0	0
Infrequent—high severity	Moderate	Low	High	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Infrequent—high severity	Moderate	Moderate	Low	0	1	0	0	0	0	0	0	0	0	0	0	0	0	0
Infrequent—high severity	Moderate	Moderate	Moderate	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Infrequent—high severity	Moderate	Moderate	High	0	0	0	0	0	0	0	1	0	0	0	0	0	0	0
Infrequent—high severity	Moderate	High	Low	1	1	11	0	0	0	0	0	0	0	4	0	0	0	0
Infrequent—high severity	Moderate	High	Moderate	0	0	0	0	0	0	1	0	0	0	0	0	0	0	0
Infrequent—high severity	Moderate	High	High	0	0	0	0	0	0	1	1	0	0	0	0	0	0	0
Infrequent—high severity	High	Low	Low	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Infrequent—high severity	High	Moderate	Low	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Infrequent—high severity	High	High	Low	0	0	2	0	0	0	0	0	0	0	0	0	0	0	0
Infrequent—high severity	High	High	Moderate	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Infrequent—high severity	Very high	High	Low	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Infrequent—high severity	Very high	High	Moderate	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Moderately frequent—mixed severity	Very low	Low	Moderate	0	0	0	0	0	0	0	1	0	0	0	0	0	0	0

Table 3-10—Temperature, precipitation, and lightning class levels (see table 3-9) of fire regimes and percentage of vegetation zones in that set of environmental classes (continued)

Regime	Temp.	Precip.	Lightning	PISI	THPL	TSHE	CHLA	LIDE	SESE	ABAM	TSME	ABLA	ABMAS	PSME	ABGRC	PIPO	PINUS	OAK
Moderately frequent—mixed severity	Very low	Low	High	0	0	0	0	0	0	0	1	0	0	0	0	0	0	0
Moderately frequent—mixed severity	Very low	Moderate	Moderate	0	0	0	0	0	0	3	4	0	0	0	0	0	0	0
Moderately frequent—mixed severity	Very low	Moderate	High	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Moderately frequent—mixed severity	Very low	High	Moderate	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Moderately frequent—mixed severity	Very low	High	High	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Moderately frequent—mixed severity	Low	Low	Moderate	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Moderately frequent—mixed severity	Low	Low	High	0	0	0	0	0	0	0	1	0	0	0	0	0	0	0
Moderately frequent—mixed severity	Low	Moderate	Low	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Moderately frequent—mixed severity	Low	Moderate	Moderate	0	0	1	0	0	0	2	1	0	0	0	0	0	0	0
Moderately frequent—mixed severity	Low	Moderate	High	0	0	0	0	0	0	0	1	0	0	0	0	0	0	0
Moderately frequent—mixed severity	Low	High	Low	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Moderately frequent—mixed severity	Low	High	Moderate	0	0	1	0	0	0	0	0	0	0	0	0	0	0	0
Moderately frequent—mixed severity	Low	High	High	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Moderately frequent—mixed severity	Moderate	Low	Moderate	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Moderately frequent—mixed severity	Moderate	Low	High	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Moderately frequent—mixed severity	Moderate	Moderate	Low	0	14	7	0	0	0	0	0	0	0	0	0	0	0	0
Moderately frequent—mixed severity	Moderate	Moderate	Moderate	0	0	4	0	0	0	1	0	0	0	0	0	0	0	0
Moderately frequent—mixed severity	Moderate	Moderate	High	0	0	0	0	0	0	0	6	0	0	0	0	0	0	0

Table 3-10—Temperature, precipitation, and lightning class levels (see table 3-9) of fire regimes and percentage of vegetation zones in that set of environmental classes (continued)

Regime	Temp.	Precip.	Lightning	PISI	THPL	TSHE	CHLA	LIDE	SESE	ABAM	TSME	ABLA	ABMAS	PSME	ABGRC	PIPO	PINUS	OAK
Moderately frequent—mixed severity	Moderate	High	Low	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Moderately frequent—mixed severity	Moderate	High	Moderate	1	0	7	0	0	0	2	0	0	0	0	0	0	0	0
Moderately frequent—mixed severity	Moderate	High	High	0	0	3	0	0	0	4	3	0	0	0	0	0	0	0
Moderately frequent—mixed severity	High	Low	Low	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Moderately frequent—mixed severity	High	Low	Moderate	0	0	1	0	0	0	0	0	0	0	0	0	0	0	0
Moderately frequent—mixed severity	High	Low	High	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Moderately frequent—mixed severity	High	Moderate	Low	0	21	5	0	0	0	0	0	0	0	0	0	0	0	0
Moderately frequent—mixed severity	High	Moderate	Moderate	0	0	8	0	0	0	0	0	0	0	0	0	0	0	0
Moderately frequent—mixed severity	High	Moderate	High	0	0	3	0	0	0	0	2	0	0	0	0	0	0	0
Moderately frequent—mixed severity	High	High	Low	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Moderately frequent—mixed severity	High	High	Moderate	0	0	3	0	0	0	0	0	0	0	0	0	0	0	0
Moderately frequent—mixed severity	High	High	High	0	0	3	0	0	0	0	1	0	0	0	0	0	0	0
Moderately frequent—mixed severity	Very high	Low	Moderate	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Moderately frequent—mixed severity	Very high	Moderate	Moderate	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Moderately frequent—mixed severity	Very high	Moderate	High	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Moderately frequent—mixed severity	Very high	High	Moderate	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Moderately frequent—mixed severity	Very high	High	High	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Frequent—mixed severity	Very low	Low	Low	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Frequent—mixed severity	Very low	Low	Moderate	0	0	0	0	0	1	0	0	1	0	0	1	0	0	0

Table 3-10—Temperature, precipitation, and lightning class levels (see table 3-9) of fire regimes and percentage of vegetation zones in that set of environmental classes (continued)

Regime	Temp.	Precip.	Lightning	PISI	THPL	TSHE	CHLA	LIDE	SESE	ABAM	TSME	ABLA	ABMAS	PSME	ABGRC	PIPO	PINUS	OAK
Frequent—mixed severity	Very low	Low	High	0	0	0	0	0	0	0	0	4	0	0	1	0	0	0
Frequent—mixed severity	Very low	Moderate	Low	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Frequent—mixed severity	Very low	Moderate	Moderate	0	0	0	0	0	3	0	0	9	0	0	0	0	0	0
Frequent—mixed severity	Very low	Moderate	High	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Frequent—mixed severity	Very low	High	Moderate	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Frequent—mixed severity	Low	Low	Low	0	0	0	0	0	1	0	0	0	0	0	0	0	0	0
Frequent—mixed severity	Low	Low	Moderate	0	0	0	0	0	0	0	0	0	0	1	2	0	0	0
Frequent—mixed severity	Low	Low	High	0	0	0	0	0	0	0	0	10	67	1	4	0	0	0
Frequent—mixed severity	Low	Moderate	Low	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Frequent—mixed severity	Low	Moderate	Moderate	0	0	0	0	0	4	0	0	2	0	0	2	0	0	0
Frequent—mixed severity	Low	Moderate	High	0	0	0	0	0	0	0	0	0	0	0	1	0	0	0
Frequent—mixed severity	Low	High	Low	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Frequent—mixed severity	Low	High	Moderate	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Frequent—mixed severity	Low	High	High	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Frequent—mixed severity	Moderate	Low	Low	0	0	0	0	0	9	0	0	0	0	0	0	0	0	0
Frequent—mixed severity	Moderate	Low	Moderate	0	0	0	0	0	1	0	0	0	0	1	3	4	0	0
Frequent—mixed severity	Moderate	Low	High	0	0	0	0	0	0	0	1	1	0	0	0	0	0	0
Frequent—mixed severity	Moderate	Moderate	Low	0	0	0	0	0	2	0	0	0	0	12	0	0	0	0
Frequent—mixed severity	Moderate	Moderate	Moderate	0	0	0	0	2	4	0	0	0	0	0	1	0	0	0
Frequent—mixed severity	Moderate	Moderate	High	0	0	0	0	1	0	0	0	0	0	0	8	0	0	0
Frequent—mixed severity	Moderate	High	Low	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Frequent—mixed severity	Moderate	High	Moderate	0	0	0	13	3	1	0	0	0	0	0	0	0	0	0
Frequent—mixed severity	Moderate	High	High	0	0	0	0	0	0	0	0	0	0	0	1	0	0	0
Frequent—mixed severity	High	Low	Low	0	0	0	0	1	28	0	0	0	0	3	2	1	0	8
Frequent—mixed severity	High	Low	Moderate	0	0	0	0	1	1	0	0	0	0	3	7	19	0	1
Frequent—mixed severity	High	Low	High	0	1	0	0	0	0	0	0	0	0	0	0	0	0	0
Frequent—mixed severity	High	Moderate	Low	0	0	0	0	0	2	0	0	0	0	4	0	0	0	0
Frequent—mixed severity	High	Moderate	Moderate	0	0	0	0	8	4	0	0	0	0	2	2	0	0	0
Frequent—mixed severity	High	Moderate	High	0	0	0	0	3	0	0	0	0	0	0	10	0	0	0
Frequent—mixed severity	High	High	Low	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Frequent—mixed severity	High	High	Moderate	0	0	0	75	11	0	0	0	0	0	1	0	0	0	0
Frequent—mixed severity	High	High	High	0	0	0	0	1	0	0	0	0	0	0	1	0	0	0

Table 3-10—Temperature, precipitation, and lightning class levels (see table 3-9) of fire regimes and percentage of vegetation zones in that set of environmental classes (continued)

Regime	Temp.	Precip.	Lightning	PISI	THPL	TSHE	CHLA	LIDE	SESE	ABAM	TSME	ABLA	ABMAS	PSME	ABGRC	PIPO	PINUS	OAK
Frequent—mixed severity	Very high	Low	Low	0	0	0	0	1	12	0	0	0	0	12	0	0	21	59
Frequent—mixed severity	Very high	Low	Moderate	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Frequent—mixed severity	Very high	Low	High	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Frequent—mixed severity	Very high	Moderate	Low	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Frequent—mixed severity	Very high	Moderate	Moderate	0	0	0	0	13	0	0	0	0	0	2	1	0	5	0
Frequent—mixed severity	Very high	Moderate	High	0	0	0	0	0	0	0	0	0	0	0	1	0	0	0
Frequent—mixed severity	Very high	High	Moderate	0	0	0	11	5	0	0	0	0	0	0	0	0	0	0
Frequent—mixed severity	Very high	High	High	0	0	0	0	1	0	0	0	0	0	0	0	0	0	0
Very frequent—low severity	Very low	Low	High	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Very frequent—low severity	Low	Low	High	0	0	0	0	0	0	0	0	0	1	0	1	0	0	0
Very frequent—low severity	Low	Moderate	High	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Very frequent—low severity	Moderate	Low	Moderate	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Very frequent—low severity	Moderate	Low	High	0	0	0	0	0	0	0	0	3	22	2	10	9	0	0
Very frequent—low severity	Moderate	Moderate	High	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Very frequent—low severity	High	Low	Moderate	0	0	0	0	0	0	0	0	0	0	1	0	1	0	0
Very frequent—low severity	High	Low	High	0	0	0	0	3	0	0	0	1	0	7	17	48	0	0
Very frequent—low severity	High	Moderate	Moderate	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Very frequent—low severity	High	Moderate	High	0	0	0	0	0	0	0	0	0	0	0	1	0	0	0
Very frequent—low severity	High	High	High	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Very frequent—low severity	Very high	Low	Low	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Very frequent—low severity	Very high	Low	Moderate	0	0	0	0	10	0	0	0	0	0	14	2	14	55	26
Very frequent—low severity	Very high	Low	High	0	0	0	0	22	0	0	0	0	0	20	8	3	13	4
Very frequent—low severity	Very high	Moderate	Moderate	0	0	0	0	0	0	0	0	0	0	0	0	0	1	0
Very frequent—low severity	Very high	Moderate	High	0	0	0	0	5	0	0	0	0	0	5	3	0	5	1
Very frequent—low severity	Very high	High	High	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0

Temp. = temperature; Precip. = precipitation; ABAM = *Abies amabilis*; ABLA = *Abies lasiocarpa*; ABMAS = *Abies magnifica* var. *shastensis*; ABGRC = *Abies grandis/concolor*; CHLA = *Chamaecyparis lawsoniana*; LIDE = *Libocedrus decurrens*; OAK = *Quercus* spp.; PIPO = *Pinus ponderosa*; PINUS = *Pinus* spp.; PISI = *Picea sitchensis*; PSME = *Pseudotsuga menziesii*; SESE = *Sequoia sempervirens*; THPL = *Thuja plicata*; TSME = *Tsuga mertensiana*; TSHE = *Tsuga heterophylla*.

Appendix 3: Summary of Fire History Studies in the Northwest Forest Plan

Table 3-11—Fire history studies in the Northwest Forest Plan area by potential vegetation zone

Vegetation zone	Study	Extent/time period	Method	Frequency/return interval	Rotation	Low/moderate/high	High-severity patch size	Interpreted regime	Mapped regime
		Hectares		*Years*		*Percent*	*Hectares*		
Redwood:									
	Stuart 1987	300 ha 1898–1940	Scars	7.8	—	—	—	Low	F-MS
	Finney and Martin 1989	~600 ha 1300–1860	Scars	10.1	—	—	—	Low	I-HS
	Brown and Swetnam 1994	<1000 ha 1714–1962	Scars	9.9	—	—	—	Low	F-MS
	Brown et al. 1999	Unknown	Age, scars	7–13	—	—	—	Low	F-MS
	Brown and Baxter 2003	20 316 ha 1550–1930	Scars	6–20	—	—	—	Low	F-MS
Western hemlock:									
	Means 1982	Unknown	Scars	100	—	—		Mixed	MF-MS
	Fahnestock and Agee 1983	Western Washington pre-1934	Age class from historical survey records	—	598	—	—	High	I-HA
	Stewart 1986	<1 ha ~1200–1982	Age, scars	50[a]	—	—	—	Mixed	MF-MS
	Yamaguchi 1986	Unknown Post-1480	Age, scars	40–150	—	—	—	Mixed	I-HS
	Teensma 1987	11 000 ha 1482–1952	Age, Scars	114	78	—	—	Mixed	MF-MS
	Agee et al. 1990a or b?	3500 ha 1573–1985	Age, Scars	137	—	—	—	Mixed	I-HS
	Morrison and Swanson 1990	1940 ha 1150–1985	Age, Scars	96	95	0–86/ 0–60/ 0–100	<110 ha	Mixed/high	MF-MS
	Garza 1995	3540 ha Pre-1910	Age, scars	93-158	134	24–41/9–23/ 25–54		Mixed	MF-MS

Table 3-11—Fire history studies in the Northwest Forest Plan area by potential vegetation zone (continued)

Vegetation zone	Study	Extent/time period	Method	Frequency/ return interval	Rotation	Low/ moderate/ high	High-severity patch size	Interpreted regime	Mapped regime
		Hectares		*Years*		*Percent*	*Hectares*		
	Impara 1997	~140 000 ha 1478–1909	Age, scars	85	271	—	—	Mixed	MF-MS/ I-HS
	Wetzel and Fonda 2000	2500 ha 1400–1985	Age, growth release	21.3[b]	—	—	—	Low	I-HS
	Agee and Krusemark 2001	26 000 ha Pre-1900	Age, live residual structure from air photos	—	296	7–9/ 18–31/ 62–90	—	High	MF-MS
	Robbins 1999	~1562 km^2 1700–1990	Age, scars	49 (2–191)	—	—	—	Low/mixed	MF-MS
	Olsen and Agee 2005	~7000 ha 1650–1900	Age, scars	2-167	—	—	—	Mixed	MF-MS
	Weisberg 2009	14 504 ha 1550–1849	Age, scars	—	162	—	—	Mixed	MF-MS
	Wendel and Zabowski 2010	1873 ha 1568–2007	Age, scars	127	140	—	—	Mixed/high	I-HS
Silver fir:									
	Hemstrom and Franklin 1982	~53 000 ha 1200–1850	Age	—	465	—	—	High	I-HS
	Fahnestock and Agee 1983	Western Washington pre-1934	Age class from historical survey records	—	834	—	—	High	I-HS
	Agee et al. 1990a	3500 ha 1573–1985	Age, scars	108–137	—	—	—	Mixed	I-HS
	Morrison and Swanson 1990	1940 ha 1150–1985	Age, scars	239	149	0–80/ 0–78/ 0–100	<50	Mixed/high	MF-MS

Table 3-11—Fire history studies in the Northwest Forest Plan area by potential vegetation zone (continued)

Vegetation zone	Study	Extent/time period	Method	Frequency/ return interval	Rotation	Low/ moderate/ high	High-severity patch size	Interpreted regime	Mapped regime
		Hectares		*Years*		*Percent*	*Hectares*		
	Garza 1995	3540 ha Pre-1910	Age, scars	154–246	—	24–57/20–22/ 45–50	—	Mixed	MF-MS
Mountain hemlock:									
	Dickman and Cook 1989	18 000 ha Post-1400	Age	—	—	—	>3200	High	I-HS
	Fahnestock and Agee 1983	Western Washington pre-1934	Age class from historical survey records	—	598	—	—	Mixed	I-HS
	Agee et al. 1990a	3500 ha 1573–1985	Age, scars	137	—	—	—	Mixed	I-HS
Subalpine:									
	Fahenstock and Agee 1983	Western Washington pre-1934	Age class from historical survey records	—	800	—	—	High	I-HS
	Agee et al. 1990a	3500 ha 1573–1985	Age, scars	109	—	—	—	Mixed	I-HS
Douglas-fir and grand fir/white fir:									
	Leiberg 1903	Southern Oregon ~1900	Historical land survey	—	—		~14 000	High	MF-MS/ F-MS
	Weaver 1959	Unknown	Scars	47	—	—	—	Mixed	VF-LS
	Agee et al. 1990a	3500 ha 1573–1985	Age, scars	52-93	—	—	—	Mixed	I-HS
	Agee 1991	197 ha 1760–1930	Age, scars	16	37-64	—	—	Low/mixed	VF-LS
	Bork 1985	~100 ha Pre-1900	Scars	8	—	—	~400	Low	VF-LS
	Wills and Stuart 1994	~20 ha 1745–1849	Age scars	10.3–17.3	—	—	—	Low	VF-LS

Table 3-11—Fire history studies in the Northwest Forest Plan area by potential vegetation zone (continued)

Vegetation zone	Study	Extent/time period	Method	Frequency/ return interval	Rotation	Low/ moderate/ high	High-severity patch size	Interpreted regime	Mapped regime
		Hectares		*Years*		*Percent*	*Hectares*		
	Taylor and Skinner 1998	1570 ha 1627–1849	Age, scars	14.5	19	59/ 27/ 14	—	Low/ mixed	VF-LS
	Van Norman 1998	45 000 ha 1480–1996	Age, scars	123	—	—	—	Mixed	MF-MS
	Brown et al. 1999	2000 ha 1820–1945	Age, scars	7.7–13	—	—	—	Low	F-MS
	Everett et al. 2000	3240–12 757 ha ~1700–1860	Scars	6.6–7	11–12.2	—	2.4–40	Low	F-MS
	Stuart and Salazar 2000	~120 ha 1614–1944	Age, scars	27 (12–161)	—	—	—	Low	VF-LS
	Taylor and Skinner 2003	2325 ha Pre-1905	Age, scars	11.5–16.5	19	–	—	Low/mixed	VF-LS
	Wright and Agee 2004	~30 000 ha 1562–1995	Scars	19–24	—	—	10-100	Low/mixed	MF-MS
	Hessburg et al. 2007	~72 000 ha ~1930	Historical aerial photos	—	—	18/ 58/ 24	~10 000	Mixed	MF-MS/ F-MS
	Baker 2012	140 400 ha ~1770–1880	Live structure from historical inventory	—	496[c]	18/ 59/ 23	—	Mixed	F-MS/VF-LS
Ponderosa pine:									
	Weaver 1959	Unknown	Scars	11–16	—	—	—	Low	VF-LS
	Soeriaatmadja 1966	1500–5000 ha Unknown	Scars	3–36	—	—	—	Low	VF-LS
	West 1969	Unknown	Age	—	—	—	<0.26	Low	VF-LS

Table 3-11—Fire history studies in the Northwest Forest Plan area by potential vegetation zone (continued)

Vegetation zone	Study	Extent/time period	Method	Frequency/ return interval	Rotation	Low/ moderate/ high	High-severity patch size	Interpreted regime	Mapped regime
		Hectares		*Years*		*Percent*	*Hectares*		
	Bork 1985	~100 ha Pre-1900	Scars	4–7	—	—	—	Low	VF-LS
	Morrow 1985	2 ha Pre-1900	Age	—	—	—	<0.35	Low	VF-LS
	Hessburg et al. 2007	~106 000 ha 1930–1940	Live structure from historical aerial photos	—	—	30/ 58/ 12	—	Low/mixed	VF-LS
	Baker 2012	123 500 ha ~1770–1880	Live structure from historical inventory	—	705[c]	40/ 44/ 16	—	Low/mixed	VF-LS

— = No value in cell

[a] Interpreted regimes are classified on fire frequency classes: low <35 years, mixed 35 to 200 years, high >200 years). In cases in which fire frequency was not available, we considered fire rotation and the percentage of high-severity fire. Mapped regimes are predicted from combinations of summer precipitation, summer temperature, and lighting frequency and follow the four class regimes used in this chapter: F-MS is frequent–mixed severity, I-HS is infrequent–high severity, MF-MS is moderately frequent–mixed severity, and VF-LS is very frequent–low severity. 1 ha = 2.47 ac.

[b] Stewart noted 15 fires over a 750-year period.

[c] Estimated at a 200-ha (494 ac) scale.

[d] Rotation for high-severity only.

A northern spotted owl in the McKenzie River Basin in Oregon.
Photo by John and Karen Hollingsworth, U.S. Fish and Wildlife Service.

Chapter 4: Northern Spotted Owl Habitat and Populations: Status and Threats

Damon B. Lesmeister, Raymond J. Davis, Peter H. Singleton, and J. David Wiens[1]

Introduction

The northern spotted owl (*Strix occidentalis caurina*) was listed as threatened under the Endangered Species Act in 1990 (USFWS 1990). Providing adequate amounts of suitable forest cover to sustain the subspecies was a major component of the first recovery plan for northern spotted owls (USFWS 1992) and a driver in the basic reserve design and old-forest restoration under the Northwest Forest Plan (NWFP, or Plan) (USDA and USDI 1994). The reserve design included large contiguous blocks of late-successional forest, which was expected to be sufficient to provide habitat for many interacting pairs of northern spotted owls. As such, the selection of reserves generally favored areas with the highest quality old-growth forests, but some areas of younger forest were also included with the expectation that they would eventually develop suitable forest structure characteristics and contribute to spatial patterns that would sustain spotted owl populations.

Northern spotted owls are now one of the most studied birds in the world. Much of the research and interest in spotted owls stem from the economic and ecological implications surrounding management for the subspecies. Courtney et al. (2004) and the U.S. Fish and Wildlife Service (USFWS 2011b) completed comprehensive reviews and syntheses of scientific information regarding the status, ecology, and threats to the northern spotted owl. In the 10-year science synthesis of the NWFP, Raphael (2006) detailed the expectations and observations for northern spotted owl populations and suitable forest types under the Plan. Here we provide a 20-year synthesis of northern spotted owl science and review key information concerning the ecology and expectations for conservation of northern spotted owls under the NWFP. We build upon previous syntheses and address guiding questions by focusing on the scientific understanding accumulated from 2005 to 2016 on the ecology, conservation, and management of northern spotted owls. We also provide an overview of the main scientific debates surrounding conservation and management of northern spotted owls. We discuss the distinction between associated forest cover types and the relative value of habitat in different forest types for the subspecies. Where needed, we review and draw inference from research related to Mexican spotted owls (*S. o. lucida*) and California spotted owls (*S. o. occidentalis*), but keep the focus of this synthesis on published literature specific to northern spotted owls (spotted owl hereafter).

Major threats to spotted owls identified at the time of design and initial implementation of the NWFP and species recovery plan included the effects of past and current timber harvest, loss of old forest to wildfire, and competition with rapidly encroaching barred owls (*Strix varia*) (USDA and USDI 1994, USFWS 1992). Studies of associations between spotted owls and forest cover published since 2005 have reinforced previous work indicating a strong association of nest and roost sites with older forest conditions and a wider range of forest cover types used for foraging and dispersal (Anthony et al. 2006; Carroll and Johnson 2008; Dugger et al. 2005, 2016; Forsman et al. 2011, 2015; Hamer et al. 2007; Irwin et al. 2012, 2013; McDonald et al. 2006; Olson et al. 2005; Sovern et al. 2015). In the southern portions of the range, abiotic environmental factors begin to play larger roles in territorial owl use (Glenn et al. 2017), and at the very southern end of the range (Marin County, California), spotted owls occur at higher densities and tend to nest in a wider variety of forest cover types and ages (Stralberg et al. 2009). The difference in localized spotted owl densities and generalist vegetation associations appear to be driven by the diversity of forest conditions and high prey density prevalent in that landscape.

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Every study that has assessed rangewide population trends of spotted owls found steady declines since standardized monitoring efforts started in 1985 (Anthony et al. 2006, Dugger et al. 2016, Forsman et al. 2011, Franklin et al. 1996). Loss of suitable forest and competitive interactions with barred owls are the primary threats that have contributed to those declines. In the following sections, we review recent information on the status and trends of spotted owl populations and suitable forest, effects of interactions with barred owls, prey ecology, disturbance impacts, climate change, and other threats. We also review population trends and range expansion of barred owls, their habitat and prey, and identify other sensitive wildlife and ecological processes that may ultimately be affected by the invasion of barred owls. We conclude by outlining considerations for management and research needs for spotted owls and forest types most critical to their persistence.

Guiding Questions

We used the following questions received from forest managers to guide our synthesis and focus on relevant spotted owl literature. Following each question, we provide the section that most effectively addresses the question, or if a question could not be adequately addressed because of a lack of published literature on the subject.

1. What is the current understanding about spotted owl population status? Will continuing to implement the NWFP reverse the downward trend in spotted owl populations?
    - Information can be found in the “Population Status and Trends” and “Conclusions and Management Considerations” sections.
2. Is the NWFP maintaining or restoring forest conditions necessary to support viable populations of spotted owls?
    - Despite old-forest loss to wildfire and timber harvest, implementation of the NWFP has been successful for putting federal lands on a trajectory for restoring forest capable of supporting spotted owls on federal lands. Information can be found in the “Habitat Status and Trends,” “Disturbance,” and “Conclusions and Management Considerations” sections.
3. What are the effects of various timber management practices and wildfire on forests used by spotted owls?
    - Information can be found in the “Habitat Status and Trends,” “Disturbance,” and “Research Needs” sections.
4. How is space use by spotted owls affected by timber management? Are there ways to modify management activities (i.e., silvicultural treatments) to benefit spotted owls? How do managed stands compare to untreated forests in terms of use by spotted owls?
    - Information can be found in the “Habitat Status and Trends,” “Disturbance,” “Research Needs,” and “Conclusions and Management Considerations” sections.
5. Do spotted owls use forests following wildfire? If so, how? Do the impacts of treatments that reduce risk of wildfire outweigh the risks of suitable forest loss resulting from wildfire?
    - The short- and long-term response by spotted owls to wildfire remains largely unknown, and scientific debate remains. We were unable to fully address this question, but do provide a synthesis of available literature in the “Habitat Status and Trends,” “Disturbance,” “Research Needs,” and “Scientific Uncertainty” sections.
6. How effective are protections for buffered areas around nest sites in retaining spotted owls across treated landscapes? Are site buffers equally effective as landscape-scale forest management in ensuring species persistence, dispersal, and habitat connectivity?
    - We were unable to address this question fully owing to the lack of published literature, but some information about the effectiveness of buffered management areas can be found in the “Habitat Status and Trends,” and “Forest protection effectiveness” sections.

7. Which provides a higher level of spotted owl persistence: the current spotted owl critical habitat or the NWFP late-successional reserve network?
    - Information can be found in the "Habitat Status and Trends," and "Forest protection effectiveness" sections.
8. Does treating late-successional stands improve spotted owl persistence when wildfire, insects, disease, and climate change threaten the ability of these forests to provide habitat for spotted owls?
    - Information can be found throughout the chapter in the "Habitat Status and Trends," "Barred Owls," "Disturbance," "Climate Change," "Other Threats," "Research Needs," "Scientific Uncertainty," and "Conclusions and Management Considerations" sections.
9. What are the effects of barred owls on spotted owls? What is the relationship of wildfires to barred owl encroachment? Can a barred owl management program be effectively implemented at a scale that will have meaningful conservation value for spotted owls?
    - Information about the effects of barred owls is found in "Barred Owls." We were unable to adequately address questions about the relationship between barred owls and wildfire, and barred owl management, because of a paucity of literature. In addition, some of this research was ongoing at the time this synthesis was being prepared. We provide further details in "Research Needs, Uncertainties, Information Gaps, and Limitations."
10. What are the management considerations and research needs for spotted owls?
    - Based on our synthesis of available literature within the context of the guiding management questions we received, we specifically address high-priority information needs in "Research Needs, Uncertainties, Information Gaps, and Limitations." We conclude the chapter with "Conclusions and Management Considerations."

Key Findings

Population Status and Trends

Understanding vital rates (e.g., birth, death) and the factors affecting those parameters over time and space can provide crucial information for management and conservation. Since the listing of the spotted owl, demographic rates have been monitored in up to 14 demographic study areas distributed across the spotted owl's geographic range. Franklin et al. (1996) developed a general framework to estimate demographic parameters and population trends of spotted owls that has been used in subsequent spotted owl population analyses. In the past 10 years, three meta-analyses (Anthony et al. 2006, Dugger et al. 2016, Forsman et al. 2011) documented a continued decline in spotted owl populations throughout their range. Those meta-analyses built upon the Franklin et al. (1996) methods to analyze survival, reproduction, and territory occupancy data that has been collected consistently for nearly three decades.

The number of study areas in which spotted owls have been monitored has changed through time owing to changes in funding and institutional support. Anthony et al. (2006) used data from 14 study areas (1985 to 2003), Forsman et al. (2011) used data from 11 study areas (1985 to 2008), and Dugger et al. (2016) used data from 11 study areas (1985 to 2013) to evaluate survival, fecundity, recruitment, and rate of population change of spotted owls throughout the subspecies' geographic range (fig. 4-1). Dugger et al. (2016) also investigated territory occupancy dynamics (gains and losses of occupied territories; i.e., local colonization and extinction rates). All three meta-analyses investigated relationships between population demography of spotted owls and the distribution of suitable forest cover types, local and regional variation in climatic conditions, and presence of barred owls. Study areas included in these meta-analyses comprised about 9 percent of the spotted owl's range, were distributed throughout the geographic range, and were selected to encompass the broad range of forest conditions used by the subspecies.

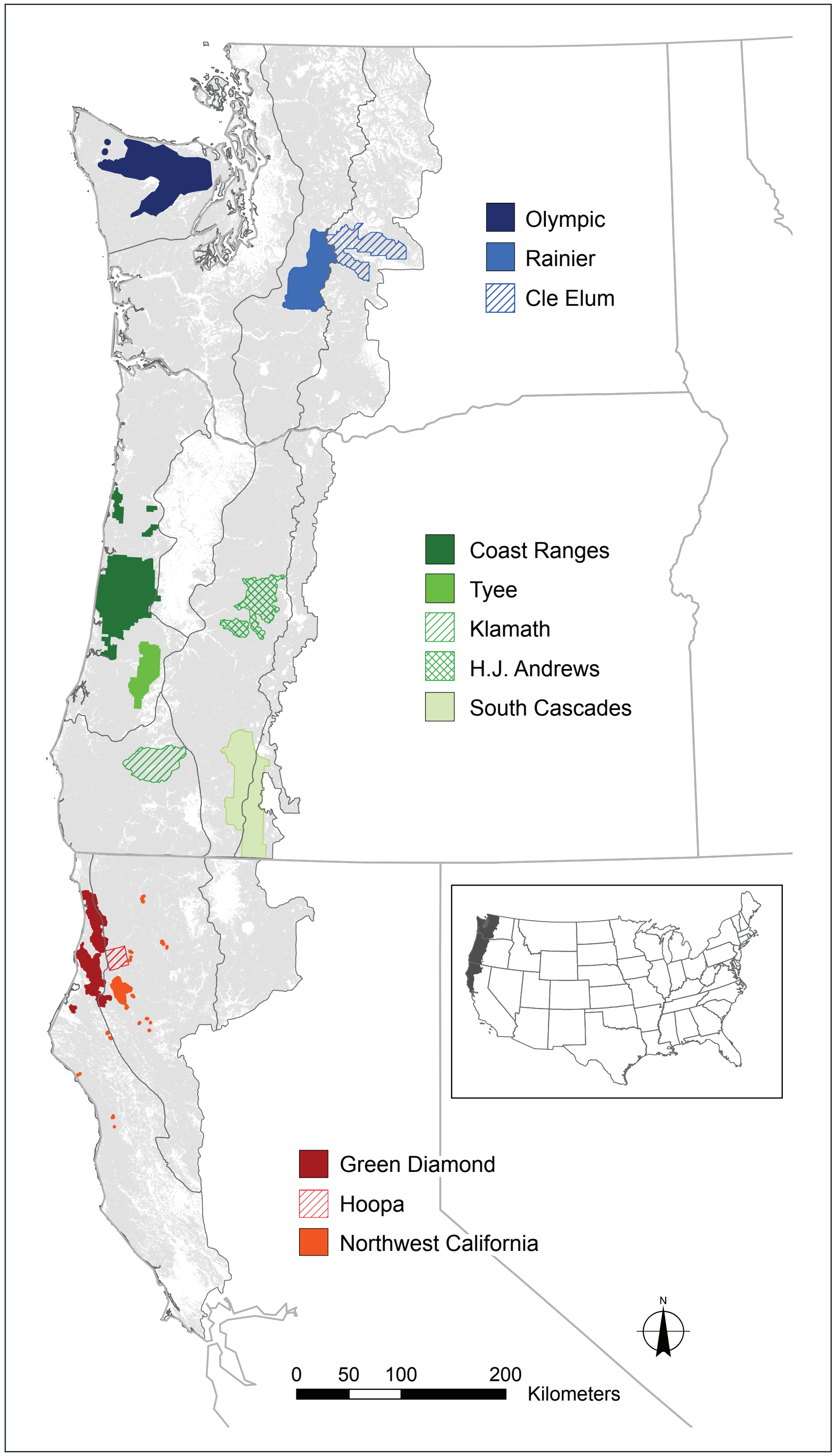


Figure 4-1—Locations of 11 study areas used in the analysis of vital rates and population trends of northern spotted owls, 1985 to 2013 (Dugger et al. 2016).

When the NWFP was developed, populations of spotted owls were estimated to be declining at about 4.5 percent (confidence interval [CI] 1.1 to 7.9) per year (Burnham et al. 1996, USDA and USDI 1994). The population was expected to continue declining for up to 50 years until younger second-growth forest in reserves matured to a point at which it would provide suitable structural conditions for nesting and roosting (Lint 2005, USDA and USDI 1994). During the first 10 years of the NWFP, the overall rate of population decline in Washington was much greater than in Oregon and California (Anthony et al. 2006, Lint 2005). Three study areas in southern Oregon had stable populations during the first decade. Anthony et al. (2006) estimated an annual decline of 3.7 percent (CI = 1.9 to 5.5) across the range, but that analysis included lands outside of the NWFP monitoring area. The eight federal study areas within the boundaries of the NWFP area (i.e., lands under federal management) used for effectiveness monitoring of the NWFP had a decline of 2.4 percent (CI = 1.0 to 3.8) compared to a 5.8 percent (CI = 2.6 to 9.0) decline for study areas composed primarily of nonfederal lands, suggesting that implementation of the NWFP had a positive effect on the demography of spotted owls (Anthony et al. 2006, Raphael 2006). Forsman et al. (2011) estimated an annual decline of 2.9 percent (CI = 1.7 to 4.0) throughout the northern spotted owl's range, and Davis et al. (2011) estimated an annual decline of 2.8 percent (CI = 1.5 to 4.2) within the eight federal study areas. The most recent meta-analysis indicated that spotted owl populations were continuing to decline throughout the range of the subspecies, and that annual rates of decline were accelerating in many areas (Dugger et al. 2016). The population was declining by about 3.8 percent (CI = 0.1 to 7.5) per year and declines ranged from 1.2 percent to 8.4 percent per year depending on the study area (fig. 4-2) (Dugger et al. 2016). For monitored populations, population change was more sensitive to adult survival than to recruitment (Glenn et al. 2010). Other studies have also documented declines in populations throughout the range of the spotted owl (Farber and Kroll 2012, Funk et al. 2010, Kroll et al. 2010).

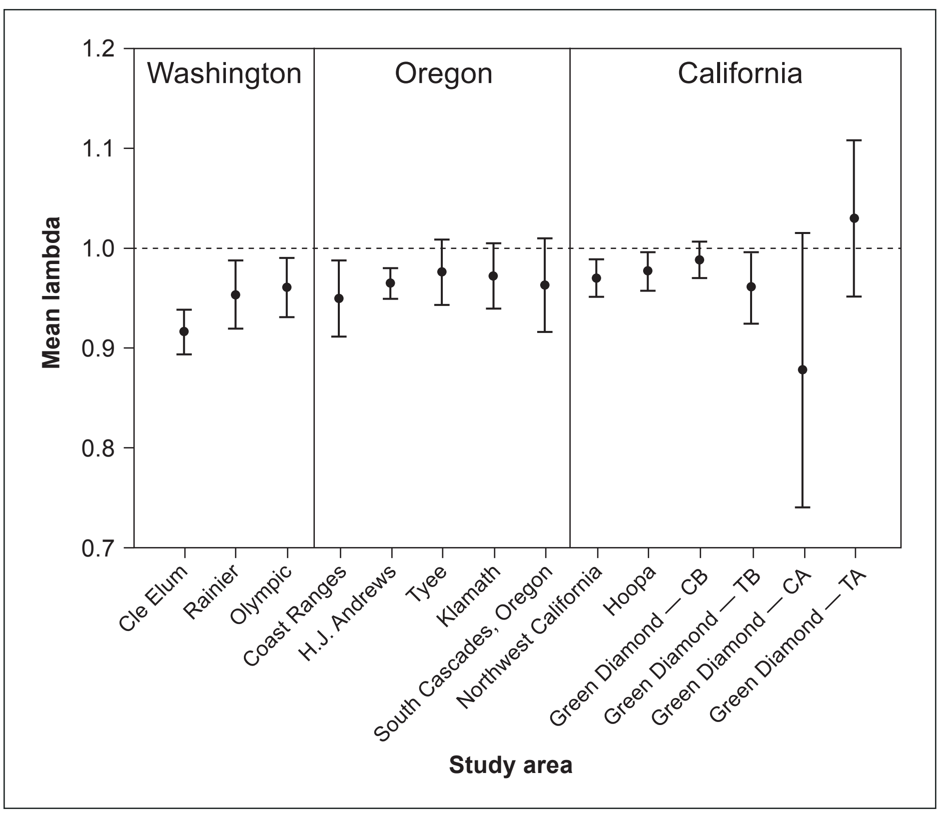


Figure 4-2—Estimated mean rates of population change (mean lambda) and 95 percent confidence limits for spotted owls from 1985 to 2013 at 11 sites: Cle Elum, Rainier, and Olympic, Washington; Coast Range, H.J. Andrews, and Tyee, Klamath, and South Cascades, Oregon; and northwest California, Hoopa, and Green Diamond, California (from Dugger et al. 2016). Estimates for Green Diamond are presented separately for control and treatment areas before (1990 to 2008) and after (2009 to 2013) barred owls were removed on the treatment area (CB = control before removal; TB = treatment before removal; CA = control after removal; TA = treatment after removal) (Diller et al. 2016, Dugger et al. 2016).

Habitat Status and Trends

Background and definitions—

Habitat for a species is an area that encompasses the necessary combination of resources and environmental conditions that promotes occupancy, survival, and reproduction of that species (Morrison et al. 2006). Typical wildlife habitat components include food, water, shelter (including nesting or denning sites), security from predators and competitors, and proper spatial arrangement of those features (Morrison et al. 2006). Although this concept of habitat may seem simple, the ways in which these individual components and animal needs interact in space and time result in very complex relationships (Mathewson and Morrison 2015).

Spotted owl habitat has often been characterized as older forest with large trees and moderate to closed canopy (Courtney et al. 2004, Forsman et al. 1984). Spotted owl site occupancy has repeatedly been shown to be influenced by the presence of these forest conditions (e.g., Dugger et al. 2016), likely because they often provide important habitat components that are suitable for nesting (e.g., cavities or platforms) (Sovern et al. 2011), abundant prey populations (Carey et al. 1992, Forsman et al. 2004, Wilson and Forsman 2013), and security from predators, including other raptors (Forsman et al. 1984, Sovern et al. 2014). An advantage of characterizing spotted owl habitat based on forest structure is that these forest types can be mapped for the entire subspecies' range using remotely sensed data (Davis et al. 2016). Other habitat components like prey abundance, predation risk, and presence of competitors are much more difficult, if not impossible, to map independently. For example, the recent colonization of the range of northern spotted owls by barred owls has confounded efforts to quantify the amount of habitat available for spotted owls because barred owls use similar forest types and can displace spotted owls from those areas (see "Barred Owl" section below).

In addition to availability, the arrangement of habitat components at a variety of scales is also important for understanding spotted owl habitat. Typically, spotted owl habitat is discussed in terms of forest cover types (stand-level forest structure and composition) most suitable for nesting, roosting, foraging, or dispersal (Davis et al. 2016, Lint 2005, Thomas et al. 1990). However, the spatial and temporal dynamics of suitable forest cover types, and how environmental conditions including climate and topography interact with vegetation patterns, are also important for producing and sustaining habitat for spotted owls (USFWS 2012a, 2012b). For example, Glenn et al. (2017) constructed habitat models using forest cover types and abiotic environmental conditions, and estimated the density of spotted owl territories on a landscape before and after barred owl invasion.

In this chapter, we define spotted owl habitat as those areas with the full suite of resources (e.g., abundant prey, available nest structures) and environmental conditions (e.g., appropriate climate, suitable forest structure, and infrequent presence of barred owls) suitable for occupancy, reproduction, and survival of the subspecies. As such, habitat is more analogous to a species' realized niche rather than the fundamental niche because habitat is more constrained than the availability of a vegetation type and a subset of environmental conditions. All published models of spotted owl habitat fall short of this definition because the distribution of spotted owls in relation to abundant prey is not known, and the distribution of an important competitor—barred owls—is not fully known. Throughout this chapter we distinguish between spotted owl habitat and components of that habitat (e.g., forest cover types used for nesting and roosting) regardless of the terms used in published literature.

Differing concepts regarding habitat definitions have long caused confusion and uncertainty in the interpretation of scientific literature (Bamford and Calver 2014, Hall et al. 1997, Morrison et al. 2006). The differences in how spotted owl habitat is defined and modeled has also caused confusion. The NWFP monitoring program estimates trends in forest types used by spotted owls (Davis and Lint 2005). The Fish and Wildlife Service (USFWS 2012a) modeled suitable forest and considered the amount and spatial arrangement of forests associated with specific life history requirements (e.g., forest types used for foraging in relation to forests used for nesting and roosting), as well as abiotic factors (e.g., slope, climate). The resulting models were used for delineation and designation of what was considered critical habitat (USFWS 2011b). The models of potential spotted owl habitat developed by the NWFP monitoring program and the Fish and Wildlife Service have important differences that result in different amounts of what is considered suitable forest for spotted owls. Estimates of the amount of suitable forest for spotted owls are highly scrutinized because of the conflict caused by the importance of that forest type for the reproduction and survival of spotted owls and because merchantable large timber is important economically for many of the rural areas where old forest occurs. The different estimates of suitable forest cover for spotted owls resulted in litigation filed in relationship to critical habitat designation. Carpenters Industrial Council et al. vs. Ashe and Salazar (District of Columbia District Court case number 1:2012cv00111 filed January 24, 2012) claimed that the USFWS (2012a) estimate of approximately 18 million ac (7.3 million ha) of suitable forest conditions (they used the term habitat) for spotted owls was an overestimate of 5.9 million ac (2.4 million ha) because previous documents produced by the agency had used estimates of approximately 12.1 million ac (4.9 million ha) as found in (Davis et al. 2011). The 2.4 million ha difference can be explained by an examination of how habitat was defined and modeled in the different efforts. For example, estimates from Davis et al. (2011) were based on a stand-level designation of forest cover suitable for nesting and roosting (fig. 4-3A), whereas USFWS (2011b) and USFWS (2012a) delineated critical habitat based on a model that included suitable forest stands (Davis et al. 2011) and other landscape components essential for spotted owls at the core-area scale (200 ha) (fig. 4-3B).

The NWFP defined suitable forest for spotted owls as an area with the species of trees, structure associated most commonly with late-successional forest, sufficient area, and adequate food source to meet some or all of the subspecies' life needs, including nesting, roosting, and foraging (USDA and USDI 1994). This definition relied heavily on the work in the Interagency Scientific Committee report (Thomas et al. 1990), which acknowledged the difficulty in defining habitat and chose to characterize the concept based on relative value or suitability of forest stands for spotted owls. Forest cover can be viewed as supporting different spotted owl life functions (e.g., nesting, roosting, foraging) and a suitability gradient in terms of its influence on individual fitness (Thomas et al. 1990). Partitioning of forest cover

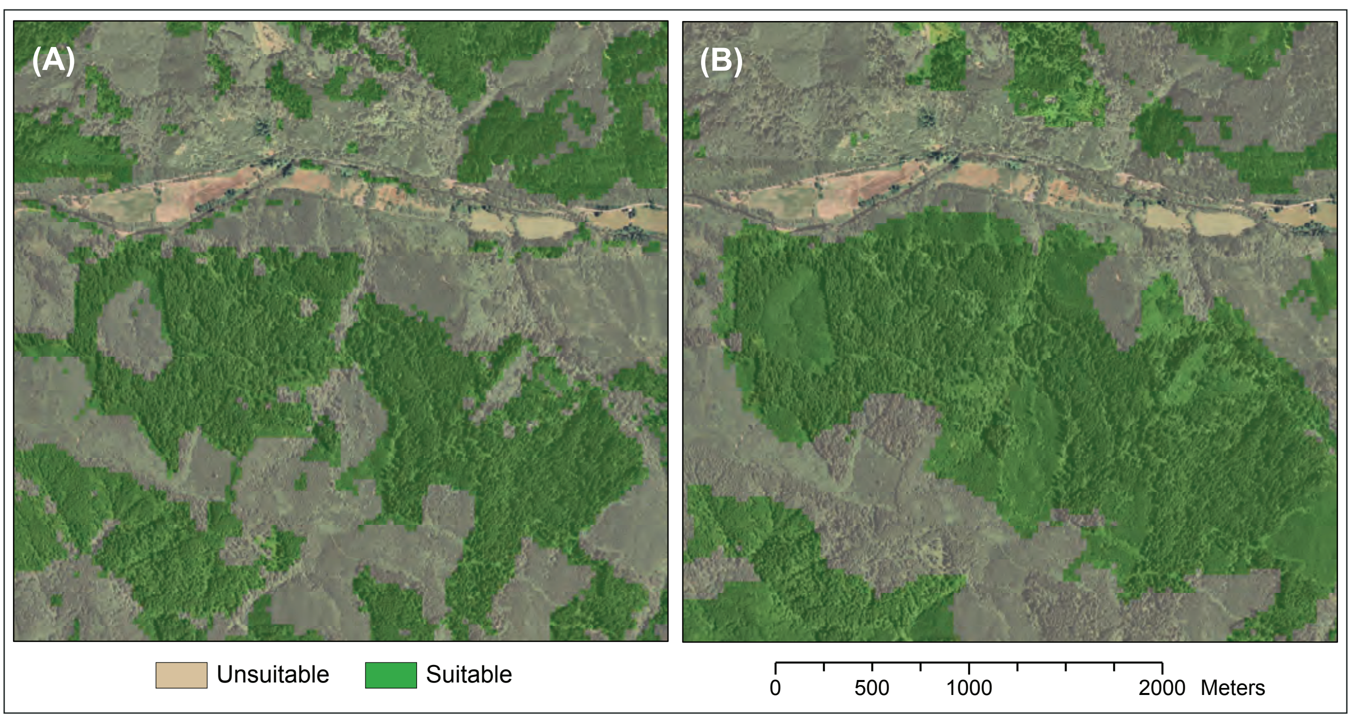


Figure 4-3—Examples of the suitable forest cover at (A) the stand scale developed by the Northwest Forest Plan monitoring program (Davis et al. 2011), and (B) the 200-ha (~250-foot radius) core-area scale used for modeling and delineating critical habitat (USFWS 2012a).

into discrete categories based on established measures of suitability for particular life functions facilitates a common frame of communication and standardization. A monitoring framework to measure relative suitability of forest cover types used by spotted owls was developed as part of a rangewide monitoring program for the subspecies (Davis et al. 2011, 2016; Lint 2005). Monitoring divided a continuous gradient of cover-type suitability into four discrete classes (table 4-1), based on use-versus-availability analyses using documented territorial pair locations. The unsuitable class was used for nesting and roosting by spotted owls less than expected by chance based on availability, the marginal class was used in proportion to its availability, the suitable class was used more often than expected by chance, and the highly suitable class was used much higher than one would expect from chance based on its availability. For monitoring purposes that dates to the life of the NWFP, the suitable and highly suitable classes were combined into a single class to identify forests that were most strongly associated with nesting and roosting locations. Thomas et al. (1990) characterized highly suitable forest cover as forests that include a multilayered, multispecies canopy dominated by large (>30 inch diameter at breast height [d.b.h.]) conifer trees; an understory of shade-tolerant conifers or hardwoods; moderate to high (60 to 80 percent) canopy cover (they used the term closure, but by definition they had described cover) (Jennings et al. 1999); substantial decadence in the form of large, live coniferous trees with deformities (e.g., cavities, broken tops, and dwarf mistletoe infections); numerous large snags; large accumulations of logs; and other woody debris. The unsuitable or marginal classes do not imply unimportance to spotted owls because the classification was restricted to describe only suitability for nesting and roosting activities by spotted owls. The marginal class is likely important for supporting dispersal, foraging, and nonbreeding (i.e., floater) individuals that can replace adult mortality and dispersal at nesting territories. Likewise, unsuitable and marginal classes may be important forest types for many prey species used by spotted owls. Forests that are suitable for nesting and roosting have similar characteristics throughout the range of spotted owls, but the path of development to those conditions typically differ based on the fire regime within the area (chapter 3; table 4-2, fig. 4-4).

Thomas et al. (1990) defined forest suitable for dispersal as having ≥11 inch (28 cm) d.b.h. trees and ≥40 percent canopy cover occurring on ≥50 percent of a 36 mi^2 township; this definition became known as the 50/11/40 rule. Analyses of movement data of spotted owls suggest that most (90 percent) dispersal occurred through landscapes meeting these criteria and are generally considered

Table 4-1—General descriptions of forest cover type classes used to estimate the amount of suitable forest available for nesting and roosting by spotted owls.

Cover type class	General description
Unsuitable	Younger forests or older forests with higher basal area of pine or high-elevation tree species or more open canopies. Usually smaller than average tree diameters, and lacking the presence of residual large trees and multiple canopy layers.
Marginal	Usually mid-seral forests, but can also be older forests lacking large-diameter trees, having simpler stand structure, or primarily composed of pine or high-elevation tree species.
Suitable	Forest stands older than 125 years of age, except in the California redwoods, where younger stands are used. Average tree diameters are usually above 20 inches (50 cm) d.b.h., with the presence of at least a few large trees exceeding 30 inches (75 cm) d.b.h. Canopy cover is usually greater than 60 percent, and the stand has multiple canopy layers.
Highly suitable	Typically forests 150 and 200 years of age or older. Average tree diameters often in excess of 30 inches (75 cm) d.b.h. except in drier portions of the range, where tree ages and sizes are typically smaller (e.g., 120 years and 24 inches). Canopy cover is usually in excess of 70 percent, and the stand has multiple canopy layers with high diversity of tree sizes.

d.b.h. = diameter at breast height.
Source: Davis et al. 2016.

Table 4-2—General descriptions of how forest cover types suitable for nesting and roosting by spotted owls typically develop within four general fire regimes within the Northwest Forest Plan (NWFP) area

Fire regime	Typical development of suitable nesting/roosting forest
Infrequent—high severity (Coast Range, fig. 4-4)	Large contiguous patches that form following infrequent, yet very large, high-severity wildfires. Once established, these large patches persist for long periods until the next large high-severity wildfire. Immediately following a large wildfire, large areas of the landscape are unsuitable for nesting and roosting for decades until closed canopies redevelop in areas that had remnant tree structures that could serve as nest trees. During this period, fine-scale gaps created by root-rot pockets, windstorms, landslides, and other small-scale processes produce complex stand structure. Complex structure sometimes does not develop over large areas for several decades following a wildfire. Produces the largest diameter and tallest nest trees; nests are usually in cavities or broken tops.
Moderately frequent—mixed severity (West Cascades, fig. 4-4)	Abundant to moderately abundant on the landscape, but very well connected across the landscape owing to the lack of extremely large high-severity wildfire patches. High-severity wildfire created smaller patches of complex early-seral forest cover type within an otherwise older forest matrix. Through time, these wildfire-created patches produced complex forest structure at the stand scale and a diverse mosaic of seral stages at the landscape scale.
Frequent—mixed severity (Klamath Mountains, fig. 4-4)	Moderately abundant on the landscape but more confined to topographic positions that functioned as wildfire refugia (e.g., lower slopes, north aspects, etc.). These areas allowed for the development and persistence of large trees required for nesting structures. In the Klamath Mountains and California Coast Range physiographic provinces, evergreen hardwoods (e.g., tanoak) are an important component that increase the suitability of use in these stands. In addition to forest stand structure and species composition, climate, and topography are important predictors of use by spotted owls.
Very frequent—low severity (East Cascades, fig. 4-4)	Not naturally abundant within the NWFP area; primarily restricted to the east side of the Cascade Mountains and eastern parts of northern California. Occurred historically in areas where the topography or soil conditions created a productive environment suitable for the development of large Douglas-fir and grand fir. Once established, these closed-canopy, structurally complex forest cover conditions can be relatively resistant to most fires, but burn with high severity under extreme weather conditions (chapter 3).

capable of supporting dispersal (Davis et al. 2011, 2016; Forsman et al. 2002; Lint 2005). However, the Thomas et al. (1990) 50/11/40 hypothesis was not based on juvenile resource selection data and remains largely untested. Only two studies (Miller et al. 1997, Sovern et al. 2015) have empirically studied forest-type selection during juvenile dispersal. Both studies found that juveniles strongly select for old forest with closed canopy (>70 percent canopy cover) and large-diameter trees (>20 inch d.b.h.), which are similar forest conditions selected by adult spotted owls for nesting and roosting (Miller et al. 1997, Sovern et al. 2015). Given the importance of forest cover classified as suitable for nesting and roosting to juvenile dispersal, the canopy cover recommendations of Thomas et al. (1990) are unlikely to be sufficient to facilitate juvenile movements on the landscape. Sovern et al. (2015) suggested that stands managed for dispersing spotted owls should be at least 80 percent canopy cover and have large average tree diameter.

Both the nesting/roosting and dispersal maps of suitable cover types produced by the NWFP monitoring program were designed to match the conceptual descriptions of forest vegetation components defined by Thomas et al. (1990) and used at the time of the NWFP development. Mapping of forests used by spotted owls is continuing to

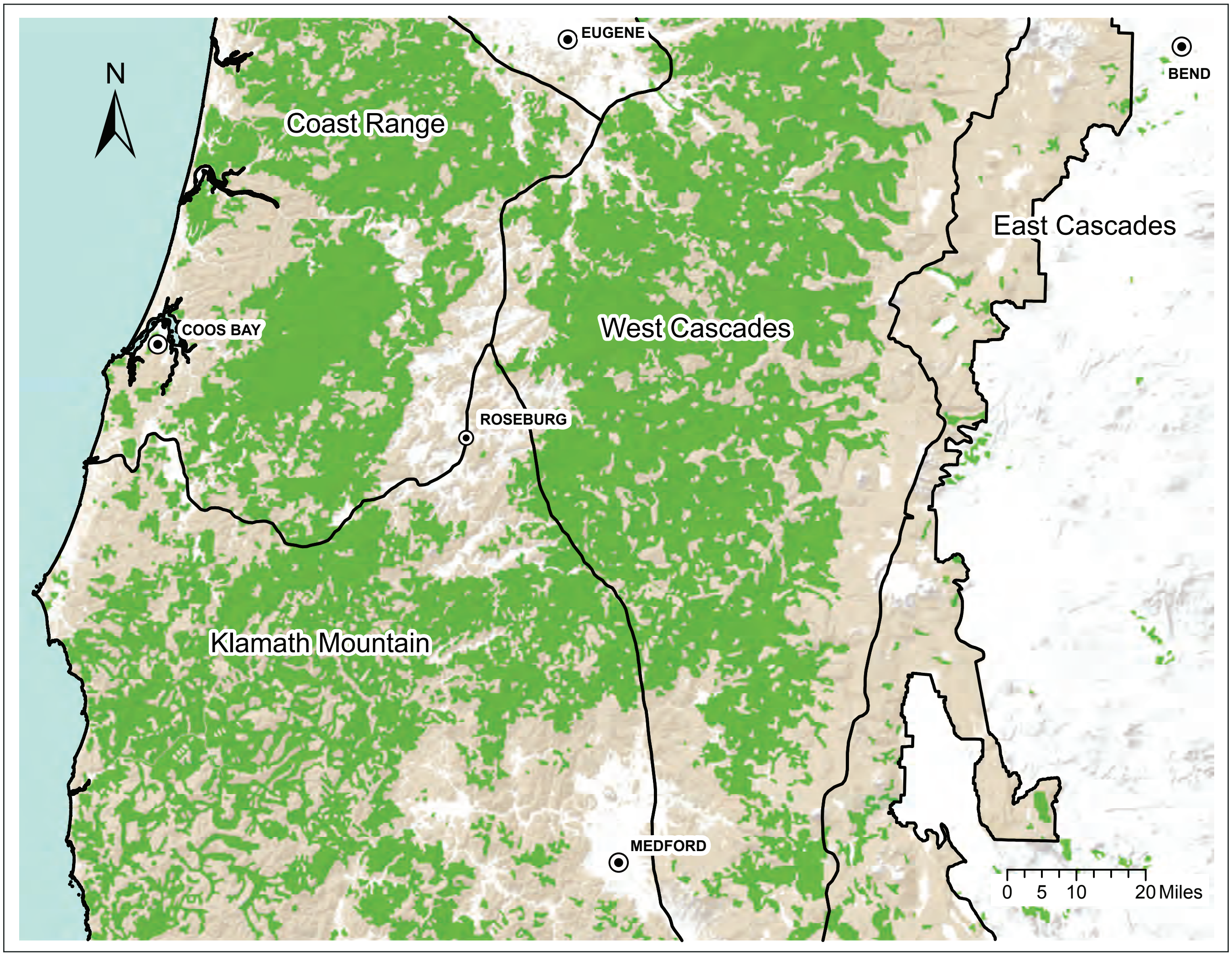


Figure 4-4—Differing historical patterns of old-growth Douglas-fir and mixed-conifer forest (green shaded areas) in west-central Oregon (Andrews and Cowlin 1940) within four areas with different fire regimes (Coast Range, infrequent—high severity; West Cascades, moderately frequent—mixed severity; Klamath Mountains, frequent—mixed severity; East Cascades, very frequent—low severity).

evolve (Ackers et al. 2015). For example, recent maps of suitable forest types (e.g., Glenn et al. 2017; USFWS 2011b, 2012a) differed from the original monitoring maps in that they factored in the spatial arrangement of discrete forest cover types (e.g., nesting, roosting, foraging) as well as abiotic factors (e.g., slope, topographic position, etc.) to produce maps describing a more comprehensive view of suitable forest (i.e., potential habitat). However, even the most recent efforts are not complete models of spotted owl habitat because they lack the impact of prey and barred owls on restricting distribution by limiting access to otherwise suitable forest for spotted owls. An important need is a better understanding and mapping of the differences between the potential and realized habitat for spotted owls. This is discussed in the "Research Needs, Uncertainties, Information Gaps, and Limitations" section below.

Patterns of change—

***Federal vs. nonfederal lands*—**Davis et al. (2016) estimated that there were about 12.6 million ac (5.1 million ha) of suitable nesting and roosting cover type distributed across the spotted owl's geographic range at the time of NWFP development (1993), the majority (73 percent) of which occurred on federal lands. By 2012, suitable nesting/roosting forest cover decreased to 12.1 million ac (4.9 million ha) (74 percent occurring on federal lands), resulting in an overall net change of -3.4 percent. Net decreases were -1.5 percent on federal lands (primarily caused by wildfire) and -8.3 percent on nonfederal lands (primarily caused by timber harvest). During those two decades, forest cover suitable for dispersal decreased from 26.2 to 25.7 million ac (10.6 to 10.4 million ha) (-2.3 percent net change) on all lands. On federal lands, forest cover suitable for dispersal increased by 2.2 percent, and it decreased by 8.6 percent on nonfederal lands (Davis et al. 2016). Gains occurred because of forest succession, whereas losses were primarily a result of wildfire, disease, and timber harvest (Kennedy et al. 2012).

Timber harvest accounted for the majority (63 percent) of the losses across all lands. The vast majority of losses on nonfederal lands was caused by timber harvest (94 percent), whereas timber harvests accounted for 18 percent of total losses on federal lands (Davis et al. 2016). In Washington alone from 1996 to 2004, most (85 percent) of the timber harvest that resulted in lost forest cover suitable for nesting and roosting of spotted owls occurred on private lands (Kennedy et al. 2012, Pierce et al. 2005). Following timber harvest, wildfire was the next largest cause of loss (31 percent of total losses), which was 73 percent of the losses on federal land and only 3 percent of the losses on nonfederal land.

***Moist vs. dry forests*—**Primary causes of loss differed by ecoregion and forest type. The loss of nesting and roosting forest cover from wildfire occurred primarily in drier, fire-prone portions of the spotted owl's geographic range (i.e., northern California, southern Oregon, and eastern Cascade Range). Losses owing to insects and disease (and other natural disturbances) was the next most significant disturbance and mainly occurred in the eastern Cascades of Washington and Oregon (Davis et al. 2016, Kennedy et al. 2012).

Recruitment of forest cover suitable for nesting and roosting by spotted owls was estimated at 257,591 ac (104 288 ha) from 1993 to 2012 (Davis et al. 2016). Most of the gain occurred on nonfederal lands within the redwood (*Sequoia sempervirens*) zone of coastal California (fig. 4-5). On federal lands, the largest net gain (40,385 ac [16 350 ha]) occurred in the eastern Cascades of Oregon, where fire suppression allowed forest succession of Douglas-fir (*Pseudotsuga menziesii*) and grand fir (*Abies grandis*) to develop in areas that historically had frequent low-severity fires and were formerly dominated by open ponderosa pine-dominated forests (*Pinus ponderosa*) (Davis et al. 2016).

Effects of forest change—

Loss of suitable forest cover for nesting and roosting, especially on nonfederal lands, has been an important contributor to declining populations of spotted owls (Dugger et al. 2016). Those spotted owls that had territories with more forest cover associated with nesting and roosting conditions typically had better survival, fecundity, occupancy dynamics, recruitment, and rate of population change (Dugger 2016; Dugger et al. 2005, 2011; Forsman et al. 2011; Seamans and Gutiérrez 2007). For example, Dugger et al. (2005) found that owl territories with the greatest fitness potential were characterized by >50 percent old-forest habitat within a 412-ac (167-ha) circle centered on used nest locations. Relationships among population parameters of spotted owls and older forests vary over different spatial scales (e.g., individual territory vs. study area), and can be independent of, or interact with, the presence of barred owls. Concentrated areas of older forest suitable for nesting and roosting, or increased amounts of heterogeneity (i.e., mixture of conditions used for foraging), have positive effects on the vital rates of spotted owls (Dugger et al. 2016, Forsman et al. 2011, Franklin et al. 2000, Olson et al. 2004).

In some landscapes, fragmentation of older forest can have negative or positive effects on spotted owl occupancy depending on the scale of fragmentation and edge characteristics. Schilling et al. (2013) found that spotted owls had decreased survival and increased home-range size with increased forest fragmentation in southwestern Oregon. In Washington, territory-level extinction rates decreased with increased amount of late-seral edge, and colonization decreased with more late-seral patches within a territory (Sovern et al. 2014). It is also important to consider spatial scale, and level of contrast between edge, when assessing the influence of forest edges on foraging and space use by spotted owls. Comfort et al. (2016) found that spotted owls radio-marked in southern Oregon were negatively associated with hard edges (high contrast in forest structure and height) at a fine scale (telemetry location), but showed a lack

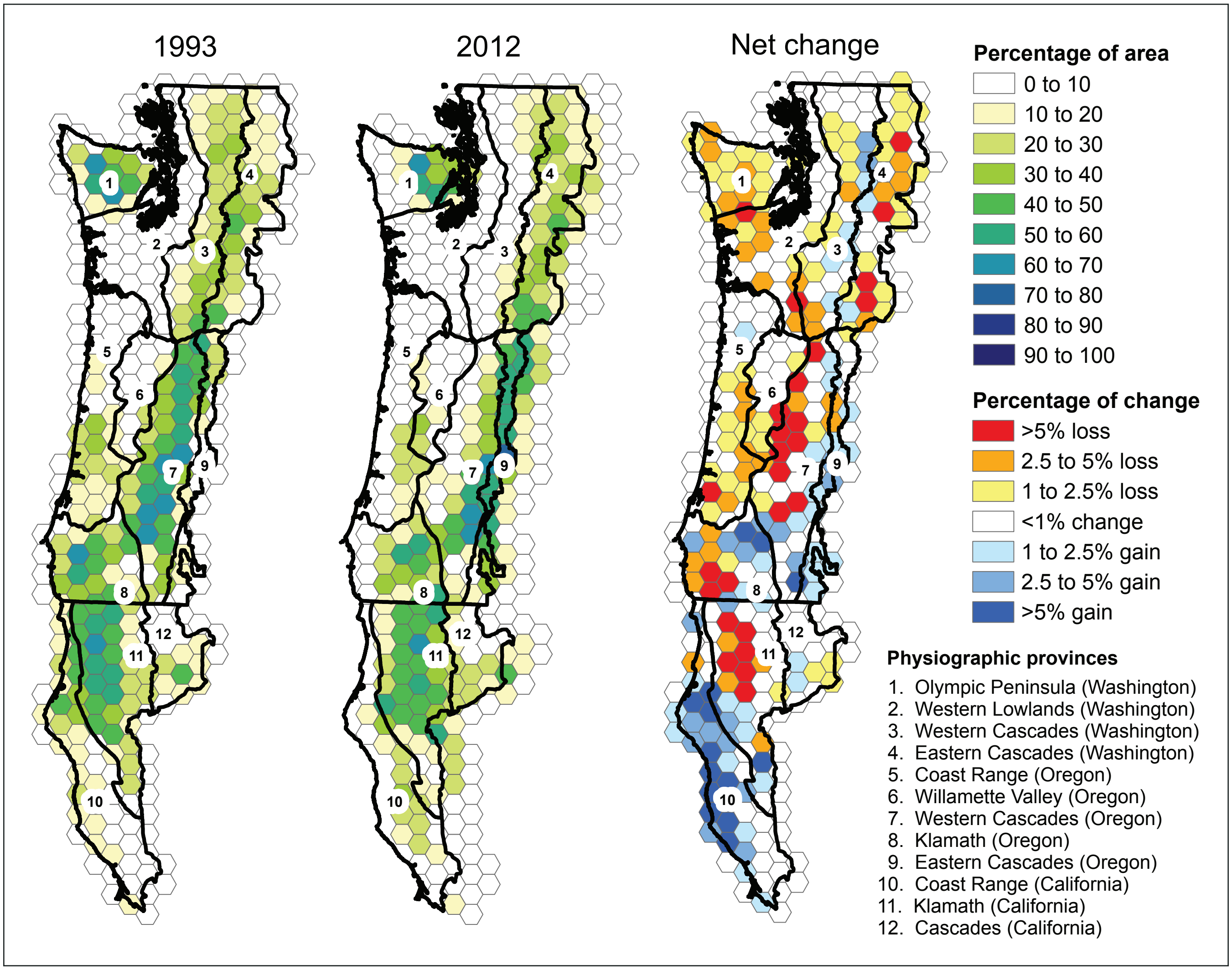


Figure 4-5—Loss and recruitment of forest types suitable for nesting and roosting by spotted owls in 1993 versus 2012, based on change in percentage of cover estimated within 18.6 mi (30 km) (center to center) hexagons across the Northwest Forest Plan area.

of negative response to hard edges at broader scales (territory or home-range scales). At least at the territory scale, heterogeneity can contribute to accessibility to different forest types. Regardless of spatial scale, spotted owls were positively associated with softer, more diffuse edge types created by disturbances such as low- and mixed-severity fire (Comfort et al. 2016). Collectively, these and other studies suggest that spotted owls select for abundant, structurally diverse closed-canopy forest with diffuse late-seral forest edge at the territory scale, and relatively lower fragmentation in nesting areas (Franklin et al. 2000, Olson et al. 2004, Sovern et al. 2014).

Forest protection effectiveness—

The NWFP included a network of large late-successional reserves (LSRs) that were designed to conserve forest for species dependent on older forests (FEMAT 1993). The LSR network was intended to meet the resource needs of many species, but a substantial focus was placed on creating and maintaining forest cover features from a draft recovery plan for the spotted owl (USFWS 1992). LSRs contained enough suitable forest cover to support multiple pairs of spotted owls and were distributed to facilitate movement of spotted owls across their geographic range. Although many of the LSRs contained large areas of older forest, a significant portion of them were delineated in fragmented landscapes that contained stands of younger forest. Dispersal between LSRs is important for spotted owl conservation, and the NWFP was expected to facilitate that dispersal by designated riparian reserves, retention of green trees in timber harvest units in the matrix, protection of 100 ac (40 ha) areas at known owl sites (managed as LSRs within the matrix), and other administratively withdrawn areas (USDA and USDI 1994). However, these assumptions are largely untested, so it remains unknown if the NWFP is sufficient to facilitate adequate dispersal, which may be a limiting factor of spotted owl populations.

In addition to broad-scale LSRs, forest protections for spotted owls include circles of varying radii centered on used nest locations, within which various amounts of suitable nesting, roosting, and foraging forest cover types are protected. For example, the Fish and Wildlife Service developed guidelines for consultation under section 7 of the Endangered Species Act that included a 2.9-km-radius circle (6,424 ac [2600 ha]) around spotted owl nest locations for evaluating ‘‘incidental take’’ for projects affecting suitable habitats (USDA and USDI 1994). The rationale for this circle size was developed based on preliminary analysis of the median home-range size of radio-marked spotted owls. States also developed rules for state and private forestry practices to protect spotted owl nest sites. For example, the 2006 Washington State Forest Practices Board Rules called for protection of 40 percent cover of suitable nesting and roosting forest within a 6,422 ac (2600 ha) circle around nest sites (WAC 222-10-041). Forsman et al. (2015) suggested that level of protection would not be sufficient because spotted owl home ranges contained more suitable forest cover than would be protected under the Washington forest practices rules. Furthermore, new methods for delineating owl territories (e.g., Thiessen polygons) used by Dugger et al. (2016) provide better representations of the territory.

At the time LSRs were delineated, it was estimated that they contained on average 43 percent older forest (USDA and USDI 1994). The expectation was that all LSRs would eventually fill in and achieve the 60-percent-or-greater area threshold needed to support multiple breeding pairs and collectively would facilitate spotted owl population recovery. The success of meeting that threshold depends on the frequency, severity, and spatial extent of disturbance (e.g., wildfire, timber harvests), as well as the rate of forest succession, and interactions among these processes on forest recruitment (chapter 3). As of the most recent monitoring report (Davis et al. 2016), the rangewide estimate for suitable nesting/roosting forest cover in LSRs was an average of 42.4 percent in 1993. As of 2012, this average decreased to 42.0 percent. Larger LSRs (≥10,000 ac) averaged 45.0 percent, decreasing to 44.5 percent by 2012. These losses were due mainly to wildfire and exceeded the regional-scale expected rate of loss (2.5 percent per decade) (FEMAT 1993). Most of the losses of nesting and roosting forest cover have been in the more fire-prone portions of the spotted owl's range (Davis et al. 2011, 2016). For example, within LSRs and other reserves (e.g., administratively withdrawn, wilderness areas, etc.) in the Klamath Mountains physiographic province, losses were as high as 18.9 percent between 1993 and 2012 (fig. 4-5), and largely the result of the 2002 Biscuit Fire, which burned 494,000 ac (>200 000 ha).

Forest cover trends on federal lands during the next two to three decades are expected to benefit spotted owls because significant recruitment of suitable nesting/roosting forest cover is expected to offset many pre-NWFP losses (chapter 3) (Davis et al. 2016). However, this expectation is based on current rates of harvest and wildfire occurrence on federal lands, which may change depending on future forest plan revisions and the predicted increased spatial extent, frequency, and severity of wildfires due to climate change (chapter 2) (Jones et al. 2016, Westerling et al. 2006). In addition, competitive pressure from established barred owls (see below) has raised uncertainties about whether recruitment of suitable forest cover will be enough to conserve spotted owls over the long term. If spotted owls are to persist in LSRs under competitive pressure from barred owls, it will likely be only in localized areas that support few barred owls. However, it remains doubtful if there are any areas where spotted owls hold a competitive advantage over barred owls (Pearson and Livezey 2007, Singleton 2013, Wiens et al. 2014).

The potential effects of climate change add to the uncertainty of how competitive dynamics with barred owls and availability of suitable habitat will affect spotted owls in the future. Carroll et al. (2010) used a climatic niche modeling approach to evaluate the regional system of LSRs for resiliency to climate change for providing necessary resources of species associated with old forest. They developed distribution models integrating climate data with vegetation variables for a large suite of species, including the spotted owl. The LSRs functioned better than expected by chance for capturing all of the species, but community composition and interspecific interactions were also important to consider in evaluating effectiveness of the reserves. A network of fixed reserves with a high level of climatic and topographic heterogeneity (i.e., designed for resilience) has an increased likelihood of retaining the biological diversity of old-forest ecosystems under climate change. Under this scenario, even those species with limited dispersal capability are able to colonize future habitat. Carroll et al. (2010) projected a northward and higher elevation movement of suitable forest for spotted owls; therefore, the current fixed system of LSRs may not have enough climatic and topographic heterogeneity to be adequate for spotted owls into the future. Other reserves designated before the NWFP, such as parks and wilderness areas, may become increasingly important for the subspecies' persistence. LSRs successfully protected areas with greater biological importance for spotted owls when the NWFP was developed, but in the face of climate change, it may be necessary to have another evaluation and planning phase that results in a reserve system designed for more robust resilience (Carroll et al. 2010) (see chapter 3 for more discussion of alternative reserve designs), especially in the dry forest zone where management for ecosystem and spotted owls may not be compatible at stand and small landscape scales (chapter 12). Even with relatively little modification in response to climate change, suitable forest conditions on the east side and southern portions of the range are at risk of losses. Dense, multilayered forests in the dry forest zone are vulnerable to a host of mortality forces, especially wildfire (see chapters 3 and 12).

Barred Owls

Barred owl range expansion and population trends— Competition with established populations of barred owls has emerged as a much more prominent and complex threat to the long-term persistence of the spotted owl than was anticipated during the development of the NWFP. Once confined to forests of eastern North America, the barred owl is a medium-size, ecologically similar species whose newly extended geographic range now completely overlaps that of the northern spotted owl (Gutiérrez et al. 2007, Livezey 2009). Newly colonizing barred owls in the Pacific Northwest have been classified as native invaders—species that, under the influence of events such as climate change or human modifications to the landscape, have become invasive by expanding their populations into new areas (Carey et al. 2012, Valéry et al. 2009, Wiens et al. 2014). The range expansion of barred owls in western North America is well documented (Dark et al. 1998, Dunbar et al. 1991, Kelly et al. 2003, Livezey 2009, Taylor and Forsman 1976). Initial colonization of different regions by barred owls was variable, but barred owls now appear to co-occupy and outnumber spotted owls throughout the entire range of the threatened subspecies (Dugger et al. 2016, Pearson and Livezey 2003, Singleton et al. 2010, Wiens et al. 2011, Yackulic et al. 2012). Barred owls have also invaded the range of the California spotted owl in the Sierra Nevada (Seamans et al. 2004). The cause of this range expansion is unknown, but landscape changes facilitated by European settlement or historical changes in climate are factors that may have enabled barred owls to expand their range from eastern to western North America (Livezey 2009, Monahan and Hijmans 2007).

With few exceptions, barred owls have not been systematically surveyed in the Pacific Northwest, and the majority of information on their distribution and population trends is limited to incidental observations during surveys of spotted owls (Dugger et al. 1991, 2016; Gutiérrez et al. 2007; Wiens et al. 2011). Despite this shortcoming, incidental field data show a rapid increase in barred owls as they expanded their populations westward and southward into the range of the spotted owl (fig. 4-6) (Dugger et al. 2016). Studies focused on barred owls found much higher densities than estimates based on incidental field observations

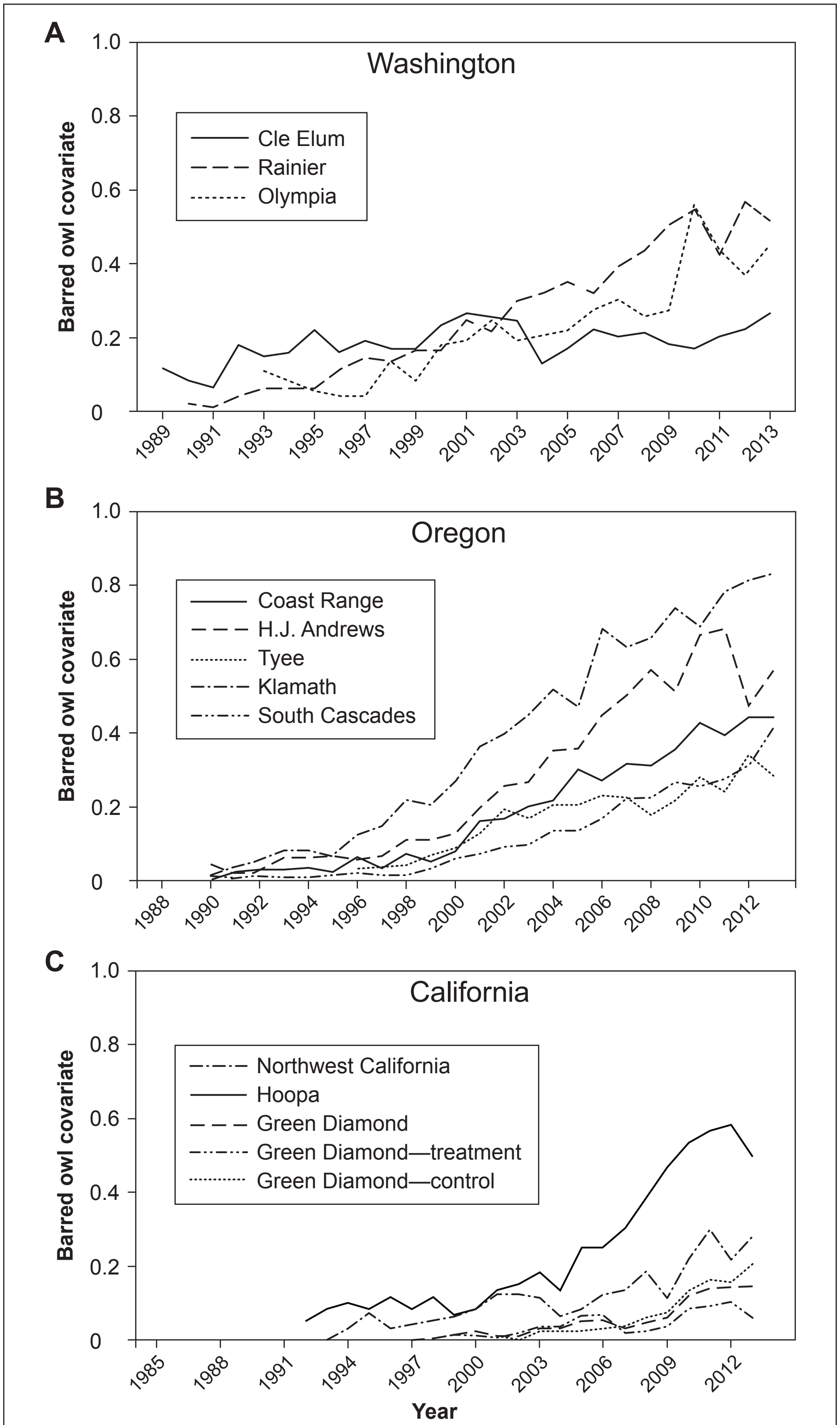


Figure 4-6—Annual increase in the proportion of spotted owl territories with detections of barred owls at (A) three sites in Washington, (B) five sites in Oregon, and (C) three sites in California from 1985 to 2013 (Dugger et al. 2016). Estimates for Green Diamond are presented separately for control and treatment areas and after (2009 to 2013) barred owls were removed from the treatment area from 2009 to 2013.

(Hamer et al. 2007; Singleton et al. 2010; Wiens et al. 2011, 2014; Yackulic et al. 2012, 2014). For example, Wiens et al. (2011) conducted surveys of barred owls during 2009 in the Oregon Coast Range and identified approximately 11 territorial pairs of barred owls per 100 km^2 (39 mi^2; 3 to 8 times higher density than spotted owls) with 89 percent of the landscape occupied, which peaked on publicly owned lands with greater amounts of mature and old coniferous forest. More recent (2015–2016) surveys of barred owls indicate an even greater probability of landscape occupancy in the Oregon Coast Range (~0.94) (Wiens et al. 2017). The degree to which the colonizing population of barred owls has reached carrying capacity within the geographic range of the spotted owl is currently unknown, but studies are underway that can help address this uncertainty (e.g., Wiens et al. 2017). Barred owl populations may continue to increase depending on the capacity of available habitat and food resources, which varies regionally with forest composition and latitudinal changes in prey communities and climate.

Barred owl effects on spotted owls—
Compared to spotted owls, barred owls are slightly larger (Gutiérrez et al. 2007), have more diverse diets (Hamer et al. 2001, Wiens et al. 2014), and use a broader range of forest conditions for nesting (Herter and Hicks 2000, Livezey 2007, Pearson and Livezey 2003) and foraging (Hamer et al. 2007, Singleton 2015, Singleton et al. 2010, Weisel 2015, Wiens et al. 2014). Barred owls also have higher annual survival (fig. 4-7), higher reproductive output, and, in most areas, use much smaller home ranges than spotted owls (Hamer et al. 2007, Singleton et al. 2010, Wiens et al. 2014). The exception is in northern California,

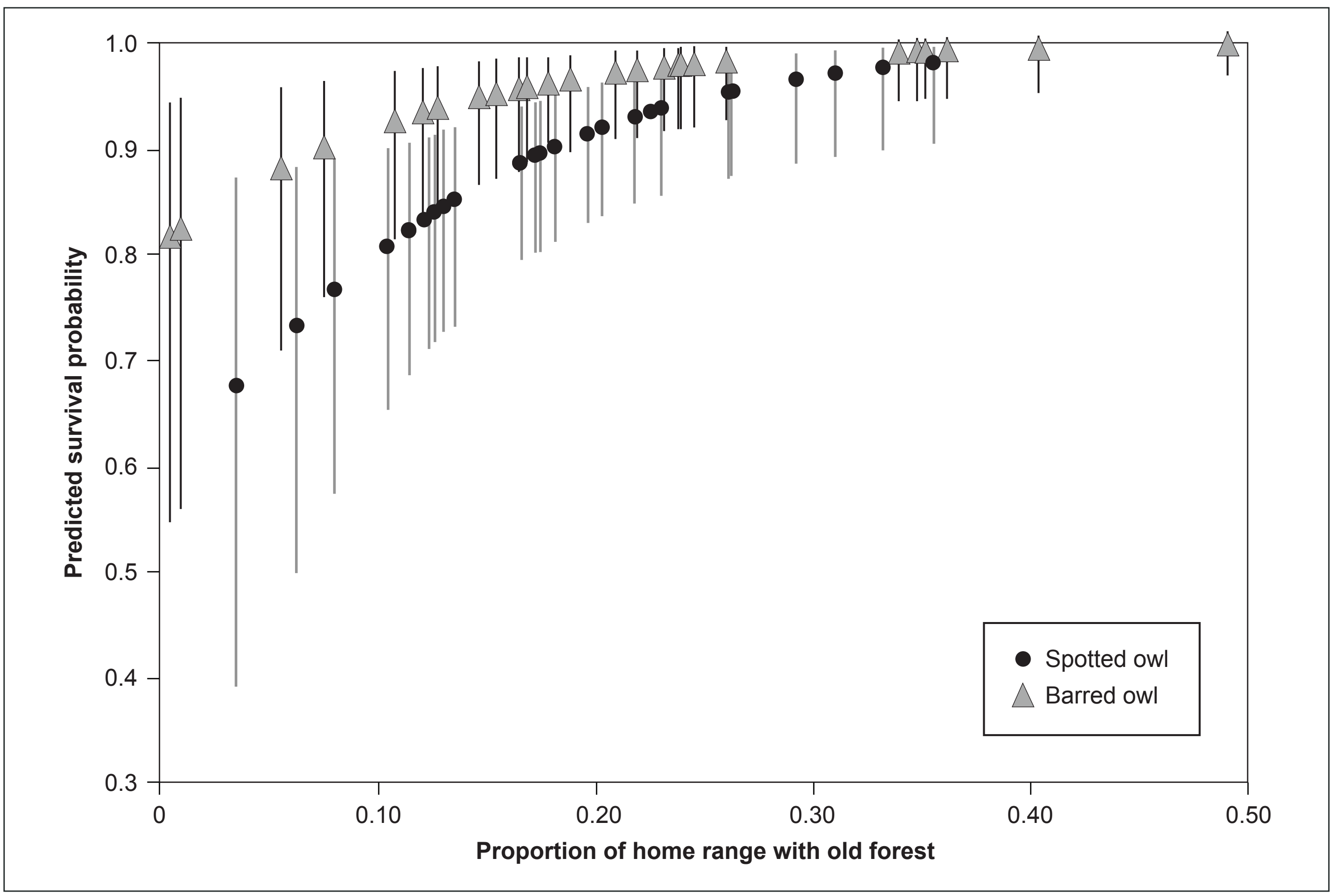


Figure 4-7—Survival (with 95-percent confidence limits) of individual adult barred owls and spotted owls increased with increasing amount of old forest (≥120 years) conifer forest within their home ranges in the Oregon Coast Range (Wiens et al. 2014).

where the two species used relatively small home ranges of similar size (Weisel 2015). Barred owls also defend their territories more aggressively than spotted owls (Van Lanen et al. 2011), which can result in increased mortality of spotted owls from agonistic interactions and direct killing of spotted owls by barred owls (Leskiwand Gutiérrez 1998, Wiens et al. 2014).

The dramatic increase in populations of barred owls since implementation of the NWFP has significant implications for management of forests inhabited by spotted owls. Several lines of evidence indicate that increases in the abundance of barred owls has had a strong and negative impact on spotted owls. Increasing abundance of barred owls has been documented to have the following effects on spotted owl populations:

1. Occupancy of historical spotted owl territories is lower (fig. 4-8) (Bailey et al. 2009, Dugger 2016, Dugger et al. 2011, Kelly et al. 2003, Kroll et al. 2010, Olson et al. 2005, Sovern et al. 2014; Yackulic et al. 2014).
2. Apparent survival is lower (Anthony et al. 2006, Diller et al. 2016, Dugger et al. 2016, Forsman et al. 2011, Glenn et al. 2011a).
3. Reproduction is lower (Dugger et al. 2016, Forsman et al. 2011, Olson et al. 2004).
4. Population size declines more rapidly (Anthony et al. 2006, Dugger et al. 2016, Forsman et al. 2011).
5. Hybridization between the species is increased (Barrowclough et al. 2005, Dark et al. 1998, Gutiérrez et al. 2007, Haig et al. 2004, Hamer et al. 1994, Kelly and Forsman 2004).
6. Detection rates during surveys are lower (Bailey et al. 2009, Crozier et al. 2006, Dugger et al. 2011, Dugger et al. 2016, Kroll et al. 2010, Olson et al. 2005, Sovern et al. 2014, Yackulic et al. 2014).

Moreover, studies of competitive interactions and resource partitioning showed that barred owls can directly alter the movements, resource use, and reproduction of spotted owls (Wiens et al. 2014). Barred owls also display demographic superiority over spotted owls; annual rate of

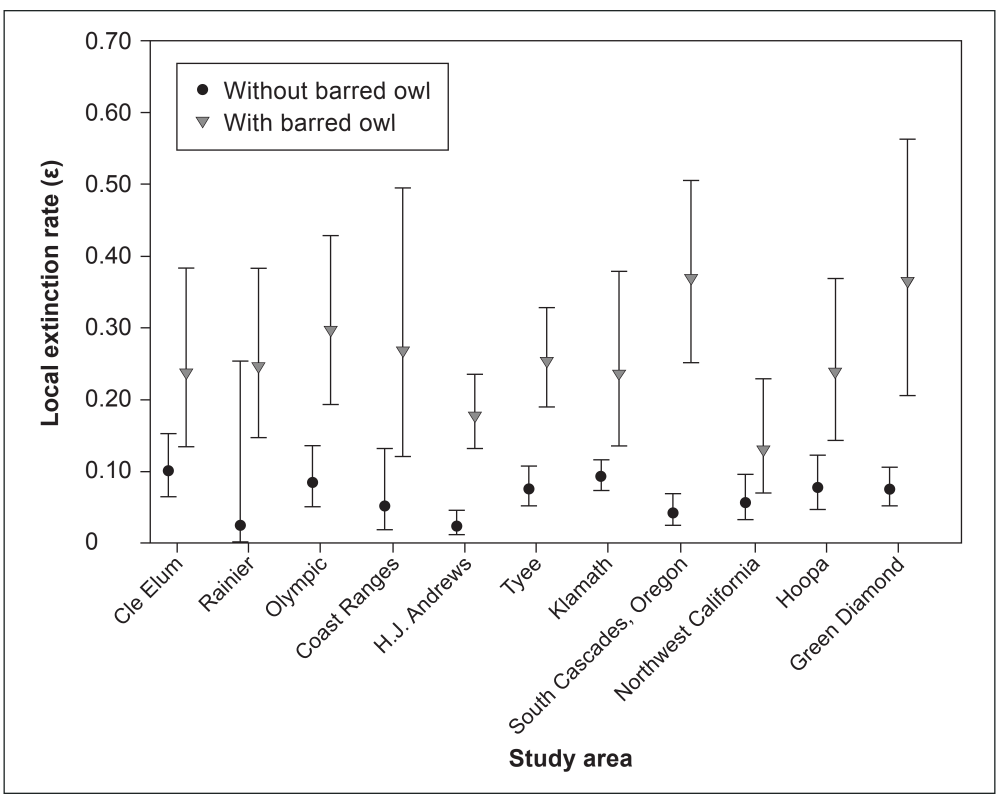


Figure 4-8—Mean annual local extinction rates (with 95 percent confidence limits) for northern spotted owls on 11 study areas in Washington, Oregon, and California relative to with (gray triangles) and without (black circles) the presence of barred owls (Dugger et al. 2016).

survival was greater for barred owls (0.92 ± 0.04) than for spotted owls (0.81 ± 0.05), and mean reproductive output of barred owl pairs was 4.4 times greater than that observed for spotted owls over 3 years in western Oregon (Wiens et al. 2014). More recently, studies in California have demonstrated a positive association between removal of barred owls and population trends of spotted owls (fig. 4-2) (Diller et al. 2016, Dugger et al. 2016). Collectively, these studies provide strong evidence that interspecific competition with an increasing number of barred owls, combined with continued loss of potentially suitable forest cover, is contributing to population declines of spotted owls despite widespread conservation of old forest under the NWFP.

Barred owl densities are now thought to be high enough across the range of the spotted owl that, despite the continued management and conservation of suitable forest cover on federal lands, the long-term persistence of spotted owls is in question without additional management intervention (Buchanan et al. 2007, Diller et al. 2016, Dugger et al. 2016, USFWS 2013). In a few cases, populations of spotted owls have responded positively to the removal of barred owls during pilot removal experiments; supporting the hypothesis that along with forest conservation and management, removal of barred owls might slow or reverse local declines in spotted owl populations in some areas (Diller et al. 2016, Dugger et al. 2016). However, the effectiveness and moreover the feasibility of large-scale barred owl removal for conservation of spotted owls remain to be demonstrated, and barred owl removal activities would likely need to be continued for the foreseeable future to maintain low barred owl densities in control areas.

Barred owl habitat and prey—

Barred owls occupy a broader range of forest types and consume a wider variety of prey than northern spotted owls (Livezey 2007), and use a variety of different forest types in the Pacific Northwest, including fragmented mixed-deciduous forest in rural and urban landscapes (Rullman and Marzluff 2014). Hamer et al. (2007) reported that, in the northern Cascade Range of Washington, barred owls tended to use old forest more than expected, but used most cover types in proportion to availability. Compared to spotted owls, barred owls occupied areas at lower elevations (Hamer et al. 2007). In the eastern Cascades of Washington, Singleton et al. (2010) reported that barred owls typically established their home ranges in areas that had canopy cover more than 72 percent, medium to large trees (tree crown diameter >21 ft [>6.5 m]), low topographic position (<25 percent), and gentle slopes (<11 degrees). Within those home ranges, barred owls used structurally diverse mixed grand fir forest more intensively than open ponderosa pine or Douglas-fir (Singleton 2015). In the Oregon Coast Range, foraging barred owls most often used patches of old (>120 years) conifer forest in addition to riparian-hardwood forests in relatively flat areas (Wiens et al. 2014). In the redwood region of coastal California, barred owls most often used sites with greater understory vegetation height and more hardwood trees, perhaps in response to greater densities of woodrats (*Neotoma* spp.) in these conditions (Weisel 2015). Collectively, these studies showed that barred owls, in areas where they were sympatric with spotted owls, were most commonly associated with relatively gentle slopes in structurally diverse, mature and old-conifer forests or lowland riparian areas containing large hardwood trees. Use of older forest in combination with moist, valley-bottom forest was also consistent with forest associations described for barred owl nesting areas (Buchanan et al. 2004, Herter and Hicks 2000, Pearson and Livezey 2003). Barred owls use the full range of forest types used by spotted owls, and a broader range of forest cover types outside of areas historically occupied by spotted owls. However, systematic studies have yet to quantify the full range of forest conditions that support barred owls in the Pacific Northwest. There are currently no known forest management actions that would benefit spotted owls more than barred owls.

Dietary studies are lacking for barred owls in California, but their diets in Washington and Oregon included a broad variety of small- to medium-size mammals, birds, frogs, salamanders, lizards, snakes, crayfish, snails, fish, and insects (Graham 2012, Hamer et al. 2001, Wiens et al. 2014). Mammalian prey of barred owls primarily included northern flying squirrels (*Glaucomys sabrinus*), woodrats, brush rabbits (*Sylvilagus bachmani*), snowshoe hares (*Lepus americanus*), moles (*Scapanus* spp.), Douglas squirrels (*Tamiasciurus douglasii*), red tree voles (*Arborimus longicaudus*), red-backed voles (*Myodes californicus*), shrews (*Sorex* spp.), and deer mice (*Peromyscus maniculatus*) (Hamer et al. 2001, Wiens et al. 2014). Although there is substantial geographic variation in diets of barred owls corresponding with differences in prey distributions, northern flying squirrels appear to be a primary contributor to diets in Oregon and Washington (Graham 2012, Hamer et al. 2001, Wiens et al. 2014).

Although there is some evidence that barred owls were more strongly associated with riparian areas than spotted owls, studies clearly indicate a high degree of ecological overlap between the two species, especially in their use of old-growth forests and associated prey species (Hamer et al. 2001, 2007; Singleton et al. 2010; Weisel 2015; Wiens et al. 2014). In the eastern Cascades of Washington, spotted owls used drier midslope areas less likely to be occupied by barred owls, possibly as a mechanism to minimize interactions with barred owls, at least in the near term (Singleton 2013). This pattern reflects displacement of spotted owls by barred owls from highly suitable forest into conditions less favorable to long-term reproduction and survival of spotted owls, a finding consistent with long-term demographic studies of spotted owls throughout the range of the subspecies (Dugger et al. 2016, Forsman et al. 2011, Singleton 2013, Wiens et al. 2014).

In addition to impacts on spotted owls, changes in the abundance and distribution of an apex predator like the barred owl can have cascading effects on prey populations and food web dynamics (Holm et al. 2016, Wiens et al. 2014), as well as populations of other small sympatric owls (Acker 2012, Elliot 2006). Differences in space use, abundance, demography, suitable forest, diets, and behavior collectively suggest that the barred owl is not a direct functional replacement of the spotted owl in old-growth forest ecosystems (Holm et al. 2016, Wiens et al. 2014). As a consequence, additional changes in community structure and ecosystem processes are anticipated as a result of barred owl encroachment into areas managed under the NWFP.

Spotted Owl Prey

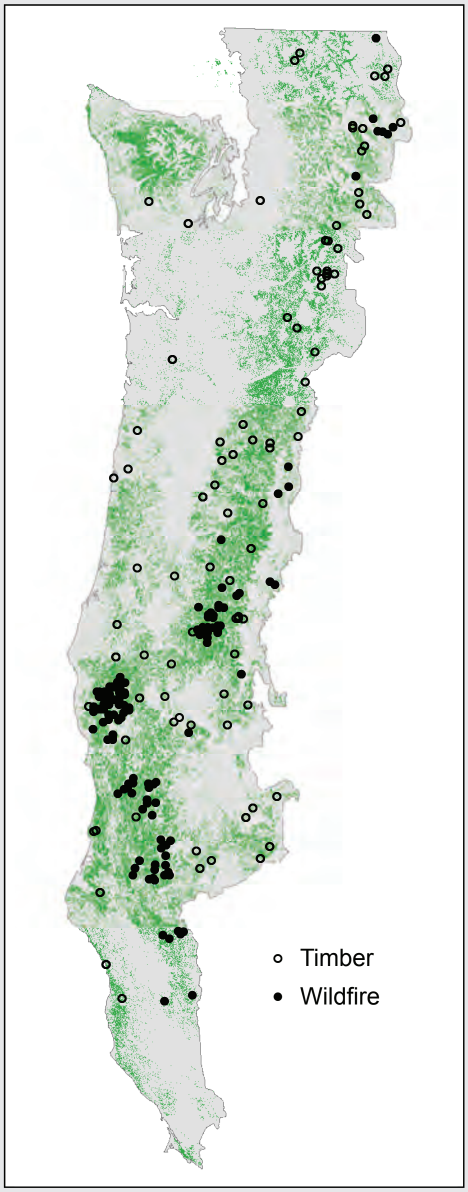
Like all predators, spotted owls are dependent on abundant and vulnerable prey. Much is known about the ecology and population demography of spotted owls, but little information exists on how fluctuations in populations of prey species influence behavior, space use, reproduction, or population growth of spotted owls. Spotted owls in some areas during some periods have had a strong 2-year cycle of high reproduction one year followed by a year of low reproduction (Anthony et al. 2006). One hypothesis for the cycle in reproductive output is variation in prey abundance. However, simple prey relationship models do not explain the highly synchronous and temporally dynamic patterns of spotted owl reproductive performance (Rosenberg et al. 2003). Northern flying squirrels, woodrats, red-backed voles, and red tree voles are the primary prey of spotted owls throughout different regions of the spotted owl's geographic range (Barrows 1980; Bevis et al. 1997; Forsman et al. 1984, 2004, 2005; Hamer et al. 2001; Rosenberg et al. 2003; Wiens et al. 2014; Zabel et al. 1995). None of these studies had data that could be used to examine relationships between annual variation in prey abundance and annual variation in survival or fecundity of spotted owls. Although deer mice are not a primary prey species (<2 percent biomass consumed), one study (Rosenberg et al. 2003) found a positive correlation ($r^2 = 0.68$) between abundance of deer mice and reproductive success of spotted owls.

Abundance and distribution of primary prey species can influence space use by spotted owls. For example, spotted owls more frequently use riparian areas within their home ranges (Wiens et al. 2014), perhaps because the cool microclimates associated with stream drainages may be favorable for thermoregulatory purposes during summer months (Barrows 1981), or more importantly, riparian areas are likely to support a rich diversity of prey (primarily small mammals) used by spotted owls (Anthony et al. 2003, Carey et al. 1999, Forsman et al. 2004). Home ranges of spotted owls tend to be smaller in the southern portion of the subspecies range, where woodrats are the primary prey, as compared to the northern portion of the geographic range, where woodrats are uncommon and northern flying squirrels are the primary prey (Forsman et al. 2005, Zabel et al. 1995). In northern California, southwestern Oregon, and the eastern Cascades, woodrats occur in fairly open forests and at much greater densities compared to northern flying squirrels (Carey et al. 1992; Lehmkuhl et al. 2006a, 2006b; Wilson and Forsman 2013; Zabel et al. 1995). Differences in space use by spotted owls in different portions of their range also relate to regional differences in the availability of prey species. Northern flying squirrels and red tree voles, for example, occur at highest densities in the complex structure of mature Douglas-fir stands with old-growth characteristics, whereas woodrats have greater densities in young stands, along edges, or in brushy areas (Carey et al. 1992, Price et al. 2015, Sakai and Noon 1993, Swingle and Forsman 2009, Walters and Zabel 1995, Zabel et al. 1995). Spotted owls used forest edges to a greater degree when forage consisted primarily of woodrats (Diller et al. 2012), but preferred forest interiors, where they foraged on red tree voles and northern flying squirrels. Timber harvest activities, including thinning of dense plantations, reduce the abundance of northern flying squirrels and red tree voles for several decades, contributing to a reduction in use by spotted owls (Carey 2000, Dunk and Hawley 2009, Gomez and Anthony 1998, Manning et al. 2012, Price et al. 2015, Waters and Zabel 1995, Wilson and Forsman 2013).

Disturbance

In this section, we define disturbances as modifiers of the structural characteristics, species composition, and landscape patterns of forest cover types used by spotted owls. The range of the northern spotted owl encompasses a variety of historical disturbance regimes that are fundamental to the health and diversity of these ecosystems (chapter 3). Important forest disturbances result from wildfire, forest management (e.g., thinning), timber harvests, extreme weather events, or forest insect and disease processes (Davis et al. 2016). Effects that forest disturbances have on spotted owls depends on spatial scale, severity, and season (McKelvey 2015). Biogeographic variation across the large range of spotted owls also results in very different levels of disturbance type, frequency, and severity (see "Wildfire" below). Major disturbance events influence forest cover types that have been used by spotted owls for many decades, and have different effects depending on the magnitude of change and the time since disturbance. For example, in the short term, a disturbance that creates open canopy conditions could reduce value for spotted owl roosting, but have long-term benefits by enhancing understory vegetation diversity and conditions for spotted owl prey. Further, disturbances can stimulate the development of large-tree, complex-structure stand conditions over time (Lehmkuhl et al. 2015). An important secondary effect of forest disturbances for spotted owls are changes in prey abundance or vulnerability. These effects can be positive by creating conditions that increase abundance or vulnerability for some prey species, or negative by removing critical forest structure required by primary prey populations (e.g., northern flying squirrel, red tree vole) (Manning et al. 2012, Wilson and Forsman 2013). Some disturbances have a neutral affect, particularly when limited in severity or spatial extent, and ample suitable forest remains available at core and home-range scales.

Spotted owls were listed as a threatened species under the Endangered Species Act largely because of concerns regarding loss of old forest resulting from commercial timber harvest (Thomas et al. 2006, USFWS 2011b). Subsequent to reductions in harvest of old forest, high-severity wildfire has become the leading cause of suitable forest loss for spotted owls on federal lands, especially in fire-prone landscapes. However, commercial timber harvest still contributes substantially to the loss of suitable forest cover in some areas, especially on nonfederal lands (Davis et al. 2016, Pierce et al. 2005). Recent research on disturbance effects on spotted owls indicates that disturbances such as mixed-severity fires that generate heterogeneity at landscape and stand scales are not necessarily adverse, provided that adequate nesting and roosting structural conditions remain after the disturbance (Clark et al. 2013, Comfort et al. 2016). High-severity disturbances that broadly alter stands and landscapes within nesting territories can remove critical components of forest structure (e.g., high canopy cover and density of large live trees) required for spotted owl survival and reproduction (Dugger et al. 2005, Franklin et al. 2000, Olson et al. 2004). Timber harvesting and wildfire can both reduce the living tree components of a stand and reduce the overall suitability for spotted owls (see sidebars on pages 265 and 266). An important difference between timber harvest and wildfire is the removal of trees and ground disturbance in a timber harvest. For most wildfires, there is limited physical soil disturbance (although fire



Effects of Forest Disturbances on Nesting/Roosting Forest Cover

Map data from the most recent northern spotted owl habitat monitoring report (Davis et al. 2016) and Forest Inventory Analysis and Current Vegetation Survey plots were used to assess changes resulting from forest disturbances on stand structure elements used in the Davis et al. (2011, 2016) nesting/roosting cover type modeling and mapping procedure.

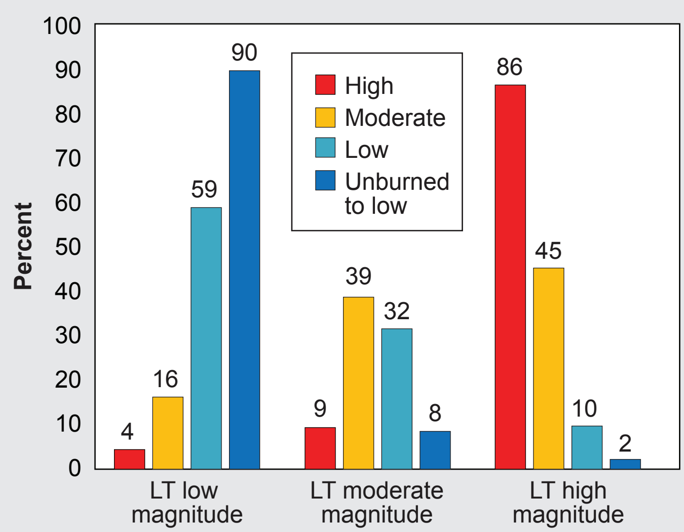
Plots used in this analysis occurred in mapped suitable nesting/roosting cover type in 1993 that experienced a disturbance between 1994 and 2012 from either timber harvesting or wildfire, which occurred between the initial plot measurement and remeasurement dates.

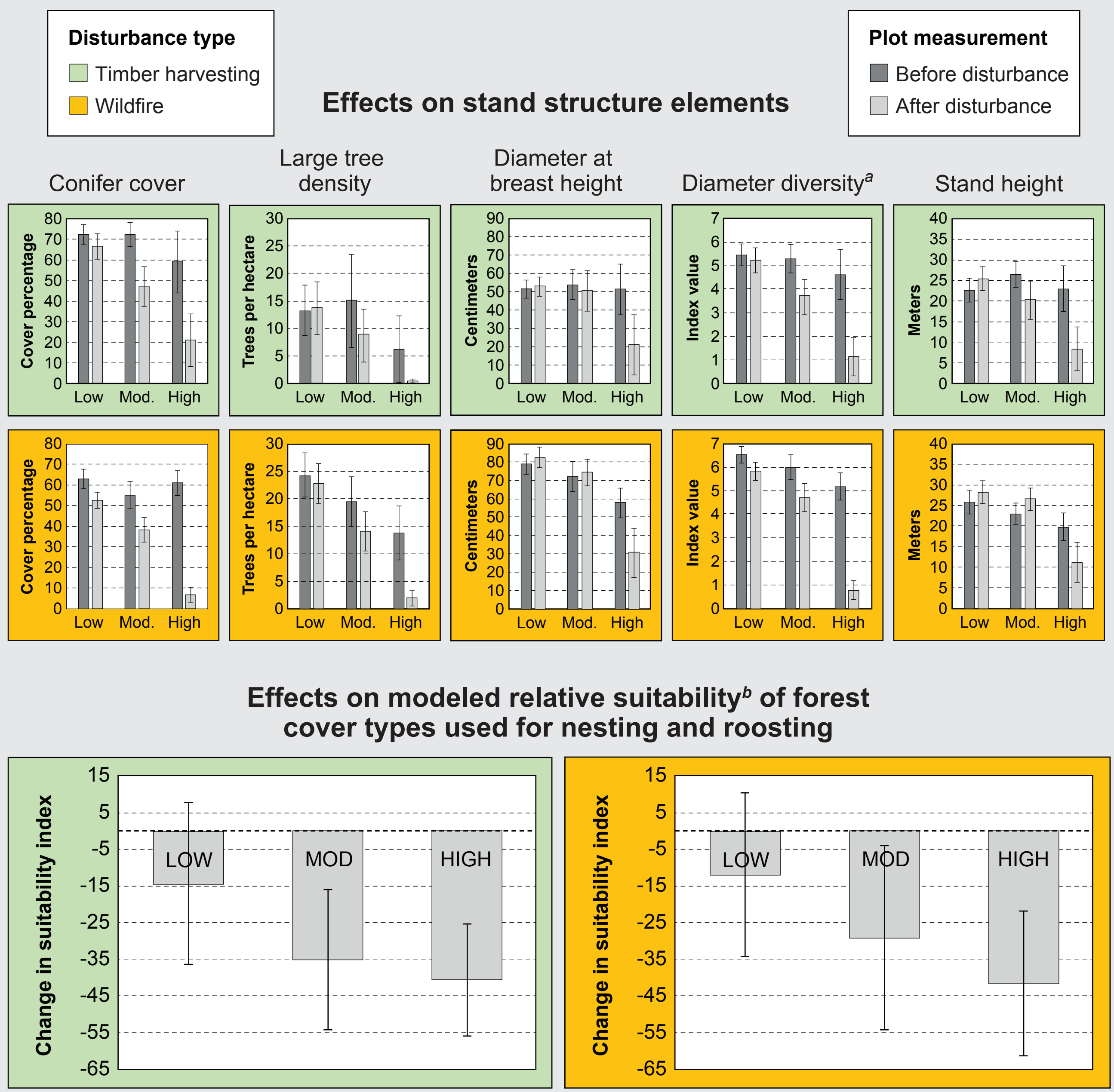
Changes in the mapped nesting/roosting relative suitability index were also analyzed by differencing the 2012 and 1993 relative suitability maps.

LandTrendr (LT) data (Kennedy et al. 2012) of forest disturbance magnitude are satellite-based measurements of loss of vegetation cover. We divided them into three classes:

- Low (<33 percent cover loss)
- Moderate (33 to 66 percent cover loss)
- High (>66 percent cover loss

The graph to the right is from Davis et al. (2015) and shows the relationship between these classes and monitoring trends in burn severity classes.





[a] The diameter diversity index is a measure of the stuctural diversity of a forest stand based on live tree densities in diameter at breast height classes. An index value of 1 represents a single-story stand. Higher index values represent more complex multistoried stand structure.

[b] Modeled relative suitability from Davis et al. (2016). The index for suitable nesting/roosting forest cover type ranges from 31 to 40, depending on the modeling region.

can have substantial impacts on soil chemistry and organic matter composition), and patches of live trees, snags, and logs remain in situ, which contributes to enhanced biodiversity, future quality of complex forest, and forest succession (Swanson et al. 2011).

Wildfire—
Wildfires occur throughout the entire range of the spotted owl. Some physiographic provinces are more environmentally suitable for wildfire occurrence at a decadal scale, while other provinces have wildfire-return intervals of several centuries (see chapter 3) (Agee 1993). Beyond frequency, the severity and spatial extent of wildfires differ across the NWFP area (Davis et al. 2011). The physiographic provinces of the eastern Cascades, southern portions of the western Cascades, and the Klamath Mountains are characterized by frequent low- and mixed-severity fire regimes (Baker 2015, Hessburg et al. 2007, Perry et al. 2011). Owing to more than a century of fire exclusion (e.g., from grazing, fire suppression, and historical forest management practices), many of these fire-prone landscapes have experienced significant increases in stand density and loss of large trees, threatening forest health and biodiversity (Hagmann et al. 2013, 2014; Hessburg et al. 2007; Perry et al. 2011). The historical extent of forest cover types suitable for nesting and roosting by spotted owls in dry and mesic mixed-conifer forests in the eastern Cascades and other fire-frequent forests was likely historically limited but has increased substantially in recent decades (Hagmann et al. 2013, 2014; Hessburg et al. 2007; Merschel et al. 2014). Moreover, in this fire-prone landscape, forest structure conditions that are more resilient to low- and mixed-severity fires (i.e., single-story old forests with large ponderosa pines) are not suitable for nesting and roosting by spotted owls. Areas occupied by spotted owls in the fire-prone landscapes of the eastern Cascade Range are often dense, closed-canopy, medium-size tree forests with a substantial true fir (*Abies* spp.) component and structural diversity enhanced by a variety of insect and disease processes, including dwarf mistletoe (Stine et al. 2014). These are the conditions that have been promoted through fire suppression and removal of fire-resilient large Douglas-fir and ponderosa pine trees. Compared to forest structure conditions that are more resilient to wildfire, areas occupied by spotted owls in these fire-prone landscapes are at higher risk to high-severity wildfire (Dennison et al. 2014, Stine et al. 2014). All forest types in these landscapes are vulnerable to substantial impacts from high-severity wildfire under extreme weather conditions, which are likely to be more common with climate change (Kennedy and Wimberly 2009, Reilly et al. 2017).

West of these fire-prone areas to the Pacific coastline, the forests become progressively moister and less prone to frequent large wildfire. In these moist forests, large wildfires tend to be infrequent to moderately frequent, and fire severity trends from mixed to high severity (see chapter 3). In less fire-prone landscapes, old and complex forest with large trees—compared to other forest types—has higher moisture retention and cooler microclimates compared to other forest types, and may enhance biodiversity under a changing climate (Frey et al. 2016). In these mixed- and low-frequency fire regime landscapes, old forest may be more resistant to wildfire than young forest with closed canopy under normal fire weather conditions (Thompson and Spies 2009).

Throughout the NWFP area, the fundamental association between spotted owls and multilayer forests with large trees and closed canopies is well established (Dugger et al. 2016, Forsman et al. 1984, Franklin et al. 2000, Olson et al. 2005, Wiens et al. 2014). The severity of the wildfire has a strong influence on the degree to which these forest cover types are altered by wildfire (see sidebar on page 265). Low-severity wildfire can have very little effect on the suitability of nesting and roosting cover types, and can even increase it. Moderate-severity wildfire can change stand structure and species composition, resulting in moderate decreases in cover-type suitability. High-severity wildfire can alter forest cover to the point at which the area is no longer be suitable for nesting, roosting, or dispersal. Multiple lines of research have confirmed the effects of wildfire on stand structure and composition, but much less is understood about the short- and long-term response of spotted owls to wildfire.

Most studies focused on wildfire effects evaluated the short-term response of spotted owls to wildfire, but in one of the few studies of the long-term effects of wildfire on spotted owls Rockweit et al. (2017) used 26 years of demographic data in a landscape with several wildfires and found that moderate and high burn severities negatively affected spotted owl apparent survival. They also found that burned territories functioned as ecological sinks where recruitment was high, but survival was lower than in nearby unburned territories. Several shorter post-wildfire studies have seemingly contradictory results regarding spotted owls and wildfire. For example, in an occupancy analysis, Jones et al. (2016) found high site extirpation rates of California spotted owls following a large, high-severity wildfire, but in a telemetry study, Bond et al. (2016) observed that burned forests were generally used in proportion to their availability. Other studies of California spotted owls and Mexican spotted owls have shown that wildfire does not necessarily decrease short-term occupancy in low- or moderate-severity burned areas (Bond et al. 2009, Ganey et al. 2011, Lee and Bond 2015, Lee et al. 2013, Roberts et al. 2011). Spotted owls can persist, at least for short periods, in landscapes that have experienced recent wildfires, as long as adequate moderate to closed-canopy nesting/roosting forest cover is retained at nesting core and home range scales. Even with high-severity wildfire, the effects can be insignificant or positive (e.g., increase vulnerability of prey) at larger spatial scales, especially if the forest cover changes caused by high-severity fire comprise only a small portion of a spotted owl's territory (Comfort et al. 2016).

Effects of wildfire interact in complex ways with other historic and current disturbances. Clark et al. (2013) found that local spotted owl site extinction probability was higher for sites with more combined area of past timber harvest, high-severity fire, and salvage logging. They also found evidence that colonization and occupancy rates were higher for sites with older forest burned at low severity (Clark et al. 2013). Coupling wildfire and salvage logging results in a high probability that a site becomes unoccupied after the first year postfire, especially if the core area burns at high severity and is subsequently logged (Bond 2016, Ganey et al. 2011, Lee et al. 2013). Beyond the effects on spotted owls, a human disturbance that directly follows a high-severity natural disturbance can have significant negative consequences to a forest ecosystem by disrupting abiotic and biotic processes, reducing or eliminating biological legacies, simplifying post-disturbance structural complexity, altering vegetation recovery, diminishing natural patterns of landscape heterogeneity, facilitating invasion of nonnative species, decreasing native biodiversity, increasing susceptibility to erosion and repeated high-severity disturbances, and eliminating restorative benefits of disturbance events (Lindenmayer-and Noss 2006, Thorn et al. 2017).

Overall, studies suggest that spotted owls are adapted to a forest landscape with a mosaic of successional stages shaped by historical disturbance regimes, accompanied by abundant prey resources, few barred owls, and structurally diverse closed-canopy forest with diffuse late-seral edge at the territory scale, and limited fragmentation occur within nesting areas (Dugger et al. 2011, Forsman et al. 1984, Franklin et al. 2000, Olson et al. 2004, Sovern et al. 2014). Research supports the premise that some spatial heterogeneity in forest conditions can have a positive effect on demography of spotted owls. At the territory scale (~500 to 1500 ha), a mosaic of older forest interspersed with other vegetation types, including early-seral and riparian forests, can promote high survival and reproduction of spotted owls (Comfort et al. 2016, Franklin and Gutiérrez 2002, Franklin et al. 2000). In terms of the effects of wildfire on spotted owls, we emphasize that most available research on impacts to spotted owls has been based to some degree on short-term responses and primarily focused on the other two spotted owl subspecies. The long-term (>5 years) effects of wildfire on spotted owl survival, reproduction, recruitment, and interactions with barred owls are not well documented.

Forest restoration and silvicultural treatments—
To meet management objectives of the NWFP, the spotted owl recovery plan, and critical habitat requirements, researchers and federal land managers have focused on ecosystem function (e.g., fire as an ecological process) in developing silvicultural practices that provide ecologically sustainable alternatives to clearcutting and old-growth harvest while still providing for timber production (chapter 3). As a result, alternative thinning methods, including variable-density thinning, have replaced clearcutting as the predominant form of active management on federal

lands, whether for restoration or timber production goals or both (Anderson and Ronnenberg 2013, Lehmkuhl et al. 2015). Ecological objectives for forest management differ by region, forest type, and historic disturbance regime (Franklin and Johnson 2012) (chapter 3).

***Moist forest*—**The focus of silvicultural treatments in moist forests of the western Cascades and Coast Ranges (historically infrequent, high-severity fire regimes) has been an attempt to accelerate development of old-forest conditions in plantations or younger closed-canopy stands (Anderson and Ronnenberg 2013). Typical thinning treatments that create canopy gaps in moist forests west of the Cascade crest can create relatively rapid increases in understory vegetation diversity and productivity (Johnson and Franklin 2013) (chapter 3). The intensity and pattern of retained trees in forest thinning can have dramatic influence on microclimate and ecological response in the short term (Aubry et al. 2009, Heithecker and Halpern 2006). Stand conditions can be either too open or too dense for foraging because spotted owls are adapted to old forest with closed canopies, and the understory must be open enough to fly and access prey (Irwin et al. 2015). In areas where dusky-footed woodrats are primary prey (e.g., southern Oregon, northern California), thinning of young dense stands may increase spotted owl use for foraging, but still not create preferred forest conditions for other life history needs such as nesting and roosting (Irwin et al. 2015). Wilson and Forsman (2013) found that the abundance of mice, terrestrial voles, and shrews increased immediately following thinning, but that northern flying squirrels and red tree voles—important prey species for spotted owls—decreased dramatically in abundance in treated areas for up to 11 years after treatment (Wilson 2010). Thus, spotted owls respond to silvicultural treatments differently where the primary prey are northern flying squirrels, which includes most of the northern and western portions of their range in Oregon, Washington, and British Columbia.

When assessing the potential effects of thinning on prey species, the landscape context should be considered. For example, the effects of thinning within heterogeneous landscapes with well-connected, intact old-forest cover may be less detrimental to northern flying squirrels than if thinning occurs within a highly fragmented forest landscape (Sollmann et al. 2016). Some degree of landscape heterogeneity resulting from forest restoration activities in west-side forests does not adversely impact spotted owls, provided that sufficient large-tree, closed-canopy forest for nesting and roosting is available at core and home range scales (Andrews et al. 2005). For example, in northern California, Franklin et al. (2000) found that territories with the highest fitness (survival and reproduction) were those with a mixture of old forest and about 40 percent of other vegetation types. Diller et al. (2012) reported that forest cover heterogeneity (i.e., juxtaposition of young and older stands) had positive effects on survival and reproduction of spotted owls on commercial timberlands in northern California, where disturbance regimes were historically of mixed severity. Highly productive growing conditions and abundant hardwoods contribute to structural complexity in these managed forests. However, survival of spotted owls decreased in southern Oregon when the amount of nesting/roosting forest cover within the territory center was less than 50 percent (Dugger et al. 2005), and a similar relationship was found in other studies (Franklin et al. 2000, Olson et al. 2004, Wiens et al. 2014).

***Dry forest*—**In the drier forests of the eastern Cascades, southern Oregon, and northern California, wildfire was historically more frequent and burned with mixed- and low-severity effects. In these areas, forest management treatments have focused on accelerating the development of old-forest conditions, but also have focused more on restoring or promoting fire-resilient forest structure, species composition, and landscape pattern (Hessburg et al. 2016, Lehmkuhl et al. 2015, Stine et al. 2014). Landscape managers implementing forest restoration treatments in drier, mixed- and low-severity fire regime forests face substantial challenges in balancing the tradeoffs between known short-term forest cover impacts on spotted owls from restoration and fuel reduction treatments versus potential benefits of reducing losses of forests with larger trees from high-severity, large-scale wildfire (Hessburg et al. 2015, 2016; Lehmkuhl et al. 2015; Stine et al. 2014). Management emphasis on wildfire suppression combined with historical harvest of large trees in these landscapes over the past 100 years has contributed to the recruitment of small-tree, closed-canopy forest

(Hessburg et al. 2016). In these regions, the moderate- to closed-canopy forest with multilayer canopy structure enhanced by dwarf mistletoe infestations are used by spotted owls for nesting and roosting areas, and appear to have increased over the latter part of the 20th century into the 21st century (Davis et al. 2016, Lint 2005). Large tree, multistory canopy typical of forest cover types used for nesting and roosting by spotted owls across their range make them less flammable under most fire conditions, but, like most cover types, these are susceptible to burning intensely in extreme weather. Standard treatments focused on increasing stand-level resilience to wildfire by using prescribed fire and removing ladder fuels (e.g., Cochrane et al. 2012, Safford et al. 2012, Stephens et al. 2009), and reducing canopy connectivity (Agee and Skinner 2005) can reduce the risk of stand-replacement high-severity wildfires, but the practices also remove important forest cover elements for spotted owls and their prey (Lehmkuhl et al. 2006a, 2006b, 2015). Prescribed fire treatments as part of fuel reduction projects can further reduce under- and mid-story canopy complexity, and burn up logs and snags, potentially causing additional negative impacts to suitable forest for spotted owls and their prey (Lehmkuhl et al. 2015). Silvicultural practices that promote spatial and structural complexity have been proposed for retaining suitable foraging conditions for spotted owls while also reducing fuel loads (Churchill et al. 2013, Gaines et al. 2010, Hessburg et al. 2016, Johnson and Franklin 2013, Lehmkuhl et al. 2015). However, the effectiveness of these management practices to restore ecological resilience and reduce risk of loss to high-severity wildfire, while maintaining components of suitable forest for spotted owls, remains to be tested in dry forest landscapes (see chapters 3 and 12 for more discussion of this issue).

Several simulation studies have used coupled wildfire and forest growth models to investigate the relative effects of wildfire and forest restoration treatments on recruitment and retention of forest cover types used by spotted owls in fire-prone landscapes. Some of these studies suggest that certain fuel treatment scenarios (i.e., active management) can reduce wildfire-caused losses of forest cover types used by spotted owls (Ager et al. 2007, Roloff et al. 2012). Other modeling efforts found that active management reduced forest cover used by spotted owls more than simulations with no management, (Roloff et al. 2005, Spies et al. 2017). As with any modeling exercise, outcomes of these studies reflect the assumptions incorporated into the simulations. Assumptions regarding wildfire severity, return intervals, and effects of treatments are particularly influential. One general theme from these simulations is that benefits of fuel treatments to forest types used by spotted owls depend on what probability of occurrence is assumed for future high-severity wildfires. If the likelihood and impacts of high-severity wildfire are assumed to be high, thinning treatments are more likely to have a positive outcome for spotted owls (e.g., Roloff et al. 2012). If the likelihood of high-severity wildfire is assumed to be low, however, then thinning treatments are more likely to produce only declines in the amount of suitable forest cover types used by spotted owls.

Climate Change

Climate change will affect spotted owl populations through changes in weather, forest cover, disturbance processes, prey availability, and other ecological interactions. Population growth of spotted owls appears to be positively associated with wetter than normal conditions during the growing season (May–October), which likely increases prey populations and thus availability (Glenn et al. 2010). Population growth and reproduction were also negatively associated with cold, wet winters (pre-nesting) and the number of hot summer days (July–August) (Diller et al. 2012, Glenn et al. 2011b). Annual survival was more closely related to regional climate conditions (Southern Oscillation Index [SOI] and Pacific Decadal Oscillation [PDO]), whereas recruitment was often associated with local weather. Projected future climate conditions have the potential to negatively affect annual survival, recruitment, and, consequently, population growth rates for spotted owls (Glenn et al. 2010). Climatic factors affecting vegetation and prey abundance likely have a greater effect on reproduction and population growth than direct effects of weather on nestlings or adult spotted owls (Glenn et al. 2011a, 2011b). Climate change models for the first half of the 21st century predict warmer, wetter winters and hotter, drier summers for the Pacific Northwest (Mote et al. 2003) (chapter 2). These conditions are expected to decrease survival of spotted owls in some areas (Glenn et al. 2011a). Climate change can affect development of forest structure by altering temperature and precipitation regimes, and disturbance frequency and intensity (Dale et al. 2001). Altered understory vegetation can reduce prey availability and thus spotted owl fitness (Carey and Johnson 1995, Franklin et al. 2000). Carroll (2010) found that vegetation rather than climate variables best explained distributions of spotted owls. Potential climate-related forest cover losses resulting from large-scale, high-severity wildfires and increased mortality of old-growth trees (Van Mantgem et al. 2009) may be particularly important for future viability of spotted owl populations (chapter 2).

Franklin et al. (2000) found that forest cover patterns explained a high amount of spatial variation in fitness potential among territories occupied by spotted owls in northern California, but climate explained most of the temporal, year-to-year variation in fitness-related traits. Survival and reproduction, for example, were lower when the early nesting period (February–March) was cold and wet. Fecundity, recruitment, and survival decreased across the range of the spotted owl when winters or early springs were colder and wetter than average (Diller et al. 2012; Dugger et al. 2005, 2016; Forsman et al. 1984, 2011). Spotted owl populations in drier forests may be especially vulnerable to climate change because hot, dry summers can reduce prey abundance or availability, and subsequently reduce spotted owl survival (Glenn et al. 2011a). Regional climate patterns, including the SOI and PDO, have also been correlated with demographic rates of spotted owls (Dugger et al. 2016; Forsman et al. 2011; Glenn et al. 2010, 2011a, 2011b). Survival of spotted owls was greater when the PDO was in a warming phase and lower when the SOI was negative (i.e., El Niño events resulting in higher than average temperatures and below normal precipitation) (Dugger et al. 2016).

Extrapolation of the best combination of vegetation-climate models to predicted future climates suggests northward expansion of high-suitability forest cover for spotted owls (Carroll 2010). Increased winter temperature under future climates might be expected to increase winter survival and nesting success, and allow range expansion of prey species such as woodrats, which currently occur at high densities only in the southern portions of the range (Noon and Blakesley 2006). However, it is uncertain how barred owls will respond to changing prey populations, and model results suggest that an initial expansion in the suitable climatic niche may be followed by a contraction as climate change intensifies (Carroll 2010). An important qualifier is that these models did not account for losses of multilayered forests to wildfire and the potential for competition with barred owls to become even more prevalent as climatic change causes shifts in forest communities that in turn further constrain both owl species to a common set of increasingly limited resources.

Other Threats

Genetic diversity and hybridization—
Loss of genetic diversity within a population can contribute to inbreeding depression and decrease adaptive potential. Increased rates of hybridization with barred owls may further compromise the genetic integrity of the spotted owl population (Funk et al. 2010, Gutiérrez et al. 2007). Genetic studies have reinforced other studies that showed spotted owl population declines. Specifically, genetic evidence indicates a loss of genetic variation and increased potential for inbreeding depression in small populations. This suggests a vulnerability of spotted owls to extinction (Funk et al. 2010). Genetic data from spotted owls have indicated population bottlenecks for the Washington eastern Cascades, northern Oregon Coast Range, and Klamath Mountains (Funk et al. 2010), which corresponded temporally with population declines in most of those regions (Anthony et al. 2006, Dugger et al. 2016, Forsman et al. 2011). There was, however, no definitive evidence that suitable forest cover associated with dispersal was limited, or that gene flow was restricted in those regions (Barrowclough et al. 2005, Davis et al. 2011)

Hybridization with barred owls is another potential threat to spotted owl persistence, especially as the spotted owl becomes increasingly rare and the invading species becomes more abundant (Gutiérrez et al. 2007, Haig et al. 2004). Spotted owls occasionally mate with barred owls (male spotted owl–female barred owl mating is most common) and produce fertile hybrids (Hamer et al. 1994, Kelly and Forsman 2004). In the southern portion of the spotted owl range, 3 percent of spotted owl genetic samples collected prior to 2004 (barred owls were still relatively rare on the landscape) contained barred owl mitochondrial DNA (Barrowclough et al. 2005). There are typical markings of hybrids that can be helpful in field identification (Hamer et al. 1994), but genotyping potential hybrids across generations has shown that field identifications were often wrong (Funk et al. 2007). Hybridization rates may also have changed substantially in recent years as barred owl populations have increased and spotted owls have decreased.

Hybridization with other spotted owl subspecies does not appear to be a concern for spotted owl conservation. The northern spotted owl and California spotted owl are two well-differentiated subspecies connected by a narrow hybrid zone in a region of low population density for both subspecies in north-central California (Barrowclough et al. 2005, 2011; Funk et al. 2008; Gutiérrez and Barrowclough 2005). Spotted owls in the contact zone are highly differentiated and may be a distinct population from other northern spotted owl and California spotted owl populations (Miller et al. 2017).

Diseases and pathogens—
Disease exposure could be a secondary consequence of climate change, blood parasites, and effects of barred owl interactions. Lewicki et al. (2015) found that spotted owls had a higher *Haemoproteus* spp. parasite diversity and probability of infection than sympatric barred owls. Further, avian malaria (*Plasmodium* spp.) is common in barred owls, and only recently was documented in spotted owls; therefore, barred owls likely have an additional competitive advantage because spotted owls are potentially immune-compromised owing to recent exposure to avian malaria (Ishak et al. 2008). Spotted owls are susceptible to West Nile virus and experience high rates of mortality when exposed (Courtney et al. 2004); however, it is unknown what, if any, population-level impacts the disease has caused. Wiens et al. (2014) reported that the leading cause of death in a sample of radio-marked barred owls was bacterial infection associated with endoparasitism.

Environmental contaminants—
Environmental contaminants, especially anticoagulant rodenticides, have recently emerged as a potential threat to spotted owls and their prey. In particular, anticoagulant rodenticides used in illegal marijuana cultivation and urban settings can have significant indirect impacts by the poisoning of nontarget forest predators, including owls (Albert et al. 2010, Gabriel et al. 2012, Riley et al. 2007, Stone et al. 1999). To our knowledge, no studies have addressed potential effects of anticoagulant rodenticides on spotted owls.

Research Needs, Uncertainties, Information Gaps, and Limitations

Research Needs

Effects of barred owls—
It has become increasingly clear that barred owls are a primary driver of spotted owl population declines, but many questions remain about the full impact of barred owls directly on spotted owls, and indirectly through alterations of forest communities. Research is needed to build on the work of Wiens et al. (2014) and others to identify potential processes by which spotted owls and barred owls use resources differently. More research is needed to establish the full suite of cause-and-effect relations of barred owl impacts on spotted owls, and how barred owls interact with other threats to spotted owls. Unfortunately, these types of studies are becoming increasingly difficult because spotted owl numbers are declining so rapidly on most study areas. In a pilot study, Diller et al. (2016) found that spotted owls responded positively to experimental removal of barred owls, but additional removal studies in other physiographic provinces, where owl populations and suitable forests are different, are needed. To determine the feasibility and effectiveness of barred owl removals as a tool for spotted owl recovery, the Fish and Wildlife Service and U.S. Geological Survey initiated a barred owl removal experiment on four study areas in Washington, Oregon, and northern California (USFWS 2013). Continued monitoring of spotted owl populations in those areas will be required to fully assess the short- and perhaps, in particular, long-term response of spotted owls to the removal of an important competitor. More genetic studies are needed to address the frequency

and impact of hybridization between spotted owls and barred owls, and how hybridization rates may have changed with changes in abundance of the two species.

It remains uncertain how climate change will affect interactions between spotted owls and barred owls, or even where barred owl populations are in terms of the invasion process. For example, little research has been conducted to investigate if populations of barred owls are continuing to increase or if carrying capacity has been met in some regions. Fundamental information on barred owl distribution and population trends is needed to address this important issue. Further, little is known about barred owl distribution and populations beyond forest cover types occupied by spotted owls. Ecologists are being challenged to predict how spotted owls will change in abundance and distribution under current climate, availability of suitable forest, and competitive interactions with barred owls. It is well documented that climate change influences species' abundances and distributions, and can have indirect effects on interspecific interactions (Angert et al. 2013). An important area of needed research related to barred owl-spotted owl interactions and climate change will be to better understand how the combined effects of barred owl competition and future changes in the amount and distribution of forests used by spotted owls might contribute to spotted owl population persistence and range shifts under a changing climate.

In addition to impacts on spotted owls, changes in the abundance and distribution of a generalist apex predator like the barred owl can have cascading effects on prey populations and food-web dynamics (Gutiérrez et al. 2007, Holm et al. 2016, Wiens et al. 2014). Barred owls have reached densities in the Pacific Northwest that are far greater than historical populations of northern spotted owls (Wiens et al. 2011, 2014). Moreover, as generalist predators, barred owls capture a greater proportion of diurnal, terrestrial, and aquatic prey than northern spotted owls (Forsman et al. 2004, Hamer et al. 2001, Wiens et al. 2014). These life-history traits indicate that barred owls are not direct functional replacements of northern spotted owls in forested ecosystems of the Pacific Northwest (Holm et al. 2016), and that a wide range of prey species may be affected if they replace northern spotted owls. Further research is needed to determine the potential effects of barred owls on other sensitive wildlife beyond spotted owls.

Finally, critical needs for managers are detailed assessments of those locations where spotted owls persist and a better understanding of the effects of forest management activities on interactions between spotted owls and barred owls, and the species individually. Many spotted owl sites with apparently suitable forest structure for nesting and roosting have been abandoned as a result of displacement by barred owls. Those sites that spotted owls have persisted in the face of barred owls may be a result of the behavioral characteristics of the territorial spotted owl, or perhaps those sites have unique forest characteristics that enhance coexistence between the two species. Thinning treatments could potentially affect competitive interactions either by displacing barred owls into areas occupied by spotted owls, or potentially increasing foraging opportunities for barred owls over spotted owls. These and many other responses are plausible, but it remains unknown how either species responds to many forest management techniques. Recent advances in lightweight geographic positioning system telemetry devices and high-resolution forest structure mapping technologies can provide new opportunities for advancing our understanding of these issues.

Prey populations and population performance— Previous studies have characterized the diet of spotted owls in different portions of the subspecies' range (Barrows 1980; Bevis et al. 1997; Cutler and Hays 1991; Forsman et al. 1984, 2001, 2004), investigated the relationship between forest cover selection, home-range size, and prey availability (Carey et al. 1992; Forsman et al. 1984, 2005; Irwin et al. 2000; Zabel et al. 1995), and evaluated diet overlap with barred owls (Hamer et al. 2001, Wiens et al. 2014). The importance of understanding relationships between spotted owl populations and their prey has repeatedly been acknowledged (Clark et al. 2011, Courtney et al. 2004, Forsman et al. 2004, Glenn et al. 2010, Olson et al. 2004, Rosenberg et al. 2003, Thomas et al. 1990, Wilson and Forsman 2013, Zabel et al. 1995). However, to our knowledge, no efforts have been undertaken to quantify the relationship between interannual fluctuations in prey abundance and long-term demography of spotted owls. Research is needed to understand how spotted owl reproduction, stress levels, and survival are influenced by prey species composition and abundance, and how prey populations are influenced by disturbance or fluctuations in weather and climate. Population fluctuations in small mammals have been linked with variation in precipitation (Avery et al. 2005, Crespin et al. 2002). However, identifying the mechanisms by which climate influences population processes of spotted owls and their prey remains a challenge (Glenn et al. 2011a).

A better understanding of the effects of thinning treatments and the impacts that anticoagulant rodenticides have on spotted owl prey populations will be critical for managers. Research and an effect analysis is needed to address thinning impacts on spotted owl prey, both within treated stands and at broader landscape scales. This information would contribute to thinning prescription development throughout the range of the spotted owl. The use of anticoagulant rodenticides in natural systems is increasing, especially in areas where illegal marijuana cultivation is prevalent. Studies are also needed to better understand the individual- and population-level impact of rodenticides on spotted owls, and development of management options to reduce the ecological impacts.

Landscape restoration, silvicultural treatments, prescribed fire, and wildfire in moist and dry forests— Research is needed in both dry and moist forest landscapes to evaluate the short- and long-term effects of silvicultural treatments and wildfire on spotted owl occupancy, forest dynamics, and prey, but research questions differ between forest types. For example, the optimization of forest restoration and conservation of spotted owls will require more knowledge about the conditions under which restoration activities can benefit spotted owls in the long term without significant detrimental impact in the short term. Restoration activities and objectives are different between moist and dry forest landscapes. Current conditions in dry forests are generally not sustainable, and some measure of treatment is needed to increase fire resiliency of forest stands in at least some locations (USFWS 2012b). In these fire-prone landscapes, a common objective is to modify and reduce fuels to alter wildfire behavior and to manage for ecological integrity based on the natural range of variability (USDA 2012). Additional information is needed to evaluate the consequences of fuels reduction and restoration treatments relative to the long-term benefits of forest restoration, particularly as large, high-severity fires are expected to become more frequent because of climate change. This is especially true in the frequent low-severity fire regime of the eastern Cascades, where environmental conditions favor open pine-dominated forests. Studies are needed to identify resilient sites for spotted owls in the face of changing forests (e.g., species composition changes) caused by climate change, active forest management, and increased wildfire occurrence.

In moist forest landscapes, research is needed to determine how or if spotted owls use forest stands where thinning has been conducted to accelerate the development of late-successional forest characteristics. If spotted owls avoid these areas in the short term, work is needed to understand the time before they begin using the areas again. To fully understand restoration effects, long-term before/after control-impact studies are needed to elucidate spotted owl and prey responses to forest restoration treatment effects in different ecotypes.

Research to address restoration and silvicultural treatment on spotted owl space use and forest structure development will also need to account for the potential confounding impact that barred owls are likely to have on spotted owl response to restoration efforts. Beyond a better understanding of spotted owl response to silvicultural treatments, managers need information regarding how sympatric populations of barred owls respond to treatments. Additionally, research is needed to understand the effectiveness of ecosystem-scale conservation versus conservation that targets one particular stage of succession (e.g., late-successional forest characteristics for spotted owls). Finally, much more information is needed to evaluate the short- to long-term effects that wildfire has on spotted owls in all landscapes, with a focus on the relative susceptibility of old forest and young forest to high-severity wildfire under a range of weather conditions. Finally, it is important to note that these research topics become increasingly difficult to address as spotted owl populations decline and fewer individual owls are available to study in some landscapes.

Physiological consequences of stress—
An animal's ability to cope with stressors is an important determinant of its physiological conditions, and therefore, health and survival. Environmental perturbations and an individual's response can affect the body's production of hormones, such as glucocorticoids, with negative physiological consequences (Carrete et al. 2013, Strong et al. 2015). For many species, the level of stress hormone corticosterone can be an effective predictor of survival probabilities, reproduction, dispersal, and can have population-level impacts (Carrete et al. 2013, Romero and Wikelski 2001, Romero et al. 2000). Quantification of corticosterone in feathers, which is stable over time, represents an integrated measure of stress levels (Bortolotti et al. 2009, Sheriff et al. 2011). Stress hormones are accumulated in feathers during growth, so can provide a measure of stress levels during that time, and can be a strong predictor for future survival of individuals (Koren et al. 2012). Variation in feather corticosterone can also be quantified among individuals of a population, as well as through time to track stress over space and time to address questions about the health and ecology of a population (Bortolotti et al. 2009).

Hayward et al. (2011) found that spotted owls had a glucocorticoid response to acute noise disturbance and that spotted owls with nests near noisy roads fledged fewer young than those near quiet roads. Corticosterone analyses are needed to determine the physiological response to acute and prolonged exposure to environmental stressors (e.g., barred owls, prey abundance, weather, and human-caused disturbance) and response activity for both juvenile and adult spotted owls. Our understanding of spotted owl ecology will be improved with studies to evaluate the associations between stress levels and survival, reproduction, and dispersal of spotted owls. From a management perspective, it is important to understand the stress response of spotted owls related to management activities like prescribed fire, road construction, various logging systems, and the timing of these activities. Additional research will be important to understand key stressors for spotted owls and inform seasonal restrictions on human activities that can increase stress levels.

Dispersal and suitable forest connectivity—
Dispersal behavior for both juveniles and adults may increase survival and reproductive success, but also increase risks to establishing a home range in an unfamiliar landscape. Juvenile spotted owls disperse within their first year and the condition of matrix forest types between natal and breeding sites can facilitate or hamper survival and movement processes (Forsman et al. 2002). Available information for spotted owls suggests that stands used for roosting during natal dispersal movements have very similar structure as those stands used for nesting and roosting activities of adults (>70 percent canopy cover and large trees >50 cm d.b.h.), but this finding is based on only two studies with no data throughout most of the geographic range (Miller et al. 1997, Sovern et al. 2015). Further research is needed to understand the contemporary dynamics of juvenile dispersal because many assumptions are made about what constitutes forest cover suitable to facilitate dispersal by spotted owls. A better understanding of the forest structure and configuration characteristics of forest conditions that facilitate juvenile dispersal is needed to ensure demographic connectivity among isolated patches of remaining old forests. Further, it remains unknown how barred owls influence juvenile spotted owl survival or dispersal. It is possible that some of these questions could be addressed with a thorough analysis of existing dispersal data from demographic study areas.

Historically, adult spotted owls exhibited strong nesting-site and mate fidelity, with fewer than 8 percent of individuals dispersing to a different territory between years (Forsman et al. 1984, 2002). In recent years, however, field observations suggest that interterritory movements by resident spotted owls are increasing, and that such movements appear to coincide with the colonization of barred owls (Dugger et al. 2011, Olson et al. 2005). Research that addresses how forest alteration and the presence of barred owls interact with social conditions on territories to affect movement decisions and survival of individual spotted owls will improve our ability to implement forest management practices that benefit spotted owls. In addition to helping land managers identify the range of conditions within individual owl territories that promote high site fidelity and survival, such data can also provide a powerful framework for testing broad ecological theories about the causes and consequences of breeding dispersal in a long-lived predatory bird with declining populations.

Testing alternative monitoring protocols—
When the NWFP was developed, mark-recapture and random census (i.e., occupancy framework; the proportion of sites occupied by spotted owls) population monitoring methods were both considered. The decision was made to use the mark-recapture method, which was already in use. Precise estimates from mark-recapture studies require large samples of marked spotted owls; therefore, Lint et al. (1999) recommended the use of an independent estimate of population trend for comparison with the results from spotted owl demographic studies. Monitoring in an occupancy framework (i.e., MacKenzie et al. 2006) could provide an independent, empirical assessment of population trends to compare with estimates of the annual rate of population change. Because of uncertainty about the precision of the occupancy-based approach, Lint et al. (1999) recommended that statistical power and cost effectiveness of the method be explored.

The low number of spotted owls in some study areas suggests that passive acoustic monitoring may be an effective solution for future monitoring of spotted owl populations. Traditional call-back surveys at night (playing spotted owl calls and listening for a spotted owl response) are labor intensive, more risky compared to daytime work, and only generate reliable data for spotted owls. Further, detection probabilities for spotted owls—using call-back surveys—are negatively influenced by the presence of barred owls, and barred owls often do not respond to spotted owl calls (Bailey et al. 2009). Call-back surveys could also have unintended consequences by exposing spotted owls to predation or harassment by barred owls or great-horned owls. Primary advantages of passive acoustic monitoring are as follows: (1) surveys do not require an elicited response from target species; (2) surveys are able to detect and do not bias against many other species (e.g., barred owl, marbled murrelet, western screech-owl, northern pygmy-owl, northern saw-whet owl, and many others); (3) increased crew safety because all work would be conducted during daylight hours; (4) biological training and expertise needed for crew members will be much less than is needed for call-back surveys and demographic studies; and (5) sound recordings provide a permanent record of the detection. A limitation of this approach is the time required to process recordings and data storage. Automated call detection technology has been developed, but improvements are needed, especially for call recognizers for rare birds in areas with excessive background environmental noise (e.g., rain, streams). Research is needed to test alternative methods that take advantage of technological advancements in noninvasive detection equipment to monitor trends in rare populations. The transition to alternative methods to monitor spotted owl populations will be most effective if new methods have spatial and temporal overlap with traditional methods so that robust comparisons can be made between historical and contemporary data.

Population simulation modeling—
The program HexSim (Schumaker 2015) provides a simulation framework for systematically investigating factors that influence population function, including forest conservation scenarios and emergent competitors. The implementation of HexSim by the USFWS (2011b) did not include spatially explicit representation of spotted owl interactions with barred owls. Modeling exercises that incorporate a more sophisticated representation of population interactions with barred owls are needed to simulate and predict responses of spotted owls to experimental removal of barred owls. Two-species models implemented in HexSim could also be used to simulate potential efficacy of long-term management programs for barred owls and spotted owls relative to critical habitat designations. Current modeling efforts are female-only models. A two-sex HexSim implementation for the spotted owl population is needed to get at small population processes (e.g., Alee effects and stochasticity in sex ratios) that can drive extinction.

Scientific Uncertainty

Survival estimates—

Adult survival is typically the most important factor influencing population performance in long-lived raptors, and survival estimates for spotted owls have been the focus of extensive research and monitoring. As in other meta-analyses of spotted owl demographic data (e.g., Burnham et al. 1996, Dugger et al. 2016, Forsman et al. 2011), Anthony et al. (2006) used capture-recapture methods to estimate apparent survival rates of spotted owls. Apparent survival is the product of probabilities that an animal survives and remains in the population. If a marked animal permanently emigrates, then it is, for purposes of the estimate, presumed dead, because emigration and mortality are confounded. Further, fates are not known for all individuals because recapture probabilities are less than one even when animals remain in the population. Therefore, models based on capture-recapture data account for imperfect encounter rates in estimates of survival (i.e., apparent survival). Apparent survival rates on individual study areas ranged from 0.75 (± 0.03) to 0.89 (± 0.01) for adults, 0.63 (± 0.07) to 0.89 (± 0.01) for 2-year-olds, and 0.42 (± 0.11) to 0.86 (± 0.02) for 1-year-olds. They found negative effects of reproduction and barred owls in survival rates on several study areas (Anthony et al. 2006).

Elsewhere, Loehle et al. (2005) used telemetry to study annual survival of spotted owls and obtained a known-fate estimate of 0.93 (± 0.07), which was considerably higher than the apparent survival estimates reported by Anthony et al. (2006). Known-fate models estimate survival rate when fates (i.e., alive or dead) of individuals can be determined with certainty. Loehle et al. (2005) used their results to cast doubt on apparent survival estimates from mark-recapture studies of spotted owls. They suggested that survival estimates from mark-recapture studies were too low because some marked individuals left the study areas and were assumed to be dead. Anthony et al. (2006) estimated a declining spotted owl population; Loehle et al. (2005) suggested that the true population change for spotted owls was likely stable and not declining. In response, Franklin et al. (2006) argued that Loehle et al. (2005) had inappropriately compared their study with the work of Anthony et al. (2006) in a number of ways, including (1) the manner in which missing radio-marked individuals were removed from analyses may have overestimated survival; (2) telemetry-based estimates of survival were not valid for estimating bias; and (3) results from the telemetry-based study should not be compared to the capture-recapture study because study areas differed dramatically in size and distribution. Both apparent survival estimates from mark-recapture data and known-fate estimates from telemetry studies are valid estimates of annual survival. However, in this circumstance it was inappropriate to compare telemetry-based survival estimates with results from capture-recapture studies, which was acknowledged by both sides of the disagreement (Franklin et al. 2006, Loehle and Irwin 2006).

Wildfire risk—

The 2008 recovery plan (now withdrawn) for spotted owls (USFWS 2008) suggested a change in the LSR network as the foundation of conservation strategies established in the NWFP. Because of concern about wildfire, the plan recommended a switch from a reserve to a no-reserve strategy in up to 52 percent of the spotted owl's range. For dry forests, the plan recommended thinning stands at regular intervals to reduce fuel loading, and thus wildfire risk. Hanson et al. (2009) suggested that the estimates of wildfire risk used by the USFWS (2008) were overestimated and that there was not a strong basis for major changes to the NWFP conservation strategy for the spotted owl. Spies et al. (2010) defended the estimates of wildfire risk and suggested that Hanson et al. (2009) had underestimated wildfire risk and were biased against active management. Hanson et al. (2010) then responded by calling for less focus on fuel treatments in the recovery plan for the spotted owl. Because of uncertainty about future wildfire occurrence, spatial extent, and severity, we cannot know with complete confidence whether wildfire risk has been over- or underestimated in these efforts. Both the 2008 critical habitat designation and the 2008 recovery plan were challenged in court, and the inspector general of the Department of the Interior issued a report concluding that the decisionmaking process for the recovery plan was potentially jeopardized by improper political influence (Devaney 2008, USFWS 2011a). The court ordered the Fish and Wildlife Service to withdraw the 2008 recovery plan and issue a revised recovery plan and critical habitat designation.

Spies et al. (2017) projected that the extent of forest cover suitable for spotted owls in the eastern Oregon Cascades is expected to increase in coming decades under recent historical frequencies and severities of wildfire (and current levels of wildfire suppression). Treating the landscape to reduce potential loss of suitable forest cover for spotted owls with high-severity wildfire still resulted in increases in that forest cover type, but not as much as would occur without management. The results suggest that managing for resilience to fire and climate change could occur without necessarily reducing forest cover from its current levels (younger forest is growing into older closed-canopy forests to replace dense forests lost thinning or wildfire). However, these outcomes are likely to be different under climate change or if an alternative landscape-scale treatment design is used (Spies et al. 2017).

Despite the potential negative effects on spotted owl habitat, the overwhelming consensus in the scientific literature is that active management in dry forests is appropriate to reduce wildfire risk and improve ecosystem function. Therefore, the 2011 revised recovery plan (USFWS 2011b) and 2012 critical habitat designation (USFWS 2012a) for spotted owls contained proposals for active management in dry forests. In some regions, project planning has moved forward, and federal land managers are consulting with the Fish and Wildlife Service on a case-by-case basis. The debate about active management related to wildfire risk for forests used by spotted owls remains unresolved and reflects different goals (e.g., ecosystem versus single species) and assumptions about wildfire risk with a changing climate. These differences of opinion highlight legitimate concerns about where to place the burden of proof regarding ecosystem versus species management, but the fundamentals of this controversy lie in the diversity of philosophical views about ecological goals and the role that active management should play on public lands (see chapter 12).

Restoration framework—

Franklin and Johnson (2012) outlined a series of recommendations for an "ecological forestry" framework and a forest restoration strategy within the Plan area that reflect many of the elements of the revised spotted owl recovery plan (USFWS 2011b). They called for reserving older forest stands, thinning plantations to accelerate development of structural complexity, and implementing variable-retention harvests in younger forests to help provide diverse early-seral ecosystems on moist forest sites. On dry forest sites, their strategy called for silvicultural treatments that retain and release older trees, reduce stand densities, shift composition toward fire- and drought-tolerant tree species, and incorporate spatial heterogeneity at multiple spatial scales (Franklin and Johnson 2012). The framework included an extensive set of large patches of dense forests on approximately 30 percent of the forested landscape to retain some suitable forest for spotted owls while reducing the potential for landscape-level high-severity wildfires.

DellaSala et al. (2013) identified seven areas in which the ecological forestry framework may fall short of the stated goals of the NWFP, and offered 14 recommendations to improve the framework and its implementation. They also criticized decisions to incorporate some of the elements of ecological forestry in the revised recovery plan and revised critical habitat designation. Henson et al. (2013) agreed with many of the recommendations made by DellaSala et al. (2013), but differed on two key perspectives. Henson et al. (2013) regarded the potential impacts of wildfire to spotted owls as higher risk to species persistence, and suggested that in many circumstances, the adverse effects associated with active management may be preferable to adverse effects of passive management. As with wildfire risk, the fundamentals of this debate reside in philosophical disagreements about ecological goals and what role active management should play in managing public lands. Most research in dry or frequent-fire forest landscapes suggests that active management is needed to achieve or accelerate restoration objectives, but more study is needed to advance our understanding of disturbance effects on wildlife dependent on old forest, especially interactions between wildfire and a range of prefire and postfire active management actions.

Modeling to inform critical habitat designation— The Fish and Wildlife Service (USFWS 2012a) produced maps of distribution of potentially suitable habitat for spotted owls that did not include the effects of barred owls on spotted owl distribution, but the effort did incorporate the spatial arrangement of forest structure associated with nesting/roosting and foraging, and abiotic factors such as slope and topographic position, to determine the extent of critical habitat. In an alternate analysis, Loehle et al. (2015) conducted an accuracy assessment of vegetation data used as input to develop the USFWS (2012a) models, used independent locations to validate model prediction, correlated model output with spotted owl reproductive success in two study areas, and developed alternate models. Their independent locations and vegetation evaluations suggested a high rate of classification errors, and productivity did not correlate well with predictions in their study areas (Loehle et al. 2015). Dunk et al. (2015) defended the critical habitat model as scientifically rigorous and as meeting the goals established by the Fish and Wildlife Service. They suggested that Loehle et al. (2015) mischaracterized the literature and the Fish and Wildlife Service species distribution model, failed to demonstrate the locations used by the agency were biased, and failed to show significant flaws in analytical methods.

Bell et al. (2015) argued that Loehle et al. (2015) underestimated the predictive performance of critical habitat maps because the field plots they used potentially biased the accuracy assessment toward older forests, and that they examined accuracy at finer scales than the model was intended to predict. Loehle and Irwin (2015) responded to Bell et al. (2015) and Dunk et al. (2015) by arguing that, although the habitat models average out at large spatial scales, errors at smaller scales may limit their utility for conservation. This debate underscores the importance of acknowledging the appropriate scale at which predictive distribution models can be used for conservation purposes. The debate also serves as another example highlighting the need to recognize and carefully evaluate how habitat is defined. The definition of habitat for spotted owls must now consider that forests that were once suitable for spotted owls are less suitable habitat if occupied by barred owls.

Conclusions and Management Considerations

Spotted owls are a resilient subspecies but are faced with significant challenges. Research and monitoring efforts over the past several decades have documented the population declines and risks to spotted owls despite measures to address their long-term sustainability. The framework, standards, and guidelines of the NWFP have been both critical and necessary for spotted owl conservation, and underlie species recovery plans. However, because of barred owls and continued forest perturbations outside of federal lands, the NWFP alone is not sufficient for spotted owl recovery. Additional measures beyond the Plan will be needed for long-term persistence of spotted owls. Suitable habitat continues to decline because of current and lingering effects of extensive forest disturbance, and the recent invasion of a formidable congeneric competitor has reduced the space available for spotted owl recovery. The need to provide habitat for spotted owls has been a critical component of conservation plans and was a major catalyst for developing the NWFP. It is now clear that barred owl presence reduces habitat suitabiliy for spotted owls, so species recovery will require protections for old forest and management actions focused on reducing the threat from barred owls. After only two decades, it is too early to evaluate if the Plan has been effective at improving the conservation status of spotted owls; however, the framework, standards, and guidelines of the NWFP have aided spotted owl conservation; if logging had continued at pre-NWFP levels, spotted owl populations certainly would have declined more rapidly over the past 20 years. Further, the NWFP has put federal lands on a trajectory for providing enough suitable forest for recovery of spotted owl populations over the next several decades. The effectiveness of LSRs established under the NWFP is linked to the frequency, severity, spatial extent, and type of disturbance, as well as how those disturbances are offset by recruitment of suitable forest, primarily through succession. Disturbance events can reduce the suitability of forests used by spotted owls for several decades by creating open canopy conditions and reducing structural complexity. Although disturbance rates have exceeded suitable forest-cover recruitment rates during the first 20 years of the NWFP, recruitment will likely outpace losses if current timber harvests and wildfire occurrence remain constant. However, climate models suggest that wildfire occurrence may increase, causing significant reductions in cover for spotted owls, and that suitable forest cover for spotted owls will move northward and occur at higher elevations. Therefore, other reserves designated before development of the NWFP, such as parks and wilderness areas, may become increasingly important for spotted owl conservation.

Several lines of compelling evidence indicate that interspecific competition between spotted owls and barred owls is causing accelerated population declines of spotted owls, despite widespread conservation of old forests under the NWFP. Competitive pressure from barred owls may negate the benefits of recruitment of suitable forest cover, because barred owls exclude spotted owls from sites that otherwise are suitable for spotted owls. It remains uncertain how, or if, spotted owls can coexist with barred owls. Although much research has been done on spotted owls, we identified many uncertainties in available information and have identified future research needs important for management of the subspecies. The long-term effects of barred owls and fine-scale partitioning of resources remain unknown, and studies are needed to identify resilient sites for spotted owls in the face of competitive interactions with barred owls, if they exist. Additionally, it remains unknown how, or if, spotted owls will respond to removals of barred owls from historical spotted owl territories.

Abundance and distribution of primary prey species can influence home range size and forest selection by spotted owls. But it remains unknown how spatially and temporally fluctuating prey populations influence the survival and reproduction of spotted owls. Studies are needed to quantify relationships between interannual fluctuations in prey abundance and long-term demography of spotted owls. The short- and long-term effects of silvicultural treatments and wildfire on spotted owl occupancy, forest dynamics, and prey remain unclear. The optimization of forest restoration and conservation of spotted owls will require more knowledge about the conditions under which restoration activities can benefit spotted owls in the long term without significant detrimental impact in the short term.

Management Considerations

Forest management and barred owls—
Wiens et al. (2014) found that adult survival of spotted owls and barred owls was higher in home ranges with greater amounts of conifer forest dominated by trees age 120 years or older. Dietary studies also showed that barred owl diet is broader than spotted owls, but both owl species relied on similar prey associated with older forest types (e.g., northern flying squirrels and red tree voles). These findings have important implications for land managers because they suggest that (1) conservation of old forest under the NWFP not only promotes survival of spotted owls, but also survival of barred owls; and (2) availability of old forests (and associated food resources) is a key limiting factor in the competitive relationship between the two owl species (Wiens et al. 2014). As barred owls continue to increase in number, it has become clear that conservation of the spotted owl and its forest cover types need to be extended from ameliorating the effects of old-forest loss and fragmentation to accounting for the impacts of a widespread invasive competitor as well. Although spotted owls are known to use recently thinned stands (e.g., Irwin et al. 2015), it remains unclear how such silvicultural treatments can affect the fitness of spotted owls in the long term or how barred owls may respond to those management actions. Those silvicultural treatments with high disturbance likely increase long-term extinction rates of spotted owls by reducing forest complexity and thus suitability for spotted owls but not necessarily for barred owls (Dugger et al. 2016, Singleton 2015, Singleton et al. 2010, Sovern et al. 2014, Wiens et al. 2014).

Barred owl densities may now be high enough across the range of the spotted owl that, despite the continued management and conservation of suitable forest cover types under the NWFP, the spotted owl population will continue to decline without intervention to reduce barred owl populations (Dugger et al. 2016). Recommendations to conduct experimental removal of barred owls to benefit spotted owls have been criticized as being too difficult to accomplish owing to the effort and cost required to maintain sufficiently low numbers of invasive barred owls (Livezey 2010, Rosenberg et al. 2012). Nonetheless, experimental removal of barred owls on one study area in California suggests that removal of barred owls may have positive, short-term effects on population trends of spotted owls (Diller et al. 2016, Dugger et al. 2016). In 2013, the Fish and Wildlife Service decided to expand removal experiments to additional sites in California, Oregon, and Washington to determine if similar results can be obtained in areas with different forest conditions and densities of barred owls (USDI 2013, USFWS 2013). Those experiments will yield information about how spotted owls respond, and will convey the economic and logistic feasibility of removal efforts as potential management actions. Such information will be useful in projecting possible long-term consequences and benefits of an active management program for barred owls in the future.

Current evidence suggests that a combination of habitat protection and active management of barred owls are the two highest priorities for stabilizing declining trends in populations of spotted owls. A recent analysis casts doubt on the likely effectiveness of barred owl removals for spotted owl conservation (Bodine and Capaldi 2017). Experimental culling of barred owls will provide information to validate those models and about how, or if, their populations can be controlled at scales sufficient to promote recovery of spotted owls. However, detailed studies of habitat associations and resource use by barred owls have been conducted in only a few limited areas within the range of the spotted owl. More detailed studies in other areas will better enable an understanding of how specific tree species, stand densities, or physiographic conditions are negatively associated with barred owls but not spotted owls.

Wildfire and active management—
Disturbance processes that increase forest or landscape heterogeneity (e.g., wildfire, management activities) can benefit spotted owls as long as the required forest structural conditions are available for foraging, nesting, and roosting activities. Processes that substantially simplify stand structure or landscapes often have negative impacts on the suitability of forest for spotted owls. Our basic understanding of forest structural conditions used by spotted owls has not substantially changed over the past 20 years, but there has been a growing recognition of the contribution of diverse forest conditions to broader ecosystem function and species diversity in conifer forests of the Pacific Northwest. This is especially true in historically moderate- and high-frequency fire regime landscapes where fire suppression and forest management have greatly reduced fire and altered forest structure and composition at stand and landscape scales (chapter 3). For example, nonconiferous vegetation, including shrubs and broad-leaved trees, makes an important contribution to the diversity of forest landscapes. Therefore, allowing shrubs and hardwood trees to develop and persist in early-seral stands, and curtailing vegetation control, will benefit many wildlife species associated with nonconiferous vegetation (Hagar 2007), including some spotted owl prey species (Diller et al. 2012). Additionally, diversity and configuration of different forest types are important for spotted owls at stand, home range, and landscape scales (Franklin et al. 2000). The function and diversity of an ecosystem is enhanced by the presence of high-quality early-seral patches (i.e., a mix of nonforest and forest) because they have high species and structural diversity (Swanson et al. 2011). These early-seral ecosystems can be created using low-intensity approaches for regeneration, combined with retention of biological legacies to promote the development of structurally diverse closed-canopy forest over time (Franklin and Johnson 2012). Indeed, under normal conditions, natural disturbances frequently result in patches of high-quality early-seral ecosystems, provided that intensive salvage and replanting does not occur after the disturbance (Swanson et al. 2011).

Disturbances have different impacts on spotted owls depending on the scale under consideration. A hypothesis that has emerged from recent research is that disturbance processes (e.g., low- and mixed-severity wildfire, light to moderate thinning) that increase stand or landscape heterogeneity can have long-term benefits for spotted owls, as long as enough suitable forest cover for nesting and roosting remain within the territory. Conversely, disturbances that substantially simplify stands or landscapes often have long-lasting negative impacts on spotted owls and their habitat. Finally, we emphasize the importance of conserving sites currently occupied by spotted owls as well as those that are known to have been historically occupied by the subspecies. Many sites, for example, have been abandoned as a result of disturbance to suitable forest cover or displacement by barred owls, but maintain structure suitable for nesting and roosting. Those remaining spotted owls and sites likely represent unique behavioral or forest characteristics that may not yet be fully recognized, thus they are an important research need. Conserving the unique forest structural conditions of those few sites that remain, particularly in the northern portion of the geographic range, will likely have a positive benefit for the long-term persistence of spotted owls.

Prognosis for the future—

In the 2011 revised recovery plan for spotted owls, the Fish and Wildlife Service's modeling team used the HexSim modeling program (Schumaker 2008) to simulate population-level responses to various conservation strategies and other threats (USFWS 2011b). They developed models based on demographic data (Forsman et al. 2011), dispersal information (Forsman et al. 2002, Thomas et al. 1990), and home range size (Carey et al. 1990; Forsman et al. 1984, 2005; Glenn et al. 2004; Hamer et al. 2007). Objectives of the modeling effort were to (1) evaluate if future viable populations of spotted owls were likely given conditions at the time (demographic rates, LSR network, amount of suitable forest cover, barred owls); (2) estimate population viability under different conservation networks of suitable forest cover; and (3) quantify the effect of forest cover and barred owl management on recovery goals for spotted owls (USFWS 2011b). The modeling results suggested that availability of suitable forest cover was critical for territory acquisition and sustained occupancy by spotted owls. Population viability models suggest that barred owls reduce spotted owl survival and act to depress populations to about half of potential population size without barred owls (fig. 4-9). Simulations did not include the barred owl impact on spotted owl reproduction, forest selection, site fidelity, or detection probability, and were based upon early rates of population growth. More recent population change estimates (Dugger et al. 2016) indicate a further declining growth

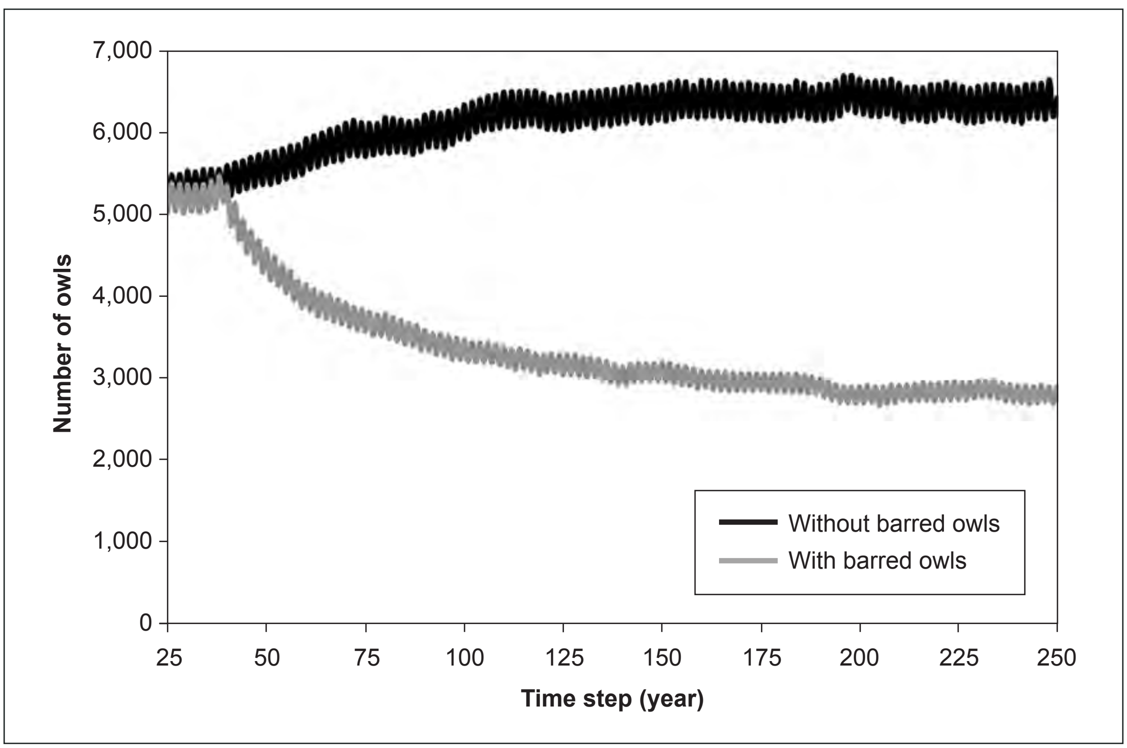


Figure 4-9—HexSim model runs with five replicates each for without barred owl impacts and with barred owl impacts for the spotted owl's geographic range in the United States. The apparent within-year variation that appears in the figure is a function of an "even-odd" year effect on reproduction (USFWS 2011b). The first 30 years of the simulation was a "burn-in" period, which provided for the simulated population to distribute according to available resources and develop an age structure determined by demographic processes. Barred owl effects were not included during the "burn-in" period and were introduced starting at year 30 (USFWS 2011b).

rate, suggesting that USFWS (2011b) projected estimates are more optimistic than what is likely to be observed in spotted owl populations. These studies provide further evidence that the framework, standards, and guidelines of the NWFP are critical components to spotted owl recovery plans, but the impacts of barred owls will likely need to be controlled if spotted owl species recovery is to be successful.

Schumaker et al. (2014) used the HexSim model originally developed by the Fish and Wildlife Service (USFWS 2011b) to simulate and quantify source-sink dynamics and landscape connectivity throughout the range of the spotted owl. Their results indicated that populations are likely to decline in most regions, but that southern Oregon and northern California may serve as source populations. Marcot et al. (2013) also used the HexSim model to evaluate how size and spacing of suitable forest cover types for spotted owls affected simulated population size and persistence. Their results indicated that long-term occupancy rates were significantly higher with suitable forest patches large enough to support 25 spotted owl pairs or more, with less than 9.3 mi (15 km) spacing between patches, and with overall landscapes of at least 35 to 40 percent suitable forest cover types for nesting and roosting. In a sensitivity analysis, Marcot et al. (2015) determined that spotted owl response variables in the HexSim model were most sensitive to the availability of highly suitable forest cover for nesting and roosting. All these studies used static habitat maps that did not incorporate climate change or wildfire impacts on spotted owls. Only the USFWS (2011b) model incorporated effects of barred owls.

Spotted owl populations have continued to decline under the NWFP, but because of slowed timber harvest on federal lands since the late 1980s, forests throughout most of the range of the spotted owl are on a trajectory—through succession—to develop suitable forest characteristics for spotted owls in coming decades. When the NWFP was adopted, spotted owl populations were expected to continue declining for up to 50 years because of lingering impacts of previous losses of suitable forest cover, yet the magnitude and characteristics of barred owl impacts were unknown and unexpected at that time. Per assumptions of the NWFP, we are unable, after only two decades, to use stable or increasing populations (i.e., improved conservation status) of spotted owls as the success criterion for the NWFP. However, if the success criterion is forests capable of supporting interconnected populations of spotted owls in the absence of barred owls, then the implementation of the framework, standards, and guidelines of the NWFP has put federal lands on a trajectory for success, despite recent losses of suitable forest cover to wildfire. In the Pacific Northwest, forest succession from early-seral to climax forest is a slow process, which is in part the reasoning for the NWFP to be a 100-year plan intended to span several human generations (USDA and USDI 1994). Further, conservation and management of spotted owls rests critically on continued implementation of the protections afforded by the NWFP and the Endangered Species Act (Noon and Blakesley 2006). It also rests on improving our understanding of how to minimize impacts of barred owls, and on fine-tuning our ability to retain needed forest structure while also increasing resiliency of forests through strategic management.

U.S. and Metric Equivalents

When you have:	Multiply by:	To get:
Inches	2.54	Centimeters
Meters (m)	3.28	Feet
Hectares (ha)	2.47	Acres

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Marbled murrelet.
Photo by Kim Nelson, Oregon State University.

Chapter 5: Marbled Murrelet

Martin G. Raphael, Gary A. Falxa, and Alan E. Burger[1]

Introduction

In this chapter, we describe expectations of the Northwest Forest Plan (NWFP, or Plan) and review recent science on the ecology and status of the marbled murrelet (*Brachyramphus marmoratus*), with an emphasis on the portion of the species' range that falls within the Plan area. The conservation strategy embodied in the NWFP evolved from designation and protection of a large number of relatively small management areas to an approach based primarily on the designation of fewer large areas, each designed to conserve functioning late-successional and old-growth ecosystems. These were intended to support multiple pairs of northern spotted owls (*Strix occidentalis caurina*) and murrelets, and to conserve habitat for other species associated with older forests.

The marbled murrelet is a small seabird of the family Alcidae (fig. 5-1) whose summer distribution along the Pacific Coast of North America extends from the Aleutian Islands of Alaska to Santa Cruz, California (fig. 5-2). It forages primarily on small fish and krill in the nearshore (0 to 2 mi [0 to 3 km]) marine environment. Unlike other alcids, which nest in dense colonies on the ground or in burrows at the marine-terrestrial interface, murrelets nest in more dispersed locations up to 55 mi (89 km) inland. In the southern portion of the range, including the Plan area

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Figure 5-1—The marbled murrelet is a small seabird of the family Alcidae.

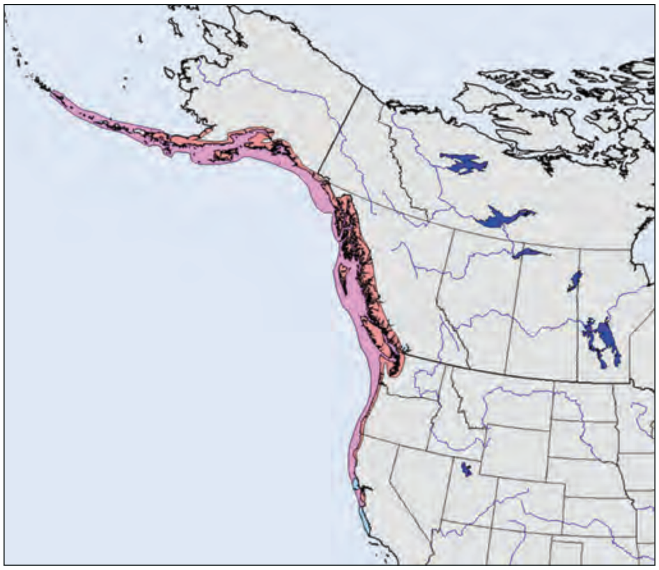


Figure 5-2—Range of the marbled murrelet in North America. Map by Terry Sohl from NatureServe data.

and the area emphasized in this chapter, murrelets typically nest in large coniferous trees in forested areas containing characteristics of older forests. Throughout the forested portion of the species' range, murrelets typically nest in areas containing characteristics of older forests (Baker et al. 2006; Binford et al. 1975; Hamer and Cummins 1991; Hamer and Nelson 1995; Hamer et al. 1994; Hébert and Golightly 2006; Quinlan and Hughes 1990; Ralph et al. 1995a; Singer et al. 1991, 1992; Wilk et al. 2016). The marbled murrelet population in Washington, Oregon, and California nests in most of the major types of coniferous forests (Hamer and Nelson 1995) in the western portions of these states, wherever older forests remain inland of the coast at elevations primarily below the extent of the true fir zone, generally <4,000 ft (1220 m) (table 5-1). Although murrelet nesting habitat characteristics may differ throughout the range of the species, some general habitat attributes are characteristic throughout its listed range, including the presence of nesting platforms, adequate canopy cover over the nest, larger patch size of mature forest, and being within commuting distance to the marine environment (Binford et al. 1975, Hamer and Nelson 1995, Nelson 1997, McShane et al. 2004, Ralph et al. 1995b). Because murrelets do not construct nests, they depend on the availability of platforms, typically tree limbs with a moss or other thick substrate, such as piles of needles collected on limbs near a tree bole, sufficiently large for laying their single egg and raising a nestling (Nelson 1997, Ralph et al. 1995).

Table 5-1—Known inland limits of marbled murrelet nests and detections

	Inland distance		
State/province	Nest[a]	Occupied site	Sources
	- - - *Miles* - - -		
Alaska	33		Nelson et al. 2010, Whitworth et al. 2000
British Columbia	39	41	Jones et al. 2006, Lougheed 1999, Nelson et al. 2010, Ryder et al. 2012
Washington	55	55	D. Lynch, personal communication[b]; Ritchie and Rodrick 2002
Oregon	32	47	Alegria et al. 2002; Dillingham et al. 1995; E. Gaynor, personal communication[c]; Witt 1998a, 1998b
California	24	24	S. Chinnici, personal communication[d]; A. Transou, personal communication[e]

Note: see table on page 338 for metric equivalents.

[a] Includes grounded fledglings and eggshell fragments.

[b] D. Lynch. Personal communication. Fish and wildlife biologist, U.S. Department of the Interior, Fish and Wildlife Service, 510 Desmond Dr., Suite 102, Lacey, WA 98503.

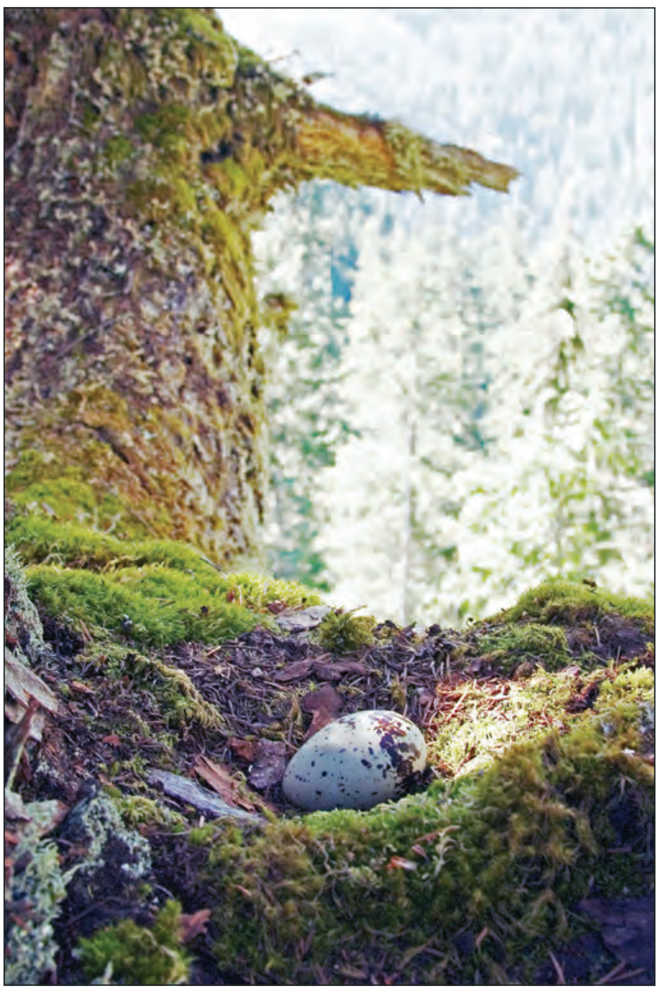
[c] E. Gainer. Personal communication. Wildlife biologist, U.S. Department of the Interior, Bureau of Land Management, 777 NW Garden Valley Blvd., Roseburg, OR 97471.

[d] S. Chinnici. Personal communication. Forest science manager, P.O. Box 712, Humboldt Redwood Company, Scotia, CA 95565.

[e] A. Transou. Personal communication. Environmental scientist, California Department of Parks and Recreation, North Coast Redwoods District, P.O. Box 2006, Eureka, CA 95502; 707-445-6547; atransou@parks.ca.gov.

Individual tree attributes that provide conditions suitable for nesting (i.e., provide a nesting platform) include large branches (ranging from 4 to 32 inches (10 to 81 cm) diameter, with an average of 13 inches (33 cm) in Washington, Oregon, and California) or forked branches; deformities (e.g., broken tops); dwarf mistletoe infections; witches' brooms; and growth of moss or other structures large enough to provide a platform for a nesting adult murrelet (Hamer and Cummins 1991; Hamer and Nelson 1995; Singer et al. 1991, 1992).

These nesting platforms (fig. 5-3) are generally located ≥33 ft (10 m) above ground (reviewed in Burger 2002 and McShane et al. 2004). These structures are typically found in old-growth and mature forests, but may be found in a variety of forest types, including younger forests containing remnant large trees. Since 1996, research has confirmed that the presence of platforms is considered the most important characteristic of murrelet nesting habitat (Burger 2002, Huff et al. 2006, McShane et al. 2004). Platform presence is more important than the size of the nest tree because tree size alone may not be a good indicator of the presence and abundance of platforms (Evans Mack et al. 2003). Tree diameter and height can be positively correlated with the size and abundance of platforms, but the relationship may change depending on the variety of tree species and forest types that murrelets use for nesting (Burger et al. 2010, Huff et al. 2006, Raphael et al. 2011). Overall, nest trees in Washington, Oregon, and northern California have been greater than 19 inches (48 cm) diameter at breast height (d.b.h.) and greater than 98 ft (30 m) tall (Hamer and Meekins 1999, Hamer and Nelson 1995, Nelson and Wilson 2002). Northwestern forests and trees typically require 200 to 250 years to attain the attributes necessary to support murrelet nesting, although characteristics of nesting habitat sometimes develop in younger western hemlock (*Tsuga heterophylla*) forests with dwarf mistletoe.



Nick Hatch

Figure 5-3—Nesting platforms usually include large branches and other structures large enough to provide a platform for a nesting adult murrelet.

Marbled murrelets are reported to nest disproportionately on lower slopes and near streams. The recovery plan for the murrelet (USFWS 1997) states, "With respect to slope, eighty percent of nests in the Pacific Northwest were located on the lower one-third or middle one-third of the slope." Hamer and Nelson (1995) showed the mean distance to streams from murrelet nests in the Pacific Northwest to be 159 m (509 ft). In southern California, Baker et al. (2006) found that murrelet nest sites were located closer to streams, and were located lower on slopes than random sites, based on analysis of variance models. Baker et al. (2006) found that nest sites were much closer to streams than would be expected based on randomly available sites within old-growth forests. Nest sites may have been located near streams because these sites afforded murrelets better access from at-sea flyways.

Other studies have also found proximity to streams or other openings to be important for murrelet nesting in other regions as well (Hamer and Nelson 1995, Meyer et al. 2004, Zharikov et al. 2006). In British Columbia, Rodway and Regehr (2002) found that forests bordering major stream channels provided high-quality nest habitat for murrelets, with large trees, high epiphyte cover, and many potential nest platforms.

Murrelets travel up to 55 mi (89 km) inland to reach suitable habitat in the northern part of their range in the Pacific Northwest; inland distances narrow in the southern portions of the range (table 5-1). Because murrelets depend on marine conditions for foraging and resting, and on forests for nesting, both marine and forest conditions could limit murrelet numbers. Population declines attributed to loss of mature and old-growth forest from harvesting, low recruitment of young, and mortality at sea, led this species to be federally listed as threatened in Washington, Oregon, and California in 1992 (USFWS 1997), and listed as threatened in British Columbia (Rodway 1990). The murrelet's association with late-successional and old-growth forests and its listed status made conservation of the murrelet an explicit goal in the design of the NWFP.

The NWFP included several elements of protection for murrelet nesting habitat. The Plan's system of reserves was not designed, as it was for the northern spotted owl, with specific goals for the number and spacing of clusters of murrelets. Rather, the system of congressionally reserved lands and late-successional reserves was designed to encompass a high proportion of murrelet nesting habitat thought to exist on federal lands. In addition to the reserve system, the NWFP requires murrelet surveys to be conducted before harvest on any other federal lands in the murrelet's range. If a survey shows likely nesting, then all contiguous existing and recruitment habitat (defined as stands that could become nesting habitat within 25 years) within a 0.5-mi (0.8 km) radius is protected. These occupied sites become small reserves, denoted as LSR3, and are managed to retain and restore nesting habitat.

Guiding Questions

The mission statement for the Forest Ecosystem Management Assessment Team (FEMAT) directed the team to take an ecosystem approach to forest management and particularly to address maintaining and restoring biodiversity on federal forests within the range of the northern spotted owl. In addressing biological diversity, the team was directed to develop alternatives that met, among other things, the objective of maintaining or restoring habitat conditions for the murrelet that would provide for viability of the species (FEMAT 1993: iv). Now, 22 years after the NWFP was initiated, national forests in the Plan area are preparing to revise their forest plans. Accordingly, U.S. Forest Service managers have asked how the NWFP has been functioning to support the murrelet and what new science is relevant to murrelet conservation and management. Managers were polled to develop questions relating to the murrelet (as well as other NWFP issues), and this chapter aims to synthesize relevant science related to these questions:

- Are murrelets maintaining viable populations under current NWFP management?
- Is forest management under the NWFP providing nesting habitat for murrelets as planned?
- What is the latest science surrounding the effects of various treatments (silvicultural and fuels) and wildfire on late-successional, old-growth forests and plantations, and what are the effects on murrelets?
- Does the murrelet use these treated forests after harvest? If so, how? Are there ways to modify harvest to benefit murrelets?
- How do these treated habitats compare to untreated habitat in terms of habitat use and reproductive success?
- How have at-sea conditions affected nearby forest use by the murrelet?

To address these questions, we conducted a thorough literature review, guided by keywords included in the questions, and we emphasized references pertaining to murrelets in the Plan area. We excluded gray literature and other unpublished work. We considered additional literature suggested by public comments. As will be apparent in the text, we found little literature bearing on questions 3, 4, and 5, as they pertain to responses of murrelets to silviculture. We direct readers to Spies et al. (this volume) for a summary of how younger forests respond to silvicultural treatments that might influence murrelet nesting habitat.

Key Findings

NWFP Expectations

The stated objective of the NWFP is to maintain and restore nesting habitat conditions that would provide for viability of murrelet populations, well-distributed along their current range on federal lands (FEMAT 1993: iv). The expectation was that the Plan "…would eventually provide substantially more suitable nesting habitat for murrelets than currently (in 1994) exists on federal lands" (USDA and USDI 1994a). FEMAT used an expert panel to assess the likelihood that nesting habitat on federal lands would support stationary and well-distributed populations of the murrelets. Following the methods described in FEMAT (1993), the murrelet expert panel assigned an 80 percent likelihood that nesting habitat would be of sufficient quality, distribution, and abundance to allow the murrelet population to stabilize, well distributed across federal lands over the next 100 years (Outcome A) under Option 9, the preferred alternative that was eventually adopted (with modifications) as the NWFP. The panel assigned a 20 percent likelihood for Outcome B, under which nesting habitat would be sufficient to allow the murrelet population to stabilize but with significant gaps in the historical distribution that could cause some limitation in interactions among local populations. The panel assigned no likelihood of Outcomes C or D. Thus, the panel's assessment was that the likelihood was high that nesting habitat conditions on federal lands would allow the murrelet population to stabilize and be well distributed throughout its range (FEMAT 1993). In recognition of the major influence of marine conditions on population viability, however, including mortality from oil spills and gill netting, and considering the potentially important role of nonfederal lands, the murrelet panel assigned a second set of ratings that considered the cumulative effects of all major factors. The murrelet panel concluded that the likelihood that the murrelet population on federal lands would be stationary and well-distributed was between 50 and 75 percent. The higher rating was meant to indicate the degree of protection conferred by nesting habitat conditions on federal lands, assuming that all other factors were not limiting; the lower rating from the cumulative effects analysis was an attempt to indicate the greater uncertainty in murrelet persistence, given the importance of other factors beyond federal nesting habitat.

Neither the assessment team nor final supplemental environmental impact statement nor subsequent monitoring plan for the murrelet (Madsen et al. 1999) provided quantitative descriptions of expected murrelet population trends or nesting habitat trends over time that now could be used to assess NWFP performance since its implementation. There are, however, some more qualitative descriptions or assumptions from the period around the start of the assessment team and the record of decision:

- The amount of murrelet nesting habitat had declined over the previous 50 years, primarily because of timber harvesting (Perry 1995, USFWS 1997).
- Murrelet populations are likely to have declined as well, largely in response to loss of nesting habitat (Ralph et al. 1995a).
- Demographic projection models estimated at the time the NWFP was initiated suggested a population decline of 4 to 7 percent per year from 1990 to 1995 (Beissinger 1995).
- Because murrelets have naturally low reproductive rates, population recovery will be slow, on the order of a maximum of 3 percent per year (USFWS 1997).
- No destruction of nesting habitat surrounding active murrelet nesting sites will be knowingly done on federal lands.
- Catastrophic and stochastic events that decrease the quality or quantity of nesting habitat would affect nesting habitat at unknown rates.
- Over the long term, the amount of nesting habitat will increase in reserves as unsuitable forest matures.
- Late-successional reserves will provide large contiguous blocks of nesting habitat with increased interior (180 ft [55 m] or more from edge) nesting habitat.

- Rates of nest depredation would decrease as the amount of interior nesting habitat increases in reserves.
- In the short term (less than 50 years), the availability of nesting habitat may remain stable or decline from losses from fire and other natural disturbances.
- The rate of increase in the amount of nesting habitat will be slow because trees do not develop structures suitable to support nests until they are large and old, often 150 or more years (USDA and USDI 1994a; USFWS 1997).
- Nesting habitat management on nonfederal lands will affect viability of murrelets on federal lands.
- Physical and biological processes in the marine environment, which operate at multiple temporal and spatial scales, also affect short- and long-term population trends of murrelets, independent of nesting habitat quantity or quality.

McShane et al. (2004) developed a population model to predict population change in each of five conservation zones comprising the Plan area (fig. 5-4). Their model, which used annual adult survival estimates obtained from detailed mark-recapture studies in British Columbia (the only such data then available) and fecundity estimates from ratios of juveniles to adults at sea or from mark-recapture studies, predicted annual rates of decline varying from 3 to 5 percent per year over the first 20 years of their simulations in murrelet conservation zones 1 through 5.[2] Rates of decline were generally greater going from north (zones 1 and 2) to south (zone 5). These predictions are in line with those of Beissinger (1995), using models based mostly on comparative demographic data from other alcid species. These models do not directly account for the amount of nesting habitat, thus model projections do not respond to expected habitat trends.

NWFP Monitoring Results for Marbled Murrelets

Population size and trends—
A specific conservation goal of the plan is to stabilize and increase murrelet populations by maintaining and increasing nesting habitat. As described below, population monitoring results to date indicate that the plan goal of stabilizing and increasing murrelet populations has not yet been achieved throughout the Plan area, because while in some areas the population may have stabilized, they have not increased substantially. Murrelet populations were thought to be declining at the start of the Plan, with loss of more than 80 percent of nesting habitat being the central cause for declines and for murrelets being listed as federally threatened (USFWS 1997). Declines were expected to continue for a period (e.g., Raphael 2006), until nesting habitat sufficiently recovers from previous losses to lead to increased fecundity, and populations stabilize and increase (USFWS 1997). The Plan goal of increasing populations recognizes the large historical population declines (Peery et al. 2010, USFWS 1997), and the conservation value of larger populations than were present in 1994.

To evaluate murrelet population status and trends under the Plan, the murrelet effectiveness monitoring program designed a coordinated sampling protocol (Madsen et al. 1999, Raphael et al. 2007) and obtained annual population estimates starting in 2000 by monitoring murrelet populations in nearshore marine waters associated with the Plan area, in Washington, Oregon, and northern California (fig. 5-4). The population monitoring uses boat-based transects and distance estimation methods in those coastal waters, which are divided into five geographic subareas corresponding to conservation zones established in the U.S. Fish and Wildlife Service's recovery plan for the murrelet (fig. 5-4). The monitoring program estimated population size and trend for each conservation zone, for each state, and for all zones combined. Through 2013, the entire Plan area was surveyed annually; starting in 2014 a reduced-sampling design was instituted because of funding constraints, in which conservation zones 1 through 4 are sampled every other year, and zone 5 every fourth year. Details about the sampling and data analysis methods used by the population monitoring program are described elsewhere (Falxa et al. 2016, Raphael et al. 2007).

[2] These zones are defined in the marbled murrelet recovery plan (USFWS 1997): Conservation zone 1 is Puget Sound and the Strait of Juan de Fuca in Washington; zone 2 is the outer coast of Washington to the Columbia River; zone 3 is Oregon from the Columbia south to North Bend (Coos Bay); zone 4 is North Bend south to Shelter Cove, California; zone 5 is Shelter Cove south to the mouth of San Francisco Bay (see fig. 5-2). Zone 6, from the mouth of San Francisco Bay south to Point Sur, California, is outside of the Northwest Forest Plan area.

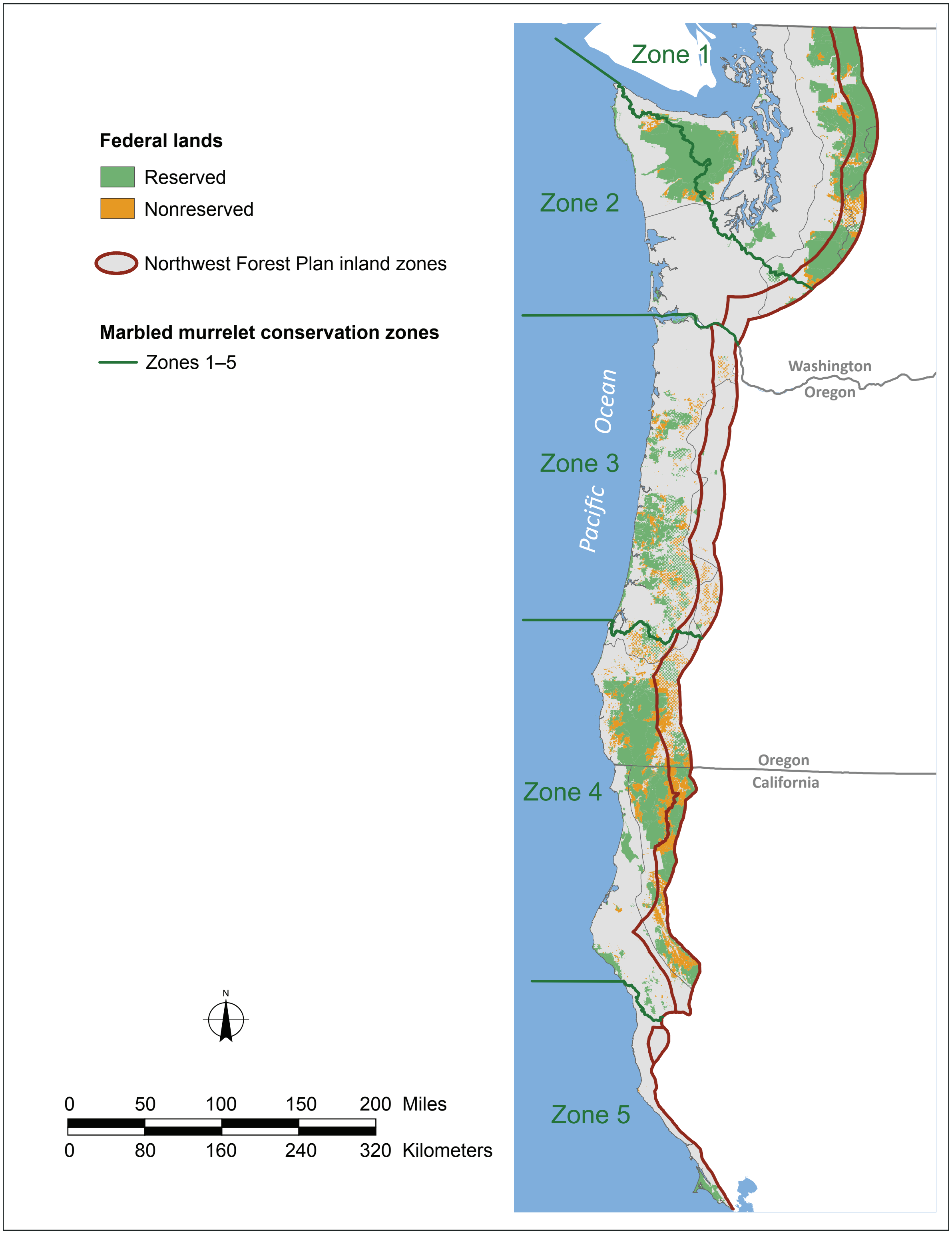


Figure 5-4—Range of the marbled murrelet with boundaries of conservation zones and locations of federal reserves, other federal lands, and nonfederal lands. Also shown are boundaries of the "inland zones" designated by the Northwest Forest Plan; see "Research Needs, Uncertainties, Information Gaps, and Limitations" for a description of these zones.

The 20-year murrelet status and trends report provided estimates through 2013 (Falxa et al. 2016); population monitoring results from 2014 and 2015 have since become available in annual reports (Falxa et al. 2015, Lynch et al. 2016). At the conservation-zone scale, the most recent population estimate shows few murrelets remaining in conservation zone 5 (San Francisco Bay north to Shelter Cove, California; estimate: 71 murrelets, 95 percent confidence interval: 5 to 118) (Lynch et al. 2016); this is consistent with estimates since 2000. Considerably more murrelets remain in the other four conservation zones within the NWFP area, with murrelet numbers, expressed as an average of annual estimates over the the past 4 years with sampling (Lynch et al. 2016) as follows: about 7,600 murrelets in conservation zone 1 (the Strait of Juan de Fuca, San Juan Islands, and Puget Sound in Washington; for 2012–2015); about 2,000 birds in conservation zone 2 (the outer coast of Washington; 2012–2015); about 7,600 murrelets in conservation zone 3 (from Coos Bay north to the Columbia River, Oregon; 2011–2014); and about 6,600 birds in conservation zone 4 (from Shelter Cove, California, north to Coos Bay, Oregon; 2012–2015). The use of averages accounts for some of the annual variation in population estimates. Single-year estimates vary among years and tend to have relatively large confidence intervals. For example, the most recent estimate for conservation zone 2 (3,204 murrelets in 2015) is higher than the 4-year average, but with a 95 percent confidence interval (1,883 to 5,609) (Lynch et al. 2016) that includes that average. All annual estimates at the conservation zone and other scales are found in recent reports from the NWFP's murrelet effectiveness monitoring program (Falxa et al. 2016, Lynch et al. 2016).

Estimated density of murrelets on the surveyed waters (generally within 2 to 3 mi [3 to 5 km] of shore, depending on conservation zone) (Raphael et al. 2007) ranged from approximately 0.1 murrelets per square kilometer in conservation zone 5 to 7.5 murrelets per square kilometer in conservation zone 4 in 2015. Annual population estimates for the entire Plan area ranged from about 16,600 to 22,800 murrelets during the 15-year period (fig. 5-5), and averaged about 21,000 birds over the past 4 years (2011–2014); the most recent estimate for the Plan area is 21,300 birds for 2014 (95 percent confidence interval: 17,500 to 25,100) (Lynch et al. 2016). The confidence intervals associated with population estimates reflect the difficulties in sampling such a mobile, patchily distributed, and relatively rare species over a large area of ocean waters. Although this sampling error decreases the power to detect population trends, the trend estimation accounts for sampling error.

The estimates from population monitoring form the basis for evaluating population trends since 2000. The monitoring program evaluated linear trends from 2000 to 2015 at multiple scales (Lynch et al. 2016), and found evidence for a declining trend in Washington, no clear trend in Oregon, and evidence for an increasing trend in the California portion of the Plan area (fig. 5-6). In Washington (fig. 5-7), there was strong evidence of a population decline in conservation zone 1 (a 5.3 percent annual decline, 95 percent confidence interval: -8.4 to -2.0) (Lynch et al. 2016), and a 4.4 percent decline per year for Washington state (conservation zones 1 and 2 combined; 95 percent confidence interval: -6.8 to -1.9) (Lynch et al. 2016). In conservation zone 2, where past analyses found a declining trend (Falxa et al. 2016), the most recent trend analysis, with 2014 and 2015 data included, indicates that a negative trend may continue in conservation zone 2, but the upper confidence interval now overlaps zero (fig. 5-7), thus the trend for this zone is uncertain (95 percent confidence interval: -7.6 to 2.3) (Lynch et al. 2016). In conservation zones 3 and 5, the most recent data provide no evidence of a trend (confidence intervals broadly overlap zero) (Falxa et al. 2016, Lynch et al. 2016); for an earlier period, Strong (2003) described a decline for central Oregon, which includes part of zone 3. In zone 4, the trend estimate was positive (3.0 percent per year), and with the addition of 2015 survey data the trend estimate's 95 percent confidence interval does not include zero (0.4 to 5.6; fig. 5-7), evidence for a positive trend on average for the 2000 to 2015 period for this zone (Lynch et al. 2016). At the state scale for Oregon and California, which combines conservation zones and portions of conservation zones, there was no evidence of a trend in Oregon (fig. 5-6). For California, as for zone 4, the trend estimate was positive for 2000 to 2015 (3.8 percent per year) and the 95 percent confidence interval for that estimate (0.9 to 6.8) lies entirely above zero, suggesting an increasing population (fig. 5-6).

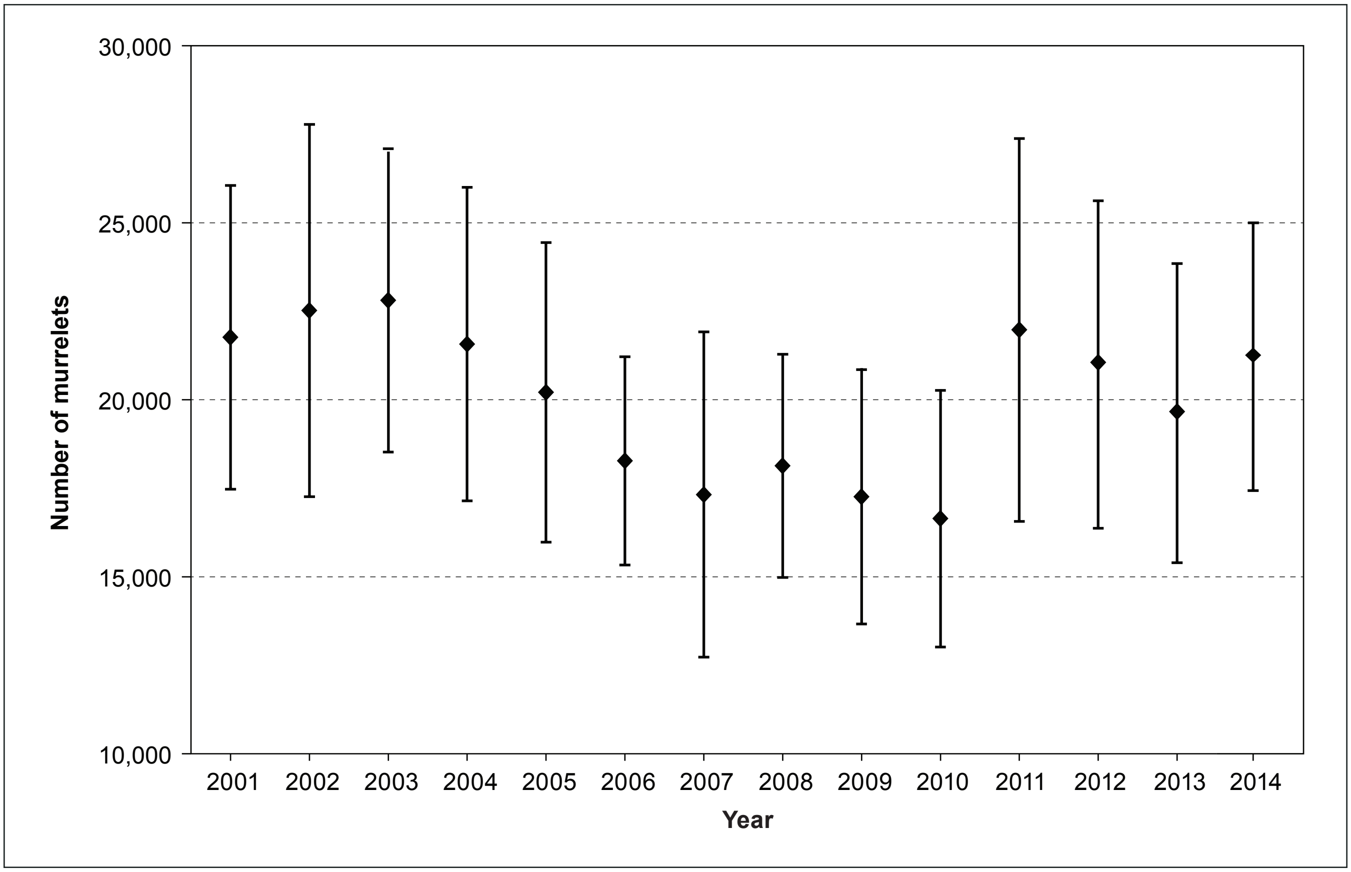


Figure 5-5—Annual marbled murrelet population estimates and 95 percent confidence intervals for the Northwest Forest Plan area (conservation zones 1 through 5 combined) based on 2000–2014 data (Falxa et al. 2016, Lynch et al. 2016).

For the entire Plan area, the estimated rate of population change for the 2001 to 2014 period was negative (-0.7 percent per year), but the confidence interval for the estimate (-2.3 to 0.8) broadly overlapped zero and there was no clear evidence for a trend (fig. 5-7). Additional years of monitoring should increase the power to detect an ongoing trend, such as where the trend is slight and power to detect low, but population trajectories can also change with time, which adds variability and difficulty in describing trends. For example, the magnitude and strength of evidence for a NWFP-wide population decline have decreased relative to a previous assessment for the 2001 to 2010 period (Miller et al. 2012). This difference may be driven by a variety of factors, most notable being the higher population estimates for 2011 through 2014 compared to the previous several years (fig. 5-5), which reduced the slope of the trend and increased variability (Falxa et al. 2016, Lynch et al. 2016). In 2011 and 2012, estimates of murrelet population size increased in all conservation zones except conservation zone 2, compared to estimates from previous years. Falxa et al. (2016) discuss and evaluate potential causes for the pattern observed, which include (1) change in the distribution of murrelets relative to shore that affects the proportion of the population sampled, (2) change in the model parameters used to estimate density, (3) shift of murrelets from nonsampled units to sampled units in conservation zone 1, (4) movement of birds into conservation zone 1 from the north or south during 2011 to 2013, and (5) potential effects of atypical timing of breeding or proportion of the population nesting. The cause(s) remain unknown, and continued monitoring and research should help managers better understand population trends and assess underlying factors that might explain trends and variability in annual estimates.

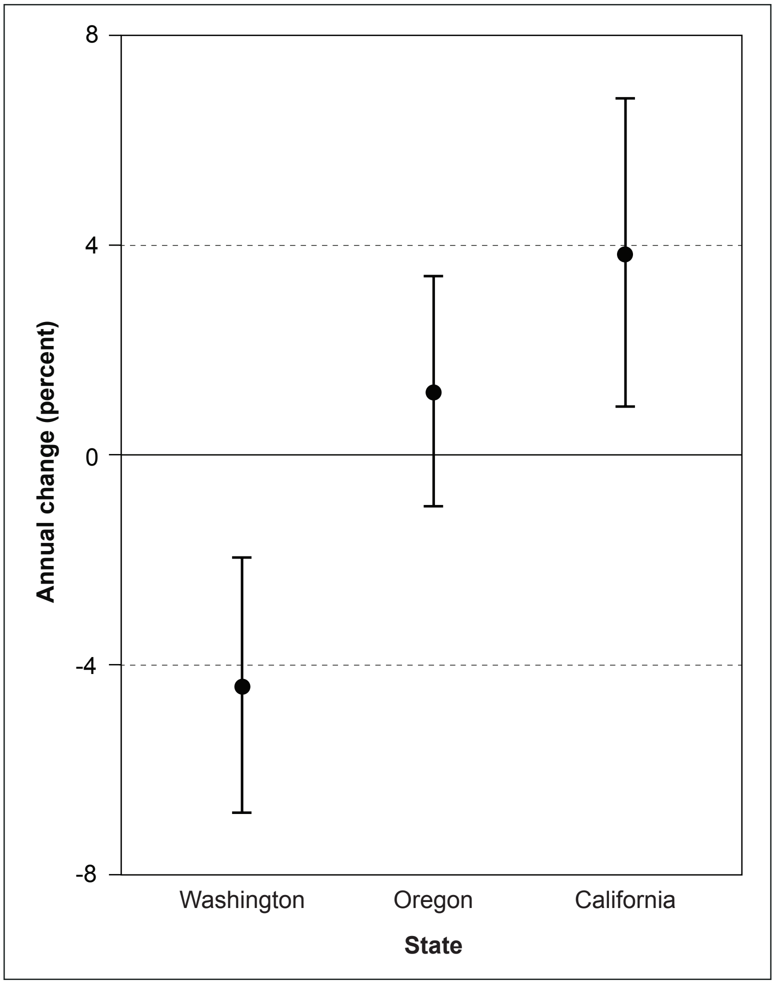


Figure 5-6—Trend results: average rate of annual change by state, 2000 to 2013, with 95 percent confidence intervals. Washington trend is based on 2001–2015 data, Oregon on 2000–2014 data, and California on 2000–2015 data (Falxa et al. 2016, Lynch et al. 2016).

The population monitoring results to date indicate that, as expected, the NWFP goal of stabilizing and increasing murrelet populations has not yet been achieved throughout the Plan area. Although the population monitoring data for 2000 through 2015 are not consistent with declining populations in Oregon and California during this period, murrelets are declining in Washington. The Washington trend results are consistent with demographic models for the murrelet (McShane et al. 2004, USFWS 1997), which predicted declining populations based on the available data on rates of murrelet survival and reproductive output. The population monitoring data suggest a north-to-south trend pattern, in which population trends appear to improve from north to south within the Plan area based on the last 15 years. The observed Oregon and California trend results are not consistent with model predictions. However, major sources of uncertainty include (1) uncertainty in estimating survivorship and fecundity (reproductive output) in the demographic models, (2) uncertainty about whether the murrelet populations being monitored are closed or open to immigration, and (3) the relatively large confidence intervals around population estimates. Murrelets occur immediately to the north of the Plan area, and monitored populations may be subsidized by immigrants from British Columbia or Alaska, where birds are more abundant (Falxa and Raphael 2016, Raphael 2006). Peery et al. (2007) found that immigration of murrelets from north of the zone 6 (Santa Cruz Mountains) population may have been sufficient to mask an intrinsic decline in the zone 6 population; this could occur elsewhere.

Status and trend of nesting habitat—
Whereas the focus of the murrelet effectiveness monitoring program is on the status and trends of murrelet populations and nesting habitat on federal lands within the Plan area, the populations monitored at sea respond to nesting habitat conditions on both federal and nonfederal lands. To better understand the murrelet's conservation status, and the relationship between population conditions and nesting habitat conditions, monitoring considered nesting habitat conditions across both federal and nonfederal lands (Raphael et al. 2016a). Also, in some areas, such as southwest Washington and northwest California, few federal lands occur within the murrelet's nesting range, and thus nonfederal lands are likely important to murrelet conservation.

***Baseline nesting habitat*—**When the NWFP was developed, no consistent map of murrelet nesting habitat was available. For purposes of the Plan, murrelet nesting habitat was then assumed to be late-successional forest with much

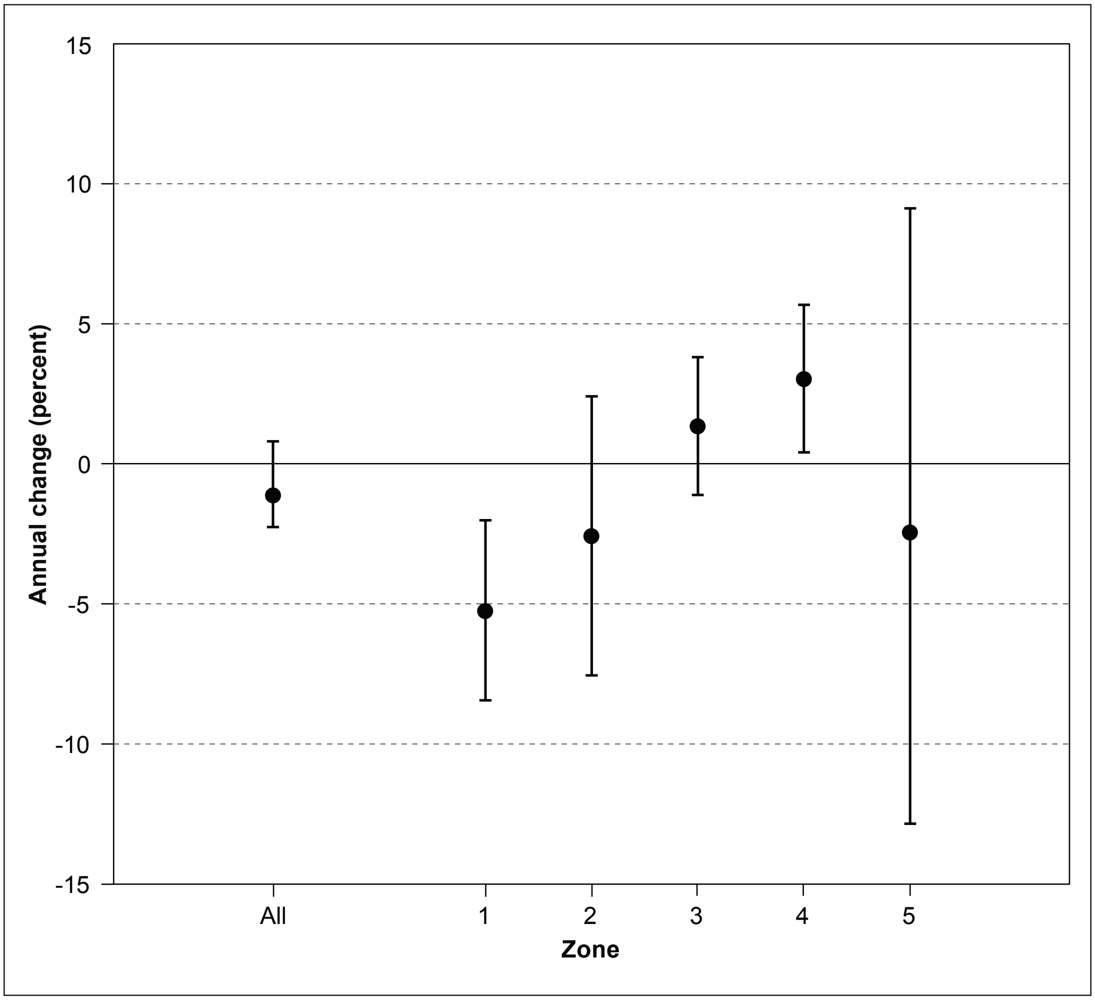


Figure 5-7—Trend results: average rate of annual change by conservation zone (see fig. 5-2 for zone locations) and for all conservation zones combined, with 95 percent confidence intervals. All zones based on 2001–2014 data, zones 1 and 2 on 2001–2015 data, zone 3 on 2000–2014 data, zone 4 on 2000–2015 data, and zone 5 on 2000–2013 data (Falxa et al. 2016, Lynch et al. 2016).

the same characteristics as northern spotted owl habitat. Therefore, the existing map of spotted owl habitat, which was itself a mosaic derived from compilations of local maps based on agency judgment, classified satellite imagery, and existing inventory maps, was constrained to the range of the murrelet and used as a proxy for murrelet nesting habitat. No estimate or map of nesting habitat on nonfederal land was available. The murrelet effectiveness monitoring group has since developed a series of maps, using a consistent vegetation base across all ownerships throughout the Plan area (Raphael et al. 2016a); the maps were based first on vegetation data from CALVEG and the Interagency Vegetation Mapping Project (Moeur et al. 2005), and then later based on Gradient Nearest Neighbor (GNN) vegetation data (Davis et al. 2015, Ohmann and Gregory 2002, Moeur et al. 2011).

The primary objectives of the effectiveness monitoring plan for the murrelet included mapping baseline nesting habitat (at the start of the NWFP in 1993) and estimating changes in that forest over time. For the NWFP 20-year analysis and report, Raphael et al. (2016a) used maximum entropy (Maxent) models to estimate nesting habitat suitability over all habitat-capable lands in the murrelet's range in Washington, Oregon, and California. "Habitat-capable" lands were defined as lands capable of supporting or developing into murrelet nesting habitat (fig. 5-8). The area of habitat-capable lands evaluated by the 20-year analysis included about 20.7 million ac (8.5 million ha) of federal plus nonfederal lands within the murrelet range portion of the Plan area (Raphael et al. 2016a).[3]

The portion of the murrelet range included in this analysis excluded inland zone 2 of Oregon and California, where no murrelet nests have been observed (see Raphael et al. 2016a for details). The models used vegetation and climate attributes, and a sample of 368 murrelet nest sites (184 confirmed murrelet nest sites and 184 occupied sites) for model training. Occupied sites are sites where murrelet behaviors associated with nesting have been observed during carefully prescribed surveys (Evans Mack et al. 2003); such sites do not have confirmed nests but are places deemed likely to have nests. Attributes used to build the model included estimates of canopy cover, mean tree diameter, diameter diversity, canopy layers, number of nesting platforms, stand age and stand height, an index of old-growth structure, percentage of a 124-ac (50-ha) area composed of older forest, and several climate variables. All of these attributes were derived from a regional vegetation database and a climate database that covered the entire Plan area as described in Raphael et al. (2016a). The model classified each 30-m pixel in the Plan area with a nesting habitat suitability score ranging from 0 (unsuitable) to 1.0 (most suitable); higher scores indicate that a pixel has vegetation and climate characteristics more similar to those in the sample of murrelet nest sites, compared to a random sample of available forest. Model validation was accomplished by withholding 25 percent of the training data, testing the model on the withheld data, and replicating the process 25 times.

Thresholds were defined that summarized land area into four classes of nesting habitat suitability; classes 1 and 2 were deemed lower suitability, and classes 3 and 4 were deemed higher suitability (see Raphael et al. [2016a] for a detailed explanation of these suitability classes and the cutoff values used to define them). The model was run 25 times for each state and then summarized to provide an estimate of model error, owing to variation in model runs themselves and variation in underlying GNN data. Raphael et al. (2016a) estimated that there were 2.53 million ac (1.02 million ha) of higher suitability nesting habitat over all lands in the murrelet's range in Washington, Oregon, and California at the start of the NWFP; this included 1.50 million ac (0.61 million ha) on federal lands. Of the 2.53 million ac of higher suitability nesting habitat, 0.46 million ac (0.18 million ha) were identified as highest suitability (class 4), matching or exceeding the average conditions for the training sites; of this, 0.25 (0.10 million ha) million ac were on federal lands. A substantial amount (41 percent) of baseline nesting habitat occurred on nonfederal land (fig. 5-9). The estimate of nesting habitat on federal land from the 1993 final supplemental environmental impact statement was 2.6 million ac. Differences between the 1993 and current nesting habitat estimates were to be expected, as the new map was derived from a nesting habitat suitability model specific to the murrelet, and was built from forest- and satellite-derived data that had not been available at the time the NWFP was written. As noted earlier, the final 1993 supplemental environmental impact statement used habitat for the northern spotted owl as a proxy for murrelet nesting habitat.

Although a substantial amount of higher suitability nesting habitat occurred on nonfederal lands, federal lands contributed proportionately more suitable nesting habitat. Of the about 20.7 million ac (8.4 million ha) of forest land capable of supporting or developing into murrelet nesting habitat, federal lands comprise only about 28 percent of the area, but provided 59 percent of the suitable nesting habitat at the start of the NWFP (Raphael et al. 2016a). The contribution of suitable nesting habitat from nonfederal land varies: in Washington, 42 percent; in Oregon, 33 percent; and in California, 80 percent (fig. 5-9). On the 1.0 million ac (0.4 million ha) of suitable nesting habitat on nonfederal lands in 1993, about 39 percent was managed by states. In Washington, the proportion of the nesting habitat on federal lands that is within reserves is 93 percent; in Oregon, 88 percent; and in California, 93 percent. The final supplemental environmental impact statement estimated that 86 percent of murrelet nesting habitat on federal lands

[3] Does not include conservation zone 6, which is south of San Francisco and outside of the NWFP area.

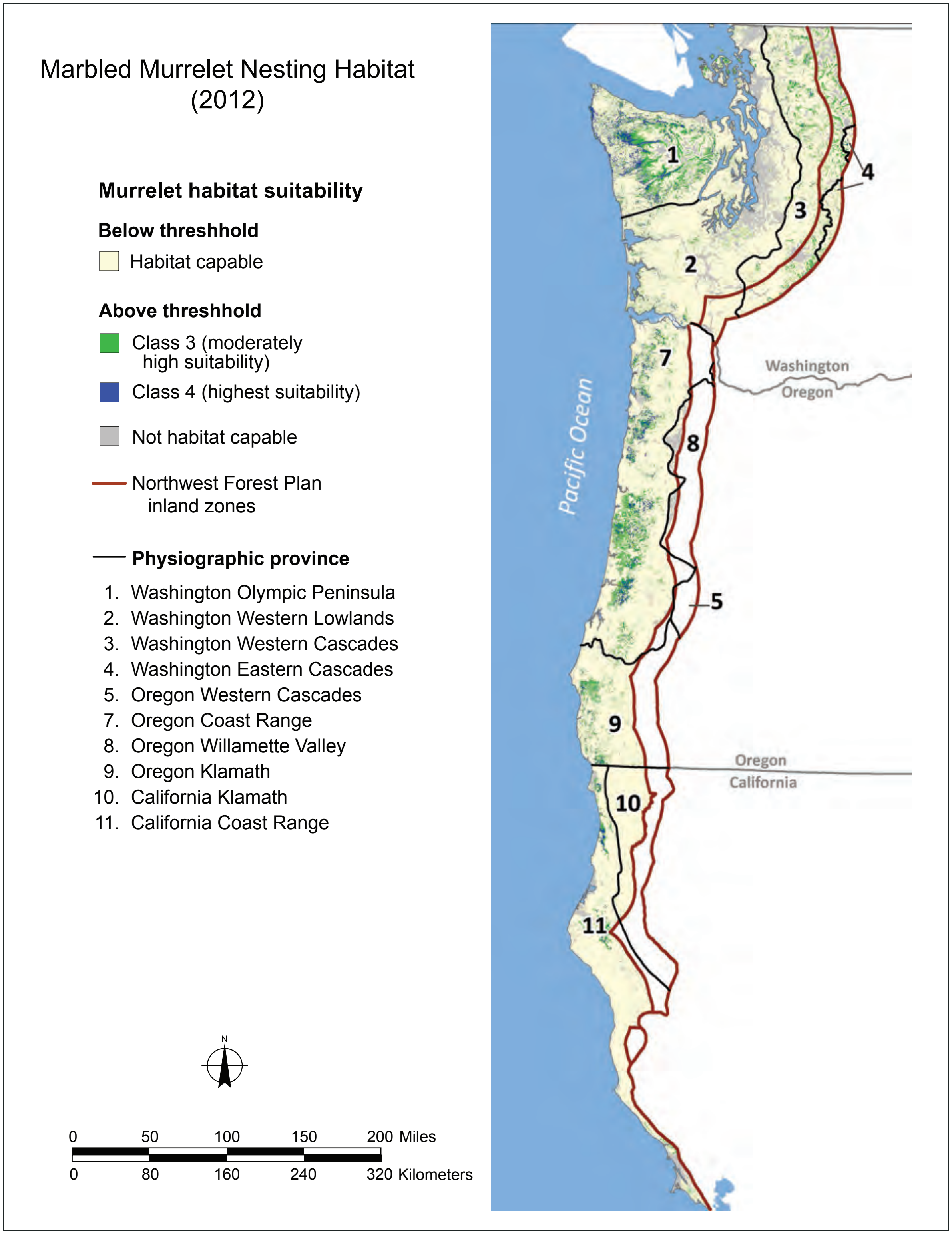


Figure 5-8—Map of suitability for marbled murrelet nesting habitat, 2012 (Raphael et al. 2016a).

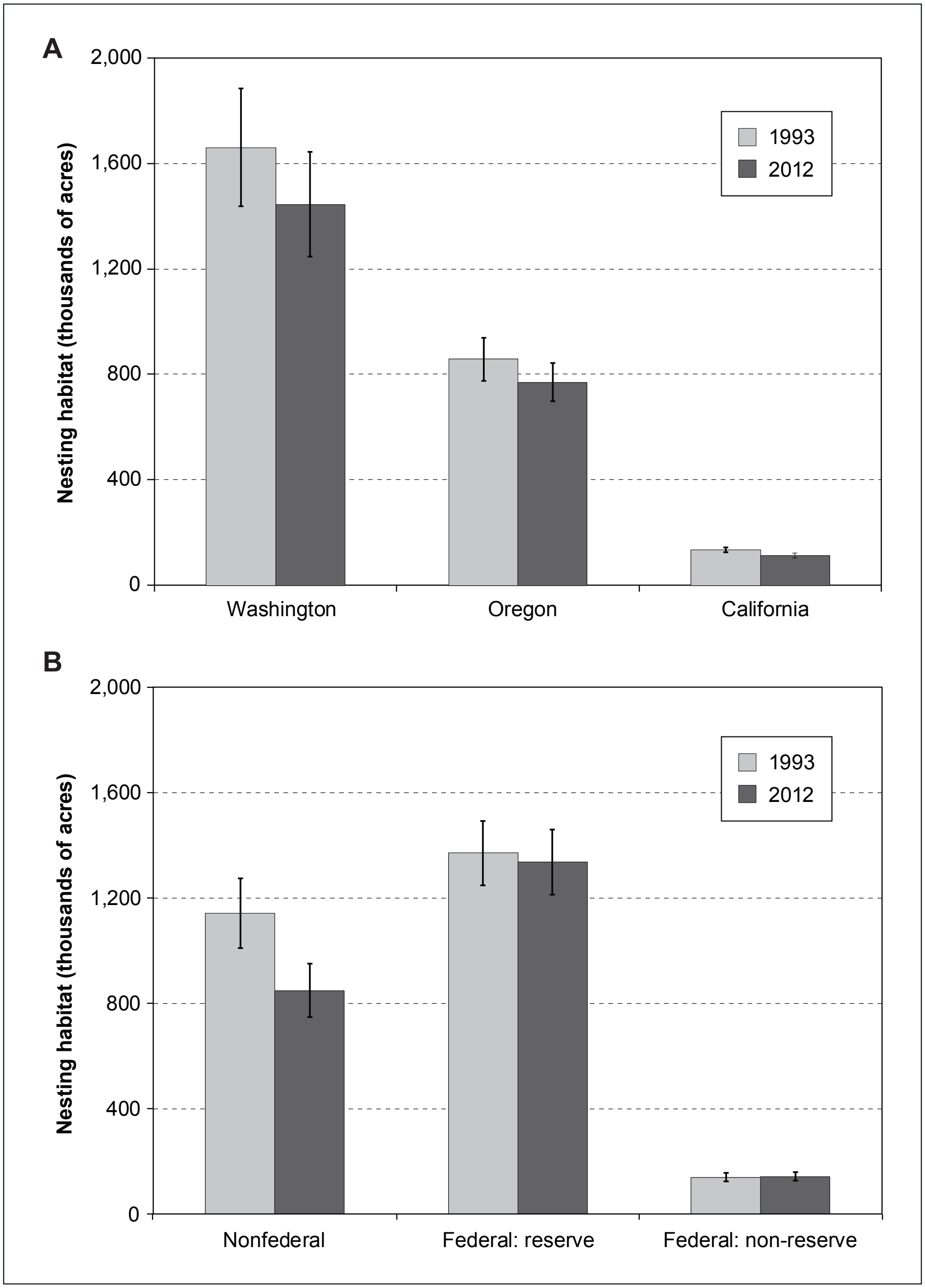


Figure 5-9—Estimated amounts of higher suitability nesting habitat of the marbled murrelet in 1993 and 2012, by (A) state, and (B) land allocation (Raphael et al. 2016a). Error bars are 95-percent confidence intervals from 25 replicated model runs. See table on page 338 for metric equivelents.

would be in reserves. The 20-year analysis found that, in 1993, 90 percent of potential nesting habitat on federally administered lands occurred within reserved-land allocations (Raphael et al. 2016a). Thus, the NWFP seems to have successfully captured most of the existing higher suitability nesting habitat on federal lands within its reserve system. We conclude that the NWFP had successfully encompassed a majority of murrelet nesting habitat within its reserve system but that a substantial amount of additional suitable nesting habitat occurs on nonfederal lands over which the NWFP has little or no control.

***Nesting habitat losses*—**The intent of the NWFP is to conserve most of the remaining murrelet nesting habitat and to prevent the subsequent loss of any nesting habitat occupied by nesting birds, wherever that nesting habitat occurred on federal lands. The amount of nesting habitat was expected to increase over time, but the rate of increase would be very slow, and changes might not be observed for many decades. In the meantime, some unoccupied nesting habitat would be lost to timber harvest on federal land, and some losses might be caused by wildfire and other disturbances.

The observed trends are in line with these expectations. Raphael et al. (2016a) used satellite imagery and change detection methods (see Davis et al. 2015) to estimate a net loss of 307,957 ac (124,692 ha) of higher suitability nesting habitat over all lands (including nonfederal) from 1993 to 2012, or a total loss of about 12 percent. Net loss was about 27 percent from the baseline on nonfederal lands, and 2.2 percent on federal lands (table 5-2). Of those losses on nonfederal lands, the highest rate of loss was on private lands (37 percent); losses on state lands were just under 10 percent (table 5-2). Of those losses on federal lands, 62 percent was due to fire (most of that in one event, the 2002 Biscuit Fire); 23 percent to timber harvest; and 16 percent to insects, disease, or other natural disturbances (table 5-3). On nonfederal lands, 98 percent of losses were due to timber harvest, and 2 percent to insects, disease, and other causes (table 5-3).

Table 5-2—Change in acres (thousands) of suitable nesting habitat from 1993 to 2012 by land ownership in the Northwest Forest Plan area (updated from Raphael et al. 2016a)

State	Owner	1993	2012	Change
		- - - - - - *Acres (thousands)*	- - - - - -	*Percent*
Washington	Federal	899.7	887.1	-1.4
	State	243.7	209.7	-29.8
	Other nonfederal	405.6	246.3	-39.3
Oregon	Federal	573.1	553.7	-3.4
	State	123.3	119.6	-3.0
	Other nonfederal	157.0	101.5	-35.4
California	Federal	26.5	26.0	-1.9
	State	32.3	31.9	-1.2
	Other nonfederal	73.7	51.3	-30.4
Plan area total	Federal	1,499.3	1,466.8	-2.2
	State	399.2	361.2	-9.5
	Other nonfederal	636.4	398.8	-37.3

Note: see table on page 338 for metric equivelents.

Table 5-3—Attribution of loss, in thousands of acres, of marbled murrelet higher suitability habitat from the Northwest Forest Plan baseline (1993) to 2012 by land allocation

	Losses[a]			
Land allocation	**Fire**	**Harvest**	**Other**	**Total**
	Acres (thousands)			
Federal reserved	19.1	4.6	5.3	34.8
Federal nonreserved	2.4	3.3	0.2	5.3
Nonfederal	0.6	308.7	6.9	316.3
Total	22.1	316.7	12.4	351.7

[a] Losses as verified by LandTrendR (see Raphael et al. 2016a for details).
Source: Raphael et al. 2016a.

***Nesting habitat increases*—**One NWFP expectation was a gradual increase in the amount of suitable nesting habitat as forests mature. Previous evidence showed that the amount of forest with large (>20 [>51 cm] inches in diameter) trees had increased by about 15 percent over the first 10 years of the NWFP, based on analyses of inventory plots on national forest lands (Moeur et al. 2005). More recent work, however, showed a decrease of about 2.8 percent in the amount of older forest on federal lands and about 6 percent over all lands within the entire NWFP area; the discrepancy may be due to the newer definitions of older forest used in the more recent estimates (Davis et al. 2015); this analysis included large areas outside (inland to the east) of the murrelet nesting range. As noted above, net losses of murrelet nesting habitat totaled about 12 percent over all lands and 2.2 percent on federal lands. At some point in the future, the extent of current young forest within the reserve system on federal land will be such that we could see a net increase in amount of suitable nesting habitat. For example, trends in the Oregon Coast Range on federal lands show that nesting habitat can increase when stand-replacement rates of disturbance are low and forest age classes are available to grow into murrelet nesting habitat in a few decades. Unfortunately, however, we are unaware of any estimates of exactly when that point will be reached. There is a need to develop models to project forest conditions forward in time and to then estimate future nesting habitat suitability. We do know, as pointed out in Raphael et al. (2016a), that there is sufficient young and mature forest within the reserve system (fig. 5-8) to eventually make up for losses since the start of the NWFP, if future nesting habitat losses on federal lands remain similar to the first 20 years of the NWFP, and the NWFP reserve system remains intact and continues to be managed for the development of old-forest conditions. While at broader scales the amount of murrelet nesting habitat declined, some gains in nesting habitat may already be occurring locally, notably on Forest Service lands in the Oregon Coast Range province, where small net gains (about 1 percent) were observed by the 20-year analysis (Raphael et al. 2016a).

Status of Marbled Murrelets Elsewhere in the Species' Range

The NWFP effectiveness monitoring program provides data on murrelet status and trends that is unparalleled elsewhere in geographic and temporal extent. Nonetheless, other monitoring programs exist elsewhere within the species' range (see fig. 5-3 for range map); these provide information on the status and trends for some areas outside of the NWFP area. The most comprehensive of these in geographic scope is conducted by the Canadian government to assess temporal trends of the murrelet in British Columbia. That program recently reported on murrelet population trends from 1996 through 2013, based on a radar-based monitoring program; they found evidence for a coastwide decline of about 1.6 percent per year in British Columbia (Bertram et al. 2015a). Trends varied strongly among the six sampling regions within British Columbia: negative trends were detected in their east Vancouver Island (-9 percent per year) and south mainland coast (-3 percent per year) regions, and a weak negative trend in Haida Gwaii. A separate program has monitored at-sea murrelet numbers from about 62 mi (100 km) of transects on the southwest coast of Vancouver Island during May to July since 1995. Results from this effort suggest an initial decline through 2006, followed by stable or increasing numbers since 2006 (Bertram et al. 2015a; Zharikov et al. in Irvine and Crawford 2012; Y. Zharikov, pers. comm.[4]). The most recent

[4] **Zharikov, Y. 2016.** Personal communication. Monitoring ecologist, Parks Canada, Ucluelet, BC V0R 3A0.

population estimate in British Columbia, using extrapolations from at-sea surveys and radar counts, gives the range as 72,600 to 125,600 birds of all ages (mid-point 99,100 birds) (COSEWIC 2012).

In central California, the small murrelet population of conservation zone 6 (from the mouth of San Francisco Bay to Point Sur in Monterey County) has been monitored with at-sea surveys almost annually since 1999. Those surveys estimated population sizes of about 400 to 600 birds between 2009 and 2014 (Henry and Tyler 2014), with no clear trend during that period, but an apparent decline compared to numbers from 1999 to 2003 (Henry et al. 2012).

Data are more limited on the murrelet's status in Alaska, where its range extends from the southeast corner of the state through the Aleutian Islands. Within that area, monitoring surveys have been conducted annually in Glacier Bay since 2009; murrelet numbers there have been variable, with the highest annual estimates in 2013 and 2015 (Sergeant et al. 2015). Monitoring surveys throughout Prince William Sound in 11 years between 1972 and 2007 suggest that murrelet abundance there declined by an annual average rate of about 4 to 5 percent per year for that period (Kuletz et al. 2011).

Less recent information is available from a 2007 evaluation of the status of the murrelet in Alaska and British Columbia (Piatt et al. 2007). That review evaluated trends for Alaska using at-sea survey data from eight different and widely distributed sample sites. Although the sites differed in methods, sampling effort, and time period sampled, the evaluation found evidence for significant declines at five of eight sites, at annual rates of -5.4 to -12.7 percent since the early 1990s (Piatt et al. 2007). While acknowledging uncertainty resulting from a lack of recent survey data from key areas, they projected the 2007 murrelet population in Alaska to be roughly 270,000 birds, representing a decline of about 70 percent over a 25-year period (Piatt et al. 2007). They concluded that the declines were likely real, and attributed them to combined and cumulative effects from climate-related changes in the marine ecosystem affecting prey resources (including a regime shift in the Gulf of Alaska that reduced the abundance of important murrelet prey), and human activities (logging, gill net bycatch, and oil pollution).

As noted below, Raphael et al. (2016b) reported a correlation between numbers of murrelets counted at sea and amounts of adjacent suitable nesting habitat within the three-state region of the NWFP. This relationship, however, seems to vary considerably in different portions of the murrelet range, as illustrated in table 5-4. Certainly, part of the reason for this variation is due to differences in methods and definitions of nesting habitat, but the magnitude of difference (e.g., 207 ac [84 ha] per bird in Washington versus 15 ac per bird in Alaska) suggests that there are real differences in relationships between offshore numbers of birds and inland nesting habitat in the various regions. We note that there is likely a higher proportion of murrelets in Alaska nesting in small patches of forest, which are likely to be excluded in forest inventories, and on cliffs or on the ground (Barbaree et al. 2014). It is also possible that foraging prey density is much greater in Alaska, supporting a larger number of birds relative to available nesting habitat compared with other parts of the range.

Table 5-4—Estimated amounts of potential nesting habitat (rounded to nearest 100 ac), murrelet population size, and ratio of habitat to population in portions of the murrelet range (as depicted in fig. 5-7)

Region	Nesting habitat	Estimated murrelet population	Habitat area per bird	Source
	Acres		*Acres*	
Southeast Alaska	2,034,700	144,200	15	Piatt et al. 2007
British Columbia	3,439,100	99,100	35	Environment Canada 2014
Washington	1,549,000	7,494	207	Lynch et al. 2016, Raphael et al. 2016a
Oregon	853,400	11,384	75	Lynch et al. 2016, Raphael et al. 2016a
California	132,600	5,666	23	Lynch et al. 2016, Raphael et al. 2016a

Nesting Habitat Relationships

Patches and edges—

Although the behavior and habitat cues used by murrelets to locate nest sites are not known, their nests tend to be widely spaced across the landscape, especially if there is extensive suitable nesting habitat (Nelson 1997). In areas where there is a wide choice of suitable trees, nest trees tend not to be re-used in successive seasons (Burger et al. 2009). Nests located using radiotelemetry in Desolation and Clayoquot Sounds, British Columbia, had mean inter-nest distances of 2.9 ± 2.5 (standard deviation [SD]) mi (4.6 ± 4.0 km) and 4.1 ± 2.6 mi (6.6 ± 4.2 km), respectively, although there were, almost certainly, undiscovered nests in between. Other telemetry studies showed similar wide spacing (Barbaree et al. 2014, Bloxton and Raphael 2009, Wilk et al. 2016), although in northern California where nesting habitat is very limited, nests were closer together and more often reused (Hébert and Golightly 2006). In some circumstances, nests might be more closely aggregated. For example, on the southern mainland coast of British Columbia, Manley (1999) found that 52 percent of nests located with tree climbing were within 300 ft of another nest, and on Naked Island, Alaska, Naslund et al. (1995) found three nests within a 43-ac (19 ha) stand. Additional evidence of co-location within stands and watersheds is reviewed by Plissner et al. (2015).

Analyzing the distribution of marbled murrelet nests relative to patch size and forest edges is limited, because many studies lacked a statistical comparison of habitat use in patches or edges versus the availability of these and alternative habitats (Jones 2001), and proximity to edges was not considered in relation to the degree of fragmentation of the landscape. Marbled murrelets are known to nest within 150 ft (46 m) of forest edges and in small, often isolated patches of suitable trees. The data summarized by McShane et al. (2004) showed that 75 percent of all nests were within 164 ft (50 m) of forest edges. Most of these edges were natural edges (streams, wetlands, natural forest gaps, and avalanche chutes) but almost a third of all nests were close to edges created by human activities. These data include nests located from ground searches and tree climbing linked to audiovisual surveys, and these nests are likely to be biased toward being found near edges (Burger 2002). When considering only the nests found by climbing randomly selected trees and radiotelemetry to remove possible bias, the results were similar: most nests were located near edges (76 percent of 152 nests), and the most common type of edge was natural (69 percent of 115 edge nests) (McShane et al. 2004). In this unbiased sample, which covered a range of modified and relatively pristine nesting habitats in Oregon, Washington, and British Columbia, 24 percent of all nests were near manmade edges, even though interior forest existed near many of these nests. Distances of nests to all edges in these samples ranged from 20 to 2,100 ft (6 to 640 m), and proximity to anthropogenic edges ranged from 9 to 1,000 ft (3 to 305 m) (McShane et al. 2004).

Studies using telemetry in British Columbia and Alaska found some murrelets nesting in small, often isolated patches of suitable forest; these patches were usually in higher elevation sites, where suitable trees are sparse and small patches of larger trees provide suitable platforms (e.g., Barbaree et al. 2014, Bradley 2002). When small patches are used in lower elevation sites, this often occurred where logging had removed most of the low-elevation suitable forest (e.g., Zharikov et al. 2006, 2007a). It is possible that murrelets persisted in such small patches because of site fidelity. Murrelets have shown a strong fidelity to sites where they have previously nested (e.g., Hébert and Golightly 2006). It is important to note that nest success may be lower in these smaller patches, probably because of higher risk of nest depredation (Barbaree et al. 2104). Fine-scale spatial analysis of the nests found with telemetry in Desolation Sound, on the southern mainland of British Columbia, showed that murrelets were more likely to nest close to natural edges, but there were insufficient data to test whether this was true for manmade edges (Burger 2002).

Two studies of nest placement did consider the use versus availability of edge habitat and patch size within the landscape. Raphael et al. (2016a) found that more than 60 percent of 162 nests in Washington, Oregon, and California were found in interior forest (defined as further than 180 ft (55 m) from any edge) (table 5-5). In that study, only 23 percent of potential nesting habitat occurred as interior forest on all lands in the study area, indicating a greater-than-expected occurrence of nests in interior forest. Wilk et al. (2016) analyzed nesting habitat at nests used by birds

Table 5-5—Number of marbled murrelet nests[a] located in core areas (interior forest) and near (within 180 ft [55 m]) edges

Location	Core	Core edge[b]	Edge[c]	Total
Washington	24	15	8	47
Oregon	29	23	4	56
California	45	8	6	59
Total	98	46	18	162
Percent	61	28	11	
Available (percent)[d]	22	28	49	

[a] Numbers of nests as sampled in Raphael et al (2016a), not the total number of known nests in this region.
[b] Edge of interior forest (core) patch.
[c] Isolated edge or stringer.
[d] Percentage of each type throughout range.
Source: Raphael et al. 2016a.

tagged with radios in the waters close to the Olympic Peninsula, Washington. Murrelet nests in Washington (n = 18) had greater core areas of older forest than random sites (235 ac [95 ha] at nest sites versus 25 ac [10 ha] in random sites). Core area is the interior area of the forest patch after buffering edge effects (180-ft buffers); this measure integrates patch size, shape, and edge-effect distance into a single measure. Raphael et al. (2016a) also found that patch cohesion, the physical connectedness of the corresponding patch type (index range 0 to 100), was greater at nests than random sites (93 at nests, 66 at random sites). They concluded that stands with nests were less fragmented than available forest across the murrelet's range that they sampled.

Edge effects on forest nesting habitat: windthrow, microclimate, and epiphytes—
A general rule of thumb used in Pacific Northwest forests has been that microclimatic effects penetrate two tree heights (240 to 300 ft [73 to 93 m]) and sometimes farther (450 ft [137 m] or more) into old-growth forests bordering clearcuts or similar sharp-gradient boundaries (Franklin and Forman 1987, Kremsater and Bunnell 1999). This is supported by some field studies, but local variables like topography, wind exposure, type of forest, and the surrounding matrix strongly influence the magnitude and influence distance of these edge effects (reviewed below).

Several studies reviewed below found differences based on edge type, in which "hard" edges are those with recent clearcuts (e.g., 0 to 20 years old) and "soft" edges are with regenerating forest (such as 21 to 100+ years old).

Windthrow refers to the uprooting or breakage of trees by wind, which can affect murrelets owing to loss of potential nest trees and nest limbs. Windthrow is increased when clearcuts, and to a lesser extent roads, increase the exposure of residual trees to wind (Sinton et al. 2000). Windthrow and physical damage to canopy branches are common problems at hard edges within the murrelet's range. In the Pacific Northwest, factors affecting the risk and degree of windthrow include orientation relative to winter winds; topography; the age, height, and density of trees; soil type; exposure to wind prior to logging (trees exposed to winds are more likely to develop stronger root systems); and the shape and size of the clearcuts and residual stands (Franklin and Forman 1987, Gratowski 1956, Mitchell et al. 2001). Although local factors have a strong influence, these impacts are generally found within 150 to 240 ft (46 to 73 m of edges, are most prevalent in patches less than 3 ha (7.4 ac), and are most likely within 25 years of clearcut logging creating the edges. In a review of data from the Pacific Northwest, Franklin and Forman (1987) suggested that wind-driven edge effects were likely to penetrate into remnant forests about two tree heights (240 ft [73 m]) from clearcut edges, but they did not distinguish between windthrow, canopy damage, and changes to microclimate.

Canopy epiphytes (mostly mosses) provide nest platforms for murrelets in much of the NWFP area. Exposure to increased wind and solar radiation at newly created edges could be detrimental (through wind-removal, thermal stress, and desiccation) or beneficial (through increased light for photosynthesis) to these epiphytes. Studies in the Pacific Northwest found variable effects of edges on bryophytes, although moss cover tended to be lower near hard edges. Local features, especially topography, time since edge creation, edge orientation, aspect, the nature of the surrounding harvested matrix, and even soil conditions have a strong effect on physical damage and changes in edge microclimates (Franklin and Forman 1987, Gratowski 1956, Mitchell et al. 2001, Muth and Bazzaz 2002, Sherich et al. 2013). These studies of edge effects on epiphytes and microclimate, although not focused on murrelet nesting, indicate that in many cases forests within 150 ft (46 m) of hard edges are likely to provide adverse conditions for nesting murrelets, and in situations with greater wind exposure, these adverse conditions could extend well beyond 300 ft (91 m). These adverse conditions are likely to diminish as the adjacent regenerating forest reduces the edge gradient (i.e., creates "soft" edges). One study, by Van Rooyen et al. (2011) at four locations in British Columbia, has specifically investigated edge effects on factors relevant to nesting murrelets. Compared to adjacent interior forest, epiphyte cover on canopy branches was slightly lower at hard edges (possibly because of the microclimate effects discussed above), about the same at soft edges, and slightly higher at natural edges. There was a large difference in the density of trees with potential nest platforms between hard edges and forest interiors (1.5 versus 6.4 platform trees per acre [0.6 versus 2.6 per hectare]); the difference was less marked at soft edges (6.5 versus 10.8 platform trees per acre [2.6 versus 4.4 per hectare]) and negligible at natural edges. The authors concluded that the creation of artificial edges by forest fragmentation would have negative consequences for epiphytic development for 20 to 30 years, and this might reduce nesting habitat for murrelets.

Natural forest edges bordering openings produced by streams, avalanche chutes, and wetlands generally do not provide adverse conditions for nesting murrelets, and if temperature and moisture regimes are favorable, such edges might be more suitable for murrelets than interior forests (Harper et al. 2005, Van Rooyen et al. 2011). Despite the evidence of negative microclimates and bryophyte development near hard edges, murrelet nests have been observed within 150 ft (46 m) of such edges, suggesting that conditions there are not always an absolute deterrent to the birds. We do not know if they avoid hard edges, i.e., whether nest densities at hard edges are lower than those elsewhere in old-growth forests. On balance, however, the evidence suggests that the creation of small patches and hard edges can be detrimental in areas where maintenance of nesting murrelets is a priority. Occurrence of nests along edges may, as noted above, be a result of site fidelity and a tendency to nest at previously used locations even when disturbances have created edges near those sites.

Microclimates within old-growth forests differ from those in clearcuts or young regenerating forests. In general, extremes of temperature and solar radiation are minimized, and humidity in summer is higher and more stable in old-growth forests than in recent clearcuts (Chen et al. 1999, Frey et al. 2016). Changes in microclimates can have both direct and indirect effects on nesting murrelets. Direct effects include thermal stress (both hot and cold) and dehydration if adults or chicks are exposed to direct sunlight or increased winds. Indirect effects are most likely to occur through changes to the availability of moss pads and other epiphyte growth on which most murrelet nests have been found.

Analysis across the Plan area indicates that the prevalence of fog is a strong contributor to predictive models of suitable nesting habitat for murrelets (Raphael et al. 2016a). In areas where fog is frequent, it might mitigate some edge effects, by promoting epiphyte growth and ameliorating stressful solar radiation. However, there is some evidence of reduced fog frequency, at least in California, over the past century (Johnstone and Dawson 2010).

Landscape-level relationships between nesting habitat and populations—

***Data from radar surveys*—**In this section, a landscape-level spatial scale considers entire watersheds and similar large areas in contrast to smaller stand- and patch-level analyses. Counts of murrelets entering watersheds obtained by detections from radar equipment have been instrumental in showing that murrelet numbers are strongly correlated with available areas of suitable old-growth nesting habitat (Burger 2001, Burger et al. 2004, Raphael et al. 2002a). In addition, Raphael et al. (2002a) also tested for the effects of habitat fragmentation in watersheds sampled with radar on the Olympic Peninsula, Washington. In their 3-year study, numbers of murrelets detected increased as the amount of core-area old-growth (defined as interior forest more than 300 ft [92 m] from an edge) increased (r^2 = 0.69, 0.82, and 0.76 in 1998, 1999, and 2000, respectively, $p < 0.01$), but decreased with increasing amounts of edge in late-seral patches. Numbers of murrelets were not correlated with patch density (number of patches per hectare), mean patch size, or spacing (proximity) of late-seral patches, nor with the overall diversity of all forest cover types within the landscape.

Cortese (2011) compared radar counts of murrelets entering watersheds with forest cover parameters within these watersheds in three regions of British Columbia: southwest Vancouver Island, and the central and southern mainland coasts. One goal of the study was to investigate the effects of forest fragmentation within these watersheds. As expected from previous radar studies (Burger 2001, Burger et al. 2004, Raphael et al. 2002a), total area of old-growth forest was included in the top predictive models for all three regions, which explained 11 to 35 percent of the variability in radar counts. Measures of mature forest edge density (including "hard" edges with clearcuts 0 to 20 years old, and "soft" edges with regenerating forest 21 to 140 years old) also were included in most predictive models, but there were marked regional differences in whether these were positive or negative associations. In the central and southern mainland coast regions, hard edges had a positive association with murrelet numbers, although there was high uncertainty in the model selection for the latter region. Cortese (2011), following Zharikov et al. (2006, 2007a), attributed this result to the preference by murrelets and the logging companies for the same patches of old-growth forest. Much of the old-growth forest in the watersheds studied in these regions has already been removed (Zharikov et al. 2006), and therefore murrelets tend to nest in the remaining forests where there is active logging and hence fragmentation. By contrast, murrelets in southwestern Vancouver Island, where a greater proportion of murrelet nesting habitat remains, showed a negative association with the density of hard edges and a strong negative association with the density of soft edges, and these edge factors were more important predictors in this region than in the other two regions (Cortese 2011).

***Data from at-sea surveys*—**Comparison of murrelet counts at sea with forest nesting habitat parameters emphasizes the value of tracts of suitable old-growth forest close to marine foraging areas (e.g., Falxa and Raphael 2016, Miller et al. 2002, Ronconi 2008, Raphael et al. 2015). In addition to the total area of accessible nesting habitat, Miller et al. (2002) found that nesting habitat patch size ($r = 0.91$) and contiguity of old-growth forest ($r = 0.95$) were the strongest predictors of murrelet densities at sea in northern California and southern Oregon. Raphael et al. (2016b) analyzed 13 years of data (2000–2012) from marine surveys in nine geographic strata across three states (Washington, Oregon, and California). Murrelet abundance at sea was most strongly correlated with the amount of higher suitability nesting habitat in the adjacent terrestrial environment ($r^2 = 0.324$), but there was considerable variance that was not explained by the factors included in the analysis. In addition, cohesion (an index of nesting habitat pattern in which higher values indicate more contiguous and less fragmented nesting habitat) was strongly and positively correlated ($r^2 = 0.76$) with murrelet abundance within the survey strata. We note, however, that amount of nesting habitat and cohesion of that habitat cannot be considered independent; cohesion tends to increase as amount of nesting habitat increases. Although the unexplained variance indicates that other factors also influence murrelet distribution and abundance, the results of Miller et al. (2002) and Raphael et al. (2015, 2016b) indicate that fragmentation of nesting habitat has negatively affected murrelet populations across the large, diverse, and highly modified NWFP area.

***Nesting habitat configuration and risk of nest predation*—**Breeding success in murrelets tends to be low (typically less than 35 percent of nests fledge chicks). A study using museum specimens indicated that historical breeding success about a century ago was sufficient to maintain stable murrelet populations, but that contemporary reproductive success is not (Beissinger and Peery 2007). Predation is the highest known cause of nest failure in recent decades and is likely to limit murrelet populations in many areas. Corvids (crows, ravens, and jays) are the nest predators most commonly documented, but owls, diurnal raptors, and arboreal mammals (squirrels and mice) (Bradley et al. 2003; Malt and Lank 2007, 2009) are also likely to be important predators. Although definitive demographic studies testing the effects of predation are limited to the edge of the species range in central California (Peery and Henry 2010; Peery et al. 2004, 2006a), those studies and cumulative evidence from across the species range indicate that nest predation is a limiting factor on murrelet populations (McShane et al. 2004, Nelson and Hamer 1995, Piatt et al. 2007). Studies in several parts of the species range show that only about a third of murrelet nests result in fledging, e.g., 0.33 fledglings per nesting attempt rangewide, n = 124 nests (McShane et al. 2004), and 0.23 to 0.46 in British Columbia (Burger 2002). Research using radiotelemetry found failure rates of 54 percent in British Columbia (Bradley 2002), 68 to 86 percent in northern California (Hébert and Golightly 2006), 84 to 100 percent in central California (Peery et al. 2004), 80 percent in southeast Alaska (Barbaree et al. 2014), and 31 percent in south-central Alaska (Kissling et al. 2015). It is possible that nesting success results from radiotelemetry studies are affected by the method: Peery et al. (2006b) found that radio-tagged murrelets had a lower survival rate, and Ackerman et al. (2004) found that radio-tagging reduced reproductive success in another small alcid, the Cassin's auklet *(Ptychoramphus aleuticus*).

Predation is the greatest known cause of failure at 78 percent, or 29 of 37 nests with known outcomes in a rangewide analysis (McShane et al. 2004). In southern British Columbia, Malt and Lank (2007) found no difference between the survival of 57 actual versus 40 artificial murrelet nests and were able to document predator discovery at 40 percent of 136 artificial nests. In northern California, Hébert and Golightly (2006, 2007) attributed a minimum of 51 percent of nest failures across 3 years to predation, and documented egg predation by ravens (*Corvus corax*) and Stellar's jays (*Cyanocitta stelleri*). In central California, rates of nest predation were consistently high (67 to 81 percent)(Peery et al. 2004).

Several studies across the southern part of the murrelet's range have investigated nest success relative to forest edges and habitat fragmentation (table 5-6). As in many studies of habitat fragmentation, separating the effects of proximity to edge to the related effects of patch size and habitat configuration is often difficult (Harper et al. 2005, Lindenmayer and Fischer 2007). Nelson and Hamer (1995) found that successful nests were significantly further from forest edges (mean 510 ± SE 241 ft [155 ± 73 m], n = 9) than nests that failed (mean 90 ± SE 20 ft [27 ± 6 m], n = 8), and all successful nests, except one, were more than 180 ft (55 m) from the forest edge. For 58 nests with known locations from Oregon and British Columbia, Manley and Nelson (1999) (see also Burger 2002) reported that the success of nests within 150 ft (46 m) of a forest edge was 38 percent (n = 29) and for those more than 150 ft from an edge, success was 55 percent (n = 29), but this difference is not statistically significant. Successful nests were significantly further from edges (mean 462 ft [141]) than failed nests (mean 184 ft [56 m]). Predation was responsible for the failure of 60 percent of all active nests in these samples, and predation rates were higher within 150 ft of edges than farther into the forest interior. All 13 nests that were more than 450 ft (137 m) from an edge were successful or failed from reasons other than predation. There was a trend for successful nests from Oregon and British Columbia to occur in larger stands (mean 1,212 ac [491 ha]) than unsuccessful nests (mean 694 ac [281 ha]), although this was not statistically significant.

Bradley (2002) analyzed the success of nests found by telemetry in Desolation Sound, British Columbia, relative to their proximity to forest edges. Successful nesting was assumed if the radio-tagged adult visited the nest up to the midpoint in the chick-rearing period and was confirmed at some nests by tree climbing after the chick had fledged. Bradley (2002) conducted two analyses. One was from ground-based measures of distance from edge and nest success from 37 accessible nest sites, analyzed at 150 and 300 ft (46 and 91 m) distances from edge. At both distances, there were no significant differences in nest success at sites adjacent to or far from forest edges. Most nests were located adjacent to natural edges rather than artificial ones. Comparing nest success at natural and artificial edges was difficult, because only two nests were located directly adjacent to artificial edges (both were successful). Bradley's (2002) second analysis was a coarse-scale geographic information system (GIS) analysis using 98 nest sites, looking at edge type within 600 ft (182 m) of sites based on 1:250,000 landscape classification maps. In this analysis, the proportions of sites adjacent to edges versus interior were similar to those in the first ground-based sample. As in the first analysis, many nest sites were adjacent to natural edges, predominantly avalanche chutes, and most of these nesting attempts were successful (79 percent, n = 42). Nest success near artificial edges (61 percent, n = 23) and in forest interiors (48 percent, n = 33) was lower. Nests adjacent to natural edges had significantly higher success than those in the forest interior, but

Table 5-6—Summary of studies investigating the effects of habitat fragmentation, small patches, and forest edges on the success of marbled murrelet nesting

Study	Location	Type of study	Conclusions
Nelson and Hamer 1995	Rangewide	Review of early studies	Successful nests significantly farther from forest edges than failed nests. Corvid predation important.
Manley and Nelson 1999	Oregon and British Columbia—using some of same data as above	Review of early studies	38-percent success in nests <150 ft; 55 percent success in nests >150 ft. Predation responsible for at least 60 percent of failures.
Bradley 2002; see also Burger 2002	Desolation Sound, British Columbia	Nest success based on telemetry and post-fledging evidence	No negative effect of natural edges (e.g., avalanche chutes); insufficient data to test effects of clearcut edges.
Luginbuhl et al. 2001, Marzluff et al. 2000, Raphael et al. 2002b	Olympic Peninsula, Washington, and Oregon	Artificial nests with mimic eggs and chicks in natural nest locations	No consistent effects of forest fragmentation on nest survival. Proximity to human activity increased predation rates. Corvid predation important. Maturing forest bordering old-growth nesting habitat reduced predation risk.
Malt and Lank 2007	Southwestern British Columbia	Artificial nests with mimic eggs and chicks in natural nest locations	Predator visits significantly higher at edges (<150 ft) than in forest interior (>450 ft from edges), but no difference between “hard,” “soft,” and natural edges. Predatory corvids more likely at “hard” edges.
Malt and Lank 2009	Southwestern British Columbia	Artificial nests with mimic eggs and chicks in natural nest locations	Predator disturbance 2.5 times more likely at hard edges than in forest interior. Soft and natural edges not so. Corvid predation important. Maturing forest (20 to 40 years old) bordering old-growth nesting habitat reduced avian predation risk.
Hébert and Golightly 2006, 2007; Peery et al. 2004, 2006	Central and northern California	Telemetry and nest observations showing nest success in highly fragmented forests	84-percent nest failure; 67 to 81 percent of nests predated. Corvid predation important. Repeated use of same nest site associated with high predation.
Zharikov et al. 2006	Desolation sound, British Columbia	Nest success based on telemetry evidence only (new analysis using Bradley 2000 data)	Breeding success was greater in areas with recent clearcuts and lower in areas with much regrowth.

there were no significant differences between nests adjacent to artificial versus natural edges and artificial edges versus interior forest. In summary, Bradley’s (2002) analysis did not support the hypothesis that nesting near forest edges was harmful to murrelets, but could not resolve whether natural or artificial edges produced differences in nest success. Bradley’s (2002) study was limited because only 38 percent of the nests were accessible for ground-based measures and tree climbing, and proximity to edges for most nests was inferred from coarse-scale global positioning system (GPS) locations with ±100 m (328 ft) accuracy. The more detailed study by Malt and Lank (2007, 2009) in the same area and using some of the same nest data did find significant negative edge effects and differences between edge types (see below).

A later analysis by Zharikov et al. (2006) studied habitat selection and breeding success at nest sites located with telemetry in Desolation Sound (heavily logged; 121 nests) and Clayoquot Sound on the west coast of Vancouver Island (relatively intact; 36 nests). Comparing nest sites with randomly located points in these same areas, they found that murrelets used either old-growth fragments proportionately to their size frequency distribution (more intact landscape) or tended to nest in disproportionately smaller fragments (heavily logged landscape). Nests were closer to clearcut edges than expected, with mean distances to forest edges of 1.2 and 1.5 mi (1.9 and 2.4 km) at nest sites and randomly chosen points, respectively). Breeding success, as inferred from nest attendance patterns by radio-tagged parents, was modelled in Desolation Sound, where sample sizes were sufficient (Zharikov et al. 2006). They found that breeding success was greater in areas with recent clearcuts and lower in areas with much regrowth, implying that marbled murrelets can continue nesting in highly fragmented old-growth forests, successfully using patches of about 25 ac (10 ha) or greater. However, they cautioned that breeding success in fragmented areas may decrease as adjacent clearcuts overgrow, and that their findings imply that the same stands of old-growth forest may be equally attractive to marbled murrelets and logging companies, versus a murrelet preference for forest fragmented by logging (Zharikov et al. 2006). The finding by Zharikov et al. (2006) that murrelets can nest successfully in highly fragmented old-growth forests differs somewhat from results of other studies from British Columbia (Burger 2002); Burger and Page (2007) suggested that the spatial resolution and scale of the Zharikov et al. (2006) analyses were not sufficient to test edge effects (see Zharikov et al. 2007b for their response).

Because of the difficulties in locating and monitoring murrelet nests, several studies have resorted to using artificial nests with eggs or chicks mimicking those of the murrelet. Justification for this approach for studying murrelets is provided by Raphael et al. (2002b) and Malt and Lank (2007, 2009). “Predation” and disturbance by predators at artificial nests was based on removal, photographic or video evidence, movements detected by implanted motion sensors, or bite and peck marks made on wax coatings of eggs or chicks (Luginbuhl et al. 2001, Malt and Lank 2007, 2009). Artificial murrelet nests do not, of course, have an attendant parent, which might affect the rates of predation, although incubating adults have been attacked by ravens, and adults do leave eggs unattended for periods of several hours (Nelson and Hamer 1995). Murrelet chicks are brooded by adults for only a few days after hatching. The use of artificial nests to test predation effects has been criticized (e.g., Faaborg 2004), but their use has also been supported as allowing more rigorous and controlled quantitative experiments (Batáry and Báldi 2004). In the only study to compare the success of real and artificial marbled murrelet nests at various edge types, Malt and Lank (2007, 2009) found that artificial nests had significantly lower probabilities of disturbance (0.18 ± 0.05) than the probabilities of failure at real nests (0.35 ± 0.07), but the patterns of disturbance/failure were similar across edge types for real and artificial nests (reviewed below). If these results apply generally, then artificial nests seem unlikely to overestimate predation rates, and there is support for their application for studying edge effects in murrelets.

Intensive research on the likely impacts of forest structure and landscape contiguity on murrelet nest predation was undertaken by Marzluff and his team in the Olympic Peninsula, Washington, and in Oregon (Luginbuhl et al. 2001, Marzluff et al. 2000, Raphael et al. 2002b). Their experiments used painted plastic eggs and dark chicken chicks placed high in forest canopies to mimic those of murrelets. Video monitoring and marks on wax coatings identified predators, and field studies were supplemented with laboratory studies to test whether potential predators would attack eggs or chicks. Their field trials were focused on determining the effects of forest structure (simple or complex and of different ages), landscape contiguity (classified as fragmented when plots were more than 75 percent surrounded by clearcuts or contiguous when plots were more than 75 percent surrounded by mature forest), and proximity to human activities (near, less than 0.6 mi [1 km]), or far, more than 3.1 mi (5 km), from towns, farms, campgrounds, dumps, highways, etc.). Survival of simulated nests differed relatively little among the various forest cover types, and there were no consistent effects of forest fragmentation on nest survival but proximity to human activity increased predation rates. At locations far from human activity, predation rates were greater in continuous stands than in fragmented stands, but close to human activity, predation rates were similar in continuous and fragmented stands. The highest nest survival occurred in mature forest with simple structure, which were either contiguous and near human activity or fragmented and far from humans. Densities of corvids were lowest in contiguous, simple-structured maturing forests, regardless of proximity to humans, and corvid numbers differed little among the other forest cover categories. It is difficult to infer generalizations from these results, apart from negative effects of proximity to human activities, but Marzluff et al. (2000) suggested that old-growth stands used by murrelets for nesting might be best buffered by surrounding the stands with maturing, simple-structured forests in which there were relatively few predators.

In the same study, Luginbuhl et al. (2001) reported a strong negative correlation between survival of simulated murrelet eggs and corvid abundance at the landscape level (2 to 20 mi^2 [5 to 52 km^2] scale). Corvid abundance explained 69 percent of the variance in predation of simulated murrelet eggs. This trend was not evident at smaller plot-level scales (60 to 120 ac [24 to 49 ha]). The cause of this scale-sensitive relationship was likely due to the large home range of some of the corvid species (ravens and crows). For monitoring and management purposes, this result implies that such negative correlations might not be evident unless large spatial scales are considered. Artificial nests in areas with high use by Steller's jays lasted only half as long as those in low-use areas (Vigallon and Marzluff 2005).

Malt and Lank (2007, 2009) used artificial nests with painted eggs and stuffed quail chicks to study predation rates likely to apply to murrelets relative to edges in four sites in British Columbia. Avian predators caused more than half of the disturbances, with squirrels and mice also frequent. Artificial eggs were disturbed more frequently than nestling mimics, and birds and squirrels disturbed eggs more than nestlings, but the reverse was true for mice. In their first study (Malt and Lank 2007), disturbances of nest contents by all predators was significantly higher at edges (less than 150 ft [46 m]) than in the forest interior (more than 450 ft [137 m] from edges), but there was no difference between "hard," "soft," and natural edges. In both studies, predation of eggs by birds (mainly corvids) was always higher at hard edges than in interior forest, but soft or natural edges did not show this effect. Predation on nest contents by squirrels and mice was more variable regionally and with forest type, but generally predation by mice was not strongly affected by edges (although higher at natural edges than in adjacent interiors). They found no edge effects from squirrel predation in their first study (Malt and Lank 2007), but in their second study squirrel predation was higher at all edge types than in adjacent interior forest (Malt and Lank 2009).

At the landscape scale, Malt and Lank (2009) found that avian predation risk was negatively affected by the percentage of regenerating forest 20 to 40 years old; i.e., the risk of egg predation decreased by more than half if the bordering regenerating forest increased from 1 to 40 percent. This matches the conclusions by Marzluff et al. (2000) on the buffering effects of regenerating younger forest. Malt and Lank (2007) also reported higher predation in landscapes with a higher proportion of old-growth forests, which might indicate that recent clearcuts and regenerating forests supported fewer predators overall.

Some important trends emerge from the work of Malt and Lank (2007, 2009). Predation risk from avian predators was considerably higher than from mammals, and the birds were more likely to target eggs than nestlings. This risk from avian predators was particularly high at hard edges, but much less likely at soft or natural edges, and the landscape-level analysis indicates that this is likely due to reduced predation risk as the regenerating matrix changes from clearcut to young (20- to 40-year-old) forest. They also found strong edge effects among squirrels, which is contrary to the general belief that squirrels are less attracted to edges than birds, such as corvids (Marzluff and Restani 1999).

The reduction and fragmentation of old-growth forests can also lead to the undesirable situation in which murrelets and some of their predators (especially old-growth-dependent species such as goshawks) are restricted to using the same small patches. This could lead to greater risk of predation. If adult murrelets are put at risk in this way, it would have serious consequences for populations.

Nesting murrelets and their eggs and chicks are at risk to a formidable array of potential predators, and the murrelet's cryptic and widely spaced nest sites, secretive and crepuscular visits to nests, and camouflaged breeding plumage are all obvious adaptations to reducing predation risk. Although it is difficult to estimate the predation impacts of the complete suite of predators (birds and mammals) in any area, it is clear that corvids, especially Steller's jays and common ravens, are the most common nest predators across the murrelet's range (McShane et al. 2004, Nelson and Hamer 1995, Piatt et al. 2007). Both of these species and, in some situations, other predators like squirrels (Malt and Lank 2007, 2009), exhibit strong affinities with forest edges (Marzluff et al. 2000). Murrelets nesting at edges, and especially hard edges bordering open areas like clearcuts, appear to be at greater risk of predation than in the forest interior. Given that nest predation appears to be a dominant demographic driver for the murrelet (McShane et al. 2004; Nelson 1997; Peery et al. 2004, 2006a; Piatt et al. 2007), any forest alteration that increases predation risk is likely to have a negative and perhaps serious impact on local murrelet populations. Reducing predator risks by minimizing edge habitats and controlling corvid access to garbage and human food (e.g., at campsites) is also likely to benefit murrelets in modified landscapes.

The situations in northern California, documented by Hébert and Golightly (2006, 2007) and in central California by Peery et al. (2004, 2006b), illustrate how massive nesting habitat loss and limited nesting options for murrelets lead to a classic habitat trap situation (Battin 2004). Murrelets nesting in those regions are concentrated in the relatively small patches of suitable redwood forests remaining, and reuse of the same trees and nest sites is higher than what is recorded elsewhere (Burger et al. 2009; Hébert and Golightly 2006, 2007). These trees and nest sites are repeatedly visited by corvids (Steller's jays and common ravens), and consequently nesting success is extremely low in conservation zone 6 at the southern end of the murrelet's breeding range, where 84 percent of nests fail and predation rates at nests are 67 to 81 percent (Peery et al. 2004). Along with periodic food shortages linked to oceanic variability, nest predation is considered to limit this population, which appears to be sustained by immigration (Peery et al. 2004, 2006a, 2007). Reducing corvid populations (Peery and Henry 2010) and aversion conditioning to reduce nest predation by Steller's jays (Gabriel and Golightly 2014) are potential management strategies to help maintain this marginal population of murrelets. This extreme situation might not be typical of other less-modified parts of the murrelet's range, but is likely similar in northwest Oregon, southwest Washington, and northern California, and on Bureau of Land Management lands in Oregon where the landscape is highly fragmented. These situations indicate the risks of excessive habitat reduction and fragmentation.

In summary, this review shows that many factors affect the risks to murrelets when they nest near forest edges or in small forest patches, including the type of edge, the type of habitat bordering the edge, the suite of predators likely, and proximity to human activity (table 5-6). In most situations, particularly where ravens and jays are common, nesting near (<150 ft [46 m]) "hard" edges (i.e., the bordering regenerating forest is less than 20 to 40 years old) will increase predation risk.

Marine habitat—

The NWFP is tightly linked to the status and trends of murrelets because its lands provide the majority of suitable nesting habitat within the species' listed range in Washington, Oregon, and California. Recent analyses indicate that nesting habitat conditions best explain the abundance and trends of murrelets at sea off the NWFP area during the breeding season (Raphael et al. 2015, 2016b). A breeding-season pattern of murrelets tending to occur offshore of nesting habitat is consistent with nesting murrelets behaving as central-place foragers, subject to energetic constraints that limit them to foraging within some radius of their nest location—the "central place" (Raphael et al. 2015). Murrelets depend entirely on marine prey, and because of this, prey conditions such as abundance and quality, and the underlying factors affecting prey conditions, are important to the future of the murrelet in the Plan area and elsewhere. Thus, the juxtaposition of productive foraging habitat offshore of nesting habitat may be important to murrelet conservation. Notably, reviews of murrelet biology (McShane et al. 2004, Piatt et al. 2007) indicate that the distribution of foraging murrelets is strongly influenced by patterns of prey availability (and perhaps juxtaposition to nesting habitat), while other studies found that prey quality or availability influence breeding success (Becker et al. 2007, Gutowsky et al. 2009, Norris et al. 2007).

Below, we summarize those recent analyses that evaluated the relative contributions of marine conditions and nesting habitat conditions to murrelet status and trend in the Plan area, and review the larger body of scientific information on the relationships between marine habitat conditions and murrelet biology throughout the species' entire range.

To understand the murrelet's marine habitat, it is helpful to introduce some key features of that habitat. First, most of the marine waters off the NWFP area are within the California Current system, the southward-moving surface current of colder water from the north Pacific. A key characteristic of the system is wind-driven upwelling of cooler and typically nutrient-rich waters to the surface in nearshore areas, particularly in spring and summer. This upwelling of nutrients results in increased productivity (Batchelder et al. 2002), and may be key to maintaining cold, productive marine conditions favorable to murrelets south of Washington state, in areas that would have warmer sea temperatures in the absence of the current system and upwelling (McShane et al. 2004).

The Puget Sound/Salish Sea region differs from elsewhere in the Plan area; it is not dominated by the California Current, and it has a more complex nearshore geography shaped by glaciation and with many islands, like many areas to the north, which creates local currents and tidal patterns that concentrate prey.

Marbled murrelet prey—Marbled murrelets prey on a wide range of marine fish and invertebrates (Burkett 1995, Nelson 1997). Murrelets appear to have a flexible foraging strategy, exploiting the prey species and foraging locations that maximize energy gain (Piatt et al. 2007). For example, murrelets selected less abundant, higher value Pacific herring (*Clupea pallasii*) at times over other, more abundant species (Ostrand et al. 2004), and sometimes foraged in deeper waters than normal, where local conditions created prey concentrations near breeding areas (Kuletz 2005).

Species composition of available prey changes across the murrelet's range, perhaps most notably between the California Current system and the Alaska Current system, which dominates the species' range north of the NWFP area. Common murrelet prey species include sand lance (*Ammodytes hexapterus*) and smelt (family Osmeridae), which are taken by murrelets in many areas, as are small herring and krill (*Thysanoessa* spp. and *Euphausia* spp.), where available. As one moves north, and particularly north of the California Current area, sand lance, capelin (*Mallotus villosus*), and small Pacific herring are frequent murrelet prey (Bishop et al. 2014, McShane et al. 2004, Piatt et al. 2007); all three of these species are of moderate to high quality in terms of energy content (Anthony et al. 2000). Of these, capelin do not occur from the Olympic Peninsula southward, and sand lances become scarce in some areas to the south of the peninsula. Within the California Current, northern anchovy (*Engraulis mordax*) and, in spring, juvenile rockfish (*Sebastes* spp.) are dominant small schooling fish in nearshore waters (McShane et al. 2004), and are taken by murrelets (Burkett 1995). Although fish tend to dominate the murrelet diet and exclusively comprise prey brought to the nest, invertebrates, particularly krill, are taken at times by adults throughout the murrelet's range.

Marine proxies for prey abundance in the NWFP area—As part of the 20-year monitoring report and related work, the NWFP effectiveness monitoring program analyzed the relative influences of marine and terrestrial factors on murrelet distribution and population trends during the first two decades of the NWFP (Raphael et al. 2015, 2016b). Although the murrelet diet has been studied to the north of the Plan area, particularly in Alaska (summarized in McShane et al. 2004 and Piatt et al. 2007), few studies have been conducted on the murrelet diet south of Canada, and monitoring data for murrelet prey species from waters off NWFP lands are equally sparse. For these reasons, Raphael et al. (2015, 2016b) used physical and biological attributes of marine habitat as proxies for local prey abundance in their analyses. The attributes that the authors measured included chlorophyll "a" concentration in ocean surface waters and sea surface temperature, which have been used in comparable analyses by others (e.g., Ainley and Hyrenbach 2010, Hazen et al. 2012), and are available at relatively fine temporal and spatial scales. The idea is that cooler water is rich in nutrients. This in turn leads to a more robust food chain, ultimately leading to a more robust supply of small fishes and invertebrates that murrelet prey upon. Cooler waters are enriched with nutrients compared with warmer waters, and are frequently associated with upwelling. Chlorophyll "a" concentration has for decades been a proxy for phytoplankton abundance and primary productivity, and performs well in this role (Huot et al. 2007). In the northeast Pacific (Ware and Thomson 2005) and California Current (Reese and Brodeur 2006), chlorophyll "a" concentration was positively associated with fish abundance, as was phytoplankton abundance in the North Sea (Frederiksen et al. 2006). In the California Current, chlorophyll "a" peak abundance was a strong predictor of seabird abundance and hotspots of seabird density (Suryan et al. 2012). For these reasons, Raphael et al. (2015, 2016b) hypothesized that murrelet prey abundance would be positively associated with primary productivity.

Marbled murrelet prey availability is likely to be affected by broader Pacific Ocean conditions, including the Pacific Decadal Oscillation (PDO) (Mantua et al. 1997) and the El Niño Southern Oscillation (ENSO) (Trenberth 1997), which have widespread effects on marine productivity and food webs, as well as on seabird populations, including other diving seabirds in the California Current system (Ainley and Hyrenbach 2010). Therefore, the 20-year NWFP analyses also included factors to account for variability in PDO and ENSO conditions (Raphael et al. 2016b). The ENSO is a pattern of periodic changes (events), typically lasting about 9 to 18 months, that produce (1) El Niño events with increased sea surface temperatures and reduced coastal upwelling, and (2) La Niña events that result in colder, more nutrient-rich waters than usual (Mestas-Nunez and Miller 2006, Schwing et al. 2002). The PDO represents long-term (20 to 30 years) climate variability in the north Pacific Ocean, in which there are observed warm and cool phases, or "regime shifts" with corresponding patterns of weaker or strong upwelling (Mantua et al. 1997). Later (see "Climate Change Considerations" below), we discuss potential effects of climate change on murrelet prey and these proxies.

Associations with marine habitat and prey—Although prey and foraging habitat conditions differ across the murrelet's wide range, murrelets forage and rest mostly in shallow nearshore waters associated with the continental shelf (Nelson 1997). Murrelets often use sheltered waters when available (Nelson 1997), but most of the coast in the Plan area (except for the Puget Sound area) lacks the complex structure and sheltered areas found farther north in the glaciated fjords and abundant islands of Alaska and British Columbia. In the Plan area, data from the at-sea work of the NWFP effectiveness monitoring program shows that most murrelet foraging during the breeding season occurs in water depths of 80 ft (24 m) or less, except for the San Juan Islands and northern Puget Sound, where murrelets used waters up to 130 ft (40 m) depth (Raphael et al. 2016b).

Analyses for the 20-year NWFP murrelet report examined variation in murrelet abundance in relation to dominant shoreline substrate within the Plan area, and found that murrelet abundance was greater offshore of fine- to medium-grained sand beaches and was also greater offshore of estuaries and marshes, compared to other substrates (Raphael et al. 2016b). In an earlier study of murrelet habitat use off southern Oregon, murrelets were most abundant near ocean bays, river mouths, sandy shores, and submarine canyons (Meyer and Miller 2002). Similarly, murrelet densities off British Columbia were highest over sandy substrate, near estuaries, and where waters are coolest (Burger 2002,, Piatt et al. 2007, Ronconi 2008, Yen et al. 2004). In a study at the southern end of the murrelet's range near Santa Cruz, California, Becker and Beissinger (2006) found that foraging murrelets appeared to prefer cooler waters associated with areas of recent upwelling.

In their review of murrelet ecology, Piatt et al. (2007) concluded that physical and biological oceanographic processes that concentrate prey (such as upwellings and rip currents) have an important influence on where murrelets forage. Although that conclusion is largely based on work in Alaska and British Columbia (e.g., Burger 2002, Day and Nigro 2000, Kuletz 2005), it is supported by work in Washington, Oregon, and California (Ainley et al. 1995, Nelson 1997, Strong et al. 1995). This suggests that, at the finer scale, across their range, murrelets select foraging areas based on similar topographic and oceanic factors associated with higher prey densities in shallower waters. This pattern is consistent with the often strong positive relationship between forage fish abundance and the abundance of fish-eating birds (e.g., Durant et al. 2009, Furness and Tasker 2000).

***Changes in foraging habitat conditions*—**There is some information from analyses of stable isotopes in murrelet tissues indicating long-term declines in murrelet diet quality in portions of its range in central California (Becker and Beissinger 2006), the Salish Sea, including northern Washington (Gutowsky et al. 2009), and British Columbia (Norris et al. 2007). At least one of these studies suggested that murrelet foraging success along the Pacific Coast is sensitive to climate variability, and that cooler ocean waters and resulting prey conditions are associated with greater reproductive success (Becker et al. 2007). Further, though murrelets have flexible foraging and life history strategies that presumably evolved in an environment of varying prey conditions, there is evidence that declines in murrelet diet quality may have contributed to reduced murrelet reproductive success in the Salish Sea (Gutowsky et al. 2009), and that foraging flexibility in murrelets (Ronconi and Burger 2009) and other alcids (Schrimpf et al. 2012) may not be sufficient to avoid low reproductive success when environmental conditions are extremely poor. Adult survival in murrelets appears less vulnerable to poor forage conditions than does reproductive success (Beissinger and Peery 2007, Peery et al. 2006a, Ronconi and Burger 2008), and Ronconi and Burger (2008) proposed that murrelets likely have a life history strategy in which adults do not initiate nesting, or abandon nesting attempts, to maximize their own survival when available forage is inadequate. Piatt et al. (2007) concluded that climate-related changes in marine ecosystems, in addition to human activities (logging, gill net bycatch, oil pollution), were the likely reasons for the wide-scale declines in murrelet populations in British Columbia and Alaska.

Environmental conditions, particularly El Niño events, have been shown to markedly reduce prey availability for some seabirds in California, leading to poor reproductive success (Ainley et al. 1995). Although El Niño events appear to reduce overall seabird prey availability, their effect on murrelets are not well known. Inner coastal waters in the Puget Sound and the Strait of Juan de Fuca, as well as estuarine areas along the outer coast, appear less influenced by El Niño conditions because of mixing and nutrients from sources other than outer coastal waters (USFWS 1997). In addition to ENSO variation, it is known that fish populations and zooplankton in the California Current generally do better during “cold” than in “warm” phases of the PDO, while in the more northerly Alaska Current, some fish populations such as salmon behave oppositely (Hallowed et al. 2001).

The 20-year NWFP analysis found only one forage-fish dataset from the Plan area of interest, and which spanned the period of that analysis (Raphael et al. 2016b). Those data provided abundance of forage fish from two transects located just north and south of the Columbia River. For this limited area, the authors found some evidence in the year-to-year variation of a positive relationship between forage fish and murrelet abundance, and concluded that direct measures of forage-fish abundance as predictors of murrelet abundance need additional investigation (Raphael et al. 2016b).

Climate Change Considerations

The Intergovernmental Panel on Climate Change (IPCC) is a scientific body that was set up in 1988 by the World Meteorological Organization and United Nations Environment Program to inform policymakers about the causes of climate change, its potential environmental and socioeconomic consequences, and the adaptation and mitigation options to respond to it. In 2014, the IPCC published its Fifth Assessment Report, which is widely considered the most comprehensive compendium of information on actual and projected global climate change currently available. Although the extent of warming likely to occur is not known with certainty at this time, the IPCC has concluded that warming of the climate system is unequivocal: that the atmosphere and ocean have warmed, sea level has risen, and continued greenhouse gas emissions will cause further warming (IPCC 2014). Ocean warming accounts for more than 90 percent of the energy accumulated and stored in the climate system between 1971 and 2010 (IPCC 2014). Although the report does not focus on changes at the scale of the NWFP, it did find with high confidence new evidence for decreasing spring snowpack in western North America.

Climate change and terrestrial nesting habitat—
Although murrelets spend most of their time in the marine environment, murrelets require suitable forest cover for nesting. The U.S. Fish and Wildlife Service reviewed potential threats to murrelet nesting habitat in its last status review (USFWS 2009). The agency concluded that, based on climate model projections, the future conditions of forests where murrelets nest will be largely unfavorable for maintaining current forest structure and composition. Projections suggest that increases in annual temperature changes within the range of the murrelet will be greatest in the summer and lowest in the spring, but predicted that temperature increases near the coast will be generally lower than in the rest of the Plan area (Dalton et al 2013). Already in the Pacific Northwest, tree mortality rates in unmanaged old forests have increased in recent decades at a rate equivalent to doubling over 17 years (van Mantgem et al. 2009), a change the authors suggested was likely due, at least in part, to documented regional warming and drought stress associated with climate change. With respect to drought stress, Johnstone and Dawson (2010) found evidence of a 33 percent reduction in fog frequency over the past century in the coast redwood forest zone of northern California, which includes most of the nesting habitat in conservation zone 5 and in the California portion of conservation zone 4. Based on tree physiological data, they suggested that redwood and other western coastal forest ecosystems may experience increasing drought stress as a result of reduced summer fog and greater evaporative demand.

During the next 20 to 40 years, climate projections for the Pacific Northwest indicate likely decreases in Douglas-fir growth from drier summers (Littell et al 2010). Heat extremes and heavy precipitation events are likely to become more frequent (Loehman and Anderson 2009). With these changes, the potential exists for increased fire frequency and severity, even in the coastal forests where murrelets nest (Millar et al. 2006). In North America broadly (Dale et al. 2001) and the Pacific Northwest specifically (Kliejunas et al. 2009; Littell et al. 2009; Mote et al. 2003, 2010), climate changes may also alter forest ecosystems via the frequency, intensity, duration, and timing of other disturbance factors such as drought, introduced species, insect and pathogen outbreaks, windstorms, ice storms, landslides, and flooding. Evidence for an increased role of fire within the range of the murrelet is mixed, with some models projecting increases and others projecting decreases (see chapter 2, "Climate," and chapter 3, "Vegetation Change"), but the historical occurrence of large, high-severity fires suggests the potential for losses in nesting habitat if fires do occur (Agee 1993). Overall, the evidence is substantial that climate change will result in changes to forest habitats where murrelets nest. The magnitude of those changes is less known, as is how nesting murrelets and murrelet populations will respond to forest changes. However, to the extent that changes such as increased tree mortality, decrease in canopy epiphytes, and increased severity and frequency of fires reduce the number of potential nest trees, impacts to murrelets appear likely to be negative.

Climate change and marine habitat—
In addition to influencing the quality and abundance of nesting habitat, as discussed above climate change is likely to result in changes to the murrelet's marine environment, with effects on murrelet food resources the most likely mechanism. Given the large body of climate change literature, we focus our review here on such potential effects on prey resources.

The U.S. Fish and Wildlife Service reviewed the potential effects of climate change on murrelets south of Canada, and concluded that—although predicting climate change effects on marine resources is complex and has many uncertainties—taken as a whole, the evidence from models and other sources suggested that few changes are likely to benefit murrelets, with many more having the potential for neutral or adverse effects (USFWS 2009). The same review found it most likely that the murrelet prey base will be adversely affected to some extent by climate change, and noted that although seabirds generally have life-history strategies adapted to variable marine environments, ongoing and future climate change could present changes of a rapidity and scope outside the adaptive range of murrelets (USFWS 2009).

Marine changes already observed may be attributable to climate change. El Niño events have become more frequent, persistent, and intense during the last decades of the 20th century (Snyder et al. 2003). There is general agreement that sea surface temperatures will increase as a result of climate change, with evidence that they have already increased in murrelet marine habitat off the NWFP area by 0.5 to 1.0 °C (about 1 to 2 °F) over the last half century, both in the California Current system (Di Lorenzo et al. 2005) and in the Strait of Juan de Fuca (Rucklehaus and McClure 2007). In the murrelet's nearshore environment, upwelling of cold waters may moderate some level of sea-surface-temperature changes, but differences in the timing, intensity, and duration of upwelling can affect productivity, resulting in considerable uncertainty regarding the ultimate effects of marine changes on murrelet foraging conditions. Climate models show inconsistent projections for the future of coastal upwelling in the Pacific Northwest (Melillo et al. 2014). Illustrating the complexities of making such projections, Sydeman and others (2014) conducted a meta-analysis of the literature on wind intensification in coastal upwelling marine systems over the prior six decades. They found support for wind intensification in the California Current system and noted that this could increase nutrient input and benefit marine populations if primary production is nutrient limited. However, they emphasized the complexity of forecasting the consequences of wind intensification in coastal ecosystems because the ecological effects are likely sensitive to diverse factors including phenology of upwelling-favorable winds, patterns of nutrient transport offshore, differing responses of food web species, and potential for increased stratification resulting from increased water temperatures (Sydeman et al. 2014).

Climate change is anticipated to result in sea-level rise and a decrease in the pH of marine waters, with unknown effects in both the California Current system and Puget Sound. Increasing acidification of marine waters caused by increased absorption of carbon dioxide from the atmosphere may have significant impacts on marine food webs. This is because acidification reduces the availability of calcium ions for the formation of calcium carbonate, an essential component of the skeletons of marine plankton, shellfish, and other organisms (Doney et al. 2012, Feely et al. 2008). In the Pacific Northwest, which includes Oregon and Washington, projected marine changes include increasing but variable acidity, more increases in surface water temperature, and possible changes in storminess (Melillo et al. 2014). In Puget Sound, changes in the timing and amount of freshwater inflow may produce fresher waters during winter and saltier waters during summer, resulting in stronger stratification in winter and weaker stratification in the summer (Rucklehaus and McClure 2007).

Although physical changes to the marine environment appear likely, much remains to be learned about the magnitude, geographic extent, and temporal and spatial patterns of change, and their effects on murrelets (USFWS 2009). However, we do know that climate variability can strongly influence the foraging and reproductive success of seabirds, including the murrelet (Becker et al. 2007, Grémillet and Boulinier 2009, Norris et al. 2007). Shifts in the intensity of upwelling influence nutrient availability and primary productivity in coastal waters, with cascading effects at higher trophic levels (Thayer and Sydeman 2007). For example, El Niño events have been associated with poor seabird survival and recruitment in the eastern Pacific (Bertram et al. 2005, Hodder and Graybill 1985). Some species respond more strongly to either the ENSO or PDO phases, but not both (Black et al. 2011, Sydeman et al. 2009), and the local effect of regional patterns such as the ENSO and PDO is modified by undersea topography, trophic interactions, bird movements to track prey, and food web impacts from commercial fisheries harvest (Doney et al. 2012). Although many seabirds have flexible foraging strategies, chronic food scarcity can compromise long-term breeding success (Cury et al. 2011) and reduce adult survival and fecundity (Kitaysky et al. 2010).

With respect to foraging strategies, Lorenz et al. (2017) reported on marbled murrelet movements during the breeding season, based on the radio-tracking of 157 birds between 2004 and 2008 in northwestern Washington. The authors did not find oceanographic conditions to substantively explain variation in movements of foraging murrelets. They did find low breeding propensity, large marine ranges, and long nest-sea commutes compared to studies elsewhere in the murrelet's range, and hypothesized that this may indicate that marine habitat in their study area was lower quality compared to elsewhere in the species' range. They also found, unexpectedly, that a recent widespread and strong delay of the onset of spring upwelling in the California Current in 2005 did not appear to substantially affect murrelet movements or breeding propensity. This finding differs from that of Ronconi and Burger (2008), who linked reduced murrelet breeding productivity in southwestern British Columbia to the 2005 upwelling delay.

If recent warm-water events are an indicator of future effects of increased sea-surface temperatures, the murrelet prey base could be negatively affected. Studies of other diving seabirds such as Cassin's auklets (Sydeman et al. 2006), historical versus recent murrelet diet (Becker and Beissinger 2006), and recent annual variations in murrelet reproductive success (Becker et al. 2007) suggest that warmer coastal waters tend to adversely affect prey quality and result in lowered reproduction.

Research Needs, Uncertainties, Information Gaps, and Limitations

The challenges of accurately sampling such a mobile and patchily distributed species result in fairly large uncertainty around each year's density and population estimates, as seen in the confidence intervals. The NWFP population monitoring data provide 15 years from which to assess population trends and, based on the observed sampling error, power analysis indicates that 15 or more years of population estimates are required to detect an annual rate of decline of 2 percent (Falxa et al. 2016). Even with these constraints, the population monitoring data for 2000 through 2015 indicate a marked decline in Washington, no evidence of a trend in Oregon, and an increasing trend in California. Additional years of population monitoring will increase the power to detect ongoing trends, such as those of 2 percent or less per year. Conversely, population trajectories can change over longer monitoring periods, resulting in nonlinear trends, which adds temporal variability and complexity in describing trends.

A major source of uncertainty is whether the murrelet population is closed or open. That is, existing population models (such as McShane et al. 2004) assume there is little or no recruitment of adults or juveniles from outside the study population, and little or no emigration out of the study population. For example, the local population may be declining but is being supplemented by immigrants, perhaps from Alaska or British Columbia, where murrelets are more numerous. Recruitment of birds from outside the local range has been proposed as the most likely explanation for the seemingly stable population estimates in central California (Peery et al. 2006a), despite demographic models that predict a decline (Peery 2004). The open population hypothesis, at least for their range from southern Alaska through northern California, is supported by genetic analyses (Piatt et al. 2007), and recent studies showing long-distance movements of murrelets tracked by satellite (e.g., Bertram et al. 2015b). However, it is not known if movements of murrelets are sufficient to affect population estimates and trends within the NWFP area.

Future population trends are difficult to predict because of uncertainties in the timing and extent of risk factors. Catastrophic loss of nesting habitat from uncharacteristically severe wildfire is an ever-present risk. Among factors other than habitat loss, murrelets at sea are subject to risk from large oil spills at sea (USFWS 1997); oil spills killed an estimated 872 to 2,024 murrelets between 1977 and 2008 in California, Oregon, and Washington (USFWS 2009). A recent review concluded that spills continue to be a threat and can cause severe localized impacts owing to direct mortality from oiling, as well as reductions in reproductive success through changes in prey base, marine habitat, and disturbance (USFWS 2009). Gill net mortality was cited as a factor for listing the murrelet in 1992. Since then, this risk has been substantially reduced in the NWFP area, with no mortality in California and Oregon because of gill net bans, and reduced mortality in Washington as a result of measures implemented to reduce seabird mortality (McShane et al. 2004, USFWS 2009). Gill net mortality remains a threat to the north in British Columbia and Alaska, however, and could be a risk to the NWFP murrelet population to the extent that murrelets move between waters off the Plan area and marine waters to the north. Future energy development, both at sea and on land, could also pose a local threat to murrelets, such as potential collisions with wind turbines (USFWS 2009).

Changes in prey base present risks as well. As discussed earlier, studies have found evidence that murrelet reproductive success is influenced by prey availability, and future prey resources can be affected by fishing as well as changes in ocean conditions, including those linked to climate change. In some other seabird species (e.g., Montevecchi and Myers 1997), changes in ocean currents can have profound effects on forage fish, leading to starvation in addition to breeding inhibition. For murrelets, one study found that adult survival appears less vulnerable to prey shortages (Ronconi and Burger 2008). To date, disease has not been found to be a significant threat to murrelets (Piatt et al. 2007, USFWS 2009), but pathogens new to the region could cause direct mortality to nesting birds, and could also have indirect effects (USFWS 2009). For example, the West Nile virus is documented to kill jays, crows, and ravens, and if mortality of these species resulted in appreciable reductions in their densities, this might increase nest success of murrelets by reducing nest depredation.

Raphael et al. (2016a) describe sources of uncertainty in estimating the amount and distribution of nesting habitat of the murrelet. But one additional source warrants further mention. Because murrelet nesting behavior is so cryptic, biologists have found very few actual nests of the species. To supplement actual nesting observations, biologists rely on locations of "occupied behaviors" to infer nesting activity. Occupied behaviors are observations of murrelets flying into the canopy, circling very close above the canopy, or landing in trees. These behaviors are typically associated with nesting, but some sites where occupied behaviors are observed may not be true nest sites. To the extent that false positives may be included in the murrelet database used to build models, these models may be less accurate than if all locations were based on verified nests (Plissner et al. 2015). A more reliable modeling solution would be to conduct intensive research to identify more known nest sites across a broad sampling of regions within the NWFP area, then build models exclusively from training sites that represent actual murrelet nests. Such intensive surveys would also help our understanding of spacing and density of nesting activity in relation to forest stand characteristics.

Some uncertainty also exists in the distance that murrelets fly inland to nest and how that varies within the Plan area. The Forest Ecosystem Management Assessment Team designated two inland zones within the area in which murrelets nest: Inland zone 1 formed the area closer to the marine environment, and inland zone 2 was further inland, extending to the eastern boundary of the species' nesting range (see fig. 5-4). Nesting was assumed to occur mostly in zone 1. Recent survey-based studies in some areas have led to local contractions of zone 2, especially in northern California and southern Oregon (Alegria et al. 2002, Hunter et al. 1998, Schmidt et al. 2000). Agencies in those areas have redefined the eastern boundary of the area in which surveys for murrelets are required prior to timber harvest, bringing it farther to the west to match study results. This revised boundary has not been formally implemented in the NWFP agency maps; to date this revision applies only to survey requirements for management units where the studies were conducted. This strategy adds uncertainty in the calculation of nesting amounts of nesting habitat to the extent that acres classified as nesting habitat may actually fall outside the species' true breeding range. This uncertainty is reduced in the most recent analysis by the NWFP monitoring program, which did not model or estimate suitable murrelet nesting habitat in inland zone 2 in California or Oregon; this is because of the lack of inland zone nest sites in those states with which to train the nest habitat models (Raphael et al. 2016a).

We found no studies documenting the response of murrelets to silvicultural activities designed to accelerate expression of mature forest conditions, and this remains an area in which much further research is needed. Foresters have conducted studies using experimental thinning prescriptions, but none of these has incorporated responses of murrelets to these treatments.

Perhaps the most important area of uncertainty is the relationship between murrelet population size and trend and the influences of either amount and trend of nesting habitat versus variation and trends in ocean conditions that affect foraging habitat. The studies that we summarize point toward nesting habitat as the primary driver, but all these studies concede that relationships are correlational. Cause-effect relationships have not been established, so further work will be needed to confirm whether these correlations reflect true underlying causes.

Conclusions and Management Considerations

Are NWFP Assumptions Still Valid?

Nesting habitat status and trend—
The NWFP has played a pivotal role in the fate of murrelet nesting habitat on federal lands. The Plan has been highly successful in conserving existing murrelet nesting habitat, and little nesting habitat has been lost to timber harvest on federal lands. Some loss of nesting habitat, especially in federal reserves, was caused by fire. Loss of murrelet nesting habitat to catastrophic events will always be a risk, and such losses were expected. The NWFP has less control over the risk of such losses, except to the extent that active management in fire-prone areas might reduce the risk of fire in younger forests in proximity to murrelet nesting habitat, and by reducing vegetation that could transmit fire to the canopy of murrelet nesting trees, such as in forests with scattered nest trees within younger forest. One caution should be recognized: managing forest cover to reduce fire risk could also lead to better habitat for corvids (nest predators); silvicultural practices near suitable murrelet nesting habitat may need to be fine-tuned to ensure they do not inadvertently impair nesting success of murrelets by increasing the rate of nest depredation. In addition to active fire management, another area for potential reduction of nesting habitat loss on federal lands is management to reduce the risk of windthrow associated with the creation of hard edges. In this case, the greatest potential benefit to murrelets would be in (1) creating and maintaining forested buffers adjacent to existing known and suitable murrelet nesting habitat, and (2) developing nesting habitat within reserves plus in adjacent buffers.

The fate of nesting habitat on nonfederal lands is beyond the scope of the NWFP; 67 percent of habitat-capable forest is in nonfederal ownership, as is 41 percent of suitable murrelet nesting habitat. The rate of loss of suitable nesting habitat on nonfederal lands (1.5 percent per year) has been far more rapid than on federal lands (0.1 percent per year).

The requirement for preproject surveys on federal land was assumed to prevent the loss of any occupied sites from timber harvest. We are not able to test this assumption because we have no way to assess whether sites on federal land were classified as unoccupied when they might actually have been occupied. Occupied behaviors are not observed at every visit to a site; a finite likelihood exists of failing to detect occupied behaviors even if the site is occupied. The protocol used to determine site occupancy (Evans Mack et al. 2003) sets the numbers of visits required to have a high likelihood (set at 0.95) of observing occupied behavior at an occupied site. Under this protocol, a 5-percent chance of failing to detect occupied behavior exists, so a small number of sites might be mistakenly classified as unoccupied and released for timber harvest. The Pacific Seabird Group (a society of professional seabird researchers and managers dedicated to the study and conservation of seabirds) is considering a revision of the current survey protocol (Evans Mack et al. 2003), which would use the best available science to ensure that the 5 percent criterion is met by the protocol. We can say that sites classified as occupied were, in fact, set aside and managed as LSR3 reserves. There apparently have been some differences among NWFP management units in applying the NWFP standards and guidelines to occupied sites, with some reserves including all forest within a 0.5-mi (0.8 km) radius (which provides a larger block and more protection), and others including only contiguous forest within the radius that is existing suitable or recruitment murrelet nesting habitat (USDI BLM 2016).

Population status and trends—
Murrelet populations are affected by a variety of factors, only some of which are under the NWFP's direct influence. The Plan most directly affects populations through its provisions for conservation and restoration of nesting habitat, but even then its influence extends only to federal lands. Although NWFP forest management may have minor or local effects on marine habitats, such as through altered input of sediment and coarse wood, overall the Plan has little to no influence on marine conditions affecting murrelet populations (including marine food sources) or on sources of mortality at sea, such as oil spills and gillnetting. This makes it more difficult to relate changes in murrelet populations to land management under the NWFP. With the NWFP conserving nesting habitat as expected, murrelet populations could still fall because of adverse marine

conditions or because of nesting habitat loss on nonfederal lands. Despite this uncertainty, circumstantial evidence suggests that inland nesting habitat conditions are the major driver setting murrelet population size at this time. This point is illustrated in Raphael et al. (2016b), in which the authors found a positive correlation with the total amount of nesting habitat and size of adjacent murrelet population for segments of the murrelet range. In addition, Raphael et al. (2015, 2016b) constructed a model to assess relative contribution of marine and terrestrial habitat attributes toward abundance and trend of murrelets throughout their range in Washington, Oregon, and California south to San Francisco Bay. In that model, amount and pattern of nesting habitat made the strongest contribution to predictions of spatial distribution and temporal trends of murrelet populations at sea; marine factors such as sea surface temperature and chlorophyll, as well as ENSO and PDO indices, had little effect. Murrelet nesting habitat seems to be the primary driver of murrelet population status and trend, at least in recent decades, but that relationship has not been tested empirically and a cause-effect relationship has not been established. Raphael et al (2016b) suggest that one test of this relationship will be whether murrelet populations are observed to increase when the net amount of suitable nesting habitat increases at some point in the future.

The fundamental assumptions of the NWFP were that the rate of loss of murrelet nesting habitat in reserves would slow or stop, and that unsuitable forest cover types would recover. Available data support this assumption and show that rates of loss on NWFP lands are low, and that forest stands in reserves are on a trajectory toward higher nesting habitat suitability. Conservation and restoration of murrelet nesting habitat are essential to population viability of the species.

Although federal protection of nesting habitat is essential to murrelet viability, it may not be sufficient given the cumulative effects of other influences on population viability. Research has documented that murrelet viability depends on a variety of factors, many of which (e.g., supply of ocean prey) are not under the control or influence of the NWFP. Nesting habitat loss on nonfederal lands, marine conditions, and threats from disease, oil spills, and gillnetting could reduce the likelihood of population viability despite the habitat protections built into the NWFP. Past timber harvest was hypothesized to have lingering effects on murrelet carrying capacity and nesting success. We are aware of no new data to challenge this hypothesis. Recent research shows that murrelet population size is reduced as nesting habitat is lost, and that birds do not pack into remaining suitable nesting habitat (Burger 2001, Raphael et al. 2002a).

A major premise of the NWFP is that large reserves will support more murrelets, eventually leading to stationary or increasing populations. Because of the long period of time required to recruit new nesting habitat in reserves, thus forming larger blocks of nesting habitat, it is too soon to fully evaluate this premise, but trends on Forest Service lands in the Oregon Coast Range suggest that this may be starting to occur there.

Fahrig (1997) suggested that habitat loss tends to far outweigh the spatial configuration of habitat (fragmentation) as a risk to species. Although habitat loss and limitation appear to best explain the observed patterns of murrelet distribution and population trends in the Plan area, spatial configuration of nesting habitat is also a factor. As discussed above (see “Landscape-level relationships between nesting habitat and populations”), fragmentation of nesting habitat and the associated greater amounts of habitat edge may increase the risk of breeding failure due to nest predation.

Also, as summarized above, nest depredation seems to be a major limiting factor on murrelet populations, and nesting habitat configuration may affect predation risk. More than half of known murrelet nests whose fate has been determined failed because eggs or chicks were lost to predators, primarily jays, crows, and ravens (Manley and Nelson 1999, and other papers cited above). The relationship of predation risk and forest configuration appears to be complex. Increased edge resulting from forest fragmentation appears to have negative effects on murrelets. For example, some research has found higher densities of nest predators near edges (primarily jays), particularly where edges are near human development such as campgrounds (Goldenberg 2013, Marzluff and Neatherlin 2006) or include berry-producing plants (Masselink 2001). Other research suggests that predator numbers are high in old-growth forests with complex forest structure, such as those expected to develop in NWFP reserves, but lower in mature forests with simpler structure (Marzluff et al. 2000, Raphael et al. 2002b). At the plot scale (90 to 260 ac), one study found predator densities higher and nest success lower in plots with a variety of tree ages intermixed with young tree/brush habitats (Luginbuhl et al. 2001). The relationship between nest predator density and predation risk may also depend on the scale of observation. Luginbuhl and others (2001) found that nest predation risk was much better predicted by corvid abundance at the landscape level (2 to 20 mi^2 [5 to 52 km^2] scale) than at a finer scale (60 to 120 ac [24 to 49 km^2]), likely because of the large home range of some corvids (ravens, crows).

Forest fragmentation will decline as young patches within reserves mature, creating more contiguous canopy cover, and where rates of nest predation would decrease as forests became less fragmented. Murrelet populations may not grow at the rate predicted from recovery of nesting habitat in reserves because nest depredation could suppress successful reproduction. We lack understanding of the full suite of factors that affect nest success, which increases uncertainty about the relations between amounts of nesting habitat and murrelet populations.

Research indicates that maintaining older, maturing forest adjacent to nesting habitat also reduces predation risk (table 5-6). Taken as a whole, research to date suggests that, apart from increasing the amount of nesting habitat and reducing its fragmentation, managing forest structure to reduce nest predation risk should be approached with consideration of local factors that might affect predator densities (e.g., overstory thinning that might result in increased abundance of berry-producing early-seral shrubs that attract corvids).

Although habitat loss and fragmentation lead as factors influencing murrelet numbers and trends, birds in the NWFP area are also affected by marine factors. Murrelets are subject to risk from large oil spills at sea, which killed an estimated 872 to 2,024 murrelets between 1977 and 2008 in California, Oregon, and Washington, and continue to be a threat, as they can cause severe localized impacts such as direct mortality through oiling, as well as other less direct effects (USFWS 2009). Gill net mortality in the Plan area has been reduced substantially since 1994, with California and Oregon banning gill net use near shore, and measures taken in Washington to reduce seabird mortality (McShane et al. 2004, USFWS 2009). As discussed above ("Marine Habitat," "Changes in foraging habitat conditions," and "Climate Change Considerations"), murrelet reproductive success is influenced by prey quality and availability, which can be affected by fishing as well as changes in ocean conditions, including those linked to climate change. Future energy development, both at sea and on land, could also pose a local threat to murrelets, such as potential collisions with wind turbines (USFWS 2009).

Cumulative effects—

Wildlife population trends reflect the cumulative effects of multiple interacting factors. Nesting habitat conditions on federal lands are but one of those factors, albeit the one over which the NWFP has the most direct influence. Monitoring both nesting habitat trends and population trends is of value: monitoring nesting habitat trends tells managers how well the Plan is meeting its primary objectives; monitoring population trends tells managers if the NWFP is having the desired effects. Ideally, population trends will track nesting habitat trends, but we may observe diverging trends. In such cases, we can dig deeper to discover whether our understanding of nesting habitat relationships is mistaken or whether other, perhaps unmeasured, factors are driving population trends. Research to date, as noted above, does support the idea that population trends track nesting habitat trends, but the evidence is still based on correlations and has not established cause-effect relationships.

Carrying capacity is a measure of the potential population size that can be supported by a given amount and distribution of suitable nesting habitat. The actual population may be lower than the carrying capacity owing to a variety of other factors such as hostile weather, interactions with other species, nesting habitat conditions outside of the planning area, disease, or other factors that might depress a population. Observing a declining population in the face of habitat conservation does not mean that habitat is not important or that habitat conservation is not important. It means we have to look at options to manage some of the other factors that might be driving the population trend. Until we have more robust models of wildlife habitat relationships, which include these other factors, continued monitoring of both population and habitat trends will be important to evaluate how well the NWFP is meeting its intended objectives.

Efficacy of large reserves for murrelet conservation— A central tenet of the NWFP was that the system of large, late-successional reserves would largely suffice to provide for species and biodiversity components associated with late-successional and old-growth forest ecosystems. We have found that, to an extent, this is true with respect to murrelets. However, the degree to which late-successional reserves—along with the set of other NWFP land allocations (e.g., riparian reserves in matrix lands)—suffice differs considerably by species. Our review has highlighted the importance of large contiguous blocks of nesting habitat in meeting the nesting needs of the murrelet, and reserves seem an essential way to create such landscapes.

One of the management dilemmas is that optimal habitat conditions differ among species. Creating shrubby foraging habitat will be good for the northern spotted owl in the southern parts of its range, but such habitat will also be good for jays and crows, which depredate nests of the murrelet. In this case, what is good for the owl may be bad for the murrelet (see chapter 12 for further discussion of interactions among NWFP goals and objectives).

Management Considerations

Some key points emerge from this synthesis:

- Maintaining and increasing the area and cohesion (creating larger blocks) of suitable nesting-habitat area on federal lands will likely contribute to stabilizing and eventually recovering murrelet populations. Within NWFP lands, the current NWFP reserve system (including riparian buffers and other set-asides) appears well designed to accomplish this. Because it can take many decades for murrelet nesting habitat to develop, protection of existing habitat for the next several decades will continue to be key to minimizing habitat losses, both within and outside of reserves.
- Defining the inland limit of the murrelet nesting range will require additional survey work and a synthesis of existing observations. A refined range will better meet management objectives and avoid problems with managing for murrelets in areas where none are really expected to exist.
- Conservation of existing nesting habitat on federal lands may not be sufficient to conserve murrelet populations in the short term. Contributions from nonfederal lands may help the NWFP or its successor to achieve objectives for the murrelet, and the larger goal of murrelet conservation and recovery. This might be approached by collaborative programs to increase murrelet conservation on nonfederal lands, particularly those adjacent to NWFP lands, and in key areas (such as southwest Washington and northwest Oregon) where few federal reserves exist.
- Restoration of old-forest/murrelet nesting habitat in reserves may be accelerated by active management toward that end. Active management actions could include thinning in plantations to accelerate growth of potential nest trees and development of nesting platforms, but care will be needed to prevent simultaneously increasing numbers of nest predators attracted to more diverse understory conditions. Moreover, such management should also be careful to not increase the suitability of older forests to harbor barred owls (*Strix varia*), which may prey on murrelets and also reduce forest suitability for northern spotted owls (see chapter 4). Development and implementation of forest management practices that protect (short term) and develop (long term, e.g., over many decades) suitable murrelet nesting habitat on NWFP lands within the murrelet range would be beneficial in recovering murrelet populations (see chapter 3 for examples of restoration treatments).
- To guide management and increase its effectiveness in achieving nesting habitat expansion, modeling tools are needed to help forecast site-specific future nesting habitat development and structural characteristics of potential murrelet nesting habitat.

- Restoration in plantations and younger natural forests can benefit murrelets by incorporating an understanding of relations among stand shape, extent of higher-contrast edges, and populations of potential nest predators, including corvids. Proximity of nest and occupied habitat should be considered. Treatments that consider risk to existing suitable nesting habitat along exposed edges from windthrow would also contribute to conservation of existing nesting habitat.
- Forest planning and management can positively affect murrelet status by managing human recreation activities that might promote murrelet nest predator populations (e.g., ravens, crows, and jays in campgrounds). The greatest benefit would be expected in areas within and near existing and developing murrelet nesting habitat. Implementing education programs, limiting garbage, and controlling predators could have positive effects.
- Future management and design of reserves will benefit from accounting for climate change, including increased risks to murrelet nesting habitat from fire and other natural disturbances. Boundaries of reserves (including making them larger) may be reconsidered if revised boundaries might better conserve nesting habitat in the face of anticipated effects of climate change.
- Maintaining a broad distribution of large nesting habitat blocks over the NWFP landscape will likely help to minimizing the risk to the population from nesting habitat loss to fire, wind or other disturbance agents.

The NWFP remains the boldest effort ever undertaken by federal agencies to meet large-scale biodiversity objectives. The Plan had a short-term objective for murrelets: conserve much of the best remaining nesting habitat. The NWFP has been very successful in meeting this objective. The NWFP also has a long-term objective: create a system of reserves containing desired sizes and distributions of large blocks for suitable nesting habitat. Evidence suggests that nesting habitat trends on federal lands are on course toward this objective, but many more decades will be needed to observe whether the Plan is successful in achieving its goal to stabilize and increase murrelet populations by maintaining and increasing nesting habitat. We have shown that the NWFP has been remarkably successful in conserving nesting habitat over its first 20 years of implementation, but much work remains. Murrelet numbers continue to decline in the northern portion of the Plan area. Assuming no large fires, we believe that the current decline in amount of murrelet nesting habitat will reverse on federal lands, leading to a net increase in the amount of nesting habitat, and that murrelet populations should also increase in response. How many decades before this reversal in trend occurs is unknown, but at-sea monitoring suggests that the first step of possible population stabilization may be occurring in the southern Plan area. Lastly, climate change has emerged as an external force that may affect future murrelet populations, their nesting habitat, and, in particular, food resources for murrelets.

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Metric Equivalents

When you know:	Multiply by:	To find:
Feet (ft)	0.3049	Meters
Miles (mi)	1.61	Kilometers
Acres (ac)	0.4049	Hectares
Square miles (mi^2)	2.59	Square kilometers

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Scientific and common names of plant species identified in this report

Scientific name	Common name
Abies amabilis (Douglas ex Loudon) Douglas ex Forbes	Pacific silver fir
Abies concolor (Gord. & Glend.) Lindl. ex Hildebr.	White fir
Abies grandis (Douglas ex D. Don) Lindl.	Grand fir
Abies lasiocarpa (Hook.) Nutt.	Subalpine pine
Abies magnifica A. Murray bis	California red fir
Abies procera Rehder	Noble fir
Acer circinatum Pursh	Vine maple
Acer macrophyllum Pursh	Bigleaf maple
Achlys triphylla (Sm.) DC.	Sweet after death
Adenocaulon bicolor Hook.	American trailplant
Alliaria petiolata (M. Bieb.) Cavara & Grande	Garlic mustard
Alnus rubra Bong.	Red alder
Amelanchier alnifolia (Nutt.) Nutt. ex M. Roem.	Saskatoon serviceberry
Anemone oregana A. Gray	Blue windflower
Apocynum cannabinum L.	Dogbane
Arbutus menziesii Pursh)	Madrone
Arceuthobium M. Bieb.	Dwarf mistletoe
Arceuthobium occidentale Engelm.	Gray pine dwarf mistletoe
Arceuthobium tsugense Rosendahl	Hemlock dwarf mistletoe
Arctostaphylos nevadensis A. Gray	Pinemat manzanita
Brachypodium sylvaticum (Huds.) P. Beauv.	False brome
Brodiaea coronaria (Salisb.) Engl.	Cluster-lilies
Callitropsis nootkatensis (D. Don) Oerst. ex D.P. Little	Alaska yellow-cedar
Calocedrus decurrens (Torr.) Florin	Incense cedar
Cannabis L.	Marijuana
Carex barbarae Dewey and *C. obnupta* L.H. Bailey	Sedges
Centaurea solstitialis L.	Yellow starthistle
Chamaecyparis lawsoniana (A. Murray bis) Parl.	Port Orford cedar
Chimaphila menziesii (R. Br. ex D. Don) Spreng.	Little prince's pine
Chimaphila umbellata (L.) W.P.C. Barton	Pipsissewa
Clematis vitalba L.	Old man's beard
Clintonia uniflora Menzies ex Schult. & Schult. f.) Kunth	Bride's bonnet
Coptis laciniata A. Gray	Oregon goldthread
Corylus cornuta Marshall var. *californica* (A. DC.) Sharp	California hazel
Cornus canadensis L.	Bunchberry dogwood
Cytisus scoparius (L.) Link	Scotch broom
Disporum hookeri (Torr.) G. Nicholson var. *hookeri*	Drops-of-gold
Fallopia japonica (Houtt.) Ronse Decr. var. *japonica*	Japanese knotweed
Gaultheria ovatifolia A. Gray	Western teaberry
Gaultheria shallon Pursh	Salal

Scientific name	Common name
Gentiana douglasiana Bong.	Swamp gentian
Geranium lucidum L.	Shining geranium
Geranium robertianum L.	Robert geranium
Goodyera oblongifolia Raf.	Western rattlesnake plantain
Hedera helix L.	English ivy
Heracleum mantegazzianum Sommier & Levier	Giant hogweed
Hesperocyparis sargentii (Jeps.) Bartel	Sargent’s cypress
Hieracium aurantiacum L.	Orange hawkweed
Ilex aquifolium L.	English holly
Iris pseudacorus L.	Paleyellow iris
Juniperus occidentalis Hook.	Western juniper
Lamiastrum galeobdolon (L.) Ehrend. & Polatschek	Yellow archangel
Lilium occidentale Purdy	Western lily
Linnaea borealis L.	Twinflower
Lithocarpus densiflorus (Hook. & Arn.) Rehder	Tanoak
Lonicera hispidula Pursh	Honeysuckle
Lupinus albicaulis Douglas	Sickle-keeled lupine
Lycopodium clavatum L.	Running clubmoss
Lythrum salicaria L.	Purple loosestrife
Mahonia nervosa (Pursh) Nutt.	Cascade barberry
Malus fusca (Raf.) C.K. Schneid.	Pacific crabapple
Notholithocarpus densiflorus (Hook. & Arn.) P.S. Manos, C.H. Cannon, & S.H. Oh	Tanoak
Notholithocarpus densiflorus (Hook. & Arn.) P.S. Manos, C.H. Cannon, & S.H. Oh var. *echinoides* (R.Br. ter) P.S. Manos, C.H. Cannon & S.H. Oh	Shrub form of tanoak
Nuphar polysepala (Engelm.)	Yellow pond lily
Nymphoides peltata (S.G. Gmel.) Kuntze	Yellow floating heart
Osmorhiza chilensis Hook. & Arn.	Sweetcicely
Phalaris arundinacea L.	Reed canarygrass
Picea engelmannii Parry ex Engelm.	Engelmann spruce
Picea sitchensis (Bong.) Carrière	Sitka spruce
Pinus albicaulis Engelm.	Whitebark pine
Pinus attenuata Lemmon	Knobcone pine
Pinus contorta Douglas ex Loudon	Lodgepole pine
Pinus contorta Douglas ex Loudon var. *contorta*	Beach pine, shore pine
Pinus jeffreyi Balf.	Jeffrey pine
Pinus lambertiana Douglas	Sugar pine
Pinus monticola Douglas ex D. Don)	Western white pine
Pinus ponderosa Lawson & C. Lawson	Ponderosa pine
Populus trichocarpa L. ssp. *trichocarpa* (Torr. & A. Gray ex Hook) Brayshaw	Black cottonwood
Potamogeton crispus L.	Curly pondweed
Potentilla recta L.	Sulphur cinquefoil

Scientific name	Common name
Prunus emarginata (Douglas ex Hook. D. Dietr.)	Bitter cherry
Pseudotsuga menziesii (Mirb.) Franco	Douglas-fir
Pteridium aquilinum (L. Kuhn)	Brackenfern
Pueraria montana (Lour.) Merr. var. *lobata* (Willd.) Maesen & S.M. Almeida ex Sanjappa & Predeep	Kudzu
Pyrola asarifolia Sweet	American wintergreen
Quercus agrifolia Née var. *oxyadenia* (Torr.) J.T. Howell	Coastal live oak
Quercus berberidifolia Liebm.	Scrub oak
Quercus chrysolepis Liebm.	Canyon live oak
Quercus douglasii Hook. & Arn.	Blue oak
Quercus garryana Douglas ex hook.	Oregon white oak
Quercus kelloggi Newberry	California black oak
Quercus lobata Née	Valley oak
Rhamnus purshiana (DC.) A. Gray	Cascara
Rhododendron groenlandicum Oeder	Bog Labrador tea
Rhododendron macrophyllum D. Don ex G. Don	Pacific rhododendron
Ribes lacustre (Pers.) Poir.	Prickly currant
Rubus armeniacus Focke	Himalayan blackberry
Salix exigua Nutt.	Sandbar willow
Senecio bolanderi A. Gray	Bolander’s ragwort
Sequoia sempervirens (Lamb. ex D. Don) Endl.	Redwood
Smilacina stellata (L.) Desf.	Starry false Solomon’s seal
Synthyris reniformis (Douglas ex Benth.) Benth.	Snowqueen
Taxus brevifolia Nutt.	Pacific yew
Thuja plicata Donn ex D. Don	Western redcedar
Tiarella trifoliate L.	Threeleaf foamflower
Trapa natans L.	Water chestnut
Trillium ovatum Pursh	Pacific trillium
Tsuga heterophylla (Raf.) Sarg.	Western hemlock
Tsuga mertensiana (Bong.) Carrière	Mountain hemlock
Typha latifolia L.	Cattails
Umbellularia californica (Hook. & Arn.) Nutt.	California bay laurel
Vaccinium alaskaense Howell	Alaska blueberry
Vaccinium membranaceum Douglas ex Torr.	Thinleaf huckleberry, big huckleberry
Vaccinium ovatum Pursh	Evergreen huckleberry
Vaccinium oxycoccos L.	Small cranberry
Vaccinium parvifolium Sm.	Red huckleberry
Vancouveria hexandra (Hook.) C. Morren & Decne.	White insideout flower
Xerophyllum tenax (Pursh) Nutt.	Beargrass

Glossary

This glossary is provided to help readers understand various terms used in the Northwest Forest Plan (NWFP) science synthesis. Sources include the Forest Service Handbook (FSH), the Code of Federal Regulations (CFR), executive orders, the Federal Register (FR), and various scientific publications (see "Glossary Literature Cited"). The authors have added working definitions of terms used in the synthesis and its source materials, especially when formal definitions may be lacking or when they differ across sources.

active management—Direct interventions to achieve desired outcomes, which may include harvesting and planting of vegetation and the intentional use of fire, among other activities (Carey 2003).

adaptive capacity—The ability of ecosystems and social systems to respond to, cope with, or adapt to disturbances and stressors, including environmental change, to maintain options for future generations (FSH 1909.12.5).

adaptive management—A structured, cyclical process for planning and decisionmaking in the face of uncertainty and changing conditions with feedback from monitoring, which includes using the planning process to actively test assumptions, track relevant conditions over time, and measure management effectiveness (FSH 1909.12.5). Additionally, adaptive management includes iterative decisionmaking, through which results are evaluated and actions are adjusted based on what has been learned.

adaptive management area (AMA)—A portion of the federal land area within the NWFP area that was specifically allocated for scientific monitoring and research to explore new forestry methods and other activities related to meeting the goals and objectives of the Plan. Ten AMAs were established in the NWFP area, covering about 1.5 million ac (600 000 ha), or 6 percent of the planning area (Stankey et al. 2003).

alien species—Any species, including its seeds, eggs, spores, or other biological material capable of propagating that species, that is not native to a particular ecosystem (Executive Order 13112). The term is synonymous with exotic species, nonindigenous, and nonnative species (see also "invasive species").

allochthonous inputs—Material, specifically food resources, that originates from outside a stream, typically in the form of leaf litter.

amenity communities—Communities located near lands with high amenity values.

amenity migration—Movement of people based on the draw of natural or cultural amenities (Gosnell and Abrams 2011).

amenity value—A noncommodity or "unpriced" value of a place or environment, typically encompassing aesthetic, social, cultural, and recreational values.

ancestral lands (of American Indian tribes)—Lands that historically were inhabited by the ancestors of American Indian tribes.

annual species review—A procedure established under the NWFP in which panels of managers and biologists evaluate new scientific and monitoring information on species to potentially support the recommendation of changes in their conservation status.

Anthropocene—The current period (or geological epoch) in which humans have become a dominant influence on the Earth's climate and environment, generally dating from the period of rapid growth in industrialization, population, and global trade and transportation in the early 1800s (Steffen et al. 2007).

Aquatic Conservation Strategy (ACS) —A regional strategy applied to aquatic and riparian ecosystems across the area covered by the NWFP) (Espy and Babbit 1994) (see chapter 7 for more details).

at-risk species—Federally recognized threatened, endangered, proposed, and candidate species and species of conservation concern. These species are considered at risk of low viability as a result of changing environmental conditions or human-caused stressors.

best management practices (BMPs) (for water quality)—Methods, measures, or practices used to reduce or eliminate the introduction of pollutants and other detrimental impacts to water quality, including but not limited to structural and nonstructural controls and to operation and maintenance procedures.

biodiversity—In general, the variety of life forms and their processes and ecological functions, at all levels of biological organization from genes to populations, species, assemblages, communities, and ecosystems.

breeding inhibition—Prevention of reproduction in healthy adult individuals.

bryophytes—Mosses and liverworts.

canopy cover—The downward vertical projection from the outside profile of the canopy (crown) of a plant measured in percentage of land area covered.

carrying capacity—The maximum population size a specific environment can sustain.

ceded areas—Lands that particular tribes ceded to the United States government by treaties, which have been catalogued in the Library of Congress.

climate adaptation—Management actions to reduce vulnerabilities to climate change and related disturbances.

climate change—Changes in average weather conditions (including temperature, precipitation, and risk of certain types of severe weather events) that persist over multiple decades or longer, and that result from both natural factors and human activities such as increased emissions of greenhouse gases (U.S. Global Change Research Program 2017).

coarse filter—A conservation approach that focuses on conserving ecosystems, in contrast to a "fine filter" approach that focuses on conserving specific species. These two approaches are generally viewed as complementary, with fine-filtered strategies tailored to fit particular species that "fall through the pores" of the coarse filter (Hunter 2005). See also "mesofilter."

co-management—Two or more entities, each having legally established management responsibilities, working collaboratively to achieve mutually agreed upon, compatible objectives to protect, conserve, use, enhance, or restore natural and cultural resources (81 FR 4638).

collaborative management—Two or more entities working together to actively protect, conserve, use, enhance, or restore natural and cultural resources (81 FR 4638).

collaboration or collaborative process—A structured manner in which a collection of people with diverse interests share knowledge, ideas, and resources, while working together in an inclusive and cooperative manner toward a common purpose (FSH 1909.12.05).

community (plant and animal)—A naturally occurring assemblage of plant and animal species living within a defined area or habitat (36 CFR 219.19).

community forest—A general definition is forest land that is managed by local communities to provide local benefits (Teitelbaum et al. 2006). The federal government has specifically defined community forest as "forest land owned in fee simple by an eligible entity [local government, nonprofit organization, or federally recognized tribe] that provides public access and is managed to provide community benefits pursuant to a community forest plan" (36 CFR 230.2).

community of place or place-based community—A group of people who are bound together because of where they reside, work, visit, or otherwise spend a continuous portion of their time.

community resilience—The capacity of a community to return to its initial function and structure when initially altered under disturbance.

community resistance—The capacity of a community to withstand a disturbance without changing its function and structure.

composition—The biological elements within the various levels of biological organization, from genes and species to communities and ecosystems (FSM 2020).

congeneric—Organisms that belong to the same taxonomic genus, usually belonging to different species.

connectivity (of habitats)—Environmental conditions that exist at several spatial and temporal scales that provide landscape linkages that permit (a) the exchange of flow, sediments, and nutrients; (b) genetic interchange of genes among individuals between populations; and (c) the long-distance range shifts of species, such as in response to climate change (36 CFR 219.19).

consultation (tribal)—A formal government-to-government process that enables American Indian tribes and Alaska Native Corporations to provide meaningful, timely input, and, as appropriate, exchange views, information, and recommendations on proposed policies or actions that may affect their rights or interests prior to a decision. Consultation is a unique form of communication characterized by trust and respect (FSM 1509.05).

corticosterone—A steroid hormone produced by many species of animals, often as the result of stress.

cryptogam—An organism that reproduces by spores and that does not produce true flowers and seeds; includes fungi, algae, lichens, mosses, liverworts, and ferns.

cultural keystone species—A species that significantly shapes the cultural identity of a people, as reflected in diet, materials, medicine, or spiritual practice (Garibaldi and Turner 2004).

cultural services—A type of ecosystem service that includes the nonmaterial benefits that people obtain from ecosystems through spiritual enrichment, cognitive development, reflection, recreation, and aesthetic experiences (Sarukhán and Whyte 2005).

desired conditions—A description of specific social, economic, or ecological characteristics toward which management of the land and resources should be directed.

disturbance regime—A description of the characteristic types of disturbance on a given landscape; the frequency, severity, and size distribution of these characteristic disturbance types and their interactions (36 CFR 219.19).

disturbance—Any relatively discrete event in time that disrupts ecosystem, watershed, community, or species population structure or function, and that changes resources, substrate availability, or the physical environment (36 CFR 219.19).

dynamic reserves—A conservation approach in which protected areas are relocated following changes in environmental conditions, especially owing to disturbance.

early-seral vegetation—Vegetation conditions in the early stages of succession following an event that removes the forest canopy (e.g., timber harvest, wildfire, windstorm), on sites that are capable of developing a closed canopy (Swanson et al. 2014). A nonforest or "pre-forest" condition occurs first, followed by an "early-seral forest" as young shade-intolerant trees form a closed canopy.

ecocultural resources—Valued elements of the biophysical environment, including plants, fungi, wildlife, water, and places, and the social and cultural relationships of people with those elements.

ecological conditions—The biological and physical environment that can affect the diversity of plant and animal communities, the persistence of native species, invasibility, and productive capacity of ecological systems. Ecological conditions include habitat and other influences on species and the environment. Examples of ecological conditions include the abundance and distribution of aquatic and terrestrial habitats, connectivity, roads and other structural developments, human uses, and occurrence of other species (36 CFR 219.19).

ecological forestry—A ecosystem management approach designed to achieve multiple objectives that may include conservation goals and sustainable forest management and which emphasizes disturbance-based management and retention of "legacy" elements such as old trees and dead wood (Franklin et al. 2007).

ecological integrity—The quality or condition of an ecosystem when its dominant ecological characteristics (e.g., 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 (36 CFR 219.19).

ecological keystone species—A species whose ecological functions have extensive and disproportionately large effects on ecosystems relative to its abundance (Power et al. 1996).

ecological sustainability—The capability of ecosystems to maintain ecological integrity (36 CFR 219.19).

economic sustainability—The capability of society to produce and consume or otherwise benefit from goods and services, including contributions to jobs and market and nonmarket benefits (36 CFR 219.19).

ecoregion—A geographic area containing distinctive ecological assemblages, topographic and climatic gradients, and historical land uses.

ecosystem—A spatially explicit, relatively homogeneous unit of the Earth that includes all interacting organisms and elements of the abiotic environment within its boundaries (36 CFR 219.19).

ecosystem diversity—The variety and relative extent of ecosystems (36 CFR 219.19).

ecosystem integrity—See "ecological integrity."

ecosystem management—Management across broad spatial and long temporal scales for a suite of goals, including maintaining populations of multiple species and ecosystem services.

ecosystem services—Benefits that people obtain from ecosystems (see also "provisioning services," "regulating services," "supporting services," and "cultural services").

ectomycorrhizal fungi—Fungal species that form symbiotic relationships with vascular plants through roots, typically aiding their uptake of nutrients. Although other mycorrhizal fungi penetrate their host's cell walls, ectomycorrhizal fungi do not.

endangered species—Any species or subspecies that the Secretary of the Interior or the Secretary of Commerce has deemed in danger of extinction throughout all or a significant portion of its range (16 U.S.C. Section 1532).

endemic—Native and restricted to a specific geographical area.

El Niño Southern Oscillation (ENSO)—A band of anomalously warm ocean water temperatures that occasionally develops off the western coast of South America and can cause climatic changes across the Pacific Ocean. The extremes of this climate pattern's oscillations cause extreme weather (such as floods and droughts) in many regions of the world.

environmental DNA (eDNA)—Genetic material (DNA) contained within small biological and tissue fragments that can be collected from aquatic, terrestrial, and even atmospheric environments, linked to an individual species, and used to indicate the presence of that species.

environmental justice populations—Groups of people who have low incomes or who identify themselves as African American, Asian or Pacific Islander, American Indian or Alaskan Native, or of Hispanic origin.

ephemeral stream—A stream that flows only in direct response to precipitation in the immediate locality (watershed or catchment basin), and whose channel is at all other times above the zone of saturation.

epicormic—Literally, "of a shoot or branch," this term implies growth from a previously dormant bud on the trunk or a limb of a tree.

epiphyte—A plant or plant ally (including mosses and lichens) that grows on the surface of another plant such as a tree, but is not a parasite.

even-aged stand—A stand of trees composed of a single age class (36 CFR 219.19).

fecundity—The reproductive rate of an organism or population.

federally recognized Indian tribe—An Indian tribe or Alaska Native Corporation, band, nation, pueblo, village, or community that the Secretary of the Interior acknowledges

to exist as an Indian tribe under the Federally Recognized Indian Tribe List Act of 1994, 25 U.S.C. 479a (36 CFR 219.19).

fine filter—A conservation approach that focuses on conserving individual species in contrast to a "coarse filter" approach that focuses on conserving ecosystems; these approaches are generally viewed as complementary with fine-filtered strategies tailored to fit particular species that "fall through the pores" of the coarse filter (Hunter 2005). See also "mesofilter."

fire-dependent vegetation types—A vegetative community that evolved with fire as a necessary contributor to its vitality and to the renewal of habitat for its member species.

fire exclusion—Curtailment of wildland fire because of deliberate suppression of ignitions, as well as unintentional effects of human activities such as intensive grazing that removes grasses and other fuels that carry fire (Keane et al. 2002).

fire intensity—The amount of energy or heat release during fire.

fire regime—A characterization of long-term patterns of fire in a given ecosystem over a specified and relatively long period of time, based on multiple attributes, including frequency, severity, extent, spatial complexity, and seasonality of fire occurrence.

fire regime, low frequency, high severity—A fire regime with long return intervals (>200 years) and high levels of vegetation mortality (e.g., ~70 percent basal area mortality in forested ecosystems), often occurring in large patches (>10,000 ac [4047 ha]) (see chapter 3 for more details).

fire regime, moderate frequency, mixed severity—A fire regime with moderate return intervals between 50 and 200 years and mixtures of low, moderate, and high severity; high-severity patches would have been common and frequently large (>1,000 ac [>405 ha]) (see chapter 3 for more details).

fire regime, very frequent, low severity—A fire regime with short return intervals (5 to 25 years) dominated by surface fires that result in low levels of vegetation mortality (e.g., <20 percent basal area mortality in forested ecosystems), with high-severity fire generally limited to small patches (<2.5 ac [1 ha]) (see chapter 3 for more details).

fire regime, frequent, mixed severity—A fire regime with return intervals between 15 and 50 years that burns with a mosaic of low-, moderate-, and high-severity patches (Perry et al. 2011) (see chapter 3 for more details).

fire rotation—Length of time expected for a specific amount of land to burn (some parts might burn more than once or some not at all) based upon the study of past fire records in a large landscape (Turner and Romme 1994).

fire severity—The magnitude of the effects of fire on ecosystem components, including vegetation or soils.

fire suppression—The human act of extinguishing wildfires (Keane et al. 2002).

floodplain restoration—Ecological restoration of a stream or river's floodplain, which may involve setback or removal of levees or other structural constraints.

focal species—A small set of species whose status is assumed to infer the integrity of the larger ecological system to which it belongs, and thus to provide meaningful information regarding the effectiveness of a resource management plan in maintaining or restoring the ecological conditions to maintain the broader diversity of plant and animal communities in the NWPF area. Focal species would be commonly selected on the basis of their functional role in ecosystems (36 CFR 219.19).

food web—Interconnecting chains between organisms in an ecological community based upon what they consume.

Forest Ecosystem Management Assessment Team (FEMAT)—An interdisciplinary team that included expert ecological and social scientists, analysts, and managers assembled in 1993 by President Bill Clinton to develop options for ecosystem management of federal forests within the range of the northern spotted owl (FEMAT 1993).

forest fragmentation—The patterns of dispersion and connectivity of nonhomogeneous forest cover (Riitters et al. 2002). See also "landscape fragmentation" and "habitat fragmentation" for specific meanings related to habitat loss and isolation.

frequency distribution—A depiction, often appearing in the form of a curve or graph, of the abundance of possible values of a variable. In this synthesis report, we speak of the frequency of wildfire patches of various sizes.

fuels (wildland)—Combustible material in wildland areas, including live and dead plant biomass such as trees, shrub, grass, leaves, litter, snags, and logs.

fuels management—Manipulation of wildland fuels through mechanical, chemical, biological, or manual means, or by fire, in support of land management objectives to control or mitigate the effects of future wildland fire.

function (ecological)—Ecological processes, such as energy flow; nutrient cycling and retention; soil development and retention; predation and herbivory; and natural disturbances such as wind, fire, and floods that sustain composition and structure (FSM 2020). See also "key ecological function."

future range of variation (FRV)—The natural fluctuation of pattern components of healthy ecosystems that might occur in the future, primarily affected by climate change, human infrastructure, invasive species, and other anticipated disturbances.

gaps (forest)—Small openings in a forest canopy that are naturally formed when one or a few canopy trees die (Yamamoto 2000).

genotype—The genetic makeup of an individual organism.

glucocorticoid—A class of steroid hormones produced by many species of animals, often as the result of stress.

goals (in land management plans)—Broad statements of intent, other than desired conditions, that do not include expected completion dates (36 CFR part 219.7(e)(2)).

guideline—A constraint on project and activity decision-making that allows for departure from its terms, so long as the purpose of the guideline is met (36 CFR section 219.15(d)(3)). Guidelines are established to help achieve or maintain a desired condition or conditions, to avoid or mitigate undesirable effects, or to meet applicable legal requirements.

habitat—An area with the environmental conditions and resources that are necessary for occupancy by a species and for individuals of that species to survive and reproduce.

habitat fragmentation—Discontinuity in the spatial distribution of resources and conditions present in an area at a given scale that affects occupancy, reproduction, and survival in a particular species (see "landscape fragmentation").

heterogeneity (forest)—Diversity, often applied to variation in forest structure within stands in two dimensions: horizontal (e.g., single trees, clumps of trees, and gaps of no trees), and vertical (e.g., vegetation at different heights from the forest floor to the top of the forest canopy), or across large landscapes (North et al. 2009).

hierarchy theory—A theory that describes ecosystems at multiple levels of organization (e.g., organisms, populations, and communities) in a nested hierarchy.

high-severity burn patch—A contiguous area of high-severity or stand-replacing fire.

historical range of variation (HRV)—Past fluctuation or range of conditions in the pattern of components of ecosystems over a specified period of time.

hybrid ecosystem—An ecosystem that has been modified from a historical state such that it has novel attributes while retaining some original characteristics (see "novel ecosystem").

hybrid—Offspring resulting from the breeding of two different species.

inbreeding depression—Reduced fitness in a population that occurs as the result of breeding between related individuals, leading to increased homogeneity and simplification of the gene pool.

in-channel restoration—Ecological restoration of the channel of a stream or river, often through placement of materials (rocks and wood) or other structural modifications.

individuals, clumps, and openings (ICO) method—A method that incorporates reference spatial pattern targets based upon individual trees, clumps of trees, and canopy openings into silvicultural prescriptions and tree-marking guidelines (Churchill et al. 2013).

Interagency Special Status and Sensitive Species Program (ISSSSP)—A federal agency program, established under the U.S. Forest Service Pacific Northwest Region and Bureau of Land Management Oregon/Washington state office. The ISSSSP superseded the Survey and Manage standards and guidelines under the NWFP and also addresses other species of conservation focus, coordinates development and revision of management recommendations and survey protocols, coordinates data management between the agencies, develops summaries of species biology, and conducts other tasks.

intermittent stream—A stream or reach of stream channel that flows, in its natural condition, only during certain times of the year or in several years, and is characterized by interspersed, permanent surface water areas containing aquatic flora and fauna adapted to the relatively harsh environmental conditions found in these types of environments.

invasive species—An alien species (or subspecies) whose deliberate, accidental, or self-introduction is likely to cause economic or environmental harm or harm to human health (Executive Order 13112).

key ecological function—The main behaviors performed by an organism that can influence environmental conditions or habitats of other species.

key watersheds—Watersheds that are expected to serve as refugia for aquatic organisms, particularly in the short term, for at-risk fish populations that have the greatest potential for restoration, or to provide sources of high-quality water.

land and resource management plan (Forest Service)—A document or set of documents that provides management direction for an administrative unit of the National Forest System (FSH 1909.12.5).

landform—A specific geomorphic feature on the surface of the Earth, such as a mountain, plateau, canyon, or valley.

landscape—A defined area irrespective of ownership or other artificial boundaries, such as a spatial mosaic of terrestrial and aquatic ecosystems, landforms, and plant communities, repeated in similar form throughout such a defined area (36 CFR 219.19).

landscape fragmentation—Breaking up of continuous habitats into patches as a result of human land use and thereby generating habitat loss, isolation, and edge effects (see "habitat fragmentation").

landscape genetics—An interdisciplinary field of study that combines population genetics and landscape ecology to explore how genetic relatedness among individuals and subpopulations of a species is influenced by landscape-level conditions.

landscape hierarchy—Organization of land areas based upon a hierarchy of nested geographic (i.e., different-sized) units, which provides a guide for defining the functional components of a system and how components at different scales are related to one another.

late-successional forest—Forests that have developed after long periods of time (typically at least 100 to 200 years) following major disturbances, and that contain a major component of shade-tolerant tree species that can regenerate beneath a canopy and eventually grow into the canopy in which small canopy gaps occur (see chapter 3 for more details). Note that FEMAT (1993) and the NWFP also applied this term to older (at least 80 years) forest types, including both old-growth and mature forests, regardless of the shade tolerance of the dominant tree species (e.g., 90-year-old forests dominated by Douglas-fir were termed late successional).

leading edge—The boundary of a species' range at which the population is geographically expanding through colonization of new sites.

legacy trees—Individual trees that survive a major disturbance and persist as components of early-seral stands (Franklin 1990).

legacies (biological)—Live trees, seed and seedling banks, remnant populations and individuals, snags, large soil aggregates, hyphal mats, logs, uprooted trees, and other biotic features that survive a major disturbance and persist as components of early-seral stands (Franklin 1990, Franklin et al. 2002).

lentic—Still-water environments, including lakes, ponds, and wet meadows.

longitudinal studies—Studies that include repeated observations on the same response variable over time.

lotic—Freshwater environments with running water, including rivers, streams, and springs.

low-income population—A community or a group of individuals living in geographic proximity to one another, or a set of individuals, such as migrant workers or American Indians, who meet the standards for low income and experience common conditions of environmental exposure or effect (CEQ 1997).

managing wildfire for resource objectives—Managing wildfires to promote multiple objectives such as reducing fire danger or restoring forest health and ecological processes rather than attempting full suppression. The terms "managed wildfire" or "resource objective wildfire" have also been used to describe such events (Long et al. 2017). However, fire managers note that many unplanned ignitions are managed using a combination of tactics, including direct suppression, indirect containment, monitoring of fire spread, and even accelerating fire spread, across their perimeters and over their full duration. Therefore, terms that separate "managed" wildfires from fully "suppressed" wildfires do not convey that complexity. (See "Use of wildland fire," which also includes prescribed burning).

matrix—Federal and other lands outside of specifically designated reserve areas, particularly the late-successional reserves under the NWFP, that are managed for timber production and other objectives.

mature forest—An older forest stage (>80 years) prior to old-growth in which trees begin attaining maximum heights and developing some characteristic, for example, 80 to 200 years in the case of old-growth Douglas-fir/western hemlock forests, often (but not always) including big trees (>50 cm diameter at breast height), establishment of late-seral species (i.e., shade-tolerant trees), and initiation of decadence in early species (i.e., shade-intolerant trees).

mesofilter—A conservation approach that "focuses on conserving critical elements of ecosystems that are important to many species, especially those likely to be overlooked by fine-filter approaches, such as invertebrates, fungi, and nonvascular plants" (Hunter 2005).

meta-analysis—A study that combines the results of multiple studies.

minority population—A readily identifiable group of people living in geographic proximity with a population that is at least 50 percent minority; or, an identifiable group that has a meaningfully greater minority population than the adjacent geographic areas, or may also be a geographically dispersed/transient set of individuals such as migrant workers or Americans Indians (CEQ 1997).

mitigation (climate change)—Efforts to reduce anthropogenic alteration of climate, in particular by increasing carbon sequestration.

monitoring—A systematic process of collecting information to track implementation (implementation monitoring), to evaluate effects of actions or changes in conditions or relationships (effectiveness monitoring), or to test underlying assumptions (validation monitoring) (see 36 CFR 219.19).

mosaic—The contiguous spatial arrangement of elements within an area. In regions, this is typically the upland vegetation patches, large urban areas, large bodies of water, and large areas of barren ground or rock. However, regional mosaics can also be described in terms of land ownership, habitat

patches, land use patches, or other elements. For landscapes, this is typically the spatial arrangement of landscape elements.

multiaged stands—Forest stands having two or more age classes of trees; this includes stands resulting from variable-retention silvicultural systems or other traditionally even-aged systems that leave residual or reserve (legacy) trees.

multiple use—The management of all the various renewable surface resources of the National Forest System so that they are used in the combination that will best meet the needs of the American people; making the most judicious use of the land for some or all of these resources or related services over areas large enough to provide sufficient latitude for periodic adjustments in use to conform to changing needs and conditions; that some land will be used for less than all of the resources; and harmonious and coordinated management of the various resources, each with the other, without impairment of the productivity of the land, with consideration being given to the relative values of the various resources, and not necessarily the combination of uses that will give the greatest dollar return or the greatest unit output, consistent with the Multiple-Use Sustained-Yield Act of 1960 (16 U.S.C. 528–531) (36 CFR 219.19).

natal site—Location of birth.

native knowledge—A way of knowing or understanding the world, including traditional ecological, and social knowledge of the environment derived from multiple generations of indigenous peoples' interactions, observations, and experiences with their ecological systems. This knowledge is accumulated over successive generations and is expressed through oral traditions, ceremonies, stories, dances, songs, art, and other means within a cultural context (36 CFR 219.19).

native species—A species historically or currently present in a particular ecosystem as a result of natural migratory or evolutionary processes and not as a result of an accidental or deliberate introduction or invasion into that ecosystem (see 36 CFR 219.19).

natural range of variation (NRV)—The variation of ecological characteristics and processes over specified scales of time and space that are appropriate for a given management application (FSH 1909.12.5).

nested hierarchy—The name given to the hierarchical structure of groups within groups used to classify organisms.

nontimber forest products (also known as "special forest products")—Various products from forests that do not include logs from trees but do include bark, berries, boughs, bryophytes, bulbs, burls, Christmas trees, cones, ferns, firewood, forbs, fungi (including mushrooms), grasses, mosses, nuts, pine straw, roots, sedges, seeds, transplants, tree sap, wildflowers, fence material, mine props, posts and poles, shingle and shake bolts, and rails (36 CFR part 223 Subpart G).

novel ecosystem—An ecosystem that has experienced large and potentially irreversibly modifications to abiotic conditions or biotic composition in ways that result in a composition of species, ecological communities, and functions that have never before existed, and that depart from historical analogs (Hobbs et al. 2009). See "hybrid ecosystem" for comparison.

old-growth forest—A forest distinguished by old trees (>200 years) and related structural attributes that often (but not always) include large trees, high biomass of dead wood (i.e., snags, down coarse wood), multiple canopy layers, distinctive species composition and functions, and vertical and horizontal diversity in the tree canopy (see chapter 3). In dry, fire-frequent forests, old growth is characterized by large, old fire-resistant trees and relatively open stands without canopy layering.

palustrine—Inland, nontidal wetlands that may be permanently or temporarily flooded and are characterized by the presence of emergent vegetation such as swamps, marshes, vernal pools, and lakeshores.

passive management—A management approach in which natural processes are allowed to occur without human intervention to reach desired outcomes.

patch—A relatively small area with similar environmental conditions, such as vegetative structure and composition. Sometimes used interchangeably with vegetation or forest stand.

Pacific Decadal Oscillation (**PDO)**—A recurring (approximately decadal-scale) pattern of ocean-atmosphere —a stream or reach of a channel that flows continuously or nearly so throughout the year and whose upper surface is generally lower than the top of the zone of saturation in areas adjacent to the stream.

perennial stream—A stream or reach of a channel that flows continuously or nearly so throughout the year and whose upper surface is generally lower than the top of the zone of saturation in areas adjacent to the stream.

phenotype—Physical manifestation of the genetic makeup of an individual and its interaction with the environment.

place attachment—The "positive bond that develops between groups or individuals and their environment" (Jorgensen and Stedman 2001: 234).

place dependence— "The strength of an individual's subjective attachment to specific places" (Stokols and Shumaker 1982: 157).

place identity—Dimensions of self that define an individual's [or group's] identity in relation to the physical environment through ideas, beliefs, preferences, feelings, values, goals, and behavioral tendencies and skills (Proshansky 1978).

place-based planning—"A process used to involve stakeholders by encouraging them to come together to collectively define place meanings and attachments" (Lowery and Morse 2013: 1423).

plant association—A fine level of classification in a hierarchy of potential vegetation that is defined in terms of a climax-dominant overstory tree species and typical understory herb or shrub species.

population bottleneck—An abrupt decline in the size of a population from an event, which often results in deleterious effects such as reduced genetic diversity and increased probability of local or global extirpation.

potential vegetation type (PVT)—Native, late-successional (or "climax") plant community that reflects the regional climate, and dominant plant species that would occur on a site in absence of disturbances (Pfister and Arno 1980).

poverty rate—A measure of financial income below a threshold that differs by family size and composition.

precautionary principle—A principle that if an action, policy, or decision has a suspected risk of causing harm to the public or to the environment, and there is no scientific consensus that it is not harmful, then the burden of proof that it is not harmful falls on those making that decision. Particular definitions of the principle differ, and some applications use the less formal term, "precautionary approach." Important qualifications associated with many definitions include (1) the perceived harm is likely to be serious, (2) some scientific analysis suggests a significant but uncertain potential for harm, and (3) applications of the principle emphasize generally constraining an activity to mitigate it rather than "resisting" it entirely (Doremus 2007).

prescribed fire—A wildland fire originating from a planned ignition to meet specific objectives identified in a written and approved prescribed fire plan for which National Environmental Policy Act requirements (where applicable) have been met prior to ignition (synonymous with controlled burn).

primary recreation activity—A single activity that caused a recreation visit to a national forest.

probable sale quantity—An estimate of the average amount of timber likely to be awarded for sale for a given area (such as the NWFP area) during a specified period.

provisioning services—A type of ecosystem service that includes clean air and fresh water, energy, food, fuel, forage, wood products or fiber, and minerals.

public participation geographic information system (PPGIS)—Using spatial decisionmaking and mapping tools to produce local knowledge with the goal of including and empowering marginalized populations (Brown and Reed 2009).

public values—Amenity values (scenery, quality of life); environmental quality (clean air, soil, and water); ecological

values (biodiversity); public use values (outdoor recreation, education, subsistence use); and spiritual or religious values (cultural ties, tribal history).

record of decision (ROD)—The final decision document that amended the planning documents of 19 national forests and seven Bureau of Land Management districts within the range of the northern spotted owl (the NWFP area) in April 1994 (Espy and Babbit 1994).

recreation opportunity—An opportunity to participate in a specific recreation activity in a particular recreation setting to enjoy desired recreation experiences and other benefits that accrue. Recreation opportunities include non-motorized, motorized, developed, and dispersed recreation on land, water, and in the air (36 CFR 219.19).

redundancy—The presence of multiple occurrences of ecological conditions, including key ecological functions (functional redundancy), such that not all occurrences may be eliminated by a catastrophic event.

refugia—An area that remains less altered by climatic and environmental change (including disturbances such as wind and fire) affecting surrounding regions and that therefore forms a haven for relict fauna and flora.

regalia—Dress and special elements made from a variety of items, including various plant and animal materials, and worn for tribal dances and ceremonies.

regulating services—A type of ecosystem service that includes long-term storage of carbon; climate regulation; water filtration, purification, and storage; soil stabilization; flood and drought control; and disease regulation.

representativeness—The presence of a full array of ecosystem types and successional states, based on the physical environment and characteristic disturbance processes.

reserve—An area of land designated and managed for a special purpose, often to conserve or protect ecosystems, species, or other natural and cultural resources from particular human activities that are detrimental to achieving the goals of the area.

resilience—The capacity of a system to absorb disturbance and reorganize (or return to its previous organization) so as to still retain essentially the same function, structure, identity, and feedbacks (see FSM Chapter 2020 and see also "socioecological resilience"). Definitions emphasize the capacity of a system or its constituent entities to respond or regrow after mortality induced by a disturbance event, although broad definitions of resilience may also encompass "resistance" (see below), under which such mortality may be averted.

resistance—The capacity of a system or an entity to withstand a disturbance event without much change.

restoration economy—Diverse economic activities associated with the restoration of structure or function to terrestrial and aquatic ecosystems (Nielsen-Pincus and Moseley 2013).

restoration, ecological—The process of assisting the recovery of an ecosystem that has been degraded, damaged, or destroyed. 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 (36 CFR 219.19).

restoration, functional—Restoration of dynamic abiotic and biotic processes in degraded ecosystems, without necessarily a focus on structural condition and composition.

riparian areas—Three-dimensional ecotones (the transition zone between two adjoining communities) 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 (36 CFR 219.19).

riparian management zone—Portions of a watershed in which riparian-dependent resources receive primary emphasis, and for which plans include Plan components to maintain or restore riparian functions and ecological functions (36 CFR 219.19).

riparian reserves—Reserves established along streams and rivers to protect riparian ecological functions and processes

necessary to create and maintain habitat for aquatic and riparian-dependent organisms over time and ensure connectivity within and between watersheds. The Aquatic Conservation Strategy in the NWFP record of decision included standards and guidelines that delineated riparian reserves.

risk—A combination of the probability that a negative outcome will occur and the severity of the subsequent negative consequences (36 CFR 219.19).

rural restructuring—Changes in demographic and economic conditions owing to declines in natural resource production and agriculture (Nelson 2001).

scale—In ecological terms, the extent and resolution in spatial and temporal terms of a phenomenon or analysis, which differs from the definition in cartography regarding the ratio of map distance to Earth surface distance (Jenerette and Wu 2000).

scenic character—A combination of the physical, biological, and cultural images that gives an area its scenic identity and contributes to its sense of place. Scenic character provides a frame of reference from which to determine scenic attractiveness and to measure scenic integrity (36 CFR 219.19).

science synthesis—A narrative review of scientific information from a defined pool of sources that compiles and integrates and interprets findings and describes uncertainty, including the boundaries of what is known and what is not known.

sense of place—The collection of meanings, beliefs, symbols, values, and feelings that individuals or groups associate with a particular locality (Williams and Stewart 1998).

sensitive species—Plant or animal species that receive special conservation attention because of threats to their populations or habitats, but which do not have special status as listed or candidates for listing under the Endangered Species Act.

sensitivity—In ecological contexts, the propensity of communities or populations to change when subject to disturbance, or the opposite of resistance (see "community resistance").

sink population—A population in which reproductive rates are lower than mortality rates but that is maintained by immigration of individuals from outside of that population (see also "source population").

social sustainability—"The capability of society to support the network of relationships, traditions, culture, and activities that connect people to the land and to one another, and support vibrant communities" (36 CFR 219.19). The term is commonly invoked as one of the three parts of a "triple-bottom line" alongside environmental and economic considerations. The concept is an umbrella term for various topics such as quality of life, security, social capital, rights, sense of place, environmental justice, and community resilience, among others discussed in this synthesis.

socioecological resilience—The capacity of socioecological systems (see "socioecological system") to cope with, adapt to, and influence change; to persist and develop in the face of change; and to innovate and transform into new, more desirable configurations in response to disturbance.

socioecological system (or social-ecological system)—A coherent system of biophysical and social factors defined at several spatial, temporal, and organizational scales that regularly interact, continuously adapt, and regulate critical natural, socioeconomic, and cultural resources (Redman et al. 2004); also described as a coupled-human and natural system (Liu et al. 2007).

source population—A population in which reproductive rates exceed those of mortality rates so that the population has the capacity to increase in size. The term is also often used to denote when such a population contributes emigrants (dispersing individuals) that move outside the population, particularly when feeding a sink population.

special forest products—See "nontimber forest products."

special status species—Species that have been listed or proposed for listing as threatened or endangered under the Endangered Species Act.

species of conservation concern—A species, other than federally recognized as a threatened, endangered, proposed,

or candidate species, that is known to occur in the NWFP area and for which the regional forester has determined that the best available scientific information indicates substantial concern about the species' capability to persist over the long term in the Plan area (36 CFR 219.9(c)).

stand—A descriptor of a land management unit consisting of a contiguous group of trees sufficiently uniform in age-class distribution, composition, and structure, and growing on a site of sufficiently uniform quality, to be a distinguishable unit.

standard—A mandatory constraint on project and activity decisionmaking, established to help achieve or maintain the desired condition or conditions, to avoid or mitigate undesirable effects, or to meet applicable legal requirements.

stationarity—In statistics, a process that, while randomly determined, is not experiencing a change in the probability of outcomes.

stewardship contract—A contract designed to achieve land management goals while meeting local and rural community needs, including contributing to the sustainability of rural communities and providing a continuing source of local income and employment.

strategic surveys—One type of field survey, specified under the NWFP, designed to fill key information gaps on species distributions and ecologies by which to determine if species should be included under the Plan's Survey and Manage species list.

stressors—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 (36 CFR 219.19).

structure (ecosystem)—The organization and physical arrangement of biological elements such as snags and down woody debris, vertical and horizontal distribution of vegetation, stream habitat complexity, landscape pattern, and connectivity (FSM 2020).

supporting services—A type of ecosystem service that includes pollination, seed dispersal, soil formation, and nutrient cycling.

Survey and Manage program—A formal part of the NWFP that established protocols for conducting various types of species surveys, identified old-forest-associated species warranting additional consideration for monitoring and protection (see "Survey and Manage species"), and instituted an annual species review procedure that evaluated new scientific and monitoring information on species for potentially recommending changes in their conservation status, including potential removal from the Survey and Manage species list.

Survey and Manage species—A list of species, compiled under the Survey and Manage program of the NWFP, that were deemed to warrant particular attention for monitoring and protection beyond the guidelines for establishing late-successional forest reserves.

sustainability—The capability to meet the needs of the present generation without compromising the ability of future generations to meet their needs (36 CFR 219.19).

sustainable recreation—The set of recreation settings and opportunities in the National Forest System that is ecologically, economically, and socially sustainable for present and future generations (36 CFR 219.19).

sympatric—Two species or populations that share a common geographic range and coexist.

threatened species—Any species that the Secretary of the Interior or the Secretary of Commerce has determined is likely to become an endangered species within the foreseeable future throughout all or a significant portion of its range. Threatened species are listed at 50 CFR sections 17.11, 17.12, and 223.102.

timber harvest—The removal of trees for wood fiber use and other multiple-use purposes (36 CFR 219.19).

timber production—The purposeful growing, tending, harvesting, and regeneration of regulated crops of trees to be cut into logs, bolts, or other round sections for industrial or consumer use (36 CFR 219.19).

topo-edaphic—Related to or caused by particular soil conditions, as of texture or drainage, rather than by physiographic or climatic factors within a defined region or area.

traditional ecological knowledge—"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" (Berkes et al. 2000: 1252). See also "native knowledge."

trailing edge—When describing the range of a species, the boundary at which the species' population is geographically contracting through local extinction at occupied sites.

trophic cascade—Changes in the relative populations of producers, herbivores, and carnivores following the addition or removal of top predators and the resulting disruption of the food web.

uncertainty—Amount or degree of confidence as a result of imperfect or incomplete information.

understory—Vegetation growing below the tree canopy in a forest, including shrubs and herbs that grow on the forest floor.

use of wildland fire—Management of either wildfire or prescribed fire to meet resource objectives specified in land or resource management plans (see "Managing wildfire for resource objectives" and "Prescribed fire").

variable-density thinning—The method of thinning some areas within a stand to a different density (including leaving dense, unthinned areas) than other parts of the stand, which is typically done to promote ecological diversity in a relatively uniform stand.

vegetation series (plant community)—The highest level of the fine-scale component (plant associations) of potential vegetation hierarchy based on the dominant plant species that would occur in late-successional conditions in the absence of disturbance.

vegetation type—A general term for a combination or community of plants (including grasses, forbs, shrubs, or trees), typically applied to existing vegetation rather than potential vegetation.

viable population—A group of breeding individuals of a species capable of perpetuating itself over a given time scale.

vital rates—Statistics describing population dynamics such as reproduction, mortality, survival, and recruitment.

watershed—A region or land area drained by a single stream, river, or drainage network; a drainage basin (36 CFR 219.19).

watershed analysis—An analytical process that characterizes watersheds and identifies potential actions for addressing problems and concerns, along with possible management options. It assembles information necessary to determine the ecological characteristics and behavior of the watershed and to develop options to guide management in the watershed, including adjusting riparian reserve boundaries.

watershed condition assessment—A national approach used by the U.S. Forest Service to evaluate condition of hydrologic units based on 12 indicators, each composed of various attributes (USDA FS 2011).

watershed condition—The state of a watershed based on physical and biogeochemical characteristics and processes (36 CFR 219.19).

watershed restoration—Restoration activities that focus on restoring the key ecological processes required to create and maintain favorable environmental conditions for aquatic and riparian-dependent organisms.

well-being—The condition of an individual or group in social, economic, psychological, spiritual, or medical terms.

wilderness—Any area of land designated by Congress as part of the National Wilderness Preservation System that was established by the Wilderness Act of 1964 (16 U.S.C. 1131–1136) (36 CFR 219.19).

wildlife—Undomesticated animal species, including amphibians, reptiles, birds, mammals, fish, and invertebrates or even all biota, that live wild in an area without being introduced by humans.

wildfire—Unplanned ignition of a wildland fire (such as a fire caused by lightning, volcanoes, unauthorized and accidental human-caused fires), and escaped prescribed fires.

wildland-urban interface (WUI)—The line, area, or zone where structures and other human development meet or intermingle with undeveloped wildland or vegetation fuels.

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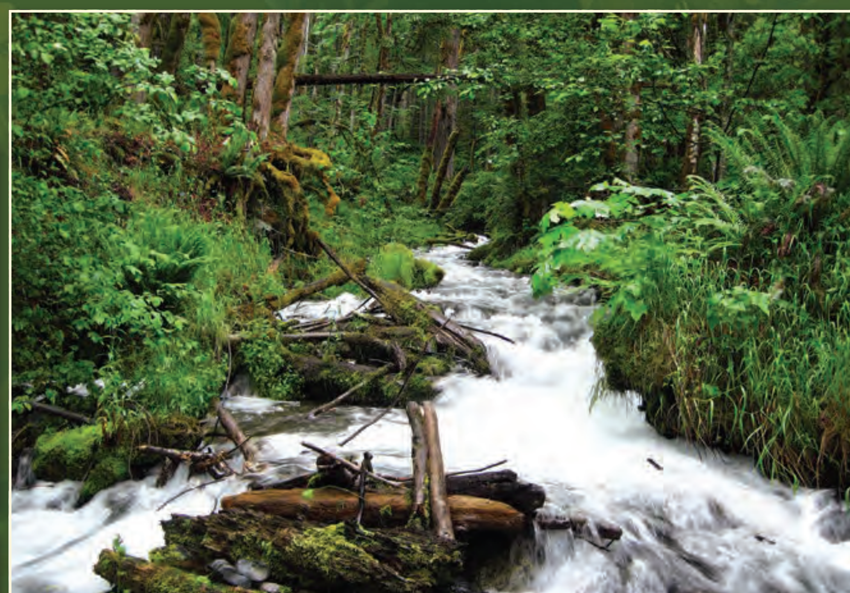
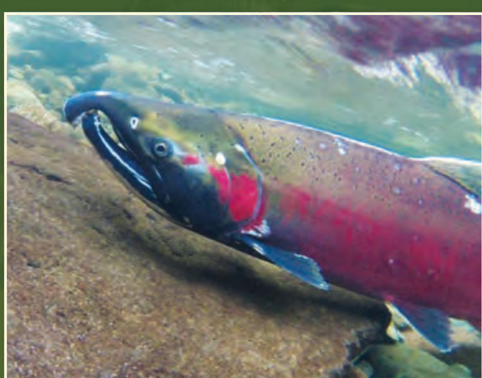
Synthesis of Science to Inform Land Management Within the Northwest Forest Plan Area

Volume 2





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Cover photos: Upper left: coho spawning on the Salmon River; photo by Bureau of Land Management. Upper right: a typical stream in the Cispus River watershed, Washington; photo by Alanna Wong. Lower right: an adult captive red tree vole mechanically removing one of two unpalatable resin ducts in a Douglas-fir needle before eating the rest of the needle; photo by Michael Durham. Lower middle: orange coral fungus, Olympic National Forest; photo by USDA Forest Service. Lower left: centipede, one component of the wide array of soil invertabrates found in late-successional and old-growth forests of the Northwest Forest Plan region; photo by Bruce Marcot.

Synthesis of Science to Inform Land Management Within the Northwest Forest Plan Area

Volume 2

Thomas A. Spies, Peter A. Stine, Rebecca Gravenmier,
Jonathan W. Long, and Matthew J. Reilly, Technical Coordinators

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Abstract

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The 1994 Northwest Forest Plan (NWFP) was developed to resolve debates over old-growth forests, endangered species, and timber production on federal forests in the range of the northern spotted owl. This three-volume science synthesis, which consists of 12 chapters that address various ecological and social concerns, is intended to inform forest plan revision and forest management within the NWFP area. Land managers with the U.S. Forest Service provided questions that helped guide preparation of the synthesis, which builds on the 10-, 15-, and 20-year NWFP monitoring reports and synthesizes the vast body of relevant scientific literature that has accumulated in the 24 years since the NWFP was initiated. It identifies scientific findings, lessons learned, and uncertainties and also evaluates competing science and provides considerations for management.

This synthesis finds that the NWFP has protected dense old-growth forests and maintained habitat for northern spotted owls, marbled murrelets, aquatic organisms, and other species despite losses from wildfire and low levels of timber harvest on federal lands. Even with reductions in the loss of older forests, northern spotted owl populations continue to decline. Moreover, a number of other goals have not been met, including producing a sustainable supply of timber, decommissioning roads, biodiversity monitoring, significant levels of restoration of riparian and dry forests, and adaptation and learning through adaptive management.

New conservation concerns have arisen, including a major threat to spotted owl populations from expanding populations of the nonnative barred owl, effects of fire suppression on forest succession, fire behavior in dry forests, and lack of development of diverse early-seral vegetation as a result of fire suppression in drier parts of moist forests. Climate change and invasive species have emerged as threats to native biodiversity, and expansion of the wildland-urban interface has limited the ability of managers to restore fire to fire-dependent ecosystems.

The policy, social, and ecological contexts for the NWFP have changed since it was implemented. The contribution of federal lands continues to be essential to the conservation and recovery of fish listed under the Endangered Species Act and northern spotted owl and marbled murrelet populations. Conservation on federal lands alone, however, is likely insufficient to reach the goals of the NWFP or the newer goals of the 2012 planning rule, which emphasizes managing for ecosystem goals (e.g. ecological resilience) and a few species of concern, rather than the population viability of hundreds of individual species.

The social and economic basis of many traditionally forest-dependent communities have changed in 24 years, and many are now focused on amenity values. The capacities of human communities and federal agencies, collaboration among stakeholders, the interdependence of restoration and the timber economy, and the role of amenity- or recreation-based communities and ecosystem services are important considerations in managing for ecological resilience, biodiversity conservation, and social and economic sustainability.

A growing body of scientific evidence supports the importance of active management or restoration inside and outside reserves to promote biodiversity and ecological resilience. Active management to promote heterogeneity of vegetation conditions is important to sustaining tribal ecocultural resources. Declines in agency capacity, lack of markets for small-diameter wood, lack of wood processing infrastructure in some areas, and lack of social agreement have limited the amount of active management for restoration on federal lands. All management choices involve social and ecological tradeoffs related to the goals of the NWFP. Collaboration, risk management, adaptive management, and monitoring are considered the best ways to deal with complex social and ecological systems with futures that are difficult to predict and affect through policy and land management actions.

Keywords: Northwest Forest Plan, science, management, restoration, northern spotted owl, marbled murrelet, climate change, socioeconomic, environmental justice.

Preface

In 2015, regional foresters in the Pacific Northwest and Pacific Southwest Regions of the USDA Forest Service requested that the Pacific Northwest and Pacific Southwest Research Stations prepare a science synthesis to inform revision of existing forest plans under the 2012 planning rule in the area of the Northwest Forest Plan (NWFP, or Plan). Managers provided an initial list of hundreds of questions to the science team, which reduced to them to 73 questions deemed most feasible for addressing through a study of current scientific literature. The stations assembled a team of 50 scientists with expertise in biological, ecological, and socioeconomic disciplines. At the suggestion of stakeholders, a literature reference database was placed online so the public could submit additional scientific literature for consideration. By spring 2016, writing was underway on 12 chapters that covered ecological and social sciences.

The draft synthesis, which was ready for peer and public review by fall 2016, went through a special review process because it was classified as "highly influential science" in accordance with the Office of Management and Budget's 2004 "Final Information Quality Bulletin for Peer Review." The synthesis was classified as such because it fit the category of a scientific assessment that is novel, controversial, or precedent-setting, or has significant interagency interest. Per the bulletin, the two research stations commissioned an independent entity, the Ecological Society of America (ESA), to manage the peer-review process, including the selection of peer reviewers.

The bulletin also stipulates that such an assessment be made available to the public through a public meeting to enable the public to bring scientific issues to the attention of peer reviewers. Accordingly, a public forum was held in Portland, Oregon, in December 2016. For those who could not travel to Portland, the forum was accessible via live Web stream, and multiple national forests within the NWFP area hosted remote viewing. Written comments on the draft synthesis were collected for 2 months. This generated 130 public comments, totaling 890 pages, which were given to the peer reviewers for consideration in their review, as they deemed appropriate. The OMB guidelines further direct that the peer-review process be transparent by making available to the public the ESA's written guidance to the reviewers, the peer reviewer's names, the peer review reports, and the responses of the authors to the peer reviewer comments—all of which are available at https://www.fs.fed.us/pnw/research/science-synthesis/index.shtml.

The peer reviewer comments, which were received in spring 2017 and informed by public input, resulted in substantive revisions to chapters of the synthesis. The result is this three-volume general technical report (an executive summary of the synthesis is available as a separate report). This document is intended to support upcoming management planning on all public lands in the Plan area, but is expected to serve primarily lands managed by the U.S. Forest Service. We hope it will be a valuable reference for managers and others who seek to understand the scientific basis and possible tradeoffs associated with forest plan revision and management decisions. The synthesis also provides an extensive list of published sources where readers can find further information.

We understand that the term “synthesis” can have many different meanings. For our purposes, it represents a compilation and interpretation of relevant scientific findings that pertain to key issues related to the NWFP that were identified by managers and by the authors of the document. Such a compilation not only summarizes science by topic areas but also interprets that science in light of management goals, characterizes competing science, and makes connections across scientific areas, addressing multilayered and interacting ecological and socioeconomic issues. In a few cases, simple analyses of existing data were conducted and methods were provided to reviewers.

The synthesis builds upon the 10-, 15-, and 20-year NWFP monitoring reports, and authors considered well over 4,000 peer-reviewed publications based on their knowledge as well as publications submitted by the public and others suggested by peer reviewers. For some of the questions posed by land managers, there was ample scientific research from the Plan area. For many of the questions, however, little research existed that was specific to the area. In such cases, studies from other regions or current scientific theory were used to address the questions to the extent possible. In many cases, major scientific uncertainties were found; these are highlighted by the authors.

The synthesis chapters characterize the state of the science but they do not develop management alternatives, analyze management tradeoffs, or offer recommendations as to what managers should do. The synthesis does identify ideas, facts, and relationships that managers may want to consider as they develop plans and make management decisions about particular issues. The final chapter attempts to integrate significant cross-cutting issues, e.g., ecological and socioeconomic interdependencies, compatibility of different management goals, and tradeoffs associated with different restoration actions. All the chapters identify where more research is needed to fill critical information gaps.

We would like to acknowledge the peer reviewers who considered hundreds of public comments as part of the process of reviewing our lengthy draft manuscripts. We also thank the many contributors to the development of the synthesis in draft and final form, including those who provided editing, layout, database, and other support services.

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Table Rock Wilderness, Oregon.
Photo by USDI Bureau of Land Management, Oregon-Washington State Office.

Chapter 6: Other Species and Biodiversity of Older Forests

Bruce G. Marcot, Karen L. Pope, Keith Slauson, Hartwell H. Welsh, Clara A. Wheeler, Matthew J. Reilly, and William J. Zielinski[1]

Introduction

This chapter focuses mostly on terrestrial conditions of species and biodiversity associated with late-successional and old-growth forests in the area of the Northwest Forest Plan (NWFP). We do not address the northern spotted owl (*Strix occidentalis caurina*) or marbled murrelet (*Brachyramphus marmoratus*)—those species and their habitat needs are covered in chapters 4 and 5, respectively. Also, the NWFP's Aquatic and Riparian Conservation Strategy and associated fish species are addressed in chapter 7, and early-successional vegetation and other conditions are covered more in chapters 3 and 12.

We begin by summarizing a set of questions provided by management. We then review the state of knowledge of species, biodiversity, and ecosystem conditions gained from studies conducted since the 10-year synthesis of monitoring and research results (Haynes et al. 2006). We review agency programs on other species and biodiversity of older forests of the Pacific Northwest, including implementation of the NWFP Survey and Manage standards and guidelines, the Interagency Special Status and Sensitive Species Program (ISSSSP), and other biodiversity consortia. We then review new findings on selected individual species and groups of species including fungi, lichens, bryophytes, and plants, as well as invertebrates. We also summarize findings on amphibians, reptiles, and birds, and on selected carnivore species including fisher (*Pekania pennanti*), marten (*Martes americana*), and wolverine (*Gulo gulo*), and on red tree voles (*Arborimus longicaudus*) and bats. We close the section with a brief review of the value of early-seral vegetation environments. We next review recent advances in development of new tools and datasets for species and biodiversity conservation in late-successional and old-growth forests, and then review recent and ongoing challenges and opportunities for ameliorating threats and addressing dynamic system changes. We end with a set of management considerations drawn from research conducted since the 10-year science synthesis and suggest areas of further study.

The general themes reviewed in this chapter were guided by a set of questions provided by the U.S. Forest Service Pacific Northwest Region (Region 6) and Pacific Southwest Region (Region 5). The scientific publications we review were selected based on the specific subjects listed above, as pertinent to science findings on other species and biodiversity of late-successional and old-growth forest ecosystems in the area of the NWFP in the Pacific Northwest, United States. We include selected references on studies outside the NWFP and Pacific Northwest and references dating prior to the previous NWFP science synthesis, when such studies are nonetheless pertinent to understanding biological and ecological topics within the NWFP and Pacific Northwest. We also address selected topics such as early-successional forest ecosystems and effects of wildfire, fire suppression, and climate change, as guided by the availability of recent literature on NWFP species and biodiversity; these topics, raised by managers, are also covered more fully in other chapters of this science synthesis. The final chapter of this synthesis discusses the conceptual and practical implications of new science findings, remaining scientific uncertainties and research needs, and overall conclusions.

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Setting and Background

Originally, the NWFP was developed as an ecosystem management plan to provide for the full suite of biodiversity at all taxonomic and functional levels, particularly in late-successional and old-growth environments, under an adaptive learning and management approach. The first decade of the NWFP, however, focused on the status of species; no biodiversity monitoring program per se was instituted under the NWFP (Marcot and Molina 2006). Since then, a broad assumption has been made that older-forest biodiversity in its full capacity would be provided by two complementary approaches of managing for "coarse filter" elements such as the dispersion and distribution of late-successional forest reserves and aquatic and riparian corridors, along with managing for "fine filter" elements of habitat needs of selected, individual late-successional and old-growth-associated species. The combined coarse- and fine-filter approach is intended to provide the same level of protection as would management and monitoring directed at specific biodiversity elements such as ecosystem processes of nutrient cycling, species' ecological functions, and population genetics and viability (e.g., Noss 1990). A current challenge is to test this assumption within the changing tapestry of ecological processes and disturbance-influenced ecosystems of the Northwest.

The 2012 planning rule for guiding land and resource management plans on national forests differs from the 1982 rule that was in place when the NWFP was first instituted and that guided the NWFP. The 2012 planning rule puts more weight on coarse-filter approaches and on ecological integrity (based in part on natural range of variation) but still calls for both coarse- and fine-filter approaches. In the Forest Service's evaluations of the alternatives to the planning rule (USDA FS 2012), the terms coarse filter and fine filter are referred to extensively as "well-developed concept[s] in the scientific literature [with] broad support from the scientific community and many stakeholders." However, the debate continues on an appropriate balance between coarse-filter (ecosystem and biodiversity) and fine-filter (species-specific) planning direction (Hayward et al. 2016,[2] Schultz et al. 2013).

Also, although not part of the NWFP per se, some previous elements of the U.S. Forest Service' Forest Inventory and Analysis (FIA) program[3]—which have since been drastically reduced or are no longer being carried out—provided much-needed information for monitoring biodiversity of trees, vegetation, and lichens.[4] The FIA program has become a de facto biodiversity monitoring program, at least for selected vegetation and floral elements.

In the previous science synthesis, Marcot and Molina (2006) concluded that NWFP directions for establishing effectiveness monitoring of forest biodiversity elements for other than selected species remained mostly unmet (beyond the FIA-identified biodiversity indicators). This remains true today, but much information has been provided by research studies and gathered by agency programs on basic occurrence, distribution, and ecology of rare and poorly known late-successional and old-growth-associated species. The 2006 synthesis also provided the following suggestions:

- Engage research partnerships to fill key information gaps on rare and little-known late-successional and old-growth species.
- Clarify objectives and expectations of implementing a coarse- and fine-filter conservation approach to managing for viable and persistent species populations.
- Validate the use of surrogates (e.g., indicator and focal species) for species and conservation objectives.
- Develop and maintain databases from ongoing inventory, survey, and any monitoring programs.

[2] Hayward, G.D.; Flather, C.H.; Rowland, M.M.; Terney, R.; Mellen-Mclean, K.; Malcolm, K.D.; McCarthy, C.; Boyce, D.A. 2016. Applying the 2012 planning rule to conserve species: a practitioner's reference. Unpublished paper. On file with: Bruce Marcot, Forestry Sciences Laboratory, 620 SW Main, Portland, OR 97205. 78 p.

[3] http://www.fia.fs.fed.us/.

[4] McCune, B. Personal communication. Professor, Department of Botany and Plant Pathology, Oregon State University, 2082 Cordley Hall Corvallis, OR 97331-2902. http://bmccune.weebly.com/.

- Develop, test, and implement species survey designs.
- Explore habitat modeling and decision-support tools to meet some conservation objectives.
- Develop and implement an effectiveness monitoring framework.

The current synthesis determines the degree to which these suggestions have been met.

New Learning and Recent Issues

Much has been learned since the 2006 synthesis (Haynes et al. 2006) about conditions and dynamics of forest ecosystems and their organisms in the Pacific Northwest and throughout the West. The issue of climate change and its known and projected impacts on systems has become a foremost research topic (Bagne et al. 2011, Vose et al. 2012). Occurrence and effects of large-scale wildfire have become major issues of research and management focus (Sheehan et al. 2015, Wimberly and Liu 2014). Studies on the effect of fire suppression on vegetation succession are needed, however, as are studies examining how suppression activities affect subsequent fire behavior but also how it changes vegetation conditions as habitat for many species (see chapter 3). Concern over invasive species also has elevated (Jones et al. 2010, Wilson et al. 2009; also see chapter 3). We address these and other issues in sections that follow.

Additionally, of increased focus is how early-successional vegetation provides habitats for many species (Hagar 2007, Swanson et al. 2011). Another topic of continued interest is the importance of conditions in the managed-forest matrix and connectivity of late-successional and old-growth forests and late-successional reserves (LSRs) in the face of fire, climate change, and increased pressure on matrix-land resources (Suzuki and Olson 2008, Wilson and Puettmann 2007). As well, the roles and conditions of rare and little-known species have been addressed (Raphael and Molina 2007). In general, much more detailed information is now available on vertebrates than on most other species groups.

Guiding Questions

This chapter reviews the scientific understanding of the ecology and conservation of species associated with late-successional and old-growth forests. We summarize science findings on the conservation strategy of the NWFP and its provision for these species; scientific progress since the previous NWFP evaluations (Diaz and Haynes 2002, Haynes and Perez 2001, Haynes et al. 2006); and the outcome of the NWFP Survey and Manage program.

We review advancements on science and conservation through the following questions:

- What is the current scientific understanding of the rarity of late-successional and old-growth-associated species?
- Is forest management under the NWFP providing habitat for rare and uncommon species as planned? Are rare and uncommon species maintaining populations under NWFP management? How effective are the management recommendations for habitat conservation in retaining these species across treated landscapes?
- Have we accumulated enough information to change the management status of these species? Are there species originally ranked as having low potential for persistence that are now of less concern, particularly with the reduction in harvest levels of late-successional and old-growth forest that has occurred under the NWFP? Are there late-successional and old-growth species originally ranked as high persistence or not initially identified as conservation concerns that have been added to lists of species of concern?
- What are results of research on the effects of prescribed fire and wildfire on rare and uncommon late-successional and old-growth species?
- What are results of research on the effectiveness of site buffers as compared with landscape-scale habitat management for ensuring late-successional and old-growth species persistence, dispersal, and habitat connectivity?
- How has the ISSSSP served to provide information on late-successional and old-growth-associated species under the NWFP?

- Does the current list of special status and sensitive species adequately represent rare late-successional and old-growth species with risks to population persistence?
- What are new issues related to conservation of biodiversity in the NWFP area?

Agency Programs on Other Species and Biodiversity of Older Forests

Survey and Manage Program

Following the 1993 report of the Forest Ecosystem Management Assessment Team or FEMAT (1993), and as part of the initial creation of the NWFP, the NWFP Survey and Manage program was instituted in 1993 as part of a final environmental impact statement and record of decision for amendments to U.S. Department of Agirculture (USDA) Forest Service and U.S. Department of the Interior (USDI) Bureau of Land Management (BLM) planning documents for federal public lands within the range of the northern spotted owl. The Survey and Manage program was then amended by a 2001 record of decision for amendment to the Survey and Manage, protection buffer, and other mitigation measures standards and guidelines. The amendment established (1) an annual species review panel process to evaluate monitoring and research findings and to recommend to the regional forester of Region 6 appropriate conservation categories for all late-successional and old-growth-associated species not otherwise provided for by the NWFP guidelines, and (2) a set of site survey protocols[5] and management recommendations for detecting and conserving sites with rare and little-known species under the NWFP. The annual species review sessions were designed as rigorous, 10-person panels consisting of 5 biologists and 5 managers and used a Bayesian network decision modeling construct to help evaluate knowledge and explicitly represent uncertainty of each species in documented, repeatable procedures (Marcot et al. 2006). Mostly because of high costs and administrative complexities, no formal annual species review has been conducted since 2003.

The Survey and Manage program was established under the NWFP as a means of collecting information on, and providing appropriate conservation direction for, rare and poorly known late-successional and old-growth-associated species under the precautionary principle (resisting implementation of untested or disputed activities that may have adverse effects) and an adaptive management process (Marcot et al. 2006, Molina et al. 2003, USDA and USDI 2001). From the initial list of 1,120 late-successional and old-growth-associated species evaluated by FEMAT (1993), various mitigation means under the NWFP Survey and Manage program narrowed the list in 2001 to 296 individual species and 4 arthropod species groups. The Survey and Manage program was then abolished, and, under a management policy decision of the agencies, 152 of the 296 species were moved to the USDA Forest Service Sensitive Species program and the USDI BLM Special Status Species program, but the court then mandated that the Survey and Manage record of decision be reinstated. Eventually, the two agencies' species programs were merged into the ISSSSP, discussed more fully below, which has since held the responsibility for evaluation of late-successional and old-growth species in the region. Also in the interim, a set of new national forest planning regulations have been instituted that provide impetus for considering other species, biodiversity, ecosystems, dynamics, and functions of both older and early-seral forests (Schultz et al. 2013). We discuss these updates to Forest Service and BLM planning guidelines and regulations further below.

The Survey and Manage program has had an unstable existence, having been established in 1994 (USDA and USDI 1994) with corrections to its standards and guidelines published in 2001 (USDA and USDI 2001), abolished by the agencies in 2004 (USDA and USDI 2004), reinstated by the court in 2006, again abolished by the agencies in 2007, challenged in 2008, and with a court ruling in 2009 that the 2007 Forest Service environmental impact statement was flawed and the court subsequently approving a settlement agreement in 2011. The timber industry then challenged the settlement agreement in 2011 (and subsequently dropped their appeal in 2015), the Ninth Circuit Court of Appeals reversed and remanded the approval of the settlement

[5] http://www.blm.gov/or/plans/surveyandmanage/protocols/.

agreement in 2013, and then in 2014, the 2007 records of decision were vacated.

Vacatur of the two 2007 records of decision (in 2007, BLM and the Forest Service each issued separate records of decision) has had the effect of returning the agencies to the status quo in existence prior to the 2007 records of decision. The status quo existing before the 2007 records of decision was defined by three previous court rulings, as follows. First is the 2006 court order reinstating the 2001 record of decision, including any amendments or modifications that were in effect as of March 21, 2004. This ruling incorporated the 2001, 2002, and 2003 annual species review changes. Second was the 2006 court-ordered categories of activities that could proceed without conducting predisturbance surveys or site management for species: (1) thinning in forests less than 80 years old; (2) replacement or removal of water culverts; (3) activities for improvement of riparian and stream areas; and (4) treatment of hazardous fuels, including use of prescribed fire; these reinstatements were retained in the later court rulings mentioned above. Third was the 2006 court ruling that vacated the 2001 and 2003 annual species review category change and subsequent removal of reference to the red tree vole in a portion of its range, returning the species to its prior monitoring status throughout its range.

At present, oversight of the Survey and Manage standards and guidelines implementation is consigned to staff members within the ISSSSP (Region 6 and Oregon BLM) and the Region 5 regional wildlife program manager within the Ecosystem Conservation staff. These individuals coordinate revision of management recommendations and survey protocols, assist field specialists in implementing the standards and guidelines, resolve issues between Survey and Manage species management and meeting other resource objectives, coordinate data management between the agencies preparing for an Annual Species Review, stay abreast of taxonomic updates, and coordinate methods for filling information gaps. To clarify, Forest Service Region 5 is not formally a part of the ISSSSP, which is unique to Forest Service Region 6 and Oregon-Washington BLM.

The list of late-successional and old-growth-associated species as provided by FEMAT (1993) had been evaluated by the Forest Service and BLM under the Survey and Manage program's annual species reviews, using a set of published guidelines (table 6-1) to determine species' potential need for more specific and additional conservation. Based on an evaluation of the occurrence of, and scientific knowledge on, the species, about 400 species of amphibians, bryophytes, fungi, lichens, mollusks, vascular plants, arthropod functional groups, and one mammal were deemed to be potentially at-risk, and the rest of the species were deemed to be adequately provided under the NWFP guidelines; the genealogy through 2006 of the many species lists are covered by Marcot and Molina (2006) and Molina et al. (2006). The annual species reviews developed and adopted use of a Bayesian network decision modeling approach to help wade through the complex evaluation guidelines (table 6-1) and to document results on each species (Marcot et al. 2006).

Under the Survey and Manage program, about 68,000 sites with presence of Survey and Manage species were identified by surveys, and new ecological knowledge was gained on about 100 species leading to their being removed from the protection list (Molina et al. 2006). Additionally, a set of field and management guides were produced on aquatic and terrestrial mollusks and fungi,[6] and guidelines were published on assessing rare species of lichens (Edwards et al. 2004), fungi (Castellano et al. 2003, Molina 2008), and other taxa. Eventually, the high cost of maintaining the Survey and Manage program, running into several tens of millions of dollars, with its annual species reviews and all other activities associated with compiling scientific and monitoring information on late-successional and old-growth-associated species, was a factor considered by managers in their decision to abolish the program and enfold it into the ISSSSP.

Understanding the distributions and disturbance responses of rare species is a perennial problem in ecology, the main issues of which include securing adequate sample sizes for statistical analyses (Cunningham and Lindenmayer 2005). Methods for increasing confidence in such studies include stratifying samples, such as demonstrated by Edwards et al. (2005) with five rare epiphytic macrolichens in the Pacific Northwest United States.

[6] http://www.blm.gov/or/plans/surveyandmanage/field.php.

Table 6-1—Guidelines for determining whether late-successional and old-growth forest (late-successional and old-growth)-associated species under the Northwest Forest Plan (NWFP) may need additional conservation consideration, as required under the Survey and Manage program 2001 record of decision (USDA and USDI 2001)

Evaluation category[a]	Description in record of decision (USDA and USDI 2001)
1. Geographic range	The species must occur within the NWFP area or near the NWFP area and have potentially suitable habitat within the NWFP area.
2. Late-successional and old-growth association	A species is considered to be closely associated with late-successional and old-growth forests if it meets at least one of the following criteria: • The species is significantly more abundant in late-successional and old-growth forest than in young forest, in any part of its range. • The species shows association with late-successional and old-growth forest and may reach highest abundance there, and the species requires habitat components that are contributed by late-successional and old-growth forest. • The species is associated with late-successional and old-growth forest, based on field study, and is on a federal U.S. Fish and Wildlife Service (USFWS) list or state threatened or endangered list; the USFWS candidate species list; a Bureau of Land Management or Forest Service special status species list in California, Oregon, or Washington; or is listed by the states of California, Oregon, or Washington as a species of special concern or as a sensitive species. • Field data are inadequate to measure strength of association with late-successional and old-growth forest; the species is listed as a federal USFWS threatened and endangered species; and the Forest Ecosystem Management Assessment Team suspected, or the panel doing the final placement in Species Review Process suspects, that it is associated with late-successional and old-growth forest.
3. Plan provides for persistence	The reserve system and other standards and guidelines of the NWFP do not appear to provide for a reasonable assurance of species persistence. Criteria indicating a concern for persistence, i.e., one or more of the following criteria must apply: • Low to moderate number of likely extant known sites/records in all or part of a species range. • Low to moderate number of individuals. • Low to moderate number of individuals at most sites or in most populations. • Very limited to somewhat limited range. • Distribution within habitat is spotty or unpredictable in at least part of its range. • Very limited to somewhat limited habitat. Criteria indicating little or no concern for persistence, usually, most of the following criteria must apply: • Moderate to high number of likely extant sites/records. • Sites are relatively well distributed within the species range. • High proportion of sites and habitat in reserve land allocations; or limited number of sites within reserves, but the proportion or amount of potential habitat within reserves is high and there is a high probability that the habitat is occupied. • Matrix standards and guidelines or other elements of the NWFP provide a reasonable assurance of species persistence.
4. Data sufficiency	Information is insufficient to determine whether Survey and Manage basic criteria are met, or to determine what management is needed for a reasonable assurance of species persistence.

Table 6-1—Guidelines for determining whether late-successional and old-growth forest (late-successional and old-growth)-associated species under the Northwest Forest Plan (NWFP) may need additional conservation consideration, as required under the Survey and Manage program 2001 record of decision (USDA and USDI 2001) (continued)

Evaluation category[a]	Description in record of decision (USDA and USDI 2001)
5. Practicality of survey	Surveys are considered "practical" if all of the following criteria apply: • The taxon appears annually or predictably, producing identifying structures that are visible for a predictable and reasonably long time. • The taxon is not so minuscule or cryptic as to be barely visible. • The taxon can authoritatively be identified by more than a few experts, or the number of available experts is not so limited that it would be impossible to accomplish all surveys or identifications for all proposed habitat-disturbing activities in the NWFP area needing identification within the normal planning period for the activity. • The taxon can be readily distinguished in the field and needs no more than simple laboratory or office examination to confirm its identification. • Surveys do not require unacceptable safety (5a) or species risks. • Surveys can be completed in two field seasons (about 7 to 18 months). • Credible survey methods for the taxon are known or can be developed within a reasonable time period, i.e., about 1 year.
6a. Relative rarity	The species is relatively rare and all known sites or population areas are likely to be necessary to provide reasonable assurance of species persistence, as indicated by one or more of the following: • The species is poorly distributed within its range or habitat. • Limited dispersal capability on federal lands. • Reproduction or survival not sufficient. • Low number of likely extant sites/records on federal lands indicates rarity. • Limited number of individuals per site. • Declining population trends. • Low number of sites in reserves or low likelihood of sites or habitat in reserves. • Highly specialized habitat requirements (narrow ecological amplitude). • Declining habitat trend. • Dispersal capability limited relative to federal habitat. • Habitat fragmentation that causes genetic isolation. • Microsite habitat limited. • Factors beyond management under the NWFP affect persistence, but special management under the NWFP will help persistence.
6b. Relative uncommonness	The species is relatively uncommon rather than rare, and not all known sites or population areas are likely to be necessary for reasonable assurance of persistence, as indicated by one or more of the following: • A higher number of likely extant sites/records does not indicate rarity of the species. • Low to high number of individuals/site. • Less restricted distribution pattern relative to range or potential habitat. • Moderate to broad ecological amplitude. • Moderate to high likelihood of sites in reserves. • Populations or habitats are stable.

[a] If criteria for any evaluation category were met, then the species may be further considered for needing additional conservation beyond what the NWFP generally provides; such further consideration was addressed during annual species reviews under the NWFP Survey and Manage program.

Interagency Special Status and Sensitive Species Program

The ISSSSP[7] was formed in 2005 as an interagency Forest Service Region 6 and BLM Oregon/Washington program for regional-level approaches for conservation and management of rare (but neither federally listed threatened nor endangered) species that would meet criteria for the two agencies' lists of special status species and sensitive species. Its geographic and ecological scope includes and exceeds that of the NWFP and late-successional and old-growth forests in Washington and Oregon. The ISSSSP is not a reformulation of the NWFP Survey and Manage program, although it has taken on some of those functions pertaining to evaluation of the conservation status of species, development of some species survey and monitoring protocols, and other items. The ISSSSP addresses species across Forest Service and BLM lands in Oregon and Washington (but not California), implementing the Forest Service sensitive species policy (FSM 2670) and BLM special status species policy (BLM 6840) and providing oversight of the Survey and Manage standards and guidelines. Criteria for determining Forest Service sensitive species are quite different from the Survey and Manage species criteria discussed above (also see table 1).

California, particularly northwest California within the NWFP area, does not have an organization equivalent to the ISSSSP, which is a collaboration unique to Washington and Oregon. In California, instead, the Forest Service Region 5 implements the national Forest Service sensitive species policy (FSM 2670) and results are overseen by various Forest Service regional office staff for the entire state, not just for the NWFP area and the six national forests therein. California BLM includes lands within the NWFP area, and those are overseen by the BLM Redding Resource Area, Arcata Resource Area, and the Kings Range National Conservation Area, all within the BLM Ukiah District.

The ISSSSP has produced a wide array of products related to conservation of rare, nonlisted species. Products include species fact sheets, conservation assessments, conservation strategies, inventory reports, inventory and survey protocols and methods workshops, and results of studies. The most recent program update[8] (June 2015) mentions reorganization of the program's conservation and inventory information on bats and fungi (covered below). The ISSSSP partners with and supports a variety of research and academic institutions to provide key information on rare species of conservation concern within its geographic venue.

Unique among federal land management agencies, the ISSSSP has developed criteria used in common with Forest Service and BLM for including species on sensitive and special status lists. The ISSSSP considers species for such listing by using independent information from the Oregon Biodiversity Information Center[9] Washington Natural Heritage Program[10] and NatureServe. [11]

The current list of Survey and Manage species[12] dates to December 2003 and includes 298 species: 189 fungi, 15 bryophytes, 40 lichens, 12 vascular plants, 36 snails and slugs (mollusks), 4 amphibians, 1 mammal (red tree vole, treated below), and 1 bird (great gray owl, *Strix nebulosa*).

Implications of Forest Service and BLM Planning Directions

The current planning rule for the U.S. Forest Service (2012: 21174)[13] states that its intent is

> ... to provide for the diversity of plant and animal communities, and keep common native species common, contribute to the recovery of threatened and endangered species, conserve proposed and candidate species, and maintain species of conservation concern within the plan area, within Agency authority and the inherent capability of the land.

[7] http://www.fs.fed.us/r6/sfpnw/issssp/.

[8] http://www.fs.fed.us/r6/sfpnw/issssp/documents3/update-2015-06.pdf.

[9] http://inr.oregonstate.edu/orbic.

[10] http://www.dnr.wa.gov/natural-heritage-program.

[11] http://www.natureserve.org/.

[12] http://www.blm.gov/or/plans/surveyandmanage/files/sm-fs-enc3-table1-1-dec2003wrtv.pdf.

[13] http://www.fs.usda.gov/detail/planningrule/home/?cid=stelprdb5359471.

The rule establishes guidelines and mandates for ecological sustainability, particularly for ecosystem integrity defined as the maintenance or restoration of terrestrial and aquatic ecosystems and watersheds, and their structure, function, composition, and connectivity. The rule also explicitly adopts a coarse- and fine-filter approach (further discussed below) to managing for diversity of plant and animal communities beginning with maintaining or restoring the diversity of ecosystem and habitat types including rare aquatic and terrestrial plant and animal communities, as well as identifying species of conservation concern to be designated by the responsible official, in coordination with the regional forester, based on scientific information. In the Pacific Northwest, the Forest Service is currently producing a draft list of potential species of conservation concern to facilitate efficiencies when the region undergoes plan revision under the 2012 planning rule.

Additional parts of the 2012 planning rule for the Forest Service provide guidance on monitoring, which it defines as "a systematic process of collecting information to evaluate effects of actions or changes in conditions or relationships" (USDA FS 2012:21271). The planning rule also provides guidance on managing ecological systems at the broad scale and for specific ecosystem elements, such as individual species, at finer scales. As such, specifically, the planning rule refers to coarse-filter management as "designing ecosystem ... connectivity based on landscape patterns of forests, grasslands, rangelands, streams, and wetlands that were created under ecological processes and landscape disturbance regimes that occurred before extensive human alteration" (section 23.11b: Ecosystem Integrity), and fine-filter management as "species-specific plan components, including standards and guidelines, for each of those species" (section 23.13: Species-Specific Plan Components for At-Risk Species).

Schultz et al. (2013) recommended directly monitoring selected species of conservation concern and focal species because of inconsistencies in the 2012 planning rule between its operational requirements and its generous discretionary allowances. They suggested that monitoring should evaluate viability of such species and that management should do no harm to species for which viability cannot be provided solely on Forest Service lands; and that monitoring should specify trigger points to spark reviews of management activities affecting species conservation.

As a point of history, BLM proposed a new rule for resource management planning nationally on BLM lands (USDI BLM 2016), but in February 2017, Congressional action nullified the regulations. In the NWFP area, BLM Resource Management Plans (RMPs) are in effect from records of decision signed in August 2016.[14] The new RMPs are intended to provide protection for northern spotted owls, listed fish species, and water resources, and provide for jobs, recreation, and timber harvest. At this point, it is unclear if the RMPs will provide additional guidelines for conservation of other old-forest species and biodiversity under the NWFP in addition to guidelines provided by ISSSSP.

Key Findings

New Information on Other Species and Biodiversity of Older Forests

In this section, we review new research information on individual species and species groups under the NWFP and in old-forest environments, conducted mostly since the previous science synthesis (Haynes et al. 2006).

Fungi—

Fungi are an important part of forest ecosystems. Fungi have always been a conservation challenge in terms of species identification, taxonomic designation, inventory and monitoring of furtive and seldom-appearing species, and understanding of their key ecological roles in late-successional and old-growth forest ecosystems. Many fungi are rare or little known, but much has been learned about some aspects of species occurrence and distribution since the previous science synthesis. Some of this work is presented in peer-reviewed publications, and other work is available through an agency peer review process. Recent regional work provides information on California fungal species. Other work provides a better understanding of the status of fungal species in the NWFP area.

[14] https://www.blm.gov/or/plans/rmpswesternoregon/.

Occurrence of fungal species is influenced, at least in part, by the type and intensity of disturbances, time since disturbance, and vegetation development, and by forest stand management and forest age class (Heithecker and Halpern 2006, Trofymow et al. 2003). Studies by Hebel et al. (2009) suggested that high-severity wildfire can reduce or prevent colonization of, and can kill, beneficial arbuscular mycorrhizal fungi. As noted in chapter 3 and in Reilly et al. (2017), current rates of high-severity fire in the NWFP area are very low even in the moist forests where fire was historically infrequent. In the dry forests, where fires were historically frequent, recent fire rotations have been well below the historical levels; however, in forests that historically had low-severity fire and little high-severity fire, the recent amounts of high-severity fire appear to be higher. It is unclear how current fire regimes might affect forest structure, age class, and disturbance intensity.

Presumably, native fungi species were able to persist across landscapes with frequent to very frequent fire (<50 years) and in landscapes with occasional, moderately frequent, mixed-severity fire (50 to 200 years). Fire suppression has affected the various forest ecosystems of the NWFP area in different ways (chapter 3), although little is known about effects of fire suppression activities on fungi. Luoma and Eberhard (2005) urged the conservation of rare truffle and mushroom species "... in a manner that recognizes their different responses to forest disturbance." They also hypothesized that fire suppression activities may have favored mushroom production over truffle production, and that presence of fire is a factor in the reproductive evolution of ectomycorrhizal fungi. Thus, they concluded that providing for ectomycorrhizal fungi would include restoring forest health from the adverse effects of decades of fire suppression.

Many fungi species are soil dwellers and mostly subterranean, and, with intermittent fruiting cycles, they are not easy to detect and collect. Determining presence can be difficult, but they can play major roles in nutrient uptake by trees and other plants and in aiding coarse-wood decomposition, contributing to soil organic matter, and maintaining overall forest health.

Luoma (2001)[15] found that retention of green trees in late-successional forests helped retain *Arcangeliella camphorata*, a rare truffle fungus, as compared to the species' loss in clearcuts. Trappe et al. (2009) provided an extensive study of the distribution, ecology, and conservation of truffle species in the Pacific Northwest. In southwest Oregon, Clarkson and Mills (1994) and Amaranthus et al. (1994) also had previously demonstrated that clearcuts ranging from 4 to 27 years since harvest nearly eliminated truffle production and that retention of mature trees and coarse woody debris promote their diversity and abundance. Marcot (2017) reviewed the role of fungi in wood decay of forest ecosystems of the Pacific Northwest.

Some fungi are dispersed in unusual ways, such as on the beaks of foraging and cavity-excavating woodpeckers (Jusino et al. 2016). Fungi such as truffles and their ectomycorrhizal sporocarps are key food resources for northern flying squirrels (*Glaucomys sabrinus*) (Lehmkuhl et al. 2004); in turn, flying squirrels are a key prey species of northern spotted owls in parts of the owl's range. Deliberate introduction of fungi in live trees is a management tactic sometimes used to induce wood decay and create partially dead trees and snags for wildlife use such as in western Washington (Bednarz et al. 2013).

In 1994, the Survey and Manage program listed some 234 rare fungi species associated with late-successional and old-growth forests (Molina 2008). Molina's (2008) review of mycology herberia eventually yielded 14,400 records of these species, with 55 percent of the species found at 20 or fewer sites and 42 percent found at 10 or fewer sites; it is unclear which of the 42 percent are rare or undersurveyed. Some 90 percent of the species had some fraction of their locations occurring within reserves, but only a third of the species had all their locations within reserves. This led Molina (2008) to conclude that fine-filter conservation of rare species outside reserves was needed to help ensure conservation of the entire late-successional and old-growth-associated fungal biota.

[15] Luoma, D.L. 2001. Monitoring of fungal diversity at the Siskiyou integrated research site with special reference to the survey and manage species *Arcangeliella camphorata* (Singer & Smith) Pegler & Young. Unpublished report. On file with: Chetco Ranger District, Siskiyou National Forest, Brookings, OR 97415.

Overall, although this work has provided much information on presence and distribution of rare fungi, in general their species-specific status and trends are still not well known.

The ISSSSP Fungi Work Group has compiled an annotated bibliography on rare fungi species on the Special Status Species list for California, Oregon, and Washington.[16] This work includes 174 references, each with annotated findings and indexed to keywords pertaining to ecological and management topics.

Mushroom collection for individual use is a popular recreational activity in forests of the Pacific Northwest (Trappe et al. 2009). Individual national forests in the NWFP area may have specific regulations on commercial mushroom harvests. For example, Siuslaw National Forest (2007) sells individual commercial collection permits, up to 1,000 permits per year, with permits being unlimited by weight or amount of mushrooms collected. Alexander et al. (2002) compared the economics of timber harvest with mushroom harvest in the Pacific Northwest and found that some mushrooms (e.g., chanterelles, morels) have lower value and some (matsutake) have about the same value as commercial timber, as measured by soil expectation value analysis. At present, though, we have encountered no studies on the impacts of mushroom collection on these species' populations in the NWFP area.

Lichens—

Much has been learned over the past decade about the occurrence and rarity of forest lichens (table 6-2) and the effects of forest management in the Pacific Northwest. Lichens play important ecological roles in late-successional and old-growth forests of the Pacific Northwest (chapter 3). Epiphytic macrolichens, because of their complete reliance on atmospheric sources of water and nutrients, are useful for monitoring air quality and climate (fig. 6-1) (Geiser and Neitlich 2007; similar work by Root et al. 2015 followed

[16] http://www.fs.fed.us/r6/sfpnw/issssp/documents3/cpt-fu-effects-guidelines-att3-annotated-bibliography-2013-10.docx.

Table 6-2—Selected recent findings on lichen species associated with late-successional and old-growth forest conditions within the Northwest Forest Plan area

Lichen taxa	Topic	Source
Bryoria subcana to *B. fuscescens*	Taxonomy of *Bryoria* section Implexae	Velmala et al. 2014
Dermatocarpon	*Dermatocarpon luridum* now found to not occur in the United States. The species in the Pacific Northwest (PNW) is *D. meiophyllizum*.	Glavich 2009, Glavich and Geiser 2004
Fuscopannaria	*Fuscopannaria saubinetii* (misidentification) does not occur in the PNW. Previous records were misidentifications of *F. pacifica*, a common species in the PNW.	Jørgensen 2000
Leptogium	*Leptogium burnetiae* var. *hirsutum* is now a synonym of *L. hirsutum*, which does not occur in the PNW. The taxon of conservation concern is *L. burnetiae*.	Esslinger 2015
Leptogium	*Leptogium rivale* is changed to *Scytinium rivale.* *L. teretiusculum* is changed to *S. teretiusculum*.	Otálora et al. 2014
Usnea	*Usnea hesperina* is changed to *U. subgracilis.* Previously*, U. hesperina* and *U. subgracilis* were both considered synonyms of *U. schadenbergiana; U. schadenbergiana* is distinct from *U. hesperina* and *U. subgracilis.*	Truong et al. 2013

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Figure 6-1—Some arboreal, epiphytic lichens, such as this *Alectoria* in Wind River Experimental Forest in the south Washington Cascade Range, are fully dependent on atmospheric sources of water and nutrients, and thus can serve as sensitive indicators for monitoring of air pollution and climate change.

mostly east of the NWFP area).[17] Trofymow et al. (2003) found that arboreal lichen abundance and species richness differed between mature (75 to 95 years old) and old-growth (>240 years old) conifer forest stands on Vancouver Island, and thus can serve as indicators of late-successional and old-growth forest conditions. Arsenault and Goward (2016) reported that, in inland wet conifer forests of British Columbia, some macrolichens, such as *Lobaria pulmonaria*, are good indicators of old-growth forests in some ecological conditions but not in others, highlighting the need for caution when determining species' association with, and dependence on, old-growth forest conditions. Sillett et al. (2000) showed that dispersal can limit the development of some lichens (e.g., *Lobaria oregana*) in late-successional and old-growth forests, as well as in young forest plantations.

Bokhorst et al. (2015) reported that lichen species in southern Norway have a large impact on associated invertebrate communities, varying with lichens that differ in nitrogen fixation and nutrient concentration, thallus structure, and terricolous versus arboreal or epiphytic habitat. Whether such contributions of lichen diversity to overall lichen-invertebrate communities occur within

[17] Also see: http://people.oregonstate.edu/~mccuneb/epiphytes.htm and http://people.oregonstate.edu/~mccuneb/biblio.htm.

late-successional and old-growth forests of the Pacific Northwest is apparently unstudied.

Effects of wildfire and fire suppression activities on late-successional and old-growth-associated lichens have been poorly studied. Large-scale, stand-replacing fires will likely reduce distribution and biomass of late-successional and old-growth-associated lichens over decades to centuries, as a function of the species' dispersal mechanisms and limitations (Sillett et al. 2000).

Recent studies suggest that variable-density thinning treatments in young forests of Douglas-fir (*Pseudotsuga menzeisii*) and western hemlock (*Tsuga heterophylla*) can provide habitat for a variety of lichen species (Root et al. 2010) that could complement lichen assemblages found in unthinned late-successional and old-growth forest environments. Earlier studies suggested that uniform thinning of dense forest stands provides only minor improvement of habitat of lichens and bryophytes, whereas some variable-density thinning can help promote development of tree branches lower in the canopy profile as substrates for lichens and bryophytes. Earlier studies (e.g., Neitlich and McCune 1997) also revealed that forest structure may play a greater role in determining lichen diversity and biomass than forest age per se from observations that lichen biomass can be greater in structurally diverse young forests with gaps and older remnant trees than in some old-growth forests. Neitlich and McCune (1997) suggested, for conserving epiphytic macrolichens, to protect forest gaps, wolf trees (large trees with wide-spreading crowns), and old remnant and legacy trees. Retention of propagule sources, such as old legacy trees in younger stands, and older-forest patches in cutting units, can be critical to providing for old-growth-associated epiphytes (Sillett et al. 2000) and cryptogams (Hofmeister et al. 2015).

Much work has also been done over the past decade on riparian lichens. Three groups of lichens depend on riparian areas:[18] (1) nonepiphytic, instream species (e.g., *Dermatocarpon meiophyllizum*, *Leptogium rivale*, and *Peltigera hydrothyria*); (2) epiphytic species occurring in streamside and lakeshore forest and woodland environments (e.g., *Leptogium polycarpum*, *Ramalina thrausta*, *Sticta fuliginosa*, *Usnea wirthii*, and many cyanolichen species); and (3) epiphytic species in forested wetlands, particularly in Oregon ash (*Fraxinus latifolia*) swamps (e.g., *Fuscopannaria mediterranea*, *Hypotrachyna riparia*, and many cyanolichens). A number of these riparian lichens are likely afforded habitat under the NWFP's Aquatic and Riparian Conservation Strategy. For further information on riparian lichens in the Pacific Northwest, see McCune et al. (2002) and Ruchty (2000).

As with fungi, many species of lichens and other rare and poorly known species in the Pacific Northwest are difficult to detect, inventory, monitor, and study because they require specialized expertise in the field and the laboratory. To address such problems, the Survey and Manage program spurred the development of methods for statistically determining the occurrence and frequency of rare species of lichens (Edwards et al. 2004, 2005). A guide was produced under the Survey and Manage program on natural history and management considerations of selected lichen species under the NWFP (Lesher et al. 2003).

Additionally, Miller et al. (2017) tested techniques for surveying *Lobaria oregana*, a rare canopy lichen, at the southern edge of its range in northwest California within the NWFP area. They found the species on branches of large trees in the mid-crown, and on boles of small trees near ground level, suggesting that the species benefits from cool, wet microclimates, and that maintaining such microhabitats is key to its long-term viability. They also concluded that ground surveys are useful for detecting abundant lichen species, but that tree-climbing to conduct canopy surveys also may be needed to detect low-abundance species associated with mid- or upper-canopy conditions.

Bryophytes—

Bryophytes—including hornworts, liverworts, and mosses (in general, nonvascular plants)—are a conspicuous component of many late-successional and old-growth forests of the Pacific Northwest, and many species have been part of the Survey and Manage program. Many late-successional and old-growth-associated bryophytes may be sensitive to disturbance and may require old-forest stands larger than

[18] McCune, B. Personal communication. Professor, Oregon State University, Corvallis, OR 97331.

2.5 ac (1 ha) (Halpern et al. 2012) or with more than 15 percent retention of older green trees in dispersed retention harvests (Heithecker and Halpern 2006).

Mölder et al. (2015) reported that 31 bryophyte species in north Germany can serve as ancient woodland indicators; a number of bryophyte species of the Pacific Northwest can serve similar functions (Rambo and Muir 1998, 2002). In an older study, Ryan et al. (1998) found that cryptogams in mixed-conifer forests of southern Vancouver Island were largely associated with humus, and especially coarse woody debris and rock substrates, and that old-growth forests variously excluded shade-intolerant species and selected for shade-tolerant species.

A recent taxonomic update has changed *Diplophyllum plicatum* to *Douinia plicata* (Konstantinova et al. 2013).

Pacific Northwest forests are the main source of commercially harvested moss in North America with dozens of species likely affected by the practice (Peck 2006, Peck and Christy 2006). The main mosses collected are epiphytes that require 15 to 25 years to recover, and which are most abundant in riparian and low-elevation forests and absent or much less abundant in young (<70 years) Douglas-fir plantations (Peck 2006). Commercial harvest methods of stripping entire moss mats can greatly impede recovery of the species (Peck and Frelich 2008) and may be a major, local threat to this group.[19] Peck and Moldenke (2011) reported a wide array of invertebrates (205 morphospecies) associated with subcanopy epiphytic bryophyte mats subject to commercial harvest in the Coast and Cascade Ranges of the Pacific Northwest, where more than 8.2 million lbs (3.7 million kg) of bryophytes are harvested per year. However, the impact of moss harvest on populations of these invertebrate species is undetermined.

Overall, few studies are available on the status and trends of the full suite of bryophyte species (and associated other taxa) considered under the NWFP and ISSSSP. In the NWFP area, some individual national forests in Region 6 (Washington and Oregon) have their own direction regarding commercial moss collection, although there is no overall regional direction. For example, Siuslaw National Forest and Willamette National Forest may have instituted local direction of moss collection under their special forest products or nontimber forest products regulations (e.g., Siuslaw National Forest 2007). Siuslaw National Forest's direction, for instance, permits commercial moss harvests to be limited to 16,000 lb (7258 kg) per year, from harvest areas that would be open for 12 years, with only one harvest area open at a time and rotated every 12 years, and harvest areas consisting of forest stands <110 years old (Siuslaw National Forest 2007).

Vascular plants—

Vascular plants are a conspicuous and important component of late-successional and old-growth forests in the NWFP area. The diversity of vascular plants tends to increase during structural development following logging in western hemlock-Douglas-fir, and to peak in old-growth forest conditions (Halpern and Spies 1995), but less is known in other vegetation zones of the area. Late-successional and old-growth-related vascular plants in the Pacific Northwest are generally considered to be protected under the NWFP guidelines and within LSRs, and have constituted only a small fraction of all Survey and Manage species thought to be closely associated with late-successional and old-growth forests (Marcot and Molina 2006).

Most studies of changes in plant communities over seral development have used a chronosequence (space-for-time) approach and compared composition or abundance in young logged stands with naturally regenerated stands in middle or later stages of development. These studies generally agree that understory plant species diversity increases following disturbance then decreases during canopy closure, and subsequently increases again and sometimes peaks in old-growth conditions (Halpern and Spies 1995, Jules et al. 2008). Few perennial understory plant species are either absent from, or restricted to, any specific stage of successional development, although the abundance of individual species changes over time (Halpern and Spies 1995). Some species are most abundant in old growth or are closely associated with old growth (table 6-3), but none of these species is on the current Survey and Manage species list and might be candidates

[19] For a compendium of literature on moss harvesting, see: http://bryophytes.science.oregonstate.edu/MossHarvest.htm.

Table 6-3—Vascular plant species found to be associated with late-successional and old-growth forests in the Northwest Forest Plan area[a]

Species	Halpern and Spies 1995[b]	Lindh and Muir 2004[c]	Jules et al. 2008[d]
Trees/shrubs:			
Abies amabilis	✓		
Acer circinatum	✓		
Taxus brevifolia	✓		
Tsuga heterophylla	✓		
Rhododendron macrophyllum		✓	
Vaccinium alaskaense	✓		
Vaccinium membranaceum	✓	✓	
Vaccinium parvifolium	✓		
Ribes lacustre			✓
Low/subshrubs:			
Berberis nervosa	✓		
Chimaphila menziesii			
Chimaphila umbellata	✓	✓	
Cornus canadensis	✓		
Gaultheria ovatifolia	✓		
Goodyera oblongifolia		✓	✓
Linnaea borealis	✓	✓	
Herbs, forbs, and fern allies:			
Achlys triphylla	✓	✓	✓
Adenocaulon bicolor	✓		✓
Anemone oregana			✓
Coptis laciniata	✓	✓	
Clintonia uniflora	✓		✓
Disporum hookeri	✓		
Galium spp.			✓
Lycopodium clavatum	✓		
Osmorhiza chilensis			✓
Pyrola asarifolia	✓	✓	
Senecio bolanderii			✓
Smilacena stellata			✓
Synthyris reniformis	✓		
Tiarella trifoliata	✓	✓	
Trillium ovatum			✓
Vancouveria hexandra	✓		

Table 6-3—Vascular plant species found to be associated with late-successional and old-growth forests in the Northwest Forest Plan area[a] (continued)

Species	Halpern and Spies 1995[b]	Lindh and Muir 2004[c]	Jules et al. 2008[d]
Saprophytes/root parasites:			
Corallorhiza maculata			✓
Corallorhiza mertensiana	✓		
Hemitomes congestum			
Hypopitys monotropa			
Monotropa uniflora			
Pterospora andromedea			

[a] None of these species is on the current Survey and Manage species list.
[b] Oregon Coast Range and Oregon and Washington Cascade Range.
[c] H.J. Andrews Experimental Forest, western Cascade Range, Oregon.
[d] California Klamath Province.

for such consideration. Bailey et al. (1998) suggested that understory communities are relatively resilient to past cutting as few species differed in frequency of occurrence between young- and old-growth forests in the Oregon Coast and western Cascade Ranges. These results contrast with those of Halpern and Spies (1995) but are based on a much smaller sample size (9 vs. 196 plots) and are only from the drier part of the moist forest region.

The affinity of some species of vascular plants to old-growth forest conditions has been attributed to multiple mechanisms including the presence of canopy gaps and unique microclimatic conditions, greater heterogeneity in resources, and sensitivity to disturbance (e.g., logging and fire) coupled with low rates of growth and reestablishment (Halpern and Spies 1995). Jules (1998) found that ~97 percent of *Trillium ovatum* were eliminated following clearcutting and planting of conifers. Furthermore, fragmentation of habitat had negative demographic consequences including decreased recruitment and seed production in this species (Jules and Rathcke 1999). Slow recovery rates of dispersal-limited, perennial herbs (e.g., *Trillium* spp., *Cornus canadensis*, *Clintonia uniflora*, *Disporum hookeri* var. *hookeri*) may take centuries for populations to recover following clearcutting (Kahmen and Jules 2005). Halpern and Spies (1995) suggested that long-term rotations (150 to 300 years) may be needed to maintain understory plant species that require long recovery times following disturbance. Observations are lacking on how these dynamics differ from historical dynamics following natural disturbance (e.g., fire).

The response of understory vascular plants to management is known primarily from western hemlock-Douglas-fir forests (e.g., Halpern et al. 1999, Puettmann et al. 2013) and generally suggest that certain management activities can increase later seral species in previously managed stands (i.e., clearcuts). Precommercial thinning can increase compositional similarity of shrubs and herbs between young- and old-growth stands and increase the abundance of late-seral herbs (Lindh and Muir 2004). Conversely, thinning in 60- to 80-year-old stands can increase diversity and the abundance of early-seral species but have little effect on late-seral species (Puettmann et al. 2013). North et al. (1995) found that richness of herb and shrub species was greater in a green tree retention harvest than in an adjacent clearcut and in an intact 65-year stand (also see Halpern et al. 2005).

Retention patches of late-successional and old-growth forest scattered between late-successional forest reserves might serve as conservation centers for some understory vascular plants. Nelson and Halpern (2005) studied the short-term (1 and 2 years) response of understory plants to patterns of aggregated retention harvest and found that old-forest patches larger than 2.5 ac (1 ha) helped retain populations of late-seral plant species, although disturbance within 33 ft (10 m) of the stand edge ("depth of edge" influence) was evident by the incursion of early-seral plant species. Late-seral herbs were more frequently extirpated in harvested portions of aggregated treatments as opposed to dispersed treatments

Fire exclusion may have affected understory communities in late-successional and old-growth forests in vegetation zones where fire was frequent historically (Loya and Jules 2008), but these effects are poorly understood and more research is needed. Some evidence suggests increases in shrub cover may be associated with fire exclusion (Loya and Jules 2008). Loss of bear grass (*Xerophyllum tenax*) in anthropogenically maintained savannahs has also been attributed to fire exclusion (Peter and Shebitz 2006). Studies from mixed-conifer forests in the eastern Washington Cascades and from ponderosa pine (*Pinus ponderosa*) forests suggest that a single application of prescribed fire can reduce shrub cover, but effects on understory response were minor and limited to slight increases in vegetation cover (Dodson et al. 2008) and diversity (Busse et al. 2000). Donato et al. (2009) found that almost all species in mature and old-growth stands were present following high-severity fire in the Biscuit Fire. There is some evidence that reintroduction of wildfire to dry forests of the Sierra Nevada had little effect on diversity of vascular plant species but may increase the distribution of species that may have been negatively affected by fire exclusion (Webster and Halpern 2010).

Invertebrates—

The ISSSSP provides a series of products on selected invertebrate species.[20] Range maps are provided on a number of mollusks species[21]and conservation assessments and species fact sheets are provided for all sensitive invertebrates including bumblebees, beetles, true bugs, butterflies and moths, damselflies and dragonflies, amphipods, and other taxa. The following sections summarize other recent studies on various invertebrate taxa including soil invertebrates, mollusks, and other insects and arthropods.

***Soil invertebrates*—**Soil invertebrates constitute a wide array of taxa, including microorganisms, nematodes, mites, springtails, microspiders, centipedes, millipedes, earthworms, and others, with an equally diverse set of ecological functional roles as comminutors (chewers), detritivores and saprophages (detritus-feeders), fungivores (fungi-eaters), prey, predators, parasites, and more (Berg and Laskowski 2005). In addition, there are four functional groups of arthropods—litter- and soil-dwelling species, coarse wood chewers, understory and forest gap herbivores, and canopy herbivores—that are listed as Survey and Manage species in the Oregon and California Klamath, California Cascade, and California Coast Range physiographic provinces.

Some functional roles of soil invertebrates are yet to be discovered and may be surprising. As an example, Duhamel et al. (2013) reported that fungivorous springtails may play the ecological role of transferring secondary metabolites (catalpol) from host plants to arbuscular mycorrhizal fungi that then helps prevent the mycorrhizae from being grazed, thus protecting the host plant that uses the symbiotic fungi as a source of soil nutrient update.

Soil microorganisms include protozoa, rotifers, bacteria, and others that also play key ecological functions contributing to forest health, resilience, and productivity (Luo et al. 2016). Overall, soil invertebrates, including microorganisms, of late-successional and old-growth forests in the Pacific Northwest have been little studied but likely include species that are undescribed and some that closely associate with closed-canopy, late-successional, and old-growth forest conditions (see chapter 3 for a discussion of closed-canopy and open-canopy late-successional and old-growth forests and associated degree of shade-tolerant understory plants). Several older identification guides and species lists for the Pacific Northwest or selected sites therein are available such as on oribatid mites (Moldenke and Fichter 1988), arthropods (Parsons et al. 1991), and spiders (Moldenke et al. 1987).

[20] http://www.fs.fed.us/r6/sfpnw/issssp/species-index/fauna-invertebrates.shtml.

[21] http://www.fs.fed.us/r6/sfpnw/issssp/planning-tools/species-distribution-maps.shtml#invertebrates.

Forest invertebrate species vary in their sensitivity to disturbance and to forest canopy closure and tree density conditions. For example, Brand (2002) found that species richness, frequency, and density of epigeic (litter-dwelling) springtails in the Midwest are sensitive to fire; findings included the springtail *Tomocerus flavescens,* which is also a component of Pacific Northwest forests. Brand suggested using scattered refuges to maintain survival of fire-sensitive invertebrates in the face of fires; whether this is practical in the face of increasing rates of fire in the drier forests of the NWFP area is untested. Within the NWFP area, Moldenke and Fichter (1988) documented that oribatid mites, which are diverse and numerically dominant in ecosystems worldwide, occurred in very different species groups in open- and closed-canopy forests. The closed-canopy species were found throughout all stages of closed-canopy succession but differed markedly in relative abundance between young and old forests.

Milcu et al. (2006) found various responses of earthworms, springtails, and soil microorganisms—all decomposers—to grass and herb species diversity, functional group diversity, and growth form, but that the response of the decomposers did not correlate with plant productivity. Niwa and Peck (2002) found variable responses to time since prescribed fire burning in the occurrence and abundance of species of spiders (Araneae) and beetles (Carabidae) in southwest Oregon.

Some late-successional and old-growth-associated invertebrates, particularly dispersal-limited species, can serve as indicators of biodiversity and climate change risk (Ellis 2015, Homburg et al. 2014), and indicators of response of species groups to thinning (Yi and Moldenke 2005). Such potential indicator species include flightless terrestrial carabid beetles (e.g., Brumwell et al. 1998, Driscoll and Weir 2005, Eggers et al. 2010), which are a component of the entomofauna of LSRs in the Pacific Northwest, and flightless saproxylic weevils (Coleoptera: Curculionidae) (Buse 2012), which are also found in late-successional and old-growth reserves in the Pacific Northwest.

Yi and Moldenke (2005) studied the effects of thinning in 40- to 60-year-old Douglas-fir forest and associated changes in forest floor environmental conditions in Willamette National Forest in the Oregon Cascade Range. They reported that Formicidae ants preferred heavy thinning intensities; Araneae spiders, Carabidae ground beetles, and Polydesmida millipedes positively correlated with litter moisture, which in turn was sensitive to season and negatively correlated with thinning intensity; and Gryllacrididae camel-crickets were negatively associated with litter moisture.

Rykken et al. (2007) found that invertebrates associated with headwater streams on the Willamette National Forest of Oregon, within the NWFP area, constitute spatially constrained but extremely species-rich communities. This occurred particularly within 3 ft (1 m) of the stream edge in mature forests and in 100-ft (30-m) riparian buffers within managed landscapes, such as may be provided as part of the Aquatic and Riparian Conservation Strategy of the NWFP (see chapter 7).

Caesar et al. (2005) assessed the genetic structure of a soil-inhabiting beetle (*Acrotrichis xanthocera*: Ptiliidae) in LSRs in the Klamath-Siskiyou ecoregion of northern California within the NWFP area. They concluded that the reserve system currently maintains high genetic variation for the species and suggested that intervening habitat gaps among the reserves should be reduced to maintain connectivity among the beetle populations within the reserves, apparently under the tacit assumption that the LSR system would continue in its current form under its current fire and fuels management approach (e.g., Fire suppression and limited use of prescribed fire) (see chapter 3).

Of concern for the conservation of invertebrates in LSRs is the incursion by invasive species. Of high concern are exotic earthworms (Ewing et al. 2015), e.g., *Lumbricus terrestris*, and about two dozen related species of Lumbricidae, introduced from Europe and currently found throughout North America including the Pacific Northwest although not yet in natural conifer stands. Exotic earthworms have been shown to reduce germination of tree seeds and survival of seedlings in southern Quebec, Canada (Drouin et al. 2014). In greenhouse and laboratory experiments, Eisenhauer and Scheu (2008) and Gundale (2002) found that exotic earthworms aid establishment of invader plants. In northern hardwood forests of Michigan and New York, Gundale et al. (2005) and Eusenhauer and Sheu (2008), respectively, found that exotic earthworms reduce nitrogen retention. Hendirx and Bohlen (2002) reported that exotic earthworms, in general, can adversely affect soil processes and can introduce pathogens. In northern temperate forests of North America, Bohlen et al. (2004a, 2004b) reported that invasive exotic earthworms can adversely alter soil nutrient content (total carbon and phosphorus, and carbon-nitrogen ratios) and soil food webs. In the Northeastern United States, increased abundance of white-tailed deer (*Odocoileus virginianus*) can facilitate the spread of invasive earthworms; the deer's herbivory promotes the spread of nonnative understory vegetation that, in turn, fosters less acidic soils that are more favorable to the invasive earthworm species (Dávalos et al. 2015). Ziemba et al. (2015) reported that invasive *Amynthas* earthworms are altering detrital soil communities in North America, adversely affecting eastern red-backed salamanders (*Plethodon cinereus*) (also see Ransom 2012). Other invasive invertebrates occur throughout the Western United States (e.g., Chen and Seybold 2014) but are largely unstudied within the NWFP reserve system.

Few such studies of the specific impacts of exotic earthworms have been conducted in the NWFP region. Bailey et al. (2002) compared the influence of native and exotic earthworm species on soil and vegetation in remnant forests in the Willamette Valley of western Oregon. They discovered two genera of native earthworms and the exotic earthworms were present in all five forest study sites, and reported that although there was no direct evidence that exotic species were affecting native species, they did not detect numerous native species presumed present in the region. They further suggested experimental studies to more clearly identify impacts of exotic species on native fauna.

Mollusks—Mollusks of the Pacific Northwest—particularly slugs and snails—have been the earlier focus of inventory, monitoring, and taxonomic study under the NWFP. Field guides have been produced under the Survey and Manage program for freshwater mollusks (Frest and Johannes 1999) and terrestrial mollusks (Kelley et al. 1999). Recent findings on mollusk species taxonomy are presented in table 6-4.

In a study in northern California, Dunk et al. (2004) sampled five species of terrestrial mollusks (*Ancotrema voyanum*, *Helminthoglypta talmadgei*, *Monadenia churchi*, *M. fidelis klamathica*, and *M. f. ochromphalus*). In a comparison across randomly selected sample plots of various forest conditions, they found that *A. voyanum* was associated with late-successional and old-growth forests, *M. churchi* was a habitat generalist, and data were insufficient to determine habitat associations of the remaining three species. As a result of the information on *M. churchi* being a habitat generalist, the species was removed from the Survey and Manage species list owing to not being specifically associated with late-successional and old-growth forests.

Table 6-4—Selected recent findings on taxonomy of mollusk species associated with late-successional and old-growth forest conditions within the Northwest Forest Plan area

Mollusk taxa	Topic	Source
Fluminicola (aquatic snail)	Updated taxonomy in northern California: *F.* n. sp. 14 = *F. potemicus* *F.* n. sp. 15, 16, & 17 = *F. multifarious* *F.* n. sp. 18 = *F. anserinus* *F.* n. sp. 19 & 20 = *F. umbilicatus*	Hershler et al. 2007
Lyogyrus (aquatic snail)	*Lyogyrus* n. sp. 1 is now described as *Colligyrus greggi*. The genus was transferred and then the species fully described.	Hershler 1999, Liu et al. 2015
Deroceras	*Deroceras hesperium* is now considered *D. leave*, a common and widespread slug in North America	Roth et al. 2013
Pristiloma	*P. arcticum crateris* is now changed to *P. crateris*	Roth 2015

In a study of species in old-forest leave islands (up to 1 ac or 0.4 ha) in western Oregon, Wessell (2005) found that leave island size positively correlated with overall slug and snail density, and density of three mollusk species groups. Studies conducted beyond the Pacific Northwest also indicate that mollusk species richness is related to habitat area (Horsák et al. 2012) or hardwood forest stand age (Moning and Müller 2009). Other studies elsewhere (Moss and Hermanutz 2010) suggest the importance of monitoring for nonnative gastropods, particularly slugs, and especially along the margins of disturbed (burned) areas. Invasive slugs can negatively affect native slug diversity and plant regeneration and should be included in the monitoring of protected areas (Moss and Hermanutz 2010).

Jordan and Black (2015) provided a conservation assessment and finding of a potential threat for the mollusk *Cryptomastix devia*, otherwise known as the Puget Oregonian (Jordan and Black 2015). This terrestrial Gastropoda snail is strongly associated with large, old bigleaf maples (*Acer macrophyllum*) (fig. 6-2) growing among conifers, typically Douglas-fir, western hemlock, and western redcedar (*Thuja plicata*), or among other hardwoods such as black cottonwood (*Populus trichocarpa* ssp. *trichocarpa* and red alder (*Alnus rubra*). A primary threat identified in the conservation assessment for this species is reduction or loss of large, old bigleaf maples through suppression by Douglas-fir and other conifers, and through selective commercial thinning of hardwoods. Further, over the past 10 to 15 years,

Bruce G. Marcot

Figure 6-2—Large, old bigleaf maples are important hosts for the terrestrial gastropod snail known as the Puget Oregonian. Loss of maples in the southern Washington Cascade Range from commercial thinning of hardwoods, suppression by conifers, and disease may be threatening this snail.

there has been widespread mortality of bigleaf maples in the Upper Cowlitz and Cispus River drainages of the south Washington Cascades,[22] the area with the vast majority of known locations of this mollusk. Other threats may include vertebrate and invertebrate predators (predatory snails, and beetles), and the occurrence of invasive slugs. As with nearly all Survey and Manage species, there is no ongoing research and monitoring program for this mollusk.

***Other insects and arthropods*—**Studies of macromoths in H.J. Andrews Experimental Forest in Oregon (and in companion study sites in Mount Jirisan National Park in South Korea) revealed that most species consisted of the families Noctuidae and Geometridae, and that more than 3 years of sampling are needed to determine 90 percent of species richness (Choi and Miller 2013). Miller et al. (2003) studied 15 species of uncommon to rare moths (Noctuidae: Plusiinae) in the Cascade Range and identified three guilds of conifer, hardwood tree and shrub, and herbaceous-feeding species. They concluded that uncommon and rare species with special or restricted habitat requirements add to overall biodiversity, and that diverse environments such as meadows and early-successional vegetation contribute to the species' diversity.

A number of regional species identification guides have been recently produced on insects (Acorn and Sheldon 2001, Haggard and Haggard 2006), lepidoptera (Miller and Hammond 2003), macromoths (Miller and Hammond 2000), butterflies (Pyle 2002), bumblebees (Koch et al. 2012), dragonflies and damselflies (Kerst and Gordon 2011), and others, adding to previous identification keys such as on arboreal spiders (Moldenke et al. 1987).

Additional arthropods of late-successional and old-growth forests of the Pacific Northwest include millipedes and centipedes. Taxonomy, ecology, and habitat elements of many of these species remain poorly known. A study of riparian-associated millipedes in southwest Washington by Foster and Claeson (2011) revealed 15 species among 10 families, and that millipede species assemblages varied between spring and fall and among sites. The authors urged further study on taxonomy and habitats and emphasized the importance of the ecological role of millipedes as detritivores in the forest ecosystem.

We did not find any recent autecology studies on centipedes in late-successional and old-growth forests of the Pacific Northwest. Insights into habitat associations of centipedes from studies elsewhere only suggest that some native species may be more closely associated with interior forest habitat conditions, and exotic species more associated with edge and disturbed habitat conditions (Hickerson et al. 2005).

A fairly recent management concern is the reduction in populations of pollinators on federal lands. In the Pacific Northwest, native bee populations are declining because of parasites, pathogens, pesticides, and invasive species (Spivak et al. 2011). Recent research has aimed at providing methods for restoring and conserving habitat for honeybee (*Apis* spp., nonnative species), bumblebees (*Bombus* spp.), and other pollinators (Decourtye et al. 2010, Wratten et al. 2012). The work highlights the economic benefit and ecosystem services provided by pollinating insects (Losey and Vaughan 2006).

Amphibians and reptiles—

***Amphibians and reptiles of forest systems*—**Recent studies on the distribution, movement, and habitat relationships of forest amphibians and reptiles help assess how well the NWFP has served to provide habitat for late-successional and old-growth forest-associated amphibians and reptiles.

Old-growth forest-associated amphibians and reptiles are often described as secretive because most are fossorial and only seasonally encountered on the ground surface under down wood, rocks, leaves, and moss. Nonetheless, creative study approaches and modern landscape modeling and genetics techniques have furthered our understanding of the distribution, life history, and status of these species in the Pacific Northwest. For example, genetics were used to identify two new species: Scott Bar salamander (*Plethodon asupak*) and forest sharp-tailed snake (*Contia longicaudae*). The Scott Bar salamander is associated with cool, moist forests and talus slopes

[22] Kogut, T. Personal communication. Wildlife biologist (retired), P.O. Box 258, Packwood, WA 98361.

of an extremely restricted range in northern California; and, similar to other plethodontid salamanders (lungless salamanders of the family Plethodontidae), it is likely sensitive to changes in its habitat and in microclimate conditions (DeGross and Bury 2007, Mead et al. 2005). The forest sharp-tailed snake is associated with mesic and dense canopied forests of northern California and southwestern Oregon (Feldman and Hoyer 2010). Although little is known about this new species, the more wide-ranging species, *C. tenuis*, displays life history characteristics such as slow growth, late maturity, and low fecundity that are consistent with functional rarity and intrinsic vulnerability to population declines (Govindarajulu et al. 2011).

However, in an analysis of conservation risk of wildlife, including amphibians and reptiles, of Washington and Oregon, Lehmkuhl et al. (2001) determined that although life history traits can be used to qualitatively determine potential vulnerability, alone they are not an adequate quantitative predictor of vulnerability without additional information on species' habitat selection, habitat breadth, demography, and other factors. Clearly, additional survey and monitoring efforts and research are needed to better delineate species ranges and to improve our understanding of species' distributions, status, habitat requirements, and potential vulnerability (Gibbs 1998).

Another example of a furtive species yielding surprises upon study is the Shasta salamander (*Hydromantes shastae*) that, prior to 2012, was thought to be endemic to limestone outcroppings in the Shasta Lake watershed (Mooney 2010). Management decisions and land use project planning were based on this assumed association. Yet, Shasta salamanders subsequently have been found at sites lacking limestone outcrops and in an adjacent watershed (Lindstrand et al. 2012, Nauman and Olson 2004).

The Survey and Manage program and the ISSSSP have greatly increased scientific knowledge of forest-dwelling salamanders and their relationships with environments in the Pacific Northwest. Conservation Assessments for seven terrestrial salamanders (*Aneides flavipunctatus*, *Batrachoseps attenuatus*, *B. wrighti*, *Plethodon larselli*, *P. stormi*, *P. vandykei*, and *P. asupak*) have been completed since 2005.[23] The Density Management and Riparian Buffer Study of western Oregon (Anderson et al. 2007, Burton et al. 2016, Olson and Burton 2014) was initiated in 1994 when the NWFP implemented riparian buffers along non-fish- bearing streams (Cissel et al. 2006). This study assessed the long-term effects of thinning and differing riparian buffer widths on the distribution, abundance, and movement of terrestrial and aquatic amphibians. It showed that terrestrial salamanders are more abundant and more mobile in unthinned, densely canopied (with visible sky ranging 5 to 7 percent) (Anderson et al. 2007) riparian buffers as compared to nearby upland thinned forest (with visible sky ranging 9 to 12 percent) (Anderson et al. 2007; Kluber et al. 2008; Olson et al. 2014a, 2014b). This work highlights the importance of maintaining areas of dense canopies within at least 50 ft (15.2 m) of perennial streams in managed forests to serve as habitat for terrestrial species and as corridors for their dispersal (Olson et al. 2014b).

Recent studies have determined that the assumed dependence of terrestrial salamanders to late-successional and old-growth forests is more nuanced than previously thought (Bosakowski 1999). Some species have been found to be more abundant or occur more frequently in late-successional and old-growth forests (*B. attenuatus*, *Ensatina eschscholtzii*, Welsh and Hodgson 2013; *P. elongatus*, Welsh et al. 2008), but studies are inconsistent (e.g., *P. stormi*, Bull et al. 2006, Suzuki et al. 2008, Welsh et al. 2008). Relationships between plethodontid salamanders and cool, moist microhabitat conditions characteristic of late-successional and old-growth forests have been identified, but these conditions can be provided by other environmental features such as proximity to streams, talus slopes, and shading by vegetation, aspect, or topography (*P. stormi*, Suzuki et al. 2008; Welsh et al. 2007; *P. larselli*, Crisafulli et al. 2008). For species associated with forest floor attributes such as amount of down wood, a high volume of down wood may mediate the otherwise adverse effects of changes to the microhabitat conditions after thinning (*E. eschscholtzii*, Welsh et al. 2015; *E. eschscholtzii* and *P. vehiculum*, Kluber et al. 2008, Rundio and Olson 2007; *B. wrighti*, Clayton and Olson 2009).

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[23] http://www.fs.fed.us/r6/sfpnw/issssp/species-index/fauna-amphibians.shtml.

Substrate type and availability are important correlates of some species (*P. vehiculum*, Kluber et al. 2008; *P. larselli*, Crisafulli et al. 2008; *H. shastae*, Mooney 2010; *P. vandykei*, McIntyre et al. 2006, Olson and Crisafulli 2014), and down logs may provide thermal refugia for salamanders in thinned upslope zones (Kluber et al. 2009). In areas deficient in preferred substrates that provide cool refugia, salamanders may be associated with old-growth forest and related microhabitat conditions (e.g., *P. larselli*, Crisafulli et al. 2008). Maintaining cool and humid refugia may be central in protecting populations in areas that may experience disturbances.

Recent studies have started to unravel the complex trophic relationships of amphibians in forest and woodland food webs. Best and Welsh (2014) noted that plethodontid salamanders serve as predators of invertebrates, and as prey for carnivores. In an experiment, the authors found that *E. eschscholtzii* predation on diverse invertebrate prey suppressed some invertebrate taxa and released others resulting in increased retention of leaf litter and increased carbon sequestration, although such functions varied by the timing and amount of precipitation. We expect many other species of reptiles and amphibians play ecological roles influencing the trophic and nutrient dynamics of forest ecosystems to varying degrees, although quantitative studies are few and wanting.

Amphibians of stream systems—

The Aquatic Conservation Strategy under the NWFP provides for conservation and restoration of selected stream and riparian systems of the Pacific Northwest (Reeves et al. 2006). Although most protections are for fish-bearing-streams, the importance of protecting non-fish bearing headwater streams and associated riparian environments has been increasingly recognized (Sedell and Froggatt 1984, Wilkins and Peterson 2000). First- and second-order headwater channels comprise the majority of stream miles (kilometers) within the Pacific Northwest and serve as cold water refugia in a warming climate (Isaak et al. 2016). In addition, they serve as headwater linkages between watersheds that provide important dispersal corridors for aquatic and terrestrial species, alike (Olson and Burnett 2009, Richardson and Neill 1998). Within stream networks, amphibians occur from the headwaters to the alluvial flood plains, and their community structure is closely tied to channel types and within-channel attributes (Welsh and Hodgson 2011). For this reason, combined with their sensitivity to environmental perturbations, stream amphibians have been recognized as biometrics of stream health (Welsh and Hodgson 2008). In a study predating the NWFP, in the Oregon Cascades and Coast Range, Corn and Bury (1989) found that four species of aquatic amphibians—coastal giant salamander (*Dicamptodon tenebrosus)*, Olympic salamander (*R. olympicus*), coastal tailed frog (*Ascaphus truei*), and Dunn's salamander (*P. dunni*)—had greater occurrence and abundance in streams flowing through uncut forests than in forests logged 14 to 40 years ago, and that tailed frogs and Olympic salamanders may be extirpated from headwaters in clearcuts. More recently, Olson et al. (2014a, 2014b) found that streambank amphibian species, including Dunn's salamander and western red-backed salamander, were sensitive to forest thinning within 20 to 49 ft (6 to 15 m) of small streams including headwaters.

Research has greatly advanced in the Pacific Northwest to incorporate amphibians to answer questions about the role of riparian buffers in the retention of aquatic ecosystem services. Studies have addressed how wide no-entry buffer zones should be to retain sensitive amphibian species or their critical habitat conditions and whether no-entry buffers are needed under differing upslope ecological forestry approaches. They are also asking whether or not it is appropriate to thin riparian zones to accelerate riparian restoration to achieve biodiversity goals in some cases. Using a meta-analysis to test the effectiveness of riparian buffers for conserving terrestrial fauna, Marczak et al. (2010) reported that riparian buffers, in general, maintain fewer amphibians compared to riparian forests in unharvested areas. Stream-associated amphibians such as coastal tailed frog and torrent salamanders are highly sensitive to timber harvest and the associated instream effects of increases in water temperature and sedimentation rates (Ashton et al. 2006, Bury 2008, Emel and Storfer 2015, Olson et al. 2007a, Pollet et al. 2010, Welsh and Hodgson 2011). Coastal giant salamanders appear to be more resilient to timber harvest practices than are other stream-associated amphibians of the Pacific Northwest (Leuthold et al. 2012, Pollet et al. 2010). While the general findings that certain species are more sensitive to timber harvest than others holds relatively consistent in the literature, alternative findings in some areas (e.g., Raphael et al. 2002) highlight the importance of recognizing that abiotic and biotic factors interact with management actions in complex ways to influence the distribution and density of stream amphibians (Kroll et al. 2009).

The preponderance of evidence suggests that streams with riparian buffers within harvested landscapes do help to ameliorate some of the adverse effects of timber removal on instream conditions (Olson et al. 2014a, Pollet et al. 2010, Stoddard and Hayes 2005. In a replicated field experiment, Olson et al. (2014a) concluded that the NWFP Aquatic Conservation Strategy riparian buffers of ~230 to 476 ft (~70 to 145 m) seem to protect the headwater amphibian biota, and in a multiscaled survey, Stoddard and Hayes (2005) found that presence of a 150-ft (46-m)-wide riparian buffer predicted increased occurrence of tailed frogs and two species of torrent salamanders in a managed landscape. In the Oregon Coast Range, Kluber et al. (2008) found that ground surface attributes such as amount of rock or fine substrate determined the response of riparian and upland amphibian species to forest thinning along headwater streams, and that variable-width riparian buffer retention can provide for such microhabitat conditions. Olson and Burton (2014) examined the effects of alternative riparian management approaches including three no-harvest buffer treatments (site-potential tree-height (~230 ft [~70 m]), variable width with a 49-ft (15-m) minimum buffer on each side of the stream, and streamside retention (~20 ft [~6 m]) and a thin-through treatment whereby overstory tree density in a ~476-ft (~145-m) buffer treatment was reduced from 430 to 600 trees per acre (hectare) to ~150 trees per acre. They found that densities of torrent salamanders decreased along streams with the narrowest buffer, ~20 ft (~6 m), and that densities of Dunn's salamanders and coastal giant salamanders decreased in thin-through buffers (Olson and Burton 2014). The authors recommend the use of a 49-ft (15-m) or wider buffer to retain headwater stream amphibians (Olson and Burton 2014).

Riparian buffers can also maintain habitat connectivity and headwater linkage areas for dispersing amphibians. Using landscape genetics, Emel and Storfer (2015) found that gene flow among populations of the southern torrent salamander (*R. variegatus*) becomes restricted in streams with low canopy cover and high heat loads. The authors suggest that maintaining stream corridors with continuous riparian buffers may increase connectivity in managed landscapes. Coastal tailed frogs disperse more readily through intact forests than clearcuts (Wahbe et al. 2004) and, thus, may benefit from headwater management for connectivity across ridgelines, or the creation of "headwater linkage areas" (Olson and Burnett 2009). Olson and Burnett (2009) recommend linking headwater drainages across 7th-code hydrologic units to maintain landscape connectivity for headwater species. They propose extending buffers or alternative forest management practices that maintain canopy structure and shading to link neighboring watersheds over ridgelines (Olson and Burnett 2009, Olson et al. 2007).

***Amphibians of still-water systems*—**In the Pacific Northwest, still-water (lentic) environments such as lakes, ponds, and wet meadows, and their adjacent riparian and terrestrial environments, also provide for species diversity. Many such sites do not provide habitat for fish, however, if they are impermanent. Lentic breeding amphibians such as red-legged frogs (*Rana aurora*), Oregon spotted frogs (*R. pretiosa*), Columbia spotted frogs (*R. luteiventris*), Cascades frogs (*R. cascadae*), northwestern salamanders (*Ambystoma gracile*), western toads (*Anaxyrus boreas*), and Pacific chorus frogs (*Pseudacris regilla*) flourish in these environments. Yet modifications and disturbances to these species' habitats (e.g., damming to increase water storage, ditching to increase drainage, siltation owing to timber harvest in the watershed, and overgrazing), combined with diseases and the spread of invasive species such as American bullfrogs (*Lithobates catesbeianus*) and brook trout (*Salvelinus fontinalis*), have resulted in some population declines (Adams 1999, Fisher and Shaffer 1996, Pearl et al. 2007).

The Oregon spotted frog is one of the most threatened amphibians in the Pacific Northwest; much information on this species can be found from the U.S. Fish and Wildlife Service Federal Register of notice of threatened status for this species (USDI FWS 2014). Blouin et al. (2010) found that low connectivity between populations has resulted in decreased genetic diversity, and they recommended maintaining habitat connectivity and expanding the availability of appropriate wetlands.

Whereas palustrine wetlands and the amphibians that depend on them are not directly protected under the NWFP, riparian buffers likely benefit these species. In a literature survey of the use of wetland buffers by reptiles and amphibians, Semlitsch and Bodie (2003) found that core habitat of wetlands, as used by breeding populations, ranged from 522 to 951 ft (159 to 290 m) for amphibians and 417 to 948 ft (127 to 289 m) for reptiles from the edge of the aquatic site, and thus that adjacent terrestrial environments also were essential for population persistence.

***Recent and ongoing issues for amphibians: fire and timber harvest*—**For most of the species discussed, effects of large disturbances such as wildfire and broad-scale timber harvest on populations, have not been adequately studied, and monitoring programs are not in place to assess the status of species and effectiveness of the NWFP in protecting these species. With recent decades of fire suppression in the Pacific Northwest, most of the forests have experienced much less fire than they did historically (see chapters 2 and 3). The effect of this change on habitats and populations of wildlife, including amphibians that require shaded, cool, moist microsites, is not known.

For the Larch Mountain salamander (*P. larselli)*, Van Dyke's salamander (*P. vandykei)*, California slender salamander (*B. attenuatus)*, and Shasta salamander (*H. shastae)*, a high proportion of known occurrences are within federal reserves associated with the NWFP where timber harvest is minimal compared to on nonreserve lands and thus these species are more protected, at least from timber harvest, under the NWFP. Species with a significant portion of their range on nonreserve land allocations (Siskiyou Mountains salamander (*P. stormi)* and Scott Bar salamander (*P. asupak)*, Clayton et al. 2005, Nauman and Olson 2008; Oregon slender salamander (*B. wrighti)*, Clayton and Olson 2009; black salamander (*A. flavipunctatus)*, Olson 2008), however, might still be negatively affected by future land management activities if they deviate from the LSR standards, and management of these lands may be crucial to protect these species. Furthermore, much of the range of many plethodontids is on nonfederal forest land (Suzuki and Olson 2008). Adaptive management areas (AMAs) and matrix lands were expected to function as experimental areas to address unresolved questions and to conduct long-term studies on timber harvesting and fire effects on forest salamanders and their response to these disturbances (Stankey et al. 2006, Suzuki and Olson 2008). However, the AMA program was never fully instituted under the NWFP, although AMAs were designated.

***Recent and ongoing issues for amphibians: climate change*—**Climate change models for the Pacific Northwest region predict a warming of 0.2 to 1.1 °F (0.1 to 0.6 °C) per decade, with wetter autumns and winters, but drier summers (Mote and Salathe 2010). Significant warming is projected for the Pacific Northwest by the end of the 21st century, with simulation results projecting an increase in warming of 5.9 to 17.5 °F (3.3 to 9.7 °C) in Washington and Oregon and 2.7 to 8.1 °F (1.5 to 4.5 °C) in northern California, with generally increasing aridity but with higher uncertainty about specific precipitation levels (see chapter 2).

Changes in the climate can affect amphibians by altering habitat features such as vegetation, soil, and hydrology directly or via interactions with other threats such as timber harvest, wildfire, and disease (Blaustein et al. 2010). Plethodontid salamanders are particularly vulnerable to climate warming because they are specialized to cool microclimates, have limited dispersal capabilities, and may lack physiological tolerance to warm-induced stresses (Bernardo and Spotila 2006, Velo-Antón et al. 2013). Climate change may influence the distribution of suitable environments, which can result in fragmenting populations and shifting or retracting species ranges (Blaustein et al. 2010). More precipitation during the autumn and winter may lengthen the period of surface activity, but warmer and drier summers could result in fewer available surface and subsurface refugia that prevent desiccation. Forest management practices that retain canopy cover, and maintain or supplement surface refugia such as down wood and logs may help to ameliorate the effects of climate change (Shoo et al. 2011).

***Recent and ongoing issues for amphibians: pests and pathogens*—**Minimal research has been conducted on the effects of nonnative animals or plants on amphibians in the Pacific Northwest, and the degree to which the NWFP influences establishment and spread of such invasives. Introduced American bullfrogs (*L. catesbeianus*) have become widely established in lentic environments in the Pacific Northwest and are considered an important predator of native pond-breeding amphibians. Most documented effects are from lowland areas (<0.2 mi [<240 m]) such as the Willamette Valley where Pearl et al. (2004) found greater effects of bullfrogs on Oregon spotted frogs compared to northern red-legged frogs. Breeding bullfrog populations have been documented from the Oregon Cascades (Garcia et al. 2009), but environmental conditions such as high UV-B levels (Garcia et al. 2015) may preclude extensive colonization.

As noted previously, the invasive Asian earthworm *Amynthas* that has invaded much of North America has been found to alter the behavior of eastern red-backed salamanders, but similar studies have not been conducted with salamander species found in the Pacific Northwest.

The threat of novel diseases entering populations of amphibians of the Pacific Northwest appears increasingly pressing. The deadly amphibian disease, chytridiomycosis, emerged in the 1970s and has become a prominent threat to amphibian biodiversity worldwide (Olson et al. 2013, Skerratt et al. 2007). Chytridiomycosis is caused by the fungus *Batrachochytrium dendrobatidis* (*Bd*) and has been identified as causing decline or extinction of over 200 amphibian species globally (Skerratt et al. 2007). It affects a broad host-range among amphibians, including salamanders, and has been reported in 516 (42 percent) of 1,240 amphibian species evaluated (Olson et al. 2013). *Bd* is widespread in the Pacific Northwest and has been found in northern red-legged frog, Columbia spotted frog, Oregon spotted frog, and Cascades frog (Pearl et al. 2007, 2009). Although the risk posed by *Bd* to these species is mostly unclear, evidence suggests that Cascades frogs at the southern extent of their range in northern California have experienced severe declines owing to chytridiomycosis (De León et al. 2016, Piovia-Scott et al. 2014, Pope et al. 2014).

More recently, in 2010, a second infectious chytrid pathogen, *B. salamandrivorans* (*Bsal*), emerged in Europe where it has decimated fire salamanders (*Salamandra salamandra*) (Martel et al. 2014). Unlike *Bd*, *Bsal* appears to affect only salamanders, not anurans (Van Rooij et al. 2015). As *Bsal* is not yet known to occur in North America, amphibian disease specialists are bracing for its arrival, which will probably occur through international pet trade routes (Grant et al. 2016, Gray et al. 2015, Martel et al. 2014). Models of amphibian habitat suitability predict that the Pacific Northwest is among the highest risk areas in North America (Richgels et al. 2016, Yap et al. 2015). The Pacific Northwest represents a hotspot for salamander biodiversity and contains numerous species from the two most *Bsal*-susceptible families, Plethodontidae and Salamandridae (Martel et al. 2014). In January of 2016, the U.S. Fish and Wildlife Service responded to the threat of *Bsal* by enacting a temporary rule restricting the importation of 201 species of salamanders for the pet trade (USFWS 2016). Although a critical step, the risk of spread of *Bsal* to North America, home to the world's richest salamander fauna, is still great and surveillance, research, and management actions are mandatory for successful mitigation of spread and response to this emerging infectious disease (Grant et al. 2016, Gray et al. 2015, Yap et al. 2015).

***Recent and ongoing issues for amphibians: connectivity*—**The movement capabilities of most Pacific Northwest amphibians, especially plethodontid salamanders, are presumed to be limited, and fragmentation of their habitats may inhibit gene flow, and thus could isolate subpopulations. In forest watersheds with harvested upland areas, amphibians may disperse along riparian corridors that provide cooler, humid conditions (Olson et al. 2007a, 2014b). Stream buffers, riparian corridors, and down wood "chains" may provide connectivity for terrestrial species between their riparian and upland habitats, their adjacent stream and terrestrial habitats, and over ridges (Olson and Burnett 2009, 2013; Olson et al. 2014b). Providing linkage areas between adjacent watersheds and using various combinations of alternative management approaches (riparian buffers, thinning, down wood, leave islands, and uncut blocks) to retain forested areas along headwater ridgelines may facilitate upland dispersal and connectivity between subpopulations (Olson and Burnett 2009, 2013).

***New methods for amphibians*—**Recently, environmental DNA has been applied to identifying the presence of endangered freshwater biodiversity including amphibians in lakes, ponds, and streams (Thomsen et al. 2012). Environmental DNA originates from cellular material shed by organisms (via skin, excrement, etc.) into aquatic environments that can be sampled, sequenced, and assigned back to the species of origin. Such methodology is important for the early detection of invasive species as well as the detection of rare and cryptic species. Environmental DNA approaches have been applied to imperiled aquatic amphibians in the Southeastern United States (McKee et al. 2015); an invasive salamander in Australia (Smart et al. 2015); and the trematode *Ribeiroia ondatrae*, a pathogenic parasite on North American amphibians (Huver et al. 2015). They have recently been applied to amphibians of aquatic systems in the NWFP area (e.g., Welsh and Cummings).[24]

Birds—

Bird species pertinent to the development of the LSRs and late-successional and old-growth forests of the NWFP are covered elsewhere (see chapters 4 and 5). Many other bird species live in or migrate through the NWPF area, contributing to the region's biodiversity; here we address new science since the 10-year synthesis.

Recent studies have addressed various aspects of avian ecology in late-successional and old-growth forests, for example on pileated woodpecker (*Dryocopus pileatus*) (Aubry and Raley 2002a, Raley and Aubry 2006) and its role as a keystone species providing such ecosystem engineering functions as cavity excavation (Aubry and Raley 2002b). Other studies have provided information on forest habitat selection by white-headed woodpeckers (*Picoides albolarvatus*) (Lorenz et al. 2015), black-backed woodpeckers (*P. arcticus*) (Bonnot et al. 2009), and others. These studies suggest the role of disturbance (fire and forest pathogens) in providing habitat for these species.

For example, white-headed woodpeckers were found to establish home ranges within forest patches that had undergone recent disturbance including small patches of prescribed burning and incidence of disease. Other studies have suggested that habitat associations of white-headed woodpeckers may differ by location and condition. Latif et al. (2015) found that the species in dry conifer forests on the east side of the Cascade Range in Oregon was closely associated with canopy openings adjacent to closed-canopy forests. In unburned, dry conifer forests of central Oregon, Hollenbeck et al. (2011) found that nest sites of white-headed woodpeckers were associated with low elevation, high density of large trees, low slope, and interspersion–juxtaposition of low- and high-canopy cover ponderosa pine patches. In postfire ponderosa pine forests of south-central Oregon, Wightman et al. (2010) found that low nest survival of white-headed woodpeckers was due to predation and was not correlated with habitat and abiotic features, and that survival in recently burned forests is likely enhanced with mosaic burn patterns with retention of larger, decayed snags.

Fontaine et al. (2009) studied response of bird communities to single fires (2 to 3 and 17 to 18 years following fire), repeated high-severity fire (2 to 3 years after fires repeated at 15-year intervals), and stand-replacement fire (>100 years following fire) in mature and old-growth mixed-evergreen forests of the Klamath-Siskiyou region of southwest Oregon. They reported that bird species richness (number of bird species) did not differ significantly among the various postfire conditions; that bird density was lowest 2 years after fire and highest 17 to 18 years after fire; and that repeat-fire conditions favored shrub-nesting and ground-foraging species, and unburned mature forests favored conifer-nesting and foliage-gleaning species. In that ecoregion, the researchers reported that bird communities were structured mostly by conditions of broadleaf hardwoods and shrubs, by extended periods of early-seral broadleaf dominance and short-interval high-severity fires.

Natural disturbance patterns, including fire, windthrow, and other forces, can leave remnant and legacy elements of older forests such as old-forest patches, large-diameter green trees and snags, and large-diameter down wood, as key habitat elements for a variety of species. Subsequent regrowth of the vegetation through secondary succession then creates early-seral structures

[24] Unpublished data.

more complex than those resulting from final timber harvests such as clearcuts. The complexity of naturally regenerating vegetation following natural disturbances can often contain habitat for a wide variety of birds and other wildlife associated with early-successional environments (Marcot 1983, 1985; Swanson et al. 2011).

Vogeler et al. (2013) adopted the use of light detection and ranging to evaluate forest structure conditions for brown creepers (*Certhia americana*), an old-forest associate. In the Eastern United States, Poulin and Villard (2011) determined that brown creeper nest survival was greater in forest interiors at least 328 ft (100 m) from the forest edge, and that young forest conditions—plantations from previous management—in the managed matrix can adversely influence productivity of the species.

In a study of the influence of forest land ownership on focal species, McComb et al. (2007) projected habitat decreases for warbling vireo (*Vireo gilvus*) and increases for pileated woodpecker, as early-successional vegetation is projected to decline in area and older forests to increase. They also noted that land ownership patterns affected dispersion of habitat for both species, and that, for wide-ranging species, public lands provided habitat less available or not present on adjacent private lands.

On southeastern Vancouver Island, British Columbia, Canada, just outside the NWFP area, Hartwig et al. (2003) noted that pileated woodpeckers selected for largest diameter trees and for patches of older forests for nest cavities in coastal forests of Douglas-fir and western hemlock. They recommend retaining large live and dead grand fir, Douglas-fir, and red alder for pileated woodpecker nest sites, and to provide mature climax stands with greater proportions of bigleaf maple and grand fir.

On the Washington Pacific coast, Aubry and Raley (2002a) found that pileated woodpeckers selected Pacific silver fir (*Abies amabilis*) for nesting and western redcedar for roosting, and selected against western hemlock for both activities; that trees used for just nesting ranged from 26 to 61 inches (65 to 154 cm) diameter at breast height (d.b.h.), and trees used for just roosting ranged from 61 to 122 inches (155 to 309 cm) d.b.h.; that trees used for both nesting and roosting were at least 90 ft (27.5 m) tall; and that trees less than 49 inches (125 cm) d.b.h. or less than 57 ft (17.5 m) tall were generally selected against. The authors suggested that habitat management for the species may need to also consider roost trees by providing snags and decadent live trees with heart-rot decay fungi. Raley and Aubry (2006) further reported that pileated woodpeckers in coastal forests foraged almost exclusively in closed-canopy stands, that they selected for relatively tall, large-diameter snags in early to moderate decay stages, and that they did not appear to use down logs for foraging because down logs did not support their primary prey of carpenter ants.

Great gray owls are considered late-successional and old-growth forest associated within the NWFP area and are still a Survey and Manage species. A conservation assessment was developed for the species in 2012, and the most recent survey protocol is from 2004, as posted on the ISSSSP site.

Carnivores—

This section summarizes new science associated with the taxa of mammalian carnivores that were either ranked as “not likely to remain well distributed” by the FEMAT or that had state or federal listing status. One species and one subspecies met both criteria (the fisher and the coastal population of the Pacific marten), and two species met the latter criterion only (the lynx [*Felis lynx*] and wolverine).

Taxonomic and listing updates of carnivores—Recent phylogenetic studies have changed the relationships of several of the species considered here. Within the Mustelidae, the fisher—formerly recognized as *Martes pennanti*—has been shown to be more closely related to the wolverine and tayra (*Eira barbara*) than to martens (*Martes* sp.) (Koepfli et al. 2008, Sato et al. 2012). Consequently, the fisher is now recognized as being the sole extant member of its own genus—*Pekania*—and is now recognized as *P. pennanti*. In April 2016, the U.S. Fish and Wildlife Service determined that the West Coast Distinct Population Segment of fisher does not warrant federal listing status. No other federal listing pertains to the species in the NWFP area.

Dawson and Cook (2012) evaluated phylogenetic and morphological evidence across the distribution of the American marten (*M. americana*) in North America and agreed with previous work (Carr and Hicks 1997) that there are two species of martens in North America. The American marten occurs east of the crest of the Rocky Mountains and across northern Alaska, and the Pacific marten (*M. caurina*) occurs west of the Rocky Mountains and includes the NWFP region. Dawson and Cook (2012) found significant genetic substructuring consistent with existing subspecific designations in the Pacific marten, and Slauson et al. (2009) agreed with these designations except in coastal California and coastal Oregon. Populations of the Pacific marten in coastal Oregon and coastal California appear to be a single evolutionary unit and have been proposed to be referred to collectively as the Humboldt marten (*M. c. humboldtensis*), a taxon formerly described as occurring only along the coast in northern California (Grinnell and Dixon 1926, Slauson et al. 2009). As of this writing, the U.S. Fish and Wildlife Service has determined that there is insufficient data to warrant federal listing of the Humboldt marten, but this decision is being appealed. The Humboldt marten is a state candidate species in California and is under a 1-year review for listing there.

The contiguous U.S. Distinct Population Segment of Canada lynx is currently listed as threatened in Washington, Oregon, and beyond[25] (USFWS 2000). On August 13, 2014, the Fish and Wildlife Service withdrew a proposal to list the North American wolverine in the contiguous United States as a threatened species under the Endangered Species Act; but then on April 4, 2016, the U.S. District Court for the District of Montana ordered U.S. Fish and Wildlife Service to reconsider whether to list the wolverine as a threatened species, and it currently sits as proposed threatened status.[26] A proposed rule for listing of wolverine will be finalized in the future (Further, recent information on life history of each species is available through Web sites noted above).

[25] http://ecos.fws.gov/ecp0/profile/speciesProfile?sId=3652.

[26] http://ecos.fws.gov/ecp0/profile/speciesProfile?sId=5123.

Current distribution of carnivores—In the NWFP area, the fisher continues to be restricted to a single native population in northwestern California and southwestern Oregon (Lofroth et al. 2010), a single small introduced population in the southern Oregon Cascades from source populations in Canada and Minnesota (Aubry and Lewis 2003), and recently reintroduced populations on the Olympic peninsula (Lewis 2014) and in the Washington Cascades (Lewis 2013).

The Pacific marten appears to remain fairly well distributed in the high-elevation forests of the inland mountain ranges but occupies less than 5 percent of its historical range in the coastal forests of California and less than 10 percent of its historical range in Oregon (Slauson et al., in press), and it is known from a single recent verifiable record from the Olympic peninsula.[27]

In the NWFP area, lynx are known only from high-elevation (>4,600 ft [1400 m]) spruce (*Picea* spp.) and lodgepole pine (*Pinus contorta*) forest habitats of Washington in the northern Cascades, and from disjunct mountain ranges in the counties bordering Canada.

Verifiable wolverine occurrences in the NWFP area continue to be limited to high-elevation alpine and subalpine habitats in the central and northern Cascades of Washington (Aubry et al. 2007). However, recent detections in northeastern Oregon[28] and of a single male in the northern Sierra Nevada (Moriarty et al. 2009) suggest that wolverines have the potential to occur in the Cascades of Oregon but have yet to be detected there.

Habitat and feeding ecology of carnivores—We summarize here the new insights on habitat relationships of each species, and how these relate to each species' key life history needs for feeding, resting, and reproducing. Where possible, we link these new insights to management actions that may benefit each species. Our emphasis is primarily on martens and fishers, which occur across a larger proportion of the NWFP area than do lynx and wolverine.

[27] K. Aubry, personal communication, Emeritus scientist, Pacific Northwest Research Station, 3625 93rd Ave., Olympia, WA 98512.

[28] A. Magoun, personal communication, Wildland Research and Management, 3680 Non Road, Fairbanks, AK 99709.

Fisher—In the NWFP area, fisher home ranges vary from 2.9 to 24.5 mi^2 (7.4 to 63.5 km^2) for males and 0.7 to 49.5 mi^2 (1.7 to 128.3 km^2) for females (Lewis 2014, Lofroth et al. 2010), with home ranges decreasing in size from north to south. Overall, home ranges tend to be composed of mosaics of forest stand types and seral stages, but still include high proportions of mid- to late-seral forest (Raley et al. 2012). Notable exceptions exist in the coastal forests in northwestern California, where fishers occur both in areas with mosaics of young regenerating forest and residual mature forest (Matthews et al. 2013), as well as in areas with high proportions of young regenerating forest (Thompson et al. 2008). Higher use of young regenerating forest in these areas is related to the abundance of dusky-footed woodrats (*Neotoma fuscipes*), which are an important prey species (Lofroth et al. 2010) and which reach peak densities in young even-aged stands 5 to 20 years postharvest (Hamm and Diller 2009). Fishers typically avoid entering open areas devoid of or with significantly reduced overhead cover and escape cover, and exhibit avoidance by positioning home ranges to minimize overlap with large open areas (Raley et al. 2012) or by moving within home ranges to avoid such large open areas.

Fishers are dependent on large-diameter live and dead trees and downed logs with features such as cavities, platforms, and chambers as daily resting sites and seasonal den sites for females raising young (Raley et al. 2012). Female fishers are obligate cavity users for reproduction, with cavities in trees and snags providing secure environments for kits by regulating temperature extremes and providing protection from predators (Raley et al. 2012). These structures take hundreds of years to develop (Raley et al. 2012), emphasizing the critical importance of retaining these features for conservation of the species.

Rest structures for fishers primarily include large deformed or deteriorating live trees and secondarily include snags and logs (Raley et al. 2012, Weir et al. 2012). The species of trees and logs used for resting appear less important than the presence of a suitable microstructure, such as a cavity or platform. A meta-analysis by Aubry et al. (2013) of resting site selection by fishers across the NWFP area and adjacent areas found that rest site characteristics selected by fishers were consistent across the Pacific States and British Columbia, and occurred on steeper slopes, in cooler microenvironments, with denser overhead cover, greater volumes of logs, and a greater prevalence of large-diameter trees and snags than were generally available. This meta-analysis provides managers with empirical support for managing for these conditions where maintenance or restoration of resting habitat conditions is an objective, even in locations where fishers have not been studied. Cavities used by female fishers for birthing and preweaning periods of kits were most often created by heart rot in older trees, and most often in the largest diameter trees available (Raley et al. 2012). During the postweaning period, female fishers continue to use tree cavities but also make use of downed logs.

Truex and Zielinski (2013) studied the influence of fuel treatments in forests of the Sierra Nevada, California, and found that mechanical treatments and fire treatments together, but not separately, reduced resting sites for fishers. Also, late-season burns adversely affected resting sites, but early-season burns did not. The authors suggested evaluating the effects of fuel treatment activities at various scales—fisher resting sites, home range, and landscape—to best balance fuel reduction with restoration and maintenance of fisher habitat.

Marten—In the NWFP area, martens occur in two distinct forest regions: (1) high-elevation forests of the inland mountain ranges, such as the Cascades of California, Oregon, and Washington and the Marble-Salmon-Trinity Mountains of California, that receive significant snowfall and (2) low- to mid-elevation coastal forests in Oregon and California that receive little snowfall. In mesic west-slope forest types in inland mountain ranges, martens select riparian areas and avoid landscapes with large patches of young forest; in contrast, in xeric east-side forests in inland mountain ranges, martens select sites with higher tree canopy cover and avoid fragmented forests (Shirk et al. 2014).

Pacific martens have recently been located in forests of the central and south coast regions of coastal Oregon (Moriarty et al. 2016a). In coastal California and Oregon, they occur in three environmental conditions: productive soil types where Douglas-fir, hemlock, and redwood forest associations occur; and two restricted environments on low-productivity soils, including near-coast serpentine forest environments restricted to south coast Oregon and north coast California, and shore pine (*Pinus contorta* var. *contorta*) forest associations found in stabilized dunes in central coastal Oregon (Slauson et al. 2007, in press). Although the tree composition and vegetation structure vary among these three environmental conditions, they have one feature in common: a dense (more than 60 percent cover) shrub understory dominated by ericaceous species such as salal (*Gaultheria shallon*) or evergreen huckleberry (*Vaccinium ovatum*) (Slauson et al. 2007). In coastal forests at the home-range scale, martens select for large patches (more than 250 ac or 100 ha) of old-growth and late-mature forest or serpentine areas (Slauson et al. 2007).

Complex vertical and horizontal forest structure is an important characteristic of marten habitat, providing features used to meet daily resting and seasonal denning needs as well as providing food and cover for prey species. Just outside the NWFP area in north-central California, Moriarty et al. (2016b) found that martens strongly selected complex-structured forest stands over simple stands and openings. Stand structure influenced martens' movement patterns, being quicker and more erratic and linear in simple stands and openings, likely related to predator behavior; and more sinuous and slower in complex stands, likely related to foraging under cover from predators.

In coastal California, the summer and fall diet of martens is dominated by sciurid and cricetid rodents (chipmunks [*Tamias* sp.], Douglas squirrels [*Tamaisciurus douglasii*], and red-backed voles [*Myodes californicus*]) and medium-sized birds. During winter, martens shift to larger bodied prey species such as medium-sized birds and northern flying squirrels (*Glaucomys sabrinus*) (Slauson et al., in press). Many of these key prey species of the Humboldt marten reach their highest densities in forest stands with mature and late-successional structural features (e.g., Carey 1991, Carey and Johnson 1995, Hayes and Cross 1987, Rosenberg et al. 1994, Waters and Zabel 1995) where their key food resources—conifer seed crops, truffles (the fruiting bodies of ectomycorrhizal fungi), and late-successional and old-growth-associated pendant lichens—typically reach their greatest abundances (Luoma et al. 2003, Smith et al. 2002). Dense ericaceous shrub layers are also positively associated with key prey species including chipmunks (Hayes et al. 1995), northern flying squirrels (Carey 1995), and small mammals in general (Carey and Johnson 1995). Coarse woody debris also has been shown to positively influence foraging efficiency for martens by making prey more vulnerable to capture; where it has been reduced from timber harvest, marten foraging efficiency declines (Andruskiw et al. 2008).

Like fishers, martens also depend on large decadent woody structures including live and dead trees and downed logs with features such as cavities, platforms, and chambers for daily use as resting sites and as seasonal den sites for females raising young (Raphael and Jones 1997, Slauson and Zielinski 2009, Thompson et al. 2012). In summer and fall, martens in coastal forests rest predominantly in large-diameter snags, downed logs, and live trees, and less commonly in slash piles, boulder piles, and shrub clumps (Slauson and Zielinski 2009). Use of the sizes of trees, snags, and downed logs is similar between coastal and inland mesic forests, but tree diameters are larger than reported for inland xeric forest (Raphael and Jones 1997). Female martens are also obligate cavity users for reproduction, with cavities in trees and snags providing secure environments for kits by regulating temperature extremes and providing protection from predators (Raphael and Jones 1997, Slauson and Zielinski 2009, Thompson et al. 2012). Reproductive den structure types are more restricted than are rest sites (Ruggiero et al. 1998), and all natal dens have occurred in aerial cavities in large-diameter hardwoods, and maternal dens have included cavities and platforms in both large-diameter hardwoods and broken top live and dead conifers (Slauson and Zielinski 2009; Slauson, in press). In contrast to inland forest types, the dens found to date in coastal forests have been predominantly in hardwoods.

Lynx—In northern Washington, Koehler et al. (2008) found that lynx exhibited the strongest selection for Engelmann spruce (*Picea engelmannii*) and subalpine fir (*Abies lasiocarpa*) forest on moderate slopes, and to a lesser degree, forests with moderate canopy cover, at elevations of 5,000 to 6,000 ft (1525 to 1829 m), and either avoided, or used proportional to their availability, the following cover types:

lower elevation Douglas-fir, ponderosa pine forests, forest openings, and recently burned areas with sparse canopy and understory cover. Lynx in Washington spend more time hunting in patches of Engelmann spruce and subalpine fir where their predominant prey, snowshoe hare (*Lepus americanus*), occur at their highest densities (Maletzke et al. 2008). Lynx den sites typically occur in sheltered spaces most often created by downed logs, root wads of fallen trees and less often by boulder fields, slash piles, and live trees (Moen et al. 2008, Squires et al. 2008). In Montana, lynx den sites typically occurred in mature and mid-seral conifer stands with abundant woody debris, located in drainages or drainage-like basins (Squires et al. 2008).

***Wolverine*—**The combination of historical records (Aubry et al. 2007) and contemporary studies of wolverines (Copeland et al. 2007, 2010) reveals that, during the denning and nondenning periods of the year, the species is closely associated with persistent spring (April to May) snow cover, and that this represents an obligate association with this bioclimatic feature. Wolverine habitat selection is strongly influenced by the distribution of seasonal prey resources, which tend to be dominated by large ungulates (e.g., moose, deer, elk, and caribou) (Copeland 1996, Lofroth et al. 2007) and large rodents (e.g., beaver, hoary marmots, and porcupine) (Lofroth et al. 2007).

Female wolverines have been shown to be more sensitive than males to predation risk and human disturbance. Females make greater use of rugged terrain to avoid predators and avoid areas with winter ski recreation and higher road densities within their home ranges (Krebs et al. 2007). Wolverine den sites tend to occur in areas with persistent spring snow cover and with large boulders and downed trees where females find secure sites for giving birth to and raising kits (Dawson et al. 2010, Magoun and Copeland 1998).

***Landscape habitat suitability and connectivity for carnivores*—**Studies and habitat modeling of carnivores conducted at the landscape scale primarily address issues related to population processes, such as dispersal, gene flow, habitat connectivity, and population persistence. Regional landscape habitat models have been developed for the fisher in various portions of the NWFP area (Carroll et al. 1999, Davis et al. 2007, Halsey et al. 2015, Lewis 2013, Zielinski et al. 2010). Collectively, the most important predictors of suitable fisher habitat measured at various scales include areas with higher annual precipitation, moderate to dense tree canopy cover, higher basal area and diameter classes of hardwoods and conifers, and topographically complex areas at mid to low elevations.

The southern portion of the NWFP area supports the largest extant fisher population in the Pacific Northwest. Modeled suitable habitat for fishers in the southern portion is projected to be well distributed and well connected, and the habitat models provide managers with a tool for planning activities and for monitoring habitat changes.

In the coastal forests of California and Oregon, Slauson et al. (n.d.)[29] modeled landscape habitat suitability for the Humboldt marten where the most important predictors, at various scales, were old-growth forest structure, amount of serpentine environment, and areas of higher precipitation, all positively correlated with marten detection locations versus nondetection locations. This model predicted that less than 15 percent of the historical habitat of this subspecies in the NWFP area is currently suitable. In the southeastern portion of the NWFP area, Kirk and Zielinski (2009) modeled landscape habitat suitability for martens in inland forest areas and found that marten occurrence was best predicted by a large amount of the biggest size classes of mesic forest types, by landscape configuration (more habitat patches), and by more habitat in public land management.

Moriarty et al. (2015) used new methods for evaluating functional connectivity of Pacific martens across fragmented forest landscapes: use of food incentive experiments, and nonincentive locations of martens collected with global positioning system telemetry. They found that Pacific martens selected complex stands and crossed open areas in the winter when enticed by bait, and without bait they avoided openings and simple-structured forest

[29] Slauson, K.M.; Zielinski, W.J.; Laplante, D.; Kirk, T. [N.d.]. Landscape habitat suitability model for the Humboldt marten (*Martes caurina humboldtensis*). Manuscript in preparation. On file with: Keith Slauson, U.S. Forest Service, Pacific Southwest Research Station, 1700 Bayview Drive, Arcata, CA 95521. kslauson@fs.fed.us.

stands. Pacific martens also selected complex stands during summer and winter alike. Baiting can induce movements and introduce bias in habitat selection analysis intended to represent bait-free conditions. Dispersal and connectivity of Pacific martens thus seemed to be afforded partially by snow cover across open areas, and more fully by networks of complexly structured forest stands during summer.

Koehler et al. (2008) developed a model of lynx habitat suitability from snow tracking and estimated that 1,467 mi^2 (3800 km^2) of habitat suitable for lynx is present in the NWFP region, far less than prior estimates based on the correlation of historical records to vegetation types (McKelvey et al. 2000). Most lynx habitat identified by Koehler et al. (2008) occurs in western Okanogan County, Washington, and it was estimated that this amount of habitat could support a population of 87 lynx. Murray et al. (2008) assessed the state of lynx research and conservation in the species' southern range, including the NWFP area, and concluded based on their review of the existing science that successful lynx conservation will require protection and management of large tracts of lynx and snowshoe hare habitat, and ensuring connectivity between lynx populations at the core and periphery of the species' range.

The global distribution of wolverines is closely associated with areas of persistent spring snow cover (Copeland et al. 2010). In the NWFP area, the most significant areas with persistent spring snow cover are high elevations of the north-central Washington Cascades, where most records of wolverines occur. Although smaller areas with these conditions occur in the Oregon and California Cascades, limited connectivity to areas known to be occupied by wolverines may impede the ability for wolverines to reoccupy these areas. Other than models of areas with persistent spring snow cover across the NWFP area (Copeland et al. 2010), no additional habitat suitability models have been developed for wolverine. Similar to the lynx, ensuring connectivity between the largest areas of suitable habitat for the wolverine in the NWFP area with areas occupied by wolverines in British Columbia will be one of the most critical requirements to help ensure wolverine conservation in the NWFP area.

Landscape-scale habitat connectivity is a significant requirement for the carnivores considered here. For lynx and wolverine, connectivity to population source areas outside the NWFP is essential for population persistence (Aubry et al. 2007, Murray et al. 2008, Schwartz et al. 2009). For the fisher in the Washington Cascades (Lewis and Hayes 2004), and for lynx (Koehler et al. 2008), a current lack of connectivity likely is a main factor inhibiting the recolonization of suitable habitat in the northern portions of the NWPF area. For species that occupy high-elevation areas, particularly in the Cascades (i.e., lynx, wolverine, marten), climate change will likely reduce the amount and connectivity of habitat (Lawler and Safford 2012, McKelvey et al. 2011). In the coastal forests of northwestern California, the lack of connectivity of habitat for the Pacific marten is likely inhibiting the recolonization of suitable habitat patches within their dispersal capability (Slauson, in press).

In general, current and future problems of habitat connectivity for carnivores include wide distances among patches of suitable habitat, low quality of intervening habitat, and adverse effects of natural and anthropogenic movement filters and barriers such as rivers and roads on dispersal behavior and survival (Crooks et al. 2011). Owing to the high cost of creating habitat corridors and enhancing connectivity such as by constructing wildlife crossings on highways, or the long time necessary for regenerating late-successional forests where they have been lost, connectivity enhancements that can benefit multiple species (e.g., Singleton et al. 2002) could be most efficient at achieving objectives for carnivore conservation.

***New threats to carnivores: predation*—**Studies in the NWFP area, as well as studies from adjacent areas in the Pacific States (Bull et al. 2001), have identified the bobcat (*Lynx rufus*) as a major predator of fishers (Wengert et al. 2014) and martens (Slauson et al., in press). Whereas predation is a natural mortality factor, its magnitude may have increased from historical times owing to the increase in early- to mid-seral forests in the NWFP area, which provide conditions suitable for predators such as bobcats and coyotes (*Canis latrans*) that have general habitat needs. In the NFPW area, studies of bobcat habitat use and diet in coastal forests indicate that they select for young regenerating stands (Slausonet al., in press; Wengert 2013) where they find key prey species such as lagomorphs, woodrats, and mountain beavers (*Aplodontia rufa*) (Knick et al. 1984; Slauson et al., in press). Halsey et al. (2015) modeled bobcat habitat to account for predation risk in evaluating reintroduction locations for fishers in the southern Washington Cascades, and found that reintroduction sites at middle to higher elevations rather than at lower elevation would likely reduce risk of predation by bobcats.

***New threats to carnivores: rodenticides*—**The widespread use of rodenticides and other toxic chemicals found at illegal marijuana-growing operations has emerged as a major new threat. In California, most [dead] fishers tested for toxicant exposure tested positive for one or more anticoagulant rodenticides, which were ingested when the fishers consumed contaminated rodent prey. Several fishers recently have been confirmed to have died from acute rodenticide poisoning on the Hoopa Valley Indian Reservation located in the southern portion of the NWFP area (Gabriel et al. 2012, 2015). Exposure rates of Humboldt martens to rodenticides is less well understood, but in California, rodenticide exposure has been confirmed in one of six (17 percent) [dead] martens tested (Slauson et al., in press). This threat may be limited to the southern portion of the NWFP area where cultivation of marijuana is illegal on federal land but common, and where most suspected use of rodenticides is at such illegal growing sites on public, tribal, and private land alike.

***New threats to carnivores: wildfire and fuels treatments*—**In 2007, the Fisher Conservation Strategy Biology Team held an expert panel workshop to conduct a threats analysis on fishers within the West Coast Distinct Population Segment and British Columbia portions of the species' range in the Pacific Northwest. The team evaluated 20 types of potential threats and ranked four equally as the greatest threats: uncharacteristically severe wildfire, forest canopy and overstory reduction, reduction of forest structural elements, and forest habitat fragmentation (Marcot et al. 2012a).

Wildfire is a natural process that can alter environments in various ways, including leaving behind remnants of older forest stands and large wood components that can serve as legacy structures as the vegetation subsequently regenerates. However, major loss or degradation of forests, especially from extensive high-intensity wildfire, can significantly affect habitat for the carnivores considered here. Effects of wildfire will be the most severe where fire suppression has resulted in the unnatural buildup of fuels. Severe wildfire occurrence is primarily in the southern portion of the NWFP region and at lowest elevations, putting the fisher at greater risk to loss of habitat than the other carnivores considered here. These are also the areas where fuels treatments such as mechanical thinning, which can potentially negatively affect fisher habitat, are most commonly prescribed. High-severity wildfire can have positive effects on some prey species, particularly herbivorous species such as Leporids (*Lepus americanus* and *Sylvilagus bachmani*) and dusky-footed woodrats (*Neotoma fuscipes*), by creating early-seral conditions that favor these populations. The degree to which high-severity wildfire can benefit the carnivores considered here depends on their reliance on these species as important components of their diets and the degree to which fires result in landscape patterns (e.g., mixed severity in the range of the fisher) compatible with supporting the rest of their life history needs.

The effects of fire suppression on carnivore forest habitat in the Pacific Northwest have received little study. Fire suppression activities often result in denser forests with more canopy layers and more dead wood from shade-tolerant trees. Whether these outcomes increase habitat quality or amount for forest carnivores is unstudied.

Initial research on the effects of fuels treatments on fishers outside the NWFP area suggests that treatments have significant indirect benefits by protecting fisher habitat from wildfire (Scheller et al. 2011) and that some types of fuels treatments may be compatible with fisher occupancy as long as they do not exceed a particular rate and extent (Sweitzer et al. 2016a, Zielinski et al. 2013) and they do not affect the density of large structures (logs, snags, live trees) used by fishers for foraging, resting, and denning.

Loss of habitat from wildfire was ranked as one of the top threats to the Humboldt marten in coastal California (Slauson et al., in press). In the last two decades, several fires have burned in marten-occupied areas with unknown effects on the population. Fuels treatments are typically used to minimize the severity of wildfire in these situations, but to avoid reducing marten habitat, they could be carefully planned and monitored. Fuels treatments that reduce surface and ladder fuels produce stand structures that martens either avoid or travel through quickly (Moriarty et al. 2016), similar to how marten respond to open linear features such as ski runs (Slauson et al 2017) and seismic lines (Tigner et al. 2015). Therefore to minimize impacts to martens from fuels treatments, they could be carefully designed to minimize their overlap with habitats supporting important aspects of marten life history such as denning and have a spatial arrangement that most likely represents a tradeoff between affecting fire behavior and minimizing negative effects on connectivity and foraging for martens (Moriarty et al. 2015, 2016b). Since 1985, fires have burned more than 386 mi^2 (1000 km^2) of forest habitat in Okanogan County, the only region in Washington where lynx has been verified recently; this loss of forest habitat from fire, coupled with the naturally fragmented distribution of suitable habitat, represent a present, significant threat for lynx conservation where they persist in the NWFP area (Koehler et al. 2008).

The effect of wildfire and fuels treatments on wolverines has not been directly studied.

***New monitoring methods for carnivores*—**New developments in methods for monitoring carnivores include methods of detection, improvement of statistical survey designs, and advances in statistical analyses of carnivore survey data. Once the objectives of monitoring are identified, a suite of survey designs can be considered (Long and Zielinski 2008, O'Connell et al. 2011). Statistical advances in occupancy modeling (MacKenzie et al. 2006, Royle et al. 2008, Slauson et al. 2012) and spatially explicit capture-recapture analysis (Royle et al. 2014) provide two new statistical frameworks to conduct carnivore monitoring. Because of the importance of designing surveys capable of reliably detecting a species, detecting magnitudes of change as specified in the monitoring objective, and optimizing the costs of such designs, Ellis et al. (2014) developed a power-analysis tool to evaluate the effects of alternative design elements for monitoring wolverines that has also been evaluated for fishers (Tucker 2013). Monitoring population status via sampling DNA from noninvasively collected hair samples is an increasingly realistic goal for carnivores (Kendall and McKelvey 2008, Pauli et al. 2011, Schwartz and Monfort 2008). Recent advances also have been made in the use of camera-trap sampling of carnivores (Mcfadden-Hiller and Hiller 2015), including fishers (Sweitzer et al. 2016b), and mountain lions (*Puma concolor*) (Caruso et al. 2015), and carnivores elsewhere (e.g., Herrera et al. 2016, LaPoint et al. 2013) including arboreal situations (Cotsell and Vernes 2016).

Red tree vole and *Arborimus* allies—

Since the 2006 synthesis (Haynes et al. 2006), much work has taken place on the morphology, genetics, and taxonomy of red tree voles and related species (Bellinger et al. 2005; Miller et al. 2006, 2010). Three species of *Arborimus* forest voles are now recognized to occur within the NWFP area. The white-footed vole (*A. albipes*) occurs in the Coast Range, Willamette Valley, and a portion of the western Cascades, from northwestern California north through Oregon to the Columbia River border with Washington. The red tree vole (*A. longicaudus*) occupies a subset of the range of the white-footed vole but does not extend as far into California, is absent in extreme northwest Oregon, and extends into the Cascades east of Portland, still constrained by the Columbia River to the north. The Sonoma tree vole (or California red tree mouse; *A. pomo*) is found only in coastal northern California, extending northward to where the red tree vole's range ends. Red tree voles and Sonoma tree voles are arboreal, making nests and using movement runways in canopies of late-successional and old-growth forests; white-footed voles, however, are only partially arboreal.

Red tree voles and Sonoma tree voles are particularly ecologically interesting for their arboreal life history and their reliance on needles and twigs of conifers as food sources. Chinnici et al. (2012) documented association of Sonoma tree voles with unharvested and partially harvested old-growth Douglas-fir forest in California. Genetics analyses by Blois and Arbogast (2006) revealed that, although Sonoma tree voles consist of one panmictic (interbreeding) population, some genetic evidence suggests that southern Sonoma tree voles may occur as a distinct management unit within the species.

Swingle and Forsman (2009) found that red tree voles occasionally moved short distances across small forest openings or across small logging roads less than 82 ft (25 m) wide. However, they saw no evidence of tree voles moving across large nonforest areas and thereby speculated that such areas are barriers to dispersal, and that long-term persistence of red tree voles depends on size and connectivity of forest cover and structure of the forest canopy.

Dunk and Hawley (2009) recognized threats to red tree vole populations, including fire and recent historical conversion of forests to nonforest uses. They developed habitat association models, and their most statistically significant model identified the best habitat correlates as site percentage of slope, basal area of trees 18 to 35 inches (45 to 90 cm) d.b.h., maximum tree height, and variation in diameters of conifers. They applied the model to the distribution of the species and determined that reserves have significantly higher quality habitat than do nonreserved lands.

Studies by Wilson and Forsman (2013) noted that silvicultural thinning of young forest stands to promote late-seral forest conditions can provide habitat for a number of vertebrate species, but thinning can delay the development of a mid-story layer of trees, which is an important habitat component for red tree voles (and also of northern flying squirrels, *Glaucomys sabrinus*). The authors suggested that long-term forest and habitat management goals should include developing more structurally and biologically complex forests across the landscape at scales appropriate to the species' ecologies.

Other recent research on red tree voles includes determining their distribution and abundance based on occurrence in pellets regurgitated by a principal predator, northern spotted owls (Forsman et al. 2004). Working in Oregon, the authors found that the incidence of red tree voles in the owl pellets occurred most commonly in the central and south coastal regions, were relatively sparse in the northern Coast Range and Cascades areas of the state, and were absent from pellets on the east slope of the Cascade Mountains and most of the dry forest area in south central Oregon. Incidence also declined by elevation in the Cascades, being most common in owl pellets below 3,200 ft (975 m) and rare above 4,000 ft (1220 m) elevation. The authors also speculated that red tree vole populations have declined in landscapes dominated by young forests and with increased occurrence of logging, fire, and human development. Their finding of low incidence of red tree voles in spotted owl pellets along the north coast area of Oregon was also supported by direct surveys there in Tillamook and Clatsop State Forests by Price et al. (2015) who found 33 tree vole nests at only 4 out of 86 randomly selected sites. They speculated that the dearth of nests there is because of logging or burning in the early 1900s and subsequent intensive forest management, and suggested that remnant old-forest stands on BLM and state lands serve as source populations.

Rosenberg et al. (2016) presented models of red tree vole habitat, comparing and combining their best performing model with previously published models. They concluded that the ensemble combination of models provided the most accurate predictions of red tree vole occurrence and habitat suitability, and that it could be used to reduce survey costs and to guide management decisions and habitat conservation strategies for the species.

Swingle et al. (2010) discovered that a key predator of red tree voles in western Oregon is weasels (*Mustela* spp.). Activity behavior and dispersal timing of red tree voles were studied by Forsman et al. (2009) using video cameras at nest sites, and additional studies of red tree vole home ranges and activity patterns were conducted by Swingle and Forsman (2009). At present, a major review has been completed on the distribution, habitat, and diet of red tree voles and Sonoma tree voles (Forsman et al. 2016), and an annotated bibliography has been compiled on all three *Arborimus* species discussed here (Swingle and Forsman 2016).

The ISSSSP's 2016 Red Tree Vole High Priority Site Management Recommendations[30] provides guidance to field personnel for identifying land use allocations consistent with red tree vole conservation, high-priority sites outside of those allocations, and connectivity areas for linking sites, as well as nonpriority sites and key information gaps.

On the International Union for Conservation of Nature and Natural Resources Red List, although relatively rare, white-footed voles are listed as of "least concern," whereas red tree voles and Sonoma tree voles are both listed as "near

[30] http://www.blm.gov/or/plans/surveyandmanage/files/RTV-HPS-MR-201604-final.pdf.

threatened" because of their restriction to mature forests—old-growth forests for the red tree vole—and from threats of logging and fragmentation of their forest habitats. In 2009, the U.S. Fish and Wildlife Service evaluated whether red tree voles in the North Coast Range of Oregon—sometimes referred to as "dusky tree voles" that seem to differ in diet and other aspects from other red tree voles—deserve designation as a distinct population segment under the Endangered Species Act. The Fish and Wildlife Service convened a panel of species experts and managers to evaluate the evidence (Marcot and Livingston 2009), and ultimately announced that the north Oregon Coast Range population warrants listing as a distinct population segment but is precluded from development of a proposed listing rule because of higher priority actions (USFWS 2011). Morphological study of the dusky tree vole (putatively *A. longicaudus silvicola*) did not find evidence to support the proposed, separate subspecies designation (Miller et al. 2010). The current listing status of red tree vole is Continuing Candidate; the Fish and Wildlife Service's species assessment[31] also provides a summary of recent biological and ecological information on the species.

Bats—

At least 14 species of bats are found within the NWFP area. Some of these species are late-successional and old-growth associated, and their forest habitats are likely provided by NWFP reserve guidelines, although studies are incomplete.

Kroll et al. (2012) reviewed the state of knowledge on birds and bats dependent on cavities and snags (dead standing trees) in the Pacific Northwest. They concluded that relatively little is known about how the viability of populations of these species is affected by distributions of snags and by the proximity and amount of mature and late-successional forests across the landscape. Rodhouse et al. (2012) provided models of the distributions of little brown bats (*Myotis lucifugus*) in the Northwestern United States, finding a link of the species with forest cover and productivity.

Lacki et al. (2012) studied use of snags for roosting by long-legged myotis (*M. volans*) in the Pacific Northwest, and found that fir snags (*Abies* spp.) had the lowest persistence rates (highest falling rates) over the course of their 5-year study (2001–2006), and that roost snags that persist standing the longest are larger in diameter, shorter in height, and with fewer branches. The authors recommend reserving sufficient, large-diameter leave-trees to provide for future use by roosting bats, given the persistence and fall rates of the snags they observed. Luszcz et al. (2016) studied three bat foraging guilds in southwest British Columbia, Canada, and reported that foraging activity of long-eared myotis increased with increasing forest age, and in general, *Myotis* spp. feeding along edges and in forest gaps was greater in black cottonwood stands and in the interior of old Douglas-fir stands than in other types of forests.

On a far broader scale, Duchamp et al. (2007) suggested managing for bats at a landscape scale to accommodate their nocturnal foraging movements, alternative forest roost sites for some species, and reduce adverse effects of forest habitat fragmentation. The authors cite evidence that some bat species seem sensitive to local habitat fragmentation but also require mixes of conditions across broad geographic areas to meet all resource needs.

Few studies are available on response of bats to fire. Buchalski et al. (2013) studied the response of bats to wildfire severity in a mixed-conifer forest in the Sierra Nevada of California and reported use in unburned, moderate-severity burn, and high-severity burn locations, with large-bodied bats showing no preference. They concluded there was no evidence that burn severity affected site selection and that bats in their region are resilient to landscape-scale fire that served to open up the stands and increase the availability of prey and roost structures.

One issue of growing concern is the recent discovery (March 2016) of white-nose syndrome in a little brown bat (Lorch et al. 2016) 30 mi (48 km) east of Seattle, Washington.[32] White-nose syndrome is caused by the fungus *Pseudogymnoascus* (prev. *Geomyces*) *destructans* (Verant et al. 2014) and is a major cause of mortality in a variety of bat species in the Eastern United States, having killed over 6 million bats since 2006. So far, white-nose syndrome

[31] http://ecos.fws.gov/docs/candidate/assessments/2015/r1/A0J3_V02.pdf.

[32] http://wdfw.wa.gov/conservation/health/wns/.

seems to exist mainly in species of bats that roost or hibernate in cave or cave-like environments; at least four species of bats in the Pacific Northwest are known to use such locations: little brown bat, big brown bat (*Eptesicus fuscus*), Townsend's big-eared bat (*Corynorhinus* [*Plecotus*] *townsendii*), and the poorly known long-eared myotis (*M. evotis*). Whether white-nose syndrome will spread geographically or among species in the Pacific Northwest, or to bat species using other roost situations such as large trees with sloughing bark and dense canopy foliage, is unknown.

The 2001 amended standards and guidelines for NWFP implementation provided specific direction on bat conservation and management, including safety considerations for conducting bat surveys, survey protocols for bat surveys in buildings, environmental education on bat ecology and presence, and plans for building bat boxes. Other issues regarding bat conservation in the NWFP area include gating of caves and mines to prevent disturbance of bat colonies, bat mortality from wind energy installations (Rodhouse et al. 2015, Weller and Baldwin 2012), and concerns for human health and safety from bat colonies occupying buildings.

The ISSSSP has developed a set of management and safety guidelines, and survey and monitoring protocols, for bat conservation in the Pacific Northwest.[33] Forest Service Region 6 has also conducted bat surveys using a Bat Grid Monitoring design and protocol.[34] The Bat Grid Monitoring study was conducted from 2003 to 2010 to provide a baseline of knowledge on bat distribution for future change detection, and produced predictive distribution maps of 11 bat species in the Pacific Northwest.

Recent Research and Development on New Tools and Data

Along with scientific studies specifically conducted on species, their habitats, and biodiversity of the NWFP area since the 2006 synthesis (Haynes et al. 2006), several advancements have been made in the development of new scientific tools, datasets, study methods, and assessments.

Much work has focused on developing geographic models and map projections, particularly related to projected effects of climate change. The Nature Conservancy has assessed and mapped the degree of resilience of ecosystems through a broad area of western North America including portions of the Pacific Northwest and NWFP area (east Cascades, west Cascades, north Cascades, Klamath Mountains, California north coast, Pacific Northwest coast, Willamette Valley, and Puget Trough).[35] They have also developed a mapping tool for assessing connectivity of "resilient terrestrial landscapes" in the Pacific Northwest (McRae et al. 2016). The U.S. Geological Survey has evaluated vulnerability of species and ecosystems to future climate in the Pacific Northwest.[36] A number of other institutions, academic studies, and agencies also have addressed climate change impacts in the NWFP area. Other sources of information on biodiversity and species in the NWFP area include the Washington Biodiversity Council[37] and the Oregon Biodiversity Information Center.[38]

Species and natural-area mapping studies and evaluations, specifically in the Klamath province, have been provided also by Sarr et al. (2015) and Olson et al. (2012). Sarr et al. compared three ecoregional classification systems and found that the top one (Omernik) served to map the distribution of tree species best, but the distribution of mammal species poorest. Olson et al. (2012) projected effects of climate change and determined that some old-growth forest conditions ("microrefugia") were not captured in the existing reserve designs. These microrefugia conditions would lend to the reserve system being more resilient under climate change; they include old-growth and intact forests on north-facing slopes and in canyon bottoms, lower and middle elevations, wetter coastal mountains, and along elevational gradients. However, the degree to which these findings for the Klamath province can be applied beyond is unevaluated, and we anticipate that other patterns might

[33] http://www.fs.fed.us/r6/sfpnw/issssp/species-index/fauna-mammals-bats.shtml.

[34] http://www.fs.fed.us/r6/sfpnw/issssp/species-index/fauna-mammals-bats-grid-monitoring.shtml.

[35] https://www.conservationgateway.org/ConservationByGeography/NorthAmerica/UnitedStates/oregon/science/Pages/Resilient-Landscapes.aspx.

[36] http://gec.cr.usgs.gov/projects/effects/vulnerability/index.html.

[37] http://www.rco.wa.gov/biodiversity/about_the_council.shtml.

[38] http://inr.oregonstate.edu/book/export/html/549.

occur in more moist forest environments farther north in the NWFP area (see chapter 2)

A number of tools for analyzing species' habitat connectivity are available (e.g., Brost and Beier 2012). Zielinski et al. (2006) used MARXAN to analyze habitat of spotted owls and fisher in northern California under the NWFP, and concluded that spotted owls require a greater percentage of planning units with habitat than do fisher, and that the current location of the late-successional reserves may not be best to maintain habitat connections for the two species. The individual movement modeling shell HexSim (Schumaker 2013) has been used in the NWFP area to assess habitat patch size and dispersion patterns for northern spotted owls (Marcot et al. 2013, Schumaker et al. 2014). Trumbo et al. (2013) used the modeling tool Circuitscape (McRae and Shah 2009) to evaluate landscape genetics relationships of Cope's giant salamander (*Dicamptodon copei*). Using an example of American pika (*Ochotona princeps*), Schwalm et al. (2015) championed the use of species distribution models for evaluating effects of climate change and habitat conditions on landscape genetics. They found that wide local variation in pika response, ranging from complete extirpation in some areas to stable occupancy patterns in others, as well as habitat composition and connectivity, were important to include in the pika model. Many other simulation and modeling tools are available.

Another recent advancement in mapping species distributions is the development of maps of probability of suitable habitat coupled with maps of uncertainty, such as the one produced by Rodhouse et al. (2012, 2015) and hosted by the ISSSSP, for bat species of the Pacific Northwest (see footnote 34), thereby providing managers with information on locations of habitat as well as confidence in the projections shown spatially. Other approaches to evaluating the effects of forest management on rare species can address the need for balanced sampling designs, such as using the application EstimateS (Colwell 2013) to statistically compare samples among various land condition categories (treatments).[39]

The modeling framework of MaxEnt (maximum entropy) has recently become popular, as used to map potential habitat in the Pacific Northwest for northern spotted owls (e.g., Carroll et al. 2010, Loehle et al.; also see chapter 4), white-headed woodpeckers (Latif et al. 2015), historical range of California condors (*Gymnogyps californianus*, D'Elia et al. 2015), fisher (Zielinski et al. 2012), and other species. Cautions in the use of the MaxEnt approach, however, have been offered by a number of researchers, such as Yackulic et al. (2013).

Olson et al. (2007b) summarized the strategic surveys conducted under the NWFP guidelines. Strategic surveys (Molina et al. 2003) are designed to fill key information gaps on species distributions and ecologies, the basic criteria for Survey and Manage, to help answer three questions: (1) Does the species occur in or occur close to the NWFP area? (2) Is the species associated with late-successional and old-growth forest? (3) Does the reserve system and other standards and guidelines of the NWFP provide for a reasonable assurance of species persistence? Olson et al. (2007b) reported that all strategic surveys were conducted under the Survey and Manage program from 1994 to 2004 and from 2006 to 2007 (other types of surveys are ongoing on a district-by-district basis). Survey results in the form of projects, publications, and reports on 10 taxa (fungi, lichens, bryophytes, vascular plants, arthropods, mollusks, amphibians, red tree voles, great gray owl, and bats) are now available in a regional repository of the ISSSSP[40] that is currently managed for a list of sensitive species, some of which are Survey and Manage species.

Recent and Ongoing Issues Adding Challenges and Opportunities

We have raised a number of issues related to other species under the NWFP, in the above text. Here we summarize these and additional challenges and opportunities, regarding conservation of other species, that have emerged since the 2006 synthesis (Haynes et al. 2006).

[39] As suggested by D. Luoma. Personal communication. Assistant professor, Department of Forest Ecosystems & Society, Oregon State University, 239A Richardson Hall, Corvallis, OR 97331.

[40] http://www.fs.fed.us/r6/sfpnw/issssp/.

Disturbance ecology and system dynamics—

***Wildfire and fire suppression*—**In this chapter, we have reviewed the mostly adverse direct effects of severe wildfire on fungi, soil arthropods, forest salamanders, forest carnivores, and tree voles. Chapter 3 reviews historical fire regimes of moist and dry forests, effects of recent historical fire suppression on increasing fuel loads and current fire risk, fire suppression effects on reducing early-successional environments, use of fire for restoration, and how the role of fire in late-successional reserves varies across the NWFP area.

Healey et al. (2008) reported that, since the initiation of the NWFP in 1994, loss of large-diameter forests to fire has exceeded loss from harvest across large areas of the region including federal and nonfederal lands (also see Davis et al. 2015). They also found that increased fire incidence is correlated with climate change and that retention of dry-site forests will hinge on coordinated, landscape-scale fire management (also see Halofsky et al. 2014, Podur and Wotton 2010, Sheehan et al. 2015). Healey et al. (2008: 1117) cited several sources for types of strategic fuel management approaches that "may have the potential to reduce the [future] impact of fire on the landscape…"

Although severe wildfire in the NWFP area is an increasing threat to conservation of existing older forest species and ecosystems, it is likely that some dense late-successional and old-growth forests of the Western United States resulted or persisted from decades of fire suppression (Sollman et al. 2016), and others likely were maintained because of topographic positions protected from fire. Fire suppression may have contributed to the development and continuation of some old-forest conditions on the east side of the Cascade Range, such as on the Deschutes National Forest (also see chapter 3). Fuels reduction activities also have changed habitat for many NWFP species in the managed forest matrix, particularly in the dry forest types in the southern part of the NWFP region, by removing fuel loads that in turn reduces presence and amounts of fine litter fuels, and in some cases, large down wood used as den sites by fishers (Sweitzer et al. 2016a).

Historically, frequent fires in dry forest types of the Pacific Northwest resulted in a diversity of vegetation ages and structures among multiple seral stages, and natural legacies of old-forest elements such as large trees and large down wood often remained postfire. But, acting as a double-edged sword, fires of various intensities and frequencies can either create standing and dead wood structures useful to wildlife, or burn them up, unless they are protected or unless fires are of generally low intensity.

As fire regimes in the region shift location and increase in intensity (see chapter 3), there may arise difficult management and social choices between fire suppression and natural fire restoration. Wildfire in the Pacific Northwest is inevitable, and massive "megafires" have the potential of posing threats to old-forest species, such as reported on spotted owls in California by Jones et al. (2016).

Prescribed fire, on the other hand, can be a valuable management tool for maintaining open old-growth forests, forest openings, and early-successional vegetation. Fuels management can help reduce risk of spread of large, high-severity fire. Management options include the use of fuelbreaks, managing for low fuel loading on the forest floor, thinning heavily stocked "doghair" and stagnant stands, pruning to eliminate fire ladders, and other methods. However, there is disagreement over the need to use such treatments, even in fire-prone systems (see chapter 3). Whereas fuel treatments and restoration management can affect fire behavior, these actions expose bare soil that, in turn, could promote invasion of exotic plant species that could spread into adjacent lands, and where nonnative plant cover is twice as high on fuel breaks as in adjacent wildlands (Merriam et al. 2006; also see chapter 3).

Modeling by Ager et al. (2007) suggested that treating 20 percent of the landscape as fuel breaks and for fuel reduction can result in a 44 percent increase in the probability of conserving late-successional and old-growth spotted owl habitat (see chapter 4). Moriarty et al. (2016) suggested that fuel treatments that simplify stand structure can adversely affect movement and habitat connectivity for martens. The authors noted that martens avoided openings at landscape- and home-range scales, but thinned stands were avoided mostly at only the home range scale because such stands were uncommon in their study area of Lassen National Forest, in northeastern California, just outside the NWFP area. The authors also suggested that fuel treatments that would create landscape- or home range-scale openings be done at elevations lower than where martens typically occur. However, the effectiveness of such fuel-treatment approaches in providing for marten habitat likely varies with other considerations, in particular, habitat considerations for other species. Such treatments also may be more effective on south slopes and ridgetops, and less effective on north aspects and in drainage bottoms.

Further, fuels treatments as mentioned above could have adverse effects on the many species, within late-successional and old-growth forest stands and other forest age and structure classes, that occupy litter and duff (the organic matter soil horizon) and down wood of various decay, size, and cover classes and tree species, as well as standing partially dead trees and snags. The flip side of this, however, is that species associated with open stands of late-successional and old-growth forest or from burn conditions might benefit from lower severity wildfire. As mentioned below, balancing reduction of fire risk with providing such microhabitat elements for species and biodiversity conservation can be the subject of a decision analysis and decision science process (see chapter 12).

***Early-seral vegetation environments*—**Early-successional vegetation conditions provide habitats for young-seral species and thus contribute to overall biodiversity of the region (Swanson et al. 2011, 2014). For example, in the Klamath-Siskiyou province, early-successional vegetation, including grass- and shrub-dominated areas and areas of young trees and shrubs, helps support abundant populations of dusky-footed woodrats, which are a primary prey of northern spotted owls in that area. Lists of special status and sensitive species of Forest Service and BLM in Washington and Oregon include a few species associated with early-successional vegetation conditions, such as green-tailed towhee (*Pipilo chlorurus*).

However, natural early-successional vegetation conditions have been reduced throughout the Western United States, including the Pacific Northwest, from plantation forestry operations of tree planting, control of competing vegetation (typically broad-leaf or hardwood species), fire suppression, and some thinning (Bormann et al. 2015). The result is reduction in habitat for some species closely associated with complex, natural early-successional conditions and with landscape mosaics of early-successional and old-growth forests. For example, Betts et al. (2010) reported declines in bird species associated with early-seral broadleaf-dominated vegetation following stand-replacing disturbance, including timber harvest and fire, in the Pacific Northwest. However, variable-density thinning and heavy thins can promote subsequent growth of hardwood shrubs and trees and provide habitat for some species of lichens, invertebrates, and other organisms (Schowalter et al. 2003, Wilson and Puettmann 2007), but more study is needed on specific conditions and effects on late-successional and old-growth species (Wilson and Forsman 2013). In general, though, early-successional vegetation created naturally from fire, windstorm, or other natural disturbances usually differs substantially in structure and composition of plant and animal species from that created by timber harvest subsequently subject to tree planting and control of competing vegetation.

Early-successional vegetation conditions created artificially also can become sources of plants, invertebrates, and other species that can compromise adjacent forest-interior conditions in late-successional and old-growth forest patches. We have noted that recent historical increases in artificially created early-successional vegetation have likely provided increased habitat for predators of mustelids, such as bobcats and coyotes.

Depending on amounts and dispersion patterns, early-successional vegetation can serve to fragment otherwise extensive and connected late-successional and old-growth forest cover and impede movement, population connectivity, and population size of canopy-associated late-successional and old-growth species such as red tree voles, martens, fishers, northern flying squirrels, and brown creepers. Sollmann et al. (2016) found that forest thinning reduced local densities of northern flying squirrels which moved into adjacent unthinned stands; that there was no difference in effect from even thinning and variable thinning; and that their results underscore the need to consider effects of stand management in a broader landscape context, particularly regarding movement and dispersal (functional response) of the squirrels.

***Full range of conditions*—**The full suite of native species and biodiversity of the Pacific Northwest occurs only with a mix of the full range of environmental conditions, including all forest successional stages that support the full array of terrestrial vertebrates (e.g., Raphael 1988, Raphael and Marcot 1986) and geophysical conditions such as montane wet meadows that support associated invertebrates. The NWFP was designed to provide aquatic and late-successional and riparian forest environments; the most appropriate balance with other conditions (e.g., mixed of early- and late-successional stages) is beyond the scope of the NWFP's original mandate and its guidelines under the 1982 planning rule, and is a matter of public policy informed by the best science. The 2012 planning rule increases emphasis on ecosystem approaches to conserving species and ecological integrity, based in part on the coarse-filter approach of providing structures and functions of ecosystems. Thomas et al. (2006) called for the NWFP to be recast as an integrative conservation strategy with federal forests managed as dynamic ecosystems.

Climate change: indicators, tipping points, and thresholds—

Complicating management objectives to provide the full array of environmental conditions are the dynamic and potentially sudden influences of climate change on vegetation, fire, species distributions, and development of so-called novel ecosystems, that is, specific groups of species in assemblages and communities that have not previously occurred in those combinations (Collier 2015, Pace et al. 2015). The current locations of late-successional forest reserves under the NWFP may not be fully resilient to expected effects of climate change. We know little about how management of vegetation, fuels, and fire will be affected by changing climate and thus the future of late-successional and old-growth forests and associated species and biodiversity (Pfeifer-Meister et al. 2013, Wimberly and Liu 2014), although some indications are that, under the NWFP, large reserves in moist forests have been relatively stable compared to smaller reserves, and that moist forests may be at less risk of disturbance from climate change than are dry forests (see chapter 3).

Changes in climate, particularly increased temperatures and aridity, can adversely affect amphibians by altering regimes of local hydrology (see chapter 7 for more discussion of climate change effects on aquatic ecosystems), fire, and disease occurrence and transmission. More directly, it may cause physiological stress on species adapted to cool and moist microclimates. If climate change reduces availability and connectivity of higher elevation older forests, populations of associated carnivores such as lynx, wolverine, and marten might suffer, as summarized above ("Carnivores" section). In a study of 34 species of small mammals in montane environments in the Sierra Nevada Mountains and Lassen Volcanic National Park in California, Rowe et al. (2015) reported that 25 species shifted their ranges since the early 20th century in response to climate change, with high-elevation species contracting their lower range limits downslope and low-elevation species showing variable responses.

Some lichens and bryophytes found in late-successional and old-growth forests could serve as useful indicators of changes in, and thresholds of, forest conditions and climate and air quality (Ellis 2015, Hofmeister et al. 2015, Mölder et al. 2015). Some late-successional and old-growth-associated invertebrates also can serve as indicators of biodiversity and climate change risk (Yi and Moldenke 2005). Halpern et al. (2012) suggested that disturbance has passed a threshold level for bryophyte conservation in Pacific Northwest forests, and that 15 percent retention of late-successional conifer forests—which the authors cited as the minimum standard on federal forest lands in the Pacific Northwest[41]—is inadequate to maintain the abundance or diversity of late-successional and old-growth species.

Turner et al. (2015) developed models to simulate potential impacts of climate change and disturbances on vegetation in the Willamette River basin of Oregon. Their projections under climate-warming scenarios suggest that potential forest types will transition from evergreen needleleaf to a mix of broadleaf and needleleaf forms, with as much as a nine-fold increase in area burned by the end of the 21st century. Implications for lowland plants and animals suggest a shift to more warm-temperate and fire-tolerant species mixes. Research by Creutzburg et al. (2017) suggests a more modest change, as climate change will increase forest productivity and carbon storage capacity that will not be offset by wildfire under the legacy of management and fire suppression activities.

Carroll et al. (2010) modeled the NWFP reserve network under climate change stressors. They concluded that using spotted owls as an umbrella species for reserve design, as under the NWFP, included habitat for other dispersal-limited ("localized") species but will not include much of their core habitats, as defined in the Zonation model they used, under projections of climate change. Thus, the current array of fixed late-successional and old-growth reserves designed for the owls may not suffice for other species that may change locations as their habitats shift under climate change. The authors suggested that a fixed-reserve system should be designed for resilience under anticipated climate change by including additional areas, particularly by adding reserves in higher elevation sites (Carroll 2010). However, not all species will respond to climate change by migrating their distributions to higher elevations. Crimmins et al. (2011) reported anomalous downhill shifts of some plant species in California, following regional changes in climatic water balance rather than responding to changes in temperature. Tingley et al. (2012) projected effects of climate change on altitudinal distributions of birds in the Sierra Nevada of California, and found that species will likely respond differently and individually at their range margins to increasing temperature and increasingly variable precipitation. Similarly, variable responses by plants, birds, and other taxa in the NWFP area are likely. The lesson is that the response to changing climate and associated weather conditions will vary, sometimes greatly, among species, and such variability will likely occur in all taxa and species groups.

Bagne et al. (2014) evaluated the efficacy of managing for special status species under climate change and its secondary effects on providing for overall biodiversity in the Southeastern United States. They concluded that 74 percent of terrestrial vertebrates potentially vulnerable to climate change were not included in current lists of special status species, and omissions were greatest for birds and reptiles. Current lists of special status species—potentially including those of the ISSSSP in the Pacific Northwest under the NWFP—could be evaluated for potentially adding species that might be vulnerable to climate threats, particularly with regard to those environmental conditions that would exceed physiological thresholds, if such species were lacking. Such analyses have not been conducted across all taxa, although this chapter and others in this compendium (e.g., chapter 2) provide insights on some species that might be vulnerable to future climate change effects, such as marten, pika, and others.

Aside from climate change effects are questions of balancing conservation between biodiversity and individual species. For example, Arthur et al. (2004) modeled tradeoffs in western Oregon between maximizing overall species richness and providing for individual endangered species. They found that the tradeoff was nonlinear, whereby increasing endangered species protection from

[41] The reference here to 15 percent retention might more clearly pertain to the 1994 Standards and Guidelines (USDA and USDI 1994, page C-41) that state "Landscape areas where little late-successional forest persists should be managed to retain late-successional patches. This standard and guideline will be applied in the fifth field watersheds (20 to 200 mi^2 [52 to 518 km^2]) in which federal forest lands are currently comprised of 15 percent or less late-successional forest."

90 to 99 percent caused a decline in all-species protection 2 to 14 percent. Their risk tradeoff approach could be applied elsewhere throughout the Pacific Northwest and NWFP area to evaluate variations in proportion of landscapes in late-successional and old-growth reserves and managed matrix lands (Arthur et al. 2002). However, their model did not quantify species persistence or viability, and it did not evaluate endemic, rare, or specialized species considerations that could be added to their analysis.

Matrix lands and habitat connections—
Many studies suggest that habitat connectivity, fragmentation, and isolation are among the most dire stressors on populations, and that, under environmental changes, movement barriers for many species are likely to shift over space and time (Caplat et al. 2016). Much has been written in the conservation biology literature on wildlife habitat corridors and connectivity among habitat patches and reserve areas (e.g., Beier and Brost 2010; Beier et al. 2008, 2011). The literature has addressed the influence of conditions in matrix lands on arthropods (Shields et al. 2008), butterflies (Ross et al. 2005), birds (Neuschulz et al. 2013), and mammals (Kurek et al. 2014). Beier et al. (2008) suggested designing habitat linkages between wildlands to provide for multiple species rather than single focal species. Brady et al. (2009) used measures of disturbance, structure, and floristics in habitat core areas, habitat patch edges, and matrix landscapes to assess small mammal populations in northeast Australia. They found that small mammal conservation entails controlling disturbances and providing for habitat preservation and restoration within matrix lands. Wessell's (2005) study of abundance and diversity of vascular plants, arthropods, amphibians, and mollusks in western Oregon suggested the value of managing forest matrix lands for habitat heterogeneity.

Several studies in the Pacific Northwest have suggested a need to view the LSR system under the NWFP as a habitat network (Molina et al. 2006), and to provide further connectivity of forest reserves to account for anticipated climate-change shifts in habitat quality for sundry species (Beier and Brost 2010; Carroll et al. 2010; Rudnick et al. 2012; see also Spies et al. 2010). Proulx and Santos-Reis (2012) determined the value of conserving late-successional and old-growth forests and structures in the Pacific Northwest as habitat for Humboldt and Pacific martens. Habitat connectivity for organisms such as northern flying squirrels and red tree voles that depend on contiguous tree canopies can be severed with heavy thinning of forests designed to accelerate diameter growth of trees (Wilson and Forsman 2013).

In general, these studies suggest also that the degree to which protection of species' habitats in the matrix may contribute to population conservation is species-specific, and depends in part on the size and configuration of habitat in core areas, patches, and habitat linkages, and in the amount of habitat occurring in land already conserved (e.g., NWFP LSRs).

Coarse and fine filters in conservation—
Since the last science synthesis, much has been written on sundry approaches to management of species, biodiversity, and ecosystems. These approaches include coarse- and fine-filter management; use of surrogate species such as indicators, umbrellas, keystones, and flagships; management of ecosystem services; management for the range of natural or historical variation; and much more. Much of this literature poses approaches with scant empirical testing and validation of their efficacy, however. In this section, we review recent literature on such topics as pertains to the Pacific Northwest.

A popular method for conservation assessment and management uses both coarse filters and fine filters (Hunter 1991, Noss 1987). This approach first addresses coarse-level, spatially broad environmental conditions to meet general habitat associations assumedly of a large number of species, and next adds specific environmental components and locations to meet the needs of additional species not sufficiently provided by the coarse-filter level. Such an approach was used to craft the late-successional forest and the riparian and aquatic reserve system under the NWFP. Further, the Survey and Manage standards and guidelines and its annual species reviews were specifically designed to determine which late-successional and old-growth-dependent species would not be sufficiently provided by the coarse-level reserves and may need additional fine-filter management. Determinations were made based on a rigorous, structured approach to evaluating new information (Marcot et al. 2006).

In the Pacific Northwest, Lehmkuhl et al. (2007) applied a coarse-filter approach to define the historical range of variability of the composition and pattern of forest conditions under historical fire regimes, and then a fine-filter approach to establish fuel reduction treatments at the landscape scale to provide conditions for the food web involving northern spotted owls and their prey of northern flying squirrels and bushy-tailed woodrats (*Neotoma cinerea*). Overton et al. (2006) used a coarse-filter regression approach with Gap Analysis Program information to evaluate use by band-tailed pigeons (*Patagioenas fasciata*) of abandoned mineral sites in western Oregon. In a more complex variant to the approach, Higgins et al. (2005) devised a four-tier spatial approach to identify areas critical to conservation of freshwater biodiversity in the Columbia River basin of the Pacific Northwest.

Thompson et al. (2009) used a coarse-filter approach to define the historical range of variability of forests in the Oregon Coast Range as a target for biodiversity conservation, coupled with what they termed the social range of variability to account for resource and land use patterns. They found that land development, shifts in anthropogenic fire regimes, and climate change will likely impede the use of historical range of variability as a management objective, and that more complex planning is needed in a continuous process of negotiation.

A coarse-filter approach to ecosystem management can also assume that emulating natural processes and disturbance dynamics, such as natural fire regimes, will provide for forest biota at natural population levels (Armstrong et al. 2003). Other variants of the coarse-filter approach include identifying and protecting biodiversity "hot spots" under the assumption that most or all biota will be provided. Some applications of the "hot spot" approach focus not on overall species richness but on identifying concentrations of locally or regionally endemic species, which tend to be range restricted and perhaps extinction prone. "Endemism hot spots" could signal climate refugia, and protecting them could be a useful tool for avoiding extinction under climate change (e.g., Harrison and Noss 2017).

The strength of the coarse- and fine-filter approach is that it can account for a wide array of species generally associated with an ecosystem condition such as late-successional or old-growth forests. The weakness of this approach is that it may require site surveys, habitat and ecological studies, and intensive assessments of individual species to determine which require fine-filter attention. Fine-filter management is typically viewed as management and conservation of species-specific habitats. This means that understanding species taxonomy is key to determining biological entities of potential conservation concern. In this chapter, we cite several instances of recent taxonomic findings describing newly identified species. Recent work on mitochondrial DNA (mtDNA) of flying squirrels now suggests that a geographically limited Pacific coastal form of northern flying squirrel ranges from southern British Columbia to southern California (Arbogast et al. 2017); the authors suggest the new species be designated Humboldt's flying squirrel, using the previously suggested epithet *Glaucomys oregonensis*. Nearly all of the occurrence of flying squirrels in the NWFP area now pertains to this newly described species, although there seems to be overlap with the northern flying squirrel (*G. sabrinus*) in western Washington. Thus, two species of flying squirrels may occupy the NWFP area, and their individual conservation status has yet to be determined.

Hunter (2005) suggested use of an additional "mesofilter" level to complement the coarse-filter strategy for conservation of entire ecosystems in reserves and the fine-filter strategy for conservation of selected individual species. The mesofilter ("middle" filter) level would focus on conserving critical ecosystem components, specifically microhabitat attributes such as logs, snags, and pools used by invertebrates, fungi, nonvascular plants, and other species groups that are often overlooked in conservation planning. Hunter suggested that the mesofilter approach is particularly useful with seminatural conditions, for sustaining biodiversity and commodity production. In a sense, management for snags, down wood, riparian conditions, and other ecosystem components, as already part of agency activities in the NWFP area, constitutes a mesofilter approach.

Assumptions of the coarse- and fine-filter approach, however, are seldom empirically tested in a controlled and rigorous manner, and their blind application may lead to inappropriate expectations for species conservation (Cushman et al. 2008, Hunter 2005). In one test in Oregon, Fagan and Kareiva (1997) found that hot spots of biodiversity (species richness) did not coincide with locations of rare and endangered butterflies. In another test in Oregon, Cushman et al. (2008) found that only 4 percent of the variation in bird species abundance was explained by using a general characterization of habitat as a proxy to bird species abundance and vegetation cover type as a proxy to habitat. Further, Molina (2008) found that 90 percent of fungal species has some fraction of their known sites occurring within LSRs in the NWFP area, but some 66 percent of all sites of Survey and Manage fungal species occurred outside the reserves, so that the coarse-filter level alone to identifying late-successional and old-growth forest reserves would not provide for many rare fungi. There may be some conservation salvation for these sites outside late-successional and old-growth forest reserves, insofar as the reserve strategy has changed, at least de facto, with little to no harvest of federal-land late-successional and old-growth forests outside the original NWFP LSRs (see chapter 12).

Related to the coarse- and fine-filter approach to conservation is the application of species- and system-level approaches to biodiversity conservation. Species-level approaches include use of population viability analyses; surrogate species approaches with the use of umbrella species, focal species, guild surrogates, habitat assemblage surrogates, management indicator species, biodiversity indicator species, apex predators, and flagship species; multiple-species approaches with use of entire guilds and entire habitat assemblages; and geographically based approaches with identifying locations of target species at risk and species hot spots or concentration of biodiversity (Marcot and Flather 2007). System-level approaches include the use of range of natural variability, key habitat conditions, keystone species, ecosystem engineers, and approaches to emulate or provide for regimes of fire, herbivory, key ecological functions of species assemblages, and food webs (Marcot and Sieg 2007). Much has been written about most of these species- and system-level approaches (e.g., Branton and Richardson 2014, Breckheimer et al. 2014, Hunter et al. 2016), including due caution in using them without clarity of definitions (Barua 2011) and empirical verification of their assumptions (e.g., Bifolchi and Lodé 2005).

Each of these species- and system-level approaches carries strengths and weaknesses depending on management objectives. No one approach is fully effective in providing for species diversity, genetic diversity, and ecosystem diversity conservation objectives, which is why the 2012 planning rule specifies use of both coarse- and fine-filter management. Raphael et al. (2007) and Raphael and Marcot (2007) suggested that a combination of approaches may be efficacious for meeting multiple conservation objectives. They suggested a series of steps to identify the suite of species- and system-level approaches for conservation of rare or little-known species and their habitats, and they emphasized the critical importance of setting clear goals, identifying measurable short- and long-term objectives, and including learning objectives to increase knowledge.

Uncertainties and Research Needs

Indicators and Surrogates

The debate over use of surrogate species in its many forms—indicators, sentinels, flagships, umbrellas, keystones, and others—bears closer scrutiny. The literature seems divided over cautioning against its use for more holistic ecosystem management, and using it with caution for more focused needs with clearly stated objectives. The Forest Service guidance on implementing the 2012 planning rule calls for the use of a surrogate-species approach to evaluating conditions contributing to viability of groups of species of conservation concern. In this approach, the viability of the surrogates is assumed to represent viability of the broader subset of species of conservation concern species with similar ecological requirements and similar responses to environmental conditions and changes. Surrogates are chosen also on the basis of their having more stringent ecological requirements than others of the group. Most important, surrogates are meant to represent ecological conditions that contribute to viability of the broader species group and are not meant to directly represent their population dynamics per se; this qualification may add some uncertainty over the response of the fuller species groups (Wiens et al. 2008) but is truer to the concerns expressed in the literature over use of indicators of population viability.

In many ways, the evolving planning and management framework under the NWFP has largely met earlier calls for conservation of forest biodiversity and other species. This includes viewing forests as dynamic ecosystems, explicitly providing for ecosystem processes, addressing little-known species, and attending to conditions in the managed forest matrix (Franklin 1993). Recent scientific publications have continued to suggest using management indicator species and umbrella species only with caution and empirical evaluation (Branton and Richardson 2010, 2014; Wiens et al. 2008), or not at all, because the terms are typically poorly defined and the concepts often untested. Simberloff (1998) argued that moving toward an ecological functional basis for species and biodiversity management, rather than using simpler proxies of indicator species, is more likely to succeed, particularly with application of ecological forestry approaches. Also, much recent scientific literature has moved beyond the contentious era of arguing whether species or systems should be the focus for conservation; the approaches are, as Lindenmayer et al. (2007) and Sergio et al. (2003) noted, complementary, including with the debate over conserving species versus ecosystem services (Hunter et al. 2014, Kline et al. 2016; also see chapter 8). Ultimately, successfully combining species and system approaches depends on clearly articulating management objectives and determining the efficacy of management through monitoring and research.

Climate Change

Climate change is at the heart of many conservation concerns in the Pacific Northwest, but much empirical study remains to better understand its effects on species, their habitats, and their ecological functions, and to avoid the easy but potentially erroneous assumption that it is the proximate cause of most or all observed perils, such as with the decline of amphibians (Davidson et al. 2001, 2002). Climate change might act indirectly, such as by reducing snowfall (Corn 2003, Forister et al. 2010) and increasing evaporation, leading to loss of wetlands (Corn 2005); by increasing fire frequency and extent; and by improving environmental conditions for diseases or pathogens that in turn could affect other organisms, again such as with the decline of amphibians (Carey and Alexander 2003).

Further, there may be higher order effects whereby climate change in turn alters biotic interactions among species, such as Preston et al. (2008) reported on an endangered butterfly and a threatened bird species in southern California, which led the authors to suggest that considering species interactions is important when designing reserve systems for conservation of sensitive species. How this can influence any redesign of reserves under the NWFP would require study.

Microbiota and ecosystem functioning—
There is a dearth of knowledge and understanding of Wilson's (1987) "little things that run the world," referring to the invertebrates, particularly soil microorganisms and mesoarthropods, along with the suite of poorly known fungi and allies, that collectively play crucial roles in organic matter breakdown, nutrient cycling, and maintenance of forest health (Berg and Laskowski 2005, Ulyshen 2016, Tolkkinen et al. 2015). There is also much to be learned about how biodiversity affects ecosystem functioning (Hooper et al. 2005, Marcot 2007). Here is where an interplay and complementarity of species-focused and ecological-process research and management can usefully provide for forest ecosystem conservation and restoration to meet early visions of achieving sustainable forestry for conservation of late-successional and old-growth forests and their biodiversity (Beebe 1991, Crow 1990).

The future of other species is not written, but is progressively being tilted by fluctuating dynamics of fire regimes, shifts of forest composition and structure from changing climates, increasing stressors of invasive species, and changes in societal values with increasing needs for progressively scarce forest resources and their ecosystem services. Successfully addressing these and related issues for restoration and conservation of other species of late-successional and old-growth forests necessitates continuing on the path to whole-ecosystem research and adaptive management in a socioecological scope (Ban et al. 2013, Schmiedel et al. 2016; also see chapters 8 and 11).

Conclusions and Management Considerations

Summary of Key Management Considerations

Here we apply our key findings to the set of guiding questions posed after the introduction of this chapter. We also address this additional question: If all late-successional and old-growth forests are now protected under the NWFP (and under designated critical habitat for the northern spotted owl), is there a need for additional fine-filter approaches, beyond those for the northern spotted owl and marbled murrelet, for species assumed to be associated with said forests? We will conclude the affirmative, that sufficient uncertainty exists, particularly with future changes in old-forest environments and species-specific habitats, particularly for species that are rare or poorly known or that are expected to be increasingly adversely affected by disturbances and changing climate. As to which species may need to be so studied and provided is at present unclear, although advances in identifying species of conservation concern under the 2012 planning rule will be a major step forward.

Current scientific understanding of late-successional and old-growth-associated species rarity—
Although no specific analysis has been conducted to quantify the degree of rarity of late-successional and old-growth-associated species in the latest Survey and Manage species list, the general trends since the 2006 synthesis (Haynes et al. 2006) are:

1. Scientific understanding has greatly expanded on the occurrence, distribution, ecology, and potential threats of many of these species, as noted by the rich array of inventory reports and conservation planning documents conducted by the ISSSSP.
2. Numerous Survey and Manage species had been removed from that list, during the last rounds conducted on the annual species reviews, largely because predisturbance site surveys suggested that the species are more common or frequent than previously thought and because they no longer met the concern for the basic criteria for persistence of Survey and Manage species.
3. The best scientific information to date on many of the remaining Survey and Manage species, as summarized by the ISSSSP, suggests continued rarity or potential vulnerability to habitat disturbance. It is true that some old-forest associated species are naturally rare and perhaps more vulnerable than are more abundant species, but neither the NWFP guidelines nor the 2012 planning rule state that naturally rare species are to be discounted. Although it may not be possible or feasible to increase the distribution or abundance of naturally rare species, their preservation or conservation is very much in the spirit of the NWFP. The specific lists of species currently considered under the ISSSSP are found on the web site cited above.

Planned provision of habitat for rare and uncommon species under the NWFP—
The NWFP was initially envisioned as providing habitats for late-successional and old-growth-associated species through a combination of LSRs, AMAs, aquatic and riparian reserves and buffers, and selected conservation of late-successional and old-growth forest stands in the managed forest matrix. At present, however, there is little to no harvest of late-successional and old-growth forest stands on BLM and Forest Service lands within the NWFP area, including within and outside reserves (see chapter 3). In this way, the current implementation of the NWFP is not the same as initially envisioned.

Further, there has been no official effectiveness monitoring program under the NWFP, so there is limited information for determining the degree to which all old-forest associated species are being provided, and determining their habitat and population trends. Without a formal effectiveness monitoring or research program for other species and biodiversity under the NWFP, much has to be inferred from vegetation and late-successional and old-growth conditions and trends, although some research is available on selected species and taxa, as reviewed in this chapter.

Rare and uncommon species populations under NWFP management—
The degree to which remaining late-successional and old-growth forest is adequately providing for persistent and sustainable populations of rare and uncommon species is known for only those few species for which demographic and monitoring studies have been conducted, such as with northern spotted owls and marbled murrelets. Lichen

sampling, if reinstituted in the Forest Service's FIA program, could provide answers to this question on late-successional and old-growth-associated species. Otherwise, as under the 2012 planning rule, assumptions need to be made on the efficacy of coarse-filter approaches to providing general ecosystem conditions and their adequacy to in turn provide habitat and resource conditions for individual species. To gain greater knowledge and understanding of the adequacy of these assumptions may entail further species-specific research and monitoring, perhaps including species previously deemed secure but now perhaps with uncertain futures from climate change and other disturbances unforeseen in the initial NWFP guidelines.

Efficacy of habitat conservation management recommendations—

There has been no effectiveness monitoring program for the bulk of Survey and Manage species. Various studies suggest some degree of success for specific species groups, such as headwater amphibians. For most species, however, no monitoring data are available.

Sufficient information to change species management status—

This would best be determined through an evaluation of recent scientific data, survey information, and other sources, through a structured species review process (e.g., the previously instituted annual species reviews under the Survey and Manage program). The ISSSSP has compiled and organized much survey and scientific information on the Survey and Manage species but has not been charged with conducting such species reviews per se. It is likely that significant information has been gathered on some species for making this determination, but at least some of the determination will not be made until or unless an adaptive learning and management procedure, along the lines of the annual species reviews (last conducted in 2003), is implemented again.

Species persistence under NWFP reduced late-successional and old-growth harvest—

No analysis has been conducted on threats to late-successional and old-growth-associated species, comparing threat levels perceived under the NWFP as initially envisioned with the NWFP as currently implemented. The NWFP as initially envisioned included late-successional and old-growth reserves, aquatic and riparian reserves and buffers, AMAs, and site-specific conservation of late-successional and old-growth forest stands. The NWFP as currently implemented includes little to no harvest of late-successional and old-growth forests on Forest Service and BLM lands within the NWFP area (see chapter 3); and provides for some management in LSRs in dry forests. Such a comparison between initial and current management conditions could recognize the greater roles and potential changes from shifts in local climate and especially in fire suppression, fuels management, and alternative ecological forestry approaches providing for the development of natural, complex early-seral and young-age forest vegetation conditions.

Under this suite of considerations, it is likely that some species originally ranked as having low potential for persistence might be viewed as having higher potential, particularly if their late-successional and old-growth forest habitats and specific habitat elements are better provided throughout the matrix as anticipated over time. Other species tied to specific locations that are vulnerable to fire and disturbances might not fare as well, but no evaluation is currently available. It is likely that location data accumulated from predisturbance and strategic surveys might suggest adequate numbers by which to reduce concern for the persistence of some late-successional and old-growth-associated species, but predisturbance surveys alone do not provide information on whether the species survived the disturbance, only that they were present before the disturbance.

Late-successional and old-growth species added to species-of-concern lists—

Several "species lists" cover the NWFP area, including the lists of Survey and Manage species; Forest Service and BLM special status and sensitive species; U.S. Fish and Wildlife Service's candidate, threatened, and endangered species under the Endangered Species Act; and various lists of species of conservation or vulnerability concern from International Union for Conservation of Nature Red Data Lists, state lists, Washington Natural Heritage Program, Oregon Biodiversity Information Center, and others. The Forest Service also is currently compiling a regional list of potential species of conservation concern that will also intersect with the NWFP area, its late-successional and old-growth forest-associated species, and many of the other lists noted here. There has been no evaluation of the composite set of species among all these lists, comparing to the list of late-successional and old-growth species originally ranked as high persistence or not identified as conservation concern under the NWFP.

This report has summarized new threats or new scientific information revealed over the past decade, suggesting potentially growing conservation concern for late-successional and old-growth forest-associated bats, should white-nosed syndrome spread in the Pacific Northwest; highly locally endemic mollusks and other invertebrates, under threat of invasive invertebrate species or reduction in key habitat elements (e.g., large, old bigleaf maples for the Puget Oregonian snail); new species recently split taxonomically and that occupy a smaller portion of the former species' range and that might associate with late-successional and old-growth forest environments that could be at risk from increasing wildfire (e.g., the new species of sharp-tail snake recently identified from northwest California and southwest Oregon); and other examples. For some species, a more extensive evaluation is in progress; for others, it awaits.

In the past annual species reviews, no species was added to the initial Survey and Manage species list.[42]

[42] New information to support adding a species to the Survey and Manage list (USDA and USDI 2001:15-16) must address the following three basic criteria including the specific factors used as a basis for determining concern for persistence, in addition to criteria for late-successional and old-growth association:

- The species must occur within the NWFP area, or occur close to the NWFP area and have potentially suitable habitat within the NWFP area.
- The species must be closely associated with late-successional or old-growth forest.
- The reserve system and other standards and guidelines of the NWFP do not appear to provide for a reasonable assurance of species persistence.

The specific factors must apply to at least an identified portion of the species range, on federal lands, within the NWFP area. One or more of the following factors may indicate that persistence is a concern. These factors must be considered in the context of other standards and guidelines (other than those related to Survey and Manage) in the NWFP:

- Low to moderate number of likely extant known sites/records in all or part of species range.
- Low to moderate number of individuals.
- Low to moderate number of individuals at most sites or in most populations.
- Very limited to somewhat-limited range.
- Very limited to somewhat-limited habitat.
- The distribution of the species within habitat is spotty or unpredictable in at least part of its range.

Effects of fire on rare and uncommon late-successional and old-growth species—

Species and ecosystems of the NWFP experience diverse forest types and disturbance regimes (Bunnell 1995). Regarding effects of fire on rare and uncommon species associated with older forests, research results are limited and mixed. For example, for some species such as white-headed woodpecker, small patches of prescribed burning can be used to create or elevate the quality of their habitat. For other species, prescribed fire (particularly associated with sanitation and safety harvests) and associated fuels reduction activities can diminish key late-successional and old-growth habitat components such as large hollow trees, large snags, and large down logs, in turn reducing habitat for a wide variety of denning and cavity-using wildlife species, including fishers, Humboldt marten, and other mammalian carnivores (fig. 6-3).

Effects of high-severity, stand-replacing wildfire are perceived as largely adverse to species that use dense, multilayered older forests, although much of the literature has not clarified levels of wildfire intensity, extent, and location when speculating or concluding on wildfire effects. Research suggests that (high-severity) fire can reduce beneficial mycorrhizal and other desirable fungi, but for the most part, the effects of wildfire on individual, rare and uncommon late-successional and old-growth species are largely unstudied. Insofar as atypically severe wildfire can reduce canopy closure and fragment dense, contiguous-canopy stands, it adversely affects habitat for red tree voles.

Little has been studied on the direct effects of suppression of wildfire on wildlife in the Pacific Northwest.

Efficacy of site buffers compared with landscape-scale habitat management for species persistence, dispersal, and habitat connectivity—

No studies have specifically compared site buffers with landscape-scale management of late-successional and old-growth forests, as influencing persistence, dispersal, and habitat connectivity of late-successional and old-growth-associated species. Similarly, little research has been done in the effectiveness of landscape-scale management in assuming a reasonable assurance of persistence

Figure 6-3—Naturally hollow Douglas-fir log in a lowland old-growth conifer forest, west Cascade Mountains of northern Oregon. Such logs are prime denning sites for fishers, Humboldt marten, and other mammalian carnivores. Some prescribed fire, timber salvage operations, and fuels reduction activities can reduce such habitat elements.

of late-successional and old-growth-associated species. However, some studies have noted that small patches of late-successional and old-growth forests, and legacy elements of late-successional and old-growth forest such as large green trees, large snags, and large down wood, can provide limited habitat for old-forest-associated species in the forest matrix. Such patches can also be used as points of restoration of older forest. Studies of old-forest remnants and fragments suggest that, by themselves, they provide only a fraction of the original forest biodiversity. Although they may be valuable to conserve for that purpose, if they are the only remaining elements of late-successional and old-growth forests in an area, they cannot be counted on to provide for persistence, dispersal, and habitat connectivity of late-successional and old-growth-associated species.

In general, riparian stream buffers, including unthinned riparian vegetation and adjacent upland forests, have been shown to have immense conservation values, particularly for amphibians. Buffers of sufficient width along streams and wetlands can serve as movement corridors and landscape linkages for aquatic frogs, salamanders, reptiles, and birds (Perry et al. 2011, Vergara 2011). Streams with riparian buffers can provide for some birds (Nimmo et al. 2016) and arthropods. Studies suggest that buffer widths differ to accommodate different species or species groups, as noted in summaries above.

The ISSSSP under the NWFP—
The ISSSSP has provided a wealth of information to agency researchers and managers on late-successional and old-growth-associated species, such as conservation assessments, summaries of surveys, threats evaluations, range maps, survey protocols, and ecological studies. It should be remembered that the ISSSSP pertains only to Washington and Oregon; in California, the Forest Service and BLM each have their own species programs and lists.

The current special status and sensitive species list—
Some Survey and Manage species also have sensitive or strategic species status, but the ISSSSP list does not include Survey and Manage species, which are maintained as separate lists based on different criteria. The current special status and sensitive species list includes those Survey and Manage species that qualify based on the ISSSSP criteria. The current ISSSSP list is broader in application covering all of Oregon and Washington with updates as recent as July 2015. The ISSSSP list includes a broad array of fungi, lichens, bryophytes, vascular plants, vertebrates, and invertebrates. Late-successional and old-growth species, or any species for that matter, with risks to persistence would likely have been included in the list update. No changes have been made to the Survey and Manage species list since 2003 owing to capacity and funding limitations to conduct an annual species review (the adaptive management process that removes or adds species or changes Survey and Manage category based on new information).

Additional fine-filter approaches for species assumed to be associated with late-successional and old-growth forest—
Not all late-successional and old-growth forests are being protected, but surveys for Survey and Manage species are being conducted in stands over 80 years of age. Much remains to be learned about demography, persistence, viability, dispersal, and habitat connectivity of most late-successional and old-growth-associated species. The coarse-filter approach assumes that providing for general habitat conditions at a broad scale, such as through LSRs and aquatic and riparian reserves, will provide for many specific elements, but research informs on situations to be wary of taking that assumption at face value without additional information and tests. Additional knowledge is needed especially on many late-successional and old-growth-associated species still known as rare or uncommon, or that are little-known—such as most soil and canopy invertebrates, fungi, and many amphibians—and on highly locally endemic species such as some aquatic mollusks, before fine-filter (species-specific) conservation measures can be confidently replaced by coarse-filter guidelines.

Issues of Species Conservation

As a program to replace or complement the Survey and Manage program under the NWFP, the ISSSSP—spanning the Forest Service Region 6 and BLM Oregon and Washington—has produced much material since the 2006 NWFP science synthesis on a wide variety of little-known or rare late-successional and old-growth-associated species. Their products pertain directly to contributing information on conservation and management of Survey and Manage species under the NWFP and sensitive species across Oregon and Washington. The products include basic ecological information, field inventories and monitoring results, protocols for surveys, and conservation guidelines. Ongoing local monitoring efforts at district levels will continue to inform on the status of little-known and rare late-successional and old-growth-associated species, including if and when individual species should be of additional conservation concern or are doing well.

The Survey and Manage program produced guidelines and approaches on natural history and management considerations for hidden, rare, or little-known species, including soil invertebrates and some lichens. Still, research is needed on their habitat associations, ecology, and degree of tolerance to disturbances of many rare and little-known, late-successional and old-growth-associated species of fungi, lichens, bryophtes, and vascular plant species.

Beyond the biological sciences, successfully conserving, restoring, and managing for other species under the NWFP will take additional attention to social needs and interests (see chapters 8, 9, and 11). Fostering or retaining public support for conservation and restoration of rare and little-known species may require addressing social beliefs and values, and clarifying impacts and rationale of management policies, context, and actions (Stankey and Shindler 2006; see chapters 9, 10, and 11). Such was the case in the formulation of the initial NWFP guidelines, and such may be required again if the Plan is to be expanded to include other species, their habitats, or objectives beyond late-successional and old-growth forest, aquatic, and riparian environments.

Accounting for dynamic systems and uncertainties— Recent science findings suggest that successfully managing for the future of other species under the NWFP needs to explicitly account for disturbance dynamics (Odion and Sarr 2007) in forest ecosystems of the Pacific Northwest, principally fire and anthropogenic changes to species' habitats in the managed forest matrix. Collectively, our findings suggest that such changes to forest age classes, structures, and dispersion patterns may have had greater impact on species metapopulation viability and on late-successional and old-growth forest resilience than initially thought by FEMAT, in large part because of increasing occurrence of high-severity fire in recent years.

Recent science findings also have highlighted the need to consider how continuing climate change will affect such disturbance dynamics and landscape-level connectivity of habitats for late-successional and old-growth-associated species. Researchers have suggested developing adaptation planning for climate change effects, in large part for (1) flexing and migrating boundaries of LSRs (as originally intended by FEMAT (1993), (2) conserving habitat in higher elevations, and (3) providing for site-specific conservation of late-successional and old-growth forest conditions within the forest matrix to serve as connections between the reserves. The federal matrix lands are not treated as they were when the NWFP was implemented; rather, late-successional and old-growth forests in the matrix lands are generally not being harvested at the rate they had been previously (see chapter 3), which may contribute to conservation of old-forest habitat for some species, although the specific degree of contribution is unstudied. Also, the Aquatic Conservation Strategy, with its riparian buffers and headwater conservation approach, seems to be providing for nonfish aquatic species, particularly stream-associated amphibians and arthropods.

There will always be uncertainty on species' ecologies and conservation effectiveness. Several researchers have suggested use of decision science and risk analysis approaches to evaluating and managing for NWFP late-successional and old-growth-associated species under uncertainty (e.g., Kerns and Ager 2007). Such approaches might prove increasingly useful as we further enter into an era of climate change-induced disturbances, by presenting evaluations of potential changes and expected effects of alternative management actions as probabilities of outcomes in a risk analysis and risk management framework. Bormann (2004) suggested a new approach to managing forests under uncertainty, which they termed "options forestry" that would entail diversifying management operations for the purpose of learning under an adaptive management framework. Franklin and Johnson (2012) suggested a restoration strategy ("ecological forestry;" see also Franklin et al. 2007, Hanson et al. 2012) for federal public forests in Washington and Oregon that would produce ecological and economic benefits by reserving older forest stands, thinning plantations to encourage complex vegetation structures, and conducting variable-retention harvests in younger forests to provide diverse early-seral environments.

Continuing the conservation of other species and biodiversity under the NWFP may benefit from use of decision science approaches including decision support models (Staus et al. 2010) and structured decisionmaking frameworks (Marcot et al. 2012b) that clearly articulate objectives; involve managers, decisionmakers, stakeholders, and analysts in all planning stages; and provide for monitoring and adaptive management to revisit, reaffirm, or revise management objectives and guidelines (Thompson et al. 2013), particularly under dynamics of changing social values, climates, and disturbance regimes (e.g., Jactel et al. 2012). The topic is explored more fully in chapter 12.

Coarse-filter, fine-filter, and landscape-scale management—

We have discussed recent findings on the use and testing of the coarse- and fine-filter approach to management, which is still the prevailing framework under the NWFP, the Survey and Manage standards and guidelines, and the ISSSSP. The approach has been conflated with the use of a variety of species- and system-level approaches, including use of umbrella species, flagship species, management indicator species, focal species, surrogate species, guild indicators, indicator groups, and others (e.g., Lawler et al. 2003). The coarse- and fine-filter approach is a simple two-tier method of (1) providing general conditions, such as late-successional and old-growth forests in LSRs, and aquatic and riparian environments in the NWFP's Aquatic Conservation Strategy, and (2) then testing the efficacy for conservation of all species and biodiversity attributes of interest, and devising and implementing additional management activities and requirements as needed to provide for the full suite of species' habitats and biodiversity elements. What recent science has found is that the results—the combination of environmental conditions—will undoubtedly shift under climate change and disturbance dynamics, and thus might need to be at least intermittently reevaluated and readjusted.

Further, the coarse- and fine-filter approach focuses on environmental conditions and species' habitats except with the northern spotted owl and marbled murrelet, which include monitoring of population-level demographic status and trends. Whether the approach assures, to a degree acceptable to management, the long-term viability of all populations for the remaining 80 years of the NWFP is not known.

Recent studies and evaluations of how well the NWFP provides for other species and biodiversity also suggest that the role and efficacy of late-successional (and aquatic and riparian) reserves needs to be put into the broader context of the managed forest matrix and other systems of the Pacific Northwest found mainly on federal public lands, specifically subalpine forests, rock and ice substrates, headwaters, complex early-successional vegetation, and unharvested postfire conditions. However, maintaining late-successional and old-growth forest ecosystems and their biota may not be fully achievable only with old-forest reserves, and only on federal forest lands, given effects of forest fragmentation and fire suppression and dynamics (e.g., Perault and Lomolino 2000, Sheehan et al. 2015). For these and other reasons, McAlpine et al. (2007) concluded that conservation of regional biodiversity under broad forest plans including the NWFP are better achieved with sustainable forest management practices implemented on—or at least coordinated across—all ownerships and by all stakeholders, not just for focused forest reserves. Social and economic considerations for any such forest management are discussed in chapters 8 and 12.

Other considerations—

Several other considerations may be pertinent and useful for managing for biodiversity and other species other than the northern spotted owl, marbled murrelet, and salmonids under the NWFP. One idea from recent science findings is the value of providing older-forest substrates and elements outside of LSRs that can contribute to species' habitats and habitat connections in the managed forest matrix (e.g., Dunk and Hawley 2009). An example pertains to dead wood. Approximately two-thirds of Survey and Manage species under the NWFP are associated in some way with, and benefit from mitigation for, partially dead trees, snags, and down wood (coarse woody debris). The Forest Service Decayed Wood Advisor (DecAID)[43] provides substantial information on wildlife use of snags, partially dead trees, and down wood, along with guidance on management of such elements at stand and landscape scales (Mellen et al. 2002).

Further, there may be an opportunity to bolster research, conservation, and restoration guidelines for providing such conditions within the managed forest matrix and across land ownerships. The objective would be to help connect LSRs, particularly now that late-successional and old-growth forests in the federal land matrix are currently being harvested at much less the rate they once were. In the past, thinning of plantations and young stands served to reduce dead wood, especially large down wood. Dead wood provides for a surprisingly vast array of life and its essential

[43] http://www.fs.fed.us/r6/nr/wildlife/decaid/; currently undergoing a major version update.

ecological processes (Brazee et al. 2014, Seibold et al. 2015). Providing more naturally regenerating, structurally and floristically complex early-successional stands, with the aim of encouraging development of dead wood and large down wood, may be part of restoration actions.

Protection of old-forest "legacies" such as large old green trees and snags and large-diameter down logs in the managed forest matrix has long been touted as a means of conserving biodiversity in Pacific Northwest old-growth forests following disturbances (Johnstone et al. 2016, North and Franklin 1990). Remnant, shade-tolerant old-growth trees (*Thuja* spp.) have been found to provide biological legacies and seed sources in postfire conditions in Pacific Northwest conifer forests (Keeton and Franklin 2005). However, retention of old-forest legacies may not fully substitute for retention of old-growth forest stands per se. For example, Price and Hochachka (2001) reported that epiphytic and alectoroid lichens were less abundant in mature (70- to 120-year-old) stands with structural retention of old-forest legacies, than in old-growth (at least 300-year-old) stands. The point here is that old-forest legacies retained in early-successional forests can complement but not fully supplant old-growth forests in providing habitat for old-forest species.

There may be considerations for tradeoffs among the kinds of species that can be provided under different silvicultural activities such as different intensities of forest thinning. For example, in the Pacific Northwest, Pollock and Beechie (2014) found that different sets of wildlife species associated with large-diameter live trees benefited more from heavy thinning in riparian areas than species associated with large-diameter deadwood benefited from light or no thinning. The authors suggested that because far more vertebrate species use large deadwood than large-diameter live trees, riparian areas may best be left to develop naturally in the absence of thinning for the benefit of terrestrial and aquatic species. Consideration for such tradeoffs of management objectives is discussed more fully in chapter 12.

Another consideration pertains to maintaining the full suite of ecological functions provided by late-successional and old-growth-associated species. This helps provide for "fully-functional" ecosystems, including full food webs and ecosystem processes of all successional stages (Marcot 2002, Marcot and Sieg 2007, Marcot and Vander Heyden 2001).

Finally, there may be efficiencies and advantages to continue coordination of NWFP implementation and any amendments or updates, with other programs including climate science centers, disturbance research programs, and other protected area network programs.

Main Findings

Much has been learned since the 2006 synthesis (Haynes et al. 2006) about a wide variety of species and their occurrence, distribution, and rarity, and some on their ecology, reaffirming the role of LSRs and conservation of aquatic systems and riparian buffers in providing habitat for such species. Greater clarity also has been developed on the role of retaining old-forest components and substrates in the managed forest matrix to serve as connections among the reserves. Providing such connections is possible through the slowing of harvests of late-successional and old-growth forests in the federal matrix lands, the current critical habitat designation for northern spotted owls to protect all remaining owl habitat, and retention of old-forest structures in the forest matrix. Based on research on early-successional vegetation and associated biotic communities, how young stands in the forest matrix are managed will have a significant bearing on the future of older forest ecosystems throughout the NWFP area.

Recent work has also called for a far more dynamic approach to account for shifting movement barriers, fire disturbance to reserves and late-successional and old-growth forest stands, and other anthropogenic and climate-mediated changes in reserves and matrix lands alike. Habitats and population connections of other species of late-successional and old-growth forests are likely more vulnerable to the static placement of existing LSRs than previously envisioned. According to the research summarized in this chapter, a dynamic approach addressing the long-term scheduling of forest management activities and additions of reserves to further connect existing reserves and supplement them in higher elevations could better account for the influence of changing fire regimes, climate, and use of natural resources.

The Survey and Manage program, and more recently the ISSSSP, have provided much information on various rare and uncommon species associated with late-successional and old-growth environments. Their conservation evaluations provide evidence that the NWFP is generally providing habitat for late-successional and old-growth-associated species either under the late-successional, aquatic, and riparian buffer reserve system (coarse-filter management), or additionally through species-specific inventories, monitoring, and site protection (fine-filter management). Additional federal listing of late-successional and old-growth species as threatened or endangered, beyond northern spotted owl, marbled murrelet, and salmonids, has generally not proved necessary. Still, there may remain concerns for some species groups such as some late-successional and old-growth-associated bryophytes (Halpern et al. 2012) and other species groups because of the legacy of past forest harvesting patterns coupled with climate-change stressors (e.g., see section above on "Bryophytes"), but species-specific information is generally scant. For most of the other, rarer species addressed in this chapter, little to no information is available on population size and trend, especially their demographics across the managed forest matrix outside of reserves, although information is available on the general distribution or occurrence of some such species.

The Survey and Manage program, with its annual species reviews, provided the basis for reducing or removing concern for conservation of many uncommon to rare late-successional and old-growth-associated species that were originally ranked as having low potential for persistence. The lowering of conservation concern was due not so much to lower levels of harvest of older forests, but more to efforts locating such species during "pre-disturbance surveys" before local harvests and other management activities proceeded.

Since the 2006 synthesis (Haynes et al. 2006), no species have been added to the Survey and Manage species list; any additions would occur through a renewed annual species review process, and none was added during the reviews in 2001, 2002, and 2003. Those reviews resulted only in removing species from the list on the basis of new findings (viz., no concern for persistence, or the species not being late-successional and old-growth associated) or changing their conservation and monitoring categories based on new information.

Other than the research summarized in this chapter, little effectiveness monitoring has been conducted on site buffers to confirm that they are indeed providing for species persistence at those sites. Limited research on riparian stream buffers suggest high value for protected associated species' habitats.

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Metric Equivalents

When you know:	Multiply by:	To find:
Inches	2.54	Centimeters
Feet (ft)	.305	Meters
Miles (mi)	1.609	Kilometers
Acres (ac)	.405	Hectares
Square miles (mi^2)	2.59	Square feet
Pounds (lbs)	.454	Kilograms
Degrees Fahrenheit (°F)	.56(°F – 32)	Degrees Celsius

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Molalla River, Oregon.
Photo by Jeff Clark, USDI Bureau of Land Management, Oregon-Washington State Office.

Chapter 7: The Aquatic Conservation Strategy of the Northwest Forest Plan—A Review of the Relevant Science After 23 Years

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Introduction

The Aquatic Conservation Strategy (ACS) is a regional strategy applied to aquatic and riparian ecosystems across the area covered by the Northwest Forest Plan (NWFP, or Plan), encompassing broad landscapes of public lands administered by the U.S. Department of Agriculture Forest Service and the U.S. Department of the Interior Bureau of Land Management (BLM) (USDA and USDI 1994a). The ACS was developed during the analysis (FEMAT 1993) that led to the NWFP, but its foundation was a refinement of earlier strategies: the Scientific Panel on Late-Successional Forest Ecosystems ("The Gang of Four") (Johnson et al. 1991), PacFish (USDA and USDI 1994b), and the Scientific Analysis Team (Thomas et al. 1993).

The ACS uses an ecosystem approach to management of riparian and aquatic habitats (Everest and Reeves 2007) and was designed to (1) protect watersheds that had good-quality habitat and strong fish populations at the time the Plan was drafted, and (2) halt further declines in watershed condition and restore ecological processes that create and maintain favorable conditions in aquatic ecosystems in degraded ecosystems (FEMAT 1993). The long-term goal (100+ years) is to develop a network of functioning watersheds that supports populations of fish and other aquatic and riparian-dependent organisms across the NWFP area (USDA and USDI 1994a). The ACS is based on preserving and restoring key ecological processes, including the natural disturbance regimes (USDA and USDI 1994a) that create and maintain habitat for native aquatic and riparian-dependent organisms, and it recognizes that periodic natural disturbances may be required to sustain ecological productivity. As a result, the ACS does not expect that all watersheds will be in favorable condition (highly productive for the same aquatic organisms) at any point in time, nor does it expect that any particular watershed will remain in a certain condition through time. If the ACS and the NWFP are effective, the proportion of watersheds in better condition (for native organisms) is expected to remain the same or increase over time (Reeves et al. 2004).

The primary objective of the ACS is to maintain and restore the distribution, diversity, and complexity of watershed-level features and processes to which aquatic and riparian species are uniquely adapted. Programs and actions under the ACS are to maintain, not prevent, attainment of this goal. The ACS designates watershed analysis as the tool for developing baseline conditions against which to assess maintenance and restoration conditions, and improvements in biological and physical processes are to be evaluated relative to the natural range of variability (USDA and USDI 1994a). ACS objectives address (1) diversity and complexity of watershed features; (2) spatial and temporal connectivity within and between watersheds; (3) physical integrity; (4) water quality; (5) sediment input, storage, and transport; (6) instream flows (e.g., both peak and low flows); (7) floodplain inundation; (8) riparian plant-species composition and structural diversity; and (9) habitat to support well-distributed populations of native, aquatic and riparian-dependent species of plants, invertebrates, and vertebrates.

The ACS sets out five components to meet its goals:

- **Riparian reserves**: Riparian reserves are specifically designated portions of the watershed most tightly coupled with streams and rivers that provide

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the ecological functions and processes necessary to create and maintain habitat for aquatic and riparian-dependent organisms over time, as well as habitat connectivity within and between watersheds. The reserve boundaries were considered interim until a watershed analysis was completed, at which time they could be modified based on suggestions made in the watershed analysis.

- **Key watersheds**: 5th-code (40,000 to 250,000 ac [16 187 to 101 171 ha]) to 6th-code (10,000 to 40,000 ac [4047 to 16 187 ha]) hydrologic units that were intended to serve as refugia for aquatic organisms, particularly in the short term for at-risk fish populations, and had the greatest potential for restoration, or to provide sources of high-quality water. At the time the NWFP was drafted, Tier 1 key watersheds had strong populations of fish, productive habitat that was in good condition, or high restoration potential. Tier 2 key watersheds provided sources of high-quality water.
- **Watershed analysis**: An analytical process that characterizes the features and processes of watersheds and identifies potential actions for addressing problems and concerns, along with possible management options. It assembles information necessary to determine the ecological characteristics and behavior of the watershed and contribute to the development of options to guide management in the watershed, including adjusting riparian-reserve boundaries.
- **Watershed restoration**: Includes actions deemed necessary to restore degraded ecological processes and habitat. Restoration activities focus on restoring the key ecological processes required to create and maintain favorable environmental conditions for aquatic and riparian-dependent organisms.
- **Standards and guidelines**: These directives impose specific requirements (standards) or recommended approaches (guidelines) for management activities in riparian reserves and key watersheds.

Note that a key philosophical shift occurred in the development of the ACS and NWFP as compared with efforts prior to 1993. The ACS, along with PACFISH (USDA and USDI 1994b) and the riparian component of the Tongass Land and Resource Management Plan (USDA FS 1997), made two substantive changes in how riparian management was formulated (Everest and Reeves 2007). First, they addressed riparian management at the watershed scale (5th- to 6th-code hydrologic units), with specific emphasis on maintaining ecological functions over the long term. Second, they rejected the previous philosophy of trying to define and achieve the absolute minimum set of practices that would meet stated riparian-management goals, and the concept that goals could be met by implementing yet another set of best management practices. The new (at that time) management philosophy under the NWFP represented a paradigm shift in how managers viewed resource coordination. In previous riparian rule-sets, riparian and aquatic technical specialists shouldered the "burden of proof" to demonstrate resource damage from forestry activities and the need for more comprehensive forest-practices rules to meet riparian-management goals. Under the NWFP, the precautionary principle was invoked—the burden of proof shifted (Thomas et al. 2006, USDA and USDI 1994a). Forest managers who wanted to alter the comprehensive default prescriptions for riparian management under the NWFP (described above) to pursue other management goals were required to demonstrate through watershed analysis that changes would not compromise established riparian-management goals.

This chapter focuses on the scientific literature related to the management and conservation of aquatic ecosystems, particularly as it has developed since the 10-year NWFP review (Reeves 2006), with particular emphasis on the area of the NWFP. Among the key issues considered are:

- The ecological, physical, and biological importance of headwater and intermittent streams.
- The contribution of periodic disturbances to the resilience and productivity of aquatic ecosystems.
- The inherent variation of aquatic ecosystems in space and time.
- A better understanding of the variation in where key ecological processes occur within the stream network and the development of new tools to identify these locations.

- An understanding of the variation in the capacity of aquatic ecosystems to provide habitat for various fish species.
- Awareness of climate change and its potential effects.

We provide an update on the status of species listed under the Endangered Species Act (ESA) and the components and the associated monitoring program (Aquatic and Riparian Effectiveness Monitoring Program [AREMP]) of the ACS. We also assess the implications for the potential evolution of the ACS in the next round of forest plans. Reeves (2006) provided a thorough review of the literature in the first 10 years of the ACS, and readers should refer to that publication for a review of the relevant science during that time.

Guiding Questions

Federal land managers submitted many questions that they deemed necessary to consider in the NWFP science synthesis to help with any revisions of forest plans. Because there was substantial overlap among and duplication in the questions, we distilled them into categories represented by the following eight questions to guide our update and assessment:

1. Is the science foundation of the ACS still valid, or does science developed since 1993 suggest potential changes or adjustments that could be made to the ACS?
2. What is the basis of trends observed in the ACS monitoring program, and what are the limitations, uncertainties, and research needs related to monitoring?
3. What is known about the variation in characteristics of unmanaged streams and riparian ecosystems in relation to stream networks across the NWFP area?
4. What has been learned about the effects of riparian vegetation on stream habitat and environments?
5. What effects have human activities had on stream and riparian ecosystems?
6. What is the scientific basis for restoration management in riparian reserves, and how does restoration relate to the ecological goals of the ACS?
7. What is the capacity of federal lands in the NWFP area to contribute water for a suite of economic, recreational, and ecological uses?
8. What are the potential effects of climate change on aquatic ecosystems in the NWFP area, and are these adequately addressed by the ACS?

These eight questions are not answered specifically in sequence because of the overlap among them and the variety of topics they involve. They are, however, answered to the extent possible in different or multiple sections of the chapter, and are addressed in outline in the conclusions.

Key Findings

Status of Species and Population Units Listed Under the Endangered Species Act on Federal Lands in the Northwest Forest Plan Area

In 1993, only the Sacramento winter Chinook salmon (*Oncorhynchus tshawytscha*), and the shortnose sucker (*Chasmistes brevirostris*), and Lost River sucker (*Deltistes luxatus*, both native to the Klamath River system) were listed as threatened or endangered under the ESA in the area covered by the NWFP. Within a few years of the development of the ACS, 23 evolutionarily significant units of Pacific salmon and 3 distinct population segments of bull trout (*Salvelinus confluentus*) were listed under the ESA (table 7-1). There have been three additions since the 10-year review (Reeves 2006): the Oregon Coast coho salmon evolutionarily significant unit (*O. kisutch*), and two other fish species, the Oregon chub (*Oregonichthys crameri*) and the Pacific eulachon (*Thaleichthys pacificus*). No population units of Pacific salmon or bull trout have warranted delisting since the ACS was developed.[2] However, the Oregon chub was delisted in 2015 (USFWS 2015), becoming the first fish to be delisted because of increases in numbers. Habitat on the Willamette National Forest contributed to its recovery.

The developers of the ACS anticipated the ESA listing of distinct population segments of various species of Pacific salmon, evolutionarily significant units, and other fish species. The ACS was not expected to prevent the listing of any species or distinct population segment because factors outside the responsibility and control of federal

[2] http://www.westcoast.fisheries.noaa.gov/publications/status_reviews/salmon_steelhead/2016_status_review.html.

Table 7-1—Evolutionarily significant units (ESUs) of Pacific salmon and trout (*Oncorhynchus* spp.), distinct population segments (DPSs) of bull trout (*Salvelinus confluentus*), and fish and amphibian species listed under the Endangered Species Act that occur in the area covered by the Northwest Forest Plan

Species[a]	ESU/DPS/species	National forests (NFs) and Bureau of Land Management (BLM) districts in which ESU, DPS, or species occur
1. Fish		
Coho salmon	Lower Columbia/ southwest Washington	Gifford Pinchot and Mount Hood NFs
	Oregon coast	Siskiyou, Siuslaw, and Umpqua NFs; Coos Bay, Eugene, Roseburg, and Salem BLM districts
	Southern Oregon/northern California	Klamath, Mendocino, Rogue River–Siskiyou, Shasta-Trinity, and Six Rivers NFs; Arcata, Medford, and Redding BLM districts; Kings Range National Conservation Area (NCA)
	Central California coast	Ukiah BLM district
Chinook salmon	Puget Sound	Gifford Pinchot, Mount Baker–Snoqualmie, and Olympic NFs
	Lower Columbia	Gifford Pinchot and Mount Hood NFs; Salem BLM district
	Upper Columbia	Okanogan-Wenatchee NF
	Upper Willamette	Mount Hood and Willamette NFs; Eugene and Salem BLM districts
	California coastal	Mendocino and Six Rivers NFs; Arcata and Ukiah BLM districts; Kings Range NCA
	Sacramento River winter run	Mendocino and Shasta-Trinity NFs; Mendocino BLM district
	Central Valley spring run	Shasta-Trinity NF; Mendocino and Redding BLM districts
Chum salmon	Hood Canal summer	Olympic NF
	Columbia River	Salem BLM district
Steelhead	Puget Sound	Gifford Pinchot, Mount Baker–Snoqualimie, and Olympic NFs
	Lower Columbia	Gifford Pinchot and Mount Hood NFs; Salem BLM district
	Mid-Columbia	Gifford Pinchot, Mount Hood, and Wenatchee NFs
	Upper Columbia	Okanogan-Wenatchee NF
	Upper Willamette	Willamette NF; Eugene and Salem BLM districts
	Northern California	Mendocino and Six Rivers NFs; Arcata, Mendocino, and Ukiah BLM districts; Kings Range NCA
	Central California coast	Arcata BLM district; Kings Range NCA
	Central Valley, California	Mendocino and Shasta-Trinity NFs Mendocino BLM
Bull trout	Klamath River	Fremont-Winema NF
	Columbia River	Deschutes, Gifford Pinchot, Mount Hood, Okanogan-Wenatchee, and Willamette NFs; Eugene BLM district
	Coastal—Puget Sound	Mount Baker–Snoqualmie and Olympic NFs
Lost River sucker		Fremont-Winema NF
Shortnose sucker		Fremont-Winema NF
Pacific eulachon		Siuslaw and Six Rivers NFs

Table 7-1—Evolutionarily significant units (ESUs) of Pacific salmon and trout (*Oncorhynchus* spp.), distinct population segments (DPSs) of bull trout (*Salvelinus confluentus*), and fish and amphibian species listed under the Endangered Species Act that occur in the area covered by the Northwest Forest Plan (continued)

Species[a]	ESU/DPS/species	National forests (NFs) and Bureau of Land Management (BLM) districts in which ESU, DPS, or species occur
2. Amphibians		
Oregon spotted frog (T)		Deschutes, Fremont-Winema, Gifford Pinchot, Mount Hood, and Willamette NFs; Klamath Falls and Medford BLM districts; Columbia River Gorge National Scenic Area (NSA) (S)
Cascades frog (petitioned)		Deschutes, Gifford Pinchot, Mount Baker–Snoqualmie, Mount Hood, Okanagan-Wenatchee, Olympic, Rogue River–Siskiyou, Umpqua, and Willamette NFs; Medford (S), Roseburg, and Salem BLM districts
Oregon slender salamander (petitioned)		Mount Hood and Willamette NFs; Columbia River Gorge NSA
Cascade torrent salamander (petitioned)		Gifford Pinchot, Mount Hood, and Willamette NFs; Eugene and Salem BLM districts; Columbia River Gorge NSA
Columbia torrent salamander (petitioned)		Siuslaw NF; Salem (S) BLM district
Western pond turtle (petitioned)		Fremont Winema, Mount Hood, Rogue River–Siskiyou, Siuslaw, Umpqua, and Willamette NFs; Columbia River Gorge NSA; Coos Bay, Eugene, Klamath Falls, Medford, Roseburg, and Salem (S) BLM districts

[a] Petitioned = under review for Endangered Species Act listing; T = threatened; S = suspected occurrence.

land managers contribute to the decline and recovery of fish populations and will strongly influence their recovery. These factors include:

- Degradation and loss of freshwater and estuarine habitats on nonfederal lands (McConnaha et al. 2006, NRC 1996).
- Excessive harvest in commercial and recreational fisheries (NRC 1996).
- Migratory impediments, such as dams (McConnaha et al. 2006, NRC 1996).
- Loss of genetic integrity from the effects of hatchery practices and introductions, combined with undesirable interactions (e.g., competition and predation) involving hatchery and naturally produced fish (Araki and Schmid 2010, NRC 1996).

Thus, the ACS was an attempt to develop a strategy to guide management of aquatic ecosystems on federal lands in the NWFP area that would meet potential ESA requirements. The ACS was expected to make significant contributions to the recovery of the ESA-listed fish by increasing the quantity and quality of freshwater habitat for Pacific salmon and protecting and enhancing habitats of other species (FEMAT 1993). Although the condition of habitat in aquatic ecosystems on federal lands appears to have improved at least slightly over the NWFP area, this has not been sufficient to change the status of most listed fish.

The potential of federal lands to contribute to the recovery of listed fish, particularly Pacific salmon, in many parts of the NWFP area is likely more limited than was recognized when the ACS was developed. The primary reason for this difference

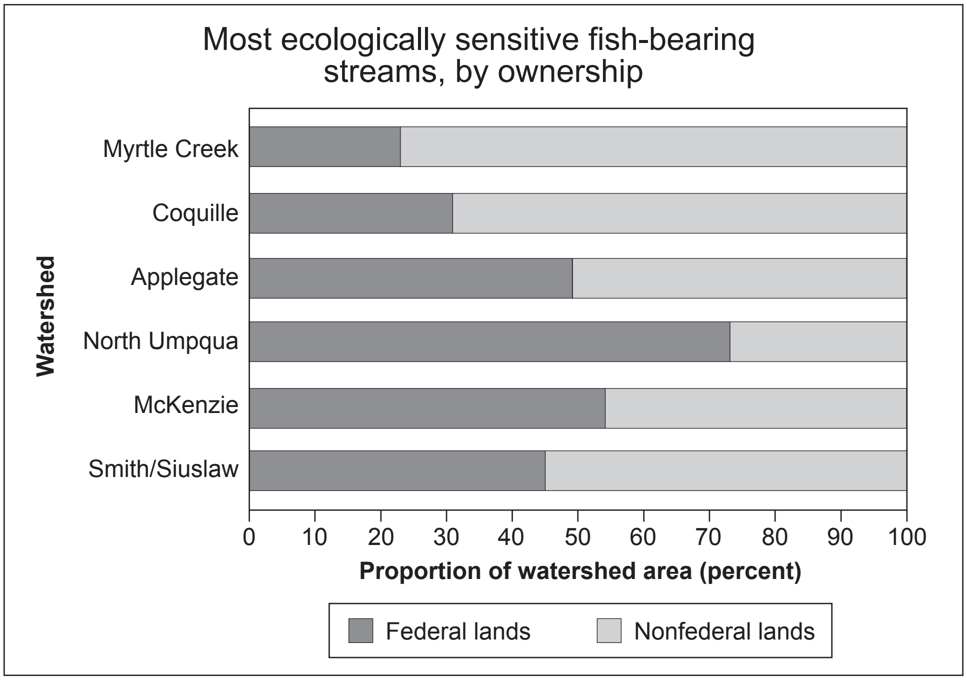

Figure 7-1—Proportion of the length of fish-bearing streams categorized as most ecologically sensitive in the six study watersheds of Reeves et al. (2016a) in western Oregon, by ownership. Ecologically sensitive areas are portions of the stream network that (1) have moderate- to high-quality habitat potential for coho and Chinook salmon and steelhead, (2) have moderate to high potential to warm if the riparian ecosystem is modified, and (3) have moderate to high potential for erosion or debris flows. From Reeves et al. 2016a.

is that, in many situations, federal lands (figs. 7-1 and 7-2) have a limited capacity to provide high-quality habitat for some of the listed fish. Federally managed lands are generally located in the middle to upper portions of watersheds, which tend to have steeper gradients and more confined valleys and floodplains, making them inherently less productive for some fish (Burnett et al. 2007, Lunetta et al. 1997, Reeves et al. 2016a). Federal lands may, however, be important sources of wood, sediment (Reeves et al. 2016a), and water (Brown and Froemke 2010, 2012) for downstream nonfederal lands, and will be important for the potential recovery of most populations. Nevertheless, their contribution to recovery may in many cases be insufficient without parallel contributions from nonfederal land ownerships elsewhere in the basin (Grantham et al. 2017).

The numbers of Pacific salmon and other anadromous fish returning to freshwater in the NWFP area are strongly influenced by ocean conditions, which are highly variable over time. Favorable conditions (cold water) tend to occur in the negative phase of the Pacific Decadal Oscillation (PDO) and the La Niña phase of the El Niño-Southern Oscillation (ENSO), when fish growth is strong and survival is high, resulting in strong returns of adults to freshwater (Mantua et al. 1997). Survival is low and numbers decline during warmer periods, especially during the positive phase of the PDO and the El Niño phase of the ENSO. Winters are cold and wet in the negative PDO–La Niña phase, which also creates more favorable conditions in freshwater (Mantua et al. 1997). A positive PDO–El Niño produces dry, warm winters, reducing streamflows, increasing water temperatures, and increasing the occurrence of fire (see chapter 3). The last extended period of high productivity was from the late 1940s to 1976 (Mantua et al. 1997), with brief periods of favorable conditions in 1984–1988, 1999–2002,[3] and 2010–2011 (Bond et al. 2015). However, beginning in 2013, abnormally warm conditions in the Pacific Ocean ("the Blob") developed because of lower-than-normal heat loss from the ocean to the atmosphere, combined with a relatively weak mixing of the upper ocean layer owing to an usually high atmospheric pressure (Bond et al. 2015). Initial effects were most notable in the North Pacific Ocean off Alaska. Ocean conditions changed noticeably along the NWFP area in 2014 as a result, and fish returns are expected to decline over the next few years.

[3] Mantua, N. 2017. Personal communication. Leader Landscape Ecology Team, National Marine Fisheries Service–Southwest Fisheries Science Center, 8901 LaJolla Shores Drive, Santa Cruz, CA 92037. nate.mantua@noaa.gov.

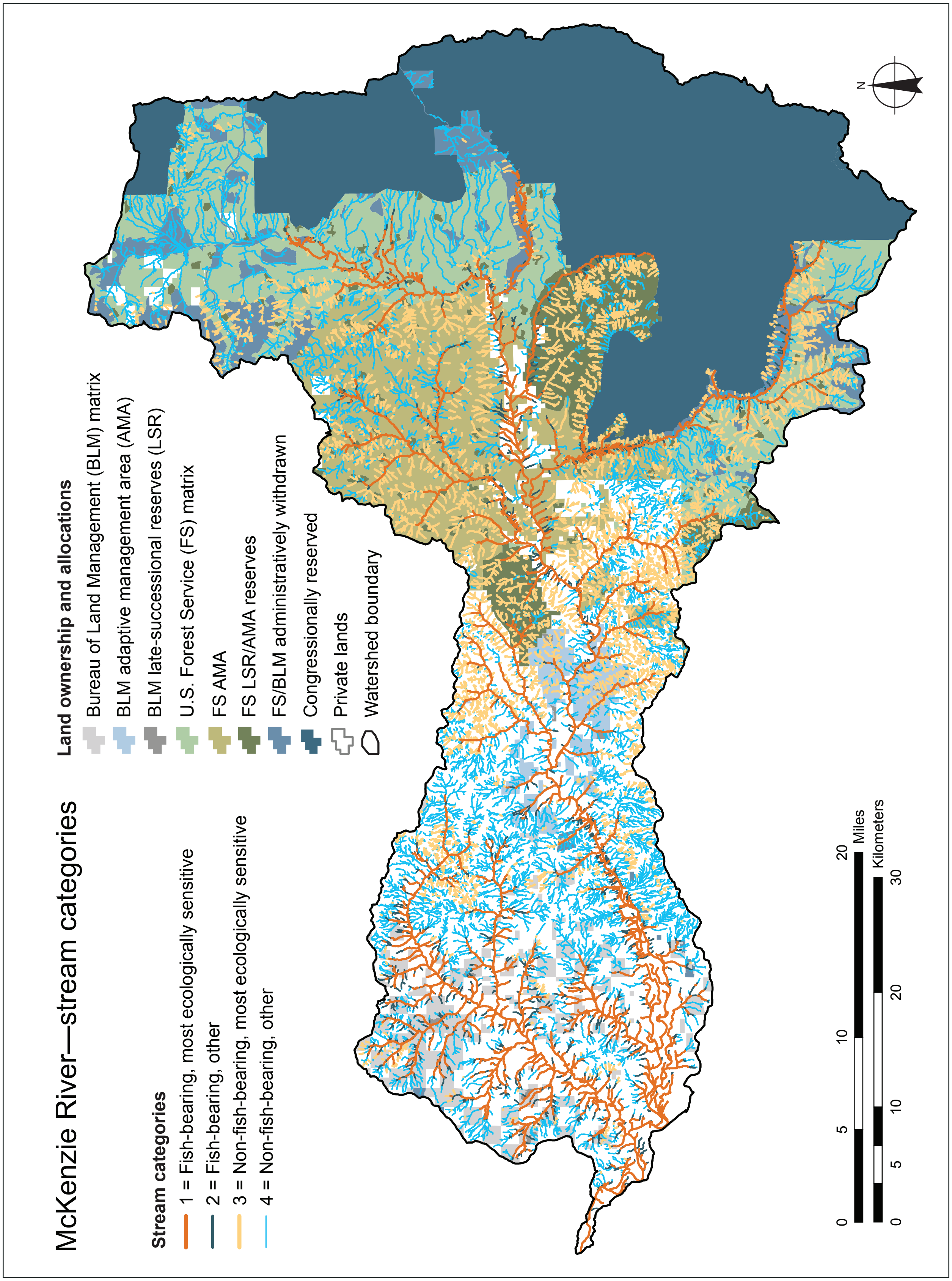

Figure 7-2—Distribution of ecologically sensitive stream reaches on federal and nonfederal lands in the McKenzie River watershed, Oregon. From Reeves et al. 2016a.

We are unable to separate the influence of ocean conditions over the last 10 years from the influence of changes in the condition of freshwater ecosystems on federal lands that may have occurred under the NWFP and ACS. The actual contribution of freshwater habitats to the persistence and recovery of anadromous salmon and trout will be relatively more important when ocean conditions move into a less-productive phase (Lawson 1993). Improvements in the quantity and quality of freshwater habitat resulting from the ACS could result in relatively greater numbers of fish entering the ocean, thus increasing the likelihood of persistence of many populations during periods of low ocean productivity. However, as noted previously, the contribution of federal lands may be more limited than expected because their potential to provide high-quality habitat is less than originally recognized when the ACS was developed.

The status of other aquatic-riparian species in the NWFP area is not as well monitored as that of Pacific salmon. The Oregon spotted frog (*Rana pretiosa*) was listed as threatened under the ESA in 2014. It is a pond-breeding amphibian now restricted to isolated populations that overlap the NWFP area in western Washington and Oregon.[4] Five other aquatic-riparian amphibian and reptile species are petitioned for ESA listing and are under status review: (1) Columbia torrent salamander, *Rhyacotriton kezeri*; (2) Cascade torrent salamander, *R. cascadae*; (3) Cascades frog, *Rana cascadae*; (4) Oregon slender salamander, *Batrachoseps wrighti*; and (5) western pond turtle, *Actinemys marmorata*. The two torrent salamanders are headwater forest species, occurring predominantly in and along the banks of small streams, with significant portions of their ranges on nonfederal lands. Nevertheless, federal riparian reserves contribute habitat for localized populations of Columbia torrent salamanders and more extensive areas for Cascade torrent salamanders. The Oregon slender salamander is found in proximity to down wood on the forest floor in riparian and upland forests, and has associations with older forest conditions (Clayton and Olson 2007). Cascades frogs are pond breeders at higher elevations in the Cascade Range, where they may be affected by multiple stressors (Pope et al. 2014). Similarly, multiple threats appear to affect western pond turtles, which may occur in stream and pond systems in the NWFP area (Rosenberg et al. 2009).

Monitoring—Aquatic and Riparian Effectiveness Monitoring Program

Watershed conditions—

The Aquatic and Riparian Effectiveness Monitoring Program is responsible for monitoring, assessing, and reporting on watershed conditions on lands governed by the NWFP. Although NWFP implementation began in 1994, AREMP implementation was delayed to accommodate the time needed for its design. The scope of the AREMP sampling design includes field-data collection across 250 watersheds in the Plan area, with a rotation of sampling among watersheds conducted each year so that the entire population of watersheds selected for monitoring would be sampled over an 8-year period. In addition, using geographic information system (GIS) and remotely sensed data are used to quantify roads and vegetation in 1,974 watersheds with federal lands in the Plan area and assess the condition of upslope and riparian areas.

Pilot monitoring of watershed conditions began in 2000, and the monitoring plan was finalized in 2003 (Reeves et al. 2004). The first full rotation of watershed visits was conducted in years 2002–2009, assessing initial status, and the second full rotation is scheduled to occur in 2010–2018 where paired assessments of most watersheds were possible owing to watersheds being resampled a second time. Reporting is on a 5-year cycle, in synchrony with NWFP establishment, with the first report covering up to year 10 of Plan implementation (Gallo et al. 2005), the second report covering up to year 15 (Lanigan et al. 2012), and the third to year 20 (Miller et al. 2017). The 20-year report includes assessment of data from the first rotation of watershed visits (2002–2009) and the first 4 years of the second rotation (2010–2013), and hence includes trend assessments based on a subset of sampled watersheds.

Changes in data collection and aggregation procedures, and in application of analytical methods, were anticipated from the onset of the development of AREMP (Hohler et al. 2001). In the late 1990s, our understanding of watershed

[4] http://www.fs.fed.us/r6/sfpnw/issssp/agency-policy/.

ecology and watershed-condition assessment approaches was acknowledged to be limited, and advances in both ecological and statistical disciplines were expected to contribute to further development of AREMP assessments. Indeed, both data sources and analyses have changed over time, with the consequence being that the results from each of the reports are not directly comparable. For example, relative to data sources, some data-collection procedures changed as attribute variability became apparent. Relative to analytical approaches, the 10- and 15-year analyses used decision support models (Reeves et al. 2004, Reynolds et al. 2014) that depended on empirical relations and expert judgment to evaluate data. The 20-year report employed a more statistical focus, with expert opinion and independent analysis of upland, riparian, and in-channel metrics. Additional discussion of adaptive processes through AREMP implementation, including anticipated next steps for research, is presented following the key 20-year findings. Although data analysis and assessment methods changed, each report reanalyzes the entire spatial and temporal dataset available at the time, and is intended to represent the most current understanding of status and trends since the beginning of the Plan.

Key 20-year findings—

The 20-year AREMP report (Miller et al. 2017) examined upslope-riparian and in-channel datasets separately. This segregation acknowledged that the source data differed significantly between these two components. Upslope-riparian data were derived from remote sensing and GIS landscape data covering all NWFP watersheds (watersheds containing more than 5 percent federal ownership, a total of 1,974 watersheds). In contrast, in-channel data were derived from annual field measurements, and therefore were limited to 213 sampled watersheds. Upslope-riparian assessments integrated five data types reflecting watershed processes: sedimentation, wood recruitment, riparian condition and processes, hydrology, and fish passage. In-channel analyses focused on three additional data types, assessed independently: physical-habitat condition, macroinvertebrate assemblages, and water temperature.

Upslope-riparian analyses integrated finer scaled data metrics reflecting indicators of key watershed processes. These processes included (1) stream-sediment delivery from landslides, based on road and vegetation disturbances, in addition to geology and climate attributes; (2) down-wood production and delivery, based on riparian and upland vegetation metrics; (3) riparian condition and associated processes as represented by stream temperature, streambank stability, and species-habitat provision based on riparian vegetation condition and riparian road density; (4) hydrology, focusing on peak flow, based on road and vegetation metrics; and (5) fish passage, based on stream gradient and assessment of barriers (e.g., dams, some road crossings). Using a multicriteria assessment approach, akin to analyses conducted in previous reports, attributes for a watershed were scored to a common 0 to 100 scale, reflecting an index of most-to-least deviation from least human-disturbed conditions.

The 20-year report found little change in the average upslope-riparian condition, from a score of 68 in 1993 to 69 in 2012. However, noticeable shifts were observed in the overall score distribution (fig. 7-3A). In particular, there was a noticeable increase in scores from the low to mid range (15 to 50) to a higher range (60 to 90), whereas the area with the highest scores (>90) decreased slightly. These patterns reflected a signature of federal land use allocations. The mean score in the most protected category of land use allocation (Congressionally reserved lands) decreased (−1), indicating greater disturbance, whereas averages for late-successional reserves and matrix lands increased (+2, +3), indicating less disturbance. The upward shift in the low-range scores is likely attributed to widespread vegetation regrowth and targeted road decommissioning in previously harvested watersheds. In contrast, the decrease in the high-end scores mainly followed the pattern of large fires during the assessment period, many of which occurred in wilderness areas, including the Biscuit Fire in southwest Oregon (2002), the B&B Complex fires in the central Oregon Cascade Range (2003), and numerous fires along the eastern edge of the North Cascade Range in Washington (2006).

It may seem counterintuitive that the most protected lands would show a trend toward more disturbance. Although this trend might be seen as negative because fire results in a loss of vegetation and an increase in riparian-upland disturbance, it is simplistic to consider this an

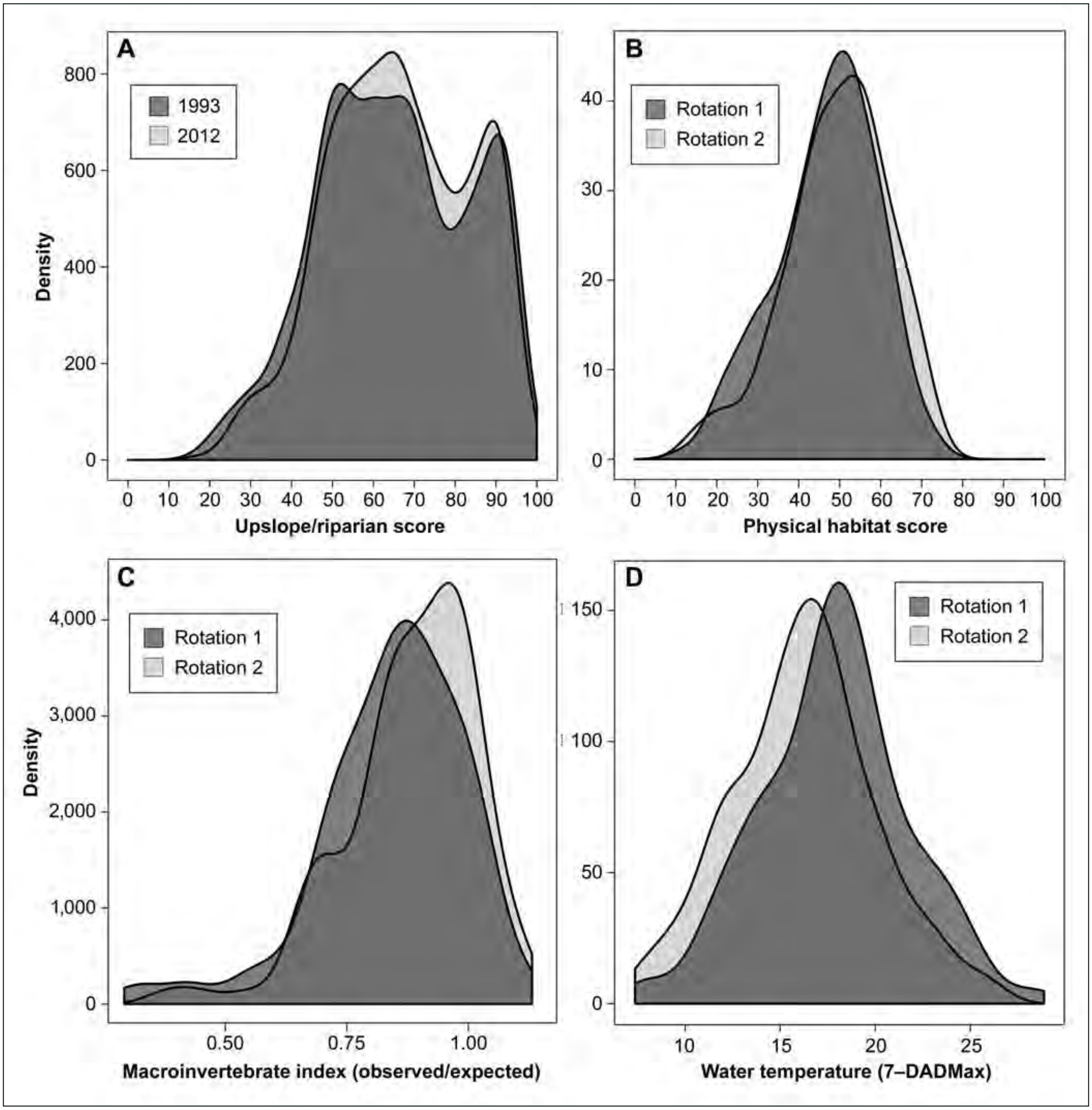

Figure 7-3—Results of the 20-year assessment of watershed conditions by the Aquatic and Riparian Effectiveness Monitoring Program (Miller et al. 2017): (A) upslope-riparian condition, (B) in-channel condition, (C) aquatic macroinvertebrates, and (D) 7-day running average of maximum water temperatures. Source: Miller et al. 2017.

adverse effect at the regional scale of forest ecosystems and their embedded watersheds. Wildfires are an integral component of long-term forest and stream ecosystem dynamics (e.g., Bisson et al. 2003, Franklin et at. 2017, Reeves et al. 1995), with direct benefits to stream habitats and biota resulting from fire (e.g., Flitcroft et al. 2016a) and related natural disturbances such as landslides (wood and sediment delivery to streams) (e.g., Benda and Dunne 1997a, 1997b; May and Gresswell 2003; Reeves et al. 2003). Aquatic-riparian ecosystems are dynamic, being multistate in space and time (Olson et al. 2017a, Penaluna et al. 2016). This recent AREMP finding highlights our nascent understanding of the range of historical aquatic-riparian conditions in the NWFP area and the cadence and extent of natural disturbance events. It also brings to the forefront the role of both passive and active management of these aquatic-riparian systems in the future to maintain and restore the dynamics of aquatic-riparian ecosystems in the region, and the importance of considering whether we need to manage for resilience. In this framework, shaped by land use allocations and trends detected therein, it is possible that development of more nuanced evaluation methods is needed to assess variation in aquatic-riparian conditions. The problem also becomes conceptually challenging, given a known shifted baseline from past anthropogenic disturbances, including the effects of fire suppression, as well as from the need to develop projections of climate change, climate extremes, and related disturbances from fire and landslides.

In-channel watershed-condition assessments conducted for the 20-year report were the first such assessments to have enough field-site revisits (about half the sample) to estimate trends. Instream conditions in subsampled watersheds in the Plan area were assessed by evaluating three separate elements: physical habitat, macroinvertebrates, and water temperature. First, for physical habitat, a composite index score (on a 0 to 100 scale, relative to unmanaged reference conditions; see below for more discussion on this topic) based on expert judgment was derived from substrate (percentage of fine substrates at a stream-reach scale), pool-tail fine substrates, and the frequency of medium- and large-size down wood. A small but statistically significant increasing trend (from 46 to 49) in overall physical habitat condition was detected when measured both on a yearly basis with all data and from the subset of watersheds that had been revisited (fig. 7-3B). Individual components of the physical habitat varied: reach-scale fine substrate showed an increasing trend in occurrence, whereas instream wood and pool-tail fine substrates showed no significant changes.

Second, an index was also derived for macroinvertebrates (fig. 7-3C), which were assessed at the site level, then grouped into taxonomic classes and compiled into a score representing the ratio of expected species based on reference sites (E = 1) (see below) to occurrence of no expected species (O = 0). Site-scale scores were aggregated into watershed scores. A positive change in the mean score was detected for macroinvertebrate diversity, suggesting a shift toward a species composition reflective of expected reference conditions.

Third, water temperature was evaluated using an integrated model that assessed 7-day maximum temperatures collected at low points in watersheds from June to September. Water temperature showed a decreasing trend, although temperature averages were still higher than federal and state standards for salmonid habitat (fig. 7-3D). Interestingly, water temperatures in all land use categories decreased, but temperatures in the most protected category showed the smallest decline, perhaps reflecting the upslope findings, which showed a signature of disturbance owing to vegetation loss in reserves.

Reflection on adaptive processes through AREMP implementation—
Here, we outline primary changes and challenges in AREMP monitoring over the past 20 years, many of which are ongoing research-emphasis areas. These topics are common, foundational aspects of many aquatic-riparian monitoring programs, representing a larger coalition of scientists and managers addressing similar themes over diverse landscapes. The science of watershed-scale ecology has followed the development of the discipline of landscape ecology, and the challenges cited here are representative of a parallel course of design and analytical adaptive processes occurring in terrestrial forest ecosystems.

Overall, NWFP monitoring, including the AREMP (Hohler et al. 2001), was framed as an adaptive-management cycle (Mulder et al. 1999). For AREMP, four primary changes and challenges over the course of the first 20 years of the NWFP have included (1) redefining the overarching objectives of watershed-condition monitoring, shifting from a salmonid habitat focus to one that was more representative of the environmental conditions of entire watersheds, while retaining selected key elements of salmonid habitats; (2) refining the indicators used in data collection and analysis; (3) reconsidering approaches to using benchmarks or reference conditions for assessment; and (4) modifying data integration methods. These four topics are discussed further below, with comparisons to other aquatic-monitoring programs for a broader perspective.

Defining objectives—

The first step in designing the NWFP monitoring modules was to define the goals and objectives of monitoring (Mulder et al. 1999). The NWFP defined the central question for aquatic ecosystems as, "Is the ecological health of the aquatic ecosystems recovering or sufficiently maintained to support stable and well-distributed populations of fish species and stocks?" (USDA and USDI 1994a, E-7). The primary fish taxa with status of concern in the NWFP area were native salmonids, hence a taxonomic focus was present from the origin of the Plan. Although particular species (owls, murrelets, salmonids) have been a principle interest of NWFP monitoring programs, concepts developed in the monitoring plan also stated that "Because of the current wide (and justified) interest in all components of biological diversity, however, the species-centric approach is no longer sufficient" (Mulder 1999, p. 29). In this vein, the language of the ACS objectives (USDA and USDI 1994a) had also included aquatic-riparian habitat conditions and species, but with a focus on multiple processes that were known to be tied to development of salmonid habitat conditions. At the time of the NWFP, emerging science on the role of disturbance in renewal of aquatic habitats also suggested a change in focus from the assessment of narrowly specified, in-channel habitat elements (e.g., a certain number of pieces of large wood per stream length) toward the ecosystem processes that form and maintain habitats (USDA and USDI 1994a). AREMP was perhaps the most ambitious of the monitoring modules in this regard, calling for monitoring a broad set of conditions in the upslope, riparian, and in-channel portions of watersheds that related to ecological processes tied to fish habitat, and evaluating these in comparison to broad distributions of conditions rather than solely on a watershed-by-watershed basis (Reeves et al. 2004).

The ACS was originally envisioned as a "coarse-filter" conservation effort (Hunter 2005, Noss 1987) (see additional discussion in chapter 12). The focus of the ACS was on restoring and maintaining ecological processes that created and maintained aquatic ecosystems for a suite of organisms, primarily ESA-listed fish, and for clean water and other ecosystem services (USDA and USDI 1994a). AREMP was, therefore, initially directed at the habitat of native salmonids, a primary responsibility of federal land managers and regulators in the NWFP area (Reeves et al. 2004). Habitat conditions for native salmonid fishes were initially used as metrics for watershed condition trend assessment, owing to their sensitivity to changes in several habitat features (e.g., water temperature, sediment, down wood). As with other coarse-filter assessments that use biotic indicators such as umbrella or flagship species (e.g., Raphael and Molina 2007), it was assumed that other aquatic and riparian-dependent organisms would benefit if watershed conditions for salmonids improved. Hence, salmonids were recognized as a focal species group, assuming that if their habitats were sustained or improved in condition over time, it would infer sustainability or improvement of the greater community of biota and the broader watershed-scale ecosystem functions and processes upon which they rely.

Development of aquatic monitoring programs requires a clear articulation of which biota and associated functional characteristics of habitats and ecosystems are being considered, and how they are likely to be altered as a result of the actions of interest (Carlisle et al. 2008, Palmer et al. 2005, Pont et al. 2006). Such species-based approaches may not fully account for the variation in species abundance or community composition present, given the spatiotemporal heterogeneity in ecosystem conditions generated by natural disturbances. Further, this natural variability in species and environmental conditions may make it difficult to identify the effects of anthropogenic disturbances and recovery from those disturbances, and thus make it difficult to assess and understand the ecological consequences of detected changes (Frissell et al. 2001). However, explicitly identifying the organism(s) of interest is essential for understanding what the monitoring results mean for those species and the fauna the species represents (Wohl 2016).

The 20-year AREMP analysis shifted from the emphasis of the 10- and 15-year analyses on evaluating habitat for salmonids to characterizing more general environmental conditions. Miller et al. (2017) used the 5^{th} and 95^{th} percentile values from a suite of physical attributes in reference sites in systems with the least human-caused disturbance (see later discussion) to determine the favorability of conditions for biota in monitored watersheds. Based on expert judgment, the 5^{th} percentile was considered the most favorable for some attributes (e.g., pool-tail fines, reach fines) and the 95^{th} for others (wood). However, Miller et al. (2017) did not identify which organisms these conditions were supposed to favor, making it difficult to understand the ecological validity of these values and the consequences of any changes detected.

This switch highlights a continuing scientific debate in the monitoring and assessment community on the merits of focusing assessments on particular flagship or umbrella taxa rather than on more general environmental processes and conditions. On the one hand, umbrella species serve as meaningful "endpoints" (Suter 2001) or "final ecosystem services" (Blahna et al. 2017) that provide relevance to monitoring results. On the other hand, in aquatic-riparian ecosystems, salmonid distributions do not reach headwater streams, which make up most of the stream length in NWFP watersheds (Gomi et al. 2002). Salmonid habitat in larger streams may not be representative of the condition of the entire watershed, unless solid ties to upstream and upslope conditions can be made. Further, the adequacy of salmonids as umbrella species has not been formally assessed (Murphy et al. 2010, Simberloff 1998), and there are questions about whether one species can be an indicator of the condition of other species (Carlisle et al. 2008). Although these two objectives for watershed-condition assessments (salmonid habitat versus watershed environmental conditions) are closely related, differences in emphasis have a ripple effect that plays out through the monitoring and assessment process, and affects how results might be interpreted relative to goals of maintaining and restoring conditions.

The original AREMP design document laid out a conceptual model that considered the interactions between upslope, riparian, and in-channel processes, all within a variable landscape (e.g., precipitation, geology) (Reeves et al. 2004). No formal assessment of relationships between upslope/riparian/in-channel indicators in the AREMP conceptual model has been published, but a number of studies are relevant to pieces of this framework. Burnett et al. (2007) developed a relative ranking, Intrinsic Potential (IP) ranging between 0 (poor) and 1 (excellent) to determine the geomorphic potential of a reach to provide habitat for coho salmon in larger streams. This work was followed by data-driven watershed-scale models of several habitat attributes important for salmon that tied upland-riparian conditions to instream habitats, including models of debris-flow-prone areas (delivering sediment to streams: Benda and Dunne 1997a, 1997b; Burnett and Miller 2007); wood recruitment (e.g., Reeves et al. 2004); and thermal loading (as a proxy to represent stream temperatures; see Reeves et al. 2016a). Several of these studies also contributed to a better understanding of instream processes connecting lower order headwater streams to higher order streams occupied by salmonids. Syntheses of these multifaceted watershed-process models have contributed to a better understanding of the multistate nature of aquatic-riparian ecosystems (Olson et al. 2017a, Penaluna et al. 2016, Reeves and Spies 2017), and integration of several of these watershed-integration models have been used to evaluate potential management options (Reeves et al. 2016a). Full incorporation of these concepts into watershed-condition assessments has been indirect to date, for example, by implicitly supporting approaches to assess upland-riparian areas and full in-channel networks from headwaters downstream. This topic deserves continued attention as AREMP procedures continue to develop, and can potentially inform the overarching objectives of the program.

Other aquatic-monitoring programs have incorporated upslope/riparian/in-channel relationships into their conceptual models, albeit in quite different ways. The National Rivers and Stream Survey (NRSA) (USEPA 2016) related four chemical and four physical habitat stressors to multimetric macroinvertebrate and fish indices using a concept of relative risk: the likelihood of finding poor biological conditions in a river or stream when stressor concentrations are high relative to the likelihood when they are low. Indirectly, these stressors, as well as three land-use metrics (urban land cover, agricultural row-crop land cover, and dam influence), were used in selecting the reference sites used to evaluate the stressor and response metrics.

PACFISH/INFISH Biological Opinion Monitoring Program (PIBO) approaches have not directly assessed upslope indicators, but have selected a set of physical-habitat indicators based on sensitivity to land-use management intensity, using road density as a surrogate (Al-Chokhachy et al. 2010). A recent assessment of the PIBO program conceptual model by Irvine et al. (2015) examined correlations between upslope (grazing, road density, percentage forested), in-channel habitat (fine sediment, temperature), and a macroinvertebrate observed/expected index. Although they found weak to no support for causal pathways related to effects of anthropogenic drivers on biological condition, they surmised that the conceptual model was sound, and the weak correlations were due to imprecision in the measurement of drivers (grazing, roads); stressors (sediment, measured in pool tails); and responses (macroinvertebrates, measured in riffles). They cited the more general issue that regional trend monitoring is not optimized for detecting causal mechanisms. A related and broader concern is that such surveys may underestimate infrequent but high-severity events (Suter 2001). In contrast, it was notable in the AREMP 20-year report that the signatures of wildfire and road decommissioning, relatively low-frequency events, were detectable in the upslope-riparian assessment because it included a full census of watersheds, rather than a limited sample. Overall, scientific work in the past 20 years has continued to support the dynamic, disturbance-based ecology of aquatic-riparian ecosystems (see "Natural Variability" section later in this chapter). Although the AREMP conceptual model has not changed, there have been numerous refinements to the indicators used, as well as to their combination and interpretation.

Selection of indicators—

***In-channel biotic indicators*—**In-channel biotic metrics have proven to be particularly challenging to monitor. Although fish populations are of principal concern to managers because of regulatory requirements and their potential as umbrella or flagship species, fish-data collection was dropped from AREMP protocol in 2007 because most in-channel watershed sample sites were above salmonid habitat, and the collection of meaningful salmonid-habitat data would have required a separate and intensive effort. Similarly, streambank amphibians were dropped because of detectability issues. Variability in detection spatially within watersheds as well as temporally within the year made streambank amphibian monitoring challenging: species that were present at a site early in a season could be missed if the site is not sampled until later in the season. Further, terrestrial salamanders are fossorial (live largely subsurface) and may not have been detected even if present (e.g., Hyde and Simons 2001).

Macroinvertebrates are the only remaining in-channel biotic indicator collected by AREMP. Macroinvertebrates have come to play a central role in many aquatic-monitoring programs because of their presumed responsiveness to local environmental conditions and ease of collection (PIBO [Al-Chokhachy et al. 2010], Oregon Department of Environmental Quality [Hubler 2009], NRSA [USEPA 2016]). This commonality creates the potential for data sharing between monitoring programs, assuming sufficient similarity in the sampling methods used. However, differing results for the fish and macroinvertebrate indices in a recent national assessment emphasized the danger of relying on one taxonomic group to represent the potential responses or condition of other groups (USEPA 2016). Understanding how to reliably collect and incorporate data from taxa other than macroinvertebrates is a challenge for ongoing research; new multispecies environmental DNA methods are promising and are undergoing trial now (see app. 1).

***In-channel abiotic indicators*—**Measurement reliability has also been a challenge with abiotic in-channel indicators. Based on quality-control sampling, AREMP dropped the evaluation of pool frequency, depth, and median particle size from the latest 20-year assessment. Measurements of these parameters are still being collected, owing to the perceived importance of these indicators for assessment of habitat conditions, and the AREMP program is actively investigating ways to make current collections of these data more reliable for use in future analyses. Remaining abiotic indicators used in the latest report were pool-tail fine substrates, reach-scale fine substrates, down wood, and water temperature (Miller et al. 2017).

***Upslope-riparian indicators*—**The basic indicators used in the upslope-riparian portion of the AREMP assessments have changed little, but their evaluation has become more context sensitive. In the latest report, their combination was reorganized into an index with a more process-based structure. The new process-based model structure includes five processes that aligned it more directly with the conceptual model in the original AREMP plan: sediment and wood delivery, riparian shading, hydrology, and habitat connectivity. All these indicators are based primarily on road and vegetation data, the two major land-use metrics that can be traced backward in time to assess trends since the beginning of the NWFP. This estimation of historical data is a challenge that appears not to have been attempted by other broad-scale programs (Gordon 2014). Road and vegetation-management effects on aquatic systems continue to be active areas of research.

Of particular relevance to the AREMP assessment is work identifying the differential effects of sediment delivery from roads, based on landscape position (Al-Chokhachy et al. 2016, Black et al. 2012) and the incorporation of potential fish-habitat considerations into the measure of aquatic connectivity (Chelgren and Dunham 2015). Advances in satellite data and their classification have considerably expanded the available vegetation metrics and the ability to track these through yearly time steps (Kennedy et al. 2010, Ohmann et al. 2011). These capabilities should be examined in light of the disturbance-ecology paradigm discussed throughout this chapter.

Examining the freshwater assessment literature more generally, Kuehne et al. (2017) found a shift in measures used from field-based responses to landscape-stressor metrics. The use of upslope indicators is attractive to managers because this is where the most extensive management activities currently occur (vegetation, roads), and also because these measures are more easily collected via remote sensing and existing GIS data, and do not require more labor-intensive field surveys. Measuring and understanding both upslope and in-channel processes, and the relationships between them, is critical. Taking into account the difficulties encountered by other parallel aquatic-monitoring programs, more formal testing of the AREMP conceptual model is warranted.

Benchmarks for assessment—
In the data-assessment step of analysis, indicators are typically compared against some benchmark (alternatively referred to as standards, thresholds, or evaluation criteria) to come up with a measure of watershed or aquatic-habitat condition. This has proven to be one of the greatest challenges, particularly given the expanding recognition of the reliance of aquatic habitats on dynamic processes of disturbance and renewal. Such monitoring faces the fundamental challenge of using static measurements (with limited temporal frequency) to measure dynamic processes.

Stoddard et al. (2006) described a number of common approaches to choosing benchmarks: reference conditions, best professional judgment, interpreting historical condition, extrapolating from empirical models, and ambient distributions. The first two AREMP reports (10 and 15 year) relied on empirical models and expert judgment to set evaluation criteria (Gordon and Gallo 2011). To accommodate environmental heterogeneity, separate thresholds were solicited for seven aquatic provinces identified in the NWFP area. To accommodate environmental heterogeneity, separate thresholds were solicited for seven aquatic provinces identified in the NWFP area. In most provinces, experts were unwilling to commit to standards in higher gradient streams for some attributes (floodplain connectivity, pool frequency, pool-tail fines, and median substrate diameter), so they were not included in the evaluation of these sites.

The most recent (20-year) AREMP report switched to using reference conditions to set criteria for in-channel conditions and upslope vegetation. Reference criteria were chosen using a nearest-neighbor approach, which matched a site to the five to seven nearest reference sites, based not on geographic distance but rather on similarity of largely invariant site characteristics (e.g., gradient, geology) (Bates Prins and Smith 2007). This approach was more empirically based than previous assessments; it set standards for higher gradient stream reaches and established a consistent method across the whole NWFP area. The use of the reference-condition approach in a monitoring program has important associated assumptions. The selection of reference sites should match the distribution of states in the ecosystem, reflecting the spatial and temporal dynamics at play. These values or thresholds are often either a direct judgment call or a chosen percentile of the overall disturbance distribution. In more highly disturbed ecoregions, few sites may qualify, as was the case with the Oregon/Washington Coast Range and the Franciscan provinces in the latest AREMP assessment (Miller et al. 2016). The nearest-neighbor approach did not rely on provinces/ecoregions, but rather incorporated environmental variability more directly through site characteristics.

Another key consideration in development of reference distributions is including the entire natural range of conditions that an ecosystem can experience (Lisle et al. 2007, NRC 2000, Stoddard et al. 2006). Relative to the ACS and development of AREMP as originally conceived (Reeves et al. 2004), it was generally assumed that there is a given condition or limited set of conditions that supports aquatic organisms, primarily fish and macroinvertebrates (e.g., Karr and Chu 1998). The panel of scientists and managers who initially framed the ACS assumed that favorable conditions for fish were constrained to areas with cold water and structural heterogeneity provided by physical habitat components such as large down wood and coarse substrates—conditions often associated with old-growth forests—thus, these conditions and the associated old-growth forested riparian habitats were assumed to be most suitable for fish.

Recent studies, however, have demonstrated that native salmonids (Howell 2006, Rieman and Isaak 2010, Sestrich et al. 2011) and aquatic invertebrates (Minshall et al. 1989) are capable of adapting to and being productive in a wide range of conditions, including those following major disturbances such as wildfire that affect stream conditions. Flitcroft et al. (2016a) found that although conditions for one life-history stage of salmonids may be unfavorable, other life-history stages may find the same conditions suitable, and populations may respond positively. Native salmonids may also change life-history tactics, such as by reducing age or size at maturity (Rosenberger et al. 2015). Evolving in naturally dynamic landscapes with infrequent to frequent fire (see chapter 3) and occasional landslides, these species appear to be resilient to a broad range of disturbances and environments that occurred under the natural range of variability. It is important for monitoring programs to incorporate this new perspective into the development of benchmarks and interpretation of results to better reflect the responses of aquatic organisms to both management and natural disturbances. The range of natural conditions likely spans recently disturbed sites as well as areas that have been undisturbed for hundreds of years or longer.

However, understanding the natural range of variability for an ecosystem is often difficult, owing to the extent and magnitude of anthropogenic effects (Miller et al. 2016, NRC 2000, Steel et al. 2016, Stoddard et al. 2006). This may especially be the case in dry-forest regions in the NWFP area where fire exclusion has altered forest and riparian plant composition and structure (see chapter 3); in areas where invasive species are now a dominant component of communities (app. 1); or where the signature of past human activities (Steel et al. 2016) and pervasive "press" disturbances (Yount and Neimi 1990) such as timber harvest have influenced the entire landscape so that current conditions, which may be a departure from the historical range, may now be seen as the norm (Pinnegar and Engelhard 2008), though they may have been rare or unknown in the past. Even areas that may appear to lack any sign of current or historical land use—areas with no discernable sign of recent human-caused disturbances—may no longer be considered pristine (see chapter 12). The Pacific Northwest moist coniferous forest region has recently been described as a "human-forest ecosystem," because people are now a foundational element of the system (Olson and Van Horne 2017).

Because pristine areas may no longer exist in many, if not most, ecosystems, the reference-condition approach sometimes has been modified to use "least-disturbed conditions" as a reference (Stoddard et al. 2006). But depending on how this approach is applied, it may not include the full range of potential ecosystem conditions, especially in naturally dynamic landscapes as described above, where disturbance had been excluded. Worse, in today's human-influenced forest landscape, there may be no locations that fit even the "least-disturbed" condition (see discussion in chapter 3).

Excluding the natural range of variability in the reference population influences the assessment of current conditions (NRC 2000). This issue can be illustrated by using the down wood data from Reeves et al. (1995), who examined three watersheds in the Oregon Coast Range that differed in the lengths of time since the last large wildfire (see details in "Attribute Integration Approaches" below). If just the values from the watersheds that were at an intermediate time point and the longest time point from disturbance were included as being in the population of reference conditions, wood values would vary from 12 to 24 pieces of wood per 100 m of stream. Twelve wood pieces per 328 ft (100 m) might be considered an extremely low value, and 24 a high value. However, if the most recently disturbed system from this dataset were included in the pool of reference conditions, the lower bound would be near 6 pieces of wood/328 ft, and the score for 12 pieces of wood/328 ft would be much higher, relatively speaking. Clearly, use of reference conditions to assess monitoring trends and their ecological relevance can be problematic if a wide range of variation is not included. Articulating how reference conditions were determined and the range of conditions they represent is essential to understanding the context for comparison, whether least-disturbed conditions, old-growth conditions, or professional judgment are used.

The relatively small number of matched sites in the reference pool used for the 20-year report is a potential drawback. Further, only one site-matching metric, quadratic mean diameter of conifers, reflected forest-ecosystem characteristics; this metric is only a limited surrogate for seral stage and does not reflect forest type, both of which may influence stream characteristics. Concerning the reference-site approach more generally, there is also some question as to whether reference sites relatively free of human disturbance still exist, given widespread fire suppression and now climate change (see chapter 3) (Herlihy et al. 2008, Stoddard et al. 2006). To the extent that these concerns are true, it likely creates uncertainty concerning monitoring results that needs to be explored.

Other recent large-scale assessments have also used a variation of the reference-condition approach, but none have incorporated much detail on surrounding vegetation conditions. The NRSA (USEPA 2016) incorporated environmental variability more generally into their reference sites by selecting a different set of sites for each ecoregion. Thus, their reference distribution includes a larger number of sites than in the AREMP analysis, but the NRSA authors recognized that this approach might not account for fine-scale variability within ecoregions and made direct comparison of results between ecoregions problematic (Herlihy et al. 2008, USEPA 2016). They incorporated some finer scale measures of environmental variability (e.g., elevation) by including them as covariates in the multiple linear regression (MLR) equations that defined reference expectations for each indicator. The PIBO analysis incorporated environmental variability by using the MLR approach, but the only covariate related to forest condition was the percentage of the 295-ft (90-m) stream buffer in forested condition (Al-Chokhachy et al. 2010).

The final step in applying the reference-condition approach involves choosing evaluation thresholds from the distribution of reference values and comparing site values to these thresholds. AREMP and PIBO selected the $5^{th}/95^{th}$ percentiles of their reference distributions to define normalized scores, assuming that more extreme values might be outliers that could skew the scoring process. They then reported these values directly, so that scores approximated the percentile in the reference distribution. In contrast, NRSA chose to place all results into three classes (good/fair/poor) based on the less than $5^{th}/5^{th}$–25^{th}/greater than 25^{th} percentiles of reference. The reference-condition approach may appear to be more empirical, but it still relies on professional judgment to set evaluation thresholds.

To promote success in ACS implementation, it is important to know how gains in watershed condition are measured. Because the natural range of variability occurring in a system over large spatial and long temporal scales occurs across a multidimensional continuum, it can be difficult, if not impossible, to incorporate into assessments (see chapter 3). One potentially useful approach is the use of "state and transition" models (e.g, Wondzell et al. 2007, 2012). Although such models can be difficult to validate, they can still be useful. It can be helpful to view aquatic ecosystems as multistate systems resulting from a variety of natural disturbances, as well as exogenous or anthropogenic processes that can alter habitat conditions, biota, and ecological processes (Penaluna et al. 2016). Having the full range of potential variability classified into discrete states provides a way to begin enumerating the ways in which variation is arrayed over large spatial scales and how it changes over long temporal scales. We suggest further exploration of reference conditions and their potential utility for analytical approaches, including consideration of how to use them in concert with state-transition models such as those developed by Wondzell et al. (2007, 2012).

The concept of the reference condition remains important in land management—not because it is a goal of management agencies to restore systems to some previous reference condition, but rather because knowledge of the historical range of variability can help inform choices about desired future conditions and, thereby, help determine management and restoration goals (chapters 3 and 12). Theoretically, departure from the reference condition would provide a relative measure to evaluate watershed conditions for managers who seek to maintain or restore ecosystem and species diversity (Nonaka et al. 2007, Safford et al. 2012). The multistate conceptual approach (Penaluna et al. 2016, Reeves et al. 1995) clearly shows that "reference conditions" are, in fact, a distribution of conditions from watersheds in various ecological states, similar to successional states in terrestrial systems (see discussion later in this chapter). As such, "departure from the reference condition" is no longer just a watershed-scale question, but rather a regional-scale problem that considers the distribution of conditions across multiple watersheds.

Equilibrium versus spatially and temporally dynamic ecosystem concepts and the choice of benchmark reference conditions will become even greater challenges in future assessments given the increasing influence of climate change. There is growing concern about the extent to which ecosystems in the NWFP area, and elsewhere, have been affected by climate change and altered disturbance regimes, such as fire exclusion (Hessburg et al. 2005, Luce et al. 2012; also see chapter 3). We are likely seeing, or will soon see, the development of ecosystems that are different from the present and at least the near past (Hobbs et al. 2009, Luce et al. 2012) (fig. 7-4). Some have called this a new geological epoch—the "Anthropocene" (see chapter 12). The conditions

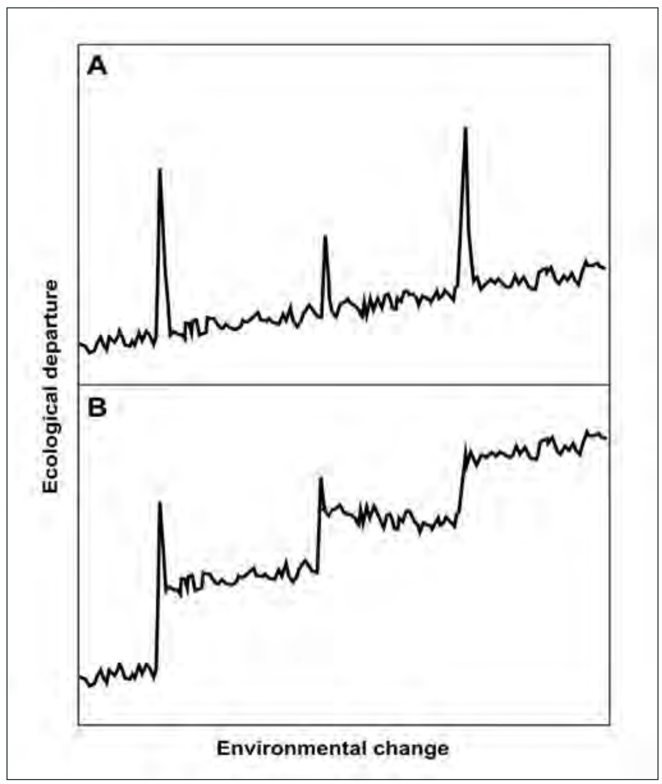

Figure 7-4—Conceptual roles for disturbance in a changing climate: (A) Disturbance could continue to operate much as it always has, with unique disturbance/recovery patterns, or (B) it could become the catalyst that forces ecosystems to shift rapidly and via alternate and uncertain pathways in response to climate. Source: Luce et al. 2012.

that result from these altered ecosystem trajectories could be very different from those that would be found in unaffected systems of the past or present, and they may not necessarily meet social or legal expectations (Luce et al. 2012). Using benchmarks based on our understanding of aquatic ecosystems today may also affect assessments of ecological consequences from natural and anthropogenic factors. We rephrase our statement from near the top of this section: the potential implications of these changes merit a primary research focus, but in the meantime it will be important that monitoring programs, whether they use reference conditions, least-disturbed conditions (e.g., Miller et al. 2017), decision-support models (e.g., Reeves et al. 2004), or other approaches, recognize and acknowledge these potential concerns in the process of analysis and the interpretation and application of results. (The topic of reference conditions is discussed in different contexts later in this report, including an expanded treatment of approaches for riparian restoration.)

Attribute integration approaches—

Integration of metrics—Watersheds and streams operate as integrated systems, and no one indicator is likely to accurately characterize their condition. A significant scientific challenge remains in how to reflect this integration in their assessment. Early efforts and ongoing regulatory guidance for assessing salmonid habitat look at a number of indicators individually, without an explicit procedure for integration (NMFS 1996, USDA FS et al. 2004). NRSA, the largest national assessment, also primarily reports on indicators separately (fish, macroinvertebrates, chemical and physical stressors), although many of their indicators are themselves composite metrics. They incorporate limited integration through their measure of relative risk, which looks at the likelihood of finding poor biological conditions in a river or stream when stressor concentrations are high, relative to the likelihood when they are low (USEPA 2016).

The AREMP monitoring plan was a pioneering attempt to integrate indicators into a composite watershed-condition index (Reeves et al. 2004). In practice, the extent of integration has declined in each of AREMP's reports. The 10-year AREMP report integrated all upslope and in-channel variables into a single score for each watershed (Gallo et al. 2005). Trend was calculated only for the upslope portion; repeated measurements of sufficient in-channel sites were not available until the 20-year report. The 15-year assessment separated upslope and in-channel metrics for two reasons. First, the upslope data (GIS and remote sensing) covered the whole region, so there was no need to restrict that analysis to the in-channel subsample. Second, little correlation was found between the upslope and in-channel results, so it was believed that these outputs offered fundamentally different types of information. In addition, the mixing of stressors and responses has been criticized in other watershed indices (Schultz 2001).

The 20-year report maintained this upslope/in-channel split, and also reported the in-channel elements of physical habitat, macroinvertebrates, and temperature separately. Temperature was split off because it was collected under a different sample design, only at the lowest point in the watershed rather than at each site. Macroinvertebrates were separated from physical habitat because they are often considered a qualitatively different type of indicator: physical habitat as a condition or stressor, and macroinvertebrates as a response. It was also believed that reporting these indicators separately would better identify problems by not obscuring high and low values in an aggregated average. Although macroinvertebrate, physical habitat, and temperature data were not integrated in the 20-year report, additional analyses may be useful to further assess a combined metric, especially as novel techniques emerge that can address issues of spatial autocorrelation along linear stream networks (Peterson et al. 2013, Ver Hoef et al. 2014).

Other monitoring programs have built condition indices by combining indicators into a more integrated value. PIBO averages its physical channel attributes into a single index score, but maintains macroinvertebrates separately (Archer and Ojala 2016). The state of Oregon combines a number of chemical metrics into an overall water-quality index but also reports macroinvertebrates separately (Hubler 2009). The national Forest Service watershed condition class combines upslope, riparian, and in-channel biotic and abiotic indicators into one overall watershed-condition score (Potyondy and Geier 2011).

Having the metrics for watershed condition assessed independently begs the question as to how to interpret overall condition relative to findings for instream habitat, stream temperature, and upland/riparian condition. Miller et al. (2017) acknowledged that reliance on a single biological metric can lead to erroneous interpretation of the biological condition of a watershed (e.g., Barbour et al. 1999), and suggested that the findings of the four separate stream metrics can be used as multiple lines of evidence to look at watershed-condition trends. So, when concordance among the measures differs within a given watershed, for example, if one or two parameters show an improving trend while the third does not change or declines, one can better understand which parameter may signal a potential issue and spur additional investigation. (See additional discussion of this issue later in this section.)

Interpreting long-term changes in a single metric can be complicated. For example, Miller et al. (2017) reported a broad-scale change in the distribution of water temperatures toward lower temperatures and an increase in watersheds with improving aquatic macroinvertebrate assemblages across the NWFP area. Nonetheless, 55 percent of waterbodies monitored by AREMP in Oregon still exceed state water-quality standards for these parameters (ODEQ 2012). One potential reason for this apparent lack of agreement between Miller et al. (2017) and the increase in miles of water-quality-impaired streams in Oregon (ODEQ 2012) is the lack of concordance among the indicators in a given watershed, as was shown in figure 7-5.

One parameter may be trending in a positive direction while another is outside or moves outside the acceptable range. Also, the number of streams surveyed by ODEQ for water-quality impairment has increased over the last 10 years, and differences in the way specific metrics are used may explain some of these apparent differences. More research is needed to fully assess whether such monitoring results represent favorable ecological changes over the long term.

Generally, analytical approaches and their interpretation for broad ecosystem assessments are still developing, including novel uses of individual metrics and multivariate methods. The initial NWFP monitoring strategy was based on identifying key stressors and a conceptual model that linked ecosystem and species components (Mulder et al. 1999). The decision-support system based on expert judgement used for interpretation of metrics in the 10-year report (fig. 7-6) is prone to subjectivity and uncertainty, however, there may be no real alternative given the needs of policymakers, the complexity of the

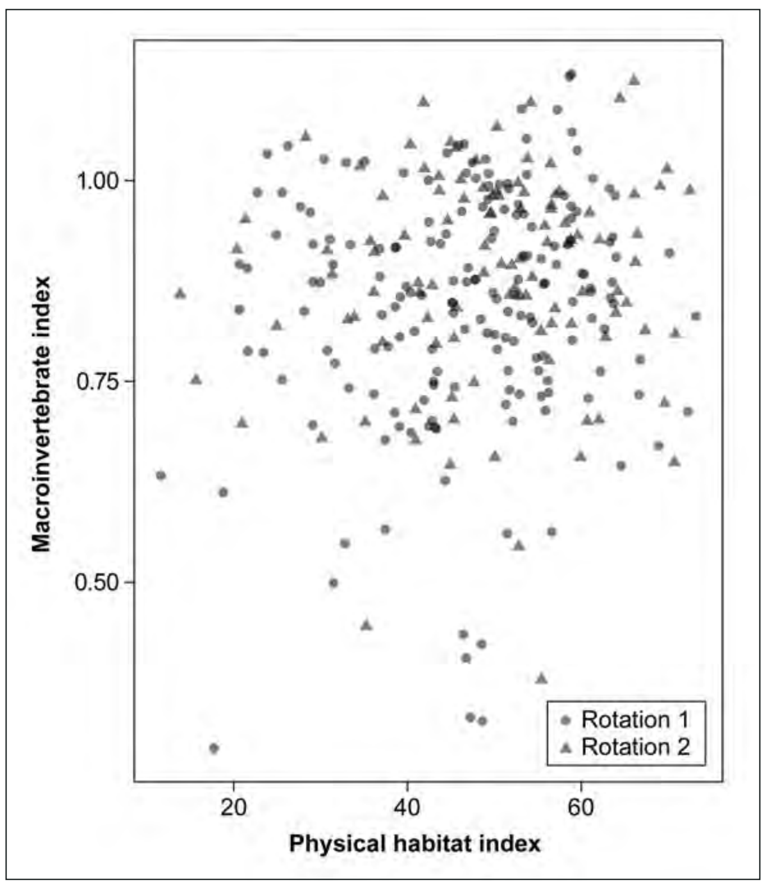

Figure 7-5—Relation between scores for overall watershed condition and condition of the macroinvertebrate communities for watersheds in the Northwest Forest Plan area in the assessment by Miller et al. (2017).

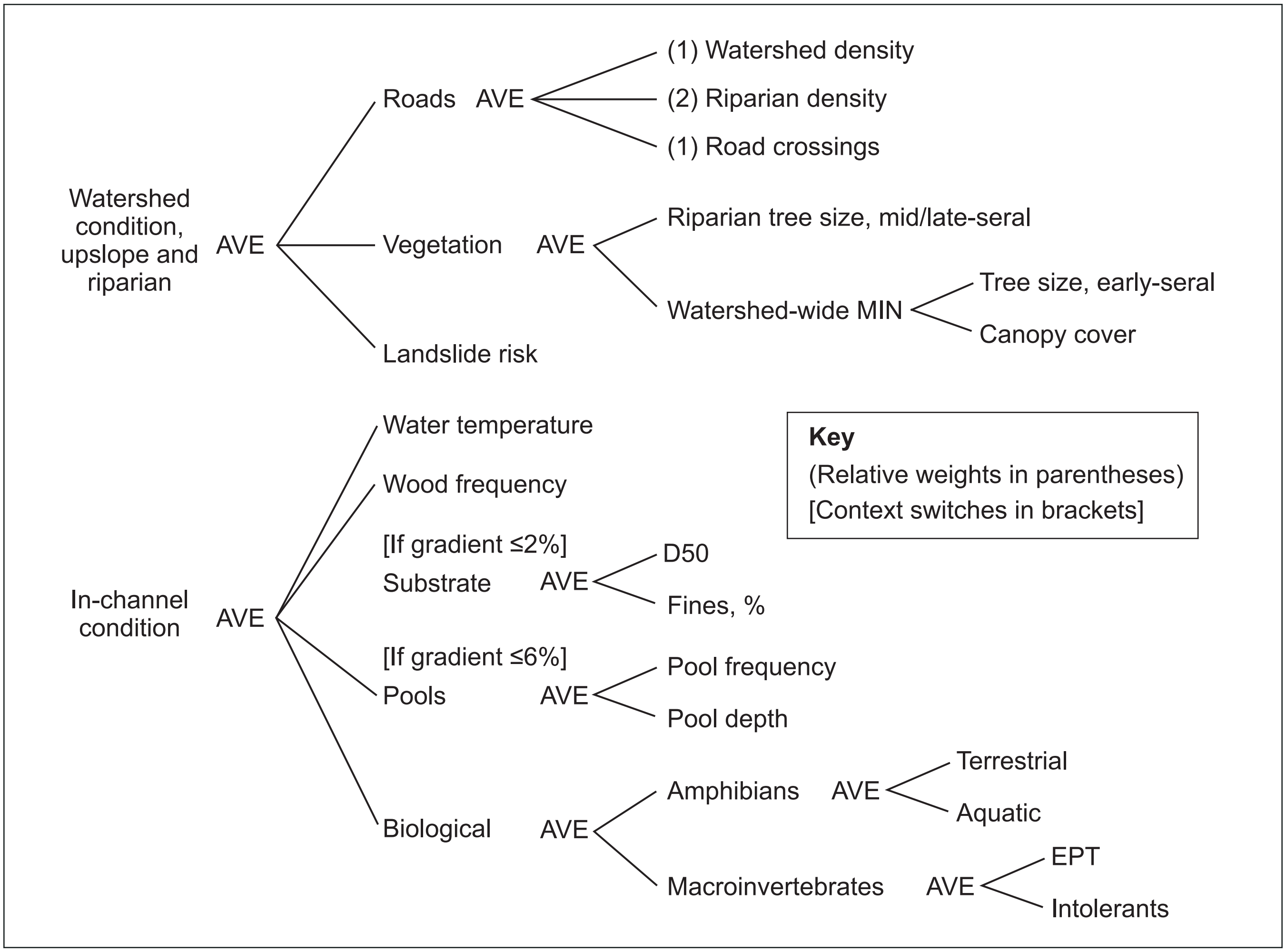

Figure 7-6—Example of a decision process for assessing the ecological condition of a watershed (Lanigan et al. 2012). AVE = average of scores; MIN = minimum; D50 = median particle size; EPT = Ephemeroptera, Plecoptera, Tricoperta index. See Reeves et al. (2004) for details.

ecological systems, and existing information gaps (Pielke 2007). The movement away from expert opinion in the 20-year report raises a new set of questions about use of single metrics interpreted independently and without an ecological framework. Carlisle et al. (2008) suggested that if one of the parameters of interest in a sampling unit (reach or watershed) is outside the threshold value for suitable conditions, the unit as a whole is outside the suitable range. A more statistically rigorous approach was described by Bowman and Somers (2006) and Collier (2009). In sum, exploration of a variety of available approaches is merited and timely.

Summarization over space and time—
A final consideration related to data integration is how to aggregate and present data over space and time. Reporting units over the spatial extents of land ownerships, land authority jurisdictions (e.g., county), and land-use allocations can address management and policy objectives, but the relationship of environmental conditions over space and time (e.g., via ecological provinces, watershed hydrological units) is also fundamental to our scientific understanding of aquatic-riparian ecosystems, particularly under the disturbance ecology paradigm (Olson et al. 2017b). Broad-scale aquatic assessments often report results by one or more aggregated spatial subunits.

Two types of units are evident in AREMP and other efforts. The first is based on ecological conditions. Similar to the other NWFP monitoring modules, AREMP reports results by seven aquatic provinces, which are similar to the physiographic zones developed in the NWFP planning process (FEMAT 1993: app. V-A). Broad ecological zones were similarly used in NRSA, both for the definition of reference conditions and the reporting of results (USEPA 2016). Hubler (2009) took a more water-centric approach, using water basins (hydrologic units, in particular HUC6, mean area ~10,000 mi^2 [~25 900 km^2]) as their principal reporting unit. Studies focused on fish populations have developed units to represent genetically and demographically independent groups of fish, referred to as evolutionarily significant units (ESUs) or distinct population segments (DPS) (McClure et al. 2003).

A second type of spatial aggregation common in the aquatic-monitoring literature is by management units. AREMP reports on indicators by NWFP land-use allocations, which generally correspond to land use intensities. The National Water-Quality Assessment targeted their sample by land use disturbance levels and so displays many of its results by urban, agriculture, and mixed-use classes (Carlisle et al. 2013). Hubler (2009) reported their results by ownerships (federal, state, private industrial, private nonindustrial). PIBO's regional reports have not broken down their data by management classes directly, but they have displayed results of all sites in concert with reference distributions, indirectly reflecting management classes (Archer and Ojala 2016).

In earlier AREMP reports, the NWFP aquatic provinces were the basis of alternate evaluation criteria. However, as described above, the most recent assessment used neighborhoods based on in-channel characteristics, and upslope zones based on forest types. Because the aquatic provinces are no longer used to set environmental parameters for the assessment, other aggregations may prove more useful. For example, the NWFP provides a common set of standards and guidelines for management in the region; they are implemented via management plans developed by each agency for each of their forests and districts. AREMP may wish to consider reporting by these unit and agency boundaries to increase their relevance to managers. Additionally, because these plans must address endangered species issues, the ESU and DPS boundaries may now be more relevant ecological units than the NWFP provinces.

AREMP's mandate is only to assess conditions on federal lands under the jurisdiction of the NWFP. However, nonfederal lands have been shown to be important, both for their effects on federal land conditions as well as for containing potentially productive fish habitat (Burnett et al. 2007, Reeves et al. 2016a, Van Horne et al. 2017). Further investigation in how to link AREMP data with assessments covering nonfederal lands could help address this gap, for an all-lands approach to watershed assessments in the moist coniferous forest ecosystem.

Because watershed conditions differ naturally over time, individual watershed ratings may be truly meaningful only when considered in some larger aggregation (Poole et al. 2004, Reeves et al. 2004). Thus, the end goal of AREMP was to look for changes in the distribution of watershed conditions in the whole NWFP area over time. As a baseline for comparison, AREMP chose to simply use conditions in the first monitoring period, because the other options considered (historical conditions, simulated natural conditions) would have been challenging to implement. Some other major programs have also used initial monitoring results as baseline conditions (Archer and Ojala 2016, USEPA 2016), whereas others have simply focused on one point in time, for example, using their most recent data (Carlisle et al. 2013, Hubler 2009). To our knowledge, historical conditions have not been estimated directly from past records, and simulation of natural conditions has been attempted only over considerably smaller spatial scales (Wondzell et al. 2007).

Finally, analytical approaches and their interpretation for broad ecosystem assessments are still developing, including novel uses of individual and multivariate methods. The rich AREMP dataset is ideal for comparison of alternative analytical approaches to assess watershed condition. Despite uncertainties and challenges faced by the 10-, 15-, and 20-year AREMP reports, reported trends support the intent of the ACS to sustain and improve conditions of federally managed watersheds in the NWFP area.

Components of the Aquatic Conservation Strategy

Riparian Reserves

Riparian reserves were intended to define and delineate the outer boundaries of the riparian ecosystem and to encompass the portions of a watershed most tightly coupled with streams and rivers (FEMAT 1993). These areas were assumed to provide the ecological functions and processes necessary to create and maintain habitat for aquatic and riparian-dependent organisms over time. This includes dispersal corridors for a variety of terrestrial and riparian-dependent organisms, and connectivity of streams within watersheds (FEMAT 1993). In 1993, the Forest Ecosystem Management Assessment Team (FEMAT) developed three management scenarios for riparian reserves along fish-bearing and non-fish-bearing streams (FEMAT 1993). Each scenario required a reserve width on fish-bearing streams of two times the height of a site-potential tree (minimum of 300 ft [91.4 m]), defined as the average maximum height the dominant tree would be expected to attain given the growing conditions at that location. On non-fish-bearing streams, the width of the riparian reserves varied from one-sixth of a site-potential tree-height (minimum of 25 ft [7.6 m]) to one-half of a site-potential tree-height to one site-potential tree-height (FEMAT 1993). One scenario was integrated into each of the 10 landscape alternatives developed and evaluated by the FEMAT (1993) scientists.

The Secretaries of the Interior and Agriculture selected FEMAT's Option 9 as their preferred option, which required a riparian-reserve network that was two site-potential tree-heights wide on fish-bearing streams and one-half of a site-potential tree-height on most non-fish-bearing streams. Interim boundaries of the riparian reserves were extended to a full site-potential tree-height on all non-fish-bearing streams between the draft and final environmental impact statements (USDA and USDI 1994a) to increase the likelihood of success of the ACS, and to provide additional protections from timber management and road building for non-fish organisms that use the area in or near streams as habitat or migratory corridors (USDA and USDI 1994a). On some fish-bearing streams, two site-potential tree-heights from the edge of a stream may not encompass the whole floodplain, which can be an important source of large wood, making it critical to recognize and protect the entire floodplain (Latterell and Naiman 2007). This was accomplished in the ACS by requiring the boundary of the riparian reserve to extend to the edge of the 100-year floodplain (USDA and USDI 1994a). These boundaries were considered interim until a watershed analysis, which could adjust the size of the riparian reserve, was completed (USDA and USDI 1994a).

Depending on the degree of dissection of the forested landscape by streams, riparian reserves along both perennial and intermittent streams may occupy between 40 and 90 percent of the landscape (FEMAT 1993, Hohler et al. 2001). Interim riparian reserves of this magnitude, coupled with key watersheds and late-successional reserves, have provided a connected watershed-level reserve system for terrestrial, riparian, and aquatic ecosystems (Everest and Reeves 2007). However, the area of the forested landscape contained in the riparian reserves has fueled a controversy regarding riparian protection, resulting in new research to evaluate prescribed widths of riparian-management areas and a reexamination of existing scientific literature on the subject (Everest and Reeves 2007). The following sections summarize some of the recent key literature relating to the functions and size of riparian reserves.

Ecological functions—

The scientific basis for delineation of interim riparian reserves in the NWFP was derived from two sets of curves showing the relationship between various ecological functions provided by riparian zones and distance from the channel (figs. 7-7 and 7-8). These curves were developed by FEMAT scientists based on the scientific literature that was available at the time, and on professional judgment when sources of information were incomplete (see table 7-2 for original sources). The original relationships (FEMAT 1993) that were incorporated into the NWFP (USDA and USDI 1994a) suggest that most ecological functions could be maintained by reserves equal to or less than the distance of one site-potential tree-height. The functions include beneficial effects of root strength for bank stability, litterfall, shading to moderate water temperatures, and delivery of coarse wood to streams (fig. 7-7A). In addition, the majority of moderating effects on sediment delivery to streams

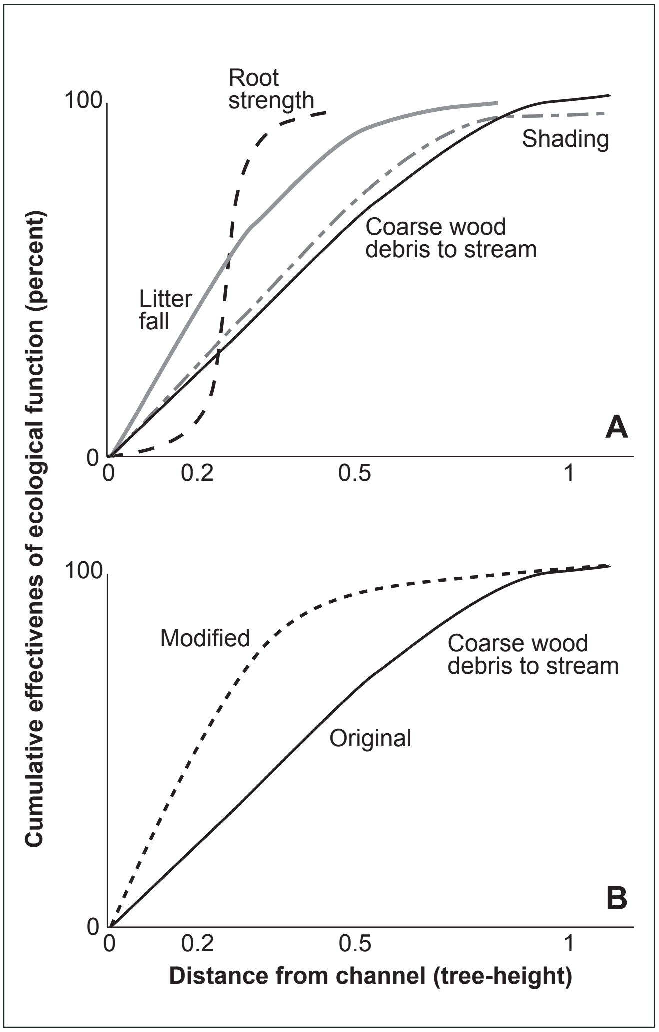

Figure 7-7—(A) Relation of distance from stream channel to cumulative effectiveness of riparian ecological functions (FEMAT 1993: V-27); (B) modified effectiveness curve for wood delivery to streams as a function of distance from the stream channel. The original curve was changed based on scientific literature developed since the original curve was portrayed in FEMAT (1993). Source: Spies et al. 2013.

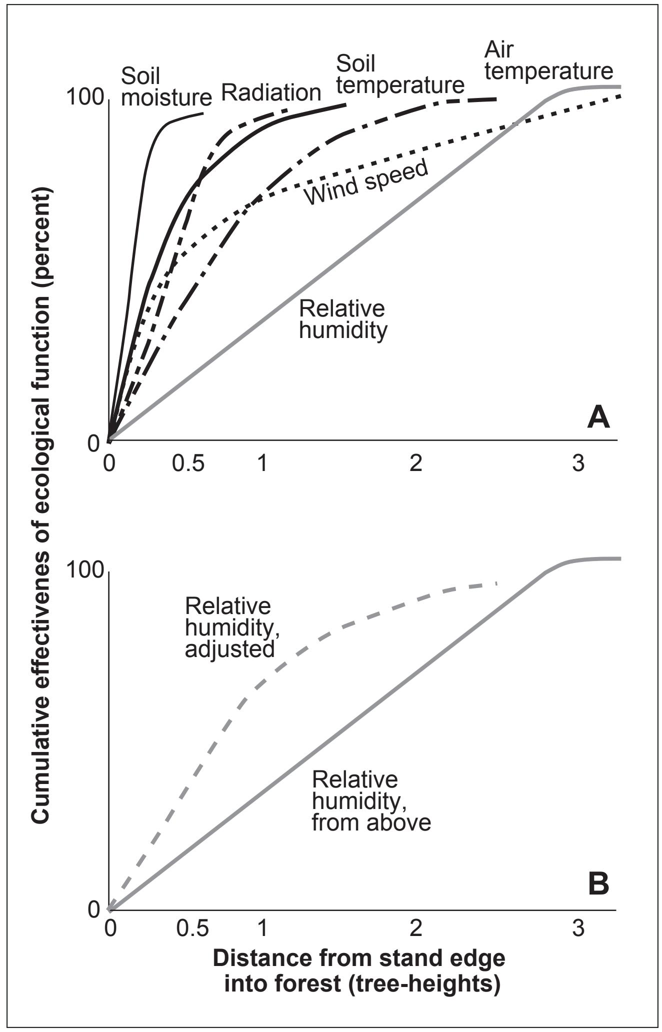

Figure 7-8—(A) Relation of distance from stream channel to cumulative effectiveness of ecological factors influencing microclimate in riparian ecosystems (FEMAT 1993: V-27); (B) modified effectiveness curve for relative humidity as a function of distance from the stream channel. The curve was changed based on scientific literature developed since the original curve was portrayed in FEMAT (1993). Source: Reeves et al. 2016a.

from overland erosion associated with upland activities generally occur within a distance of one site-potential tree-height (Castelle et al. 1994, Naylor et al. 2012). The FEMAT scientists also provided a margin for error allowing for incomplete science, unknown cumulative effects, or strategic uncertainty in defining interim riparian reserves prior to watershed analysis. Everest and Reeves (2007) concluded that science published since original development of the FEMAT curves has generally supported the original assumptions and judgments.

Recent studies of wood recruitment suggest that changes in some of the ecological function curves may be supported. According to the graph of the relationship between the cumulative effectiveness of an ecological process and the distance for wood recruitment from the immediately adjacent riparian area in fish-bearing streams, developed in FEMAT (1993), about 60 percent of wood recruitment from the immediate riparian area along fish-bearing streams occurs within one-half of a tree-height (fig. 7-7A). This graph was based on a limited number

Table 7-2—Literature sources used to develop the original curves of ecological functions in riparian reserves in FEMAT (1993)

Function	Sources
Root strength	Burroughs and Thomas 1977 Wu et al. 1986
Wood delivery	Beschta et al. 1987 McDade et al. 1990 Van Sickle and Gregory 1990
Litterfall	Professional judgment
Shading	Beschta et al. 1987 Steinblums 1977
Microclimate	Chen 1991

of studies (McDade et al. 1990, Van Sickle and Gregory 1990) and the professional judgment of scientists involved with FEMAT. More recent studies on the sources of wood (Gregory et al. 2003, Spies et al. 2013, Welty et al. 2002) found that, at least in the Cascade Range of western Oregon and Washington, about 95 percent of the total instream wood inputs from the adjacent riparian area along fish-bearing streams came from distances of 82 to 148 ft (25 to 45 m) from the stream, representing a distance of 0.6 to 0.7 of a site-potential tree-height for this area (fig. 7-7B). The shape of this curve differs from the FEMAT curve (fig. 7-7A), which showed that 95 percent of the wood-recruitment function of the same streams occurs within a distance equal to about 0.95 of the height of a site-potential tree.

A primary purpose for the extension of the boundary of the riparian reserve of the NWFP from one site-potential tree-height to two on fish-bearing streams was to protect and enhance the microclimate of the riparian ecosystem within the first tree-height (USDA and USDI 1994a). At the time the ACS was developed, the only research on the effects of clearcutting on microclimatic conditions in adjacent forests had been done in upland forests on level terrain (Chen 1991). Those studies found that the influence of recent clearcuts (10 to 15 years old) extended from tens of yards (e.g., soil moisture and radiation) to hundreds of yards (e.g., wind velocity) into adjacent unharvested stands. Based on the initial work of Chen (1991), FEMAT (1993) hypothesized that a second tree-height could provide a considerable safety margin from negative effects of intensive management on riparian areas, in terms of relative humidity and other microclimatic effects in the riparian reserve along fish-bearing streams (FEMAT 1993) (fig. 7-8A).

Since the ACS and associated ecological-function curves were originally formulated, a number of research efforts have examined the effects of forest management on microclimate in riparian areas. The vast majority of this work has focused on air temperature and relative humidity in small, headwater streams; few studies were conducted along larger streams (see review by Moore et al. 2005; also Olson et al. 2007, 2014). The magnitude of harvest-related changes in microclimate in riparian areas is usually inversely related to the width of the riparian buffer and the type and extent of management activities on the outer (upslope) edge. Some studies failed to show any edge effect between clearcuts and riparian buffers composed of intact mature forest (i.e., the extent of change in microclimatic conditions resulting from the presence of a clearcut on upslope edge of the riparian area) (Anderson et al. 2007, Rykken et al. 2007). Other studies have found that edge effects varied from a distance of 98.5 ft (30 m) (Anderson et al. 2007, Rykken et al. 2007) to 148 ft (45 m) (Brosofske et al. 1997) from the stream. At the other extreme, Ledwith (1996; as cited by Moore et al. 2005) found that above-stream temperature decreased and relative humidity increased as buffer widths increased up to 492 ft (150 m). Rykken et al. (2007) attributed the lack of an edge effect to a "stream effect," described by Moore et al. (2005), who noted that the stream can act as a heat sink and a source of water vapor during the day, thus keeping near-stream microclimates cooler and more humid than areas farther from the stream. Rykken et al. (2007) suggested that this stream effect might counteract edge effects of harvest on microclimate, thereby reducing the distance that harvest effects penetrate into riparian zones, relative to the distances measured in upland forest edges (e.g., from those projected by Chen et al. [1993] in uplands). Moore et al. (2005) also posited that cool, moist air might be carried by down-valley breezes, contributing to this stream effect.

The FEMAT microclimate curves were based on upland studies of forest-edge effects and thus they do not necessarily apply to riparian areas with a strong stream effect, protected topographic position, and retention of some canopy in the adjacent managed stand. Reeves et al. (2016a) suggest that a one tree-height buffer on fish-bearing streams (fig. 7-8B) would reduce most potential effects on microclimate and water temperature in near-stream environments from timber harvest in areas on the edge of the riparian reserve, particularly when some trees are retained in the harvest unit. In general, most studies show that microclimatic changes in temperature and relative humidity seldom extend farther than one site-potential tree-height from the clearcut edge into an intact riparian buffer composed of mature forest (see review by Moore et al. 2005 and references cited therein). However, the large variety of effects measured in different studies demonstrates that substantial uncertainties remain about the size and management of riparian reserves. These uncertainties have important implications when considering changes in the width of the NWFP riparian reserves.

Increased stream temperature following forest harvest is one of the most frequently mentioned management concerns, and one that retention of riparian buffers is clearly designed to mitigate. Generally, the smaller the riparian area and the more extensive the activities, the greater the effect on stream temperature. Clearcut logging without riparian buffers usually leads to large, post-harvest increases in stream temperature, and the width of the riparian buffer needed to limit, or even eliminate, temperature increases remains uncertain (see reviews by Moore et al. 2005 and Leinenbach et al. 2013). Given these uncertainties, management prescriptions that reduce the width of the riparian reserve or allow some tree harvest within the reserve remain controversial.

The NWFP area encompasses a wide array of bioclimatic conditions, across its latitudinal span, west to east with distance from the ocean and rain-shadow effects of mountain ranges, and with increasing elevation. Given this variation, we describe only a few broad, general patterns in riparian vegetation here. In subsequent sections, we contrast these patterns with present-day patterns in areas that were previously logged, especially where logging pre-dated the establishment of current forest-practices rules and allowed harvest right up to streambanks. Although these general trends are important considerations, we also emphasize that more detailed local knowledge will be critical for determining appropriate management goals and planning specific actions.

Riparian and upland vegetation along headwater streams in moist or wet forest types is typically dominated by conifers (Nierenberg and Hibbs 2000, Pabst and Spies 1999, Sheridan and Spies 2005) (fig. 7-9A). However, conifer density can be lower in riparian zones compared to adjacent terrestrial areas (Sheridan and Spies 2005), hardwoods are uncommon in both riparian and upslope

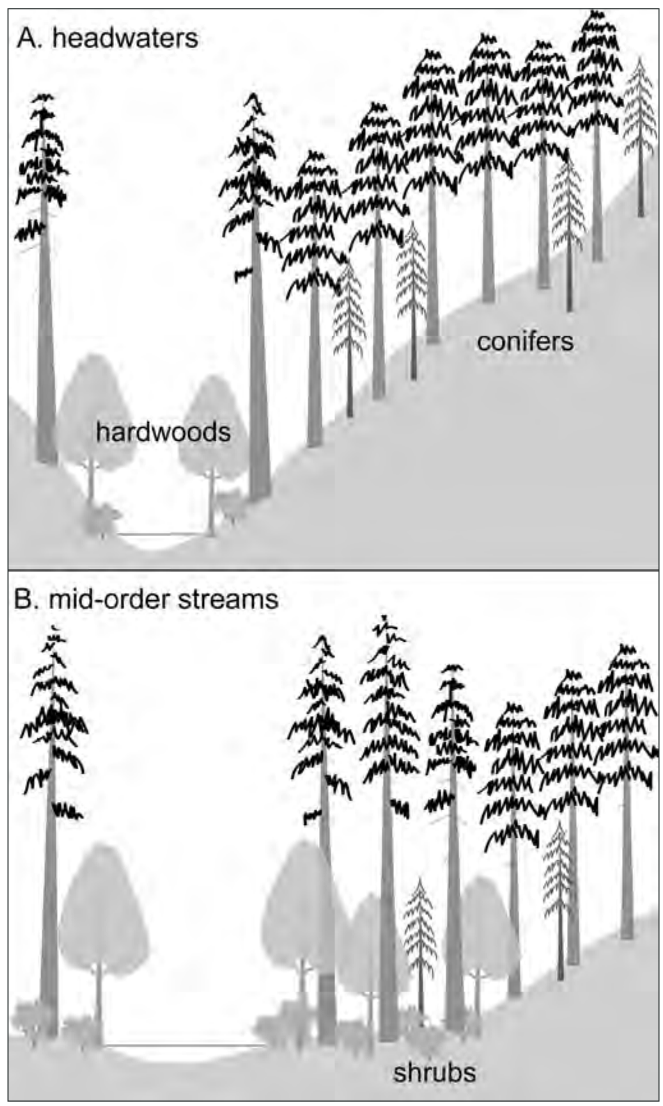

Figure 7-9—Conceptual representation of (A) vegetative conditions in headwater and (B) mid-order streams in the Northwest Forest Plan area.

areas, and there are no clear differences in shrubs between the two zones (Sheridan and Spies 2005). Many mosses and liverworts are also found at the wetted edges of small streams or on wood and rock in and along the channels (Hylander et al. 2002). In these wet forest types, tree canopies are often dense, limiting sunlight and therefore primary productivity. As a result, headwater streams depend on allochthonous (coming from outside the stream) inputs of litter and terrestrial invertebrates from the adjacent riparian forests, with as much as 95 percent coming from within 45 to 83 ft [13.6 to 25.3 m] of the channel (Bilby and Heffner 2016), as the primary energy source for aquatic and riparian organisms in these streams and for those lower in the network (Baxter et al. 2005, Gomi et al. 2002, Leroy and Marks 2006, Richardson et al. 2005, Wallace et al. 1997, Wipfli and Baxter 2010). Allochthonous material is exported to downstream areas as dissolved organic carbon, coarse (>1 mm, >0.04 inches) particulate organic matter, and to a much lesser extent, as fine particulate organic matter (Gomi et al. 2002, Richardson et al. 2005), which contribute to the productivity of fish-bearing streams. The structure and composition of the riparian vegetation determines the quality, quantity, and timing of the allochthonous input (Cummins et al. 1989, Frady et al. 2007), all of which influence overall stream productivity.

Forests in riparian zones and adjacent uplands become increasingly different as the size of streams increases (fig. 7-9B). In the middle portions of stream networks, the riparian forest is more diverse than along headwater streams (Acker et al. 2003, Johnson et al. 2000, Sarr and Hibbs 2007). Riparian forests along many mid-sized streams still remain dominated by conifers, but they are often mixed with deciduous trees such as alder (*Alnus* spp.), big-leaf maple (*Acer macrophyllum*), willow (*Salix* spp.), and cottonwood (*Populus* spp.). Big-leaf maple and California black oak (*Quercus kelloggii*) can be common in riparian zones in the southern portion of the NWFP area.

Some studies in mid-sized streams have shown that conifer basal area nearest the stream can be lower owing to reduced survival from disturbances such as flooding (Hibbs and Giordano 1996, Pabst and Spies 1999). Also, the availability of growing sites might be limited; conifers preferentially establish on "microtopographic ridges" created by old tree-falls and behind wood jams (Fetherston et al. 1995). As a result, tree density near the stream can be about half of that of upland stands (Acker et al. 2003, Rot et al. 2000, Wimberly and Spies 2001). In contrast, Pollock et al. (2012) suggested that there is little difference in tree density between upland and riparian stands. All these studies, however, did find that the basal area of conifers in streamside stands is greater than in stands farther from the channel or in adjacent uplands. Streamside trees can be among the largest in a watershed (Poage and Tappeiner 2002), and thus can be the source of the largest down trees (conifers) found in the channel (i.e., the key pieces) (Rot et al. 2000), which are generally recruited to the channel by undercutting at high water (Abbe and Montgomery 2003, Acker et al. 2003, Benda et al. 2003).

In the Oregon Coast Range, hardwoods were most abundant in the area closest to the channel of streams in the middle portion of the stream network (Pabst and Spies 1999, Wimblerly and Spies 2001), particularly in unconstrained reaches (Acker et al. 2003), and they decreased in density moving away from the channel. This mix of hardwoods and conifers is important ecologically (Sponseller and Benfield 2001, Sponseller et al. 2001) and is frequently maintained by periodic flooding (Sarr and Hibbs 2007). The vegetative diversity provides diverse habitat for a suite of terrestrial and riparian organisms; hardwoods are especially important for riparian mollusks (Foster and Ziegltrum 2013) and Neotropical migrant and resident bird species (Pearson and Manuwal 2001). Riparian areas dominated by hardwoods, particularly nitrogen-fixing red alder (*Alnus rubra*), have the potential to increase primary (Cornwell et al. 2008, Kominoski and Pringle 2009, Kominoski et al. 2011, Schindler and Gessner 2009, Swan et al. 2009) and secondary productivity and invertebrate diversity in adjacent streams (Piccolo and Wipfli 2002, Srivastava et al. 2009, Wipfli and Musslewhite 2004). Watersheds with mixed hardwood/coniferous riparian vegetation in the Oregon Coast Range received nearly 30 percent greater influx of terrestrial invertebrate biomass than streams with conifer-dominated riparian areas (Romero et al. 2005). The loss or reduction of deciduous litter could potentially influence the structure, composition, and productivity of riparian and aquatic biota (Wallace et al. 1997, 1999)

Although these broad general patterns hold across much of the NWFP area, we do not mean to give the impression that the riparian forests and their adjacent uplands were uniformly forested. Rather, these forests were a complex, shifting mosaic of vegetation patches, presenting a landscape with great spatial variability and temporal dynamics (see more detailed discussion in chapter 3). Wildfire was the primary factor driving forest dynamics across the Oregon Coast Range (Wimberly et al. 2000) and other parts of the region (see chapter 3), although windthrow, insects, and disease can also be important. As a consequence, upland forests, even when assessed at large spatial scales, showed substantial variation in the area and ages of forests. For example, over long periods, the proportion of upland forest in old-growth condition, when summed over areas of more than 4.9 million ac (2 million ha), ranged from 25 to 75 percent; at the scale of late-successional reserves as specified in the NWFP (~98,842 ac [40 000 ha]), the amounts of old growth could range from 0 to 100 percent (Wimberley et al. 2000).

In addition to the factors described above for upland forests, riparian forests are also influenced by fluvial and geomorphic processes such as floods, debris flows, and bank erosion. State-and-transition simulations of the natural disturbance regime in the Oregon Coast Range (Wondzell et al. 2012) showed that 51 percent of the riparian network was in mature forest (stand age of 66 to 200+ years). The simulations also showed that the long-term average forest composition was highly variable. Only 2 percent of the riparian network was in a nonforested condition, 28 percent was alder dominated, 40 percent was in mixed alder/conifer stands, and only 29 percent was in conifer-dominated stands (see table 5 in Wondzell et al. 2012). The specific results cited above pertain only to the central Oregon Coast Range, but it is clear that no single condition—defined by stand composition, structure, and tree age—can represent the full distribution of naturally occurring conditions over large areas. Rather, riparian forest conditions, when assessed over broad landscapes, showed a distribution of conditions that resulted from the combined influences of natural disturbances and plant succession.

Human impacts and restoration—

Riparian forests throughout much of the NWFP area have been changed by the land use activities that have taken place over the past century. As a consequence, the present-day forests may frequently differ in structure and composition from the presettlement forests that preceded them (McIntyre et al. 2015, Naiman et al. 2000, Swanson et al. 2011). This is particularly evident in an estimated 30 to 50 percent of the riparian ecosystems in the NWFP area that have been converted to plantations, based on the percentages of plantations in upland forests (see chapter 3). Riparian forested areas were harvested extensively, often to the edge of the stream, prior to the advent of current policies (Everest and Reeves 2007). In many cases, the riparian zones were subsequently planted with the most commercially valuable conifers, primarily Douglas-fir (*Pseudotsuga menziesii*), resulting in the development of dense, relatively uniform conifer stands and a decrease in hardwoods. In other cases, conifers were not successfully reestablished in logged riparian zones that are now dominated by alder with a dense salmonberry (*Rubus spectabilis*) understory, as observed at the reach scale by Hibbs and Giordano (1996). In watershed-scale simulations, Wondzell et al. (2012) estimated that, under historical conditions, 28 percent of the stream network in the Oregon Coast Range was in alder-dominated riparian forests, and that presently it is more than 40 percent. Fire suppression in dry forests with high- or moderate-frequency fire regimes has likely altered the structure and composition of riparian vegetation in ways similar to those described for upslope forested areas—namely an increase in the density of shade-tolerant conifers and a reduction in hardwoods (see chapter 3). In moist forests, with infrequent fire regimes, fire suppression has likely reduced the area of early-seral conditions in uplands and riparian areas (see chapter 3). Similarly, the removal of large conifers along rivers in the coast redwood (*Sequoia sempervirens*) range of northern California has been associated with increased dominance by alder (Madej et al. 2006). Clearly, the direct effects of logging on the structure and composition of present-day riparian forest can be varied, but overall, the distribution of conditions has changed dramatically relative to those under natural disturbance regimes.

Indirect effects of logging have also modified riparian forests. For example, rates of landslides and debris flows have increased in heavily roaded and logged watersheds (Goetz et al. 2015, Guthrie 2002, Jakob 2000), which has led to systematic changes in riparian vegetation. Debris-flow tracks are frequently scoured free of large wood and subsequently recolonized by red alder (Russell 2009, Villarin et al. 2009), with large wood deposited in runout zones. Further, the frequency of debris flows and landslides has contributed additional sediment to stream channels, driving more severe floods, with the combined effect of increasing the width of stream channels (Lyons and Beschta 1983). Exposed gravel bars within these channels are most often colonized by hardwoods, leading to substantial changes along the stream corridor.

Restoration challenges—

The changes to riparian forests described above create substantial challenges for restoration. For example, thinning of dense riparian Douglas-fir stands could open stands, allowing increased hardwood presence and thereby increasing the diversity of riparian vegetation, while also promoting growth of the remaining trees to decrease the time needed to grow trees large enough to act as key structural elements in the stream channel. Although such restoration treatments may speed the restoration of some ecological functions (USDA and USDI 1994a), they also may reduce dead wood (chapter 3), and may present risks, such as development of novel conditions and loss of a particular species or ecological condition. Concerns about the tradeoffs between potential gains and potential losses, or other management issues, appear to have limited restoration activities, particularly within the first site-potential tree-height of streams. Reeves (2006) estimated that 48,000 ac (19 400 ha) of riparian reserve in the matrix of the NWFP area was treated for restoration purposes using some form of vegetation management, primarily thinning, in the first 10 years of the Plan. Between 2010 and 2015, an additional estimated 38,719 ac (15 669 ha) in the Forest Service's Pacific Northwest Region (Region 6) were commercially or noncommercially thinned (table 7-3). FEMAT (1993) estimated that 2.2 million ac (890 000 ha) of riparian reserves were outside of other reserves and congressionally withdrawn areas in the NWFP area.

Table 7-3—The estimated area of riparian reserves in the Northwest Forest Plan area in the Pacific Northwest Region of the U.S. Forest Service where active management that produced trees for commercial (primarily in the second tree-height of the riparian reserve) and restoration (noncommercial) purposes has occurred in 2010–2015

National forest	Area of riparian reserve managed			
	Commercial		Noncommercial	
	Hectares	*Acres*	*Hectares*	*Acres*
Deschutes	168	415	461	1,139
Fremont-Winema	0	0	0	0
Gifford-Pinchot	1031	2,548	301	744
Mount Baker–Snoqualmie	125	309	0	0
Mount Hood	674	1,665	0	0
Okanogan–Wenatchee	331	818	2150	5,313
Olympic	750	1,853	454	1,122
Rogue River–Siskiyou	142	351	616	1,522
Siuslaw	3923	9,694	203	502
Umpqua	883	2,182	622	1537
Willamette	2835	7,005	No data	No data
Total	10 862	26,841	4807	11,878

Source: USDA Forest Service Pacific Northwest Region.

Because this is not the total area of riparian reserves, it is not possible to estimate the fraction of the riparian reserve in the NWFP area that has undergone restoration. However, it is clear that the area that has been treated represents a relatively small proportion of the riparian reserves in total, and of the amount that has been altered by past activities.

Primary reasons for the limited amount of restoration activity are various and probably include (1) differing perspectives about the characterization of reference conditions, conservation, and management; (2) concerns about the potential effects of mechanical treatments on stream temperature and wood recruitment; (3) concerns about rare and little-known organisms that made managers reluctant to alter default prescriptions (Reeves 2006); and (4) trust (see chapter 12). We explore the potential challenges associated with these restoration activities below.

Reference condition versus restoring function— Restoration activities necessarily require a "target" condition or conditions toward which the restoration activity is intended to move a system. One way to select a target for restoration goals is to identify a minimally disturbed condition and use it as a reference to which the current condition can be compared. The minimally disturbed condition is commonly called the reference condition. Although intellectually appealing, the selection of a reference condition is fraught with potential biases. For example, Pollock et al. (2012) set very stringent requirements on stand attributes that would be acceptable as a reference condition for Douglas-fir-dominated stands in riparian forests of western Washington state: choosing undisturbed, single-storied, conifer-dominated stands ranging in age from 80 to 200 years, and excluding stands dominated by hardwoods or shrubs that showed evidence of recent severe disturbance—including disturbances such as wildfire, insects, and disease—because these disturbances may themselves have been a product of fire exclusion or climate change. Also, stands with these features were assumed to be the successional climax forest for this size of stream, and such stands had been greatly reduced by logging.

The study by Pollock et al. (2012) illustrates some of the challenges inherent in finding reference conditions—they are often rare and may not represent the historical range of conditions that existed before extensive anthropogenic modification of upland and riparian vegetation. Ideally, reference conditions would be identified in areas with similar potential vegetation and in relatively close proximity, or at least within the same ecoregion, so that the reference provides an appropriate comparison for similar forest stands (NRC 2000). Because of their focus on older conifer-dominated patches and the assumption that these types of stands represented the primary natural vegetation of streamsides in more confined terrain, Pollock et al. (2012) were able to identify only a small portion of the existing riparian forests in which stands meeting their criteria for dominance by older Douglas-fir trees could be found (only 117, or 3.3 percent, of the 3,521 potential sites met the filtering criteria). These stands were widely scattered, spanning a broad latitudinal and climatic range in western Washington. Further, they lumped together both upland stands and riparian stands, and the only riparian reference stands were located in the western Washington Cascade Range. Thus, the applicability of the results to other stand types and locations is very limited. Nonetheless, Pollock et al. (2012) illustrated that a stringent filtering approach to identifying reference sites could contribute to characterization of reference conditions at the patch or stand scale (e.g., stand density and tree size) for evaluating riparian-management options for Douglas-fir riparian stands in western Washington. The Douglas-fir patch-scale reference conditions could also be used in setting management goals for the entire landscapes of a larger riparian zone. A similar approach could be applied to other stand types or regions to provide a more complete system of reference conditions for riparian management in the Pacific Northwest.

Another approach to setting reference conditions (although they did not call them "reference conditions" at the time) for riparian zones was used by Pabst and Spies (1999) and Nierenberg and Hibbs (2000) in the Oregon Coast Range. They sampled along first- through fourth-order streamsides without roads or a history of logging, and having no evidence of wildfire at least 80 years. Vegetation was sampled in transects from randomly selected starting points. Hence, the samples contained areas dominated by older conifers as well as patches of hardwoods and shrubs, in proportion to their occurrence in the riparian area. Smaller, recent areas of geomorphic disturbances, disease, and windthrow would have been included in the samples. The studies focused in particular on how vegetation differed between streamsides and uplands at a site, controlling for differences in environment and disturbance history. Generally, these studies found that conifer dominance decreased from the uplands to the streamside, and that many areas within 53 ft (16 m) of streams were typically a mosaic of conifers, hardwoods, and shrubs even along streams in relatively confined topographic settings. Conifer-snag densities were relatively low 17 snags/ac (6.9 snags/ha) within 53 ft (16 m) of streams, and about one-quarter of the densities found at distances of more than 53 ft (16 m) from streams (Pabst and Spies 1999). These studies could be used in developing management guidelines for riparian forests in the Oregon Coast Range. Note, however, that this approach (random samples of unmanaged riparian vegetation) did not sample many areas that had grown for several centuries since wildfire and also did not sample in large areas that were recently affected by fluvial, geomorphic, and fire disturbances, which would have been an important part of the historical range of variability in these ecosystems at watershed scales (Spies et al. 2002).

Wondzell et al. (2012) used state-and-transition models to explore the range of ecological states of the riparian network of a large river network in the central Oregon Coast Range. They used GIS methods to partition the stream network and its valley floor into discrete reaches, which were classified into potential geomorphic and vegetation types. A state-and-transition model was then developed for each potential type that included all possible states that could result from succession, natural disturbance, and land use activities. Wondzell et al. (2012) found that the structure and composition of the current riparian vegetation differed from the historical; there were fewer conifers, particularly the largest (>30 inches [76.2 cm]), and more alder-dominated patches. They clearly stated that their simulation results “should be interpreted as hypotheses of likely outcomes,” and that, despite several model limitations, they can be used to “hindcast” expected historical distribution of riparian forest conditions.

Part of the debate about restoration needs for riparian areas may derive from differing views of riparian reference conditions (as a goal for restoration), and how they differ with scale and across watersheds and the NWFP region. Although many studies (e.g., Acker et al. 2003, Hibbs and Sarr 2007, Pabst and Spies 1999) have found that riparian vegetation and upland vegetation frequently differ in structure, composition, and dynamics depending on stream size, some have noted that differences between riparian and upland vegetation may be small for some stand types, and that in some cases upland sites can supplement riparian sites to increase sample size for describing target conditions for riparian management. For example, Pollock et al. (2012) noted that, for Douglas-fir-dominated stands in western Washington, “both forest types [upslope and riparian] are generally similar, but riparian stands have more live tree wood volumes and basal areas, suggesting they may be growing on sites that are more productive.” Therefore, they concluded that riparian restoration in Douglas-fir-dominated riparian zones should aim to produce stand characteristics with densities and sizes of live and dead trees that are within the range of reference conditions (both upland and riparian). On the other hand, others (Gregory 1997, Pabst and Spies 1999, Welty et al. 2002, Wimberly and Spies 2001) have found that the type and magnitude of differences in features between upslope and riparian forests can be large, suggesting that upslope vegetation should not be assumed to be a reference for designing and assessing managed strategies for riparian vegetation in other stand types, or where riparian stands differ significantly from upland stands (e.g., in floodplains).

This variety of findings makes it difficult for managers and regulators to design and implement management actions in riparian reserves. We suggest that each of the approaches examined above—that of Pollock et al. (2012), Pabst and Spies (1999), Nierenberg and Hibbs (2000), and Wondzell et al. (2012)—offers important information that would contribute to building a “reference condition”-based strategy to examine current conditions, to project likely outcomes of planned management activities, and to help evaluate the tradeoffs between potential risks and benefits of any overall management strategy. For example, a modeling approach like the state-and-transition models of Wondzell et al. (2012) could be used to generate an expected historical distribution of states for the riparian vegetation within a stream network. The results could be used to identify relatively little-disturbed watersheds across the ecoregion and the monitoring plots located within those watersheds; individual plots could then be compared to specific states in the state-and-transition model. Those states could be attributed with values for various metrics (e.g., cover, basal area, tree densities, snag densities, species composition), as was done by Pabst and Spies (1999). Because anthropogenically disturbed states are also included in the state-and-transition models, something similar could be done to attribute these states with empirical data. The models could then be used to hindcast the historical distribution of state classes, and descriptive metrics from the empirical data could then be linked to the historical distribution. This result could be compared to the current condition. Also, the models could be used in forward simulations, incorporating different land use choices, to project how the distribution of conditions might be expected to change over time in response to various management strategies. Whatever approach is used, it will be important for managers and regulators to understand the limitations of the research they use to design and support proposed actions, which in turn necessitates that researchers clearly identify the limitations of their research (such as how broadly or to what ecosystem type they can be applied), and recognize the large variation in the inherent structure and composition of riparian areas across the NWFP area.

Riparian thinning and water temperature—Because the current distribution of conditions of riparian forests in many stream networks is far different from the historical distribution, there is substantial interest in active restoration treatments—especially thinning dense conifer plantations (Reeves et al. 2016a) or logging hardwood-dominated stands and replanting to convert them to conifer dominance (Cristea and Janisch 2007). Although these treatments are not inconsistent with the ACS, which generally allowed thinning for ecological objectives in the area beyond 120 to 150 ft (36.6 to 45.7 m) to a distance of one site-potential tree-height, they could potentially exceed the 0.3 °C "non-degradation standard" for water-quality effects of logging. The 0.3 °C standard is important from a regulatory perspective, limiting potential cumulative effects from multiple actions, none of which individually might be sufficient to impair water quality. Alternatively, restoration treatments might speed the attainment of desired future conditions. These decisions pose critical management challenges. Clearly, there are risks from any active restoration treatment, but choosing not to act also poses risks, not only by increasing the time needed to attain a desired future condition, but also leaving the riparian zone at greater risk of uncharacteristic disturbance—for example, dense conifer stands in dry forest zones are more prone to high-severity wildfire (see chapter 3). Also, there may be increases in primary production (Warren et al. 2016) and fish growth (Wilzbach et al. 2005) with the opening of the canopy along small and medium streams.

Reach-scale studies clearly demonstrate that solar radiation is the primary factor affecting stream-water temperatures during summer (Leinenbach et al. 2013). Thus, the likely effect of forest harvest on stream temperatures will be a function of the amount of shade lost. The largest effects will generally be seen with clearcut logging right to the streambanks, whereas retention of forested buffers tends to reduce these effects, as does thinning rather than clearcutting outside the buffer. The actual magnitude of stream-temperature increases can vary greatly and is determined by factors such as discharge, water depth, width, flow velocity, hyporheic exchange, and groundwater inflows (Janisch et al. 2012, Johnson 2004, Moore et al. 2005). Topographic shading can also influence water temperatures, particularly in small streams flowing in narrow, steep-sided valleys, as much as or perhaps more than shade from streamside forests (Zhang et al. 2017). It is important to remember that canopy removal also results in nighttime long-wave radiation loss, leading to lower water temperatures. This effect contributes to increased thermal variability, with poorly understood biological consequences.

Relatively few studies have examined the effects of riparian thinning on stream-water temperature. A few studies have examined clearcut harvesting combined with partial harvest of riparian buffers (Kreutzweiser et al. 2009, Macdonald et al. 2003, Mellina et al. 2002, Wilkerson et al. 2006). These studies, like those cited above, suggest that the effect of riparian thinning on summer stream temperatures will be correlated positively with the amount of forest canopy removed and inversely with the distance from the stream that the activity occurs, and thus the amount of shade lost (Leinenbach et al. 2013). However, the amount of shade lost from a given thinning treatment can be highly variable, and the small number of studies makes it difficult to draw strong generalities. The amount of shade lost can be smaller than the amount of tree basal area removed, and in one study, removal of 10 to 20 percent of the basal area had no measureable effect on angular canopy density (Kreutzweiser et al. 2009). Further, any shade loss and stream-temperature increases from riparian thinning are likely to be short lived because riparian forest canopies can close relatively quickly (within 3 years) after thinning (Chan et al. 2006, Yeung et al. 2017). The potential magnitude of stream-temperature increases in response to riparian thinning will be highly dependent on forest attributes outside the riparian buffer, the buffer size, the prethinned riparian forest attributes (Leinenbach et al. 2013), the thinning prescription, and the thermal sensitivity of the stream (Janisch et al. 2012). Further research is needed to improve our understanding of the impacts of thinning, but there is some evidence that light thinning may not substantially increase stream temperatures.

Managers thus face the following question: Are there places in the stream network in which riparian thinning would help speed attainment of the reference distribution, and where present-day thermal regimes would suggest that small temperature increases would not have significant detrimental effects on fish (Groom et al. 2011) or other organisms of interest? This question tends to be investigated at the reach scale. For example, Pollock et al. (2012) examined the potential effects of a thinning treatment on the development of riparian forest-stand attributes, and Groom et al. (2011) looked at summer maximum temperatures in the treated reach. Rarely are these questions expanded to consider the context of the distribution of reference conditions across the larger watershed. If they were asked, the question would then become: Are the conditions of the treated reach overrepresented with respect to the reference distribution, or underrepresented? In the Oregon Coast Range, for example, it is clear that not all reaches would be maintained in conifer-dominated mature forest under a natural disturbance regime (Wondzell et al. 2012). If dense, young, conifer-dominated stands are currently more abundant than expected from the reference distribution, then should some of those stands be thinned, perhaps mimicking windthrow events that open stand canopies and allow development of multistoried, mixed stands? If so, how many should be treated to better change the long-term trajectory of conditions from the current distribution toward one that is closer to the reference distribution?

Riparian thinning and large wood—The absence or diminished quantity of wood in streams throughout the NWFP area is a primary concern for managers and regulators because wood is important for creating habitat and performing other ecological functions. Thinning and other active management in plantations in riparian zones can reduce the potential amount of wood that can be delivered to streams (Beechie et al. 2000, Pollock et al. 2012) and the forest floor (Pollock and Beechie 2014, Pollock et al. 2012) if the trees are removed from the site. Additionally, thinning may negatively affect habitat, at least in the short run, for some species that are favored by dense conifer cover (see chapter 3 for more details), potentially increase water temperature (Leinenbach et al. 2013), and reduce carbon storage (D'Amore et al. 2015). However, there are also many potential benefits to thinning, including increasing structural diversity, species richness, flowering and fruiting of understory shrubs and herbs (Burton et al. 2014, Carey 2003, Hagar et al. 1996, Muir et al. 2002), and faster development of mature-forest conditions, including very large trees with thick limbs that may be used for nesting by marbled murrelets (*Brachyramphus marmoratus*) (Carey and Curtis 1996, Franklin et al. 2002, Tappeiner et al. 1997) (see chapter 5). Furthermore, variable density thinning of the overstory in the second-growth riparian forest could accelerate recovery of old-growth characteristics by promoting dominance of redwood in the southern portion of the NWFP area (Keyes and Teraoka 2014) (see chapter 3).

Considerable research on wood dynamics in the NWFP has been done in wet forests of California, Oregon, and Washington, but there has generally been less research in areas with drier forest types, including northern California. Riparian areas in redwood-dominated forests are particularly distinctive owing to the exceptional productivity, low mortality, and slow decay of those trees (Benda et al. 2002). Benda and Bigelow (2014) compared wood volumes across four different regions of northern California, including the Coast Range, Klamath Mountains, Cascade Range, and Sierra Nevada, as well as variation associated with forest management. They noted that coastal streams had much greater wood volumes, which they attributed to greater forest biomass and higher growth rates of redwood forests, as well as slower decay of large wood pieces. They also observed that some second-growth forests along streams in that region had wood volumes comparable to those in old-growth forests, owing to heavy debris remaining from tractor-era logging before the 1970s. Although the volumes were similar, streams in old-growth areas had fewer but larger logs (Benda et al. 2002). Benda and Bigelow (2014) also found that streams in the Cascades and Sierras that they characterized as more heavily managed had larger volumes of stream wood than less intensively managed areas in the same regions. They conjectured that managed forests could have higher rates of tree mortality because of stem exclusion (successional phase characterized by the rapid growth and biomass accumulations of a particular species, and intense competition among cohorts [Oliver 1981]) than more mature, but not yet decadent, unmanaged forests with lower tree densities.

A panel of scientists from the Forest Service and the National Marine Fisheries Service recently reviewed the published literature on the effects of thinning in riparian areas (Spies et al. 2013). Their major conclusions are summarized below:

- Accurate assessment of thinning effects requires site-specific information. The effects of thinning regimes on dead-wood creation and recruitment (relative to no thinning) will depend on many factors, including initial stand conditions, particularly stand density, and thinning prescriptions.
- Conventional thinning generally produces fewer large dead trees. Thinning with removal of trees (conventional thinning) will generally produce fewer large dead trees across a range of sizes over the several decades following thinning and the lifetime of the stand relative to equivalent stands that are not thinned.
- Thinning to develop old-growth structure is most beneficial in dense young stands less than 80 years old and especially those less than 50 years old.
- Conventional thinning can accelerate the development of very large diameter trees. In stands that are conventionally thinned, the appearance of very large diameter dead trees (greater than 40 in [102 cm]) may be accelerated by up to 20 years relative to unthinned plantations, depending on thinning intensity and initial stand conditions.
- To produce down wood immediately, thinning can leave trees that are cut as part of the restoration program (see Benda et al. 2016 for details).
- Thinning can increase the amount of pool-forming wood only when the thinned trees are larger in diameter than the average diameter of pool-forming wood (which varies with stream size).
- Effects of thinning on instream wood need to be placed in a watershed context. Assessing the relative effects of riparian thinning on instream wood loads at a site and over the long term requires an estimation of the likely wood recruitment that will occur from both the banks and downstream movement from upstream sources, and the rate of decay and downstream transport of wood from the site.
- The ecological effects of thinning on instream habitat will vary depending upon location in the stream network. Riparian-management practices can be altered to match the ecological functions of streams.
- Variation in thinning is essential to increase species diversity and heterogeneity (i.e., do not use the same prescription everywhere).

Since Spies et al. (2013) summarized the state of the science, other studies have increased our understanding of the effect of restoration thinning in riparian areas. Benda et al. (2015) simulated the idea of adding wood to channels during thinning by modeling the amount of instream wood that would result from thinning a 50- to 80-year-old Douglas-fir stand from below (i.e., removing the smallest trees to simulate suppression mortality) from 400 to 90 trees/ac (988 to 222 trees/ha), which is considered a moderate amount of thinning, then directionally falling or pulling over varying proportions of the harvested trees into the stream (table 7-4). This wood loading was compared to the amount that would be expected in the stream if the existing stand was not thinned. Not surprisingly, the amount of wood increased above the "no-thin" level immediately after the tipping simulation in all the wood-addition options. However, the cumulative total amount of wood expected in the stream over 100 years relative to the unthinned stand varied depending on the amount of wood delivered. Adding ≤10 percent of the wood that would be removed during thinning resulted in less wood in the channel over time than the unthinned option (i.e., if the stand were not actively managed). When 15 to 20 percent of the volume of thinned trees from one side of the stream was directed to the stream at each entry, the total amount of dead wood in the channel exceeded the unthinned scenario over time (table 7-4). Thinning the stand again 25 years after the first thinning further increased wood levels (table 7-4). Carah et al. (2014) found that adding unanchored wood into the stream was less costly than securing the wood, and improved habitat conditions for coho salmon. Reeves et al. (2016a) included wood addition (tree-tipping) as a component of options for managing the riparian reserves on Oregon and California Railroad Revested lands of the BLM in western Oregon to accelerate attainment of restoration objectives.

***Ecological tradeoffs*—**There are potential ecological consequences of limiting tree harvest (thinning) only to the outer portions of the riparian reserves. A myriad of ecological processes create and maintain the freshwater habitats of Pacific salmon (Bisson et al. 1997, 2009) and the ecological context in which they evolved (Frissell et al. 1997). This is especially relevant to the goals of the ACS, which are broad and include more than aquatic conditions. Holling and Meffe (1996) contended that uniform management prescriptions often fail when applied to situations in which processes are complex, nonlinear, and poorly understood, such as in aquatic ecosystems in the NWFP area, and may lead to further degradation or compromising of the ecosystems and landscapes of interest (Dale et al. 2000, Hiers et al. 2016, Rieman et al. 2006). For example, managing for a single purpose (e.g., maximizing dead wood) may compromise or retard other ecological functions, such as development of hardwoods and shrubs or growing large trees, in areas near the stream (see previous discussion), and ultimately may alter the structure of the food web (Bellmore et al. 2013). Pollock and Beechie (2014) stated that "species that utilize large-diameter live trees will benefit most from heavy thinning, whereas species that utilize large-diameter deadwood will benefit most from light or no thinning. Because far more vertebrate species utilize large deadwood rather than large live trees, allowing

Table 7-4—Predicted change in the estimated volume of wood in a stream channel under two different harvest options (single and double entry) over the simulated 100 hundred years (includes decay) when varying proportions of harvest wood are placed or felled into the stream channel

	Change from no treatment	
Scenario	**Single-entry thin**	**Double-entry thin (25 years after first entry)**
	Percent	
No treatment (reference)	0	0
Thin entire stand, no tipping	-33	-42
Thin entire stand except with a buffer 32.8 ft (10 m) no-harvest buffer	-7	-11
Thin with a buffer 32.8 ft (10 m) and tip 5 percent of the harvested wood	-15	-15
Thin with a buffer 32.8 ft (10 m) and tip 10 percent of the harvested wood	-6	+1
Thin with a buffer 32.8 ft (10 m) and tip 15 percent of the harvested wood	+1	+16
Thin with a buffer 32.8 ft (10 m) and tip 20 percent of the harvested wood	+6	+24

Source: Benda et al. 2016.

riparian forests to naturally develop may result in the most rapid and sustained development of structural features." We agree that tradeoffs exist and that prioritization will be needed (see chapter 12 for more discussion of tradeoffs).

The choice of priority conservation targets (e.g., dead wood, plant-community diversity, large live trees, geomorphic disturbances) for riparian management is a difficult one to make, involving scientific criteria, risk assessment, and social values. Given the diversity of conditions in riparian areas at watershed and regional scales, it would make sense not to apply one-size-fits-all strategies, but rather to develop priorities based on a watershed-scale view (see "A context-dependent approach to riparian conservation and management" below). For example, Pollock and Beechie (2014) stated that "management strategies that seek to create a range of large live and dead tree densities across the landscape will help to hedge against uncertain outcomes related to unanticipated disturbances, unexpected species needs, and unknown errors in model assumptions." It will be important to consider the full suite of ecological functions across a watershed; focusing only on one condition or metric may limit recovery of riparian ecosystems in ways that prevent full achievement of the broad objectives of the ACS. Given these broad objectives, a more comprehensive watershed- and regional-scale consideration of all ecological processes, and studies to develop new and more complete approaches, may be more fruitful than focusing on only one or two metrics.

A context-dependent approach to riparian conservation and management—

A key component of the ACS is watershed analysis (FEMAT 1993), which is supposed to provide the context of a given location for adjusting the boundaries of, and allowing activities within, riparian reserves. However, the intent of watershed analysis was never realized (Reeves et al. 2006), owing to a number of factors including cost of analysis and the need to consider a multitude of species and their ecological requirements. Neither FEMAT (1993) nor the NWFP (USDA and USDI 1994a) provided explicit criteria for changing the riparian reserve boundaries or demonstrating that proposed changes would meet or not prevent attainment of ACS objectives over the long term. In addition, at the time, credible analytical tools to aid decisionmaking were lacking (Reeves 2006); a fixed-width approach is easy to administer and apply and is less costly than developing site-specific recommendations (Richardson et al. 2012). As a result, adjustments have proven difficult for the agencies to make (Naylor et al. 2012, Richardson et al. 2012), and interim boundaries of the riparian reserves remained intact in the vast majority of watersheds (Baker et al. 2006).

Since the development of the ACS, there has been a call in the scientific literature to allow discretion in setting site-specific activities (Kuglerová et al. 2014, Lee et al. 2004, Richardson et al. 2012), which can be economically beneficial (Tiwari et al. 2016). Greater flexibility in the management of riparian areas would depend on the "context" of the area of interest (Kondolf et al. 2006, Montgomery 2004), and the primary management objective for the specific area (Burnett and Miller 2007). However, development of such an approach has been limited because of the reliance on "off-the-shelf" and one-size-fits-all concepts and designs, rather than on an understanding of specific features and capabilities of the location of interest (Kondolf et al. 2003, Naiman et al. 2012). A mix of approaches could be undertaken, recognizing ecological and other goals such as timber harvest, especially if applied over larger spatial scales (Burnett and Miller 2007, Miller and Burnett 2008, Olson and Rugger 2007), and if consideration is given to the distribution of populations of concern and connectivity among them (Olson and Burnett 2009, Olson and Kluber 2014, Olson et al. 2007).

There have been a few attempts to design and implement a site-specific approach. Cissel et al. (1999) proposed a plan based on variation in the disturbance patterns (in this case, wildfire) in the target watershed, and called for harvest of some older trees and a revision of the interim riparian reserves for the Central Cascades Adaptive Management Area. Olson and Rugger (2007) proposed a two-tiered approach to riparian management to first identify reaches in which sensitive species occur, then manage their critical habitat elements, hence varying riparian reserve management with species distributions. Olson and Burnett (2009) applied sensitive-species filters to criteria for designations of habitats for connectivity within and among watersheds. Interwatershed connections provided by riparian areas are critical avenues of movement to new habitats. None of these approaches have been implemented to date.

Reeves et al. (2016a) proposed a context-dependent approach for management of the riparian reserves in the matrix of federal lands in western Oregon that divided the riparian reserve into inner and outer zones, with management tailored to the specific features and characteristics of individual stream reaches (Option B of Reeves et al. 2016a). The context-dependent option was informed by new research, tools, and concepts, including:

- The influence of the width of riparian area on microclimate (see earlier discussion).
- Movement of amphibians along non-fish-bearing streams (Olson and Burton 2014, Olson et al. 2007).
- The distance to, and sources of, wood for fish-bearing streams (Spies et al. 2013).
- Intrinsic potential, a concept for assessing the capability of a given set of geomorphic conditions in a stream reach to provide habitat for selected species of Pacific salmon (Burnett et al. 2007).
- NetMap (Benda et al. 2007), a geospatial platform for watershed analysis that can, among other things, identify the location of some key ecological processes that influence aquatic and riparian ecosystems on the landscape and in the stream network.
- Concepts for managing riparian ecosystems and the activities that affect them, such as ecological forestry (Franklin and Johnson 2012) and tree-tipping (Benda et al. 2016).

Under the context-dependent option, current interim riparian reserves of two site-potential tree-heights along fish-bearing streams and one site-potential tree-height along non-fish-bearing streams would be retained in late-successional reserves and other special land designations (Reeves et al. 2016a). In lands allocated as matrix under the NWFP, the area of interest for aquatic conservation (Reeves et al. [2016a] referred to this as the riparian conservation area) extended upslope from the stream for a distance equal to the height of one site-potential tree along fish-bearing and non-fish-bearing streams. The riparian conservation area was divided into an inner and an outer zone depending on "ecological context," based on four characteristics of each stream reach—susceptibility to surface erosion, debris flows, thermal loading, and habitat potential for target fish species—to determine the width of the inner zone. The entire riparian conservation area of the most ecologically sensitive stream reaches along fish-bearing and non-fish-bearing streams could be managed solely for ecological goals for fish and other aquatic and riparian-dependent biota. In other fish-bearing and non-fish-bearing streams, the inner zone was 100 ft (30. 5 m) and 50 ft (15.3 m) wide, respectively (Reeves et al. 2016a). Active management was limited to stands age 80 years or younger in reserves (Spies et al. 2013), and tree-tipping (Benda et al. 2016) was used throughout the riparian reserve to ensure that harvest did not negatively affect wood recruitment to the stream (table 7-4).

Using the matrix in BLM-managed lands in western Oregon to illustrate the application, Reeves et al. (2016a) estimated that an average of 46 percent of the riparian reserve in a watershed would be managed solely for ACS goals. Also, an estimated average of 36 percent would achieve ACS goals along with other potential goals, which could include timber production, and 18 percent could be managed for a variety of purposes, including wildlife and timber, in accordance with NWFP requirements (Reeves et al. 2016a). In late-successional and other reserve allocations, which cover approximately half of the BLM lands in western Oregon, interim riparian reserves would remain unchanged. Assuming that half of the interim riparian reserves on BLM lands in western Oregon would remain unchanged, and applying their study estimates of changes in matrix, Reeves et al. (2016a) estimated that about 72 percent of the interim riparian reserves would remain solely devoted to ACS goals, and 19 percent would likely meet ACS goals and could also provide opportunity for achievement of matrix goals including limited timber production. The reduction of the width of the riparian reserve along fish-bearing streams to one tree-height would return an estimated 9 percent of interim riparian reserves to matrix on these lands.

The analysis of Reeves et al. (2016a) was not intended to provide a single recommendation for managing riparian ecosystems. The primary purpose was to reevaluate riparian-conservation strategies using the latest scientific evidence. This or other options should be viewed as working hypotheses to be tested with monitoring and adaptive-management experiments. The analysis provides an example of how a context- and landscape-dependent approach could be designed to address multiple conservation goals of the ACS, the commodity goals of the NWFP, and the significant challenges of climate change. Although new science has refined our understanding of the ecological processes in riparian ecosystems, uncertainties and information needs remain. Thus, an adaptive-management approach and further research are critical to continual improvement and evaluation of this and other options for meeting the goals of the ACS (Stankey et al. 2005).

Key Watersheds

Tier 1 key watersheds (a total of 141, covering 8,154,500 ac (3 300 000 ha) (fig. 7-10) were intended to serve as refugia for aquatic organisms or to have high potential for restoration (USDA and USDI 1994a). Tier 2 key watersheds provide sources of high-quality water, and comprised 23 watersheds covering a total of about 1,112,000 ac (405 000 ha) (fig. 7-10). Key watersheds are aligned as closely as possible with the late-successional reserves of the NWFP (areas designated to protect late-successional and old-growth ecosystems) and other officially designated reserve areas to maximize ecological efficiency (USDA and USDI 1994a), and to minimize the amount of area in which timber-harvest activities were restricted. A primary objective for tier 1 key watersheds is to aid in the recovery of ESA-listed fishes, particularly in the short term (FEMAT 1993). Tier 1 key watersheds in good condition at the time of the Plan's inception were assumed to serve as centers for potential recovery of depressed populations. Those with degraded conditions were expected to have the greatest potential for restoration and to become future sources of good habitat.

The trend in the condition of key watersheds differed among assessments. Gallo et al. (2005) reported that a greater proportion of the key watersheds had their condition scores improve than did non-key watersheds. Lanigan et al. (2012) found key watersheds to be in better condition than

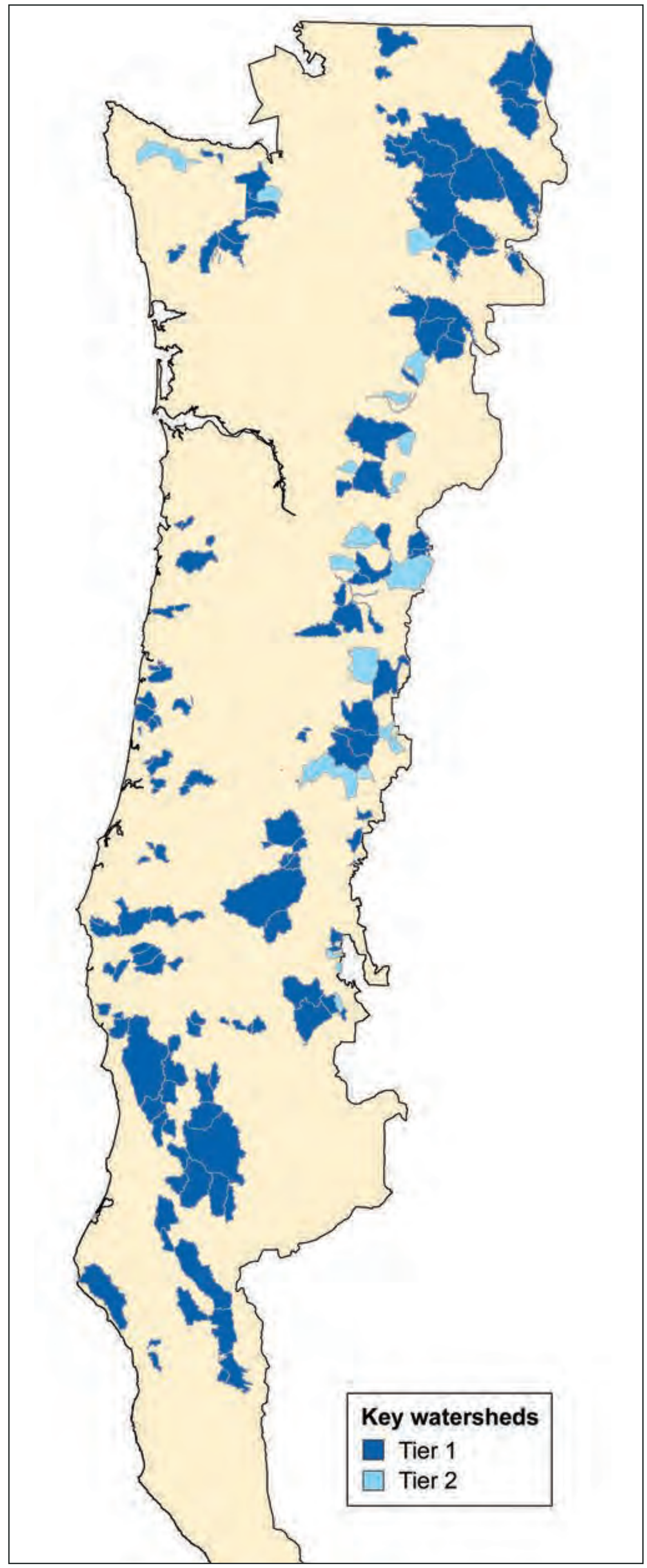

Figure 7-10—Location of key watersheds in the Northwest Forest Plan area.

non-key watersheds, primarily because more than twice as many miles of roads were decommissioned in key watersheds as in non-key watersheds (Gallo et al. 2005, Lanigan et al. 2012), suggesting that land management agencies appear to recognize key watersheds as priority areas for restoration. Miller et al. (2017), however, saw no statistical differences between the two groups.

Key watersheds were originally selected based on the professional judgment of the scientists involved with the development of the ACS, in consultation with fish and aquatic biologists and hydrologists from the national forests and BLM districts covered by the NWFP. Also, they were tightly aligned with late-successional/old-growth reserves, based in part on the assumption that streams in old-growth forests would be most favorable for fish (FEMAT 1993). New techniques (e.g., NetMap, Benda et al. 2007) and understanding of aquatic ecosystems now provide a different perspective on aquatic ecosystems and how they operate in space and time.

New concepts such as intrinsic potential of fish habitat (Burnett et al. 2007), projections of climate change, and new questions as to whether stream conditions in old-growth forests are actually most favorable for native salmonids (Bisson et al. 2009, Reeves and Bisson 2009, Reeves et al. 1995) are pivotal concepts that reframe our understanding of aquatic ecology and ecosystems. No formal evaluation of the potential effectiveness of the network of key watersheds was conducted during development of the NWFP, or has been undertaken since it was implemented. Fish populations in need of attention are clearly identified now, and it would be useful to investigate whether the current system is beneficial to those fish in terms of the overall distribution and the suitability of individual watersheds. Additionally, the distribution of other sensitive aquatic-riparian species (e.g., ESA-listed or petitioned herpetofauna) could contribute to this assessment.

Watershed Analysis

Watershed analysis was designed to provide the context for management activities in a particular sixth-field watershed as the basis for developing project-specific proposals and determining restoration needs. It was envisioned as an analytical and not a decisionmaking process, involving individuals from a variety of scientific disciplines (USDA and USDI 1994a). Management agencies were expected to complete a watershed analysis before activities (other than minor ones) were initiated in key watersheds or riparian reserves (USDA and USDI 1994a). The version of watershed analysis advocated in the NWFP differs from previous versions (e.g., Washington Forest Practices Board 1993) and involves multiple disciplines and issues other than those that are specifically aquatic.

Baker et al. (2006) estimated that about 500 watershed analyses had been done by 2003, but that the quality and effectiveness of these analyses differed widely. No formal assessment of watershed analyses has been completed as of this writing, so it would be prudent to conduct a comprehensive review and evaluation, and consider incorporating new analytical tools such as NetMap (Benda et al. 2007) to help improve the process and reduce costs while increasing the usefulness of the product. The watershed analysis process could also be reexamined so that it is conducted more efficiently and considers the appropriate spatial scales, including the watershed of interest and its context within the larger basin or region. The latter could be particularly relevant for effective planning at a landscape scale and to deal with climate change.

New Perspectives on Conservation of Riparian and Aquatic Ecosystems

The ACS was premised on the view that aquatic ecosystems were dynamic in space and time, exhibiting a range of potential conditions, similar to the terrestrial systems in which they are embedded (FEMAT 1993). Aquatic ecosystems in Pacific Northwest forests are multifaceted and complex, and can be conceptualized as a set of ecological states (Penaluna et al. 2016, Reeves et al. 1995, Rieman et al. 2015), each with particular abiotic and biotic conditions, functions, and processes at any given time. The number and variety of ecological states in a domain (i.e., the range of conditions or range of natural variability for an ecosystem) is in constant flux in response to changes in local conditions, stochastic processes, legacies of past disturbance, and time since past disturbance (Beechie et al. 2010; Benda

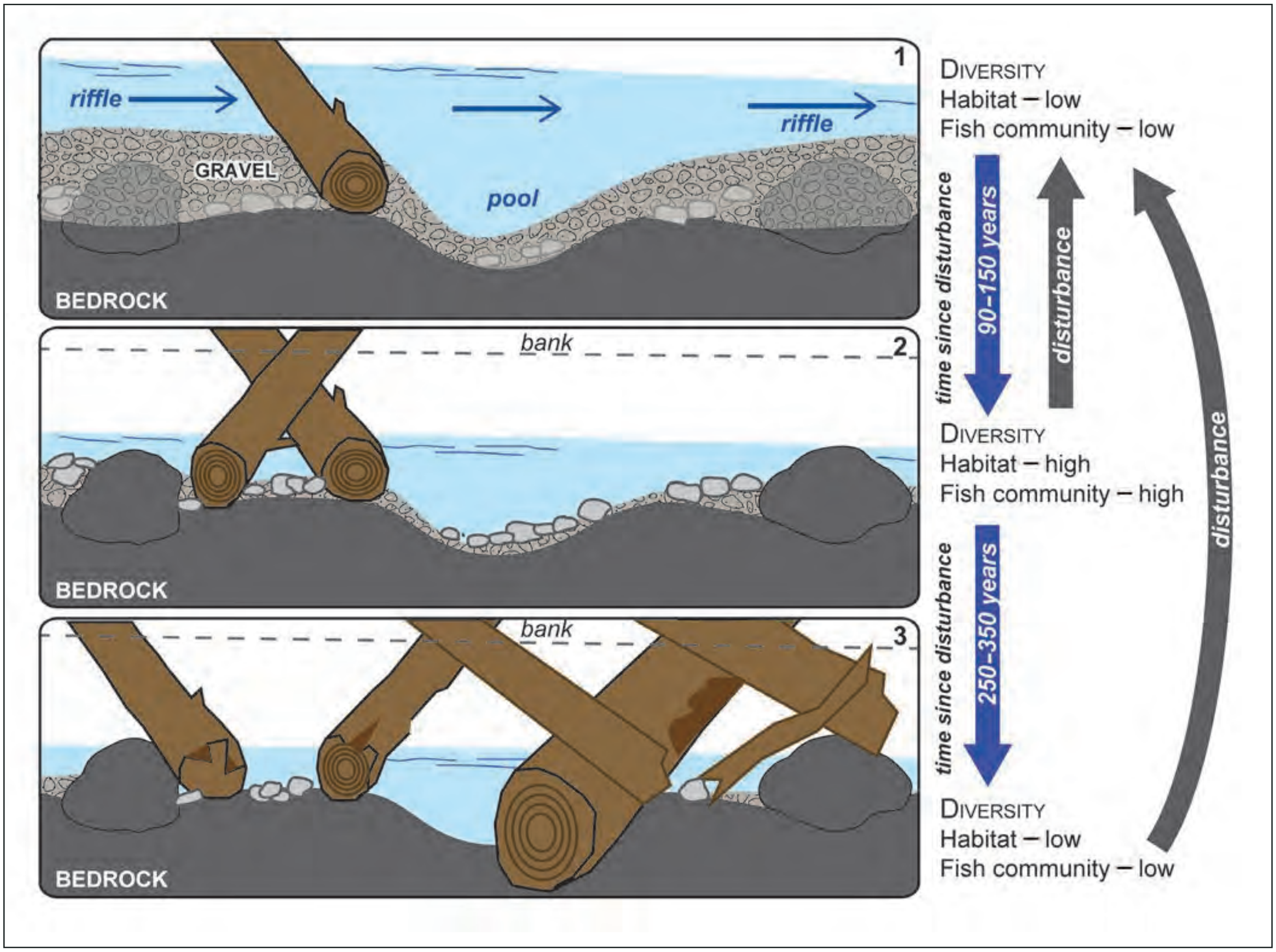

Figure 7-11A—Examples of the range of conditions that aquatic ecosystems experience: (1) a stream 90 to 100 years after the last disturbance, (2) 160 to 180 years, and (3) more than 330 years (Reeves et al. 1995) (see table 5 for specific details).

et al. 1998; Liss et al. 2006; Miller et al. 2003; Reeves et al. 1995; Resh et al. 1988; Rieman et al. 2006, 2015; Wondzell et al. 2007). Examples of the variation that aquatic ecosystems can experience through time are shown for the central Oregon Coast Range (Reeves et al. 1995) (fig. 7-11A and table 7-5), and eastern Oregon (Wondzell et al. 2007) (fig. 7-11B). Larger streams and rivers in the lower portion of the network are less variable through time; those in the upper and middle portions are more dynamic (Naiman et al. 1992). Because of the variation in the size and asynchronous nature of disturbance events (Allen et al. 1982, Malard et al. 2002, Schindler et al. 2010, Wiens 2002), conditions will vary over time among watersheds, resulting in a mosaic of biophysical conditions across the landscape. Unmanaged and minimally disturbed aquatic systems may actually exhibit a wider range of conditions than more heavily managed systems (Lisle 2002, Lisle et al. 2007).

A contrasting view holds that aquatic ecosystems tend to be in an equilibrium or steady state, and when disturbed, they are expected to return to predisturbance conditions relatively quickly (Resh et al. 1988, Swanson et al. 1988). Biological (Vannote et al. 1980) and physical conditions (Rosgen 1994) are presumed to be relatively constant through time and to be "good" (barring human interference) in all systems at the same time. Conditions in aquatic systems with little or no human influence and natural

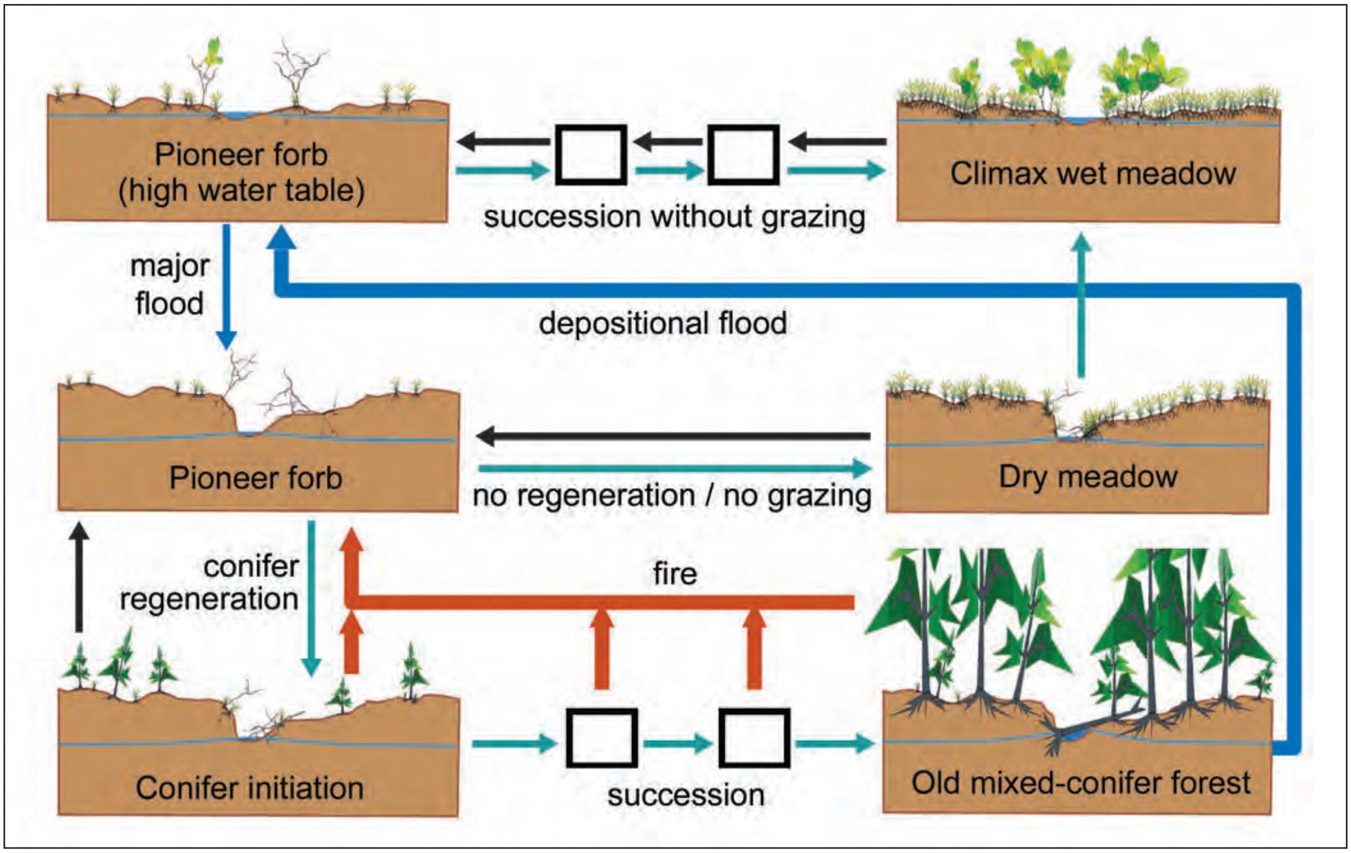

Figure 7-11B—Examples of the range of conditions that aquatic ecosystems in eastern Oregon can experience through time (Wondzell et al. 2007).

Table 7-5—Features of streams from the Oregon Coast Range used in figure 7-11A

Feature/stream	Harvey Creek (1)	Franklin Creek (2)	Skate Creek (3)
Time since disturbance (years)	90–100	160–180	More than 330
Number of pieces of wood/100 m	7.9	12.3	23.5
Mean depth of pools (m)	0.9	0.35	0.1
Dominant substrate	Gravel	Gravel	Bedrock
Percent of fish assemblage juvenile coho salmon	98.0	85.0	100.0
Percent of fish assemblage juvenile steelhead	1.0	12.5	0
Percent of fish assemblage juvenile cutthroat trout	1.0	2.5	0

Source: Reeves et al. 1995.

disturbance, particularly those associated with late-successional and old-growth forest, are assumed to have the most favorable conditions for fish (Fox and Bolton 2007, Murphy and Koski 1989, Pollock and Beechie 2014, Pollock et al. 2012) and other aquatic organisms, and are most frequently used as references against which the condition of managed streams (e.g., Index of Biotic Integrity) (Karr and Chu 1998) and effects of management actions can be assessed. Systems experiencing disturbances, such as wildfire or floods, are often immediately “restored” by attempting to reduce or eliminate erosional processes. For example, fences were placed in headwater streams following a wildfire in Colorado to reduce the potential for erosion and debris flows (Chin et al. 2014). However, reaches lower in the stream network downcut, creating other concerns. Although this static ecosystem view is being questioned in the general ecological literature (Hiers et al. 2016, Jackson et al. 2009, Montgomery 1999, White and Jentsch 2001, Wohl et al. 2014), it is still being used to guide management and assess effects on aquatic ecosystems, and persists in environmental laws and policies developed in the 1970s, such as the Clean Water Act (Craig 2010).

Resilience is the capacity of an ecosystem to absorb change and remain within the ecosystem state and domain in the face of natural disturbances and human stressors (Desjardins et al. 2015). As ecosystems undergo larger shifts from human stressors, the ecosystem can be redefined, with a completely different set of characteristics and a compromised or altered range of conditions (Bisson et al. 2009, Reeves et al. 1995). Some ecosystem components may persist through this transition, whereas others may be new components arising from human or climatic alterations, including the development of novel states that may result in the loss of selected ecosystem services and conditions for at least some native species (Penaluna et al. 2016).

The physical aspect of these dynamics is understood conceptually (see review in Buffington 2012), but few mechanistic models currently exist to help us understand the potential effects of management on dynamic ecosystems (but see Wondzell et al. 2007), especially under climate change. As a result, consideration of dynamics remains largely conceptual, and holistic models of basin function (i.e., watershed analyses) are generally lacking, limiting the development of process-based applications of river management and restoration (Beechie et al. 2010). Also, there is also a tendency to focus on mean or median conditions while overlooking temporal variability as "noise" and losing sight of the considerable inherent variability that characterizes riparian and aquatic ecosystems (Fausch et al. 2002, Montgomery 1999), which is ecologically critical (Hiers et al. 2016). Accounting for this variability and for nonstationarity of fluvial processes is central to assessing potential effects of climate change on riverine ecosystems (Buffington 2012, Miller et al. 2003, Montgomery 1999). However, being able to incorporate this variability into restoration and mitigation actions may be limited by social concerns (Kondolf et al. 2006) (see chapter 12).

Consideration of large spatial and temporal scales is critical to the development of management and conservation strategies for ecosystems (Dale et al. 2000, Holling and Meffe 1996), including a range of conditions for aquatic ecosystems (Fausch 2010, Fausch et al. 2002, IMST 1999, Liss et al. 2006, NRC 1996). This shift requires moving from the current focus on relatively small spatial scales, with little or no consideration of the relevance of time, to a focus that considers large spatial scales, specifically ecosystems and landscapes, over relatively long periods (tens to hundreds of years) (Bisson et al. 2009, Naiman and Latterell 2005, Poff et al. 1997, Reeves et al. 1995). An example of the importance of relations between scales can be seen in the "portfolio effect" of the behavior of populations of sockeye salmon (*O. nerka*) in Bristol Bay, Alaska, identified by Schindler et al. (2010). This study found large variation in the number of fish in any local population over time. However, the variation among the many populations was asynchronous—not all were high or low at the same time. As a result, the total number of fish was relatively constant at the landscape scale, a pattern similar to the amount of old growth historically found in the Oregon Coast Range (Wimberly et al. 2000). This pattern appears to be disrupted in heavily managed systems (Moore et al. 2010).

Both the NWFP and new Forest Service planning rule (USDA FS 2012a) require managers to consider large spatial scales in designing, implementing, and evaluating management actions. The new planning rule also emphasizes ecosystem goals based on ecological integrity. This can be daunting given the lack of scientifically sound examples of how to design and implement forest management at large temporal and spatial scales (North and Keeton 2008, Reeves and Duncan 2009, Thompson et al. 2009) and the lack of adequate tools and guidance. Shifting the management focus to the landscape level and longer time intervals requires recognition of the principles of hierarchy theory and the relation among levels of organization to increase the potential for success of future riparian policies and practices (Fausch 2010, Fausch et al. 2002).

Regulators may recognize the need to apply policies and regulations across broad areas, but may be constrained by the regulatory framework in which they are operating, and generally default to single standards that are applied across broad areas (e.g., National Marine Fisheries Service's matrix of pathway and indicators) (NMFS 1999). This premise is inappropriate for addressing complex, multifaceted landscapes, however (Allen and Starr 1982; O'Neill et al. 1986, 1989); instead, it is important to recognize that a multiwatershed landscape operates differently through time than does a single watershed, and that smaller spatial scales tend to be more variable over time than larger scales (Benda et al. 1998, Wimberly et al. 2000). Increasing levels of aggregation, especially as spatial scales increase, may obscure important system processes (Clark and Avery 1976) and may result in unrealistic expectations for ecosystems and contribute to the contention that often surrounds large-scale management proposals (Allan and Curtis 2005, O'Neill et al. 1986, Shindler et al. 2002). Also, the failure to recognize the different levels of ecological organization and the potential response of each to component parts of disturbance and management may incur unintended economic and social costs, such as repeated investment in ineffective restoration and management strategies (Caraher et al. 1999, Dale et al. 2000).

The emerging consideration of ecosystem dynamics and large spatial and temporal scales has implications for approaches to restoration of aquatic-riparian ecosystems. Many restoration efforts have focused mainly on improving habitat attributes, primarily wood placement, and to a lesser degree on shade improvement for water temperature. These efforts too often aim to bring "stability" to degraded systems, and are viewed as the final phase of restoration (see Palmer et al. 2014). The dynamic approach, not yet broadly practiced, focuses on restoring ecological processes (Beechie et al. 2009, 2010; Bernhardt and Palmer 2011), including periodic inputs or reoccurrences of these important habitat attributes. This requires a shift from reliance on striving only to develop a particular condition (e.g., number of pieces of wood per unit length) or channel classification (e.g., Rosgen 1994) to a quantitative approach based on ecological processes, theory, empirical field methods, and limited modeling (Kline and Cahoon 2010, Wohl et al. 2005).

Some researchers have pointed out that although restoration of ecological processes, such as flow, water temperature, habitat complexity, and connectivity, is a critical consideration in restoring many streams, it may not be sufficient for degraded channels, and can even worsen the ecological condition of the stream (Louhi et al. 2011, Tullos et al. 2009). For example, in restoring floodplain overflow potential, if riparian trees are removed from a previously closed-canopy stream, the underlying energy regime may change from one based on allochthonous resources to one driven by primary production. This may shift the stream farther from the desired ecological state and often toward algae-dominated streambeds and higher temperatures (Robinson 2012, Sudduth et al. 2011). Similarly, if the hydrologic regime is restored but there is no nearby source of invertebrate colonists, then the instream communities will remain altered (Sundermann et al. 2011). Finally, an overreliance on an in-channel focus (small-scale) may not address the stressor(s) that most limit recovery of the aquatic ecosystem; quite often this factor is water quality, and thus ecological recovery will not occur until the stressor is addressed (Beechie et al. 2010, Kail et al. 2012, Selvakumar et al. 2010). Examples of process-focused restoration are presented below in the section on climate change.

In addition to considering spatial complexity, temporal dynamics are particularly important to understand because many key ecological processes such as canopy closure, tree-fall, and fuel loading are related to the age of trees in riparian areas as well as time since disturbance. Temporal dynamics can be examined using models, but long-term studies and monitoring are needed to understand how systems respond over time (Hassan et al. 2005). One strategy that may be appropriate is to design monitoring to focus more on changes following major disturbances rather than focusing simply on short-term trends.

The other challenge posed by a dynamic perspective of aquatic ecosystems is the consideration of large spatial scales. Restoration efforts are generally performed at small spatial scales, with only a relatively small percentage of any watershed actually treated (Ogston et al. 2014, Roni et al. 2010). Roni et al. (2010) estimated that a minimum of 20 percent of the habitat of a given species in a watershed should be restored to detect a 25 percent increase in smolt (salmon or steelhead) numbers, the minimum detection level for most monitoring programs. They found that floodplain restoration yielded greater increases than in-channel restoration. However, because of the large variability in numbers for most populations (Bisson et al. 1997, Schindler et al. 2010), Roni et al. (2010) suggested that 100 percent of the habitat should be restored to have a significant ecological impact.

Non-Fish-Bearing Streams

The ecological importance of headwater streams, which generally make up 70 percent or more of the stream network (Downing et al. 2012, Gomi et al. 2002), was not well known or understood at the time the ACS was developed, but it is now better established in the scientific literature (Leigh et al. 2016, Richardson and Danehy 2007). Headwaters are sources of sediment (Benda and Dunne 1997a, 1997b; May and Lee 2004; Zimmerman and Church 2001; see review by MacDonald and Coe 2007) and wood (Bigelow et al. 2007; May and Gresswell 2003, 2004; Reeves et al. 2003) for fish-bearing streams; provide habitat (Kelsey and West 1998, Olson et al. 2007) (see chapter 6) and movement corridors (Olson and Burnett 2009, Olson and Kluber 2014) for several species of native amphibians and macroinvertebrates (Alexander et al. 2011, Meyer et al. 2007), including recently discovered species (Dieterich and Anderson 2000); and may be important sources of food for fish (Kiffney et al. 2000, Wipfli and Baxter 2010, Wipfli and Gregovich 2002, Wipfli et al. 2007; also see reviews by MacDonald and Coe 2007 and Clarke et al. 2008). Wood jams in small streams are important sites of carbon storage (Beckman and Wohl 2014), and these streams export large amounts of carbon; one-third is emitted to the atmosphere and the remainder transported downstream (Argerich et al. 2016).

Tributary junctions of headwater streams with larger channels are important nodes for regulating material flows (Benda et al. 2004, Gomi et al. 2002, Montgomery et al. 2003) and cold water (Ebersole et al. 2015) in a watershed, and are the locations where site-scale effects from management activities are often observed (Richardson and Béraud 2014). These locations have unique hydrologic, geomorphic, and biological attributes and differ in the types and amount of materials delivered to the channel, making them sites of high biodiversity (Benda et al. 2004, Danehy et al. 2012)

Headwater streams are among the most dynamic portions of aquatic ecosystems (Benda et al. 2005, Hassan et al. 2005, MacDonald and Coe 2007, Naiman et al. 1992). Headwater habitats may range from simple to complex, depending on the amount of time since disturbance (such as landslides and debris flows). Following evacuation by a debris flow, headwater depressions and channels fill with material from the surrounding hillslopes, including large wood that falls into these channels, forming obstructions behind which sediments and wood accumulate (Benda and Cundy 1990, May and Gresswell 2004), and then empty again with the next landslide or debris flow (fig. 7-12). As a result, headwater streams are likely to exhibit a range of conditions across the landscape at any point in time.

This cycle of filling and emptying results in a punctuated movement of sediment and wood to larger, fish-bearing streams (Benda et al. 1998, Naiman et al. 1992), contributing to the long-term productivity of many aquatic ecosystems (Benda et al. 2003, Hogan et al. 1998, Reeves et al. 1995). A common consequence of past clearcutting is an absence of down wood to replenish the refilling process. This lack of wood may result in a chronic movement of sediment to larger channels, which could lead to both non-fish-bearing and fish-bearing channels developing characteristics different from those that occurred before forest management. Such conditions could be outside the range of variability to which native biota are adapted (Beschta et al. 2004), limiting the effectiveness of conservation and recovery programs.

Wood enters streams via chronic and episodic processes (Bisson et al. 1987). Chronic processes, such as tree mortality and bank undercutting (Bilby and Bisson 1998, Murphy and Koski 1989), generally introduce single trees or a relatively small number of trees at frequent intervals. Wood from headwater streams, which originates from within 131 ft (40 m) of the channel (May and Gresswell 2003), is delivered to fish-bearing streams by large, infrequent events, such as windthrow (Harmon et al. 1986), wildfire (Agee 1993), severe floods, landslides, and debris flows (Benda et al. 2003; May and Gresswell 2003, 2004; Reeves

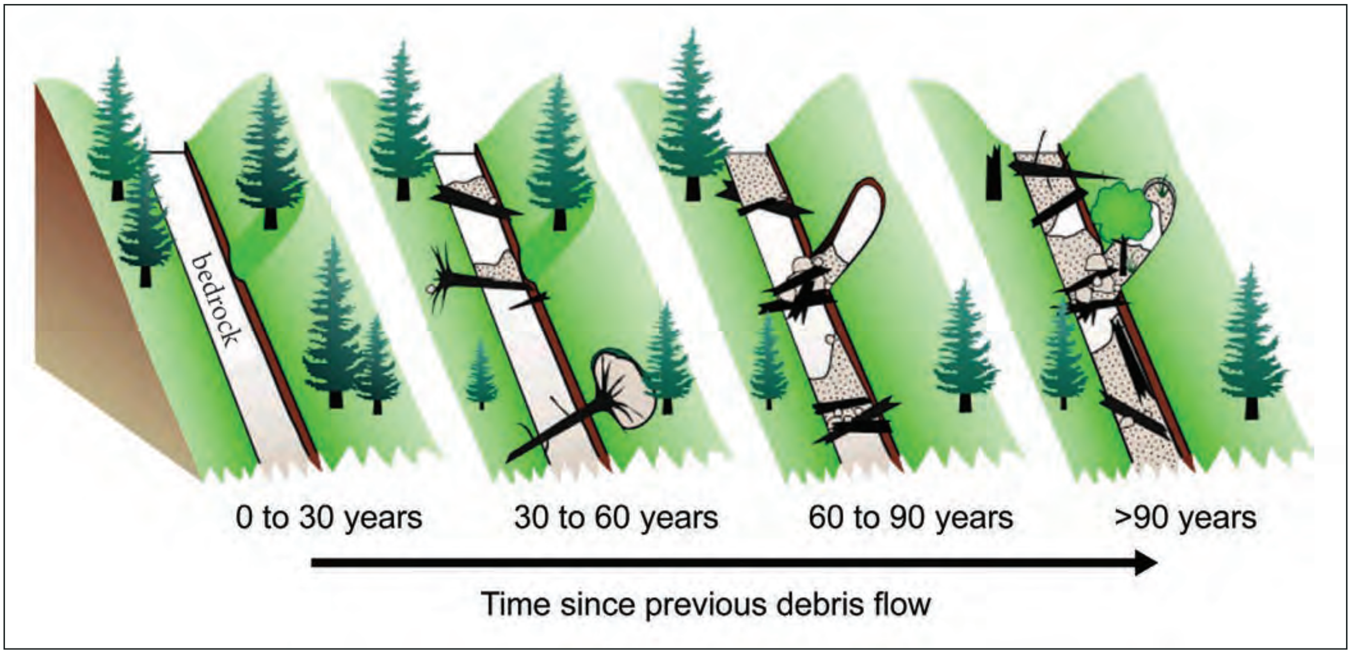

Figure 7-12—Conceptual illustration of the changes in channel morphology based on the time since the previous debris flow (May and Gresswell 2004).

et al. 2003). Geomorphic features of a watershed influence the potential contribution of upslope wood sources. Steeper, more highly dissected watersheds will likely have a greater proportion of wood coming from upslope sources than will watersheds with lower stream densities and gradients. Also, there is wide variation in the potential of headwater streams to deliver sediment and wood to fish-bearing streams, depending on channel steepness and angle of entry along the run-out track, among other factors (Benda and Dunne 1997a, 1997b; Brayshaw and Hassan 2009; Burnett and Miller 2007; May 2007). Culverts and other stream crossings can also impede wood movement from smaller to larger streams (Trombulak and Frissell 2000).

The presence of large wood from headwater streams influences the behavior of landslides and debris flows and the response of the channel to such events. Large wood in debris flows and landslides influences the run-out length of these disturbance events (Lancaster et al. 2003). Debris flows without large wood move faster and farther than those with wood, and they are less likely to stop high in the stream network. A debris flow without wood is likely to be a concentrated slurry of sediments of various sizes that can move at relatively high speeds over long distances, scouring substrate and wood from the affected channels. These types of debris flows are more likely to negatively affect fish-bearing channels, as compared to the potentially favorable effects that result from the presence of wood. Woodless debris flows can further delay or impede the development of favorable conditions for fish and other aquatic organisms. In contrast, those containing wood can help store sediments (Bunn and Montgomery 2004) and build terraces that can persist for extended periods (Lancaster and Casebeer 2007, May and Lee 2004).

Intermittent streams, which can make up half the total length of the stream network (Datry et al. 2014), connected to larger fish-bearing streams can provide important seasonal habitats for spawning and rearing by fish (Boughton et al. 2009, Wigington et al. 2006). In the Oregon Coast Range, growth and survival of juvenile coho salmon was higher in intermittent streams than the perennial mainstem (Ebersole et al. 2006, 2009; Hance et al. 2016). Identification, protection, and restoration of these streams is important to the success of conservation efforts for native fish across the NWFP area.

A rich non-fish community inhabits headwater streams throughout the NWFP area. For example, Olson and Weaver (2007) found 3 species of fish and 12 species of amphibians in stream reaches in 12 western Oregon study sites ranging from Mount Hood to Coos Bay. In this study and Olson and Burton (2014), torrent salamanders (*Rhyacotriton* spp.) dominate intermittent streams and appear to be sensitive to thinning in narrow riparian-management areas; NWFP riparian reserves appear to be benefiting retention of this aquatic-dependent community in abundant small streams in the region. Nevertheless, two torrent salamander species are currently petitioned for listing under the ESA; both have ranges that include significant tracts of nonfederal lands.

Continuing and Emerging Topics of Concern

Water

Federal lands are important sources of fresh water for human consumption, recreation, agriculture, and environmental needs in the United States. These lands produce an estimated 24.2 percent of the Nation's water supply, 18 percent and 1.5 percent from Forest Service and BLM lands, respectively (Brown et al. 2008). In the West,[5] federal lands contribute 66 percent of the mean annual water supply, 51 percent of which comes from Forest Service lands and 5.4 percent from BLM lands (Brown et al. 2008). Management strives to maintain the quality and quantity of this water. The extent of the contribution of federal lands to regional water supplies was not well quantified at the time the NWFP and ACS were developed.

The contribution of water from federal lands specifically in the NWFP area is also important; however, exact estimates are not currently available and were beyond the scope of this review. At the state level, the majority of water in the three NWFP states (California, Oregon, and Washington) originates from federal lands (table 7-6), with the bulk coming from Forest Service lands. Within the NFWP area, the amount of water flowing from federal lands varies among national forests and watersheds. Some forests, such as the Deschutes and Willamette National Forests, make relatively large contributions, 40 percent or more, to the flow of rivers whose watersheds they include (figs. 7-13A and 7-13B, respectively). Contributions from other forests are smaller (less than 20 percent) (fig. 7-13C) but nonetheless important. See "Climate Change" below for potential future issues pertaining to water supply and stream temperatures.

Roads

Roads provide necessary motorized access for forest management, recreation, and other beneficial purposes (Gucinski et al. 2001), but they can also have detrimental effects on native biodiversity and ecosystem function. The focus of the NWFP and ACS has been to address the negative effects of roads on aquatic ecosystems through a broad program of road decommissioning and upgrading, including remediation of chronic sedimentation and barriers to aquatic organism movement. Several syntheses describe the types, causes, and effects of road networks on streams, and meta-analyses concerning the ecological effects of

[5] The West is defined here as including the states of Arizona, California, Colorado, Idaho, Montana, New Mexico, Nevada, Oregon, Utah, Washington, and Wyoming.

Table 7-6—Contribution of federal lands and agencies to the total mean annual water supply of states in the Northwest Forest Plan area (percentage of mean annual water supply)

State	All federal lands	Forest Service	Bureau of Land Management	Other
		Percent		
California	61.1	46.6	5.5	9.0
Oregon	55.3	44.0	9.4	2.0
Washington	60.2	41.5	0	18.7

Source: Brown et al. 2008.

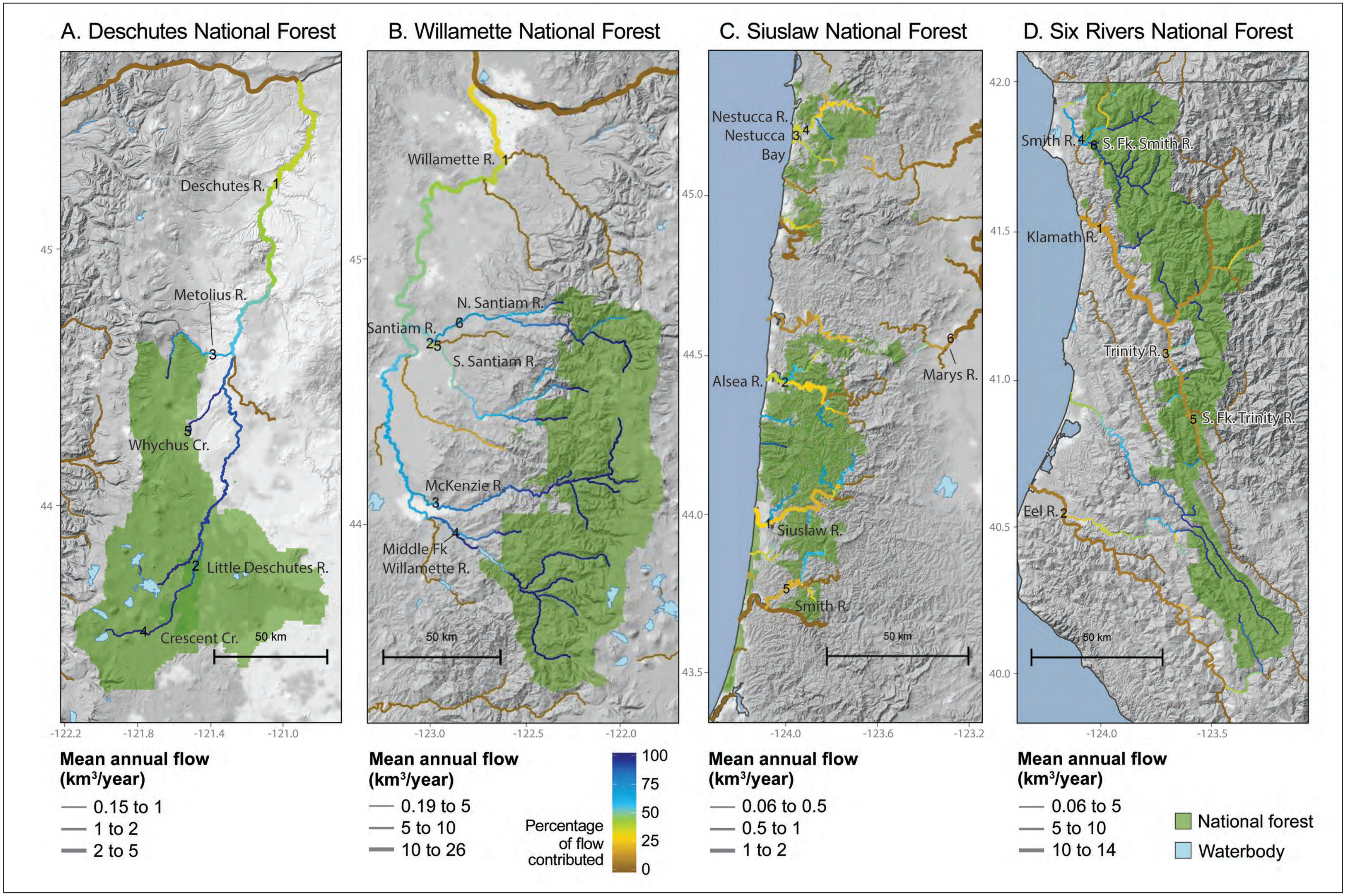

Figure 7-13—Contribution of selected national forests in the Northwest Forest Plan area to the mean annual flow of adjacent watersheds: (A) Deschutes National Forest, central Oregon Cascades; (B) Willamette National Forest, western Oregon Cascades; (C) Siuslaw National Forest, Oregon Coast Range; and (D) Six Rivers National Forest, coastal northern California. (https://www.fs.fed.us/rmrs/projects/national-forest-contributions-streamflow).

roads in general and specifically the delivery of sediment from mountain roads with low maintenance standards have been published (Croke and Hairsine 2006, Forman and Alexander 1998, Jones et al. 2000, Trombulak and Frissell 2000). Reducing the effect of roads and associated infrastructure remains a challenge for federal management agencies and others.

The vulnerability of roads to hydrologic changes and the associated effects on aquatic and riparian ecosystems differ based on topography, geology, slope stability, design, location, and use. Roads can affect streams directly by:

1. Accelerating erosion and increasing sediment loading (Allan 2004, Daigle 2010, MacDonald and Coe 2008, Suttle et al. 2004).
2. Imposing barriers to the migration of aquatic organisms, including access to floodplains and off-channel habitats (Clarkin et al. 2005, Daigle 2010, Gibson et al. 2005, Sagar 2004, Trombulak and Frissell 2000).
3. Increasing stream temperatures (Wenger et al. 2011).
4. Causing changes in channel morphology (Daigle 2010, Hassan et al. 2005).
5. Introducing exotic species (Daigle 2010, McKinney 2001).
6. Increasing harvest and poaching pressure (Lee et al. 1997, Trombulak and Frissell 2000).
7. Changing hillslope hydrology and resulting peak flows (Daigle 2010, Jones and Grant 2001).

Roads penetrating remote and otherwise intact forested landscapes can have particularly significant effects on aquatic ecosystems (Forman 2003, Havlick 2002, Trombulak and Frissell 2000). The ecological consequences of these effects are shown in table 7-7.

The effects of roads differ widely depending on local features (Al-Chokhachy et al. 2016). Recently developed techniques, such as the Geomorphic Roads Analysis and Inventory Package (Black et al. 2012), can be employed to identify priority locations of sources of sediment, including culvert failures, landslides, and gullies. A modified version of this technique has been incorporated into NetMap (Benda et al. 2007) to reduce the amount of field time needed to assess roads. Evaluating the effectiveness of new analytical approaches and focusing on treating limited lengths of roads could be a research priority.

A significant number of watershed-improvement actions implemented under the NWFP and other large-scale forest planning efforts involve decommissioning roads that have a high probability of contributing to landslides, and that are not regarded as essential to meeting local forest objectives, as well as removing road-related impediments to upstream and downstream movements of aquatic organisms (Switalski et al. 2004). The watershed-analysis component of the ACS identified forest roads where (1) drainage systems hastened runoff from storms and promoted sedimentation of streams, (2) unstable fill materials concentrated water and increased the risk of landslides, and (3) the roadbed encroached on riparian zones (Kershner 1997). Since NWFP implementation, the Aquatic and Riparian Effectiveness Monitoring Program estimated that 6.7 percent of the road network has been removed or closed (5,390 out of 80,750 mi total [8674 of 129 954 km]) in the NWFP area.[6] Additionally, 10 percent of the road crossings that impeded the movements of aquatic and riparian organisms (209 of 2,114) have been made passable on Forest Service Region 6 lands in the NWFP area since NWFP implementation.[7]

Though restoration of fish passage is often listed as a top priority for stream restoration in the Pacific Northwest (Roni et al. 2002, USGAO 2001), recent work has contributed much to our understanding of just how complex this issue is in practice (McKay et al. 2016). Advances have been made in culvert inventory and assessment (Clarkin et. al 2005), ecosystem-based restoration approaches (USDA FS 2008), and effectiveness monitoring (Heredia et al. 2016, Hoffmann et al. 2012). Until recently, however, the ecological benefit of these efforts has been difficult to quantify beyond the level of individual projects. A new

[6] Miller, S. 2016. Personal communication. National riparian program lead, U.S. Department of the Interior, Bureau of Land Management, 1849 C Street NW, Washington, DC 20240.

[7] Capurso, J. 2017. Personal communication. Regional fish and aquatic program manager, U.S. Forest Service, 333 SW First Ave., Portland, OR 97204, jcapurso@fs.fed.us.

Table 7-7—Summary of effects of road on aquatic ecosystems and associated biota

Ecological effect	Habitat loss/degradation	Habitat fragmentation	Direct mortality
Low population density	✓	✓	✓
Low population reproductive rates	✓		✓
Area occupied restricted	✓	✓	✓
Decreased habitat connectivity	✓	✓	
Overharvest			✓
Changes in water quality	✓		✓
Changes in hydrologic functions	✓	✓	✓
Change in wood and sediment recruitment	✓	✓	✓

Source: Modified from Robinson et al. 2010 and Daigle 2010.

study evaluating the effectiveness of passage restoration at the level of an entire forest (the Siuslaw National Forest) (Chelgren and Dunham 2015) found that individual culvert replacements successfully increased the probability of upstream access for all fishes in the study area. Results of this work also showed that the net benefit of culvert replacements was fairly modest across the extent of the forest when expressed in terms of gains in kilometers of stream occupied or increases in fish numbers resulting from restoration. The authors hastened to add that some limitations of the study design could have influenced these findings (Chelgren and Dunham 2015), but results of this study nonetheless point to the value of programmatic (vs. project-only) evaluations of culvert restoration. This echoes more general recommendations for following the cycle of adaptive management on national forests (Marcot et al. 2012) and the recommended scales for managing watersheds (Fausch et al. 2002, Neeson et al. 2015).

Much of the guidance for fish-passage restoration on federal lands in the Pacific Northwest was issued by an assessment in 2001 (USGAO 2001). This assessment highlighted the need for larger scale assessments, as noted above, as well as the economic challenges associated with passage restoration, which are only beginning to be addressed. For example, a followup to the Chelgren and Dunham (2015) study by Reagan (2015) evaluated costs and benefits of remaining culvert replacement opportunities on the Siuslaw National Forest in relation to multiple objectives, including benefits to fish (estimated from Chelgren and Dunham 2015), maintenance of transportation networks, and the probability of culvert failures based on culvert size and influences of floods and major erosional events in streams. The Reagan (2015) analysis explicitly quantified economic costs and benefits of restoration in relation to these objectives and their relative assumed values. This work (along with others in the region, e.g., Chelgren and Dunham 2015) has demonstrated the value of a proactive, economic analysis of multiple objectives to identify priorities for restoration investments in a programmatic context. These new tools, if applied, can more completely address standing recommendations to land management agencies in the Pacific Northwest (e.g., USGAO 2001) to more efficiently invest limited resources to benefit fisheries and other management objectives through culvert replacements.

Because road access management must take into account social, economic, and environmental objectives (Daigle 2010), the decisionmaking process for dealing with roads is complex. A decision matrix for identifying actions is shown in figure 7-14. In many cases, limited funds or socioeconomic issues may preclude closing or removing roads identified as high priority for treatment on the basis of their effects on riparian ecosystems. Also, a road network may be needed to effectively implement landscape-scale restoration projects that might involve widespread thinning and prescribed fire (Franklin and Johnson 2012), and for fire management, fuel reduction, and fire control. Studies

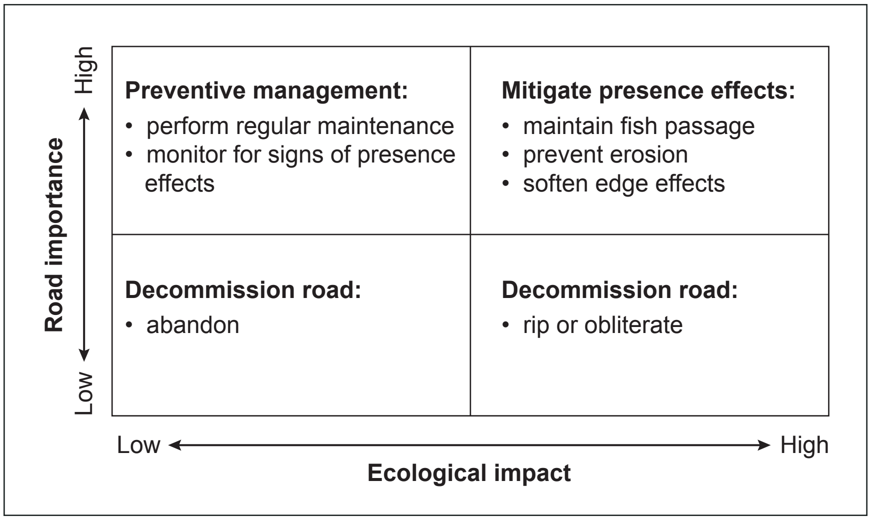

Figure 7-14—A decision matrix for identifying potential options for managing roads. (Modified from Robinson et al. 2010).

in Redwood National Park suggest that removal of logging roads can yield carbon-storage dividends, in particular by preventing soil erosion (van Mantgem et al. 2013). The vast road system on private and state lands that abut federal lands also needs to be considered in road assessments. Understanding how to balance fire management, recreation, and other needs against potential negative aspects of roads will require a concerted cooperative effort of managers and physical, biological, and social scientists, other organizations, and the public. (See next section for additional discussion of roads.)

Climate Change

Since 1994, our knowledge of climate change in the NWFP area has greatly improved, just as dealing with climate change has become an important aspect of environmental planning in the Forest Service and BLM. Many advances have come from models that forecast trends in temperature, precipitation, and snowpack, and associations of these trends with the habitat conditions for various species. Although there is general agreement about the direction of trends in many meteorological parameters, the rates and amounts of change at specific locations in the NWFP area differ among models (Climate Impacts Group 2009; see also chapter 2). Further, other climate-related changes such as increases in forest insect and disease outbreaks and uncharacteristically severe wildfires may accentuate the undesirable effects of meteorological and hydrological trends, resulting in threats to the integrity of both terrestrial and aquatic ecosystems. Although developing proactive measures that would ameliorate undesirable effects of climate change on forest resources was not a centerpiece of the NWFP, one of the Plan's main objectives was to restore forest ecosystems that were resilient in the face of natural and anthropogenic disturbances. The question is: how well does the NWFP address climate-related threats to native fishes and other aquatic biota as they are currently perceived? (See chapter 2 for further details.)

In this section, we focus on a review of recent advances in our knowledge of the likely effects of climate change on native fishes of the NWFP area. We examine climate-change effects on fish life cycles, with a principal focus on anadromous salmonids, a group of species that has received the most scientific attention, as well as significant conservation effort (table 7-8) (see additional discussion in app. 2). Watershed improvements undertaken through the NWFP are related to potential climate effects on fish life cycles, and to the capacity of populations to adapt and persist through time. Finally, we discuss the role that federally managed forests in the NWFP area play in conserving native fishes in a changing climate, when viewed in a broader matrix of different land ownerships and other landscape-scale uses.

Table 7-8—Species of Pacific salmonids considered in this section and their typical freshwater and marine residence times

Species		Residence time	
		Freshwater	**Marine**
Pink salmon	*Oncorhynchus gorbusha*	Less than 30 days	2 years
Chum salmon	*O. keta*	Less than 30 days	2 to 5 years
Sockeye salmon	*O. nerka*	Few months to 2 years	2 to 5 years
Coho salmon	*O. kisutch*	1 to 2 years	1.5 years
Chinook salmon	*O. tshawytscha*	Few months to 1 year	2 to 6 years
Steelhead	*O. mykiss*	1 to 3 years	2 to 4 years
Coastal cutthroat trout	*O. clarkii clarkii*	2 to 4 years	Short forays into nearshore environment

Climate change in the Northwest Forest Plan area—
Projected changes in climate are usually derived from models based on historical data coupled with scenarios incorporating alternative assumptions about future greenhouse gas emissions. These assumptions range from high global rates of economic development and human population growth to conservative industrial and population-growth rates and widespread implementation of "clean" technologies. Model outcomes are often displayed as incremental changes in an environmental parameter of interest such as air temperature, sea level, or precipitation over a fixed period. Projected changes in climate under different scenarios are plotted to provide a range of outcomes at a given point in time, with scenarios incorporating intermediate assumptions about future greenhouse gas emissions generally believed to represent the most realistic expectations.

Air and water temperatures—
Virtually all climate models forecast a gradual rise in air temperature by the end of this century. Recent changes in climate appear to be happening more rapidly than in at least the past 1,000 years (IPCC 2007), and have included a global average warming of 1.4 °F (0.8 °C) during the past 120 years. According to the IPCC (Intergovernmental Panel on Climate Change) (IPCC 2014), most general circulation models predict that 2 to 7 times more warming will occur by early in the next century, with projected increases in mean global surface temperatures by 2100, ranging from 2.7 to 3.6 °F (1.5 to 2.0 °C) relative to the 1850–1900 time frame, depending on carbon dioxide (CO_2) emission scenarios (IPCC 2014). The 2014 IPCC synthesis report (IPCC 2014: 10) states:

> It is **virtually certain** that there will be more frequent hot and fewer cold temperature extremes over most land areas on daily and seasonal timescales, as global mean surface temperature increases. It is **very likely** that heat waves will occur with a higher frequency and longer duration. Occasional cold winter extremes will continue to occur. [emphasis theirs]

The finding that climate change will include both gradual long-term temperature trends as well as increases in the frequency and duration of extreme events has important implications for aquatic ecosystems in the NWFP area.

Air-temperature changes in forests of the NWFP area are predicted to be generally consistent with global climate models, although somewhat more variable, with forecast increases ranging from 1 to 6.3 °F (0.5 to 3.5 °C) in the remainder of this century, depending on the greenhouse gas emission scenario used in the model and on forest location (Latta et al. 2010). Overall, these authors noted that relative temperature increases were more apparent at higher elevations than at lower elevations, and that proximity to the Pacific Ocean moderated the rate of change. Mote and Salathé (2010) examined a broad suite of IPCC climate models and found that, by the 2080s, average air temperatures in the Pacific Northwest were predicted to increase 2.9 °F (1.6 °C) under the coolest scenario and by 10.3 °F (5.7 °C) under the warmest scenario. In most models, the greatest absolute temperature increases were projected for summer months, although warming was also forecast in other seasons. Sea-surface temperatures showed less warming over the same period than those modeled over land.

Similar to air temperatures, water temperatures are expected to rise in much of the NWFP area as a result of climate change (Isaak et al. 2011 [NorWeST model]). Modeled water temperatures were developed primarily from models of the relation between air and water temperatures, and are projected to be stressful to lethal for many native salmonids (e.g., Isaak et al. 2012, Wade et al. 2013) (see app. 2 for more details). More recent studies suggest that the extent of temperature change may not be as great as originally projected, particularly at higher elevations (Isaak et al. 2016). However, other researchers (Arismendi et al. 2014) have questioned the ability to project future water temperature from past relations between air and water temperatures. In addition, Arismendi et al. (2013a) found that recent trends in water temperature have been more variable than those reported by Isaak et al. (2012)—using empirical records, they found that water temperatures increased in some systems and decreased in others. Also, Leach et al. (2016) also found variability in water temperature in a headwater stream of the Oregon Coast Range that was not captured by the NorWeST model (Isaak et al. 2010), but noted that the model was not designed to track such small-scale effects. Although there is some uncertainty about the extent of temperature changes that streams in the NWFP area will experience, it is clear that dealing with water temperatures will be a major challenge for managers.

Potential patterns of changes in water temperature are highly variable when examined at the local scale (Leach et al. 2016, Reeves et al. 2016b, Turschwell et al. 2016) (fig. 7-15). This variability is a result of local conditions such as stream orientation, topographic shading, and elevation, and strongly influences physical and biological attributes and resultant ecosystem integrity (Gomi et al. 2002, Thorp et al. 2006). Understanding this variability will be crucial to developing effective restoration and mitigation programs and prioritizing specifically where to target efforts. Watershed analysis tools such as NetMap (Benda et al. 2007) can help identify areas that can provide thermal refugia and areas in which riparian restoration efforts (fig. 7-16) could help reduce water temperatures to levels less stressful or even optimal for native fish, despite climate change (Justice et al. 2017, Lawrence et al. 2014, Ruesh et al. 2012).

Hydrology—

Predicted future changes in streamflow on national forests in the Pacific Northwest are fundamentally tied to changes in the region’s climate. Predicted changes in annual precipitation are much less certain, and most models project that future precipitation will remain approximately the same as it has been for the past 50 years (Salathé et al. 2007). Most predictions of changing streamflows for the Pacific Northwest therefore focus primarily on the effects of changes in temperature. Seasonal changes in precipitation are showing up in the data (Safeeq et al. 2013) but are difficult to resolve regionally, and consequently are not as well understood.

A key factor affecting both high and low streamflows in the future will be the fate of snow and the seasonal snowpack. Snowpack dynamics are important to understanding streamflow regimes because snow represents a dominant form of storage on the landscape. When precipitation falls as snow, it is not available for runoff or groundwater recharge until it melts. Similarly, the rate and timing of snowmelt are first-order controls on both peak and low streamflows, as discussed below.

A particularly crucial dimension of snowpack dynamics is the geographic location of the rain-snow transition on the landscape. This transition is controlled by elevation and determines how much of the winter precipitation falls as rain versus snow. Although often visualized as a fixed elevation, this transition is better seen as a stacked sequence of elevationally controlled zones or ranges with imprecise and regionally varying boundaries (Klos et al. 2014, Nolin and Daly 2006). In general, for any area, there is an elevation below which virtually all winter precipitation falls as rain and above which it falls as snow. Elevations in between are defined as the transitional snow zone (TSZ) that receives both rain and snow; snow and the snowpack usually will not persist all winter.

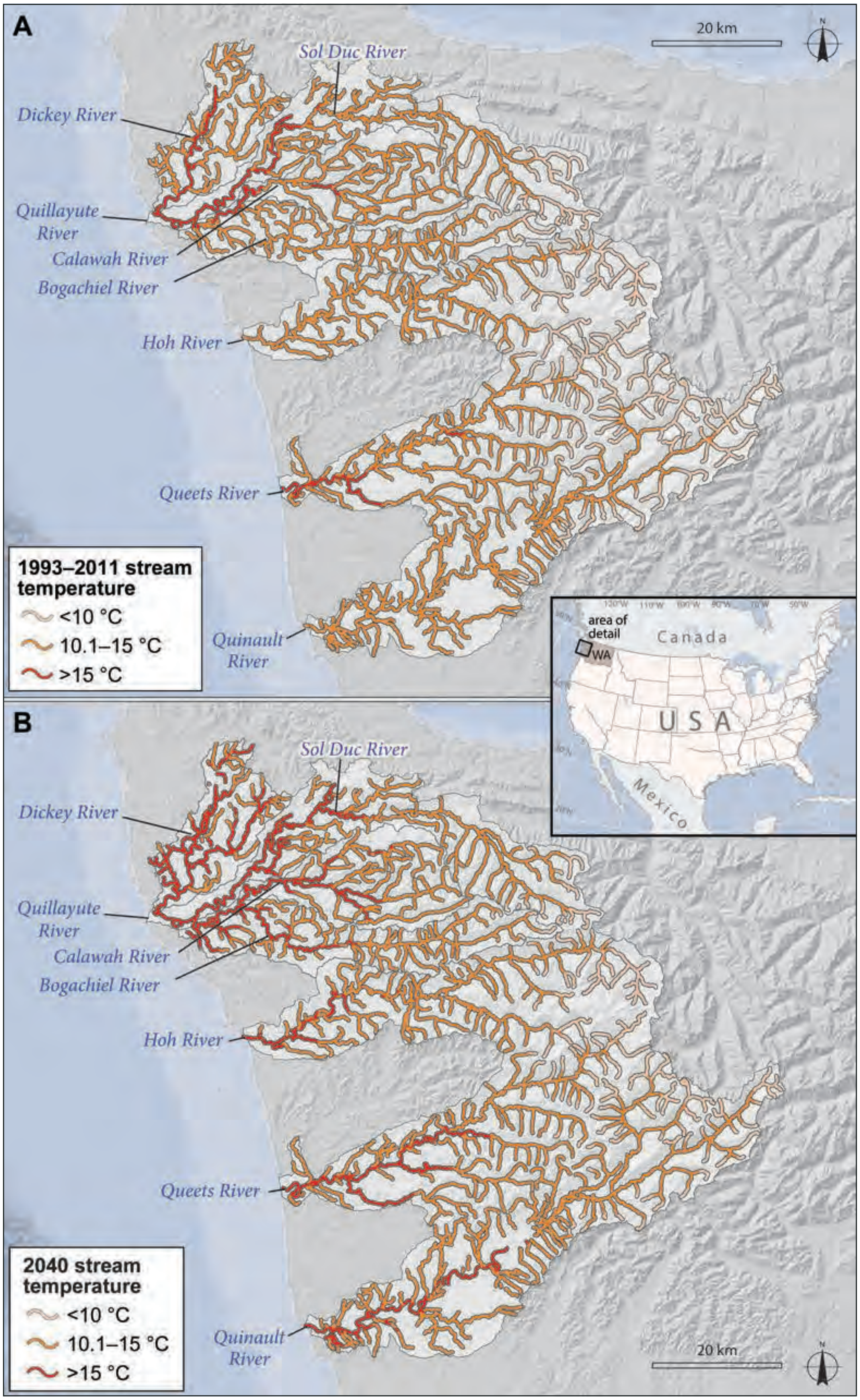

Figure 7-15—(A) Current and (B) projected (2040) summer water temperatures (°C) in the study basins in the Treaty of Olympia area (Reeves et al. 2016b).

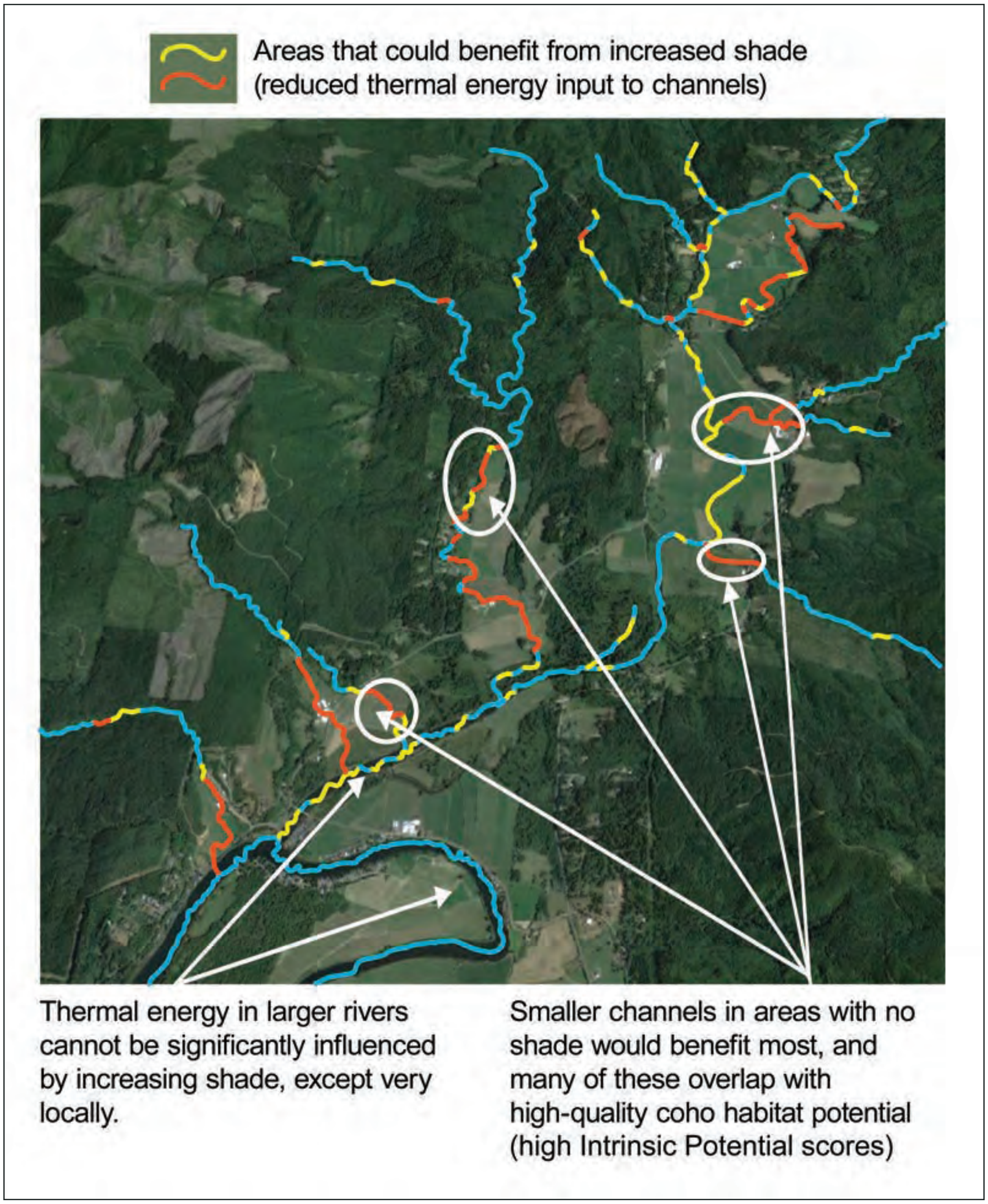

Figure 7-16—Example of identification of potential locations where riparian restoration could provide potential thermal refugia for native fish. Analysis from NetMap. (Modified from Benda et al. 2007.)

The widely varying elevational gradients in the Pacific Northwest in general, and on national forest lands in particular, therefore impose considerable variability in the response of the landscape to changing climate. Depending on the proportion of the landscape that occupies each of these zones, a warming climate, hence a rising snow line, may transition the landscape from a zone dominated by seasonal snow accumulation and melt (snow zone) to one that receives a mixture of rain and snow (and rain-on-snow)—the TSZ. Or it may push the landscape out of the TSZ and into the rain zone (Klos et al. 2014, Luce et al. 2014a)

The effects of a changing climate are already apparent in the snow data for the Pacific Northwest. As winter and spring temperatures have warmed over the past 50 to 70 years, spring snowpacks have been smaller (Hamlet et al. 2005, Mote 2003, Mote et al. 2005) and have melted out earlier (Hamlet and Lettenmaier 2007, Stewart et al. 2005). Moreover, the aforementioned zonal changes are already occurring in some landscapes as snow zones transition to the TSZ, and the TSZ becomes rain dominated (Tohver et al. 2014). These trends are expected to continue across much of the region (Luce et al. 2014a).

However, snowpack dynamics alone do not determine what future streamflow regimes will look like on national forests in the NWFP area. Recent work has shown that another first-order control is the landscape-scale drainage efficiency: the inherent, geologically mediated efficiency of landscapes in converting recharge (precipitation) into discharge (Safeeq et al. 2013, 2014; Tague and Grant 2009). In essence, the drainage efficiency determines how quickly precipitation, either as rain or snowmelt, becomes streamflow. Although drainage efficiency is "hard-wired" into the landscape on millennial timescales, and thus is not changing with climate, it mediates the climate-influenced signals and therefore has to be considered in predicting future streamflow regimes. This is particularly true for low-flow regimes, but influences peak flows as well. Basically, the drainage efficiency of a landscape is determined by the rate at which water moves through the subsurface. In steep landscapes with shallow soils, water rapidly moves laterally through the subsurface via both saturated and unsaturated pathways, drainage efficiency is high, and streams respond quickly to recharge events. In flatter landscapes with deep, permeable, porous, or fractured bedrock, water moves slowly as deep groundwater, drainage efficiency is low, and streams respond slowly to recharge events but may have sustained high base flows.

Effects of climate change on peak flows—Here we broadly consider how both climate and drainage efficiency can shape predictions of future streamflows on national forest lands. We distinguish between effects on peak and low flows, as the mechanisms of streamflow generation are different in each case. Finally, we discuss how these broad predictions can be refined for individual forests, a topic beyond the scope of the current analysis.

There are several hydrologic mechanisms by which climate could increase peak flows in rivers and thus their propensity to flood. More intense or frequent rainstorms are one mechanism, and some research has suggested that a warming atmosphere may result in a more northerly storm track for the North Pacific, potentially resulting in more intense precipitation (Salathé 2006). However, these results have large uncertainties and are not well represented in most global circulation models. A somewhat better-understood mechanism is the shifting potential for rain-on-snow (ROS) events in the Pacific Northwest as the climate warms. ROS events are known to be a potent flood-producing mechanism in steep mountain landscapes in the Pacific Northwest (Harr 1981, Marks et al. 1998, McCabe et al. 2007). In general, landscape susceptibility to ROS events is determined by climate and topography; the effects of climate warming on ROS are similarly influenced by the same controls; and climate warming may increase, decrease, or not affect the risk, depending on whether snowpacks are cold or warm (i.e., near the freezing point). As summarized by Hamlet and Lettenmaier (2007):

> Cold river basins where snow processes dominate the annual hydrologic cycle (<6 °C average in midwinter) typically show reductions in flood risk due to overall reductions in spring snowpack. Relatively warm rain-dominant basins (greater than 5 °C average in midwinter) show little systematic change. Intermediate or transient basins show a wide range of effects depending on competing factors such as the relative role of antecedent snow and contributing basin area during storms that cause flooding. Warmer transient basins along the coast in Washington, Oregon, and California, in particular, tend to show increased flood risk.

A more recent analysis looked at a range of factors influencing peak flows, including ROS in Oregon and Washington, and developed a model of sensitivity to peak-flow increases based on perturbing the temperature in the model using warming scenarios from 2020 to 2080 and the A1B[8] emissions scenario (Safeeq et al. 2015). The analysis yielded regional sensitivity maps for Oregon and Washington that can be used to characterize the risk on individual national forests and landscapes. They concluded

[8] This scenario assumes a future world of rapid economic growth and global populations peaking in the mid-21st century, then declining with the rapid introduction of new technology, with a balance between the use of fossil fuels and non-fossil-fuel sources.

that corresponding changes in snowpack dynamics may result in large (more than 30 to 40 percent) increases in peak flows, primarily in the Cascade Range and Olympic Mountains. The North Cascades, in particular, were most vulnerable (fig. 7-17). Lower elevation areas were less likely to be affected but were still vulnerable to larger floods generated from upstream reaches in vulnerable landscapes. These watersheds are also likely more susceptible to warming (Arismendi et al. 2013a, 2013b; Poole and Berman 2001; van Vliet et al. 2011, 2013). Streams at higher elevations should retain flows; with stable, cooler water temperatures, they will be critical cool-water refugia for native fish (Isaak et al. 2012, 2015; Luce et al. 2014b; Lusardi et al. 2016; Wenger et al. 2011).

Effects of climate change on low flows—Snowpack dynamics and drainage efficiency combine to determine the sensitivity of individual landscapes to a warming climate (Safeeq et al. 2013, Tague and Grant 2009). There has been a general trend over the past 50 years for less snow in winter and earlier snowmelt, resulting in reductions of spring, early-summer, and late-summer flows in the Western United States (Leppi et al. 2012, Safeeq et al. 2013), with the lowest flows showing the greatest decreases across the Pacific Northwest (Luce and Holden 2009). Hydrologic models such as the variable infiltration capacity (VIC) model, coupled with downscaled climate simulations, have been used to generate predictions of future low flows across much of the Pacific Northwest (e.g., Hamlet et al. 2013).

However, snowpack changes are not the only factor determining future low flows. Other recent work has shown that the drainage efficiency (slow versus fast) mediates the signal from climate-induced changes in snowpack and snowmelt. Employing a simple exponential-decay model to describe the recession behavior of streams, coupled with a regional-scale estimation of variations in aquifer-drainage characteristics, Safeeq et al. (2014) developed a sensitivity map for changes in summer streamflow across Oregon and Washington. As with the VIC products and peak-flow maps previously described, these maps provide water and landscape managers with a spatially explicit representation of where future changes in low flows are likely to be most pronounced. For example, these maps show that areas drained by young volcanic rocks with deep, slow groundwater systems, such as the High Cascades, may be particularly vulnerable to declines in summer streamflow, whereas areas with shallow subsurface aquifers and limited potential to store water are less sensitive to changes in low flows. Climate-change effects on summer low flows may be compounded by effects of forest-vegetation conditons. Perry and Jones (2017) found that average daily streamflow in smaller streams in summer in watersheds with 34- to 43-year-old plantations of Douglas-fir was 50 percent lower than streamflow from reference basins with 150- to 500-year-old forests. The change in flows is also likely to be highly variable among watersheds in a given area (fig. 7-18).

Assessing climate change effects on streamflow at the scale of individual national forests—The discussion above highlights how existing tools and models can be used to give technically sound predictions about the magnitude and timing of streamflow changes in specific landscapes. Although not a trivial exercise, any national forest can use the spatially explicit models already developed to make first-order forecasts for changes in streamflow regimes. The products to date cover most but not all forests in the area of the NWFP. Extending results to these unmapped forests (mostly in northern California) would require some extrapolation, but is well within the scope of the existing data. Tools and approaches such as the concept of "hydrologic landscapes" can expedite this process (Patil et al. 2014, Wigington et al. 2013, Winter 2001).

Furthermore, there are several examples to date of individual forests or groups of national forests and other federal and nonfederal landholders that have coordinated efforts to develop detailed assessments of likely hydrologic changes that can serve as models for other forests and regions. Specific examples include the Olympic National Forest (Halofsky et al. 2011), the Quinault Indian Nation on the Olympic Peninsula (Reeves et al. 2016b), the Blue Mountains Adaptation Partnership (Halofsky and Peterson 2017), and the upcoming report from the South Central Oregon Adaptation Partnership.

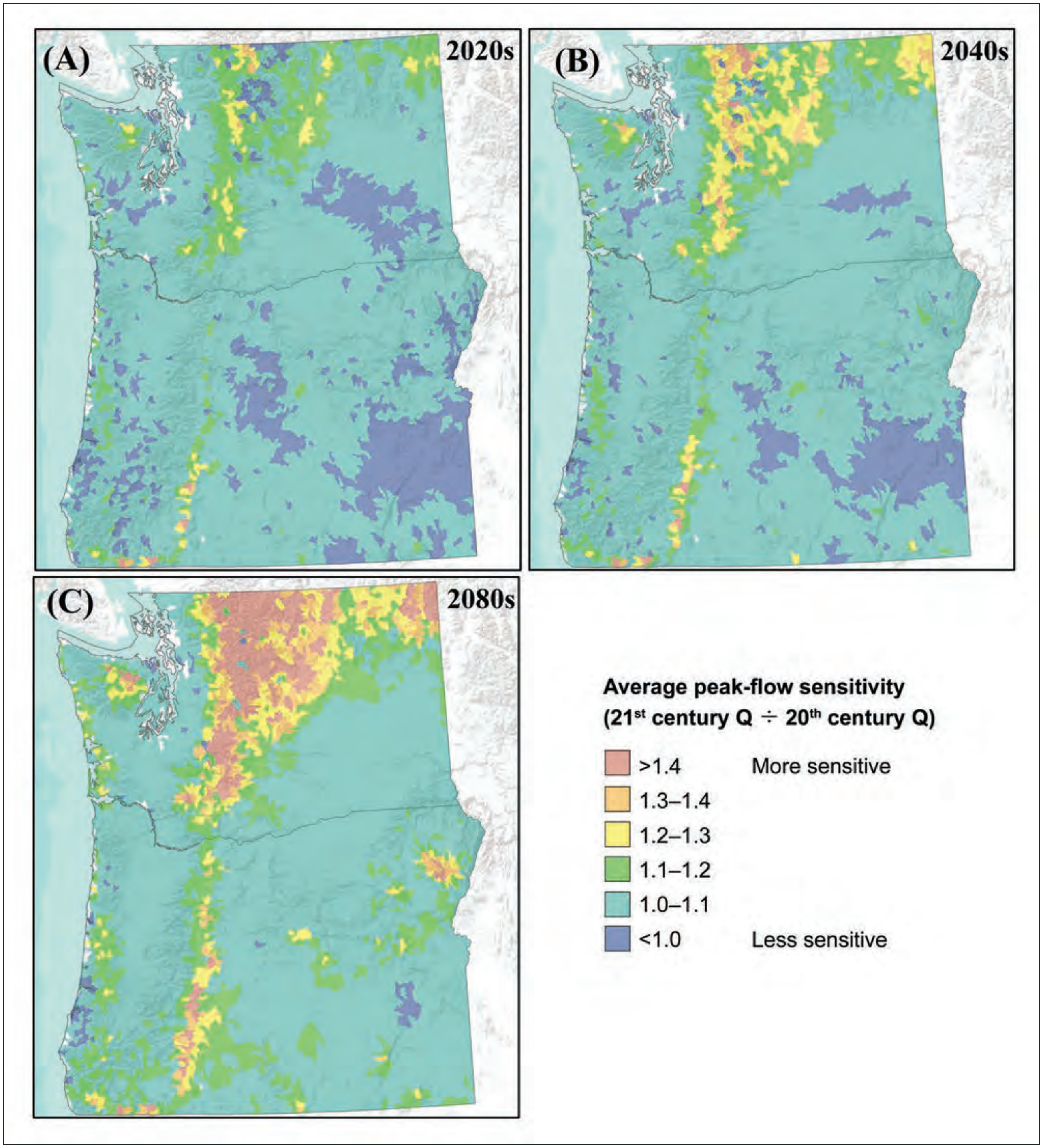

Figure 7-17—Sixth-field hydrologic unit-scale average peak-flow sensitivities across all flood magnitudes (Q2, Q10, Q25, Q50, and Q100) under A1B emission scenario for the (A) 2020s, (B) 2040s, and (C) 2080s, in which red is more sensitive and blue is less sensitive. (Modified from Safeeq et al. 2014.)

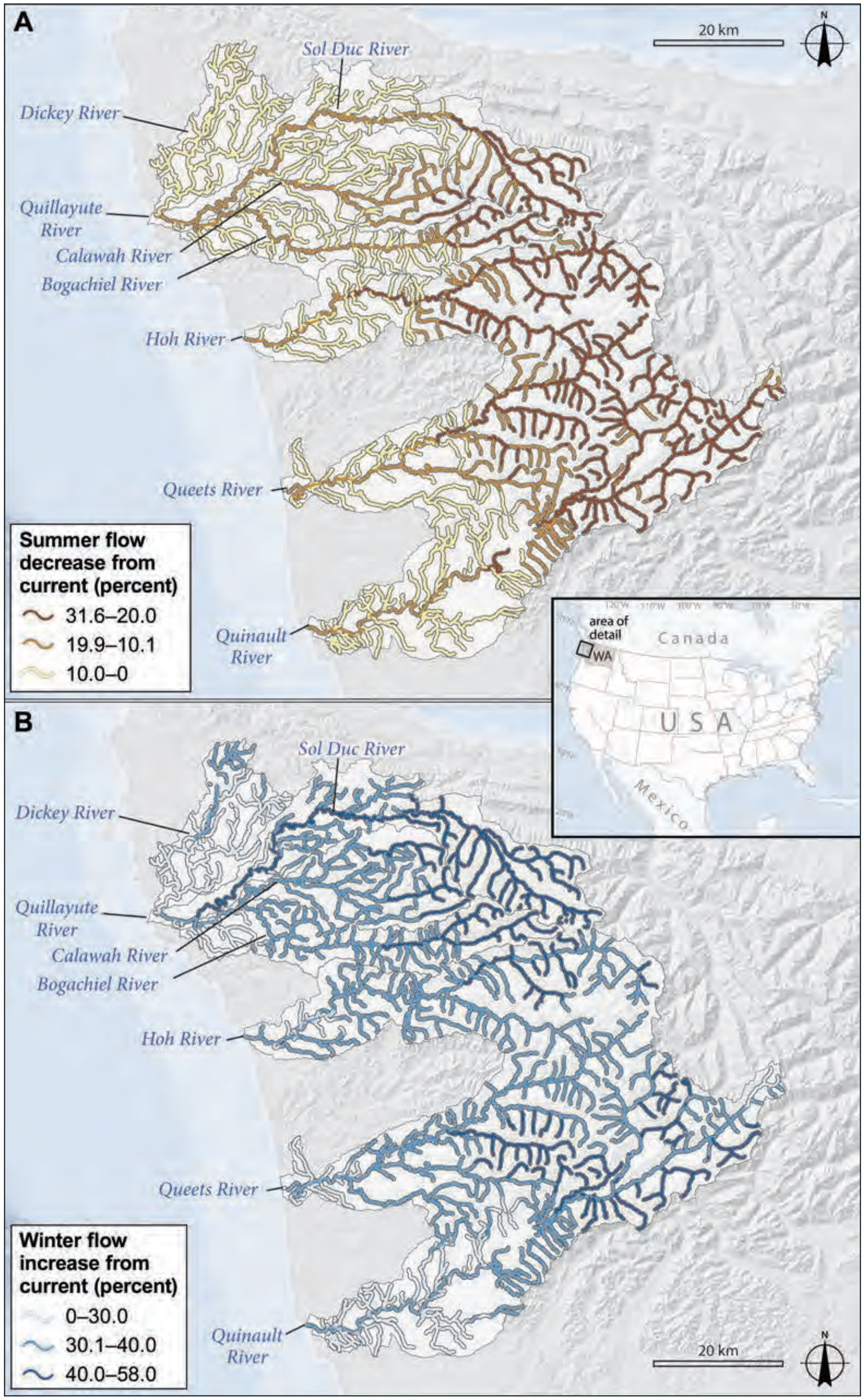

Figure 7-18—Percentage of reduction in average (A) summer and (B) winter flow levels from current to 2040 in study basins in the Olympic Peninsula area (Reeves et al. 2016b).

Extreme events—Increased frequency of extended, severe droughts and intense winter-storm events (IPCC 2014) will also affect aquatic ecosystems and fish populations in forested landscapes (Ward et al. 2015). The pattern of changes will differ widely within and among watersheds depending on local features, making it difficult to generalize the effects. However, changes in the seasonal timing of annual hydrographs and more frequent extremely low or high flows are very likely to affect native fish populations. Changes in flows that lead to earlier spring runoff and prolonged periods of summer low flows could have important implications for the habitat of (ISAB 2007) and food chains encompassing fish and other aquatic organisms (Power et al. 2008, Wooton et al. 1996) (see discussion in app. 2 for more details). Amphibians that inhabit ephemeral ponds and streams are likely to be especially vulnerable to drought and general climate change effects (Blaustein and Olson 1991, Shoo et al. 2011).

Ocean conditions—

Over the past several decades, the importance of the marine environment to fish that spend part or all of their lives at sea has been recognized as a major factor regulating population abundance. Climate-related changes in the ocean that are potentially important to native fishes in the NWFP area include acidification (Orr et al. 2005), increased sea-surface temperatures (IPCC 2007), changes in wind and current patterns (Rykaczewski and Dunne 2010), and sea-level rise (IPCC 2007). Absorption of anthropogenic CO_2 by the upper ocean decreases pH and carbonate-ion concentrations (Hendriks et al. 2010, Orr et al. 2005), increasing acidity and inhibiting the ability of planktonic organisms to form calcium carbonate, a key component of their exoskeleton. Many of these organisms form the base of the food chain that supports anadromous fishes during the marine phase of their life cycles. The subarctic Pacific Ocean has naturally higher carbon concentrations than most other ocean basins, and the effects of acidification are expected to occur sooner and be more pronounced there (Cooley et al. 2012).

Rising sea-surface temperatures may reduce the amount of preferred thermal habitat for anadromous salmonids in the ocean and potentially limit their marine distribution (Aziz et al. 2011, Welch et al. 1995). As areas with suitable temperatures decrease or shift northward, Pacific salmon could become concentrated in smaller foraging zones, resulting in increased competition for limited food resources (Grebmeier et al. 2006, Johnson and Schindler 2009, Mantua et al. 2009, Welch et al. 1995). Salmon may be able to partially compensate for these changes by using cooler subsurface waters; however, deeper water may provide reduced food resources, increased competition with other marine species, or greater exposure to predation (Hinke et al. 2005, Myers et al. 1996).

Other potentially important climate-related changes in the marine environment include sea-level rise (IPCC 2007) and altered patterns of coastal upwelling (Wang et al. 2015). The consequences of sea-level rise for nearshore fishes are uncertain and will be strongly influenced by local topography; new habitat could be created in some areas but lost in others (Flitcroft et al. 2013). Saltwater inundation may affect species that sometimes spawn immediately above tidewater (e.g., pink and chum salmon, *Oncorhynchus gorbuscha* and *O. keta*). Both positive and negative effects on estuaries are also possible as new land is inundated, and the degree to which estuarine productivity is altered will be influenced by the extent of human development. Where development of estuary and coastal shorelines is extensive, sea-level rise will likely result in more seawalls, channelization, and other measures to prevent flooding during storm surges (Neumann et al. 2015).

Changes in the patterns of coastal upwelling in the NWFP area could have very significant effects on anadromous fishes as well as other animals that depend on marine food webs. Wind-driven ocean currents regulate the strength of coastal upwelling along the Pacific Coast, where nutrients from deep-ocean waters fuel plankton blooms that are critical to marine food webs that support salmon (Francis and Sibley 1991). Long-term shifts in the timing and intensity of coastal currents and upwelling have accompanied climate change in the eastern Pacific Ocean, with winter and spring storm tracks gradually shifting northward (Salathé 2006) and upwelling along the coast in the NWFP area becoming more erratic and unpredictable (Bylhouwer et al. 2013). Anadromous salmonids are particularly vulnerable to changes in upwelling because survival of fish in the first few weeks after entering the ocean depends heavily on their ability to feed and grow large enough to avoid predation (Beamish and Boullion 1993, Pearcy 1992, Walters et al. 1978).

The occurrence of interdecadal shifts in sea-surface temperatures and related weather patterns (Pacific Decadal Oscillation—PDO) from cool/wet to warm/dry conditions (Mantua et al. 1997) further complicates the productivity of marine environments along fish migration routes, with more favorable ocean conditions occurring when the NWFP area is in a cool/wet phase than in a warm/dry phase. Wang et al. (2015) used a suite of climate models to forecast upwelling over the remainder of this century and found that, by the year 2100, coastal upwelling will likely start earlier, end later, and be more intense in the northern latitudes (British Columbia, Canada, and southeast Alaska) than in southern latitudes (northern California and Oregon). Wang et al. (2015) also noted that an intensification of upwelling could actually promote plankton productivity, but in extreme cases could also result in large swaths of anoxic conditions developing over broad areas, leading to massive die-offs of marine life where such conditions develop. Taken together, the new information on climate-related PDO cycles and trends in upwelling patterns suggest that the marine environment along the Pacific Coast is becoming more variable spatially and temporally, with northern California and Oregon being somewhat more likely to exhibit unpredictable ocean conditions than more northerly latitudes. For migratory organisms such as anadromous salmonids whose life cycles are adapted to being in the right place at the right time for feeding and reproduction, introducing more variability into the part of their life cycle where most growth occurs is likely to add to population destabilization.

Climate effects on fish life cycles—
Although the extent to which a particular fish population in the NWFP area will be affected by climate change depends to a large degree on changes that occur at the local level, climate-related effects, both favorable and unfavorable, can accumulate across multiple life-history stages. Restricting an understanding of climate influences to a single life-history stage may well underestimate the total effect on the population. Further, because of the wide geographic distributions of many native fishes and the heterogeneity of aquatic environments in which they reside, climate effects may be expressed differently across the range of a given species. Locally adapted life histories differ over broad landscapes and among different species; even stocks of the same species can exhibit dissimilar responses to similar climate trends (Schindler et al. 2010). A number of papers have investigated the potential effects of climate change on Pacific salmon, but these have primarily been overviews (e.g., Bryant 2009, ISAB 2007) or results of modeled effects on a given life-history stage (e.g., Crozier and Zabel 2006, Rand et al. 2006) and its associated habitat (e.g., O'Neal 2002). A comprehensive review of the effects of climate change on native fishes in the NWFP area across their ranges, including effects accumulated across multiple life-history stages, is lacking.

Understanding the potential consequences of altered future conditions, particularly where the perceived effects may not be lethal, requires consideration of the effects at each life-history stage (Fleming et al. 1997, ISAB 2007, Jonsson and Jonsson 2009). Changes at one life stage can cascade throughout the remaining stages, significantly altering population response. Focusing on anadromous Pacific salmonids, it is possible to examine the overall impacts of climate change by identifying effects at each life-history stage and discussing how those effects might be propagated through succeeding stages. These effects and potential management options are listed in table 7-9. It is also possible to identify attributes of Pacific salmon life cycles that promote their adaptive capacity to climate change, along with options for managers and decisionmakers to enable and enhance those attributes to mitigate potential effects of climate change in the NWFP area.

Other climate-related factors—
Climate warming will lead to an increase in the area burned by wildfires (IPCC 2014) (chapter 2). Wildfire trends in the NWFP area will be complex because the area includes a wide array of forest types, elevations, weather regimes, and forest-management histories (Hessburg and Agee 2003); hence risks of damage to native fish habitats are likely to be highly variable across the region. In addition to altering wildfire frequency and intensity, climate change will also

Table 7-9—Potential effects of climate change on anadromous salmonids of the Pacific Northwest, by life-history stage

Life stage and habitat	Potential effect of climate change	Ecological consequences	Ecological implications	Potential actions
Adults:				
Ocean	Acidification (Hendriks et al. 2010, Orr et al. 2005)	Reduced growth and survival	Lower productivity of freshwater because of reduced amount of marine-derived nutrients and eggs (Bilby et al. 1996, Cederholm et al. 2001, Garner et al. 2009, Gende et al. 2004, Helfield and Naiman 2001, Lang et al. 2006, Schindler et al. 2003)	
	Increased sea-surface temperatures (Aziz et al. 2011, IPCC 2007)	Smaller size at return	Reduced population reproductive capacity (Hankin and McKelvey 1985, Healey and Heard 1984)	
		Change in life-history expression (*Oncorhynchus mykiss*)	Loss of steelhead life history (migratory) and increase in rainbow trout life history (resident) (Benjamin et al. 2013, Quinn and Myers 2004, Rosenberger et al. 2015, Sloat and Reeves 2014)	Population monitoring with consideration of life-history types
Freshwater	Sea-level rise (IPCC 2007)	Increased estuary habitat	Increased life-history diversity (Bottom et al. 2005)	Population monitoring with consideration of life-history types
		Loss of spawning habitat in areas close to coast	Reduced population productive capacity	
		Increased flooding during surge events	Reduced egg survival	
	Increased water temperature (Isaak et al. 2010)	Increased stress	Reduced survival to spawning grounds (Martins et al. 2011, Rand et al. 2006) and population reproductive capacity (Miller et al. 2011, Pankhurst et al. 1996) Increased susceptibility to disease and parasites (Chiaramonte et al. 2016, Johnson et al. 1996, Miller et al. 2011, Ray et al. 2012)	Monitoring water temperatures in entire stream network to identify and protect areas of thermal refugia

Table 7-9—Potential effects of climate change on anadromous salmonids of the Pacific Northwest, by life-history stage (continued)

Life stage and habitat	Potential effect of climate change	Ecological consequences	Ecological implications	Potential actions
Eggs and alevins:				
Freshwater	Elevated winter water temperatures	Increased rates of development (McCullough 1999, Neuheimer and Taggart 2007)	Smaller size at emergence (Beacham and Murray 1990, Elliott and Hurley 1998)	Year-round monitoring of water temperatures
		Earlier time of emergence (Holtby 1988)	Increased growth rates, earlier timing of smolting, and smaller size at ocean entry, but decreased marine survival (Holtby and Scrivener 1989, Schindler et al. 2005)	Increase availability of floodplain and off-channel habitats
	Increased winter flows (Hamlet and Lettenmaier 2007, Hamlet et al. 2005, Tague and Grant 2009)	Increased scour of redds (Battin et al. 2007)	Reduced survival (Battin et al. 2007, Leppi et al. 2014, Shanley and Albert 2014)	Increase connection to floodplain, remove roads and infrastructure that restrict access to floodplain, with wood placement in and near low-gradient spawning areas
		Increased landslides and flooding (Dale et al. 2001, Hamlet and Lettenmaier 2007, Miller et al. 2003)	Decreased habitat quality in short term but improved conditions in long term (Bisson et al. 2009; Flitcroft et al. 2016a; Reeves et al. 1995; Rieman et al. 2006, 2015)	Road removal and culvert improvements in most susceptible areas Silvicultural treatment of plantations in most susceptible areas to increase size of trees
Juveniles:				
Freshwater	Higher spring flows (Hamlet and Lettenmaier 2007, Hamlet et al. 2005, Tague and Grant 2009)	Increased access to floodplain and off-channel habitats	Increased growth and survival if floodplains and off-channel habitats available (Brown and Hartman 1988, Moore and Gregory 1988, Peterson 1982a); decreases if not	Increased access to floodplain and off-channel habitats
	Earlier onset of low flows (Hamlet and Lettenmaier 2007, Hamlet et al. 2005, Tague and Grant 2009)	Reduced habitat availability (Battin et al. 2007, Luce and Holden 2009, Mantua et al. 2010, Stewart et al. 2005)	Reduced survival (Battin et al. 2007, Mantua et al. 2010)	Identify areas in network that are likely to be refugia during low-flow period and improve habitat conditions, including improving riparian conditions to reduce water temperature

Table 7-9—Potential effects of climate change on anadromous salmonids of the Pacific Northwest, by life-history stage (continued)

Life stage and habitat	Potential effect of climate change	Ecological consequences	Ecological implications	Potential actions
	Increased summer water temperatures (Isaak et al. 2010)	Reduced growth and survival if temperature increases are beyond favorable range (Crozier and Zabel 2006, Isaak et al. 2010, Marine and Cech 2004, Royer and Minshall 1997, Scarnecchia and Bergersen 1987)	Smaller size and reduced survival (ISAB 2007, Quinn and Petersen 1996)	Improve riparian conditions
		Increased growth if temperatures move into more favorable range		
		Altered outcomes of interactions with other species (ISAB 2007, 2012; Petersen and Kitchell 2001; Reeves et al. 1987)	Reduce growth and survival (ISAB 2007, Petersen and Kitchell 2001, Reeves et al. 1987) Change in structure and composition of fish communities (warmwater species increase) (ISAB 2012)	
	Increased nonsummer water temperatures	Increased growth rates (Ebersole et al. 2006, Sogard et al. 2010)	Increased growth and survival	Year-round monitoring of water temperatures Provide access to intermittent streams, off-channel habitats, and floodplains
Smolts:				
Freshwater	Earlier onset of low flows (Hamlet and Lettenmaier 2007, Hamlet et al. 2005, Tague and Grant 2009)	Altered timing and rate of ocean migration	Decreased marine survival (Holtby 1988, Rechisky et al. 2009)	
	Warmer water temperatures (Isaak et al. 2010)	Smaller size at ocean entry	Reduced marine survival (Holtby and Scrivener 1989, Quinn and Peterson 1989, Slaney 1988)	Improve riparian conditions
Ocean	Altered timing of ocean upwelling (Barth et al. 2007, Snyder et al. 2003)	Reduced availability of food resources (Nickelson 1986, Scheurell and Williams 2005)	Reduced marine survival (Holtby and Scrivener 1989, Quinn and Peterson 1989, Slaney 1988)	

influence outbreaks of insects and forest diseases (Spies et al. 2010) in some cases, leading to alterations of forest stands that affect aquatic habitats. Wildfires, insects, and forest diseases should not be viewed strictly as threats to native fishes, however—they may also provide benefits. They can create openings and patches along water bodies that result in more complex stream channels and greater biodiversity (Flitcroft et al. 2016a, Reeves et al. 1995, Rieman et al. 2006). In addition, the erosional processes that accompany these disturbances are important for recruiting wood and coarse sediment that form essential habitats for many aquatic organisms (Benda et al. 2004). Thus, actions that seek to control erosion and other ecological processes that occur following wildfire may have long-term and unintended negative consequences for aquatic ecosystems (Chin et al. 2016, Harris et al. 2015).

The effects of climate change on aquatic ecosystems in the NWFP area expressed through wildfires, insects, and diseases will be complex and difficult to predict, but it will be important to examine the current responses to wildfire and consider making potential changes to allow fire to be more ecologically beneficial. Climate change will likely influence the expansion of nonnative plant and animal species in the NWFP area, while at the same time either reducing or even extirpating native species (Dale et al. 2001, Garcia et al. 2014, Urban 2015). Nonnative species include undesirable invasives, species undergoing expansion of their native ranges, and nonnative species deliberately introduced for commercial, recreational, or cultural reasons. They can occur in both terrestrial (riparian) and aquatic ecosystems. Nonnative species are not always harmful to native fishes or their habitats, but in many instances they can (1) compete with, prey upon, hybridize with, or infect native species with novel pathogens; (2) greatly alter the structure of food webs; or (3) cause habitat changes that reduce the productivity of desirable aquatic organisms. See appendix 1 for a detailed discussion of invasive species in the NWFP area.

Sanderson et al. (2009) provided a useful summary of underappreciated threats to salmon posed by nonnative vertebrates, invertebrates, and plants. They concluded that threats posed by nonnative species may equal or outweigh threats posed by traditionally perceived causes of decline—habitat alteration, harvest, hatcheries, and hydroelectric production. Many of the nonnative fishes known to harm native fishes of the NWFP area are warmwater fish species deliberately introduced from eastern North America. In some river basins, these forms have largely displaced native fishes from dominant roles in the aquatic food webs of low-elevation, low-gradient rivers (ISAB 2012). Continued warming will favor the expansion of warm-adapted species in western North America (Rahel et al. 2008), and shrinking headwater flows resulting from longer, drier summers (Moore et al. 2007) could force cool-adapted native species lower in drainage systems, where there will be greater opportunity for unwanted interactions with established populations of introduced game fishes. Restoration of riparian areas, however, can help reduce water temperatures and the potential negative consequences of climate change related to elevated water temperatures (Justice et al. 2017, Lawrence et al. 2014)

Restoration and response to climate change under the Aquatic Conservation Strategy—

Watershed improvements implemented in the Northwest Forest Plan area—An important goal of the NWFP was to create a managed federal forest landscape in which natural ecological processes sustained healthy populations of native fish and wildlife (USDA and USDI 1994a). Architects of the ACS recognized that federally managed forests might anchor the recovery of imperiled native fishes, but because of their location relative to state and private forests as well as other types of land use (which tended to be located in lowland areas), they could not ensure that appropriate conservation measures would be applied throughout the full suite of freshwater environments to which many native species, particularly anadromous salmonids, were exposed (Sedell et al. 1997). Nevertheless, many of the aquatic-conservation actions that emerged from the NWFP were considered at the time to provide more protection to aquatic and riparian habitats than had ever before been implemented on multiple-use forests in the Pacific Northwest (NRC 1996). The region's national parks and designated wilderness areas were also considered to possess high-quality habitats in which natural ecosystem processes could operate. However, because of their scarcity and location (Reeves et al. 2016a, Sedell et al. 1994), such areas were generally believed to be inadequate to prevent species or their distinct population segments (evolutionarily significant units) from becoming imperiled, or to hasten recovery. Given the impact of climate change on fish life cycles as discussed above, how the framework and standard of guides of future forest plans could address these potential effects merits priority for future research.

Restoration of mid- and late-seral forest stands—Concurrent with the restoration of mid- and late-seral stands in the NWFP area, the region will likely see a reduction in large openings caused by regeneration harvests (clearcuts) and by wildfire, as a result of continuing fire suppression (see chapter 3). As forest stands grow older in the seasonally transient snow ("rain-on-snow") zone, snowfall interception by branches will diminish the accumulation of ground-level snow and will prolong melting and runoff processes during subsequent rain events (Harr 1986). Peak flows were found to increase by as much as 20 percent in small watersheds and 30 to 100 percent in larger basins over a 50-year period in the western Cascade Range of Oregon in response to road building and clearcutting (Jones and Grant 2001). However, a recent synthesis of peak-runoff studies in western Oregon and Washington (Grant et al. 2008) concluded that the incremental contribution of clearcutting to peak flows in the transient snow zone was minor relative to other types of human disturbance, and would likely be confined to stream reaches possessing 2 percent gradients with sand and gravel substrates. In areas in which climate change results in an expansion of the transient snow zone, restoration of late-seral stands is likely to reduce the frequency and possibly duration of flows that are capable of mobilizing substrates of some fish-bearing streams, which could benefit survival of developing fish eggs and alevins as well as the abundance of amphibians and benthic macroinvertebrates.

One climate trend with important implications for native fishes is the lengthening of low-flow periods during the warm season; aquatic organisms in watersheds with reduced snowpack will be especially affected by lower summer flows. Although not thoroughly investigated, the capture of fog by tree branches in areas with summer fog can result in "fog drip" that contributes to runoff during times when rainfall is scarce (Harr 1982). Whether climate change will alter the frequency of foggy days in the NWFP area is poorly understood, but there is preliminary evidence that the intensification of wind-driven upwelling in the California current as a result of increased CO^2 could lead to more fog and increased moisture flux along the Pacific Northwest coast during the upwelling season (Snyder et al. 2003). However, Johnstone and Dawson (2010) reported that fog frequency along the northern California coast declined by 33 percent in the 20th century. Nonetheless, restoration of late-seral stands will result in taller trees with larger limbs, which could capture more moisture and deliver some of it to streams during a season when water is in short supply.

Increasing the amount and sources of large wood will help aquatic ecosystems and associated biota meet the challenges of climate change. The progressive impoverishment of large wood in Pacific Northwest streams, particularly large-diameter, habitat-forming tree trunks and rootwads, has long been recognized (Bisson et al. 1987, Sedell and Swanson 1984). Climate change is expected to change the frequency and severity of fires and the incidence of forest-pathogen outbreaks in many parts of the NWFP area (see chapters 2 and 3). However, the ensuing recruitment of large wood to streams, a key component of fish habitat, may be limited if landslide-prone headwalls that normally deliver this material to channels during and following natural disturbances no longer contain trees of the size needed to form and maintain structural fish habitats. The importance of wood recruited to streams from unstable hillslopes is often underappreciated. For example, Reeves et al. (2003) found that 65 percent of the large wood pieces and 47 percent of the large wood volume in an Oregon coastal stream originated from upslope sources. Measures that could take advantage of this source of wood include inventorying and mapping unstable headwall areas, protecting them from forestry-related disturbance, permitting natural wood-delivery processes to occur, and allowing late-seral stands to develop in these areas where appropriate (Cissel et al. 1999).

Reducing the effects of roads and passage barriers—Reducing the hydrological and biological effects of forest roads in the NWFP area should improve watershed resilience to the adverse effects of climate change on aquatic ecosystems. Road cuts are known to be a major contributor to accelerated runoff during storms by intercepting subsurface flow and capturing it in ditches, which rapidly deliver water and fine sediment to streams (Wemple and Jones 2003). As the intensity of storms increases with gradual warming and, in some parts of the NWFP area, with greater precipitation, the risk of streambed-mobilizing runoff events will rise as well. Reducing the exacerbating effects of road drainage networks on peak flow in watersheds where roads have been decommissioned could lessen the potential for severe storms to scour eggs and alevins in stream gravels and likewise reduce the intrusion of harmful fine sediment into spawning substrates. In addition, eliminating road-related initiation points for landslides through road decommissioning will help return the frequency of mass wasting in watersheds to more natural levels.

Road corridors can serve as important invasion routes for nonnative species, especially nonnative plants (González-Moreno et al. 2015, Heckman 1999, Menuz and Kettenring 2013), and climate change is likely to favor continued expansion of nuisance and harmful exotic herbaceous species in watersheds (Dale et al. 2001). The effect of invasive plants on riparian ecosystems in federally managed forests has received relatively little study, but some plants (e.g., Asian knotweed, *Polygonum* spp.) are capable of displacing native vegetation (Urgenson et al. 2009) and disrupting the transfer of organic material from streamside vegetation to stream channels. Invasive plant-control programs are costly, and even in riparian zones where treatments have been applied, the long-term reestablishment of native plants has been difficult to achieve (Claeson and Bisson 2013). Therefore, reducing road densities in a watershed and across large areas should help forestall the movement of unwanted nonnative plants into sensitive riparian areas and protect the integrity of native plant assemblages.

Floodplain protection—One of the key tenets of the ACS was that connections between streams and rivers and their associated floodplain and wetland habitats should be protected and, if necessary, restored (Reeves et al. 2006). In valleys where rivers are unconstrained and riparian forests are well developed, off-channel habitats such as braided streams, oxbow lakes, springs, and other floodplain features provide important seasonal rearing habitats for a wide variety of aquatic and terrestrial species and are considered to be among the most biophysically complex and diverse systems on Earth (Bayley 1995). Additionally, they can be important areas of carbon storage (Sutfin et al. 2016). Flood pulses that redistribute sediment and organic matter create a dynamic mosaic of physical habitat features on floodplains (Junk et al. 1989, Stanford et al. 2005), which support diverse and productive biological communities. In forested regions of the Pacific Northwest, flood-induced channel migration creates a variety of aquatic habitat patches that differ in age and connectivity with the main channel, from connected side channels that reside within the active flood zone to disconnected side channels that become connected only during larger flood events.

Flood-induced erosion and deposition of substrate also create dynamic and heterogeneous plant communities. Early-successional species such as alder, willow, and cottonwood are generally found on newly deposited sediments, whereas mixed-species (deciduous and coniferous) mature forests and old-growth coniferous forests are found on older and more stable floodplain surfaces (Naiman et al. 2010). This spatial heterogeneity can also create highly complex and spatially structured food webs (Bellmore et al. 2013), which may be important for mediating the strength of predator-prey interactions and promoting biodiversity and resilience (Bellmore et al. 2015)

In the context of large-scale environmental stressors such as climate change, intact floodplains may be hubs of ecological resilience. The biological and physical diversity found across floodplains may promote ecological resilience in river networks via at least two pathways. First, enhanced species diversity in floodplains may provide functional redundancy within species guilds, whereby individual species extirpations may not significantly reduce ecological function (e.g., primary/secondary production, nutrient cycling) until some critical threshold is exceeded (Walker 1992). Second, the physical heterogeneity or spatial complexity found across floodplains may provide critical refugia for individual species (Boughton and Pike 2013, Sloat et al. 2017). For example, groundwater upwelling in floodplain springbrooks can provide cold-water thermal refugia when main-channel waters exceed thermal optimums for a particular species (Ebersole et al. 2003, Torgersen et al. 1999). Unfortunately, many river-floodplain systems have been severely altered by human disturbance, which has constrained the physical processes that create and maintain habitat heterogeneity in floodplains (Tockner and Stanford 2002), and the associated resilience these habitats may provide. Although active restoration efforts are frequently targeted at recreating specific floodplain habitats (e.g., side channels), the reestablishment of natural channel-forming processes (Beechie et al. 2013), such as the "natural flow regime" (Poff et al. 1997), may be most successful at restoring the biophysical complexity of floodplains throughout the stream network over the long term and help negate potential effects of climate change.

Winners and losers—
Climate change is projected to lead to changes in the distribution and abundance of native fishes and a host of other aquatic-riparian organisms in the NWFP area. Some species will be adversely affected by climate-mediated shifts in environmental conditions; others may actually benefit from the changes. Whether conditions will become more or less favorable for a particular species depends on physiological requirements, life-history and migratory patterns, habitat preferences, shifts in aquatic-community composition, and geographic location within the region covered by the NWFP. In general, we expect that fishes that prefer warm water and benefit from alterations in aquatic food webs and hydrologic regimes that accompany climate change will likely increase in abundance and expand their ranges. Other native fishes that prefer cool water will likely suffer losses from recently established predators and competitors; elements of their habitats that are needed at different points in their life cycles will likely decrease in abundance; and their ranges will either contract or shift northward. Population fragmentation in cool-water fishes is also likely to increase as favorable thermal conditions retreat to higher elevations, and smaller populations may suffer reduced genetic variability that threatens long-term survival (Kovach et al. 2015).

For anadromous species, survival and growth at sea will depend on how climate change alters upwelling patterns, plankton blooms, forage-fish populations, predator abundance, and other potentially limiting variables. In the fisheries management community, there is no clear consensus on whether freshwater or marine environments are "more important" to regulating the abundance of Pacific salmon, but it has become apparent that both ecosystems can exert a strong influence on run size, and that there are many uncertainties about how these two ecosystems interact to govern population viability and resilience.

In the NWFP area, climate change will lead to freshwater alterations that will be more or less favorable for some fish species relative to others. In figure 7-19, we list life-history strategies of fish that could increase vulnerability to the types of habitat change discussed earlier in this chapter. These include inflexible habitat specialization; extended freshwater rearing (1 year or more); low movement and spawning stray rates; potential for extended exposure to high water temperatures in their preferred habitats; and autumn spawning, placing them at risk of exposure to flow extremes. We also list life-history and habitat requirements that are likely to fare better in future climates. These include being able to use many different habitat types (habitat generalist); an abbreviated period in fresh water prior to seaward migration; high movement and spawning stray rates; either brief exposure to high water temperatures or a tolerance of prolonged elevated temperatures; and spring spawning that occurs after peak winter flows. Fishes are then arrayed along a risk scale, ranging from those we believe to be less vulnerable to harm from climate change to those that may be moderately vulnerable, and finally to those that may be at high risk of long-term harm. No species possesses all life-history attributes that are well adapted to thriving under predicted climate regimes, just as no species possesses only attributes that are ill-adapted to all projected future conditions. However, based on what is known about climate-related trends in freshwater habitats and on detailed knowledge of the life-history requirements of native Pacific Northwest fishes, we suggest that there will be winners and losers among fish assemblages. To some extent, the NWFP addresses many of the habitat changes likely to be associated

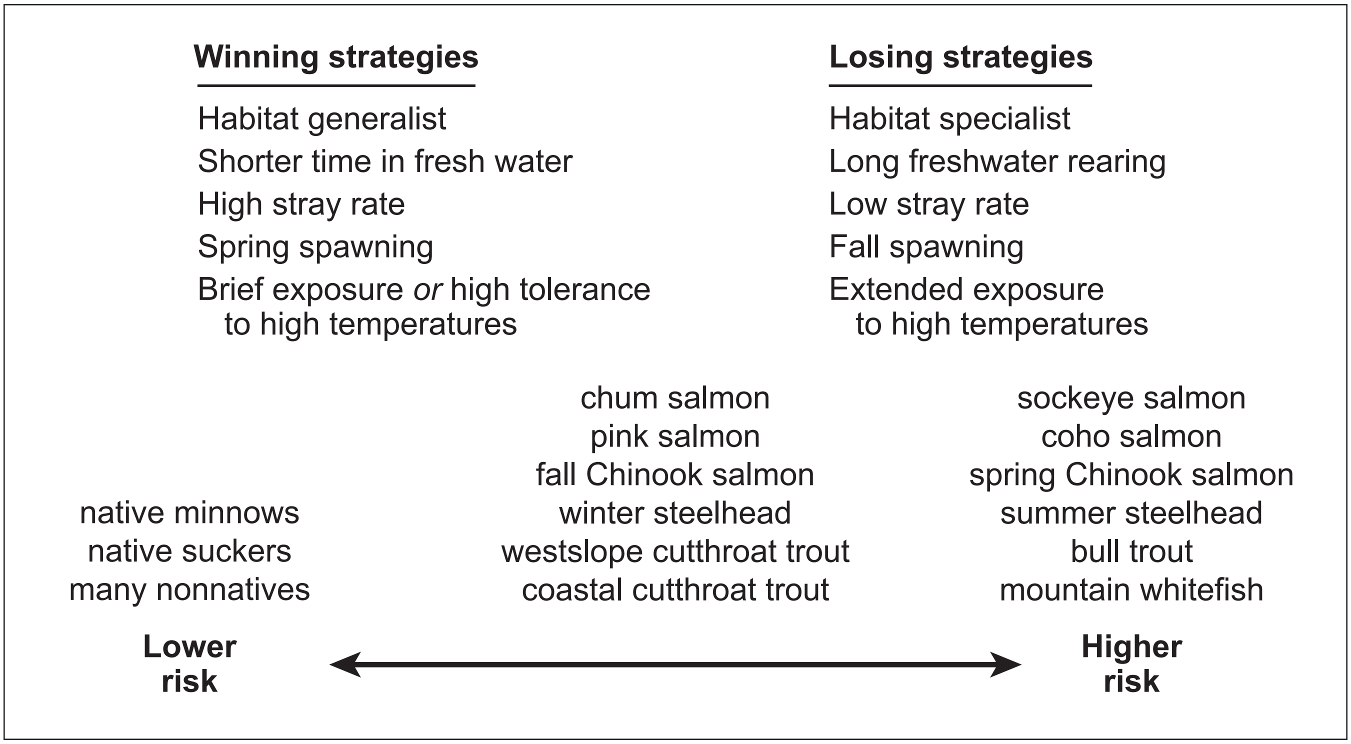

Figure 7-19—Life cycle and habitat-preference strategies of freshwater fishes that are considered in this report to be favored (“winning”) and disfavored (“losing”) in future climates of the Northwest Forest Plan area. Beneath the lists of winning and losing strategies is a grouping of fishes along a gradient of low to high risk from climate effects. These groupings, which are somewhat subjective, are based on current knowledge of each species’ life histories, spawning and rearing locations in watersheds, and residence time in fresh water.

with climate-related alterations in federally managed forests by creating and maintaining functional riparian areas within a watershed and focusing on road removal. However, some changes (e.g., trends in marine conditions) will not be materially affected by NWFP implementation.

The geographical distribution of native fishes and the variation in their life histories, combined with the wide range of effects of climate change on freshwater environments, make it difficult to predict which species will benefit most from NWFP aquatic-habitat protections. In figure 7-20, we divided the NWFP area into four zones: eastern, western, northern, and southern. The western zone includes watersheds draining coastal mountain ranges, whereas the eastern zone includes central lowlands of the NWFP area (Puget Sound, Willamette Valley, and California’s Central Valley) and western drainages of the Cascade Range and Siskiyou Mountains. The northern zone includes all river systems north of the Columbia River; the southern zone includes river systems southward into the Sacramento River. The zones are not mutually exclusive because the northern and southern zones include both eastern and western areas; however, some fishes occur primarily in western coastal systems and others are found primarily in eastern portions of the NWFP area.

Based on different types of improvements to aquatic habitats from implementation of the NWFP that mitigate harmful effects of climate change as discussed above, figure 7-20 lists native salmonids that are likely to benefit in some way from the framework and standards and guidelines introduced by the NWFP. A few of the fishes (e.g., Chinook and coho salmon, steelhead [anadromous *O. mykiss*], and coastal cutthroat trout [*O. clarkii clarkii*]) are found throughout the NWFP area and therefore occur on each list; others (e.g., westslope cutthroat trout, *O. clarkii lewisi*) are limited to relatively small regions of the NWFP area. Figure 7-20 does not include nonnative species or nonsalmonids. In general, nonsalmonids (e.g., native minnows and suckers) are likely to benefit from climate warming (although see

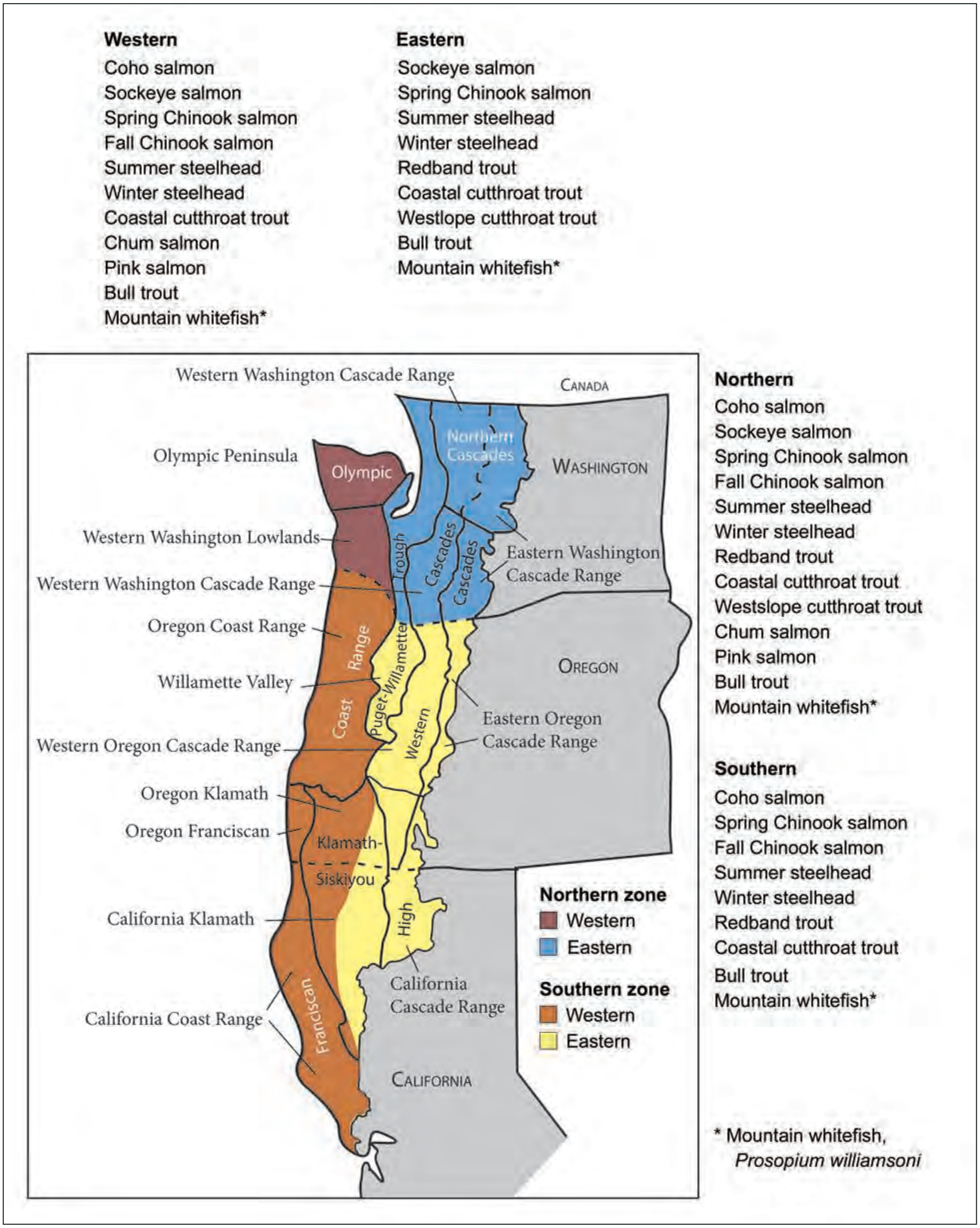

Figure 7-20—Native salmonid fishes in the Northwest Forest Plan area that are likely to benefit in some way from environmental protections from the harmful effects of climate change, grouped by different geographical zones (see text).

Moyle et al. [2013], who suggested that this may not be true in California) and may or may not respond to NWFP aquatic-habitat improvements such as fish-passage barrier removal. Nonnative salmonids (e.g., introduced chars—brook and lake trout, *Salvelinus fontinalis* and *S. namaycush*) will probably be adversely affected by climate change, but also may or may not benefit from NWFP actions. Other introduced species, especially warmwater fishes (e.g., sunfishes and basses, Centrarchidae and *Micropterus* spp.) will likely become more abundant and may increase the risk of predation, competition, and exotic disease exposure to native fishes. However, restoration of riparian habitats may reduce water temperatures and restrict expansion of these fish (Lawrence et al. 2014). Also, the effects of hatchery fish may reduce the potential of wild populations to respond to climate change (Quiñones et al. 2014a, 2014b).

Climate refugia can also be projected for amphibian species (Shoo et al. 2011), with myriad ecological consequences. For lentic-breeding amphibians in the NWFP area, higher-elevation-adapted Cascades frogs (*R. cascadae*) may be faced with shifts in their breeding-habitat conditions. In addition, they may encounter novel interactions with species associated with warmer, lower elevation habitats, such as native northern red-legged frogs (*Rana aurora*), which may spread to higher elevations with altered climate. The low- to mid-elevation-adapted red-legged frogs may in turn encounter invasive American bullfrogs (*Lithobates catesbeianus*), which now occur at warm, low elevations. Similarly, because mountain streamflows are projected to change, torrent salamanders (*Rhyacotriton* spp.) associated with intermittent streams, could have a truncated active season, retreating below ground as small streams dry earlier in the season, possibly affecting survival and reproduction.[9] They may also move downstream and be faced with new interactions with larger predatory salamanders or fish in perennial reaches. If they migrate downstream, their over-ridge dispersal to new watersheds may be affected, as distances between flowing water bodies increase. Hence their populations could become more isolated and vulnerable to stochastic events. For terrestrial-breeding salamanders, we can project the consequence of warmer, drier conditions by examining the distribution of current species in the drier portion of the Northwest; these are species for which climate change has already occurred. Optimal habitat for the Siskiyou Mountains salamander (*Plethodon stormi*) is modeled to occur on the shaded side of mountain ridges and in cooler riparian areas in the dry and warm southern Oregon landscape (Suzuki et al. 2008), and the black salamander (*Aneides flavipunctatus*) appears to become a riparian associate in dry portions of its range (Nauman and Olson 2004). Hence for cool, moisture-dependent species, riparian areas and north-facing slopes with hill shading may become more important with projected changes in climate. Alternatively, as for torrent salamanders, their activity pattern may be altered, with reduced surface activities during dry times and possible consequences for survival. Range shifts for temperature- and moisture-dependent species have also been projected for pathogens of aquatic organisms, such as the amphibian chytrid fungus (*Batrachochytrium dendrobatidis*), which is predicted to increase in occurrence probability in the NWFP area with climate change (Xie et al. 2016).

Implementation of the NWFP represented a significant change in the approach to protection and management of freshwater habitats in federal forests of the Pacific Northwest. Although not directed at mitigating the negative effects of climate change on native aquatic organisms at its outset, the protections provided under the standards and guidelines of the NWFP will benefit populations of native coldwater fishes throughout their life cycles and will help maintain the diverse mosaic of habitat types on the landscape that is essential for population resilience (Beechie et al. 2013, Bisson et al. 2009). However, although many aquatic and riparian habitats in federal forests are likely to retain favorable conditions for aquatic-riparian biota or to slowly improve as watershed-restoration actions are undertaken, it is important to recognize that federally managed forests are usually embedded in a landscape that includes many different types of landowners and uses, and that the standard of environmental protection for other lands is quite different from ACS-based standards and guidelines of the NWFP (Reeves et al. 2016a). Climate-related changes

[9] Unpublished data. On file with: Deanna Olson, Forestry Sciences Laboratory, 3200 SW Jefferson Way, Corvallis, OR 97331.

in aquatic and riparian habitats on nonfederal lands may be much less favorable for native aquatic organisms and more favorable for a variety of nonnative species.

As the biological communities of whole river systems are transformed under a changing climate, there will be a continuing need to monitor the role that federal forests play in conserving native aquatic organisms in the NWFP area. It will be critical for planners to identify vulnerabilities to climate change, and to incorporate approaches that allow management adjustments as the effects of climate change become apparent (Joyce et al. 2009). Because of the nature of environmental variability, the inevitability of novelty and surprise, and the range of management objectives and situations across the NWFP area, no single approach will fit all situations. A range of management options could include practices focused on mitigating or negating the effects of climate change by building resistance and resilience into current ecosystems, and on managing for change by enabling ecosystems and associated biota to adapt to climate change (Joyce et al. 2009, Perry et al. 2015). Better and more widespread implementation of already known practices that reduce the effects of existing stressors represents an important "no-regrets" strategy (Joyce et al. 2009). These management opportunities will require consideration of the Forest Service's adaptive capacity, including availability of personnel with the expertise to conduct required technical analyses, and being able to work cooperatively with the public and other federal agencies to develop and implement the resulting management strategies.

The marine environment is likely to be a major challenge for Pacific salmon in the NWFP area. The predicted effects of climate change on the oceans, including acidification and increased temperatures, and their potential ecological consequences, reduced survival and size of returning adult fish, were described earlier. Pacific salmon have survived climate shifts in the past (Waples et al. 2009) and likely have the ability to persist in many areas of their current range even under more pessimistic climate change scenarios. Salmonid populations exhibit large genetic and phenotypic diversity relative to many other bony fishes (Crozier et al. 2008, Schindler et al. 2010, Waples 1991) and can adapt to changing conditions rapidly (Healey and Prince 1995, Quinn et al. 2001). This diversity has allowed for persistence in highly dynamic and ecologically diverse environments in the past (Greene et al. 2009, Moore et al. 2014, Waples et al. 2009) and will be a key to future survival (Copeland and Vendetti 2009, Mangel 1994). However, we note that Gienapp et al. (2008) cautioned that our knowledge about the role of genetic variation and the ability of natural populations to respond adaptively to current and future environmental change is limited, and that assuming that adaptation can or will happen is risky because of the uncertain rate and extent of climate change, effects of invasive species, and altered ecological processes. The challenge to managers will be to conserve natural environmental complexity in space and time so it can provide the physical template for maintaining genotypic and phenotypic diversity in populations that are currently strong, or to restore environmental complexity where it is currently compromised.

Research Needs, Uncertainties, Information Gaps, and Limitations

The scientific basis of the ACS is still sound and is supported by new science produced since its inception by FEMAT in 1993. However, we have learned much about relationships of riparian vegetation to stream habitats and environments that has refined and modified some hypotheses that were used to develop the ACS in the early 1990s. A major knowledge gain has related to the behavior of aquatic and riparian ecosystems in space and time. At the time the ACS was developed, it was assumed that these systems were relatively stable through time. However, recent science is suggesting that these systems may be very dynamic in space and time, similar to terrestrial systems, and that aquatic organisms are adapted to this dynamism. Implementing this perspective in management actions will be challenging. It is not consistent with many current regulatory approaches, which require aquatic and riparian ecosystems to meet a given standard. Also, a dynamic perspective could be incorporated into the requirements for range of natural variability and all lands consideration of the 2012 planning rule, but will likely require close coordination between managers and researchers.

Emerging science also suggests that the absence of disturbance and management in upland terrestrial ecosystems, primarily fire, may be affecting vegetation, and combined with climate change, is likely altering these ecosystems (see chapters 2 and 3). The same trends are likely occurring in riparian and aquatic ecosystems in a manner that is not fully understood at present; this could be a useful subject for research conducted in an adaptive-management context to provide information to managers, regulators, and policymakers in a timely manner.

Climate change is expected to affect aquatic and riparian ecosystems throughout the NWFP area, though with much uncertainty. Effects will likely differ widely within and among watersheds and geographic areas, necessitating the development of new approaches to identify this variation and help craft strategies and programs for mitigation and adaptation. Much of the focus has been on individual species. Research that focuses on understanding potential effects over the life history of species and how effects may cascade through life-history stages, as well as consideration of community-level effects, is critical. Understanding the effects on water quantity and quality is also important, particularly across spatial scales within watersheds, among watersheds, and across seasons and years. It is likely that aquatic and associated terrestrial ecosystems will change in uncertain, and maybe unpredictable, ways under a changing climate, and that this change will vary widely across the NWFP area. Having the capacity to do the needed analysis will also be critical for the involved agencies to successfully meet this challenge in a timely and effective manner, particularly in an era when budgets and personnel for federal land-management agencies are declining (see chapter 8). Thus, development of cost-effective and scientifically sound analysis procedures performed with close collaboration between research and management is key to addressing this need.

The contribution of federal lands to the conservation and recovery of ESA-listed fish continues to be important. However, federal lands alone are likely to be insufficient in geographic scope to reach the comprehensive goals of the NWFP relative to recovery of listed fish, particularly many evolutionarily significant units of Pacific salmon, as originally expected by FEMAT (1993) and the record of decision (USDA and USDI 1994a). Although the geomorphic setting of streams on federal lands may be as capable as originally expected of providing sufficient favorable habitat, particularly for salmon, streams on state and private lands may have a much greater potential to provide habitat in many watersheds. Thus, it will be important to work closely with adjoining landowners and other interested parties to develop more comprehensive efforts across species ranges. The development of incentive programs is likely to be important to build partnerships for fish-habitat management across land ownerships. Developing an understanding of the variation in the capacity of watersheds to provide favorable conditions for fish and other aquatic biota could be critical to the success of such programs.

Monitoring the effectiveness of the ACS will continue to be important. Some meaningful uncertainties remain regarding the aquatic-riparian monitoring approaches, especially relative to whether they are capable of capturing the effects of the ACS on a wide range of ecological processes and species of aquatic and riparian ecosystems. Research is needed to test the ecological validity of individual metrics and different ways of combining metrics to represent different components of complex and diverse aquatic and riparian ecosystems and communities. It would, therefore, be prudent to compare alternative approaches in the face of the new understanding about the behavior of aquatic and riparian ecosystems in time and space and the yet-to-be-understood effects of disturbance or lack of disturbance, climate change, and novel ecosystems. A related research need is to better understand the relationship of the productivity of aquatic biota, which include organisms other than salmonids, in the context of different upland vegetation and in-channel successional stages or restoration treatments. This type of information can feed into watershed assessments to better ensure that the effects of the ACS are captured more comprehensively relative to the biota that are a key ecosystem service of aquatic-riparian ecosystems. In particular, we lack information about the amount, pattern, and type of restoration activities that have occurred in upland and riparian forests. Implementation monitoring has not been adequate to enable a sufficient understanding of the consequences of restoration actions (or lack of actions), especially relative to how they may have altered aquatic ecosystems in space and time.

Roads and their effects will continue to be a major issue in the NWFP area. Both research on road effects and the continued development of analysis tools such as Geomorphic Roads Analysis and Inventory Package (Black et al. 2012) are important. In particular, understanding the consequences of focusing on small segments rather than the entire network should be a priority. The same is true for effects of culverts on ecological processes and the movement of aquatic biota. These are current priorities given the uncertainties of climate change. Also, understanding how to balance fire management, recreation, and other needs against potential negative aspects of roads will require a concerted cooperative effort of managers and physical, biological, and social scientists.

A key uncertainty that has emerged from our analysis is how to understand and assess the effects of "no-action" management options and tradeoffs of managing for one factor (e.g., water temperature or wood recruitment) on other ecological processes or attributes. The assumption has been that focusing on one concern would not influence other processes or attributes, and that taking no action was synonymous with having no effect. However, these assumptions are questionable and deserve increased consideration and focus by researchers.

Several other topics relating to the components of the ACS merit further research. Watershed analysis could be reexamined so that it is conducted more efficiently and considers the appropriate spatial scales, including a smaller watershed of interest and its context within a larger basin. The larger scale context is particularly relevant for effective landscape-scale planning. In addition, no formal evaluation of the potential effectiveness of the network of key watersheds was conducted during development of the NWFP, nor has such an evaluation been attempted since it was implemented. New concepts, tools, and emerging understandings about aquatic ecosystems are now available to better assess and increase the potential effectiveness of key watersheds. Our understanding of aquatic ecosystems is incomplete (though evolving) at this time, but because there could be significant implications for the productivity of these systems, they will continue to be a major focus of research.

Conclusions and Management Considerations

The goal of the ACS was to maintain and restore aquatic-riparian ecosystems on federal lands within the range of the northern spotted owl. A review of monitoring efforts and the pertinent scientific literature suggests that (1) aquatic ecosystems in the NWFP area are likely improving as expected, albeit slowly; (2) the fundamental tenets and ecological framework of the ACS are sound, and we are gaining more explicit understanding of several components that over time will have important implications for future management; and (3) opportunities exist for implementing parts of the ACS differently while continuing to achieve its goals. The third finding is particularly applicable to the riparian-reserve component of the ACS, where more active management may help to address potential concerns about the effects of the lack of natural disturbance (primarily wildfire), and climate change.

The following is a detailed summary of our main findings and conclusions. We also note to which guiding questions the conclusion applies.

Guiding Questions

1. Is the scientific foundation for the ACS valid, or does the science developed since 1993 suggest potential changes or adjustments that could be made to the ACS?
2. What is the basis of trends observed in the ACS monitoring program, and what are the limitations, uncertainties, and research needs related to monitoring?
3. What is known about variation of characteristics of unmanaged streams and riparian ecosystems in relation to the stream networks across the NWFP area?
4. What has been learned about the effects of riparian vegetation on stream habitat and environment?
5. What effects have human activities had on stream and riparian ecosystems?
6. What is the scientific basis for restoration management in riparian reserves, and how does restoration relate to the ecological goals of the ACS?
7. What is the capacity of federal lands in the NFWP area to contribute water for a suite of economic, recreational, and ecological uses?

8. What are the potential effects of climate change on aquatic ecosystems in the NWFP area, and are they adequately addressed by the ACS?

Science Foundation for the ACS (Question 1)

The scientific foundation of the ACS is generally sound.

1. It is a coarse-filter approach designed to protect and restore ecological processes that create and maintain favorable habitat conditions for native anadromous salmonids. This assumes that if conditions are favorable for these organisms, then they should be suitable for other aquatic and riparian associated organisms.
    a. Verifying these assumptions could be a research priority.
    b. There is growing scientific support for larger scale ecological processes acting at small-to-large watershed scales affecting salmonid habitats and populations; these include landslides delivering sediment and wood, canopy closure, and hill-shading effects on aquatic-riparian temperatures, and the contribution of headwaters to downstream conditions and populations.
    c. The ecological process and species-habitat emphasis areas of the ACS are supported, but since 1993 additional factors have come to the forefront.
        i. More aquatic species have been considered for listing as threatened and endangered, some requiring more focused attention than the regional scale of the ACS, and consideration of threats on a case-by-case basis.
        ii. Aquatic invasives have emerged as an elevated concern because of their effects on native species.
        iii. Anthropogenic disturbances from timber-harvest activities, including road building and maintenance, remain key concerns for aquatic-riparian ecosystems, but new concerns about the extent and severity of wildfire and climate change have emerged as research and monitoring priorities.
        iv. Reliance on federal lands alone cannot address the conservation need to maintain or restore well-distributed populations of all aquatic-riparian species; key salmonid habitats rely on nonfederal lands, and fragmented federal land ownerships affect aquatic-riparian-terrestrial habitat connectivity for organisms dependent upon aquatic-riparian ecosystems.
2. The scientific foundation for the riparian reserve network is valid. The riparian reserve network was intended to identify the outer boundary of the aquatic/riparian ecosystem.
    a. Since 1993, new science supports riparian buffers to maintain aquatic-riparian processes, habitat conditions, and species.
    b. Our ecological knowledge about non-fish-bearing streams has increased tremendously since 1993, and the approach for protecting them is supported.
    c. However, there are suggestions that the second site-potential tree-height on fish-bearing streams may not be required to maintain microclimatic conditions within the first tree-height.
    d. There are potential options available to move away from fixed-width riparian buffers toward riparian management that considers the variability in ecological context within the stream network and specifies management depending on ecological importance and risk.
    e. Passive restoration approaches of riparian forests in streamside buffers have dominated management choices; active restoration might be acceptable in some locations and could accelerate achievement of goals such as growth of large trees to supply key pieces of large wood in the future.
    f. Although science has addressed reach-scale effects of riparian reserves, research on effects of larger scale management activities (e.g., small to large watersheds) is becoming a new research priority.
    g. Implementation, effectiveness, and validation monitoring of NWFP riparian reserves has not formally occurred, and is an emerging priority.

3. The use and structure of the key watershed network are supported by recent science.
   a. There is emerging evidence that the key watersheds do not have the capacity to support and provide favorable habitat for ESA-listed fish to the extent that was originally assumed.
   b. Also, the assumption that habitat conditions in old-growth forests are the most favorable for native salmonids is being questioned.
   c. A review of the key watershed network and the criteria for selecting watersheds would be useful and timely.
4. Watershed analysis remains an important process for developing and assessing management options.
   a. New analytical tools and processes are available that could be used to improve these analyses and make them more cost effective; this is a research priority relative to individual watershed analysis as well as assessment of multiple watersheds across the region that may have differing contexts.
   b. The ability of the Forest Service and other federal land managers to conduct such analyses may be limited by a declining workforce and technical competencey; see chapter 8 for more detailed discussion.
5. A tremendous amount of effort has been directed at restoring degraded watersheds and the associated aquatic and riparian ecosystems.
   a. The vast majority of this effort has been directed toward fish-bearing streams. More effort could be directed at the non-fish-bearing portions of the stream network, which will be important to addressing potential effects of climate change.
   b. Implementation, effectiveness, and validation monitoring is needed to assess restoration activities and contribute to adaptive-management processes.

Monitoring of the ACS (Question 2)

The AREMP results suggest that the condition of aquatic and riparian ecosystems in the NWFP area is improving, albeit slowly, as was originally expected owing to the extensive amount of degradation and lengthy time needed for recovery.

1. The primary reasons for improvement are likely a reduction in the extent of roads, primarily in key watersheds, and an increase in the number of large trees in the riparian reserve.
2. Regionally, there is a signature of wildfire interacting with AREMP restoration criteria in some places in the NWFP area. The ecological significance of this interaction is unclear and merits examination.
3. Assessing watershed condition is inherently challenging, and this synthesis has highlighted a number of areas for further research and management focus:
   a. Use of multiple independent measures of watershed attributes makes it difficult to assess the overall condition of a watershed.
   b. Development of aquatic-riparian monitoring programs requires a clear articulation of which biota and associated functional characteristics of habitats and ecosystems are being considered, tying these to our understanding of patterns of change over space and time, and how they are likely to be altered as a result of the actions of interest.
      i. It is important to clearly describe the ecological context of aquatic-riparian monitoring, for example, to determine to what extent AREMP should focus on environmental parameters that measure habitat for native salmonids or on other aquatic and riparian-dependent organisms.
      ii. Within a dynamic aquatic-riparian ecosystem framework, change is anticipated, but a challenge for monitoring is to assess alterations in conditions that may reflect restoration or other trajectories of patterns in response to a variety of human actions and other events.
   c. A key consideration in development of reference distributions for comparison with current conditions is including the entire natural range of conditions that an ecosystem can experience (natural range of variability). This is critical to be able to evaluate the implications of change.
      i. Reference conditions that are too narrowly or broadly defined can skew the interpretation of monitoring results and introduce uncertainty

into the process. This is an active area of research for the NWFP area: Are there neighborhood, provincial, or regional patterns to consider? We suggest further exploration of the use of reference conditions and their potential utility for diverse analytical approaches, including consideration of how to use them in concert with state-transition models and the potential development of novel conditions in the future.

d. Refinement of the objectives and approaches of aquatic-riparian monitoring programs, including AREMP, is anticipated as our understanding of these ecological systems improve and new analytical tools are developed. Advances in watershed-condition assessment procedures will be important to ensure the validity and reduce the uncertainty of future results and their implications for management.

Aquatic and Riparian Ecosystems (Questions 3–6)

1. Unaltered aquatic and riparian ecosystems likely exhibit a wide range of conditions in space and time, locally and across the NWFP area, depending to a large degree on the magnitude and frequency of the associated disturbance regime. (See chapter 3 for more details.)
    a. Headwater streams tend to be dominated primarily by conifers much of the time.
        i. The biological processing of vegetation that falls into the stream (allochthonous material) is a primary energy source for downstream fish-bearing streams.
    b. In the middle parts of the stream network, the riparian zone is composed of a mixture of conifers and deciduous hardwoods.
        i. Hardwoods are important sources of high-quality allochthonous material important for system productivity.
        ii. Hardwoods are scarce in many areas because of the conversion of riparian areas to conifer-dominated plantations. There may be important implications to system productivity that need to be explored.
2. Human impacts have extensively altered riparian ecosystems.
    a. An estimated 30 to 50 percent of the riparian reserve has been converted to single-species plantations, primarily conifer, as a result of past management that harvested trees to the edge of the streams throughout the network. In headwater streams that have experienced increased rates of landsliding, riparian zones are frequently dominated by alder.
    b. The trajectory of riparian and aquatic ecosystems has been altered as a result, reducing ecological variability across the area of the NWFP.
    c. Additionally, fire exclusion (see chapter 3) and climate change are likely altering aquatic and riparian ecosystems in ways that are not fully recognized or appreciated at this time.
3. Restoration activities in riparian reserves have been limited because of concerns about potential negative effects, particularly increased water temperatures and decreased wood-delivery potential, and lack of trust of the Forest Service (see chapter 12).
    a. Restoration activities have primarily been restricted to fish-bearing streams.
        i. Assessment of these activities has been extremely limited, so it is not possible to quantify the effects.
        ii. More active management may be needed.
        iii. The question of whether the increased risk is sufficiently offset by the long-term gains realized from active restoration or other activities within portions of the riparian buffers is a key research need.
    b. Passive restoration has been the dominant policy in riparian reserves.
    c. This approach assumes that “no activity equals no effect.” However, this assumption is questionable, and “no activity” may actually compromise or eliminate key ecological processes such as development of the largest trees.

4. Non-fish-bearing streams have received little attention in terms of restoration.
    a. These streams can be important sources of large wood for streams lower in the network. Improving stand conditions is critical to maintaining this important ecological function.

Water Contributions From Federal Lands (Question 7)

The capacity of federal lands in the NWFP area to contribute water for a suite of purposes varies widely among forests, ranging from less than 20 percent of the total flow in some basins to more than 40 percent in others.

Climate Change (Question 8)

The primary effects of climate change in the NWFP area will be increased water temperatures, decreased streamflows in summer, and increased winter streamflows. The extent of these effects will vary widely depending on location and local topographic features.

1. The ACS has the potential to meet these challenges, but it will take a focused effort to do so, including:
    b. Conducting local-scale analyses.
    c. Considering "all lands."
    d. Shifting the focus of management and restoration from increasing population sizes to increasing the life-history diversity of aquatic and riparian organisms.
    e. Recognizing that there will be "winners" and "losers"—because of their inherent capacity to adapt, some organisms will increase while others are likely to decrease.
2. With regard to anadromous fish, changes in ocean conditions (water temperature, acidification, and timing of upwelling), which are beyond the capacity of the federal land-management agencies to influence, may exert a stronger influence on populations than changes in freshwater ecosystems.

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Appendix 1: Aquatic-Riparian Invasive Species of the Northwest Forest Plan Area

Deanna H. Olson and Rebecca L. Flitcroft[1]

Invasive species are generally considered novel species that are not native to established systems, and their introductions are harmful ecologically (Vitousek et al. 1997) or economically (Pimentel et al. 2000). Nuisance-species lists have been developed for various jurisdictions, including species that both have been or potentially could be introduced to an area with subsequent adverse effects. Priority aquatic-riparian invasive species (ARIS) include those that have the potential to greatly alter food webs or ecosystem structure, economic interests such as fisheries, and recreation opportunities or human safety—for example, by fouling waterways or affecting water transportation. Priority invasive species include pathogens that can trigger disease die-offs, predators that may restructure native communities via trophic cascades, ecosystem engineers that alter physical or biological habitat conditions, and macroinvertebrates and plants that may produce population booms in systems, altering their ecosystem structure or function.

ARIS were not raised as a priority concern during development of the Aquatic Conservation Strategy (ACS) for the Northwest Forest Plan (NWFP, or Plan) in 1993–1994. As described in the 10 ACS objectives (USDA and USDI 1994a), the focus at the time was to maintain and restore watershed, landscape, riparian, and aquatic habitat conditions to which species, populations, and communities are uniquely adapted—hence emphasis was placed on native species. More explicitly, ACS objective 10 refers to the maintenance and restoration of habitat to support well-distributed populations of **native** plant, invertebrate, and vertebrate riparian-dependent species. Since 1994, ARIS concerns have intensified, and several state and federal agency groups with species jurisdictions overlapping the range of the NWFP have been addressing ARIS. In particular, modifications to the Aquatic-Riparian Effectiveness Monitoring Program (AREMP) now address ARIS during the program's annual monitoring efforts. Herein, we provide an overview of ARIS that are priorities for natural resource managers in the NWFP area, highlight key science findings of recent research, and describe the development of invasive species monitoring programs.

Priority Aquatic Invasive Species

Overall, across the Plan area, we identified 63 species and species groups as top regional aquatic-riparian invasive or nuisance-species priorities (table 7-10). Of these, 31 (49 percent) species or species groups were designated as "high concern" and inventoried by AREMP in 2016. Our broader top-priority list of 63 taxa was derived from lists compiled by state government departments in the region, interagency collaborative groups such as state invasive species councils, regional U.S. Forest Service personnel, or other entities identifying nuisance species or emerging infectious diseases. Specifically, our 63 priority taxa include those aquatic-riparian species on Oregon's "100 Worst List" (OISC 2015), Washington's "50 Priority Species" list (WISP 2009), a focal species list for U.S. Forest Service Pacific Northwest Region (Region 6) lands (Flitcroft et al. 2016b; S. Bautista, pers. comm.[2]), and the AREMP list (Raggon 2017). We recognize that top priorities identified by California and the U.S. Forest Service Pacific Southwest Region include other aquatic-riparian taxa, but upon inspection, species identified only in California and not by these other sources appeared to be of lesser immediate concern in northwestern California forests in the Plan area. We acknowledge that some other important California invasive species may merit consideration if our list were to be refined further. Lastly, some pathogens were included here because of their national and international priority status from other entities (Auliya et al. 2016, Bern Convention 2015, Conservation Institute 2013, OIE 2017, Schloegel et al. 2010, USFWS 2016). Note that priority species differ between

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Table 7-10—Aquatic invasive species of concern in the Pacific Northwest states of Oregon and Washington and within the administrative boundaries of the U.S. Forest Service Pacific Northwest Region

Scientific name	Common name	Oregon	Washington	Pacific Northwest Region	AREMP
Pathogens and parasites:					
Phytophthora alni, *P. kernoviae*, *P. pluvialis*, *P. lateralis*; *P. ramorum*	Alder root rot; *Phytophthora* taxon C; needle cast of Douglas-fir, Port Orford cedar root disease, sudden oak death	✓		✓	
Batrachochytrium dendrobatidis, *B. salamandrivorans*	Amphibian chytrid fungi (*Bd*, *Bsal*)				
Bothriocephalus acheilognathi	Asian tapeworm	✓			
Orthomyxoviridae isavirus	Infectious salmon anemia virus (ISAV)	✓			
Ranavirus	Ranavirus				
Rhabdovirus SVCV	Spring viremia of carp virus (SVCV)		✓		
Novirhabdovirus spp.	Viral hemorrhagic septicemia virus (VHSV)	✓	✓		
Myxobolus cerebralis	Whirling disease	✓			
Aquatic plants:					
Lagarosiphon major	African waterweed or African elodea	✓		✓	
Phragmites australis	Common reed	✓	✓		✓
Potamogeton crispus	Curly-leaf pondweed			✓	✓
Butomus umbellatus	Flowering rush	✓		✓	✓
Salvinia molesta	Giant salvinia	✓			✓
Arundo donax	Giant reed				✓
Hydrilla verticillata	Hydrilla, water thyme	✓	✓	✓	✓
Myriophyllum spp. including *M. spicatum, M. aquaticum*	Milfoils: Eurasian, parrotfeather		✓	✓	✓
Lythrum salicaria, Lysimachia vulgaris	Purple loosestrife, garden yellow loosestrife		✓	✓	✓
Phalaris arundinacea; P. arundinacea var. *picta*	Reed canary grass; ribbongrass			✓	
Didymosphenia geminata	Rock snot (Didymo)	✓		✓	✓
Chondrilla juncea	Rush skeletonweed		✓	✓	
Spartina spp. including *S. alterniflora, S. densiflora*	Spartina (cordgrass)	✓	✓		
Prymnesium parvum, Cylindrospermopsis raciborskii	Toxic algae (golden, toxic cyanobacteria)	✓			
Trapa natans	Water chestnut (European)	✓	✓		

Table 7-10—Aquatic invasive species of concern in the Pacific Northwest states of Oregon and Washington and within the administrative boundaries of the U.S. Forest Service Pacific Northwest Region (continued)

Scientific name	Common name	Oregon	Washington	Pacific Northwest Region	AREMP
Ludwigia spp.	Water primrose			✓	✓
Egeria densa; Elodea nuttallii, E. canadensis, E. canadensis × *E. nuttallii* hybrid	Brazilian elodea, western waterweed (*Elodea*)		✓		✓
Iris pseudacorus	Paleyellow iris			✓	✓
Nymphoides peltata	Yellow floating heart	✓		✓	✓
Riparian-terrestrial plants:					
Hedera helix	English ivy			✓	✓
Alliaria petiolata	Garlic mustard		✓		✓
Geranium robertianum, G. lucidum	geraniums (Herb-Robert, shining)			✓	✓
Heracleum mantegazzianum	Giant hogweed	✓	✓		✓
Rubus ulmifolius	Himalayan blackberry		✓	✓	✓
Fallopia japonica var. *japonica; Polygonum bohemicum*	Knotweeds (Japanese, Bohemian)		✓	✓	✓
Pueraria lobata	Kudzu	✓	✓		✓
Clematis vitalba	Old man's beard			✓	✓
Hieracium aurantiacum	Orange hawkweed				✓
Potentilla recta	Sulphur cinquefoil			✓	
Tamarix spp.	Tamarix (salt cedar)		✓	✓	✓
Lamiastrum galeobdolon	Yellow archangel				✓
Aquatic invertebrates:					
Potamocorbula amurensis	Asian clam	✓		✓	✓
Radix auricularia	Big-eared radix			✓	✓
Eriocheir sinensis	Chinese mitten crab	✓	✓		
Cipangopaludina chinensis	Chinese mystery snail			✓	✓
Orconectes spp., *Orconectes virilis, Procambarus* spp.	Crayfish (red swamp, rusty, ringed, virile, marbled, signal, northern)		✓	✓	✓
Carcinus maenas	European green crab		✓		
Potamopyrgus antipodarum	New Zealand mud snail		✓	✓	✓
Philine auriformis	New Zealand sea slug	✓			
Bythotrephes longimanus [*cederstroemi*], *Cercopagis pengoi*	Waterfleas	✓			
Dreissena polymorpha, D. rostriformis bugensis	Zebra and quagga mussels	✓	✓	✓	✓

Table 7-10—Aquatic invasive species of concern in the Pacific Northwest states of Oregon and Washington and within the administrative boundaries of the U.S. Forest Service Pacific Northwest Region (continued)

Scientific name	Common name	Oregon	Washington	Pacific Northwest Region	AREMP
Aquatic vertebrates:					
Lithobates catesbeianus (*Rana catesbeiana*)	American bullfrog		✓	✓	✓
Hypophthalmichthys spp., *Mylopharyngodon piceus*	Asian carp, black carp	✓	✓		
Salmo salar	Atlantic salmon	✓	✓		
Didemnum vexillum	*Didemnum* tunicate		✓		
Chelydra serpentina serpentina	Eastern snapping turtle	✓			
Neogobius melanostomas, Rhinogobius brunneus, Tridentiger bifasciatus	Goby	✓			
Noteigonus crysoleucas	Golden shiner	✓			
Esox spp.	Muskellunge/northern pike	✓			
Gymnocephalus cernuus	Ruffe	✓			
Channa spp.	Snakehead	✓	✓		
Dorosoma petenense	Threadfin shad (yellow tails)	✓			
Riparian-terrestrial vertebrates:					
Sus scrofa	Feral swine	✓	✓	✓	✓
Cygnus olor	Mute swan	✓			
Myocaster coypus	Nutria		✓	✓	✓

Note: Three aquatic pathogens (ranavirus, *Batrachochytrium* spp.) are included in this table owing to other national and international priority status (Auliya et al. 2016, Bern Convention 2015, Conservation Institute 2013, OIE 2016, Schloegel et al. 2010, USFWS 2016). Some species clustering within rows was conducted. "✓" denotes priority species from authority listed (above), not occurrence of species within jurisdiction.

Source: Flitcroft et al. 2016b; Bautista, S., personal communication (see footnote 2 on page 585); and Aquatic-Riparian Effectiveness Monitoring Program (AREMP) of the Northwest Forest Plan (Raggon 2017).

lists created by different jurisdictions or entities because of their variable selection criteria or jurisdiction-specific habitats and issues. A species listed as a priority for one jurisdiction and not another may have established populations and be of concern in both areas, yet because of different perspectives not be considered a top priority everywhere. Hence, the broader species list may be important to consider as regionally representative taxa of ecological or economic concern from an all-lands perspective across the Plan area. A few estuarine species are included and may be relevant to consider here, because tidally influenced areas are critical ecosystems interfacing with the coastal forest land base. Other primarily marine-associated species are not included here.

Altogether, as potentially representative of the Plan area, Northwest taxa identified as regional ARIS priorities fall into six categories (table 7-10): 8 pathogens; 19 aquatic plants; 12 riparian-terrestrial plants; 10 aquatic invertebrates; 11 aquatic vertebrates; and 3 riparian-terrestrial vertebrates. Specifically for the Plan area, AREMP's 31 invasive taxa fall into 5 categories: 13 aquatic plants; 11 riparian-terrestrial plants; 6 aquatic invertebrates; 1 aquatic vertebrate; and 2 riparian-terrestrial vertebrates.

Taxonomic Summaries

Pathogens and Parasites

Pathogens and parasites are considered invasive when their spread has been documented coincident with devastating disease effects on host species. There is heightened concern for disease-causing pathogens and parasites affecting sensitive host taxonomic groups that provide important ecosystem services (goods and services that people desire) (Blahna et al. 2017, Penaluna et al. 2016). These host taxa include culturally and economically important species such as salmonid fishes; species with broad distributions that may be central to ecosystem structural integrity and biodiversity such as Douglas-fir (*Pseudotsuga menziesii*) and alder (*Alnus* spp.); and those diseases with multiple host taxa that could affect several native species, with consequences for the organization of wild, native communities, such as the fungi that cause amphibian chytridiomycosis.

Northwest aquatic-riparian pathogens of key concern are viruses and fungi; parasites include cnidarian and cestode worms. Aquatic invasive pathogen species infect vertebrates, for example: (1) cnidarian myxosporean parasites (*Myxobolus cerebalis*) cause whirling disease in salmonid fishes (first described in Germany); (2) Asian tapeworms infect cyprinid fishes in their native range in Asia; (3) viral hemorrhagic septicemia virus (VHSV) infects salmonids, historically known from Europe; (4) ranavirus is considered an emerging infectious disease of fishes, turtles, and amphibians; and (5) chytrid fungi of the genus *Batrachochytrium* can cause the emerging infectious disease chytridiomycosis in amphibians. Riparian pathogens of key concern are fungi of the genus *Phytophthora* (table 7-10) that infect trees typical of riparian zones, such as alder, Douglas-fir, and Port Orford cedar (*Chaemaecyparis lawsoniana*).

The World Organization of Animal Health (OIE) lists species as notifiable because of the extent of their effects, the availability of diagnostic tests for detection, and the role of humans in disease spread. The United States is one of 180 member nations of OIE, hence OIE listing is relevant for consideration here. For pathogens identified here as top ARIS priorities in Oregon, Washington, and Alaska, three are OIE notifiable (OIE 2017): VHSV, ranavirus, and *B. dendrobatidis* (*Bd*). *Bd* is also listed by the Conservation Institute (2013) on their world list of the Top 100 Invasive Species. A second amphibian chytrid fungus included on our Top 60 list, *B. salamandrivorans* (*Bsal*), was recently described in Europe (Martel et al. 2013), and early challenge experiments found numerous North American taxa to be vulnerable to disease effects, with rapid mortality after infection (Martel et al. 2014). The Bern Convention (2015) and others (Auliya et al. 2016) have endorsed legislation to forestall the spread of *Bsal*, and an interim rule to the U.S. Lacey Act (USFWS 2016) has listed host salamander species that may be susceptible to *Bsal* infection as injurious, hence restricting their transportation among jurisdictions. *Bsal* risk models (Richgels et al. 2016, Yap et al. 2015) show Oregon and Washington to be extremely vulnerable to *Bsal* introduction owing to the presence of susceptable host amphibian taxa such as the rough-skinned newt (*Taricha granulosa*), suitable *Bsal* habitat conditions, and proximity to U.S. ports of entry. The pet industry has placed a moratorium on some salamander trade imports to the United States, significantly forestalling the transmission of this pathogen.

Aquatic Plants

The 19 aquatic invasive plants of concern (table 7-10) include two algae, a diatom (rock snot or Didymo), multiple species of submerged aquatic plants (e.g., *Elodea*, *Hydrilla*, milfoils [*Myriophyllum* spp.]), emergent plants (e.g., reeds, cordgrass [*Spartina* spp.], loosestrife [*Lysimachia* spp.], rushes, *Salvinia*, reed canarygrass [*Phalaris arundinacea*], paleyellow iris [*Iris pseudacorus*], water primrose [*Ludwigia* spp.], and floating plants (e.g., curly-leaf pondweed [*Potamogeton crispus*], water chestnut [*Trapa natans*], yellow floating heart [*Nymphoides peltata*]). Once established, these taxa may affect ecosystems, water quality, human health, navigation, and recreation. Emergent plants can dominate wetland and floodplain areas, outcompeting or displacing native species, thereby reducing biodiversity and altering ecosystem functions. Toxic algae are a health concern for native vertebrates and humans because they create powerful toxins known to kill fish, ducks, geese, marine mammals, and other wildlife (Edwards 1999). The diatom commonly called rock snot (Didymo) is native to the Pacific Northwest but is included on priority invasive species lists owing to a change in its growth habits in the mid-1980s (Bothwell et al. 2014), becoming more prolific in its distribution and affecting recreational experiences and activities. Some Northwest ARIS plants were initially brought to the region by the aquarium trade (the elodeas) or for ornamental use (reed canary grass), and then spread to other areas. Further, established populations may be spread by waterfowl as they move from one location to another, or by human vectors (e.g., boats and fishing gear/tackle).

Riparian-Terrestrial Plants

The 12 invasive riparian plants listed (table 7-10) are problematic in both upland and riparian environments. Species such as Himalayan blackberry (*Rubus armeniacus*), Japanese knotweed (*Fallopia japonica* var. *japonica*), and giant hogweed (*Heracleum mantegazzianum*) tend to shade out smaller native plants, reducing plant diversity and altering habitat and food resources for native wildlife. For example, Japanese knotweed (native to Europe and Asia) and giant hogweed (native to the Caucasus region of Eurasia) can grow as tall as 15 to 20 ft (4.5 to 6 m) and spread rapidly. Japanese knotweed is known globally as one of the world's most destructive invasive species because its large underground root system can damage structures, walls, and architectural sites, and reduce stream-channel capacity. Giant hogweed is considered a public-health hazard because it causes a phototoxic reaction when skin is exposed to sap and ultraviolet radiation. Species in the genus *Tamarix* are riparian shrubs or small trees that are aggressively invasive and well known in the Southwestern United States. These riparian trees are known to decrease streamflows, lower biodiversity, and create salinization issues, among other problems. Some of the listed invasive riparian plants were imported to the Northwest as ornamentals and have become invasive (e.g., English ivy [*Hedera helix*] and old man's beard [*Clematis vitabla*], native to the United Kingdom; garlic mustard [*Alliaria petiolate*], native to Europe and Asia). Garlic mustard was initially introduced to the east coast of North America as a medicinal herb, but it has spread through forest understories, where it competes with native species.

Aquatic Invertebrates

The 10 invertebrates on the northwest aquatic-riparian invasive species list (table 7-10) include several mollusks (Asian clam [*Potamocorbula amurensis*], big-eared radix [*Radix auricularia*], Chinese mystery snail [*Cipangopaludina chinensis*), New Zealand mud snail [*Potamopyrgus antipodarum*), New Zealand sea slug [*Philine auriformis*), zebra mussel [*Dreissena polymorpha*), quagga mussel [*D. rostriformis bugensis*]), and crustaceans (crayfish and crab species; waterfleas). Mollusks may spread rapidly and attain large population sizes that displace native species. These taxa can deplete prey resources rapidly, affecting foundation levels of food webs (algae, phytoplankton) in aquatic systems. Along with abundant populations come abundant waste products—in some systems, the tissues or waste products of zebra mussels may accumulate contaminants to 300,000 times the level available in the habitat they occupy, with subsequent effects on their environment, including their predators (Snyder et al. 1997). Another concern is that large numbers of some mollusks can foul human structures. Introductions of some species are likely tied to inadvertent human transmission, such as in ship ballast water or on boats or fishing gear (e.g., zebra/quagga mussels, waterfleas, green crabs). Deliberate introduction and consequent escape of some species is also associated with food and medical markets, biological supplies for education, and the aquarium and bait trade (Chinese mystery snails, crayfish, mitten crabs).

Aquatic Vertebrates

One frog (American bullfrog [*Lithobates catesbeianus*]), one turtle, eight fishes, and a tunicate (*Didemnum*) are included in the priority aquatic-riparian invasive species list (table 7-10). These taxa are strongly tied to human introductions. For example, American bullfrogs are native to the Eastern United States and were brought to the West to be farmed for food and out of nostalgia for their calls. Bullfrogs are carriers of the amphibian chytrid fungus *Bd* but do not always exhibit disease symptoms and hence may serve as a reservoir species of the pathogen, another invasive species of concern. Additional concerns surrounding bullfrog introductions include alterations of the native ecosystem via food-web changes, an issue associated with the snapping turtle (*Chelydra serpentina*) as well.

The invasive fishes include a mix of species introduced for human food, as bait for recreational fisheries, or from the aquarium or ornamental industry. Atlantic salmon (*Salmo salar*) are native to the North Atlantic Ocean, where they are anadromous, occurring in the ocean and returning to spawn in rivers. Farms in Washington and British Columbia are thought to be the origin of Atlantic salmon found elsewhere in the Northwest. Concerns arise in conjunction with their potential competition with native salmonids, pollution from the farms, and the potential for farm-raised animals to carry pathogens to native stocks. Gobies are of Asian origin, occurring in fresh and brackish water. They are thought to have been introduced in ballast water, and may compete with or prey upon native species. Golden shiners (*Noteigonus crysoleucas*) are from the Eastern United States and are pond-cultured fishes that are also used as bait. Where numerous, golden shiners may result in displacement of native species.

Didemnum vexillum is commonly called the carpet sea squirt, or ascidian. It is a colonial tunicate in the chordate phylum, hence is included here together with vertebrates—a sister chordate lineage. It seems to be native to Japan, and has been detected along the Washington coast since 2009, in two Oregon bays since 2010, and near Sitka, Alaska, in 2010. It is a fouling organism in marine and estuarine systems that grows rapidly to cover vast surfaces as mats, displacing native biota and encrusting dock pilings and aquatic equipment. It can be introduced in ballast water, or may hitchhike on the hulls of boats or on commercial shellfish stock or equipment.

Riparian-Terrestrial Vertebrates

The category of terrestrial vertebrates is the smallest, with only three species (table 7-10), but these can have extensive aquatic-riparian effects, ecologically and socioeconomically. Feral swine are escaped domestic pigs with rooting behavior that degrades waterway habitat, provides an invasion pathway for nonnative plants, and causes damage to agricultural crops and lands. The mute swan was introduced from New York for aesthetic enjoyment. These aggressive, large (2- to 30-lb [0.9- to 14-kg]) birds may consume significant quantities of aquatic plants, competing with native birds for food and habitat. Nutria (*Myocastor coypus*) were initially brought to the Pacific Northwest for fur farming in the 1920s. After the collapse of this element of the fur industry, escaped and released animals subsequently spread throughout the region. Nutria burrow into the banks of streams and agricultural canals, destabilizing natural stream systems and human agricultural infrastructure, and they consume vegetable crops.

Research and Development, Monitoring, and Management

Research and Development

Invasive-species disturbance ecology has developed conceptually in the past few decades. Aquatic-riparian ecosystems, like their terrestrial counterparts, are heterogeneous in space and time, occurring in multiple states within an ecosystem domain (e.g., Penaluna et al. 2016). This domain is highly resilient to many natural disturbances, yet larger disturbances can push an ecosystem beyond its "tipping point" to a new domain, a novel ecosystem. Novel ecosystems (e.g., Hobbs et al. 2009) are developing on our planet from a variety of disturbances, including the effects of invasive species. As discussed above, invaders may engineer habitat structures and functions, or become key players in food webs and trophic cascades, altering the native community and ecosystem. Biotic homogenization may result when the variety of initial states of ecosystems becomes equalized as a result of domination of invasive species over natives (e.g., McKinney and Lockwood 1999, Olden et al. 2004). "The New Normal" is a pragmatic description of today's ecological systems that seem to be undergoing irrevocable change, a newly developing status quo (Marris 2010). However, it may be premature to characterize such changes as irrevocable, because there are many examples of restoration successes (e.g., Murcia et al. 2014). Nevertheless, without preempting invasions, vigilance at the early stages of invasion, and concerted, often-continuous restoration efforts, transformation to a novel ecosystem can occur.

These concepts are playing out with aquatic invasive species globally. As cases of ARIS are analyzed, costs to native biodiversity are being claimed. For example, the introductions of the European brown trout (*Salmo trutta*) into South America (Soto et al. 2006) and New Zealand (Townsend 1996), and the eastern mosquitofish (*Gambusia holbrooki*) into Australia (Hamer et al. 2002), have caused major reductions in native fishes. Similar adverse effects on native amphibians and other ecosystem components have been documented by fish-stocking practices (reviews: Dunham et al. 2004, Kats and Ferrer 2003). Despite regionwide efforts to control ARIS spread, some species are recognized as requiring continuous management, or the efficacy of control methods is low. As a result, some invasive species (e.g., bullfrogs, New Zealand mud snails, Himalayan blackberry) seem fully established in some watersheds; the specter of a pragmatic New Normal with diminished native aquatic biodiversity in forests may be realized, owing to our lack of capacity to effectively control invasions everywhere. Furthermore, a clear conflict exists between maintenance of native biodiversity and pursuit of high-value recreational fisheries through nonnative fish-stocking programs.

Nevertheless, restoration tools are being applied to maintain habitats for key native species despite nearby invasive species occurrences (Biebighauser 2011). The solution appears to be purposeful management of the multistate ecosystem across landscapes and regions beyond that which has thus far occurred, in order to designate both wild and nonwild states, in which some places retain a semblance of pristine native ecosystems, whereas in other places different ecosystem services (e.g., fishing experiences) can be fostered. This managed multistate condition is likely part of our regional, if not global, future.

In the Plan area, limited research on ARIS has been conducted recently; several examples of case studies or syntheses follow. First, in a study of invasive fishes in the Willamette River, Oregon, LaVigne et al. (2008) documented an increase in invasive fish diversity and abundance since the 1940s. They also noted the significant contribution to overall fish biomass in the river contributed by the top three most common invasive fishes (smallmouth bass [*Micropterus dolomieu*], largemouth bass [*M. salmoides*], and common carp [*Cyprinus carpio*]). They argued for increased river monitoring and the use of double-pass electrofishing as a means of fish capture and eradication. Carey et al. (2011) assessed the threat to native salmonids posed by smallmouth bass. They described the tension between conservation of native salmon and angling opportunities provided by invasive warm-water fishes, such as smallmouth bass. They argued for more specific management that targeted locations for native fishes only, and others in which invasive species would be allowed to enhance angling opportunities. This notion supports the wild versus nonwild ecosystem-management approach described above.

Sanderson et al. (2009) completed a comprehensive assessment of the potential effect of invasive species on Pacific salmon in the Pacific Northwest. They found that invasive species may pose an even greater threat to salmonid persistence in the region than the four traditional factors generally thought to affect abundance and survival of native salmonids (habitat alteration, harvest, hatcheries, and the hydrosystem). They considered invasive-species management to be a significant component of salmonid-recovery planning.

Yamada and Gillespie (2008) described how initial assessments of the effects of European green crab (*Carcinus maenas*) invasion in the Pacific Northwest (in 1989), which predicted that the crabs would naturally die out, have been proven incorrect. Instead, they found that changing environmental conditions and coastal currents stemming from El Niño cycles have resulted in shifting habitat characteristics amenable to the crab, promoting its spread. They concluded that management for crab eradication is a more pressing issue than first thought.

Pearl et al. (2013) also showed that invasive crayfish in the Pacific Northwest displace native crayfish. In their work in the Rogue, Umpqua, and Willamette/Columbia River basins, they found that invasive crayfish (in particular, *Procambarus clarkia*) tended to be associated with anthropogenic effects on streams, and that these crayfish appeared to have a negative effect on occupancy of native crayfish. They argued that there is still time to control invasive nonnative crayfish, but that the window of opportunity for management to have a meaningful effect is closing.

Claeson and Bisson (2013) conducted a study of the efficacy of invasive knotweed removal by herbicidal application, and effects on the riparian plant community in western Washington. They found that sites where knotweed had been removed, followed by passive restoration, had more nonnative species and vegetative cover than reference (no knotweed) sites, and reference sites had more native species. This finding was especially true for riparian areas along larger streams in their sample, as riparian areas along smaller 2^{nd}- to 3^{rd}-order channels had primarily native plant-species assemblages. They suggested active restoration to control secondary invaders, such as replanting native species. Also, they proposed that effectiveness monitoring of invasive species control projects could help to refine and improve restoration approaches.

Kinziger et al. (2014) described establishment pathways of an introduced fish from a nearby source area to two coastal rivers of northern California. Using genetic techniques, they reported that the Eel River invasion of Sacramento pikeminnow (*Ptychocheilus grandis*) likely was the result of only three or four founding individuals, whereas the Elk River invasion likely came from seven founders. This reflects an astounding adaptive capacity for rapid invasion, and highlights the threat posed by such close-range invaders. This species is not included on our priority list (table 7-10).

In a study of western Oregon wetlands, Rowe and Garcia (2014) reported that native anuran amphibians were negatively associated with invasive plant cover, nonnative fish presence, and invasive bullfrog counts. More generally, Bucciarelli et al. (2014) conducted a comprehensive review of the effects of nonnative species on amphibians and broader ecosystem services, painting a complex picture of negative and potentially positive effects. Their conclusion points to the need for additional research on the interactions of native and nonnative species in many ecosystems.

Globally, invasive species experts are using Web-portal technologies to expedite communication among multiple stakeholders, including natural resource management communities, research, and the public sector. Web-portal information can be used to address scientific hypotheses, aid decisionmaking regarding surveillance priorities, and support local-to-regional management actions. To aid communication about emerging invasive pathogens, for example, the Pacific Northwest (PNW) Research Station has partnered with disease and bioinformatics experts internationally to create online data and mapping portals for ranavirus (https://mantle.io/grrs) and the amphibian chytrid fungus *Bd* (Olson et al. 2013) (https://www.Bd-maps.net). The online *Bd* database has been used in subsequent research (e.g., Grant et al. 2016, Xie et al. 2016). For land-management applications, the *Bd* point-locality and watershed-scale occurrence maps have been used during firefighting for decisions about which water sources might be used for water draws and whether water disinfection procedures may be necessary (NWCG 2017). A new portal is being populated now with data on both chytrid fungi, *Bd* and *Bsal*, including both planned and completed research and monitoring reports (www.amphibiandisease.org). In addition, EDDMapS (Center for Invasive Species and Ecosystem Health 2017) is a new, national, real-time tracking system for invasive species that employs global positioning system (GPS)-based mobile applications technology to allow users to report invasive species occurrences.

Monitoring and Management

Northwest Forest Plan implementation has required ACS monitoring, conducted by AREMP, with adjustments and modifications over time to address new knowledge and aquatic priorities. AREMP assessments for invasive species began in 2007 during annual field surveys at watersheds across the NWFP area (Gruendike and Lanigan 2008). The initial focus was on 13 species of primary concern for Northwest national forest waterways, with an additional 14 species considered of secondary concern (27 species total). The number of species assessed during annual sampling of watersheds in the Plan area has since fluctuated between 23 and 41 species, with 38 total species (i.e., 31 species groups in table 7-10) being included in the 2017 survey season. Survey methods have also been modified as needed, examining streams and adjacent banks within the bankfull width of the channel during summer low-flow periods. Surveys now include subsampling for benthic snails, mussels, crayfish, and ARIS plants. Few invasive species have been detected annually (table 7-11), with Himalayan blackberry being the most common species reported. Because the design of the AREMP surveys revisits watersheds every 8 years, some of the detections do not represent

Table 7-11—Invasive species detections by the Aquatic-Riparian Effectiveness Monitoring Program (AREMP) of the Northwest Forest Plan, 2007–2016

Year	Watersheds surveyed	Sites surveyed	Number of invasive species detections	Species detected (number of detections)	Reference
2007	31	149	7	Himalayan blackberry (7)	Gruendike and Lanigan 2008
2008	31	167	17	Himalayan blackberry (15) reed canary grass (2)	Gruendike and Lanigan 2009
2009	28	189	17	Himalayan blackberry (14) ringed crayfish (2) Japanese knotweed (1)	Andersen and Lanigan 2010
2010	28	185	7	Himalayan blackberry (4) reed canary grass (2) Robert geranium (1) Also reported common mullein (*Verbascum thapsus*) from the Region 6 invasive species list	Raggon and Lanigan 2011
2011	29	184	15	Himalayan blackberry (14) English ivy (1)	Raggon and Lanigan 2012a
2012	28	177	10	Himalayan blackberry (9) ringed crayfish (1)	Raggon and Lanigan 2012b
2013	28	187	4	Himalayan blackberry (2) English ivy (1) ringed crayfish (1)	Raggon and Lanigan 2013
2014	27	157	10	Himalayan blackberry (9) herb Robert, geranium (1)	Raggon 2014
2015	34	177	18	Himalayan blackberry (17) parrotfeather watermilfoil (1)	Pennell and Raggon 2016
2016	25	140	10	Himalayan blackberry(10)	Raggon 2017
Overall totals	289	1,712	125	8 species total	
Unique totals	225	1,376			

Note: Ten-year overall totals do not represent unique watersheds and sites, as some resampling among years was conducted as per the AREMP design. Totals of unique watersheds and sites sampled are also provided.

sampling of unique watersheds or sites. Taking unique sites and watersheds into consideration, overall, only 125 invasive species detections of 8 species have occurred across 1,376 unique sites sampled in 225 unique watersheds for the 10 years spanning 2007 to 2016.[3][4]

ARIS monitoring is also a priority for the Aquatic Invasive Species Network (AISN) (AISN 2017). Initiated by the U.S. Geological Survey and Washington State University in 2010 as the Columbia River Basin Aquatic Invasive Species group (CRBAIS 2011), the network integrates federal, state, academic, and tribal organizations over the area of the Columbia River basin, which includes parts of seven states and British Columbia, Canada, and is roughly the size of France. This region overlaps the NWFP area, in watersheds along the western Washington-Oregon border. AISN objectives are to develop an integrated monitoring

[3] Hirsch, C. 2017. Personal communication. Fish Program Manager, Siuslaw National Forest, Corvallis, OR 97331, chirsch@fs.fed.us

[4] Unpublished data. On file with: Aquatic and Riparian Effectiveness Monitoring Program, 3200 SW Jefferson Way, Corvallis, OR 97331.

and information system, coordinate early-detection efforts, assess invasion pathways, and contribute to an evaluation of the effects of climate change on native and nonnative biota (CRBAIS 2011). Zebra and quagga mussels have been the main taxonomic emphasis; the 2016 map of monitoring sites shows no occurrences of zebra and quagga mussels in the NWFP area (AISN 2017).

In 2012, Forest Service Region 6 developed the Regional Aquatic Invasive Species Strategy and Management Plan (USDA FS 2012b). Its three goals were to (1) prevent new introductions of ARIS into waters and riparian areas of the region; (2) limit the spread of established populations of ARIS into uninfested waters; and (3) provide a cooperative environment that encourages coordinated activities among all affected parties throughout the region. The strategy to achieve these goals is multifaceted, including educational and training programs; implementation of biosecurity protocols (e.g., equipment use and cleaning, inspections); mapping of known invasive species occurrences to inform decisions for water draws for firefighting; coordination of inventory and monitoring efforts; and advance knowledge for eradication procedures. The 2012 list of focal species in this document includes 26 species and species groups listed in table 7-10; known species occurrences were mapped in 2012 (USDA FS 2012b).

As part of the Region 6 ARIS strategy (USDA FS 2012b), surveillance and management is conducted by the joint Region 6 and PNW Research Station dive team.[5] Four specific incidences of invasive species establishment in the NWFP area have been addressed by the dive team in recent years. First, yellow floating heart infestation at a lake in Rogue River–Siskiyou National Forest, Oregon, was evaluated, and control measures were applied and monitored for efficacy. The plants were pulled out, and the area covered by a geotech-style (hardware weed-control) cloth; after 3 years, the area has remained clear of the plant. Second, Eurasian water milfoil was detected in Coldwater Lake at the Mount St. Helens National Volcanic Monument, Washington. The plants were pulled out by the dive team and the lake has been monitored for 2 years. Third, in an estuary adjacent to the Siuslaw National Forest, an invasive tunicate was found. The dive team continues to monitor this situation, and, as yet, no control measures have been implemented. Fourth, the dive team partners with the Siuslaw National Forest to survey for freshwater mussels, work that includes documenting occurrences of the Asian clam. Additional surveillance by this regional dive team occurred in summer 2017 within the Plan area on the Deschutes National Forest, Oregon. The dive team also conducts half-day annual training sessions in aquatic-riparian invasive species identification and management for Region 6 personnel, to be applied as stream field crews conduct stream and lake inventories on the national forests.

Management actions implemented when there are known infestations of invasive species on federal lands in the Plan area differ depending on the species considered. In the best scenario, invasive species are identified when their population is small enough for control to be effective. Early Detection and Rapid Response is considered the cornerstone of effective invasive species management (USDA 2017). In situations in which identification of invasive species occurs before the species becomes overly abundant on the landscape, control techniques may be quite effective, as in the above examples implemented by the regional dive team. Additionally, across this region and elsewhere, biosecurity protocols are in effect for personal disinfection of aquatic field gear to prevent spread of emerging diseases and invasive organisms (e.g., Gray et al. 2017, NWCG 2017). In riparian and upland settings, when invasive species have long been present in the environment and are ubiquitous in landscapes surrounding Forest Service land, plans for complete eradication are less feasible. Instead, measures are taken annually to control spread of invasive species, or with the understanding that additional treatments to combat recolonization will be necessary. This is particularly true of several of the invasive riparian and wetland plant species, including reed canary grass, Himalayan blackberry, and purple loosestrife. Mechanical means of control using masticating machines are combined

[5] Hansen, B. 2017. Personal communication. Ecologist, Pacific Northwest Research Station, Corvallis, OR 97331, bhansen@fs.fed.us

with pesticide applications for the most comprehensive and long-term control of these species.

In addition to invasive species control measures, management actions implemented by Region 6 include development of more effective monitoring frameworks, along with preventative measures to forestall ARIS invasion. To prevent the spread of invasive species, educational programs have been initiated, and biosecurity programs have been implemented. For example, changes have been made to fire equipment contracts that now require equipment such as water tanks to arrive at a fire clean and drained (see ARIS website at https://www.fs.fed.us/r6/fire/aquatic-invasive-species/). A two-pronged ARIS monitoring program has recently been developed by the Region 6 and the PNW Research Station. First, to leverage existing freshwater monitoring programs for ARIS detection, a review was completed of ongoing aquatic habitat monitoring, with particular focus on those programs that assess ARIS risk factors. Results of the evaluation found that wadeable stream sections were adequately represented in monitoring on lands administered by Region 6. However, non-wadeable river sections, lakes, and reservoirs that may be at highest risk for invasion by aquatic species were not well represented in existing survey programs (Flitcroft et al. 2016b). The second element was developed as a followup to this finding—a Pilot Monitoring Project of "big water," began in the summer of 2017. The pilot project will implement multispecies environmental DNA (eDNA) methods to facilitate a consistent and rigorous sampling program. Environmental DNA refers to the residual DNA found in water that is shed by species present in (aquatic species) or near (e.g., tree fungi) the water. Water samples are filtered and then processed in the laboratory, allowing for the identification of DNA from species present in the water sample. Traditionally, eDNA has been used to identify one target species at a time. Techniques being developed by the PNW Research Station in collaboration with Region 6 will allow for up to 48 species to be identified per sample. This approach could provide a breakthrough for ARIS monitoring.

Last, in an attempt to leverage existing resources for ARIS monitoring and mitigation, Region 6 has been able to involve forest law enforcement officers (LEOs) in AIS monitoring. LEOs interact regularly with users of Forest Service lands, making them ideal partners in ARIS monitoring. For LEOs to have authority to inspect vehicles, trailers, and boats, Region 6 completed a National Environmental Policy Act review in 2016 to activate two relevant federal regulations that prohibit the transfer of animal and plant invasive species across National Forest System lands. LEOs in the region are trained on invasive species inspection and identification. This effort parallels invasive species law enforcement at the state level, in which state fish and wildlife agencies or invasive species councils work with state police as the law enforcement entities to help remove illegal alien and invasive species. For example, in Washington state, the 2015 Report to the Legislature (http://wdfw.wa.gov/publications/01697/) reported results from 2011 to 2013, including (1) more than 27,000 boat inspections, with decontamination of 83 boats with aquatic invasive species, of which 19 boats had zebra or quagga mussels, and (2) six new infestations of New Zealand mud snails. The Oregon state "report card" for 2013 (http://www.oregoninvasivespeciescouncil.org/oregons-report-card) similarly reports the results of required boat inspections, including a 73 percent compliance rate, mandatory decontamination of ~4 percent (n = 289) of boats because of the presence of invasive plants or animals, including 17 boats decontaminated for quagga or zebra mussels. It is easy to envision how truncating the transmission pathway along our nation's roadways can be an effective ARIS mitigation.

Future Considerations

Our focus has been on describing those invasive species that have become leading priorities for state or Forest Service management actions in the region of the NWFP, and summarizing recent research and management advances. There is a much longer list of nonnative species detected in western Oregon and Washington that are raising local to widespread concerns. Considerable attention is being paid to novel species in the region, with efforts to prevent introductions of nonnative species into new areas and to understand their potential effects. There are also species that are naturalized to the extent that they are considered long-term, somewhat intractable problems, and that do not appear on state or other types of species lists; this situation explains why some priority species are identified in some Northwest states, but not others.

Other Nonnative Species on the Radar in the Northwest

It is important to note that species priorities change with time as new knowledge or events trigger new concerns. For example, the 2011 earthquake and tsunami in Japan resulted in an alert for alien aquatic species crossing the Pacific Ocean and reaching the eastern Pacific shores of Oregon and Washington. Heightened attention to potential introductions of aquatic nuisance species has occurred, as over 300 nonnative species have been found on debris from Japan that reached North America through the summer of 2016,[6] with more than 100 Japanese marine species drifting across the Pacific on a single dock from Misawa, Japan, to Agate Beach, Oregon (Lam et al. 2015). The potential importance of estuaries for such invasions also supports our rationale for their inclusion here.

Barred owls (*Strix varia*) have naturally dispersed into Oregon and Washington from the south and are being recorded as having ripple effects through western forested ecosystems. In addition to interactions affecting the native northern spotted owl (*S. occidentalis caurina*), barred owls have been found to have a broader prey base, including aquatic prey such as crayfish, amphibians, and fish (Wiens et al. 2014). The effects of barred owls on aquatic ecosystems await further research.

Many nonnative fish species have been released in the Western United States for sports fisheries (Schade and Bonar 2005). Effects on native species have been implicated most frequently for amphibians in the lower 48 states (e.g., Knapp and Matthews 2000). Among nonnative fishes released in Oregon and Washington lakes and rivers are smallmouth bass (*Micropterus dolomieu*) and brook trout (*Salvelinus fontinalis*). Both are predators, and effects on native aquatic prey are a concern. For example, bass are implicated as having adverse effects on state-sensitive species such as the foothill yellow-legged frogs (*Rana boylii*) in Oregon (Paoletti et al. 2011). Nonnative stocked fishes are an example of a conflict between ecosystem services: recreation versus native species. In the NWFP area, there have been increasing efforts to eradicate nonnative fish from wild areas, such as Crater Lake National Park, Oregon. To date, stocking continues at some historically stocked sites in Oregon and Washington, and native vertebrates in these systems appear to be persisting—yet may have declined from historical numbers or distribution. This is an evolving issue, and monitoring may be needed for sensitive native species under additional stressors such as disease.

Numerous additional species could be mentioned here. Newly identified nonnative species can gain quick attention with the hope of rapid eradication, forestalling a new invasive species gaining a foothold in the region. For example, chemical treatment for African clawed frogs (*Xenopus laevis*) was conducted in 2016 at a pond in Lacy, Washington, (WDFW 2016). The clawed frogs were eradicated, and surveillance is ongoing to continue efforts as needed. The role of human releases of nonnative species into the wild focuses attention on how these animals enter the region, to state laws and their enforcement for alien species generally, and to the pet trade more specifically.

Climate Change Projections

Climate factors (temperature and precipitation regimes) strongly affect seasonal conditions in upslope, riparian, and freshwater environments. As with native species, invasive species survival is also tied to these same parameters, and projected climate change can likewise affect them. Hence, the distributions of both native and invasive species are likely to synchronously respond to changing conditions. Effects on some species have not yet been modeled, but for many species, the effects of climate change on invasions can be assessed. For example, American bullfrogs require water temperatures greater than 20 to 21 °C (68 to 70 °F) for breeding (Hayes and Jennings 2005), and suitable breeding sites are projected to be found at higher latitudes and altitudes in the future. Similarly, a northward expansion of the relatively cold-water-adapted amphibian chytrid fungus *Bd* has been projected with a variety of climate

[6] Chan, S. 2017. Personal communication. Extension Watersheds and Aquatic Invasive Species. Oregon Sea Grant, Oregon State University, samuel.chan@oregonstate.edu.

futures (Xie et al. 2016). Temperature and precipitation often constrain the range of many invasive plants and limit their successful establishment. With climate change, new habitat may become available, enabling plants to survive outside their historical ranges and expand beyond their current range. For plants, disturbances such as wildfire or logging can provide a "fast-track" for changes in plant communities or even type conversion. For aquatic-dependent species, coldwater refugia are now being considered as localized areas that may be used to more practically protect native species and assemblages from the projected increase in biotic homogenization that is occurring from climate change effects and warm-water species invasions. Finally, climate-related drought and flooding events are also associated with invasive species dispersal; these effects merit additional consideration for the Northwest and elsewhere.

Research and Monitoring Priorities

Despite their increasing recognition as a potentially dominant force in restructuring ecosystems, relatively few research studies have been conducted on Northwest ARIS; their effects on the composition, function, or processes of ecosytems; ecosystem services valued by people; and mitigation efficacies. As evidenced by the above selected studies, support is growing for the importance of invasive species in altering native ecosystems. For example, Sanderson et al. (2009) considered invasive species more important for native salmonids than four other leading concerns combined, including habitat alteration and overexploitation from fishing pressures (harvest).

Several areas stand out as potentially meriting additional research attention. First, aquatic-riparian pathogens and parasites appear underrepresented on Forest Service regional lists of invasive concern species. None are included in AREMP annual surveys, and only tree fungi are included in the Region 6 watch list. Recognition of pathogens and parasites as taxa for regional monitoring could enable their early detection and help forestall invasions. New eDNA techniques could aid in this regard, as detection of cryptic invaders such as pathogens and parasites may otherwise require significant time commitment or costly laboratory analyses applied after disease events are large enough to be easily detectable. Second, only one study summarized above (Claeson and Bisson 2013) addressed effectiveness of invasive species mitigation approaches with a scientific study design. This is a topic that deserves research for all categories of invasive species in table 7-10, and likely a species-by-species comparison of approaches is needed. For example, field intervention strategies for novel species such as *Bsal* have never been attempted, and foreknowledge of fungicidal or other approaches could be vital to control spread. Nevertheless, trial of some invasive species control methods is ongoing via case-by-case management actions with monitoring, like those being conducted by the regional dive team. Although this adds significantly to our knowledge, it is critical to apply the rigor of hypothesis testing with a scientific design. Lastly, the notion of managing for wild and nonwild ecosystems has been broached, but several questions arise about how this might be developed into an effective long-term strategy. For example, relative to federal lands of the NWFP area, if reserved land use allocations are desired to be wild, can that goal be effectively achieved relative to aquatic-riparian ecosystems given that streams often are contiguous across wild and nonwild areas, potentially promoting invasive species dispersal? Furthermore, can forest restoration practices aid in forestalling nonnative species introductions, or altering the existing heterogeneity within the aquatic ecosystem to avoid crossing "tipping points" to establishment of a novel ecosystem domain (Penaluna et al. 2016)?

Appendix 2: Influence of Climate Change on Life Stages of Pacific Salmon

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Adults

The species of anadromous Pacific salmonids found in the Northwest Forest Plan (NWFP) area and their freshwater and marine residence times are shown in table 7-8. The freshwater environment is used for both growth and reproduction; the marine environment is used for growth and the initiation of sexual maturity. Depending on species, fish may spend from 1 to 5 or more years in the eastern Pacific Ocean before returning to fresh water to spawn. An exception is coastal cutthroat trout (*Oncorhynchus clarkii*), which generally make limited forays into nearshore areas and typically do not range more than 65 mi (100 km) from natal rivers (Trotter 1989). Owing to marine heterogeneity, the disparate migration patterns of various stocks, and the widely varying amount of time spent at sea, the influences of climate change on survival and growth of different populations of salmon in the ocean will differ.

Although the specific effects of climate change on marine survival and growth of salmon will depend on the location of their natal rivers and their movements at sea, some trends seem to be common to populations along the Pacific Coast. Possibly as a result of decreasing pH and increasing temperature, salmon are becoming smaller and sometimes younger upon return to fresh water, and exhibit reduced marine survival rates. The size of returning adults of most Pacific salmon species has generally trended downward over the past three decades of the 20th century (Bigler et al. 1996), although there have been multiyear periods when both sizes and abundances have increased (Helle et al. 2007). Some populations of sockeye salmon in Bristol Bay, Alaska, have returned to spawn at a younger age in the second half of the 20th century (Hodgeson et al. 2006, Robards and Quinn 2002).

In the past 69 years, the size of the largest fish caught in a Juneau, Alaska, fishing derby for Chinook salmon (*O. tshawytscha*) has declined (fig. 7-21). Although the origin of these salmon is not known with certainty, it is possible that some originated from rivers in the NWFP area, as migratory routes for some Pacific Northwest Chinook salmon

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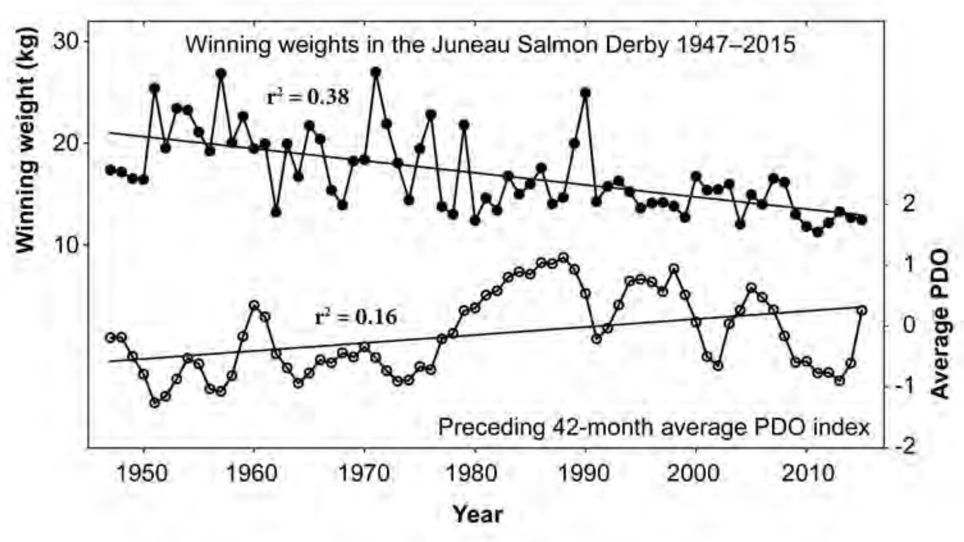

Figure 7-21—Winning weights of Juneau's Golden North Salmon Derby from 1947 through 2015 and the preceding 42-month average Pacific Decadal Oscillation (PDO) index. Positive deviations in the PDO index occur in warmer-than-average PDO cycles, and negative deviations occur in cooler cycles. See also Fagen (1988) and Reid et al. (2016).

stocks include southeast Alaska. There are many possible explanations for the observed declines in fish size, but several previously published examinations of adult salmon sizes from either commercial fishing records (Helle et al. 2007) or Alaskan fishing derbies (Fagen 1988) attributed at least some of the decline to increased competition, owing to large numbers of hatchery-produced salmon (Bigler et al. 1996, Francis and Hare 1997). However, such relationships are not simple. Helle et al. (2007) analyzed data for different species and stocks from northern Alaska to Oregon and concluded that adult body size resulted from both density-dependent factors (competition) and density-independent factors (environmental conditions). Long-term trends in body size observed over time in a Juneau, Alaska, fishing derby are weakly correlated with the gradually warming 42-month average Pacific Decadal Oscillation (PDO) index observed prior to when the fish were caught (fig. 7-21), suggesting a linkage between ocean conditions and fish size that portends future size declines under a warming climate. The relationship between gradual warming and shifts in the frequency and intensity of PDO fluctuations is unclear, but significant PDO regime shifts can signal major changes in the Earth's biophysical systems (Reid et al. 2016).

Decreases in adult body size resulting from changing environmental conditions in the ocean could also lead to reduced reproductive success. In Pacific salmon, both the number of eggs (Hankin and McKelvey 1985, Healey and Heard 1984) and egg size (Quinn and Vøllestad 2003) are directly related to the weight of adult females. Reproductive capacity of populations could decline if females have fewer eggs (McElhany et al. 2000). Egg size, primarily related to yolk reserve, can also be an adaptation to the environment in which eggs develop. Fish that spawn in warmer areas tend to have larger eggs compared to those from cooler areas because the efficiency of yolk conversion to body tissue is reduced at higher temperatures (Fleming and Gross 1990). The survival and body mass at hatching of eggs incubating at warmer future temperatures could therefore be compromised if egg size does not increase as well.

Food webs in aquatic and riparian ecosystems are supported by the influx of marine-derived nutrients from returning adult salmonids (Bilby et al. 1996, Schindler et al. 2003). The productivity of many streams and rivers within the range of Pacific salmon is influenced by the quantity of marine-derived nutrients from salmon carcasses (Gende et al. 2004, Helfield and Naiman 2001, Willson et al. 2004). A reduction in the size and number of returning adult salmon could compromise the capacity of freshwater ecosystems to produce new salmon, with carryover effects on the wide variety of aquatic and terrestrial organisms that may also benefit from the consumption of eggs during the spawning period (Cederholm et al. 2001, Garner et al. 2009). The growth of juvenile salmon during the spawning season is important for their overwinter survival (Lang et al. 2006). Energy derived from eggs consumed by returning adults can also allow for longer migrations and extended spawning times (Copeland and Venditti 2009); thus fewer, smaller eggs could diminish this potential energy source.

According to climate change predictions for most rivers in the NWFP area, returning adult salmon will face warmer temperatures and lower flows if migrations take place in summer. Some species and life-history types, such as stream-type ("spring") Chinook salmon and summer steelhead in the southern and middle portions of the Pacific Coast range of Pacific salmon, return to fresh water in spring or early-summer months, and hold in rivers and streams for several months before spawning. Adults feed infrequently and usually rest in large pools with cool water. Such pools are not abundant in late summer and early autumn, with coolwater refuges likely to become even less available at those times as climate continues to warm. This circumstance suggests that holding and migrating adults may become increasingly stressed, which will diminish their reproductive potential and increase prespawning mortality. Beechie et al. (2006) believed that the loss of summer prespawn staging habitats in rivers entering Puget Sound, Washington, could result in the replacement of stream-type Chinook salmon by ocean-type Chinook salmon, whose autumn run timing avoids exposure to warm, low-flow summer conditions. For populations undertaking long upstream migrations to spawning grounds, elevated stream temperatures will incur higher metabolic costs and mortality (Rand et al. 2006), and fish that do arrive at spawning grounds may have reduced reproductive capacities (Miller et al. 2011).

Warmer temperatures may also limit gonadal development; Pankhurst et al. (1996) found that female steelhead did not ovulate when temperatures exceeded 70 °F (21 °C). The extirpation of Atlantic salmon in the southern portion of their distributional range is attributed to reproductive failure associated with elevated water temperatures in freshwater spawning areas (McCarthy and Houlihan 1997).

Elevated water temperatures during migration can have indirect effects on returning adults. Returning adults may be more vulnerable to disease and parasites if conditions are warmer in fresh water (Johnson et al. 1996, Ray et al. 2012). However, Stocking et al. (2006) found no relation between water temperature and infection of salmonids with *Ceratomyxa shasta* in the Klamath River, California. Juveniles (Chiaramonte et al. 2016) that are unable to find coolwater holding areas during migration in warmer water may be particularly vulnerable to disease because warm water will favor rapid disease transmission and virulence of warm-adapted pathogens that could lead to fish kills. For example, Miller et al. (2011) presented evidence that elevated temperatures in British Columbia's Fraser River have likely contributed to the virulence of a virus that infects adult sockeye salmon (*O. nerka*) prior to entering the Fraser River, resulting in a high incidence of prespawning mortality.

Rising sea level (IPCC 2007) may affect the reproductive success of species that spawn close to tidewater, particularly some pink (*O. gorbuscha*) and chum (*O. keta*) salmon populations. For small populations that spawn in streams just above the high-tide level, elevated sea levels could reduce the available spawning habitat if suitable spawning sites upstream are inaccessible.

The development and persistence of less favorable ocean conditions could potentially influence the degree of anadromy in populations that possess both anadromous and nonanadromous (fully freshwater-resident) life cycle options. Steelhead, the anadromous form of *O. mykiss*, persist at least in part because there is a fitness advantage associated with migrating to the ocean to feed and returning to fresh water to spawn (Quinn and Myers 2004). If this advantage is reduced or lost, residency could increase in populations, assuming that changes in the freshwater environment are suitable for the persistence of the freshwater life-history variant of rainbow trout (Benjamin et al. 2013, Rosenberger et al. 2015, Sloat and Reeves 2014). Other Pacific Coast populations of *O. mykiss* maintain primarily resident populations in locations where the marine environment is believed to be unfavorable for survival and growth, as in southern California (Behnke 2002).

Eggs and Alevins

Eggs and developing embryos will likely be affected by two different aspects of climate change—increased temperatures during egg incubation and altered hydrographs. Under some climate scenarios, winter temperatures are predicted to increase at faster rates than are summer temperatures for Alaska (IPCC 2007), whereas the opposite is true for the more southerly NWFP region (Mote and Salathé 2010).

Most research on climate effects on native fish has focused on the potential for elevated summer temperatures (e.g., Crozier and Zabel 2006, Isaak et al. 2010). However, the effect of elevated winter temperatures may be as, and perhaps even more, pronounced and ecologically significant than increases in summer temperatures. Increased winter temperatures in the NWFP area will result in more precipitation falling as rain rather than snow. Watersheds that historically developed a seasonal snowpack will experience a trend from snow to rain, resulting in more rapid runoff in winter and early spring when snow usually falls, and lower late-spring and early-summer flows owing to reduced snowmelt (Hamlet and Lettenmaier 2007, Hamlet et al. 2005, Tague and Grant 2009). In Washington state's transitional drainage systems that historically possessed both autumn/winter and spring/summer runoff peaks, the shift to a rain-dominant hydrograph is expected to be the most dramatic. Substantial increases are anticipated in the magnitude and frequency of extremely high-flow events in winter, coupled with substantial reductions in summer low flows (e.g., Elsner et al. 2010, Mantua et al. 2010). However, because snowpack will be reduced, rivers with snowmelt-dominated hydrographs could likely see a reduction in the magnitude of high flows during spring runoff. Loukas and Quick (1999) predicted that floods in the snowmelt-dominated continental portions of British Columbia will decrease in magnitude by 7 percent and in volume by 38 percent, and occur as many as 20 days earlier, as a result of the snow-to-rain transition. In coastal areas, Loukas and Quick (1999) projected that there would be little change in the timing of floods, but that, on average, peak-flow magnitude (+14 percent), flood volume (+94 percent), frequency (+11 percent), and duration (+44 percent) would all increase.

High-flow events will influence egg and alevin survival, depending on the depth of the redd, the size of the female, and the location of spawning in the stream network. Eggs in shallower redds will be more susceptible to being scoured than will those in deeper redds, and smaller salmon often excavate shallower redds than larger salmon (van den Berghe and Gross 1989). It has been speculated that increased peak flows during the incubation period could result in decreased survival of eggs and embryos in populations exposed to hydrologic regimes that have become more prone to gravel-mobilizing flows (Battin et al. 2007)

Potential effects of hydrographs altered by climate change are likely to differ among species and life-history forms. In most drainages of the NWFP area, scour is likely to increase the most in small streams or in confined, steep rivers, affecting fish such as bull trout (*Salvelinus confluentus*) that spawn in the late autumn and early winter when the most severe storms tend to occur along the northwestern Pacific Coast (Isaak et al. 2012). Fish spawning in lower gradient, unconfined areas, such as coho (*O. kisutch*), Chinook, pink, and chum salmon, could be less affected. Studies that have examined potential effects of increased flows on streambed scour (Battin et al. 2007, Leppi et al. 2014, Shanley and Albert 2014) assumed a uniform relationship between flood magnitudes and the vulnerability of salmon populations and their habitat. However, the geographic range of Pacific salmon is characterized by exceptional topographic complexity and watershed dynamism (Montgomery 1999), which can generate considerable diversity in watershed- and stream reach-scale responses of habitat to flood disturbance (Buffington 2012, Montgomery and MacDonald 2002). Thus, effects of increased flows are unlikely to be similar among watersheds or even among reaches within stream networks.

Previous research has demonstrated that stream-channel response potential varies according to position within the dendritic structure of stream networks (Benda et al. 2004), variation in valley and reach-scale confinement (Coulthard et al. 2000, Montgomery and Buffington 1997), and differences among species in their use of habitats created by this physiographical complexity (Goode et al. 2013). In terms of management, floodplain connectivity may ameliorate the effects of future increases in discharge on streambed dynamics. Floodplain connectivity in unconfined reaches provides a "stress release valve" (McKean and Tonina 2013) that limits vulnerability of salmon spawning habitat even in large floods with return intervals of decades to centuries (Goode et al. 2013, Lapointe et al. 2000, McKean and Tonina 2013). In this regard, maintaining or restoring connectivity between streams and adjacent floodplains will mitigate near-term responses to increased flood magnitudes. Additionally, maintaining or restoring channel complexity and hydraulic roughness from large wood may further mitigate the effect of higher flows on salmon spawning habitat (Montgomery et al. 1996, Sloat et al. 2017).

The rate of development of eggs and the size of fish at emergence is related to water temperature. Egg development depends on the accumulation of degree days (Neuheimer and Taggart 2007). Even slight increases in temperature can accelerate rate of development and ultimately result in earlier time of emergence from the gravel (McCullough 1999) (fig. 7-22). Accelerated development leads to smaller

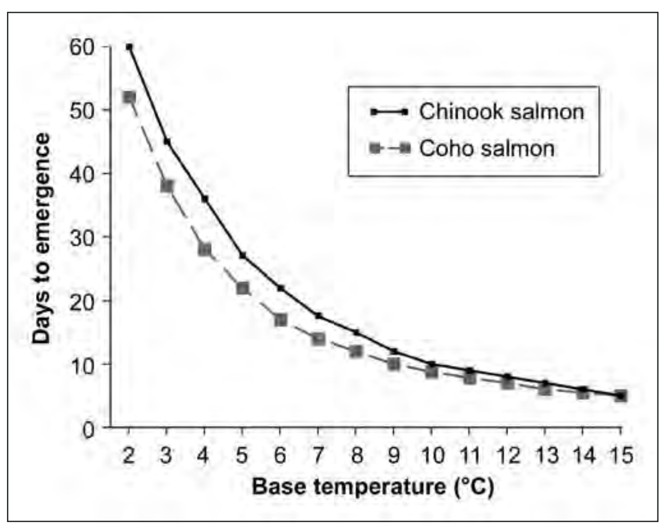

Figure 7-22—Changes in time of emergence of Chinook and coho salmon as a result of a 1 °C-increment increase in water temperature during egg development. From McCullough 1999.

individuals at emergence because metabolic costs decrease the efficiency of yolk use (Beacham and Murray 1990, Elliott and Hurley 1998). Upon emergence, smaller fish are more susceptible to displacement at higher flows. Some fish species may be more influenced by thermal shifts during incubation than others; Beacham and Murray (1990) suggested that coho salmon are adapted for cool water temperatures during development and could experience poorer survival under warming climate scenarios.

There are important ecological implications of climate-related changes in the time and size of fish at emergence. Earlier emergence can result in an extended growing season, a benefit that can lead to increased fitness. Holtby (1988) found that an increase of 1.3 °F (0.7 °C) in winter water temperatures following timber harvest in Carnation Creek on the west coast of Vancouver Island, British Columbia, resulted in coho salmon emerging 6 weeks earlier. Size at age increased because of the extended growing season, resulting in more fish completing their freshwater-rearing life history in one year rather than two. Coho salmon in Carnation Creek also smolted and moved to sea about 2 weeks earlier following timber harvest (which raised stream temperatures); however, marine survival declined, possibly as a result of the decoupling of the timing of smolt migration from marine plankton blooms (Holtby and Scrivener 1989). Similarly, warmer winter temperatures increased the length of the growing season of recently emerged sockeye salmon in southwest Alaska. Like coho salmon in Carnation Creek, sockeye salmon grew faster, and more underwent smolt transformation at age 1+ during warm periods rather than at age 2+ in cooler periods (Schindler et al. 2005). However, age-1+ smolts were smaller than age-2+ smolts and were expected to have decreased marine survival.

Juveniles

Juvenile Pacific salmon (defined here as recently emerged fry up to, but not including, smolts) face a number of challenges from the potential effects of climate change. These challenges will include elevated temperatures and altered streamflows, both of which can affect physical and biological aspects of stream habitats. The type and extent of flow effects will differ depending on the time of emergence. For example, fish emerging in the late winter and early spring may experience high flows caused by earlier snowmelt. The consequences of a changing hydrograph will depend to a large degree on the geomorphic setting in which spawning and emergence occurs. In some settings, increased flooding could improve use of floodplain habitats when fish in wide, geomorphically unconstrained channels have access to habitats where floodplain vegetation is intact and secondary channels are available.

Low-gradient streams and rivers can be important areas for postemergent and seasonal growth (Brown and Hartman 1988, Moore and Gregory 1988, Peterson 1982a), and marginal areas with reduced water velocities provide refuge against downstream displacement. Fry that emerge at a smaller size if water temperature is warmer can potentially overcome their size disadvantage by gaining an early start on the growing season (Holtby 1988). Juvenile salmonids in rain-dominated hydrographic regimes often move into the lower reaches of the channel network or into off-channel habitats in autumn to seek refuge from unfavorable water velocities in the main channel (Ebersole et al. 2006, Everest 1975, Peterson 1982b, Solazzi et al. 2000). In high-elevation snowfall-dominated drainage systems, however, climate warming might not significantly increase mid-winter flood flows and facilitate access to floodplain habitats if precipitation still falls as snow.

Under several climate scenarios, the onset of the low-flow period is expected to occur up to 4 to 6 weeks earlier in most areas as a result of warming (Hamlet and Lettenmaier 2007, Hamlet et al. 2005, Tague and Grant 2009). An extended period of low discharge over the dry season would likely decrease the amount of habitat suitable to juvenile salmonids, and this effect could be most pronounced in small to mid-sized streams (Stewart et al. 2005), resulting in some reaches that formerly held surface flows throughout the year becoming intermittent or even drying completely. As noted by Battin et al. (2007), flow reductions in headwater areas during the dry season could force resident fishes downstream in the stream network, as well as compromise their ability to cope with drought, by reducing the network of connected, perennially flowing channels. Additionally, the downstream displacement of headwater-rearing fish will expose them to warmer temperatures than those to which they are adapted, and possibly to harmful biological interactions with native and nonnative species inhabiting the lower watershed.

The consequences of climate-induced changes in low flows for juvenile salmonids such as Chinook salmon and steelhead that often rear in rivers are likely similar to those in smaller streams, although the risk of river reaches becoming intermittent is less because drainage areas are larger. Mantua et al. (2010) found widespread declines in summer discharge for many rivers in Washington state under climatic warming scenarios. Likewise, Luce and Holden (2009) examined hydrographic records from drainage systems throughout the Pacific Northwest and found that summer flows in all types of hydrologic regimes have been declining, thus providing increasingly smaller rearing areas to river-dwelling species.

In addition to lower flows, elevated summer water temperatures will likely have strong ecological effects on juvenile Pacific salmon, with the direction and magnitude of influence varying geographically, by species, and by life-history type. Water temperature influences the metabolism, food consumption, and growth of an individual (Brett et al. 1969, Warren and Davis 1967, Wurtsbaugh and Davis 1977). Age and size of individuals also influence thermal effects; younger and smaller fish are most susceptible to thermal extremes (Brett 1952) and to short-term thermal variation (Elliott 1994). There is a temperature range in which an individual performs best given a certain level of food resources, and beyond that range, metabolic costs increase such that growth declines (Warren 1971). Increased temperature could potentially affect juvenile salmonids in opposing ways (Li et al. 1994). Warmer water could enhance primary and secondary aquatic production, leading to greater food availability; however, if the increased metabolic demands of warmer temperatures reduce food-conversion efficiency or if the organisms benefiting from warmer temperatures are not preferred food items, the net effect of warming could be reduced growth (Bisson and Davis 1976). In southern portions of a species' range, elevated temperatures could reduce the suitability of rearing areas for juveniles during the summer as temperatures exceed the point at which gains resulting from increased aquatic production are offset by physiological costs, resulting in reduced summer growth rates (Marine and Cech 2004). In contrast, growth rates of juveniles in more northern areas could increase if projected temperature changes stimulate aquatic productivity while remaining within the preferred physiological range for the species.

If the net effects of elevated temperatures resulting from climate change in southern areas reduce summer growth (Isaak et al. 2010, Royer and Minshall 1997, Scarnecchia and Bergersen 1987), juveniles will be smaller entering the winter (ISAB 2007), and overwinter survival may decrease (Quinn and Petersen 1996). However, thermal increases may be beneficial for growth during other seasons if abundant food is present. Sogard et al. (2010) found that juvenile steelhead on the central coast of California attained the most growth in the spring and autumn, and that juvenile coho salmon grew in the winter in coastal Oregon (Ebersole et al. 2006, 2009).

Outcomes of interactions between salmonids and nonsalmonids can be influenced by changing water temperatures. Rearing salmonids tend to outcompete nonsalmonids for food resources and preferred feeding areas at cooler temperatures, whereas nonsalmonids have the advantage at warmer temperatures (Petersen and Kitchell 2001, Reeves et al. 1987). The susceptibility of juvenile salmonids to disease could also increase at warmer temperatures and could be compounded by the presence of competitors that are less susceptible to the pathogens infecting salmon and trout (Reeves et al. 1987). Additionally, warmer temperatures could lead to increased predation from nonnative warmwater fish (ISAB 2007, Petersen and Kitchell 2001). The aggregate results of these indirect effects are likely to be changes in the structure and composition of fish communities in the affected stream systems (ISAB 2012), particularly in the southern portions of the NWFP area where the potential for interaction with warmwater species is greatest owing to widespread introduction and proliferation of nonnative warmwater fishes.

The effects of climate change on rearing habitats for juvenile salmon at the local level will depend, to some degree, on the geomorphic features of a particular location.

Crozier and Zabel (2006) suggested that two climate-influenced factors—stream temperature and flow—could affect habitat in different ways: narrow, confined streams were predicted to be more responsive to flow changes, and geomorphically unconfined streams would be more sensitive to temperature changes. In addition, the future quantity and quality of freshwater rearing habitat of Pacific salmon may also be influenced by predicted increases in the magnitude and frequency of large disturbances. Climate change scenarios predict an increase in exceptional flood events caused by transitions from snow to rain, accelerated glacial melt, wildfires, and forest pathogen outbreaks (Dale et al. 2001, Hamlet and Lettenmaier 2007). Frequent large floods promote landsliding and stream sedimentation in many areas (Miller et al. 2003). The effects of floods and associated erosion events on freshwater habitat will differ depending on the geomorphic setting, the magnitude and legacy of the event, the interval between succeeding disturbances, and the extent to which the affected ecosystem has been altered by past human activities (Reeves et al. 1995, Rieman et al. 2006).

Increased disturbance frequency and severity can have short-term negative consequences for fish populations, including substrate scour and fine-sediment intrusion that reduces egg and alevin survival and macroinvertebrate abundance in confined channels, displacement of juveniles downstream, and loss of surface flow in summer in reaches where porous material has been deposited in the channel. However, in functionally intact systems there is a strong potential for aquatic habitat complexity to improve with flooding because floodplain linkages can be reestablished and large wood will be recruited to the channel network (Bisson et al. 2009). Long-term changes could be favorable to rearing salmon if the cumulative effects of climate change on water temperature, fine-sediment levels, and surface flows remain within limits tolerable to juvenile salmon or exceed those thresholds only for a short duration.

Population productivity after large disturbances will also be enhanced by the presence of adjacent fish populations that provide sources of colonizers to help initiate recovery and that add to the phenotypic and genetic diversity of affected populations (Schtickzelle and Quinn 2007). But it is also possible that in greatly altered watersheds, where the cumulative harmful effects of climate change exceed environmental tolerance limits, the damage caused by large-scale disturbances will be too great, and if there are no nearby populations to provide new colonists, local population extirpation will occur.

Lakes are important rearing habitats for sockeye salmon and will also be affected by climate change, although there are relatively few drainage systems in the NWFP area that support sockeye salmon runs. Potential effects will vary greatly depending on the location and features of the lake, but a primary effect will be the magnitude and seasonality of warming, with epilimnetic water and the timing of spring and autumn turnover experiencing the greatest changes (Stefan et al. 2001). Slight warming of deep lakes could lead to increased sockeye growth rates if temperatures stimulate primary and secondary production without significantly affecting the availability of cooler water during periods when the epilimnion becomes too warm for efficient metabolism. This benefit could be offset during the growing season by a reduction in the delivery of inorganic nutrients and dissolved organic carbon from terrestrial systems as a result of decreased spring and summer flows. Reduced inputs of nutrients and dissolved organic carbon from the surrounding watershed could result in diminished algal production, which would result in deeper light penetration and additional warming of the lake (Schindler et al. 1990).

The productivity of zooplankton, the principal food of juvenile sockeye salmon in lakes, will be affected by climate change, but whether or not the changes are beneficial will depend on ambient thermal and hydrologic regimes. In Alaska, warming temperatures have resulted in earlier ice melt, greater densities of zooplankton, and increasing sockeye growth rates (Schindler et al. 2005). In contrast, earlier onset of spring in western Washington's Lake Washington has advanced lake stratification by 20 days in recent years, resulting in earlier diatom blooms and a decline in cladocerans (*Daphnia* spp.), important prey species for juvenile sockeye rearing in the lake (Winder and Schindler 2004).

Smolts

Anadromous salmonids typically undergo the smolting process and move to the ocean in spring, although seaward migrations of some salmon stocks occur throughout the year. Water temperature, day length, and changes in flow are the principal cues influencing the timing of parr-smolt transformations. Environmental signals affecting smolting can be divided into regulating and controlling factors (Byrne et al. 2004). Regulating factors act on juvenile salmon before the migration and influence the physiological aspects of smolting. Controlling factors operate during migration and affect the speed of downstream movement. Water temperature and day length appear to be key regulating factors (Jonsson and Jonsson 2009). Day length is not influenced by climate change, but increased temperature will affect the onset of smoltification. For Pacific salmon, elevated winter temperatures can result in earlier migration times of smolts. Chinook salmon have been observed to migrate earlier in warmer years than in cooler years (Achord et al. 2007, Roper and Scarnecchia 1999), but Jonsson and Jonsson (2009) cite a suite of other studies on Atlantic salmon, brown trout (*Salmo trutta*), and steelhead in which water temperatures did not affect the timing of smolt migration. Under certain conditions, elevated temperatures may even inhibit parr-smolt transformation. Adams et al. (1973) found that smolting in steelhead held at 59 °F (15 °C) or warmer led to reductions of ATPase activity needed to initiate the smolt transformation process. Thus, the effect of altered temperature on timing of smolt migration remains unpredictable and likely will vary widely across populations.

To a large extent, streamflow determines the rate at which smolts move downstream (Connor et al. 2003, Smith et al. 2002). Climate model projections of stream runoff (Snover et al. 2003, Tague and Grant 2009) suggest that the onset of the low-flow period will occur 4 to 6 weeks earlier over much of the NWFP area in the next century. Projections of the annual cycle of elevated flows from melting snow for more northerly areas are not currently available, but we assume that they will be similar. The consequences of altered flows are likely to be population-specific, with the timing and smolt survival rates of those populations that tend to migrate later or are required to move long distances likely to be the most affected by climate change.

The survival of smolts entering the ocean depends on a number of factors (Pearcy 1992). Larger smolts tend to have higher survival rates than do smaller fish (Holtby and Scrivener 1989, Quinn and Peterson 1996, Slaney 1988), possibly because they are better able to avoid predation. The size of an individual at smolting is influenced by its size at the beginning of the previous winter. Brown and Hartman (1988) found that stream and groundwater warming caused by logging in a coastal Vancouver Island watershed resulted in increased overwinter growth of presmolt coho salmon, and Holtby and Scrivener (1989) suggested that this growth advantage led to higher smolt-to-adult return rates through improved ocean survival.

Conditions in marine nearshore areas at the time of ocean entry are known to strongly influence ocean survival (Rechisky et al. 2009). In the coastal area influenced by the California current—primarily the southern half of the distributional range of many Pacific salmon species—potential changes in the timing and intensity of upwelling have important implications for smolts (Barth et al. 2007). Cold, nutrient-rich waters are pushed into nearshore areas by northerly winds in the late spring and early summer, producing favorable conditions for plankton production (Nickelson 1986, Scheuerell and Williams 2005). Under one climate change scenario, upwelling is projected to intensify but occur later in the summer (Snyder et al. 2003), decoupling the timing of smolt migration relative to plankton blooms for early-entry salmon smolts.

The abundance of predators in nearshore areas can also influence marine survival of smolts (Pearcy 1992). Coho salmon from Carnation Creek on the west coast of Vancouver Island, British Columbia, entered the ocean about 2 weeks earlier as a result of increased growth as juveniles (Holtby 1988), but survival declined compared to the timing of pre-logging smolt migration. It was believed that predation by mackerel (*Scomber japonicus*) and hake (*Merluccius productus*) contributed to the decline, as both species moved into Barkley Sound during periods of warm sea-surface temperatures. Elevated ocean temperatures could also result in the expansion of subtropical predators such as the Humboldt squid (*Dosidicus gigas*) into Pacific Northwest waters, further increasing predation pressure on salmon smolts (Christensen and Trites 2011, ISAB 2007).

Nearshore conditions in northern portions of the NWFP area will also be influenced by climate change. In some locations, melting glaciers could increase iron levels in nearshore areas (Westerlund and Ohman 1991). Iron levels are often considered limiting to primary production in the North Pacific, and increased iron levels in freshwater plumes could potentially enhance marine food webs (Rose et al. 2005) and thus improve growth and survival of young salmon. The projected effects of climate change on the ocean ecology of Pacific salmon will therefore result from the combined influences of several factors, notably predation, food resource abundance, and both intra- and interspecific competition.

Scientific and common names of plant species identified in this report

Scientific name	Common name
Abies amabilis (Douglas ex Loudon) Douglas ex Forbes	Pacific silver fir
Abies concolor (Gord. & Glend.) Lindl. ex Hildebr.	White fir
Abies grandis (Douglas ex D. Don) Lindl.	Grand fir
Abies lasiocarpa (Hook.) Nutt.	Subalpine pine
Abies magnifica A. Murray bis	California red fir
Abies procera Rehder	Noble fir
Acer circinatum Pursh	Vine maple
Acer macrophyllum Pursh	Bigleaf maple
Achlys triphylla (Sm.) DC.	Sweet after death
Adenocaulon bicolor Hook.	American trailplant
Alliaria petiolata (M. Bieb.) Cavara & Grande	Garlic mustard
Alnus rubra Bong.	Red alder
Amelanchier alnifolia (Nutt.) Nutt. ex M. Roem.	Saskatoon serviceberry
Anemone oregana A. Gray	Blue windflower
Apocynum cannabinum L.	Dogbane
Arbutus menziesii Pursh)	Madrone
Arceuthobium M. Bieb.	Dwarf mistletoe
Arceuthobium occidentale Engelm.	Gray pine dwarf mistletoe
Arceuthobium tsugense Rosendahl	Hemlock dwarf mistletoe
Arctostaphylos nevadensis A. Gray	Pinemat manzanita
Brachypodium sylvaticum (Huds.) P. Beauv.	False brome
Brodiaea coronaria (Salisb.) Engl.	Cluster-lilies
Callitropsis nootkatensis (D. Don) Oerst. ex D.P. Little	Alaska yellow-cedar
Calocedrus decurrens (Torr.) Florin	Incense cedar
Cannabis L.	Marijuana
Carex barbarae Dewey and *C. obnupta* L.H. Bailey	Sedges
Centaurea solstitialis L.	Yellow starthistle
Chamaecyparis lawsoniana (A. Murray bis) Parl.	Port Orford cedar
Chimaphila menziesii (R. Br. ex D. Don) Spreng.	Little prince's pine
Chimaphila umbellata (L.) W.P.C. Barton	Pipsissewa
Clematis vitalba L.	Old man's beard
Clintonia uniflora Menzies ex Schult. & Schult. f.) Kunth	Bride's bonnet
Coptis laciniata A. Gray	Oregon goldthread
Corylus cornuta Marshall var. *californica* (A. DC.) Sharp	California hazel
Cornus canadensis L.	Bunchberry dogwood
Cytisus scoparius (L.) Link	Scotch broom
Disporum hookeri (Torr.) G. Nicholson var. *hookeri*	Drops-of-gold
Fallopia japonica (Houtt.) Ronse Decr. var. *japonica*	Japanese knotweed
Gaultheria ovatifolia A. Gray	Western teaberry
Gaultheria shallon Pursh	Salal

Scientific name	Common name
Gentiana douglasiana Bong.	Swamp gentian
Geranium lucidum L.	Shining geranium
Geranium robertianum L.	Robert geranium
Goodyera oblongifolia Raf.	Western rattlesnake plantain
Hedera helix L.	English ivy
Heracleum mantegazzianum Sommier & Levier	Giant hogweed
Hesperocyparis sargentii (Jeps.) Bartel	Sargent's cypress
Hieracium aurantiacum L.	Orange hawkweed
Ilex aquifolium L.	English holly
Iris pseudacorus L.	Paleyellow iris
Juniperus occidentalis Hook.	Western juniper
Lamiastrum galeobdolon (L.) Ehrend. & Polatschek	Yellow archangel
Lilium occidentale Purdy	Western lily
Linnaea borealis L.	Twinflower
Lithocarpus densiflorus (Hook. & Arn.) Rehder	Tanoak
Lonicera hispidula Pursh	Honeysuckle
Lupinus albicaulis Douglas	Sickle-keeled lupine
Lycopodium clavatum L.	Running clubmoss
Lythrum salicaria L.	Purple loosestrife
Mahonia nervosa (Pursh) Nutt.	Cascade barberry
Malus fusca (Raf.) C.K. Schneid.	Pacific crabapple
Notholithocarpus densiflorus (Hook. & Arn.) P.S. Manos, C.H. Cannon, & S.H. Oh	Tanoak
Notholithocarpus densiflorus (Hook. & Arn.) P.S. Manos, C.H. Cannon, & S.H. Oh var. *echinoides* (R.Br. ter) P.S. Manos, C.H. Cannon & S.H. Oh	Shrub form of tanoak
Nuphar polysepala (Engelm.)	Yellow pond lily
Nymphoides peltata (S.G. Gmel.) Kuntze	Yellow floating heart
Osmorhiza chilensis Hook. & Arn.	Sweetcicely
Phalaris arundinacea L.	Reed canarygrass
Picea engelmannii Parry ex Engelm.	Engelmann spruce
Picea sitchensis (Bong.) Carrière	Sitka spruce
Pinus albicaulis Engelm.	Whitebark pine
Pinus attenuata Lemmon	Knobcone pine
Pinus contorta Douglas ex Loudon	Lodgepole pine
Pinus contorta Douglas ex Loudon var. *contorta*	Beach pine, shore pine
Pinus jeffreyi Balf.	Jeffrey pine
Pinus lambertiana Douglas	Sugar pine
Pinus monticola Douglas ex D. Don)	Western white pine
Pinus ponderosa Lawson & C. Lawson	Ponderosa pine
Populus trichocarpa L. ssp. *trichocarpa* (Torr. & A. Gray ex Hook) Brayshaw	Black cottonwood
Potamogeton crispus L.	Curly pondweed
Potentilla recta L.	Sulphur cinquefoil

Scientific name	Common name
Prunus emarginata (Douglas ex Hook. D. Dietr.)	Bitter cherry
Pseudotsuga menziesii (Mirb.) Franco	Douglas-fir
Pteridium aquilinum (L. Kuhn)	Brackenfern
Pueraria montana (Lour.) Merr. var. *lobata* (Willd.) Maesen & S.M. Almeida ex Sanjappa & Predeep	Kudzu
Pyrola asarifolia Sweet	American wintergreen
Quercus agrifolia Née var. *oxyadenia* (Torr.) J.T. Howell	Coastal live oak
Quercus berberidifolia Liebm.	Scrub oak
Quercus chrysolepis Liebm.	Canyon live oak
Quercus douglasii Hook. & Arn.	Blue oak
Quercus garryana Douglas ex hook.	Oregon white oak
Quercus kelloggi Newberry	California black oak
Quercus lobata Née	Valley oak
Rhamnus purshiana (DC.) A. Gray	Cascara
Rhododendron groenlandicum Oeder	Bog Labrador tea
Rhododendron macrophyllum D. Don ex G. Don	Pacific rhododendron
Ribes lacustre (Pers.) Poir.	Prickly currant
Rubus armeniacus Focke	Himalayan blackberry
Salix exigua Nutt.	Sandbar willow
Senecio bolanderi A. Gray	Bolander's ragwort
Sequoia sempervirens (Lamb. ex D. Don) Endl.	Redwood
Smilacina stellata (L.) Desf.	Starry false Solomon's seal
Synthyris reniformis (Douglas ex Benth.) Benth.	Snowqueen
Taxus brevifolia Nutt.	Pacific yew
Thuja plicata Donn ex D. Don	Western redcedar
Tiarella trifoliate L.	Threeleaf foamflower
Trapa natans L.	Water chestnut
Trillium ovatum Pursh	Pacific trillium
Tsuga heterophylla (Raf.) Sarg.	Western hemlock
Tsuga mertensiana (Bong.) Carrière	Mountain hemlock
Typha latifolia L.	Cattails
Umbellularia californica (Hook. & Arn.) Nutt.	California bay laurel
Vaccinium alaskaense Howell	Alaska blueberry
Vaccinium membranaceum Douglas ex Torr.	Thinleaf huckleberry, big huckleberry
Vaccinium ovatum Pursh	Evergreen huckleberry
Vaccinium oxycoccos L.	Small cranberry
Vaccinium parvifolium Sm.	Red huckleberry
Vancouveria hexandra (Hook.) C. Morren & Decne.	White insideout flower
Xerophyllum tenax (Pursh) Nutt.	Beargrass

Glossary

This glossary is provided to help readers understand various terms used in the Northwest Forest Plan (NWFP) science synthesis. Sources include the Forest Service Handbook (FSH), the Code of Federal Regulations (CFR), executive orders, the Federal Register (FR), and various scientific publications (see “Glossary Literature Cited”). The authors have added working definitions of terms used in the synthesis and its source materials, especially when formal definitions may be lacking or when they differ across sources.

active management—Direct interventions to achieve desired outcomes, which may include harvesting and planting of vegetation and the intentional use of fire, among other activities (Carey 2003).

adaptive capacity—The ability of ecosystems and social systems to respond to, cope with, or adapt to disturbances and stressors, including environmental change, to maintain options for future generations (FSH 1909.12.5).

adaptive management—A structured, cyclical process for planning and decisionmaking in the face of uncertainty and changing conditions with feedback from monitoring, which includes using the planning process to actively test assumptions, track relevant conditions over time, and measure management effectiveness (FSH 1909.12.5). Additionally, adaptive management includes iterative decisionmaking, through which results are evaluated and actions are adjusted based on what has been learned.

adaptive management area (AMA)—A portion of the federal land area within the NWFP area that was specifically allocated for scientific monitoring and research to explore new forestry methods and other activities related to meeting the goals and objectives of the Plan. Ten AMAs were established in the NWFP area, covering about 1.5 million ac (600 000 ha), or 6 percent of the planning area (Stankey et al. 2003).

alien species—Any species, including its seeds, eggs, spores, or other biological material capable of propagating that species, that is not native to a particular ecosystem (Executive Order 13112). The term is synonymous with exotic species, nonindigenous, and nonnative species (see also “invasive species”).

allochthonous inputs—Material, specifically food resources, that originates from outside a stream, typically in the form of leaf litter.

amenity communities—Communities located near lands with high amenity values.

amenity migration—Movement of people based on the draw of natural or cultural amenities (Gosnell and Abrams 2011).

amenity value—A noncommodity or “unpriced” value of a place or environment, typically encompassing aesthetic, social, cultural, and recreational values.

ancestral lands (of American Indian tribes)—Lands that historically were inhabited by the ancestors of American Indian tribes.

annual species review—A procedure established under the NWFP in which panels of managers and biologists evaluate new scientific and monitoring information on species to potentially support the recommendation of changes in their conservation status.

Anthropocene—The current period (or geological epoch) in which humans have become a dominant influence on the Earth’s climate and environment, generally dating from the period of rapid growth in industrialization, population, and global trade and transportation in the early 1800s (Steffen et al. 2007).

Aquatic Conservation Strategy (ACS) —A regional strategy applied to aquatic and riparian ecosystems across the area covered by the NWFP) (Espy and Babbit 1994) (see chapter 7 for more details).

at-risk species—Federally recognized threatened, endangered, proposed, and candidate species and species of conservation concern. These species are considered at risk of low viability as a result of changing environmental conditions or human-caused stressors.

best management practices (BMPs) (for water quality)—Methods, measures, or practices used to reduce or eliminate the introduction of pollutants and other detrimental impacts to water quality, including but not limited to structural and nonstructural controls and to operation and maintenance procedures.

biodiversity—In general, the variety of life forms and their processes and ecological functions, at all levels of biological organization from genes to populations, species, assemblages, communities, and ecosystems.

breeding inhibition—Prevention of reproduction in healthy adult individuals.

bryophytes—Mosses and liverworts.

canopy cover—The downward vertical projection from the outside profile of the canopy (crown) of a plant measured in percentage of land area covered.

carrying capacity—The maximum population size a specific environment can sustain.

ceded areas—Lands that particular tribes ceded to the United States government by treaties, which have been catalogued in the Library of Congress.

climate adaptation—Management actions to reduce vulnerabilities to climate change and related disturbances.

climate change—Changes in average weather conditions (including temperature, precipitation, and risk of certain types of severe weather events) that persist over multiple decades or longer, and that result from both natural factors and human activities such as increased emissions of greenhouse gases (U.S. Global Change Research Program 2017).

coarse filter—A conservation approach that focuses on conserving ecosystems, in contrast to a "fine filter" approach that focuses on conserving specific species. These two approaches are generally viewed as complementary, with fine-filtered strategies tailored to fit particular species that "fall through the pores" of the coarse filter (Hunter 2005). See also "mesofilter."

co-management—Two or more entities, each having legally established management responsibilities, working collaboratively to achieve mutually agreed upon, compatible objectives to protect, conserve, use, enhance, or restore natural and cultural resources (81 FR 4638).

collaborative management—Two or more entities working together to actively protect, conserve, use, enhance, or restore natural and cultural resources (81 FR 4638).

collaboration or collaborative process—A structured manner in which a collection of people with diverse interests share knowledge, ideas, and resources, while working together in an inclusive and cooperative manner toward a common purpose (FSH 1909.12.05).

community (plant and animal)—A naturally occurring assemblage of plant and animal species living within a defined area or habitat (36 CFR 219.19).

community forest—A general definition is forest land that is managed by local communities to provide local benefits (Teitelbaum et al. 2006). The federal government has specifically defined community forest as "forest land owned in fee simple by an eligible entity [local government, nonprofit organization, or federally recognized tribe] that provides public access and is managed to provide community benefits pursuant to a community forest plan" (36 CFR 230.2).

community of place or place-based community—A group of people who are bound together because of where they reside, work, visit, or otherwise spend a continuous portion of their time.

community resilience—The capacity of a community to return to its initial function and structure when initially altered under disturbance.

community resistance—The capacity of a community to withstand a disturbance without changing its function and structure.

composition—The biological elements within the various levels of biological organization, from genes and species to communities and ecosystems (FSM 2020).

congeneric—Organisms that belong to the same taxonomic genus, usually belonging to different species.

connectivity (of habitats)—Environmental conditions that exist at several spatial and temporal scales that provide landscape linkages that permit (a) the exchange of flow, sediments, and nutrients; (b) genetic interchange of genes among individuals between populations; and (c) the long-distance range shifts of species, such as in response to climate change (36 CFR 219.19).

consultation (tribal)—A formal government-to-government process that enables American Indian tribes and Alaska Native Corporations to provide meaningful, timely input, and, as appropriate, exchange views, information, and recommendations on proposed policies or actions that may affect their rights or interests prior to a decision. Consultation is a unique form of communication characterized by trust and respect (FSM 1509.05).

corticosterone—A steroid hormone produced by many species of animals, often as the result of stress.

cryptogam—An organism that reproduces by spores and that does not produce true flowers and seeds; includes fungi, algae, lichens, mosses, liverworts, and ferns.

cultural keystone species—A species that significantly shapes the cultural identity of a people, as reflected in diet, materials, medicine, or spiritual practice (Garibaldi and Turner 2004).

cultural services—A type of ecosystem service that includes the nonmaterial benefits that people obtain from ecosystems through spiritual enrichment, cognitive development, reflection, recreation, and aesthetic experiences (Sarukhán and Whyte 2005).

desired conditions—A description of specific social, economic, or ecological characteristics toward which management of the land and resources should be directed.

disturbance regime—A description of the characteristic types of disturbance on a given landscape; the frequency, severity, and size distribution of these characteristic disturbance types and their interactions (36 CFR 219.19).

disturbance—Any relatively discrete event in time that disrupts ecosystem, watershed, community, or species population structure or function, and that changes resources, substrate availability, or the physical environment (36 CFR 219.19).

dynamic reserves—A conservation approach in which protected areas are relocated following changes in environmental conditions, especially owing to disturbance.

early-seral vegetation—Vegetation conditions in the early stages of succession following an event that removes the forest canopy (e.g., timber harvest, wildfire, windstorm), on sites that are capable of developing a closed canopy (Swanson et al. 2014). A nonforest or "pre-forest" condition occurs first, followed by an "early-seral forest" as young shade-intolerant trees form a closed canopy.

ecocultural resources—Valued elements of the biophysical environment, including plants, fungi, wildlife, water, and places, and the social and cultural relationships of people with those elements.

ecological conditions—The biological and physical environment that can affect the diversity of plant and animal communities, the persistence of native species, invasibility, and productive capacity of ecological systems. Ecological conditions include habitat and other influences on species and the environment. Examples of ecological conditions include the abundance and distribution of aquatic and terrestrial habitats, connectivity, roads and other structural developments, human uses, and occurrence of other species (36 CFR 219.19).

ecological forestry—A ecosystem management approach designed to achieve multiple objectives that may include conservation goals and sustainable forest management and which emphasizes disturbance-based management and retention of "legacy" elements such as old trees and dead wood (Franklin et al. 2007).

ecological integrity—The quality or condition of an ecosystem when its dominant ecological characteristics (e.g., 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 (36 CFR 219.19).

ecological keystone species—A species whose ecological functions have extensive and disproportionately large effects on ecosystems relative to its abundance (Power et al. 1996).

ecological sustainability—The capability of ecosystems to maintain ecological integrity (36 CFR 219.19).

economic sustainability—The capability of society to produce and consume or otherwise benefit from goods and services, including contributions to jobs and market and nonmarket benefits (36 CFR 219.19).

ecoregion—A geographic area containing distinctive ecological assemblages, topographic and climatic gradients, and historical land uses.

ecosystem—A spatially explicit, relatively homogeneous unit of the Earth that includes all interacting organisms and elements of the abiotic environment within its boundaries (36 CFR 219.19).

ecosystem diversity—The variety and relative extent of ecosystems (36 CFR 219.19).

ecosystem integrity—See "ecological integrity."

ecosystem management—Management across broad spatial and long temporal scales for a suite of goals, including maintaining populations of multiple species and ecosystem services.

ecosystem services—Benefits that people obtain from ecosystems (see also "provisioning services," "regulating services," "supporting services," and "cultural services").

ectomycorrhizal fungi—Fungal species that form symbiotic relationships with vascular plants through roots, typically aiding their uptake of nutrients. Although other mycorrhizal fungi penetrate their host's cell walls, ectomycorrhizal fungi do not.

endangered species—Any species or subspecies that the Secretary of the Interior or the Secretary of Commerce has deemed in danger of extinction throughout all or a significant portion of its range (16 U.S.C. Section 1532).

endemic—Native and restricted to a specific geographical area.

El Niño Southern Oscillation (ENSO)—A band of anomalously warm ocean water temperatures that occasionally develops off the western coast of South America and can cause climatic changes across the Pacific Ocean. The extremes of this climate pattern's oscillations cause extreme weather (such as floods and droughts) in many regions of the world.

environmental DNA (eDNA)—Genetic material (DNA) contained within small biological and tissue fragments that can be collected from aquatic, terrestrial, and even atmospheric environments, linked to an individual species, and used to indicate the presence of that species.

environmental justice populations—Groups of people who have low incomes or who identify themselves as African American, Asian or Pacific Islander, American Indian or Alaskan Native, or of Hispanic origin.

ephemeral stream—A stream that flows only in direct response to precipitation in the immediate locality (watershed or catchment basin), and whose channel is at all other times above the zone of saturation.

epicormic—Literally, "of a shoot or branch," this term implies growth from a previously dormant bud on the trunk or a limb of a tree.

epiphyte—A plant or plant ally (including mosses and lichens) that grows on the surface of another plant such as a tree, but is not a parasite.

even-aged stand—A stand of trees composed of a single age class (36 CFR 219.19).

fecundity—The reproductive rate of an organism or population.

federally recognized Indian tribe—An Indian tribe or Alaska Native Corporation, band, nation, pueblo, village, or community that the Secretary of the Interior acknowledges

to exist as an Indian tribe under the Federally Recognized Indian Tribe List Act of 1994, 25 U.S.C. 479a (36 CFR 219.19).

fine filter—A conservation approach that focuses on conserving individual species in contrast to a "coarse filter" approach that focuses on conserving ecosystems; these approaches are generally viewed as complementary with fine-filtered strategies tailored to fit particular species that "fall through the pores" of the coarse filter (Hunter 2005). See also "mesofilter."

fire-dependent vegetation types—A vegetative community that evolved with fire as a necessary contributor to its vitality and to the renewal of habitat for its member species.

fire exclusion—Curtailment of wildland fire because of deliberate suppression of ignitions, as well as unintentional effects of human activities such as intensive grazing that removes grasses and other fuels that carry fire (Keane et al. 2002).

fire intensity—The amount of energy or heat release during fire.

fire regime—A characterization of long-term patterns of fire in a given ecosystem over a specified and relatively long period of time, based on multiple attributes, including frequency, severity, extent, spatial complexity, and seasonality of fire occurrence.

fire regime, low frequency, high severity—A fire regime with long return intervals (>200 years) and high levels of vegetation mortality (e.g., ~70 percent basal area mortality in forested ecosystems), often occurring in large patches (>10,000 ac [4047 ha]) (see chapter 3 for more details).

fire regime, moderate frequency, mixed severity—A fire regime with moderate return intervals between 50 and 200 years and mixtures of low, moderate, and high severity; high-severity patches would have been common and frequently large (>1,000 ac [>405 ha]) (see chapter 3 for more details).

fire regime, very frequent, low severity—A fire regime with short return intervals (5 to 25 years) dominated by surface fires that result in low levels of vegetation mortality (e.g., <20 percent basal area mortality in forested ecosystems), with high-severity fire generally limited to small patches (<2.5 ac [1 ha]) (see chapter 3 for more details).

fire regime, frequent, mixed severity—A fire regime with return intervals between 15 and 50 years that burns with a mosaic of low-, moderate-, and high-severity patches (Perry et al. 2011) (see chapter 3 for more details).

fire rotation—Length of time expected for a specific amount of land to burn (some parts might burn more than once or some not at all) based upon the study of past fire records in a large landscape (Turner and Romme 1994).

fire severity—The magnitude of the effects of fire on ecosystem components, including vegetation or soils.

fire suppression—The human act of extinguishing wildfires (Keane et al. 2002).

floodplain restoration—Ecological restoration of a stream or river's floodplain, which may involve setback or removal of levees or other structural constraints.

focal species—A small set of species whose status is assumed to infer the integrity of the larger ecological system to which it belongs, and thus to provide meaningful information regarding the effectiveness of a resource management plan in maintaining or restoring the ecological conditions to maintain the broader diversity of plant and animal communities in the NWPF area. Focal species would be commonly selected on the basis of their functional role in ecosystems (36 CFR 219.19).

food web—Interconnecting chains between organisms in an ecological community based upon what they consume.

Forest Ecosystem Management Assessment Team (FEMAT)—An interdisciplinary team that included expert ecological and social scientists, analysts, and managers assembled in 1993 by President Bill Clinton to develop options for ecosystem management of federal forests within the range of the northern spotted owl (FEMAT 1993).

forest fragmentation—The patterns of dispersion and connectivity of nonhomogeneous forest cover (Riitters et al. 2002). See also "landscape fragmentation" and "habitat fragmentation" for specific meanings related to habitat loss and isolation.

frequency distribution—A depiction, often appearing in the form of a curve or graph, of the abundance of possible values of a variable. In this synthesis report, we speak of the frequency of wildfire patches of various sizes.

fuels (wildland)—Combustible material in wildland areas, including live and dead plant biomass such as trees, shrub, grass, leaves, litter, snags, and logs.

fuels management—Manipulation of wildland fuels through mechanical, chemical, biological, or manual means, or by fire, in support of land management objectives to control or mitigate the effects of future wildland fire.

function (ecological)—Ecological processes, such as energy flow; nutrient cycling and retention; soil development and retention; predation and herbivory; and natural disturbances such as wind, fire, and floods that sustain composition and structure (FSM 2020). See also "key ecological function."

future range of variation (FRV)—The natural fluctuation of pattern components of healthy ecosystems that might occur in the future, primarily affected by climate change, human infrastructure, invasive species, and other anticipated disturbances.

gaps (forest)—Small openings in a forest canopy that are naturally formed when one or a few canopy trees die (Yamamoto 2000).

genotype—The genetic makeup of an individual organism.

glucocorticoid—A class of steroid hormones produced by many species of animals, often as the result of stress.

goals (in land management plans)—Broad statements of intent, other than desired conditions, that do not include expected completion dates (36 CFR part 219.7(e)(2)).

guideline—A constraint on project and activity decision-making that allows for departure from its terms, so long as the purpose of the guideline is met (36 CFR section 219.15(d)(3)). Guidelines are established to help achieve or maintain a desired condition or conditions, to avoid or mitigate undesirable effects, or to meet applicable legal requirements.

habitat—An area with the environmental conditions and resources that are necessary for occupancy by a species and for individuals of that species to survive and reproduce.

habitat fragmentation—Discontinuity in the spatial distribution of resources and conditions present in an area at a given scale that affects occupancy, reproduction, and survival in a particular species (see "landscape fragmentation").

heterogeneity (forest)—Diversity, often applied to variation in forest structure within stands in two dimensions: horizontal (e.g., single trees, clumps of trees, and gaps of no trees), and vertical (e.g., vegetation at different heights from the forest floor to the top of the forest canopy), or across large landscapes (North et al. 2009).

hierarchy theory—A theory that describes ecosystems at multiple levels of organization (e.g., organisms, populations, and communities) in a nested hierarchy.

high-severity burn patch—A contiguous area of high-severity or stand-replacing fire.

historical range of variation (HRV)—Past fluctuation or range of conditions in the pattern of components of ecosystems over a specified period of time.

hybrid ecosystem—An ecosystem that has been modified from a historical state such that it has novel attributes while retaining some original characteristics (see "novel ecosystem").

hybrid—Offspring resulting from the breeding of two different species.

inbreeding depression—Reduced fitness in a population that occurs as the result of breeding between related individuals, leading to increased homogeneity and simplification of the gene pool.

in-channel restoration—Ecological restoration of the channel of a stream or river, often through placement of materials (rocks and wood) or other structural modifications.

individuals, clumps, and openings (ICO) method—A method that incorporates reference spatial pattern targets based upon individual trees, clumps of trees, and canopy openings into silvicultural prescriptions and tree-marking guidelines (Churchill et al. 2013).

Interagency Special Status and Sensitive Species Program (ISSSSP)—A federal agency program, established under the U.S. Forest Service Pacific Northwest Region and Bureau of Land Management Oregon/Washington state office. The ISSSSP superseded the Survey and Manage standards and guidelines under the NWFP and also addresses other species of conservation focus, coordinates development and revision of management recommendations and survey protocols, coordinates data management between the agencies, develops summaries of species biology, and conducts other tasks.

intermittent stream—A stream or reach of stream channel that flows, in its natural condition, only during certain times of the year or in several years, and is characterized by interspersed, permanent surface water areas containing aquatic flora and fauna adapted to the relatively harsh environmental conditions found in these types of environments.

invasive species—An alien species (or subspecies) whose deliberate, accidental, or self-introduction is likely to cause economic or environmental harm or harm to human health (Executive Order 13112).

key ecological function—The main behaviors performed by an organism that can influence environmental conditions or habitats of other species.

key watersheds—Watersheds that are expected to serve as refugia for aquatic organisms, particularly in the short term, for at-risk fish populations that have the greatest potential for restoration, or to provide sources of high-quality water.

land and resource management plan (Forest Service)—A document or set of documents that provides management direction for an administrative unit of the National Forest System (FSH 1909.12.5).

landform—A specific geomorphic feature on the surface of the Earth, such as a mountain, plateau, canyon, or valley.

landscape—A defined area irrespective of ownership or other artificial boundaries, such as a spatial mosaic of terrestrial and aquatic ecosystems, landforms, and plant communities, repeated in similar form throughout such a defined area (36 CFR 219.19).

landscape fragmentation—Breaking up of continuous habitats into patches as a result of human land use and thereby generating habitat loss, isolation, and edge effects (see “habitat fragmentation”).

landscape genetics—An interdisciplinary field of study that combines population genetics and landscape ecology to explore how genetic relatedness among individuals and subpopulations of a species is influenced by landscape-level conditions.

landscape hierarchy—Organization of land areas based upon a hierarchy of nested geographic (i.e., different-sized) units, which provides a guide for defining the functional components of a system and how components at different scales are related to one another.

late-successional forest—Forests that have developed after long periods of time (typically at least 100 to 200 years) following major disturbances, and that contain a major component of shade-tolerant tree species that can regenerate beneath a canopy and eventually grow into the canopy in which small canopy gaps occur (see chapter 3 for more details). Note that FEMAT (1993) and the NWFP also applied this term to older (at least 80 years) forest types, including both old-growth and mature forests, regardless of the shade tolerance of the dominant tree species (e.g., 90-year-old forests dominated by Douglas-fir were termed late successional).

leading edge—The boundary of a species’ range at which the population is geographically expanding through colonization of new sites.

legacy trees—Individual trees that survive a major disturbance and persist as components of early-seral stands (Franklin 1990).

legacies (biological)—Live trees, seed and seedling banks, remnant populations and individuals, snags, large soil aggregates, hyphal mats, logs, uprooted trees, and other biotic features that survive a major disturbance and persist as components of early-seral stands (Franklin 1990, Franklin et al. 2002).

lentic—Still-water environments, including lakes, ponds, and wet meadows.

longitudinal studies—Studies that include repeated observations on the same response variable over time.

lotic—Freshwater environments with running water, including rivers, streams, and springs.

low-income population—A community or a group of individuals living in geographic proximity to one another, or a set of individuals, such as migrant workers or American Indians, who meet the standards for low income and experience common conditions of environmental exposure or effect (CEQ 1997).

managing wildfire for resource objectives—Managing wildfires to promote multiple objectives such as reducing fire danger or restoring forest health and ecological processes rather than attempting full suppression. The terms "managed wildfire" or "resource objective wildfire" have also been used to describe such events (Long et al. 2017). However, fire managers note that many unplanned ignitions are managed using a combination of tactics, including direct suppression, indirect containment, monitoring of fire spread, and even accelerating fire spread, across their perimeters and over their full duration. Therefore, terms that separate "managed" wildfires from fully "suppressed" wildfires do not convey that complexity. (See "Use of wildland fire," which also includes prescribed burning).

matrix—Federal and other lands outside of specifically designated reserve areas, particularly the late-successional reserves under the NWFP, that are managed for timber production and other objectives.

mature forest—An older forest stage (>80 years) prior to old-growth in which trees begin attaining maximum heights and developing some characteristic, for example, 80 to 200 years in the case of old-growth Douglas-fir/western hemlock forests, often (but not always) including big trees (>50 cm diameter at breast height), establishment of late-seral species (i.e., shade-tolerant trees), and initiation of decadence in early species (i.e., shade-intolerant trees).

mesofilter—A conservation approach that "focuses on conserving critical elements of ecosystems that are important to many species, especially those likely to be overlooked by fine-filter approaches, such as invertebrates, fungi, and nonvascular plants" (Hunter 2005).

meta-analysis—A study that combines the results of multiple studies**.**

minority population—A readily identifiable group of people living in geographic proximity with a population that is at least 50 percent minority; or, an identifiable group that has a meaningfully greater minority population than the adjacent geographic areas, or may also be a geographically dispersed/transient set of individuals such as migrant workers or Americans Indians (CEQ 1997).

mitigation (climate change)—Efforts to reduce anthropogenic alteration of climate, in particular by increasing carbon sequestration.

monitoring—A systematic process of collecting information to track implementation (implementation monitoring), to evaluate effects of actions or changes in conditions or relationships (effectiveness monitoring), or to test underlying assumptions (validation monitoring) (see 36 CFR 219.19).

mosaic—The contiguous spatial arrangement of elements within an area. In regions, this is typically the upland vegetation patches, large urban areas, large bodies of water, and large areas of barren ground or rock. However, regional mosaics can also be described in terms of land ownership, habitat

patches, land use patches, or other elements. For landscapes, this is typically the spatial arrangement of landscape elements.

multiaged stands—Forest stands having two or more age classes of trees; this includes stands resulting from variable-retention silvicultural systems or other traditionally even-aged systems that leave residual or reserve (legacy) trees.

multiple use—The management of all the various renewable surface resources of the National Forest System so that they are used in the combination that will best meet the needs of the American people; making the most judicious use of the land for some or all of these resources or related services over areas large enough to provide sufficient latitude for periodic adjustments in use to conform to changing needs and conditions; that some land will be used for less than all of the resources; and harmonious and coordinated management of the various resources, each with the other, without impairment of the productivity of the land, with consideration being given to the relative values of the various resources, and not necessarily the combination of uses that will give the greatest dollar return or the greatest unit output, consistent with the Multiple-Use Sustained-Yield Act of 1960 (16 U.S.C. 528–531) (36 CFR 219.19).

natal site—Location of birth.

native knowledge—A way of knowing or understanding the world, including traditional ecological, and social knowledge of the environment derived from multiple generations of indigenous peoples' interactions, observations, and experiences with their ecological systems. This knowledge is accumulated over successive generations and is expressed through oral traditions, ceremonies, stories, dances, songs, art, and other means within a cultural context (36 CFR 219.19).

native species—A species historically or currently present in a particular ecosystem as a result of natural migratory or evolutionary processes and not as a result of an accidental or deliberate introduction or invasion into that ecosystem (see 36 CFR 219.19).

natural range of variation (NRV)—The variation of ecological characteristics and processes over specified scales of time and space that are appropriate for a given management application (FSH 1909.12.5).

nested hierarchy—The name given to the hierarchical structure of groups within groups used to classify organisms.

nontimber forest products (also known as "special forest products")—Various products from forests that do not include logs from trees but do include bark, berries, boughs, bryophytes, bulbs, burls, Christmas trees, cones, ferns, firewood, forbs, fungi (including mushrooms), grasses, mosses, nuts, pine straw, roots, sedges, seeds, transplants, tree sap, wildflowers, fence material, mine props, posts and poles, shingle and shake bolts, and rails (36 CFR part 223 Subpart G).

novel ecosystem—An ecosystem that has experienced large and potentially irreversibly modifications to abiotic conditions or biotic composition in ways that result in a composition of species, ecological communities, and functions that have never before existed, and that depart from historical analogs (Hobbs et al. 2009). See "hybrid ecosystem" for comparison.

old-growth forest—A forest distinguished by old trees (>200 years) and related structural attributes that often (but not always) include large trees, high biomass of dead wood (i.e., snags, down coarse wood), multiple canopy layers, distinctive species composition and functions, and vertical and horizontal diversity in the tree canopy (see chapter 3). In dry, fire-frequent forests, old growth is characterized by large, old fire-resistant trees and relatively open stands without canopy layering.

palustrine—Inland, nontidal wetlands that may be permanently or temporarily flooded and are characterized by the presence of emergent vegetation such as swamps, marshes, vernal pools, and lakeshores.

passive management—A management approach in which natural processes are allowed to occur without human intervention to reach desired outcomes.

patch—A relatively small area with similar environmental conditions, such as vegetative structure and composition. Sometimes used interchangeably with vegetation or forest stand.

Pacific Decadal Oscillation (**PDO**)—A recurring (approximately decadal-scale) pattern of ocean-atmosphere —a stream or reach of a channel that flows continuously or nearly so throughout the year and whose upper surface is generally lower than the top of the zone of saturation in areas adjacent to the stream.

perennial stream—A stream or reach of a channel that flows continuously or nearly so throughout the year and whose upper surface is generally lower than the top of the zone of saturation in areas adjacent to the stream.

phenotype—Physical manifestation of the genetic makeup of an individual and its interaction with the environment.

place attachment—The "positive bond that develops between groups or individuals and their environment" (Jorgensen and Stedman 2001: 234).

place dependence— "The strength of an individual's subjective attachment to specific places" (Stokols and Shumaker 1982: 157).

place identity—Dimensions of self that define an individual's [or group's] identity in relation to the physical environment through ideas, beliefs, preferences, feelings, values, goals, and behavioral tendencies and skills (Proshansky 1978).

place-based planning—"A process used to involve stakeholders by encouraging them to come together to collectively define place meanings and attachments" (Lowery and Morse 2013: 1423).

plant association—A fine level of classification in a hierarchy of potential vegetation that is defined in terms of a climax-dominant overstory tree species and typical understory herb or shrub species.

population bottleneck—An abrupt decline in the size of a population from an event, which often results in deleterious effects such as reduced genetic diversity and increased probability of local or global extirpation.

potential vegetation type (PVT)—Native, late-successional (or "climax") plant community that reflects the regional climate, and dominant plant species that would occur on a site in absence of disturbances (Pfister and Arno 1980).

poverty rate—A measure of financial income below a threshold that differs by family size and composition.

precautionary principle—A principle that if an action, policy, or decision has a suspected risk of causing harm to the public or to the environment, and there is no scientific consensus that it is not harmful, then the burden of proof that it is not harmful falls on those making that decision. Particular definitions of the principle differ, and some applications use the less formal term, "precautionary approach." Important qualifications associated with many definitions include (1) the perceived harm is likely to be serious, (2) some scientific analysis suggests a significant but uncertain potential for harm, and (3) applications of the principle emphasize generally constraining an activity to mitigate it rather than "resisting" it entirely (Doremus 2007).

prescribed fire—A wildland fire originating from a planned ignition to meet specific objectives identified in a written and approved prescribed fire plan for which National Environmental Policy Act requirements (where applicable) have been met prior to ignition (synonymous with controlled burn).

primary recreation activity—A single activity that caused a recreation visit to a national forest.

probable sale quantity—An estimate of the average amount of timber likely to be awarded for sale for a given area (such as the NWFP area) during a specified period.

provisioning services—A type of ecosystem service that includes clean air and fresh water, energy, food, fuel, forage, wood products or fiber, and minerals.

public participation geographic information system (PPGIS)—Using spatial decisionmaking and mapping tools to produce local knowledge with the goal of including and empowering marginalized populations (Brown and Reed 2009).

public values—Amenity values (scenery, quality of life); environmental quality (clean air, soil, and water); ecological

values (biodiversity); public use values (outdoor recreation, education, subsistence use); and spiritual or religious values (cultural ties, tribal history).

record of decision (ROD)—The final decision document that amended the planning documents of 19 national forests and seven Bureau of Land Management districts within the range of the northern spotted owl (the NWFP area) in April 1994 (Espy and Babbit 1994).

recreation opportunity—An opportunity to participate in a specific recreation activity in a particular recreation setting to enjoy desired recreation experiences and other benefits that accrue. Recreation opportunities include non-motorized, motorized, developed, and dispersed recreation on land, water, and in the air (36 CFR 219.19).

redundancy—The presence of multiple occurrences of ecological conditions, including key ecological functions (functional redundancy), such that not all occurrences may be eliminated by a catastrophic event.

refugia—An area that remains less altered by climatic and environmental change (including disturbances such as wind and fire) affecting surrounding regions and that therefore forms a haven for relict fauna and flora.

regalia—Dress and special elements made from a variety of items, including various plant and animal materials, and worn for tribal dances and ceremonies.

regulating services—A type of ecosystem service that includes long-term storage of carbon; climate regulation; water filtration, purification, and storage; soil stabilization; flood and drought control; and disease regulation.

representativeness—The presence of a full array of ecosystem types and successional states, based on the physical environment and characteristic disturbance processes.

reserve—An area of land designated and managed for a special purpose, often to conserve or protect ecosystems, species, or other natural and cultural resources from particular human activities that are detrimental to achieving the goals of the area.

resilience—The capacity of a system to absorb disturbance and reorganize (or return to its previous organization) so as to still retain essentially the same function, structure, identity, and feedbacks (see FSM Chapter 2020 and see also "socioecological resilience"). Definitions emphasize the capacity of a system or its constituent entities to respond or regrow after mortality induced by a disturbance event, although broad definitions of resilience may also encompass "resistance" (see below), under which such mortality may be averted.

resistance—The capacity of a system or an entity to withstand a disturbance event without much change.

restoration economy—Diverse economic activities associated with the restoration of structure or function to terrestrial and aquatic ecosystems (Nielsen-Pincus and Moseley 2013).

restoration, ecological—The process of assisting the recovery of an ecosystem that has been degraded, damaged, or destroyed. 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 (36 CFR 219.19).

restoration, functional—Restoration of dynamic abiotic and biotic processes in degraded ecosystems, without necessarily a focus on structural condition and composition.

riparian areas—Three-dimensional ecotones (the transition zone between two adjoining communities) 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 (36 CFR 219.19).

riparian management zone—Portions of a watershed in which riparian-dependent resources receive primary emphasis, and for which plans include Plan components to maintain or restore riparian functions and ecological functions (36 CFR 219.19).

riparian reserves—Reserves established along streams and rivers to protect riparian ecological functions and processes

necessary to create and maintain habitat for aquatic and riparian-dependent organisms over time and ensure connectivity within and between watersheds. The Aquatic Conservation Strategy in the NWFP record of decision included standards and guidelines that delineated riparian reserves.

risk—A combination of the probability that a negative outcome will occur and the severity of the subsequent negative consequences (36 CFR 219.19).

rural restructuring—Changes in demographic and economic conditions owing to declines in natural resource production and agriculture (Nelson 2001).

scale—In ecological terms, the extent and resolution in spatial and temporal terms of a phenomenon or analysis, which differs from the definition in cartography regarding the ratio of map distance to Earth surface distance (Jenerette and Wu 2000).

scenic character—A combination of the physical, biological, and cultural images that gives an area its scenic identity and contributes to its sense of place. Scenic character provides a frame of reference from which to determine scenic attractiveness and to measure scenic integrity (36 CFR 219.19).

science synthesis—A narrative review of scientific information from a defined pool of sources that compiles and integrates and interprets findings and describes uncertainty, including the boundaries of what is known and what is not known.

sense of place—The collection of meanings, beliefs, symbols, values, and feelings that individuals or groups associate with a particular locality (Williams and Stewart 1998).

sensitive species—Plant or animal species that receive special conservation attention because of threats to their populations or habitats, but which do not have special status as listed or candidates for listing under the Endangered Species Act.

sensitivity—In ecological contexts, the propensity of communities or populations to change when subject to disturbance, or the opposite of resistance (see "community resistance").

sink population—A population in which reproductive rates are lower than mortality rates but that is maintained by immigration of individuals from outside of that population (see also "source population").

social sustainability—"The capability of society to support the network of relationships, traditions, culture, and activities that connect people to the land and to one another, and support vibrant communities" (36 CFR 219.19). The term is commonly invoked as one of the three parts of a "triple-bottom line" alongside environmental and economic considerations. The concept is an umbrella term for various topics such as quality of life, security, social capital, rights, sense of place, environmental justice, and community resilience, among others discussed in this synthesis.

socioecological resilience—The capacity of socioecological systems (see "socioecological system") to cope with, adapt to, and influence change; to persist and develop in the face of change; and to innovate and transform into new, more desirable configurations in response to disturbance.

socioecological system (or social-ecological system)—A coherent system of biophysical and social factors defined at several spatial, temporal, and organizational scales that regularly interact, continuously adapt, and regulate critical natural, socioeconomic, and cultural resources (Redman et al. 2004); also described as a coupled-human and natural system (Liu et al. 2007).

source population—A population in which reproductive rates exceed those of mortality rates so that the population has the capacity to increase in size. The term is also often used to denote when such a population contributes emigrants (dispersing individuals) that move outside the population, particularly when feeding a sink population.

special forest products—See "nontimber forest products."

special status species—Species that have been listed or proposed for listing as threatened or endangered under the Endangered Species Act.

species of conservation concern—A species, other than federally recognized as a threatened, endangered, proposed,

or candidate species, that is known to occur in the NWFP area and for which the regional forester has determined that the best available scientific information indicates substantial concern about the species' capability to persist over the long term in the Plan area (36 CFR 219.9(c)).

stand—A descriptor of a land management unit consisting of a contiguous group of trees sufficiently uniform in age-class distribution, composition, and structure, and growing on a site of sufficiently uniform quality, to be a distinguishable unit.

standard—A mandatory constraint on project and activity decisionmaking, established to help achieve or maintain the desired condition or conditions, to avoid or mitigate undesirable effects, or to meet applicable legal requirements.

stationarity—In statistics, a process that, while randomly determined, is not experiencing a change in the probability of outcomes.

stewardship contract—A contract designed to achieve land management goals while meeting local and rural community needs, including contributing to the sustainability of rural communities and providing a continuing source of local income and employment.

strategic surveys—One type of field survey, specified under the NWFP, designed to fill key information gaps on species distributions and ecologies by which to determine if species should be included under the Plan's Survey and Manage species list.

stressors—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 (36 CFR 219.19).

structure (ecosystem)—The organization and physical arrangement of biological elements such as snags and down woody debris, vertical and horizontal distribution of vegetation, stream habitat complexity, landscape pattern, and connectivity (FSM 2020).

supporting services—A type of ecosystem service that includes pollination, seed dispersal, soil formation, and nutrient cycling.

Survey and Manage program—A formal part of the NWFP that established protocols for conducting various types of species surveys, identified old-forest-associated species warranting additional consideration for monitoring and protection (see "Survey and Manage species"), and instituted an annual species review procedure that evaluated new scientific and monitoring information on species for potentially recommending changes in their conservation status, including potential removal from the Survey and Manage species list.

Survey and Manage species—A list of species, compiled under the Survey and Manage program of the NWFP, that were deemed to warrant particular attention for monitoring and protection beyond the guidelines for establishing late-successional forest reserves.

sustainability—The capability to meet the needs of the present generation without compromising the ability of future generations to meet their needs (36 CFR 219.19).

sustainable recreation—The set of recreation settings and opportunities in the National Forest System that is ecologically, economically, and socially sustainable for present and future generations (36 CFR 219.19).

sympatric—Two species or populations that share a common geographic range and coexist.

threatened species—Any species that the Secretary of the Interior or the Secretary of Commerce has determined is likely to become an endangered species within the foreseeable future throughout all or a significant portion of its range. Threatened species are listed at 50 CFR sections 17.11, 17.12, and 223.102.

timber harvest—The removal of trees for wood fiber use and other multiple-use purposes (36 CFR 219.19).

timber production—The purposeful growing, tending, harvesting, and regeneration of regulated crops of trees to be cut into logs, bolts, or other round sections for industrial or consumer use (36 CFR 219.19).

topo-edaphic—Related to or caused by particular soil conditions, as of texture or drainage, rather than by physiographic or climatic factors within a defined region or area.

traditional ecological knowledge—"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" (Berkes et al. 2000: 1252). See also "native knowledge."

trailing edge—When describing the range of a species, the boundary at which the species' population is geographically contracting through local extinction at occupied sites.

trophic cascade—Changes in the relative populations of producers, herbivores, and carnivores following the addition or removal of top predators and the resulting disruption of the food web.

uncertainty—Amount or degree of confidence as a result of imperfect or incomplete information.

understory—Vegetation growing below the tree canopy in a forest, including shrubs and herbs that grow on the forest floor.

use of wildland fire—Management of either wildfire or prescribed fire to meet resource objectives specified in land or resource management plans (see "Managing wildfire for resource objectives" and "Prescribed fire").

variable-density thinning—The method of thinning some areas within a stand to a different density (including leaving dense, unthinned areas) than other parts of the stand, which is typically done to promote ecological diversity in a relatively uniform stand.

vegetation series (plant community)—The highest level of the fine-scale component (plant associations) of potential vegetation hierarchy based on the dominant plant species that would occur in late-successional conditions in the absence of disturbance.

vegetation type—A general term for a combination or community of plants (including grasses, forbs, shrubs, or trees), typically applied to existing vegetation rather than potential vegetation.

viable population—A group of breeding individuals of a species capable of perpetuating itself over a given time scale.

vital rates—Statistics describing population dynamics such as reproduction, mortality, survival, and recruitment.

watershed—A region or land area drained by a single stream, river, or drainage network; a drainage basin (36 CFR 219.19).

watershed analysis—An analytical process that characterizes watersheds and identifies potential actions for addressing problems and concerns, along with possible management options. It assembles information necessary to determine the ecological characteristics and behavior of the watershed and to develop options to guide management in the watershed, including adjusting riparian reserve boundaries.

watershed condition assessment—A national approach used by the U.S. Forest Service to evaluate condition of hydrologic units based on 12 indicators, each composed of various attributes (USDA FS 2011).

watershed condition—The state of a watershed based on physical and biogeochemical characteristics and processes (36 CFR 219.19).

watershed restoration—Restoration activities that focus on restoring the key ecological processes required to create and maintain favorable environmental conditions for aquatic and riparian-dependent organisms.

well-being—The condition of an individual or group in social, economic, psychological, spiritual, or medical terms.

wilderness—Any area of land designated by Congress as part of the National Wilderness Preservation System that was established by the Wilderness Act of 1964 (16 U.S.C. 1131–1136) (36 CFR 219.19).

wildlife—Undomesticated animal species, including amphibians, reptiles, birds, mammals, fish, and invertebrates or even all biota, that live wild in an area without being introduced by humans.

wildfire—Unplanned ignition of a wildland fire (such as a fire caused by lightning, volcanoes, unauthorized and accidental human-caused fires), and escaped prescribed fires.

wildland-urban interface (WUI)—The line, area, or zone where structures and other human development meet or intermingle with undeveloped wildland or vegetation fuels.

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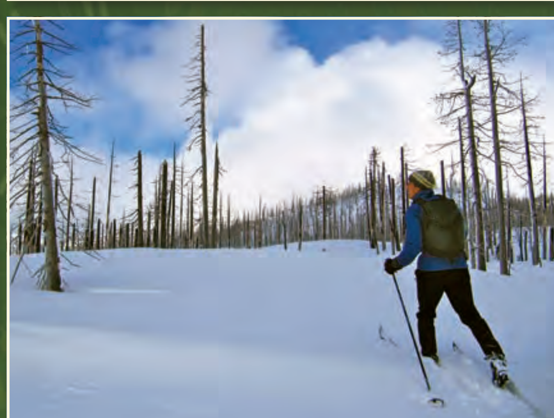

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Cover photos: Left: Inner City Youth Institute, High School Natural Resources Camp, Multnomah Falls, Oregon; photo by USDA Forest Service. Upper middle: roasting salmon over an open fire; photo by Jon Ivy, Coquille Indian Tribe. Lower middle: backcountry skiing on the Willamette National Forest; photo by Emily Jane Davis. Right: a field tour with the Lakeview Forest Landscape Collaborative in the Fremont-Winema National Forest; photo by Tom Spies, USDA Forest Service.

Synthesis of Science to Inform Land Management Within the Northwest Forest Plan Area

Volume 3

Thomas A. Spies, Peter A. Stine, Rebecca Gravenmier,
Jonathan W. Long, and Matthew J. Reilly, Technical Coordinators

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Abstract

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The 1994 Northwest Forest Plan (NWFP) was developed to resolve debates over old-growth forests, endangered species, and timber production on federal forests in the range of the northern spotted owl. This three-volume science synthesis, which consists of 12 chapters that address various ecological and social concerns, is intended to inform forest plan revision and forest management within the NWFP area. Land managers with the U.S. Forest Service provided questions that helped guide preparation of the synthesis, which builds on the 10-, 15-, and 20-year NWFP monitoring reports and synthesizes the vast body of relevant scientific literature that has accumulated in the 24 years since the NWFP was initiated. It identifies scientific findings, lessons learned, and uncertainties and also evaluates competing science and provides considerations for management.

This synthesis finds that the NWFP has protected dense old-growth forests and maintained habitat for northern spotted owls, marbled murrelets, aquatic organisms, and other species despite losses from wildfire and low levels of timber harvest on federal lands. Even with reductions in the loss of older forests, northern spotted owl populations continue to decline. Moreover, a number of other goals have not been met, including producing a sustainable supply of timber, decommissioning roads, biodiversity monitoring, significant levels of restoration of riparian and dry forests, and adaptation and learning through adaptive management.

New conservation concerns have arisen, including a major threat to spotted owl populations from expanding populations of the nonnative barred owl, effects of fire suppression on forest succession, fire behavior in dry forests, and lack of development of diverse early-seral vegetation as a result of fire suppression in drier parts of moist forests. Climate change and invasive species have emerged as threats to native biodiversity, and expansion of the wildland-urban interface has limited the ability of managers to restore fire to fire-dependent ecosystems.

The policy, social, and ecological contexts for the NWFP have changed since it was implemented. The contribution of federal lands continues to be essential to the conservation and recovery of fish listed under the Endangered Species Act and northern spotted owl and marbled murrelet populations. Conservation on federal lands alone, however, is likely insufficient to reach the goals of the NWFP or the newer goals of the 2012 planning rule, which emphasizes managing for ecosystem goals (e.g. ecological resilience) and a few species of concern, rather than the population viability of hundreds of individual species.

The social and economic basis of many traditionally forest-dependent communities have changed in 24 years, and many are now focused on amenity values. The capacities of human communities and federal agencies, collaboration among stakeholders, the interdependence of restoration and the timber economy, and the role of amenity- or recreation-based communities and ecosystem services are important considerations in managing for ecological resilience, biodiversity conservation, and social and economic sustainability.

A growing body of scientific evidence supports the importance of active management or restoration inside and outside reserves to promote biodiversity and ecological resilience. Active management to promote heterogeneity of vegetation conditions is important to sustaining tribal ecocultural resources. Declines in agency capacity, lack of markets for small-diameter wood, lack of wood processing infrastructure in some areas, and lack of social agreement have limited the amount of active management for restoration on federal lands. All management choices involve social and ecological tradeoffs related to the goals of the NWFP. Collaboration, risk management, adaptive management, and monitoring are considered the best ways to deal with complex social and ecological systems with futures that are difficult to predict and affect through policy and land management actions.

Keywords: Northwest Forest Plan, science, management, restoration, northern spotted owl, marbled murrelet, climate change, socioeconomic, environmental justice.

Preface

In 2015, regional foresters in the Pacific Northwest and Pacific Southwest Regions of the USDA Forest Service requested that the Pacific Northwest and Pacific Southwest Research Stations prepare a science synthesis to inform revision of existing forest plans under the 2012 planning rule in the area of the Northwest Forest Plan (NWFP, or Plan). Managers provided an initial list of hundreds of questions to the science team, which reduced to them to 73 questions deemed most feasible for addressing through a study of current scientific literature. The stations assembled a team of 50 scientists with expertise in biological, ecological, and socioeconomic disciplines. At the suggestion of stakeholders, a literature reference database was placed online so the public could submit additional scientific literature for consideration. By spring 2016, writing was underway on 12 chapters that covered ecological and social sciences.

The draft synthesis, which was ready for peer and public review by fall 2016, went through a special review process because it was classified as "highly influential science" in accordance with the Office of Management and Budget's 2004 "Final Information Quality Bulletin for Peer Review." The synthesis was classified as such because it fit the category of a scientific assessment that is novel, controversial, or precedent-setting, or has significant interagency interest. Per the bulletin, the two research stations commissioned an independent entity, the Ecological Society of America (ESA), to manage the peer-review process, including the selection of peer reviewers.

The bulletin also stipulates that such an assessment be made available to the public through a public meeting to enable the public to bring scientific issues to the attention of peer reviewers. Accordingly, a public forum was held in Portland, Oregon, in December 2016. For those who could not travel to Portland, the forum was accessible via live Web stream, and multiple national forests within the NWFP area hosted remote viewing. Written comments on the draft synthesis were collected for 2 months. This generated 130 public comments, totaling 890 pages, which were given to the peer reviewers for consideration in their review, as they deemed appropriate. The OMB guidelines further direct that the peer-review process be transparent by making available to the public the ESA's written guidance to the reviewers, the peer reviewer's names, the peer review reports, and the responses of the authors to the peer reviewer comments—all of which are available at https://www.fs.fed.us/pnw/research/science-synthesis/index.shtml.

The peer reviewer comments, which were received in spring 2017 and informed by public input, resulted in substantive revisions to chapters of the synthesis. The result is this three-volume general technical report (an executive summary of the synthesis is available as a separate report). This document is intended to support upcoming management planning on all public lands in the Plan area, but is expected to serve primarily lands managed by the U.S. Forest Service. We hope it will be a valuable reference for managers and others who seek to understand the scientific basis and possible tradeoffs associated with forest plan revision and management decisions. The synthesis also provides an extensive list of published sources where readers can find further information.

We understand that the term “synthesis” can have many different meanings. For our purposes, it represents a compilation and interpretation of relevant scientific findings that pertain to key issues related to the NWFP that were identified by managers and by the authors of the document. Such a compilation not only summarizes science by topic areas but also interprets that science in light of management goals, characterizes competing science, and makes connections across scientific areas, addressing multilayered and interacting ecological and socioeconomic issues. In a few cases, simple analyses of existing data were conducted and methods were provided to reviewers.

The synthesis builds upon the 10-, 15-, and 20-year NWFP monitoring reports, and authors considered well over 4,000 peer-reviewed publications based on their knowledge as well as publications submitted by the public and others suggested by peer reviewers. For some of the questions posed by land managers, there was ample scientific research from the Plan area. For many of the questions, however, little research existed that was specific to the area. In such cases, studies from other regions or current scientific theory were used to address the questions to the extent possible. In many cases, major scientific uncertainties were found; these are highlighted by the authors.

The synthesis chapters characterize the state of the science but they do not develop management alternatives, analyze management tradeoffs, or offer recommendations as to what managers should do. The synthesis does identify ideas, facts, and relationships that managers may want to consider as they develop plans and make management decisions about particular issues. The final chapter attempts to integrate significant cross-cutting issues, e.g., ecological and socioeconomic interdependencies, compatibility of different management goals, and tradeoffs associated with different restoration actions. All the chapters identify where more research is needed to fill critical information gaps.

We would like to acknowledge the peer reviewers who considered hundreds of public comments as part of the process of reviewing our lengthy draft manuscripts. We also thank the many contributors to the development of the synthesis in draft and final form, including those who provided editing, layout, database, and other support services.

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Log deck resulting from a mechanical forest thinning operation in a Washington Douglas-fir forest. Photo by Robert Keefe.

Chapter 8: Socioeconomic Well-Being and Forest Management in Northwest Forest Plan-Area Communities

Susan Charnley, Jeffrey D. Kline, Eric M. White, Jesse Abrams, Rebecca J. McLain, Cassandra Moseley, and Heidi Huber-Stearns[1]

Introduction

Given the need to conserve forest biodiversity and produce forest products, President Clinton's vision for the Northwest Forest Plan (NWFP, or Plan) was that it would provide "a balanced and comprehensive strategy for the conservation and management of forest ecosystems, while maximizing economic and social benefits from forests" (USDA and USDI 1994: E-1). The Plan was expected to support the production of a predictable, sustainable level of timber and nontimber resources from federal forests to contribute to the stability of local and regional economies over the long term (Charnley et al. 2006a). The Plan also aimed to help rural communities affected by cutbacks in federal timber production by providing economic assistance programs to promote long-term economic development and diversification and minimize the adverse effects of job loss from reductions in timber harvesting (Dillingham 2006).

To monitor effectiveness in achieving these goals, the NWFP record of decision contained two socioeconomic monitoring questions: (1) Are predictable levels of timber and nontimber resources available and being produced? (2) Are local communities and economies experiencing positive or negative changes that may be associated with federal forest management? (USDA and USDI 1994: E-9). After the first 10 years of socioeconomic monitoring, the Regional Interagency Executive Committee identified a new monitoring question: what is the status and trend of social and economic well-being in the Northwest Forest Plan area (at the county level) (Grinspoon et al. 2016)? Socioeconomic well-being in relation to federal forest management continues to be an important concern among agency managers.

Thus, the goal of this chapter is to synthesize findings from NWFP monitoring and scientific research on the relationship between federal forest management and socioeconomic well-being in forest communities in the NWFP area (which includes 72 counties in western Washington, western Oregon, and northwestern California), recognizing that there is a reciprocal relationship between them. We build on Breslow et al. (2016) and define socioeconomic well-being as a state of being with others and the environment that arises when human needs are met, when people can act meaningfully to pursue their individual and collective goals, and when people and communities enjoy a satisfactory quality of life.

"Community" has been defined in many ways in the literature, making it difficult to adopt one general definition here. However, our main focus is on communities of place having social and economic ties to nearby forests, which are typically located in rural areas, where the effects of the NWFP were greatest. Communities are not homogenous; they contain residents with diverse socioeconomic circumstances, values, interests, and relations to federal forests, and federal forest management affects different community residents differently. Although our focus is on the community as a unit of analysis, where possible we draw attention to the diversity that exists among subpopulations in the Plan area. Chapter 10 complements this chapter with a focus on low-income and minority populations and their relations to federal forests in the Plan area.

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Guiding Questions

This chapter focuses on six key questions pertaining to socioeconomic well-being in NWFP-area communities and federal forest management:

1. What is the statutory and policy foundation for considering socioeconomic well-being in federal forest management, and how does it reflect changing understandings of the relation between community well-being and federal forest management over time?
2. What has been the impact of the NWFP on rural communities in the Plan area?
3. How have social and economic conditions in rural communities in the Plan area changed over the past two decades?
4. How do goods, services, and opportunities from federal forests contribute to socioeconomic well-being in rural communities?
5. How do rural communities contribute to federal forest management?
6. What implications do changes in land use and land ownership over the past two decades have for federal forest management?

We summarize key findings pertaining to these questions at the beginning of the sections, below, which address each one in depth.

Key Findings

Statutory and Policy Foundation and Evolving Understandings of Socioeconomic Well-Being and Federal Forest Management

The relationship between federal forest management and community well-being has been understood from different perspectives over time, with both the Forest Service and Bureau of Land Management (BLM) being concerned with community well-being historically. The National Forest System was inspired in part by concerns about the predominant timber harvesting practices of the late 19th century, in which mobile logging camps exploited forests and then moved on without considering reforestation needs. Not only was this pattern of timber exploitation detrimental to U.S. forest stocks, it also raised concerns about the unstable

Summary—
Laws that direct the U.S. Forest Service and Bureau of Land Management (BLM) to create social and economic benefits for communities and the public date back to the inception of the agencies. Legislation in the first half of the 20th century emphasized provision of a continuous flow of timber from federal forests to promote economic stability in the forestry industry and forest communities. Legislation passed in the second half of the 20th century strengthened environmental goals and planning requirements associated with federal forest management, but also reaffirmed the economic goals of the Forest Service, and added or expanded social goals. Law and policy have also often given special consideration to people living near national forests and BLM-managed Oregon and California (O&C) Railroad Revested Lands in the form of payments to counties, for example.

With adoption of the NWFP, the goal of providing social and economic benefits to communities continued alongside an increased focus on environmental protection and restoration. At the same time, community benefit began to be conceptualized as coming from activities beyond traditional timber harvest and milling activities, such as ecosystem management, forest and watershed restoration, outdoor recreation, and the harvest of nontimber forest products. This shift reflected a change in thinking about well-being in forest communities from being a product of nondeclining, even flows of timber, to being influenced by a host of commodity and noncommodity benefits from federal forest lands.

Subsequent to the adoption of the NWFP and the occurrence of several large, high-visibility wildfires, wildfire became the central focus of national forest management-related law and policy. In parallel to the adoption of the NWFP, wildfire policy has shifted from a 20th-century focus on using fire suppression to protect natural resources (i.e., timber), to a focus on protecting firefighters and communities—especially

homes and other structures, community preparedness and forest restoration to create wildfire-resilient landscapes. In turn, the concept of community resilience has emerged, which focuses on the ability of a community to successfully cope with and adapt to natural disturbances and change. Wildfire is now a critical issue to address in the context of federal forest management and community socioeconomic well-being.

livelihoods and lifestyles of forest workers, and communities experiencing boom and bust economic cycles associated with unsustainable logging practices (Hibbard 1999, Quirke et al. 2017). Given many rural communities' high degree of economic dependency on lands that were designated as national forests, there has been a longstanding public policy concern with the effects of national forest management on community "stability" (Dana 1918, Kaufman and Kaufman 1946). Although the BLM came to manage forest lands within the NWFP area under a different set of historical circumstances, the policy framework for managing these Oregon and California (O&C) Railroad Revested Lands has likewise shown a long-standing concern with providing local community benefits (Richardson 1980). Thus, the NWFP focus on the impacts of reduced federal timber harvesting on rural community well-being has continuity with broader policy goals reflected throughout the histories of these agencies.

Conceptually, the social and economic dimensions of laws and policies associated with the Forest Service and BLM can be broken into two categories: (1) those that require or authorize the agencies to create social and economic benefits for the nation or particular populations, and (2) those that authorize or require the agencies to provide opportunities for input into the planning and management process by the public as a whole, or particular subpopulations. The former is the focus of this section.

Social and economic goals in federal forest management law and policy—

Laws that direct the Forest Service and BLM to create social and economic benefits for communities and the public date back to their inception. In the Forest Service's Organic Act of 1897, for example, forest reserves (later national forests) were to provide for water flow and a continuous supply of timber (Wilkinson and Anderson 1987). Under the Organic Act, a central goal of creating forest reserves was to ensure that western timber did not end up in the hands of private industry monopolies and was continually accessible for the "greatest good." Throughout the second half of the 20th century, the focus on timber as the primary public benefit of national forest and BLM O&C land management increasingly came into conflict with other uses and benefits of federal forest lands. Although the National Forest Management Act (NFMA), the BLM's Federal Land Policy and Management Act, the Wilderness Act, and other laws passed in the 1960s and 1970s strengthened environmental goals and planning requirements, Congress also reaffirmed the economic goals of the Forest Service, and added or expanded social goals in these same laws. For example, NFMA expanded the authority of the agencies to harvest timber by legalizing clearcutting, and the Wilderness Act was as much about protecting special places for recreation and scenic beauty as it was about environmental protection in its own right.

In parallel to the "greatest good" concept embedded in much of federal land management legislation, law and policy have also often given special consideration to people living near national forests and BLM O&C lands. The most well known of these laws is the 1908 Twenty-Five Percent Fund Act (Public Law 60-136), which requires the Forest Service to pay 25 percent of its revenue generated from timber sales and other goods and services from national forests to counties to help fund roads and schools. On the BLM side, although the revesting of O&C lands in western Oregon to BLM management was an effort to get timberlands out of the hands of a corrupt railroad company, decisions about what to do with those lands revolved around the likely local economic impacts on communities, specifically the local timber industry and local taxation (Richardson 1980). Ultimately, sustained-yield timber production, and paying counties a portion of agency timber revenues, also became an obligation of O&C forest management (Richardson 1980). Fifty percent of timber revenues from BLM O&C and Coos Bay Wagon Road lands were returned to counties to use for any general county purpose (Phillips 2006b).

The Sustained-Yield Forest Management Act of 1944 (16 U.S.C. Section 583), which authorized the secretaries of the Department of Agriculture and Interior to create sustained-yield units (or "cutting circles") on federal, or combined federal and private lands, is another example of local community consideration in forest policy. The act provided local lumber mills with exclusive access to federal timber and encouraged a continuous supply of timber that would stabilize forest industries, employment, and communities near federal forests. As reflected in the act, from the 1940s through the 1980s, national forest management was thought to be important in contributing to "community stability," defined in terms of stable timber industry employment and income in forest communities (Le Master and Beuter 1989). Contributing to community stability through a policy of sustained-yield timber harvesting to provide a nondeclining, even flow of forest products and associated jobs and income was a central goal of national forest management between the 1940s and 1980s (Le Master and Beuter 1989, chapters in Lee et al. 1990) (fig. 8-1).

The belief that national forest management can ensure community stability was questioned in the 1980s as it was recognized that many variables influence social and economic well-being in rural communities (Charnley et al. 2008b, Cook 1995, Force et al. 1993, Nadeau et al. 2003, Power 2006, Sturtevant and Donoghue 2008). Federal forest managers cannot ensure community economic stability through their management actions alone, particularly if such stability is assumed to arise from a consistent flow of timber. However, management of federal forests and investments in federal forest management (including the presence of a federal workforce) can contribute to community stability and business vitality. The positive economic and social outcomes in the Blue Mountains of Oregon from the Pacific Northwest Region's "eastside strategy" and the state of Oregon's Federal Forest Restoration Program (previously the Federal Forest Health Program) illustrate how investment in federal forest management can promote community well-being (Bennett et al. 2015, White et al. 2015).

Under the NWFP, the goal of providing social and economic benefits to communities continued even as an increased focus on environmental protection and restoration challenged the provisioning of traditional timber-based benefits from federal forest lands. At the same time, community benefit began to be conceptualized as resulting from activities beyond traditional timber harvesting and milling, such as ecosystem management, forest and watershed restoration, outdoor recreation, and the harvest of nontimber forest products (Hibbard and Lurie 2013, Kruger et al. 2008). As the Forest Service adopted

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Figure 8-1—Coos Bay, Oregon, historically supported a diversity of logging and milling operations.

ecosystem management as its new management paradigm (Thomas 1996), it actively invested in job training and management projects with the goal of creating a new class of quality jobs in ecosystem management and restoration for displaced timber workers and communities affected by this transition in forest management (Spencer 1999). One effort to do so was the Jobs in the Woods Program, which began as part of the NWFP and included waivers of federal procurement law that allowed the Forest Service and BLM to set aside service contracts for ecosystem management to benefit contractors located in counties affected by the plan (Moseley 2005). Although this program was too small to offset the number of jobs lost in the timber industry, it did provide short-term employment for some displaced timber workers (Dillingham 2006). Moreover, its intent—to create jobs in local communities associated with restoration and ecosystem management—carried forward into subsequent agency programs (e.g., Secure Rural Schools Act projects, stewardship contracting, and community-focused National Fire Plan projects, described below).

Along with this shift toward ecosystem management, the 1990s gave rise to new understandings of community-forest relations that acknowledged the diverse contributions federal forests make to "community well-being." Studies recognized that well-being in forest communities included quality of life attributes beyond jobs and income, such as health, safety, educational attainment, political participation, social equity, empowerment, community cohesiveness, and access to social services (Beckley 1998, Doak and Kusel 1996, Harris et al. 2000). Studies also recognized that federal forests can contribute to community well-being in multiple ways, including both commodity (e.g., timber, grazing, minerals, nontimber forest products) and amenity (e.g., outdoor recreation, scenic beauty, clean air and water, open space, landscape) values they provide (Beckley 1998, Kusel 2001, Nadeau et al. 2003, Sturtevant and Donoghue 2008). 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.

In the past two decades, little congressional lawmaking has related to federal forest management. That which has occurred has tended to include some attention to local community social and economic needs. Laws that were designed to shore up payments to counties as timber harvest declined, first in the Plan area and then nationwide, are good examples. Timber-sale receipts comprised the vast majority of payments to county governments and dropped dramatically with the spotted-owl-related injunctions on timber harvesting in the early 1990s and subsequent implementation of the NWFP. Consequently, Congress passed a series of measures starting in 1991 to mitigate the lost revenues to counties using new formulas to calculate payments, the most recent of which was the Secure Rural Schools and Community Self-Determination Act of 2000 (Phillips 2006b). Although the Secure Rural Schools Act was initially set to expire in 2006, it has been reauthorized and extended several times, most recently on April 16, 2015, for 2 more years.[2] The Act was allowed to expire in 2017, prompting agencies to revert to making payments to counties from revenues generated by timber sales (25 percent for the Forest Service, 50 percent for the BLM) under the 1908 Payments to States Act. Congress continues to debate reauthorization; this is a subject of ongoing political debate and economic uncertainty in NWFP-area counties that relied heavily on these payments (Hoover 2015). In addition to payments to counties to backstop declining timber revenues, the Secure Rural Schools Act created local resource advisory committees to advise the Forest Service on priority ecosystem management and restoration projects that could be funded through Title II of the act. In addition, stewardship contracting, permanently authorized through legislation in 2014, has meeting local community needs as one of its central goals (P.L. 106-393; P.L. 106-291, Sec 323) (Kitzhaber 1998; Moseley and Charnley 2014). Similarly, for much of the 2000s, Congress provided appropriations language authorizing the Forest Service and BLM to consider local economic benefit when awarding restoration-related service contracts (e.g., PL 108-7, Sec 333). Although the exact language varies from

[2] http://www.fs.usda.gov/pts/.

law to law, typical beneficiaries include workers and businesses in forest communities, local communities, or isolated communities.

An area of significant rulemaking in the decades following NWFP adoption were efforts to revise the Forest Service planning rule, which elaborates how national forests should create long-term plans as required under the NFMA.[3] The planning rule had last been modified in 1982 under the Reagan Administration. Several subsequent revisions were attempted but never completed, so forest planning (either full plan revisions or plan amendments) continued to follow the 1982 planning rule (Schultz et al. 2013). From the beginning, the Obama Administration placed a strong emphasis on creating a new planning rule that could become successfully institutionalized, including provisions for significant public involvement and collaboration. The planning rule, as finalized in 2012,[4] requires assessment of numerous social values including social, cultural, and economic conditions and benefits that people obtain from forest plan areas and of recreation opportunities (FR 88 no 68. Sec. 219.6 (6)-Sec 291.6(13)); it directs plans to provide for social and economic sustainability (Sec. 219.8(b)). The planning rule also calls for multiple uses of national forests, including not only timber harvest but also aesthetic values; access to fishing, hunting, and gathering; and access to recreation and water supplies. Among many shifts in the planning rule from prior versions is the introduction of the concept of "ecosystem services," which is framed as the range of social, economic, and ecological benefits from national forests to be provided presently and into the future (Subpart A. Sec. 219.1).

Wildfire policy—

During the early years of the NWFP, the focus of forest management was centered around reconciling competing demands for timber production and threatened and endangered species conservation. However, subsequent to the adoption of the NWFP and the occurrence of several large, high-visibility wildfires in the region (Reilly et al. 2017), wildfire became the central focus of national forest management, eventually consuming over half of the agency budget by the mid-2010s (see chapter 3 for discussion of the wildfire issue). Wildfire policy and practice have also undergone dramatic transformation, although with only relatively little congressional involvement. With wildfire costs increasing from 16 percent of the Forest Service budget in the 1980s to more than 50 percent in 2015,[5] wildfire management now affects every corner of the agency by dramatically reducing funds available for other management activities.

Prior to the NWFP era, wildfire was rarely mentioned in law and policy (Nelson 1979), perhaps because wildfire occurrence nationwide was relatively low from the 1940s through the 1980s (Agee 1993). Nevertheless, wildfire management has deep roots in the founding and early management of the Forest Service (Pyne 1981), and there were decades of wildfire suppression capacity-building prior to the NWFP (Davis 2001). As noted above, the focus of wildfire policy has largely shifted from fire suppression to protect timber, to ensuring firefighter safety and protecting homes and other structures. Restoration for ecological objectives, including increasing the resilience of forests to fire and drought, has also become a forest management goal (chapter 3). The 2001 National Fire Plan increased the focus on community preparedness for wildfire, hazardous fuels reduction, ecosystem restoration, reintroduction of prescribed fire, and other management changes (Steelman and Burke 2007) (fig. 8-2). The Healthy Forest Restoration Act of 2003, among other things, created a community wildfire protection planning process that allowed national forests that had participated in community planning to use expedited planning processes for hazardous fuels reduction projects in the Community Wildfire Protection Plan (CWPP)-designated wildland-urban interface (WUI) (Vaughn and Cortner 2005). Increasingly, there are calls for managing wildfire more to meet the goals of reducing forest fuels and wildfire risk to communities and ecosystems (e.g., North et al. 2015), though it has been difficult to manage wildfire for resource benefits in practice in many landscapes (Calkin et al. 2015).

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[3] http://www.fs.usda.gov/Internet/FSE_DOCUMENTS/stelprdb5362536.pdf.

[4] http://www.fs.usda.gov/Internet/FSE_DOCUMENTS/stelprdb5362538.pdf.

[5] http://www.fs.fed.us/sites/default/files/2015-Fire-Budget-Report.pdf.

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Figure 8-2—In the 2000s, wildfire policy has shifted to focus on community wildfire protection and preparedness.

The Federal Land Assistance, Management, and Enhancement Act of 2009 (FLAME Act) sought to reduce the growing impacts of wildfire expenditures on the rest of the Forest Service budget. It also required the creation of the National Cohesive Wildland Fire Management Strategy, which increases the focus on creating resilient landscapes, fire-adapted communities, and safe and effective wildland fire response. From the National Fire Plan of 2001 to the Cohesive Strategy adopted a decade later, there have been significant policy efforts to change wildfire management, many of which have increased focus on community preparedness and protection in wildfire. Both the use of fire (prescribed or naturally ignited) and the use of silvicultural treatments to alter fuels conditions are complicated by ecological, economic, and social challenges that reflect decades of past land use patterns and policies (Carroll et al. 2007). Although much change has occurred, there has been a significant pattern of stasis as well, making clear that wildfire management is an increasingly complex social-ecological problem with few easy solutions (Carroll et al. 2007, Fischer et al. 2016). Nevertheless, it is a critical issue to address in the context of federal forest management and community socioeconomic well-being.

As wildfire law and policy have shifted to emphasize community preparedness, hazardous fuels reduction, and reintroduction of prescribed fire to create wildfire-resilient landscapes, a parallel paradigm shift has occurred in thinking about community-forest relations. Much of this thinking now revolves around the concept of "community resilience" (e.g., Daniel et al. 2007, Lynn et al. 2011, McGee 2011, Paveglio et al. 2009), which focuses on a community's ability to cope with and adapt to natural disturbances and change. Building on Folke (2006), Magis (2010), 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, while still retaining its basic function and structure. Federal land management policies that help promote community capacity to adapt to change may contribute to socioeconomic well-being (Anderson and Kerkvliet 2011).

The Impact of the Northwest Forest Plan on Rural Communities

From a social standpoint, the primary concern relating to socioeconomic well-being and federal forest management in Plan-area communities historically has been the impacts of reduced timber harvesting from federal lands on forest products workers, businesses, and timber-dependent communities in particular. In the Plan area, a steep harvest decline followed the 1990 listing of the northern spotted owl (*Strix occidentalis caurina*) as threatened under the Endangered Species Act (Charnley et al. 2008b) (fig. 8-3). In the 1980s, timber sales from Forest Service and BLM lands in the Plan area averaged 5.5 billion board feet annually (Charnley et al. 2008b). Intensive timber management on federal lands ended in the early 1990s owing to a series of lawsuits over the protection of the owl and associated species under the Endangered Species and National Forest Management Acts (Thomas et al. 2006), and related injunctions on federal timber sales within the range of the owl (Charnley 2006b). The social controversy engendered by the "owl wars," in which the interests of environmentalists concerned with the impacts of timber harvesting on old-growth forests and associated species were pitted against the interests of forest products workers and forest communities, is well documented (e.g., Carroll 1995, FEMAT 1993, Satterfield 2007). The NWFP was an attempt to balance these interests, and offer a solution that would provide "a sustainable level of human use of the forest resource while still meeting the need to maintain and restore the late-successional and old-growth forest ecosystem" (USDA and USDI 1994: 26–27).

Over the past two decades, a body of literature has emerged that assesses the impacts of the owl listing and NWFP on communities. This literature is composed of the results of NWFP socioeconomic monitoring (Charnley 2006a, Charnley et al. 2008a, 2008b; Grinspoon and Phillips 2011, Grinspoon et al. 2016) and a number of additional studies by economists and other social scientists. It is important to note that changes in the forest products industry in Plan-area communities and economies were not solely a result of declines in timber harvesting on federal forest lands. The most significant factors influencing the

Summary—

Numerous factors have influenced socioeconomic well-being in rural communities in the NWFP area; here we focus on the impacts of the NWFP. We begin by describing regional and national trends in the wood products industry to provide context for understanding Plan impacts. Regarding wood products production, market conditions facing the forest products industry are driven by overall consumer demand for wood products (e.g., lumber, paper, and engineered wood products), global competition, and technological change. Construction and remodeling account for the greatest demand for lumber and engineered wood products; therefore, changes in the housing market over the past 20 years have affected the forest products industry in the Plan area. Over and above changes in demand, industry restructuring and technological improvements have generally led to contractions in wood products manufacturing and a reduction in the number of workers required in the milling process. Nevertheless, demand fluctuations do influence employment levels in wood products manufacturing over short time periods, such as the increase in employment in wood products manufacturing that occurred when the overall economy improved post-2010, as the economic recession that began in December 2007 subsided.

Private forests currently contribute the vast majority of logs processed by mills in the Plan area. Greater timber harvest on federal forests would increase the number of logs available to mills and create additional work opportunities for logging contractors in the short term. If long-term mill output within the Plan area increased as

a result of higher federal harvest levels, these short-term changes in timber supply and harvesting contracts could extend for longer periods and could include additional work in processing facilities. However, log supply is not the sole determinant of the level of output from mills. Rather, demand for wood products in the United States and globally, mill production technology, currency exchange rates, and competition from other domestic and international wood product producers combine with other factors to influence levels of wood products production. As elsewhere in the West (and Nation as a whole), the wood products manufacturing sector in the Plan area has experienced mill closures and employee reductions. However, mills remaining in operation and those coming into production have greater production capacity and lower labor demands than those that closed. This trend results in the seemingly contradictory pattern of falling mill numbers and reductions in mill workers, but smaller declines (or even increases) in aggregate milling capacity, and increasing average mill capacity. Further, within the Plan area, mills are using more of that available capacity relative to mills elsewhere in the West, generally a sign of mill strength and demand for workers.

Within the Plan area, and especially in Oregon, much of the federal timber log supply comes from thinning harvests in plantations that are less than 80 years of age. Recent discussions about future federal forest management within the Plan area have proposed variable-retention harvests and ecological forestry within matrix lands to create more early seral vegetation through regeneration harvests, conserve older forests, and provide a more reliable flow of ecosystem services, including timber.

NWFP-related impacts on communities are associated primarily with cutbacks in federal timber harvesting, loss of federal agency jobs, reductions in federal contract spending, and the setting aside of reserve lands that exclude intensive timber production. Research examining the nature and extent of these impacts on communities has produced different findings. These differences may be attributed to the unit of analysis used to assess impacts (i.e., region, county, community); the period considered (first vs. second decade of the Plan); and the different datasets and indicators used to assess impacts. Most studies evaluate NWFP socioeconomic impacts using secondary indicator data pertaining to population change and economic variables such as employment, income, poverty levels, and property values, rather than primary data (data gathered at the community scale directly from community residents).

The findings of these studies can be generalized as follows:

1. Impacts attributed to the NWFP include population growth and decline, increases and decreases in socioeconomic well-being, and increases and decreases in economic indicators. Some studies found no NWFP impact on population and economic indicators.
2. NWFP impacts on communities differed at the community and county scales, and depended on local social, cultural, economic, and environmental contexts.
3. Impacts (both positive and negative) were greater during the first decade of the NWFP than they were during the second decade.
4. Impacts (both positive and negative) were greater in communities located close to national forests, or to reserved lands set aside by the NWFP, and in communities that had experienced a mill closure (not necessarily a result of the Plan).
5. Impacts were greater at the community scale than at the county and regional scales, and were greater in nonmetropolitan counties than they were in metropolitan counties.
6. Given the growing incidence of large and severe wildfires in the NWFP area, one important way in which federal forest management will affect rural communities moving forward relates to management for forest restoration and wildfire.

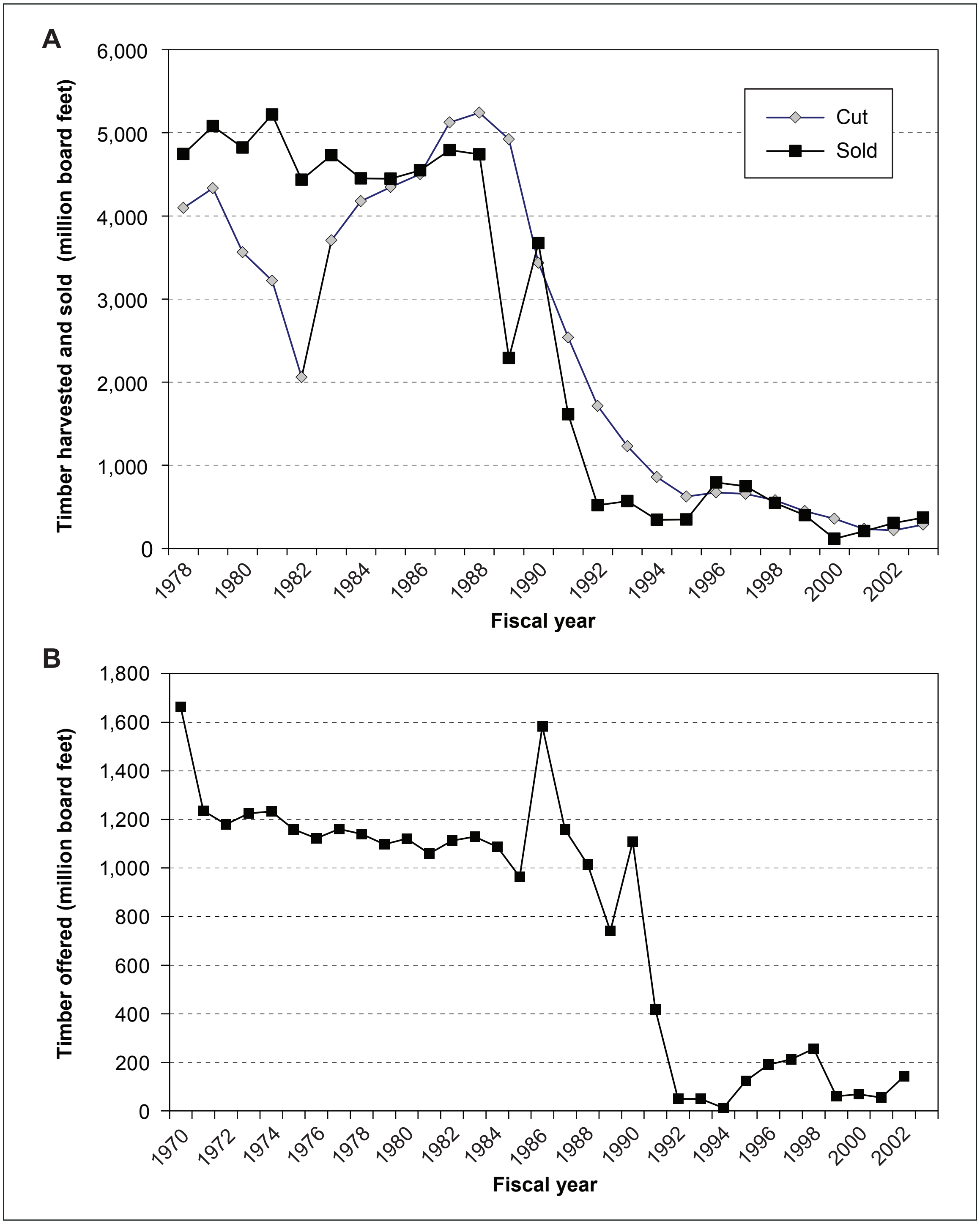

Figure 8-3—Volume of timber offered for sale, sold, or harvested from (A) Forest Service and (B) Bureau of Land Management units in the Northwest Forest Plan area, 1970s–2002. Source: Charnley 2006c.

industry are market conditions (e.g., demands for lumber and paper products), technological advances in wood processing, foreign and domestic competition, the cost of labor and manufacturing equipment, currency exchange rates, and timber availability (Keegan et al. 2006, Ince et al. 2011, Skog et al. 2012). Thus we begin this section by providing a broader picture of changes taking place in the wood products industry in the NWFP area and more broadly during the past three decades. We then focus on the role of federal forest management by discussing the impacts of the owl listing and the NWFP. We also briefly discuss the effects of wildfire management on local communities because wildfire on federal forests has become a salient factor affecting socioeconomic well-being there.

The wood products production market—
The primary wood products manufactured in Oregon, Washington, and northern California are dimensional lumber and plywood used in housing construction and remodeling. For the most part, the wood products produced within the NWFP area are commodity products, meaning they compete, in many cases, with products of the same quality produced from forests in different regions of the United States and around the world (Skog et al. 2012). Consumption of wood and paper products in the United States has risen in recent decades, but that consumption has been increasingly met through imports from other countries with lower costs of production (Skog et al. 2012). Further, wood products produced in the NWFP area must compete with nonwood products, such as concrete, steel, and composites that can be used in the same construction applications. These substitutes have been slowly taking market share from wood products over the past few decades because of consumer preferences, technological advances in materials, and cost (Ince et al. 2007). Although both heavy competition from other countries and substitute materials are anticipated, U.S. lumber production is still projected to increase through 2040, from a low point in 2010, under a variety of alternative future scenarios because of expanding domestic demand for wood products (Ince et al. 2011). The magnitude of the projected increase depends, however, on assumptions about the magnitude of increases in housing starts, gross domestic product (GDP) growth, and global demand for wood to use in energy production (Ince et al. 2011). Smaller increases in housing starts and GDP, and lower demand for wood for energy in foreign markets, yield lower levels of projected future U.S. lumber production.

Lumber production—
In the last decades of the 20th century, the Western United States was the Nation's "wood basket" and supplied the majority of softwood lumber produced nationally. That changed in the first decade of the 2000s, when the South became the predominant lumber-producing region. In 2010, lumber production in the Pacific Northwest states—the largest lumber producers in the Western United States—was at its lowest level since the 1950s (Keegan et al. 2011). The case of Oregon is illustrative. Since the mid-1950s, lumber production in Oregon has gone through cyclical ups and downs, but has generally declined over the long term (fig. 8-4) (Gale et al. 2012). The period since the early 1990s has been especially volatile, with dramatic swings influenced by changing timber availability and surges and collapses in the housing market.

The changing role of the Pacific Northwest in the nation's wood products industry reflects the combined effects of broad-scale changes that affect the industry across the United States and globally (i.e., changing demand for wood products, improved milling technology, foreign competition), and regional steep reductions in federal timber supply within the NWFP area. Despite this downturn, the wood products industry remains an important contributor to the economies of Oregon, Washington, and California, although not to the degree that it was in the past. For example, although wood products manufacturing in Oregon slipped from about 8 percent of the state's gross domestic product in the late 1980s to about 1 percent in 2009 (Lehner 2012), in many rural communities it remains an important source of jobs and income. Overall, the economies of the three states have diversified and expanded into other sectors, but this diversification has not necessarily occurred in some local communities.

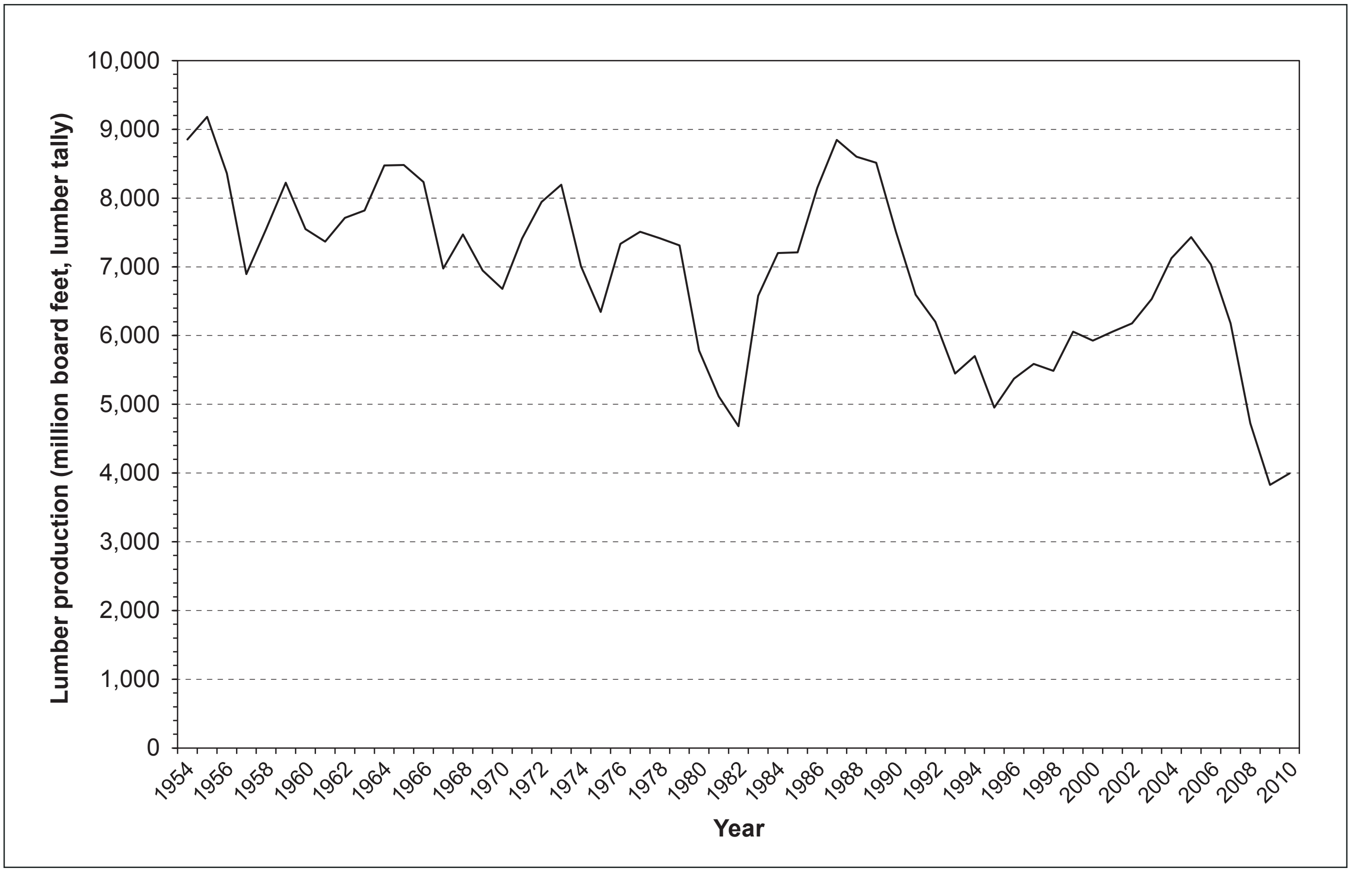

Figure 8-4—Oregon lumber production, 1954 to 2009. Source: Gale et al. 2012.

The role of timber supply—

In California, Oregon, and Washington, since the early 1990s, private (especially private industry lands) and state-owned forests have provided the majority of timber to wood processing facilities (Oswalt et al. 2014). Similarly, in the NWFP area, the majority of timber harvested has come from nonfederal lands (fig. 8-5). Increases in log supply from public or private lands can increase the employment at mills when there is unutilized mill capacity, a healthy market for wood products, and sufficient volume of new logs to warrant adding an additional shift at the mill, or opening another processing line. For example, a sawmill with unutilized capacity in John Day, Oregon, recently increased mill employment over the short term when Forest Service harvest volumes were increased (Bennett et al. 2015). Aside from the amount of federal timber supplied, mill employment remains influenced by market conditions for lumber and other wood products, and changes in milling technology that reduce the amount of necessary labor. Cyclical ups and downs in mill employment (e.g., Lehner 2012) for lumber production follow changing conditions in the economy and markets for housing construction, regardless of federal timber supply conditions (Keegan et al. 2011). Even when timber supply changes are happening, mill employment remains influenced by technological improvements to mill operations. For instance, Helvoigt and Adams (2009) found that 38 percent of the decline in employment at sawmills between 1988 and 1994 (when federal timber harvests declined precipitously) can be attributed to technological change that reduced labor requirements.

Increases in federal timber supply may lead to expansion in lumber production and hiring of mill employees if timber supply is constrained, demand for lumber products is strong, and mill capacity is underutilized. Within the Pacific Northwest, these mill conditions are thought to

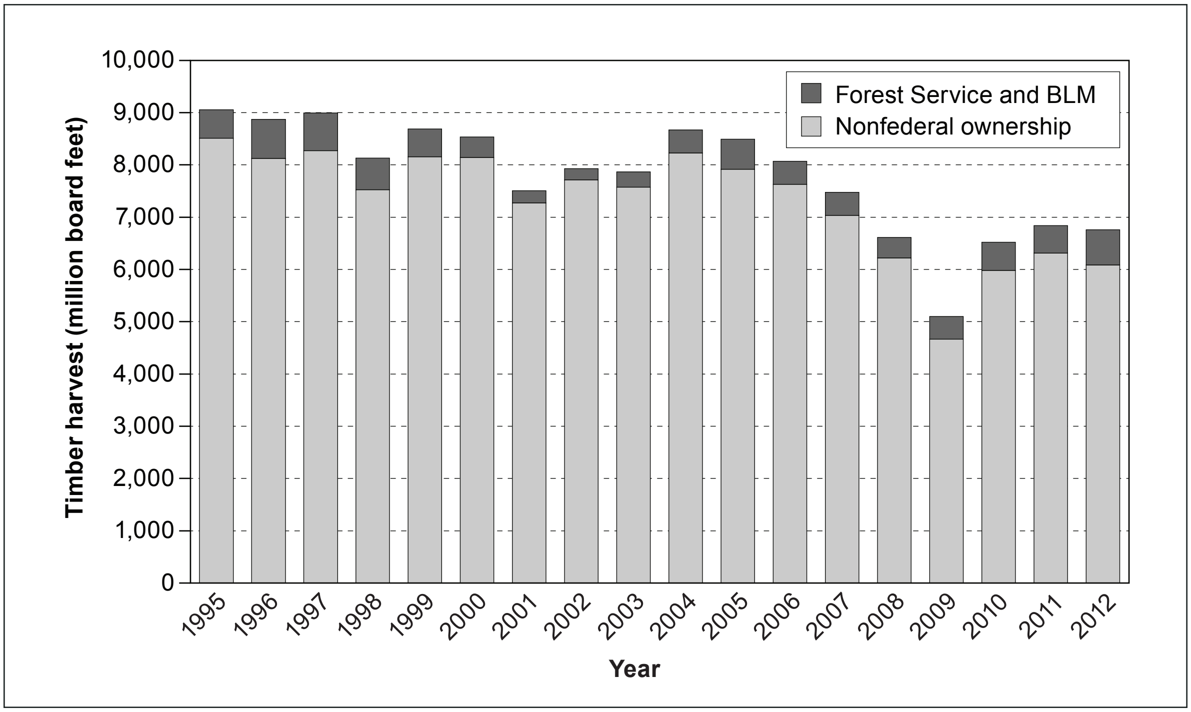

Figure 8-5—Since the 1990s, the majority of timber harvested in the Northwest Forest Plan area has come from nonfederal lands. Source: Grinspoon et al. 2016.

be more commonly found east of the Cascade Range, where productive forests are usually owned by the federal government, and severe losses in milling capacity (Swan 2012) have led to very limited processing infrastructure. In general, economic models have found that significant increases in federal harvest levels benefit wood products manufacturers because more timber is available at lower prices, but pose a disadvantage to private forest owners because the price of stumpage falls, forcing them to sell for less (e.g., Abt and Prestemon 2006, Adams and Latta 2005, Adams et al. 1996, Ince et al. 2011). Stumpage prices may rebound over time if private landowners reduce harvest levels in response to lower stumpage prices. Increased federal timber harvest might improve the well-being of local wood products producers and private forest landowners in situations in which all local milling capacity is in danger of closing, and the addition of federal timber supply helps to keep mills above the tipping point of having to close operations (e.g., Adams and Latta 2005); or where supply increases last for a long time (e.g., Abt and Prestemon. 2006). The potential increased timber supply from "ecological forestry," including variable-retention harvesting[6] (e.g., Franklin and Johnson 2012) in plantations, may well promote improved community well-being if the early seral vegetation created supported long-term timber production, especially in areas with a higher share of dry forest, and in communities that have, or can recreate, a forest products workforce. However, the wood products sector within the NWFP area would remain subject to market conditions and competition from other wood products manufacturers nationally and globally.

Because of the relatively high transport cost, species preference of mills, and supply from private forests, the majority of the wood processed in the NWFP area comes

[6] Franklin and Johnson (2012) identified the key elements of ecological forestry as (1) retaining structural and compositional elements of the preharvest stand during regeneration harvests, (2) using natural stand development principles and processes in manipulating established stands to restore or maintain desired structure and compositions, (3) using return intervals for silvicultural activities consistent with recovery of desired structures and processes, and (4) planning management activities at landscape scales. Variable-retention harvesting is clearcut harvesting that retains a portion (e.g., 10 to 15 percent) of the original forest in undisturbed patches or aggregates distributed across the harvest unit.

from within the region. Historically, there has been relatively little procurement of federal timber from outside the NWFP area by local mills. Under the Forest Resources Conservation and Shortage Relief Act of 1990 (as amended), federal timber in the NWFP area is barred from international export, and, in most cases, purchase by an entity that sells timber into the export market. With that export restriction, federal timber can be a source of wood supply for businesses that have difficulty purchasing logs when there are high prices in the log export market. Additionally, providing a consistent flow of federal timber could offer some certainty to wood processors that some wood volume would be accessible to domestic purchasers in the face of a strong log export market.

Following adoption of the NWFP, the limited social acceptability of harvesting large-diameter and old-growth trees from matrix land allocations on federal lands and of clearcutting (Charnley and Donoghue 2006a), has largely confined harvests west of the Cascades to existing plantations within matrix lands that have younger, smaller trees. Timber harvest prescriptions in these cases often apply commercial variable-density thinning (see chapter 3) to stands younger than 80 years. The focus on harvesting trees under 80 years old in the matrix is counter to the calculation of probable sale quantity (PSQ)[7] in the NWFP (Charnley 2006b), which relied substantially on volume produced from stands over 80 years of age within the timber-suitable base of matrix lands (Johnson 1994, Johnson et al. 1993). One modeling study undertaken in a large landscape in the Coast Range of Oregon estimated that continuing current federal forest management practices that were focused on thinning smaller, young trees in plantations under 80 years of age would ultimately result in a 71-percent decline in federal harvest levels by 2050 (Johnson et al. 2007). The reason for the decline was reduced availability of small- and medium-diameter stands on federal forest lands because thinning did not establish new young stands, and the existing plantations aged beyond 80 years.

[7] Probable sale quantity is an estimate of average annual timber sale levels likely to be achieved over a decade; it is a decadal average. The NWFP identified matrix lands and adaptive management areas as being suitable for producing a predictable and sustainable timber supply, thus only timber produced from these locations counts toward PSQ volume (Charnley 2006c).

Potential future declines in harvest volumes from federal forests would further reduce the contribution of federal timber supply to the traditional forest and wood products sectors of local economies within the NWFP area. As a consequence, the forest and wood products sectors would become more reliant on the supply of timber from private and state-owned forests. Increased use of ecological forestry (Franklin and Johnson 2012) to create early seral vegetation (Swanson et al. 2011) that has been reduced by fire exclusion (chapter 3) and other practices in moist and dry forests could be a way to maintain some level of timber harvest from plantations and other younger forests over the longer run. Challenges to expanded use of ecological forestry and regeneration harvests in the NWFP area include (1) lack of public trust of federal agencies, (2) the scale of restoration needed in dry forests, and (3) the legal and social obstacles to implementing regeneration harvests in moist forests (Franklin and Johnson 2012). In addition, it could be difficult to plan and schedule timber production from early-seral vegetation projects when landscape goals for these conditions can also be met by wildfire, which is unpredictable.

Trends in the number of wood-processing facilities— Reductions in demand for wood products, technology, and reduced log supply from federal forests during the 1980s and 1990s have led to declines in wood-processing infrastructure throughout the United States. Consistent with national trends, over the long term and under varying levels of federal timber supply, the number of operating timber mills and employees in the wood products sector has declined in Oregon, Washington, and California (Gale et al. 2012, Keegan et al. 2011, McIver et al. 2015); the case of Oregon is illustrative (figure 8-6). For example, Oregon had 405 lumber mills in 1980, 282 of which closed over the next three decades for a reduction of two-thirds (Chen and Weber 2012). Similarly, in 1980, 113 rural communities in Oregon had mills (roughly half of them), and by 2007 only 58 communities had mills. Direct job loss per mill closure averaged 100 jobs, a large impact on rural communities whose median population was 2,000 people or fewer (Chen and Weber 2012). It is unknown how many mills in the Pacific Northwest closed specifically because of the NWFP. A variety of factors (e.g., technological change, industry

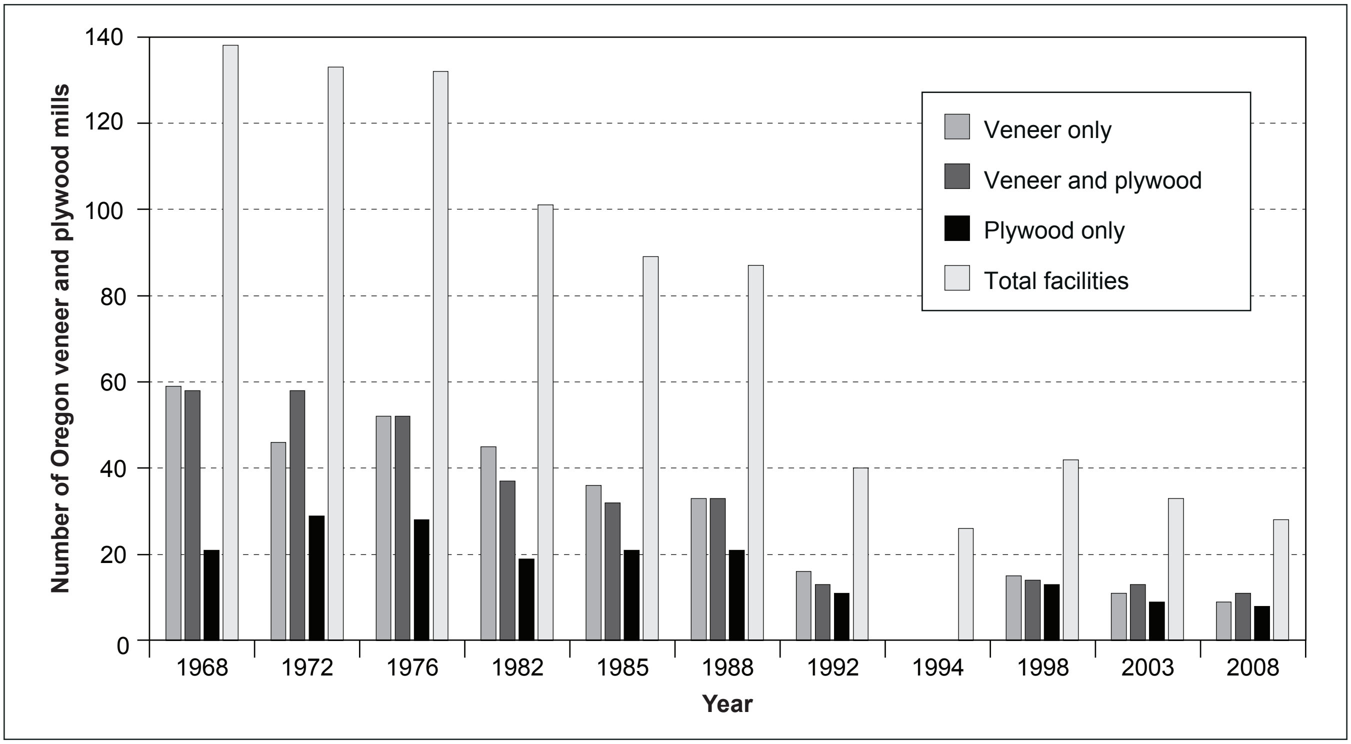

Figure 8-6—Number of veneer and plywood mills in Oregon, 1968–2008. Note that, for 1994, veneer and plywood mills were not counted separately. Adapted from Gale et al. 2012.

restructuring, and competition) have combined to precipitate mill closures in the region. For example, Helvoigt and Adams (2009) found that 38 percent of jobs lost in sawmills in Oregon and Washington between 1988 and 1994 were related to technology improvements in log processing. The remaining jobs losses were due to a variety of factors, including changes in log supply.

More recently, between 2000 and 2003, an estimated 142 wood products plants closed in the United States (Quesada and Gazo 2006). During that time, 20 plants closed in Oregon (the second most in the nation), 13 closed in Washington, and 5 closed in California (Quesada and Gazo 2006). Plant closures (when a cause could be determined) were most commonly attributed to general financial difficulty and reorganization; only 5 of 94 cases cited material shortages as a reason for plant closure (Quesada and Gazo 2006). Between 2005 and 2009, an additional 300 mills temporarily or permanently closed in the Western United States in response to the steep decline in demand for lumber in the housing sector, and competition from other mills (Keegan et al. 2011). The national pattern of mill closures in the 2000s was mirrored in Oregon, Washington, and California (McIver et al. 2015, WDNR 2014).

Mill capacity—

The capacity of operating mills (mill capacity) can be a better indicator of the size of the wood products industry and the potential use of, and demand for, timber harvested from public and private forest lands than the number of mills (Keegan et al. 2011). Because of technological improvements and loss of small mills, the number of mills and mill employees may decline while total aggregate mill capacity across states or regions declines more slowly, remains steady, or even increases. For example, although the number of sawmills in Washington declined from more than 200 in 1968 to 75 in 2002, aggregate mill capacity in the state increased during the period as mills adopted new technology and became larger (Helvoigt and Adams 2009). The average capacity of the mills in operation in 2002 in Washington was three times what it was in 1968 (Helvoigt and Adams 2009).

Historically and currently, the Pacific Coast states (Washington, Oregon, California, and Alaska) have accounted for the majority of the West's milling capacity (Keegan et al. 2006). The change in mill capacity across the West sets the context for considering changes in mill capacity within the NWFP area. Between the late 1980s and 2010, mill capacity in the Western United States declined from about 25 billion board feet to 13 billion board feet—a nearly 50-percent decline (Keegan et al. 2011). Mill capacity losses in the NWFP area during that time reflected, in part, conditions facing the industry elsewhere in the West. Between 1986 and 2003, the Pacific Coast states lost 35 percent of their mill capacity, but this decline was the smallest percentage decline in the West during that period. Post-2005, and influenced in large part by the Great Recession, milling capacity in the Pacific Coast states dropped another 10 percent to a little under 11 billion board feet by 2010. Although that loss was significant, the Pacific Coast region again had smaller percentage declines in mill capacity than elsewhere in the West during that period (Keegan et al. 2011). Within the Pacific Coast states, Oregon and Washington have typically fared better than California and Alaska in rates of change in the industry. For example, in Oregon, mill capacity in 2010 was roughly the same as it was in 1996 (Gale et al. 2012); and in Washington, aggregate milling capacity in 2002 was slightly greater than it was in 1968 (Helvoigt and Adams 2009).

The percentage of mill capacity in use gives an indication of how much additional timber could be processed in the short term with minimal infrastructure investment. Capacity utilization in the Western United States from the 1980s through 2005 (just prior to the Great Recession) remained steady at about 70 to 80 percent (Keegan et al. 2011). In the early 2000s, with high demand for lumber during the housing peak, capacity utilization in the Western United States peaked at a little over 80 percent before subsequently falling to about 56 percent at the height of the recession of the late 2000s (Keegan et al. 2011). After the Great Recession, in 2012, Oregon was utilizing 57 percent of its overall timber processing capacity and 61 percent of its sawmill capacity (Gale et al. 2012); California was using 72 percent of its sawmill capacity (McIver et al. 2015).

Employment in the wood products industry— The U.S. wood products manufacturing sectors have experienced consistent, long-term contraction in employment since the early to mid-1990s (Keegan et al. 2011, Quesada and Gazo 2006, Woodall et al. 2012). Employment in wood products manufacturing in the Pacific Northwest mirrors that pattern. For example, in Oregon, employment in wood products manufacturing has been in a general decline since the late 1970s (Lehner 2012). At various times during that period, contraction in employment has resulted from changes in the demand for lumber and paper products, plant closures, technological advances in manufacturing that led to lower labor requirements, closing of product lines, and consolidation of companies. Demand for softwood lumber closely tracks conditions in the U.S. housing market. Steep declines in demand for new housing and housing remodels in the late 2000s that occurred in association with the Great Recession led to sharp reductions in lumber production, to levels not seen since World War II (Woodall et al. 2012). As result of that decline, the U.S. wood products sector lost nearly 209,000 jobs between 2005 and 2009. This pattern mirrored that seen in other manufacturing sectors, such as the automotive industry, during the same time frame (Woodall et al. 2012).

In the Western United States specifically, employment in the wood products industries dropped by about 50,000, to about 250,000, between 2000 and 2010 (Keegan et al. 2011). Oregon and Washington each experienced wood products manufacturing employment in the 2000s that was below employment levels of the late 1990s (Eastin et al. 2007, Lehner 2012). Subsequent to 2010, there has been a recovery in this sector in Oregon, in line with an overall economic recovery (Rooney 2015). In California, employment remained flat through 2012. Comparable reporting is not available for Washington. Employment in the wood products sector in Oregon is cyclical over the long term, and often tracks in a pattern similar to overall nonfarm employment (although the swings in wood products employment are generally of higher magnitude) (Lehner 2012). Regardless, wood products manufacturing now requires fewer employees than in earlier decades (see Grinspoon et al. 2016), but recovery in recent years has been good relative to employment levels in the 1990s and early 2000s.

It is challenging to predict the complex interactive outcomes of changes in timber production, wood products markets, technologies, and other factors relevant to future timber economies as they interact with global climate trends. However, various climate change scenarios anticipate steady or increasing flows of forest products production worldwide (Alig 2010, Irland et al. 2001, Kirilenko and Sedjo 2007, Latta et al. 2010). Such outcomes could benefit those communities that contain infrastructure for harvesting and processing timber, though effects on wood products prices will influence the distribution of benefits (Alig 2010, Joyce 2007). Within the NWFP area specifically, gains in productivity may be offset by increased incidence of fire, disease, and insect outbreaks, especially in drier forest types within the region (Klopfenstein et al. 2009) and in areas that become more susceptible to other pathogens (Kliejunas et al. 2009).

Effects of the Northwest Forest Plan on timber production and timber industry jobs—
As noted at the start of this section, economic concerns over the impacts of the NWFP on forest communities in the Plan area stemmed mainly from cutbacks in federal timber harvesting. During the 1980s, the allowable sale quantity (ASQ) of timber from federal forests in the Plan area averaged 4.5 billion board feet (BBF) annually (Charnley 2006c). Under the Plan, the PSQ varied during the first decade but averaged 776 million board feet (MMBF) annually between 1995 and 2003. The total volume of timber offered for sale from Forest Service and BLM lands in the Plan area averaged 526 MMBF annually between 1995 and 2003. Of this volume, an estimated 80 percent was from adaptive management areas and matrix lands, and 20 percent from reserve lands. Under the NWFP, only timber offered for sale from adaptive management areas and matrix lands counts toward PSQ, meaning that an annual average of 421 MMBF of PSQ volume was offered for sale between 1995 and 2003 (Charnley 2006c). Reflecting this shift, the total contribution of federal timber to the regional supply dropped from roughly 25 percent in 1990 to under 5 percent in 2000 (Phillips 2006a). By 2003, the expected PSQ volume from federal forests in the Plan area was 805

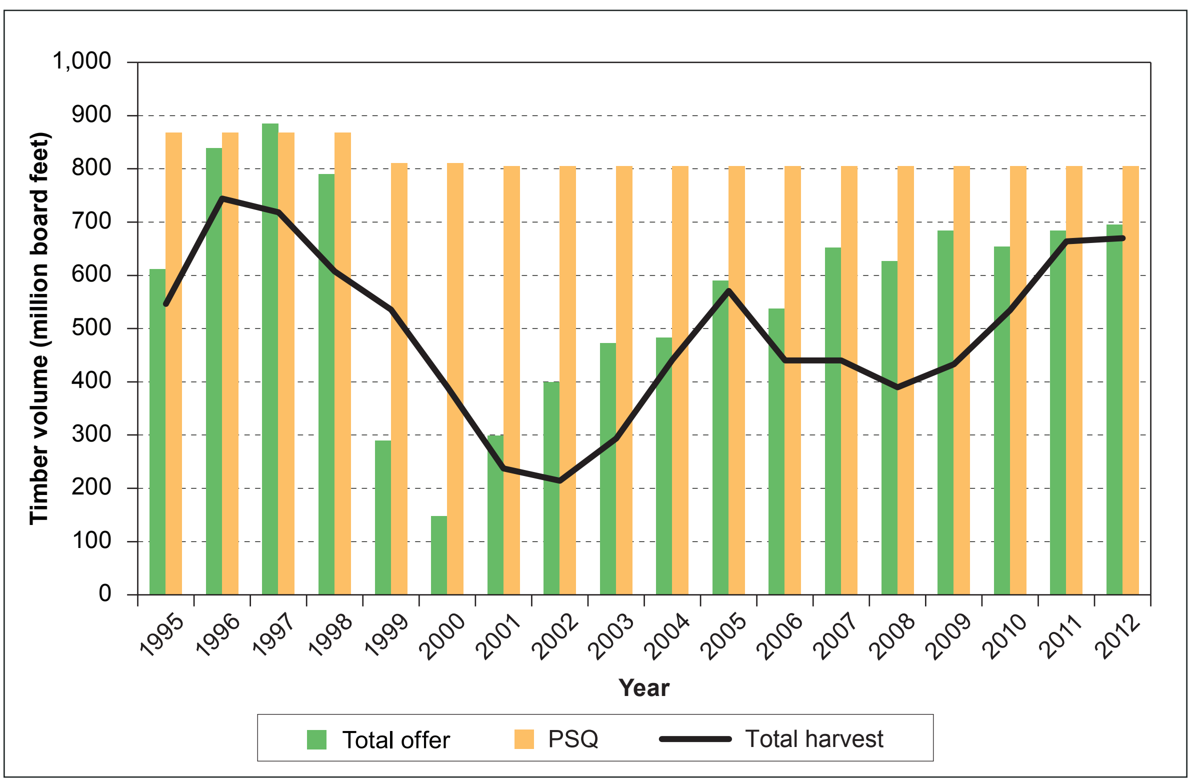

Figure 8-7—Timber offered for sale and harvested from federal forests in the NWFP area, in relation to the probable sale quantity (PSQ), 1995-2012. Source: Grinspoon et al. 2016.

MMBF. During the second decade of the Plan, the volume of timber offered for sale from Plan-area federal forests increased gradually and became more stable and predictable, but remained below the PSQ (fig. 8-7) (Grinspoon et al. 2016). By 2012, federal timber accounted for about 10 percent of the regional timber supply from all land ownerships (Grinspoon et al. 2016).

Regarding employment, jobs in primary wood products manufacturing declined in the NWFP area by 30,000, or 26 percent, between 1990 and 2000, and stood at roughly 85,000 in 2000 (Phillips 2006a). The bulk of the 30,000 job losses (all but 400 of them) occurred between 1990 and 1994, after injunctions on federal timber harvesting were put into place following the owl listing in 1990. An estimated 39 percent of these jobs were lost as a result of cutbacks in federal timber harvesting; the majority of the job loss (the remaining 61 percent) is attributable to technological changes in the industry (Phillips 2006a). In 2001, there were over 100,000 jobs in the NWFP area in the timber sector/forest products industries (logging, primary and secondary wood processing) associated with production from all forest ownerships; by 2012, there were 65,000, a drop of about 40 percent (Grinspoon et al. 2016). In 2001, 12 percent of the jobs in nonmetropolitan counties in the NWFP area were in the timber sector, and by 2012 only 3 percent were in the timber sector (Grinspoon et al. 2016). During this same period, the volume of federal timber sales within the NWFP area increased from about 150 MMBF in 2000, to about 650 MMBF in 2012, meaning that despite the overall job decline the number of industry jobs associated with timber harvesting from Forest Service and BLM lands increased (Grinspoon et al. 2016). In 2012, timber harvested from federal forests in the Plan area supported an estimated 2,300 direct jobs, and 2,500 indirect and induced jobs in the 72 NWFP-area counties (Grinspoon et al. 2016). Total employment in nonmetropolitan counties of the Plan area increased between 2001 and 2012, more than offsetting job losses in the wood products industries. Nevertheless, if people do not have the skills to take advantage of new job opportunities, they may still suffer unemployment.

Adding to the economic effects of changing timber harvest levels on employment in the private sector, additional economic losses resulted from the contraction of public sector agency jobs: the five BLM units in the NWFP area lost 13 percent of their full-time-equivalent positions between 1993 and 2002 (166 jobs), and 15 of the 17 national forests in the NWFP area (excluding the Lassen and Modoc) together lost 36 percent of their full-time-equivalent positions (3,066 jobs). These trends continued during the second decade of the NWFP, especially on Plan-area national forests in Oregon and Washington, which had about 5,700 full-time-equivalent employees in 1993, and 2,300 in 2012 (Grinspoon et al. 2016). Forest Service job loss during the first decade of the plan was associated with declining budgets. Despite growth in Forest Service and BLM budgets at the national scale during the decade (owing largely to increased appropriations for fire and fuel management), national forest budgets for the Plan area as a whole dropped 35 percent, even with increased allocations for fire and fuel management (Stuart 2006). Budget declines were tied to reduced timber harvest levels (Charnley et al. 2008b). BLM job loss was associated with reduced timber sales, but not with reduced budgets; BLM unit budgets rose overall during the first decade of the NWFP, mainly because of stable O&C funding appropriations and additional budget allocations for NWFP-related programs such as Jobs in the Woods and Survey and Manage (Charnley et al. 2008b, Stuart 2006).

Another way in which federal agencies create local community benefit is through procurement contracting, which can provide jobs for local businesses. Although BLM procurement contract spending remained constant during the first decade following NWFP implementation, Forest Service procurement contract spending declined from $103 million in 1991 to $33 million in 2002, meaning that the agency supported substantially fewer external jobs through contracts for services such as road maintenance, forest management, and professional services (Charnley et al. 2008b). Trends in Plan-area procurement contract spending were not analyzed during the second decade of the Plan.

Mitigation measures designed to offset the negative economic impacts of the NWFP included the Jobs in the Woods Program, the Northwest Economic Adjustment Initiative (NEAI), and changes in federal payments-to-counties formulas so that these payments were not tied to subsequent annual timber revenues from federal forest lands. Community economic assistance provided through the NEAI was generally viewed as having some successes, but as being "too little, too late" overall (Dillingham 2006). Although changes in legislation related to payments to counties have been successful in mitigating the effects of declining timber receipts (Graham 2008, Phillips 2006b), ongoing uncertainty associated with Secure Rural Schools Act reauthorization makes the future uncertain.

Impacts of job loss on wood products workers—
Job loss can have severe impacts on affected workers. Employees who lose their jobs in wood products manufacturing face the challenge of finding work in other sectors of the economy, either where they currently live or elsewhere, including perhaps in other states. Helvoigt et al. (2003) examined Oregon employment records to study employment transitions of those displaced from the wood products market in the early 1990s. In Oregon, about 51 percent of wood products sector employees who lost their jobs during industry downturns in the early 1990s found employment by 1998 in other industries within the state, primarily in the service sector, retail trade, manufacturing, and construction (Helvoigt et al. 2003). The remainder of those who lost their jobs either stayed unemployed, left the state, or became self-employed. Those who were able to find employment in another sector within Oregon had median annual wages that were about 1 percent lower than their former wages. However, that small change in median wage was buoyed by the high incomes of those former wood products manufacturing employees who found new jobs in the technology sectors. Many workers who lost their jobs were working in relatively low-paying service-sector jobs by 1998. Aside from changes in wages, there may have been additional losses in benefits coverage not reported in these figures. In southern and eastern Oregon, about one-third of those who lost their mill jobs moved elsewhere in the state for work (Helvoigt et al. 2003).

The impacts of job loss on wood products workers were not purely economic; they were also social. Existing literature finds that mill workers were concerned about economic stability, and have a strong attachment to their home communities (Lee et al. 1991). This finding implies that moving for a new job elsewhere would have strong social impacts. Loggers' sense of identity was closely tied to their occupation, which fostered independence, pride in their work, and the feeling of having a unique job (Carroll et al. 2005). They were also part of an "occupational community" that included other loggers, social interactions with whom strengthened their sense of identity (Carroll et al. 2005). This attachment to a logging way of life meant that many loggers were willing to move or migrate seasonally in order to pursue it (Carroll et al. 2000b). Thus, not only did job loss represent a loss of jobs and income; it also undermined loggers' sense of identity and personal empowerment, which were tied to working in the woods, making finding a substitute occupation difficult. Moreover, loggers and the timber industry were often vilified during the years of the so-called "owl wars," leading to occupational stigmatization, which had a negative social and psychological impact on loggers and their families (Carroll 1995, Carroll et al. 1999). A study of job loss among company loggers in Idaho (Carroll et al. 2000a) found that many loggers chose to stay in logging if they could, even if it meant lower wages and fewer benefits than they had previously enjoyed. Reasons included the relatively high income from logging, attachment to their local community and region, desire to maintain a rural way of life, and sense of identity tied to logging.

Northwest Forest Plan impacts on communities and counties—
The impacts of reduced federal timber harvesting following the spotted owl listing and the NWFP on jobs, wood products workers, and communities in the NWFP area have been debated since the 1990s (e.g., Carroll et al. 1999, Freudenburg et al. 1998). Often, different findings emerge depending on the unit of analysis used to assess impacts (region, county, census tract, definition of community, individual or household), time considered, and datasets and indicators used to assess impacts. Thus, studies on the socioeconomic impacts of the NWFP on communities and counties find mixed results. Most studies evaluate NWFP socioeconomic impacts using secondary indicator data, rather than primary data gathered at the community scale from community residents.

The NWFP caused some 11.5 million ac (4.65 million ha) of federal land to be reallocated from commodity production to ecosystem management and conservation status (Chen et al. 2016, Eichman et al. 2010). A number of studies have looked at the effects of federal lands conservation policies and protected areas generally on local counties and communities in the Western United States. Some have found these policies to undermine the local economic base associated with natural resource production, causing job loss, lower wages, and outmigration (e.g., Duffy-Deno 1998). Others have found that they can be good for communities because they may increase amenity migration and associated amenity-driven economic development (Holmes and Hecox 2004, Lorah and Southwick 2003, Power 2006, Rasker et al. 2013). And some analyses find no significant impacts on employment or wages from proximity to public lands that are protected from, or experience reduced levels of, resource extraction (Duffy-Deno 1997; Lewis et al. 2002, 2003; Pugliese et al. 2015; Rasker 2006). Eichman et al. (2010) pointed out that because the impacts of conservation policies can be both negative and positive, one must analyze their aggregate effects, including how the positive impacts mitigate the negative ones, to fully understand their effects.

Community-scale research conducted as part of NWFP socioeconomic monitoring during the first decade of the NWFP used a community socioeconomic well-being index derived from six U.S. Census variables[8] to evaluate change in 1,314 nonmetropolitan communities in the Plan area (Donoghue and Sutton 2006). Socioeconomic well-being was evaluated based on index scores that ranged from 0 to 100. The index was used to examine change in well-being for a number of parameters; those reported here are (a) number of communities regionwide whose socioeconomic well-being scores increased, decreased, or remained the same between 1990 and 2000; (b) change in socioeconomic well-being scores between 1990 and 2000 in communities based on their proximity to federal forest lands (<5 miles versus ≥5 miles away); and (c) number of communities having very low (0 to 48.72), low (48.73 to 61.07), medium (61.08 to 73.36), high (73.37 to 85.58), or very high (85.59 to 100) socioeconomic well-being scores in relation to proximity to federal forests. Donoghue and Sutton (2006) also looked at variation in the individual indicators comprising the socioeconomic well-being index between 1990 and 2000, and between communities within and greater than 5 miles of a federal forest, also reported here. The authors compared change in socioeconomic well-being in NWFP-area communities within 5 miles of a federal forest, with those 5 miles or more away, because they inferred that communities near federal forests have distinct connections to those forests that differ from those farther away.

The study found that, regionwide, 27 percent of NWFP-area communities experienced little change in socioeconomic well-being between 1990 and 2000 (scores in 2000 were within +3 to -3 points of the 1990 scores); 37 percent experienced a decrease in well-being (ranging from -51 to < -3 points), and 36 percent experienced an increase in well-being (ranging from >3 to 44 points) (Donoghue and Sutton 2006). When comparing means between 1990 and 2000 for each of the six indicators comprising the socioeconomic well-being index, they found that change in the means of five of these indicators were statistically significant at a regional scale ($p < 0.001$). At a regional scale, the percentage of the population in communities with a bachelor's degree or higher went up, the percentage of the population in poverty went down, employment diversity increased slightly, income inequality increased, and average commute time to work also increased during the decade. Change in unemployment between 1990 and 2000 at the regional scale was not statistically significant (Donoghue and Sutton 2006).

Among communities within 5 miles of a federal forest, 40 percent had socioeconomic well-being scores that decreased during the decade, compared with a 33 percent decrease in scores among communities 5 miles or farther from a federal forest. Moreover, most of the communities with very low or low socioeconomic well-being scores in 2000 (71 percent) were within 5 miles of a federal forest. However, 43 percent of the communities with high or very high socioeconomic well-being scores in 2000 were also within 5 miles. Thus, although some communities close to federal forest lands were doing well in 2000, in general, communities farther away had higher socioeconomic well-being scores. When disaggregating the index indicators and comparing their means for 1990 and 2000,

[8] The variables were diversity of employment by industry, percentage of population 25 years and older having a bachelor's degree or higher, percentage of the population unemployed, percentage of persons living below the poverty level, household income inequality, and average travel time to work.

Donoghue and Sutton (2006) found that, on average, communities farther from federal forests had a greater percentage of the population with a bachelor's degrees or higher, less poverty, less unemployment, and less income inequality during both time periods, and a higher diversity of employment by industry in 1990 (but not 2000). Communities farther away also had higher commute times, but there was a positive correlation between average travel time to work and median household income. There were no statistically significant correlations between community socioeconomic well-being scores and community population size or population change (Donoghue and Sutton 2006).

Another study examined how 2000 poverty and unemployment rates (indicators of community well-being) traced to prior high rates of timber industry employment, the share of minority populations, and other characteristics of communities on the Olympic Peninsula in the context of the establishment of the NWFP (Kirschner 2010). The study used panel regression with U.S. Census data from 1990 and 2000, and the census tract as the unit of analysis (which is larger than a community but smaller than a county). In the study region, the poverty rate in 1990, a high minority population in 2000 (primarily American Indians and Latinos), and the share of the population with college degrees were significant predictors of the poverty rate in 2000. The poverty rate in 1990 was believed to reflect the lingering impacts of timber industry restructuring that occurred in the 1980s. The presence of minorities was the only variable tested that was a statistically significant predictor of the unemployment rate in 2000. These findings likely reflect a history of prejudice and discrimination toward, and disadvantage among, these populations, influencing community socioeconomic well-being (Kirschner 2010). The level of reliance on the timber industry as a local employer (used as a proxy for the potential magnitude of the effect of the NWFP) was not found to be a statistically significant predictor of poverty or unemployment in 2000 on the Olympic Peninsula.

Eichman et al. (2010) studied the effects of the NWFP on employment growth rates and net migration rates during the first decade of the NWFP at the county scale for 73 counties that either contain NWFP reserved land (late-successional reserves, riparian reserves), or are adjacent to such counties. They were interested in how the economic effects of net migration might offset those associated with reduced timber production from the reserved lands. They found that in counties having land reserved by the NWFP, there was a negative effect on annual employment growth rates, reducing them by 0.2 percent for every 1 percent of land in a county that was reserved. Thus the presence of reserved lands (12 percent on average across the 73 counties studied) decreased the average annual employment growth rate from 1.75 to 1.52 percent The percentage of decline in annual employment growth was higher in nonmetropolitan counties than in metropolitan counties. This study also found that the NWFP had a slightly positive effect on net migration to the 73 counties, which the authors attribute to the natural features associated with reserved land that attract amenity migrants (e.g., retirees, telecommuters) or help retain residents. However, the positive economic effects of migration only slightly offset the negative impacts of reduced timber harvesting on employment growth rates (-0.019 [total effect] versus -0.021 [without net migration offset]).

Chen and Weber (2012) examined the impact of the NWFP on 234 rural communities (incorporated cities having less than 50,000 people) in Oregon whose economies were based in the wood products industry before NWFP implementation. The authors found complex relationships between community population change and wealth growth (measured by residential and commercial real estate value), mill closures, and proximity to NWFP-reserved land in the decades around establishment of the NWFP. They found that, during the 1990s, proximity to NWFP reserved land (i.e., within 10 miles of reserved land) had a statistically significant positive effect on community population growth and wealth growth compared to communities located farther away. They attributed this finding to positive amenity-related growth effects of the Plan on communities. This positive effect of proximity to reserved lands on population and wealth disappeared by the early 2000s; it was also not evident in the 1980s. In that decade, mill closures caused by the general downturn in the wood products sector and early reductions in federal timber harvest had a direct negative effect on community population, but no statistically significant effect on wealth change in communities. In the 1990s, with the NWFP in place, mill closures had a direct negative effect on wealth and an indirect (through wealth loss) negative effect on population. That is, the mill closures did not directly influence population change, but the effect of mill closures reduced community wealth, which in turn led to population loss. Oftentimes these negative effects were not limited to communities close to NWFP reserved land because mills are often located away from the log source. By the early 2000s, the relationship between mill closures and wealth creation disappeared, and there was a direct positive relationship between communities with mill closures and communities with population growth. The authors postulated that relationships between mills closures and population and wealth found for the early 2000s may reflect the arrival of amenity migrants in mill towns (after they had already arrived in communities closest to reserved land), and the corresponding increase in residential housing value that offset (in real estate values community-wide) any continued loss in commercial property values.

Chen et al. (2016) extended this analysis by testing for any effect of proximity to NWFP reserved areas on population, income, and wealth through the late 2000s. The authors found that small communities (100 to 2,500 people) within 5 miles of protected NWFP land experienced positive increases in all three attributes relative to those that were farther away. They attribute the correlation between proximity to protected NWFP lands and income, population, and property value growth to the amenity values associated with conservation lands set aside by the NWFP, where land uses were restricted. Because a share of amenity migrants are often individuals with strong purchasing power who can purchase existing homes or build new ones, amenity migration can lead to increases in property values within a community without an associated increase in income in the community. In this study, the authors did find that property values in NWFP-proximate small communities grew more than median income, resulting in a decrease in real income in those communities. The authors found no effect of NWFP proximity for medium-size communities (2,500 to 20,000 residents).

It is difficult to generalize about the effects of the NWFP on rural communities and counties, and its role as a driver of change there, from quantitative studies based on secondary data because the body of research encompasses different periods, different geographic scales and locations, and different indicators. Moreover, although several studies find correlations between different social and economic indicators and lands protected by the NWFP, these correlations do not necessarily imply causation. For example, some studies attribute their findings to the NWFP when they may be the result of proximity to federal lands generally, instead of a specific forest management policy such as the NWFP (Charnley et al. 2008c). Nevertheless, to summarize the results of these studies: impacts attributed to the NWFP include population growth and population decline, both increases and decreases in socioeconomic well-being, and both increases and decreases in economic indicators. Some studies found no NWFP impact on population and economic indicators. Studies also found that NWFP impacts on communities differed at the community and county scales, and depended on local social, cultural, economic, and environmental contexts. In general, impacts (both positive and negative) were greater during the first decade of the NWFP than they were during the second decade. Impacts (both positive and negative) were also greater in communities located closer to national forests, or to reserved lands set aside by the NWFP; and in communities that had experienced a mill closure (not necessarily as a result of the Plan). Finally, impacts were greater at the community scale than at the county and regional scales; and were greater in nonmetropolitan counties than in metropolitan counties.

Qualitative accounts providing insight into causal relationships between the NWFP and socioeconomic conditions in rural communities are less common. Seventeen community case studies that included primary qualitative data collection were undertaken in communities surrounding federal forests in the NWFP area to evaluate its impacts on community well-being during the first decade (Buttolph et al. 2006, Charnley et al. 2008a, Dillingham et al. 2008, Kay et al. 2007, McLain et al. 2006). Charnley et al. (2008c) and Charnley and Donoghue (2006b) summarize the findings of these case studies.

They found that not all communities were affected in the same way, or to the same extent. The NWFP's impacts depended on the relative strength of the wood products industry as an economic sector around 1990; the extent to which federal timber supported that sector; and the degree to which local residents depended on federal jobs (as agency employees or contractors). Communities that participated heavily in the wood products industry in the late 1980s and early 1990s, where loggers worked mainly on federal forest lands and local mills obtained most of their wood from federal forests, were heavily affected. Communities having a large number of Forest Service or BLM employees were also heavily affected. In communities where tribal or private forest lands were the main source of supply for the industry, the NWFP had a minor impact. Although timber workers and agency employees experienced impacts, at the community level, the effects of the NWFP also depended on economic activity in other sectors. In places where other industries were also in decline (e.g., the fishing industry in coastal communities), the NWFP added to these impacts. In places with more diversified local economies, its impacts were somewhat mitigated, although jobs in other sectors did not necessarily provide opportunities for those who experienced NWFP-related job loss. In communities where the timber industry had declined prior to the late 1980s, or was never prominent—as in some agriculturally oriented communities—the NWFP had little impact.

Effects of wildfire management on communities— Several of the studies reviewed here suggest that rural communities near federal forests are more affected by federal forest management policy than communities located farther away. Communities near federal forests—no matter what their economic orientation—are also likely to face greater risks from the heightened incidence of wildfires that occur there, and that are predicted to increase under a warming climate (see chapter 2). These risks will likely be greatest in areas of WUI expansion (Wimberly and Liu 2014) (fig. 8-8). Socially vulnerable WUI populations may be at

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Figure 8-8—Home expansion into the wildland-urban interface increases the risk of losses from high-severity wildfire on federal forest lands.

even greater risk (Ojerio et al. 2011). Beyond the strictly economic impacts of wildfire, there are multiple social and health concerns associated with wildfires generally, and large wildfires specifically (Finlay et al. 2012). Recent large wildfires have resulted in injuries, property loss, and death among WUI residents. Wildfire smoke has been associated with increased risk of respiratory disease, and may also be associated with increased cardiovascular disease and mortality (Kochi et al. 2010, Liu et al. 2015, Moeltner et al. 2013, Mott et al. 2002).

Displacement of residents, stress, psychological trauma, and conflict have also been documented in communities affected by wildfires (Carroll et al. 2006, Finlay et al. 2012). The activities of federal fire managers during fires that threaten or damage the built environment can influence trust and relationships between community members and agency managers in the future (Carroll et al. 2006, 2011; Paveglio et al. 2015a). Management activities intended to alter fire behavior, restore forest conditions so they are more resilient to wildfire, or protect human values from fire are often warranted in various forest types throughout the NWFP area (see chapter 3 of this volume). Thus, eliminating fire from these systems is not possible, nor is it possible to eliminate smoke impacts, especially where prescribed fire is a needed forest restoration tool to increase forest resilience to wildfire.

Social and Economic Change in Rural Communities in the Northwest Forest Plan Area

Social science research from the Plan area that examines how communities have changed in the two decades since the NWFP was implemented forms part of a broader literature on rural restructuring in the American West that followed the decline in natural resource extraction as a prominent economic activity in rural communities. Following a brief overview of demographic change in the region, we discuss key findings of this body of research.

Demographic change—
Published accounts of demographic change in the 72 counties of the NWFP area as a whole since the Plan was implemented come from the Plan's socioeconomic monitoring reports. These are inconsistent in their data sources and

Summary—
The population of the NWFP area has been increasing at a faster rate than for the United States as a whole, with the majority of population growth occurring in metropolitan areas. Population trends in nonmetropolitan communities have been variable. Over the past two to three decades, many rural communities in the Plan area have undergone changes in demographic and economic conditions following declines in commodity production. One general trajectory is the "amenity" trajectory, in which communities that are relatively accessible and situated near natural amenities such as mountains and water bodies experience population growth owing to in-migration by people who are seeking an improved quality of life or are fleeing cities, telecommuting, becoming creative entrepreneurs, and living off of retirement or investment incomes. Amenity migration may drive local community development. A second trajectory is for communities to continue with traditional modes of production, albeit at lower levels, or to attract new forms of commodity production or service-oriented economic activity to bolster the local economy. These new businesses may be less desirable but provide jobs, at least in the short term; illegal (e.g., marijuana production on federal lands); or may seek to use natural resources in new and diverse ways through investments in sustainable agriculture and natural resource management. Many communities pursue a range of strategies, with diverse development pathways increasing their resilience. A third trajectory, however, is one in which communities find it difficult to recover from declines in commodity production, and therefore experience population and employment declines. Nevertheless, these communities have latent potential for development associated with the availability of labor, land, natural resources, or infrastructure that may become valuable in the future.

scale of analysis, making simple reporting of trends difficult. Socioeconomic monitoring of the NWFP area during the first decade (1994 to 2003) occurred at the community scale and used decennial U.S. Census data from 1990 and 2000 (Donoghue and Sutton 2006). Socioeconomic monitoring during the second decade (2004 to 2013) occurred at the county scale and used annual mid-year population estimates from the U.S. Census Bureau (reported by the Bureau of Labor Statistics and Bureau of Economic Analysis) for the years 1999 through 2012 (Grinspoon and Phillips 2011, Grinspoon et al. 2016). All of these reports distinguish between trends in metropolitan and nonmetropolitan areas. A metropolitan area is a core urban area with a population of 50,000 or more people, and can be composed of several counties.[9] The 10-year socioeconomic monitoring report identifies 10 metropolitan areas and 1,314 nonmetropolitan communities in the NWFP area (Donoghue and Sutton 2006), and identifies trends for these communities. The 15- and 20-year monitoring reports distinguish 32 metropolitan counties and 40 nonmetropolitan counties (Grinspoon and Phillips 2011, Grinspoon et al. 2016), and show population trends for these two groups of counties. General findings from the two reports are as follows:

1. Between 1990 and 2000, the total population of the NWFP area went from 8.57 million in 1990 to 10.26 million in 2000, a population increase of 19.8 percent (Donoghue and Sutton 2006). The population of the United States as a whole grew by 13.2 percent during this decade.[10] Population in the 1,314 nonmetropolitan communities went from 4.13 million in 1990 to 4.98 million in 2000, increasing by 20.6 percent. However, 21 percent of communities lost population during this period; these tended to be small (under 2,000 people). About 40 percent of communities grew at a slower rate than for the region as a whole, and about 40 percent grew more quickly. The fast-growing communities were typically bigger than the slow-growing communities (Donoghue and Sutton 2006).
2. Between 2000 and 2012, the total population of the NWFP area grew to 11.87 million, an increase of 15 percent since 2000 (Grinspoon et al. 2016). In comparison, the U.S. population grew by 11.6 percent during this period (based on 2012 population projections from the 2010 Census).[11]
3. The population of NWFP-area counties grew by 10 percent in California, 16 percent in Oregon, and 19 percent in Washington between 1999 and 2012. Population growth between 1999 and 2012 in metropolitan counties overall was twice what it was in nonmetropolitan counties, and accounted for nearly all of the population growth in the Plan area during this period. And, NWFP-area counties (both metropolitan and nonmetropolitan) grew faster than non-NWFP-area counties in the three states (Grinspoon et al. 2016), perhaps because they contain the largest metropolitan areas. These trends obscure changes occurring in individual counties and at the community scale.
4. Overall, people residing in nonmetropolitan communities and counties in the NWFP area are aging.

Changing socioeconomic conditions—
Over the past two to three decades, many rural communities in the NWFP area and elsewhere in the Western United States have undergone "rural restructuring"—changes in their demographic and economic conditions (Nelson 1997)—owing to declines in natural resource production and agriculture, which previously were the economic mainstays of these communities. Researchers investigating this phenomenon in rural forest communities in the United States and in the West have identified general trajectories of change in response, leading to different community/county types that have emerged today. This does not mean that communities were static prior to the 1980s, nor that they can be neatly categorized into one ideal type today. Nevertheless, researchers have distinguished several rural community development pathways, typically integrating

[9] http://www.census.gov/population/metro/.
[10] https://www.census.gov/prod/2001pubs/c2kbr01-2.pdf.
[11] http://factfinder.census.gov/faces/tableservices/jsf/pages/productview.xhtml?src=bkmkPl.

considerations of economic activities, “connectedness” to urban areas, population, and stocks of financial, social, or other forms of capital in doing so. These different development pathways can be used to characterize change in the NWFP area as well.

The degree of community economic dependence upon “traditional” resource use (e.g., logging, ranching, and mining) is one common variable used to differentiate rural Western communities. For example, so-called “old West” economic activities are typically contrasted with “new West” economic activities associated with the service industries, particularly tourism and real estate (Winkler et al. 2007). We apply three general trajectories of socioeconomic change documented in rural forest communities in the United States (based on Morzillo et al. 2015) to the Plan area because they are consistent with the literature from the region: (1) amenity-driven development, (2) development driven by new production strategies, and (3) economic decline. These are archetypes; communities following different trajectories can occur in the same county, and individual communities may pursue a combination of development strategies (fig. 8-9).

Gaps in the published literature prevent us from quantifying the number of communities in the NWFP area that have followed these different trajectories, and from identifying their geographic distribution. However, other researchers have developed typologies that classify counties according to variables that help to characterize socioeconomic conditions there. For example, the U.S. Department of Agriculture’s Economic Research Service (ERS) developed nine different rural-to-urban continuum codes, which classify metropolitan counties based on the size of the population in their metropolitan area (three categories), and nonmetropolitan counties based on their

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Figure 8-9—Weaverville in Trinity County, California, retains a sawmill and has also experienced amenity-driven development.

degree of urbanization and adjacency to a metropolitan area (six categories).[12] Rasker et al. (2009) developed a similar typology of urban connectivity for counties in the U.S. West that further differentiate nonmetropolitan counties. In that typology, counties are classified as metropolitan, connected, and isolated based on location within a metropolitan area or location within one hour of an airport with daily commercial passenger service. About 50 percent of the counties in the U.S. West were classified as "isolated;" 18 counties within the NWFP area (25 percent) were classified as "isolated."

The ERS has also typed counties based on several social and economic characteristics (not necessarily mutually exclusive).[13] Examples include economic dependence on recreation (fig. 8-10); economic dependence on manufacturing (fig. 8-11); retirement-destination counties (fig. 8-12); and low-employment counties (fig. 8-13). In the NWFP area, the majority of recreation-dependent counties are located along the Pacific Coast or on the east side of the Cascade Range, in areas commonly perceived as being rich in natural amenities. Manufacturing-dependent counties are rare, and are all metropolitan. Two of the manufacturing-dependent counties are focused on advanced manufacturing: Snohomish County, Washington, is a key manufacturing center for the aerospace industry, and Washington County, Oregon, is home to semiconductor and bioscience manufacturers. Retirement counties are sprinkled throughout the Plan area and are in a mix of metropolitan and nonmetropolitan locations. In general, the retirement counties tend to be associated with areas that are rich in natural amenities (e.g., Deschutes County, Oregon; Skagit County, Washington; and Shasta County, California) or that have a relatively low cost of land and housing (e.g., Douglas County, Oregon, and Lewis County, Washington). Low-employment counties are predominantly nonmetropolitan, and within the NWFP area are concentrated in northern California, southern Oregon, and the Olympic Peninsula of Washington. It is important to bear in mind that county-scale typologies do not necessarily reflect conditions at the community scale.

[12] http://www.ers.usda.gov/data-products/rural-urban-continuum-codes/.aspx.

[13] http://www.ers.usda.gov/data-products/county-typology-codes.aspx.

Amenity communities—

The most studied form of rural restructuring in forest communities nationwide, and in the Western United States, is the one that follows the commodity production → decline → amenity trajectory (Morzillo et al. 2015), in which rural communities or counties become places that attract people who wish to enjoy the natural amenities they offer, rather than because they are pursuing employment in natural resource production (Lawson et al. 2010, Morzillo et al. 2015). Natural amenities include water bodies, mountains, and public lands, and communities following this trajectory of change are typically located in or near places that offer nearby natural amenities and are relatively accessible from urban areas (McGranahan 1999, Rasker et al. 2009). Amenity communities are characterized by high population growth rates owing to in-migration by amenity migrants—people who seek an improved quality of life outside of cities, telecommute, are entrepreneurs, or who live on retirement or investment income (McGranahan and Wojan 2007, Winkler et al. 2007). For overviews of the phenomenon of amenity migration see Gosnell and Abrams (2011) and Waltert and Schläpfer (2010).

High-amenity communities and counties draw people and businesses, which in turn can drive economic development (Rasker et al. 2013). Waltert and Schläpfer (2010) identified five ways that natural amenities have been found to affect rural development: (1) new residents with flexible income sources move to the area to be closer to natural amenities; (2) new residents accept lower pay or higher costs of living in rural areas to be close to natural amenities; (3) entrepreneurs willing to accept lower profits move to rural areas to be closer to natural amenities; (4) natural amenities provide a basis for tourism, recreation and outdoor industries; and (5) amenities provide benefits from nature that improve the well-being of individual people or make businesses more profitable. In some cases, population change that provides a potential labor force with desirable skills may attract new businesses looking for workers (Waltert and Schläpfer 2010).

Research on amenity migration and amenity communities in the Northwest is relatively sparse compared to research on this topic from other parts of the American West. In the Northwest, amenity counties have been found

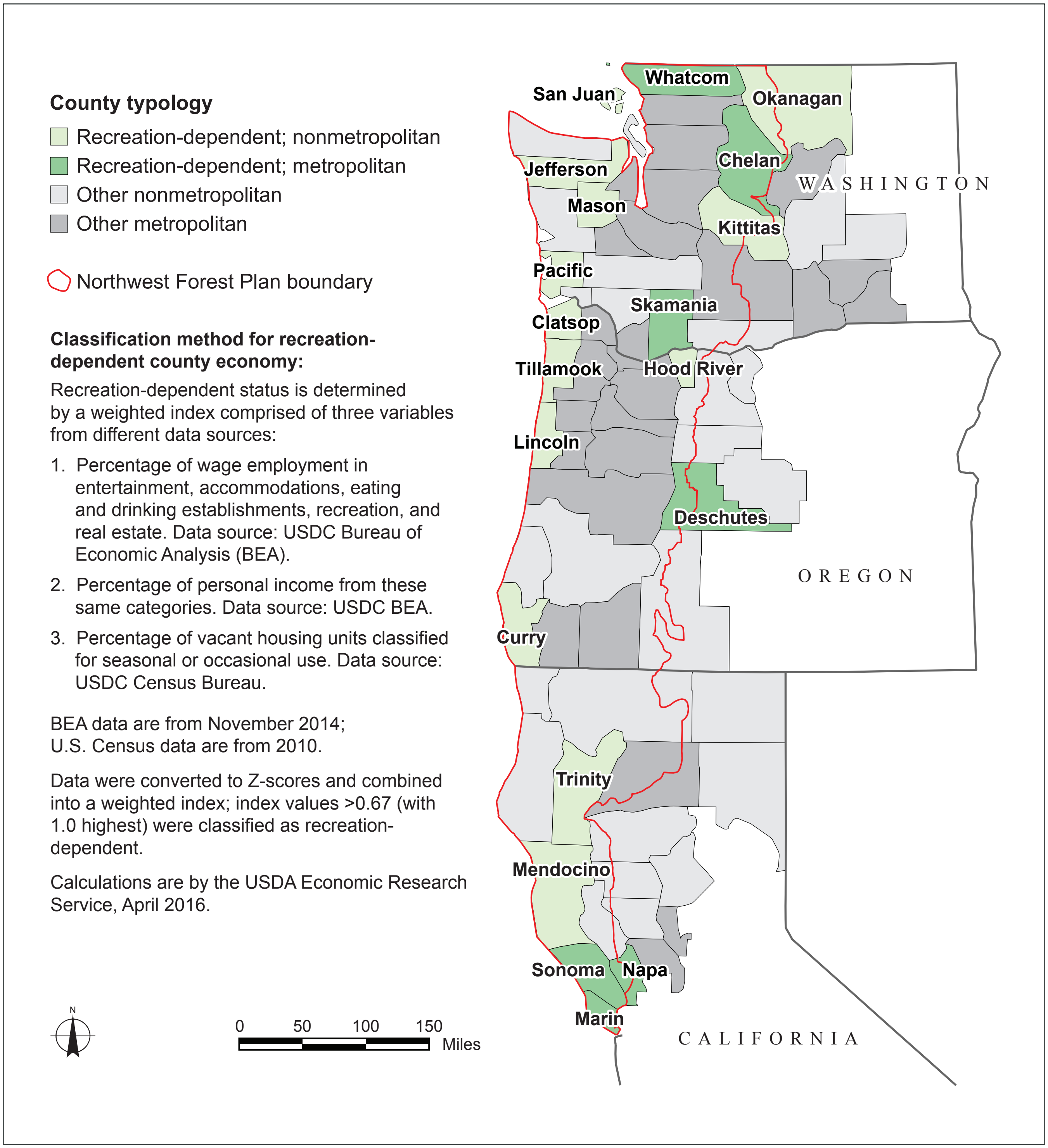

Figure 8-10—Recreation-dependent counties in the Northwest Forest Plan area. Source: U.S. Department of Agriculture, Economic Research Service.

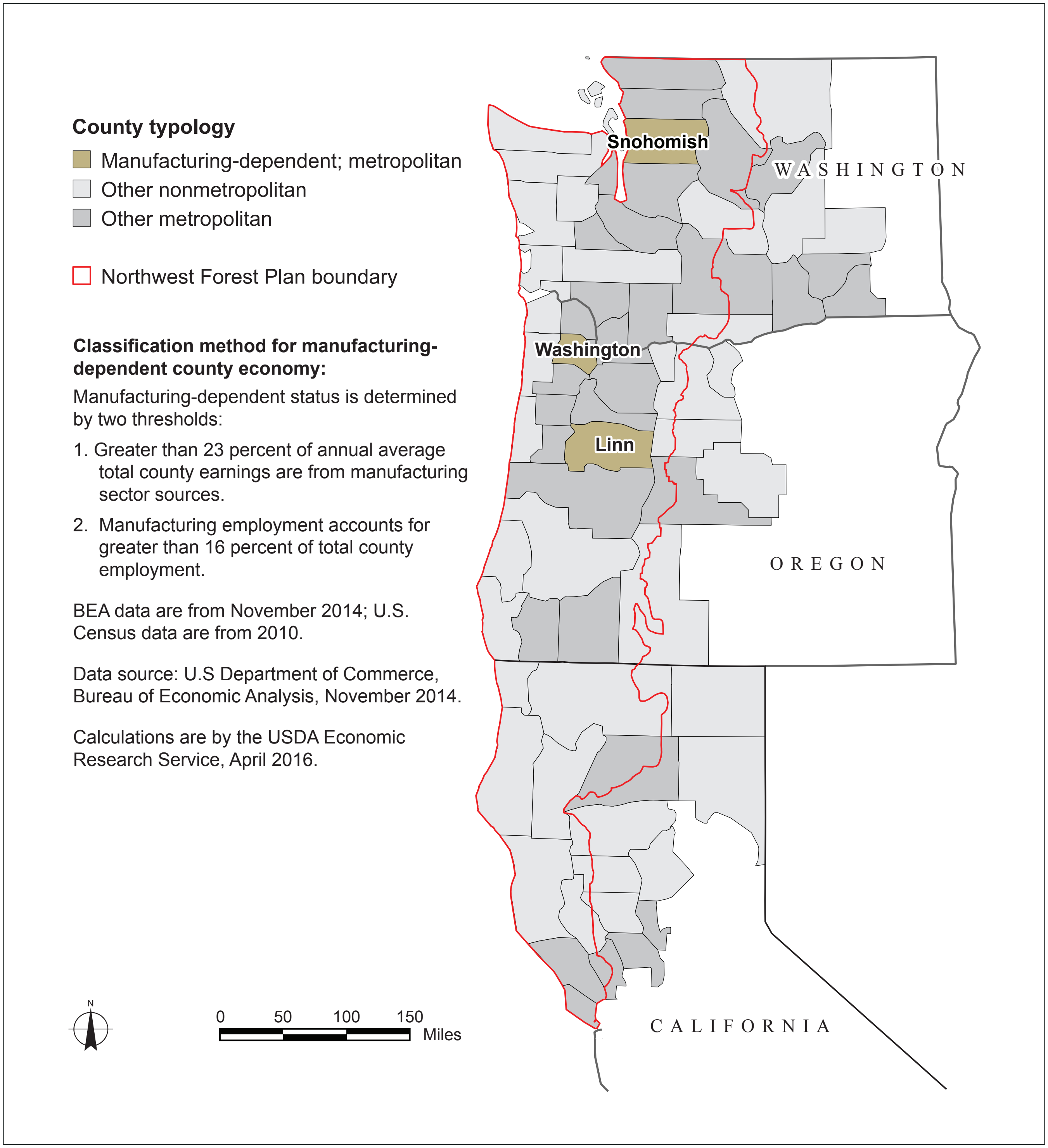

Figure 8-11—Manufacturing-dependent counties in the Northwest Forest Plan area. Source: U.S. Department of Agriculture, Economic Research Service.

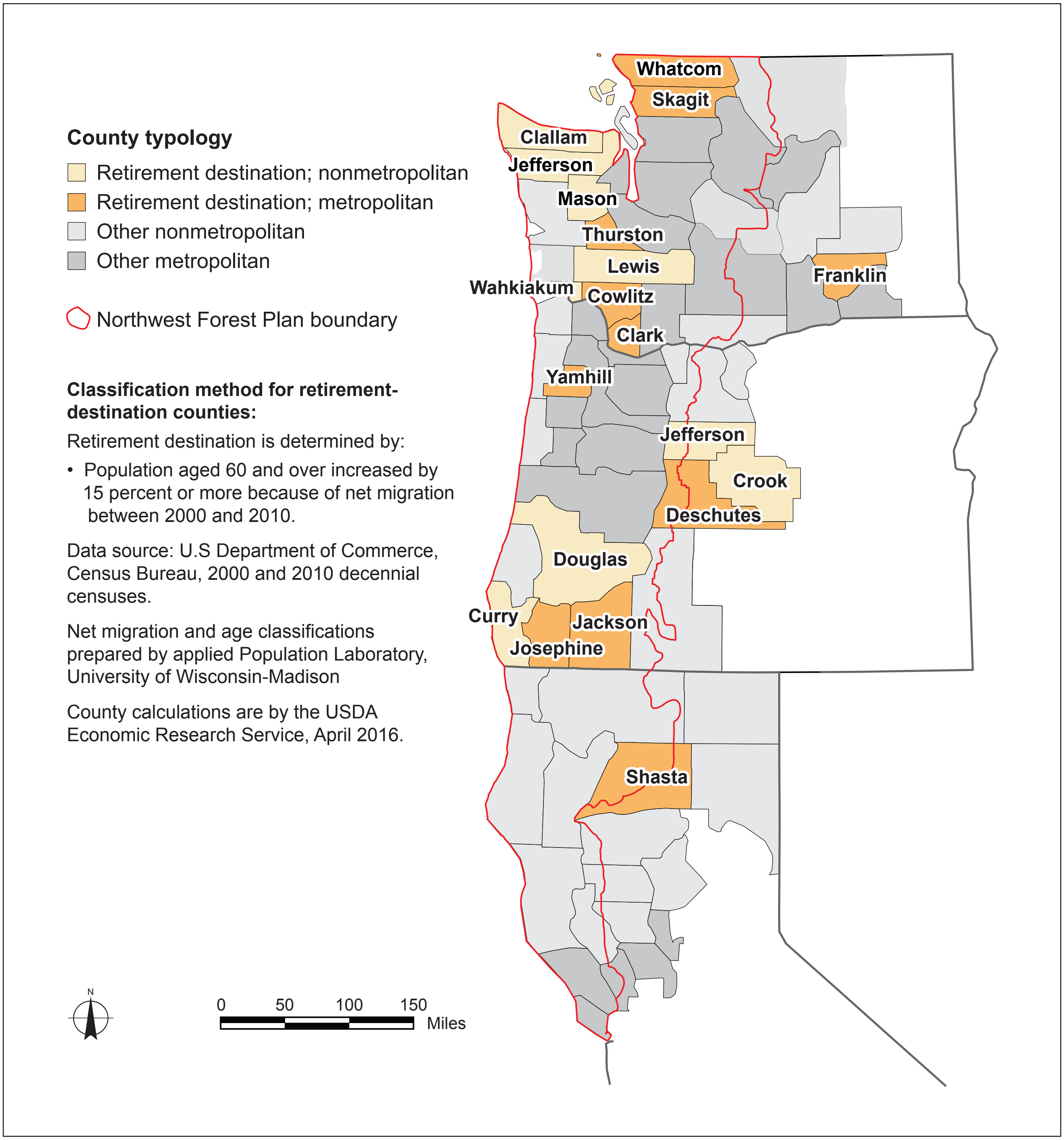

Figure 8-12—Retirement-destination counties in the Northwest Forest Plan area. Source: U.S. Department of Agriculture, Economic Research Service.

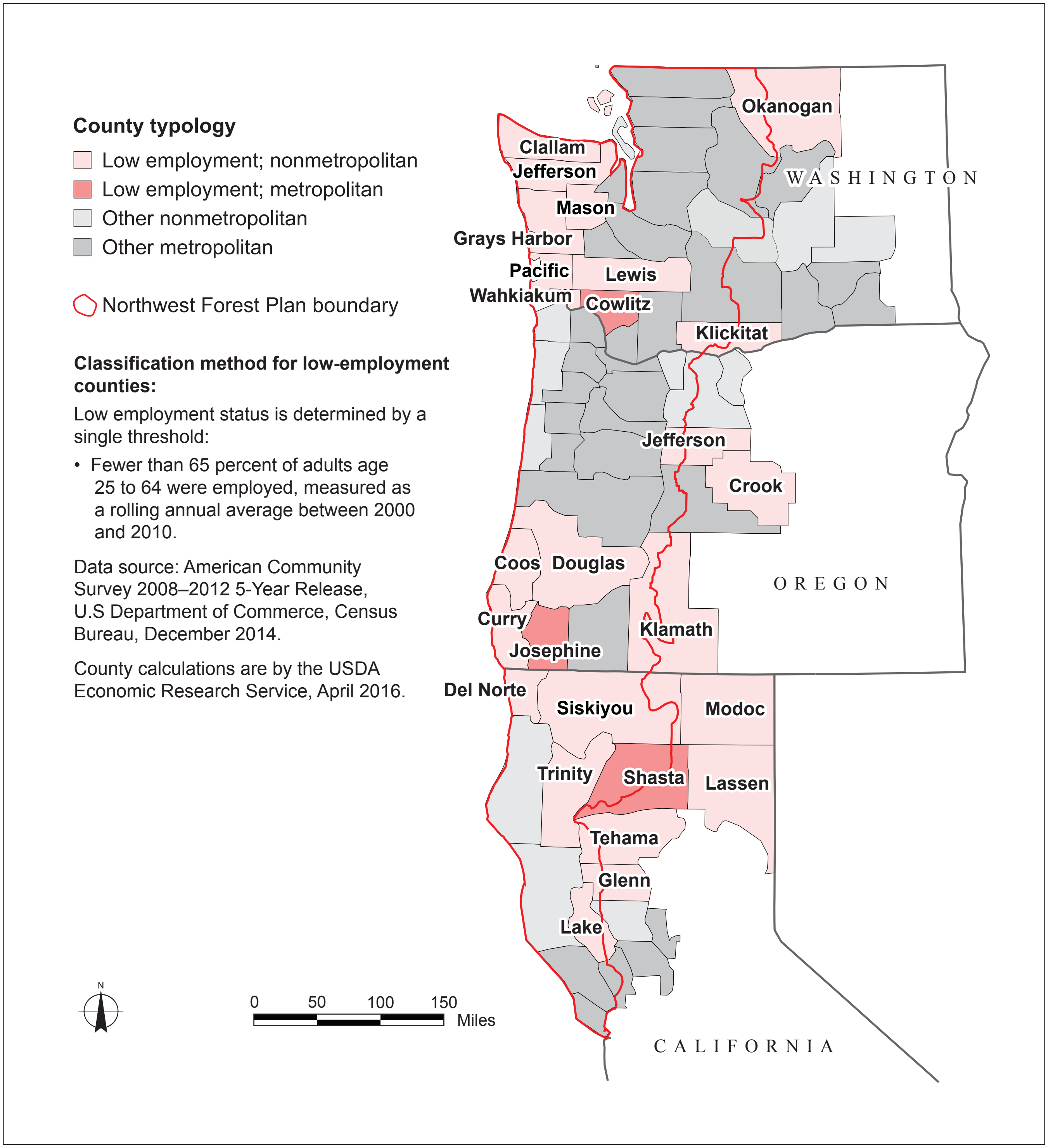

Figure 8-13—Low-employment counties in the Northwest Forest Plan area. Source: U.S. Department of Agriculture, Economic Research Service.

to attract investments in recreation and tourism, to draw middle- and high-income residents, and to be economically diversified relative to other rural counties (Lawson et al. 2010). Amenity counties also often have a high proportion of second homes; nonmetropolitan Washington counties had an average of 17 percent of their housing stock in second homes in 2010, with the number increasing rapidly (Kondo et al. 2012). Employment in the retail and services sectors in these areas is typically more important economically than employment in agriculture or natural resource extraction (Lawson et al. 2010) (fig. 8-14). Although poverty has been found to be relatively low in high-amenity counties in the Northwest compared to other nonmetropolitan counties (Lawson et al. 2010), these places are often characterized by high social and economic inequality, and by sociocultural divisions between long-time residents and newcomers (Kondo et al. 2012, Morzillo et al. 2015, Nelson 1997, Ohman 1999). In Oregon and Washington, high-amenity rural counties are concentrated along the Pacific Coast and the Cascade Range (Lawson et al. 2010). One example is Hood River County, Oregon (Pierce 2007).

The presence of public lands can be an important driver in attracting amenity migration; new arrivers often wish to live near public land boundaries. A study of housing growth within 50 km of designated wilderness areas, national parks, and national forests in the conterminous United States between 1940 and 2000 found that national forests experienced the highest absolute growth in number of housing units in their vicinity (from 484,000 to 1.8 million within 1 km of a national forest; and from 9.0 to

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Figure 8-14—Services and retail are important economic sectors in amenity-based communities.

34.8 million within 50 km) (Radeloff et al. 2010). Population growth and associated housing and road development can lead to habitat fragmentation and threats to water quality and biodiversity on federal lands (Radeloff et al. 2010), and other patterns of ecological degradation (Abrams et al. 2012). For example, Radeloff et al. (2010) found that, between 1940 and 2000, 940,000 housing units were built on private inholdings within national forests nationwide. Housing growth and associated road development near these protected areas can make them ecologically isolated by causing habitat fragmentation around their boundaries, disrupting habitat corridors between them, increasing the spread of invasive species, and increasing predation by pets (Radeloff et al. 2010). The study does not provide comparable statistics for the Pacific Northwest.

The expansion of the WUI also poses challenges for fire managers (Hammer et al. 2007). During the 1990s, 61 percent of the new housing units built in Oregon, Washington, and California (combined) were built in the WUI, causing 18 percent growth in the number of WUI housing units in these states during the decade (Hammer et al. 2007). Most of this growth occurred in the intermix, where homes and forests intermingle, making fire management especially difficult. In 2000, about two-thirds of the WUI in these states occurred in places with a 35 to 100+-year fire-return interval, the vast majority of which had departed from its historical range of variability (Hammer et al. 2007). These past patterns may portend future trends in WUI development in the NWFP area.

Communities pursuing new production strategies— A second trajectory of change in rural forest communities in the United States has been characterized as commodity production → decline → (new) production (Morzillo et al. 2015). Places that follow this trajectory find ways to continue traditional forms of commodity production, albeit often reduced or altered, or they find new forms of commodity production or service-oriented economic activity to bolster the local economy (Morzillo et al. 2015). Research indicates that change along this trajectory has various outcomes.

On the one hand, it can lead to industrial recruitment (Lawson et al. 2010). Research from the Northwest characterizes such communities as being as remote or less attractive then amenity communities, and as having weak farming and natural resource production sectors. Thus, community leaders try to lure in new businesses such as hog farms, food processing plants, corporate dairies, or prisons in the hope that they will lead to job creation. To be competitive, they may loosen environmental, labor, and zoning standards, and provide economic incentives and cheap land. Although such industries may be deemed undesirable—providing low-wage jobs, paying low property taxes, having undesirable environmental consequences, or departing after a few years—they are pursued as a means to create large numbers of jobs in the short term to keep the local economy afloat (Crowe 2006, Lawson et al. 2010). In Washington state, local control over land and resources, physical space for expansion, and accessibility to markets were found to be important community characteristics associated with industrial recruitment. Well-developed social infrastructure (e.g., schools, health care services, active community organizations, and links to agencies or organizations in nearby communities or at the state or national levels) also positively influenced industrial recruitment (Crowe 2006).

An alternative to industrial recruitment is the emergence of new but illegal production economies, exemplified by the marijuana economy that has developed in the California portion of the NWFP area since the 1980s (Polson 2013). An estimated 60 to 70 percent of the marijuana consumed in the United States is produced in California (Carah et al. 2015). The collapse of the mining and timber industries in northern California, economic stagnation, and the rise of service-oriented industries—in which many jobs are low paying, temporary or seasonal, and lack benefits—created conditions of economic vulnerability (Keene 2015). This lack of economic opportunity led many people to experiment with marijuana production. Initially illegal, marijuana production increased substantially in the 1990s and 2000s as a result of local economic restructuring and legislative changes in California legalizing the use, cultivation, and possession of marijuana for medicinal purposes (although some illegal modes of production continued, e.g., growing on federal lands). Marijuana production now plays a significant role in sustaining rural livelihoods in the region and in shaping land values there (Keene 2015, Polson 2013). This role may increase because California legalized marijuana for recreational use by adults in 2016. Large-scale production (hundreds to thousands of plants) on private lands funded by nonlocal residents for investment purposes can create conflict by driving up land prices, taking land out of food production, affecting water use, and failing to consider or contribute to local community interests (fig. 8-15). Washington and Oregon have also legalized marijuana for medicinal and recreational use, but we are not aware of any published literature on marijuana production in Oregon and Washington and its effects on local communities, economies, and the environment.

The environmental impacts of commercial-scale, outdoor marijuana cultivation in northern California's forested landscapes are beginning to be documented (Bauer et al. 2015, Carah et al. 2015, Gabriel et al. 2012). They include forest clearing, land terracing, and road construction; and diversion of large quantities of surface water for irrigation during summer when water flows are low, posing a threat to fish, amphibians, and other wildlife in watersheds important for their aquatic biodiversity. These impacts can occur on both public and private lands. Chemical pollution from heavy use of pesticides, herbicides, and fertilizers is another threat that has been documented on public lands, with these pollutants contaminating watersheds and

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Figure 8-15—Large-scale marijuana production funded by nonlocal community members and its impacts on Karuk and Yurok ancestral lands in northern California is controversial.

entering local food chains, poisoning wildlife, including fishers (*Pekania pennanti*), recently considered for listing under the Endangered Species Act (Bauer et al. 2015, Carah et al. 2015, Gabriel et al. 2012). Whether these kinds of environmental impacts will decrease in response to recent legislation legalizing marijuana cultivation remains to be seen.

Another distinct development pathway for communities pursuing new production strategies is what Hibbard and Lurie (2013) refer to as the "new natural resources economy." This strategy entails using natural resources in new and diverse ways to help drive local economic development through investments in sustainable agriculture and natural resource management (fig. 8-16), including restoration. Such activities draw on the natural resource base of rural communities in ways that both diversify the local economy and promote socioeconomic well-being by producing new goods and services for export, generating new jobs and income-earning opportunities, and producing goods and services for local use rather than importing them, thereby increasing self-sufficiency. Examples of such activities in Oregon communities include (1) sustainable farming/ranching, forest products production, and alternative energy production (production related); (2) ecotourism and agritourism (consumption related); and (3) watershed restoration, wildlife habitat protection and restoration, forest restoration, and environmental education (protection related) (Hibbard and Lurie 2013).

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Figure 8-16—Mount Adams Resource Stewards' small business incubator and log yard in Glenwood, Washington.

Examples of NWFP-area communities that are developing new natural resource economies are Hayfork, California, (Abrams et al. 2015) and Vernonia, Oregon (Hibbard and Lurie 2013). In Hayfork, a local community-based organization—the Watershed Research and Training Center—helped the community transition by developing workforce training and job opportunities associated with ecosystem management work and hazardous fuels reduction on national forests. It also invested in a small-log processing facility and a business incubator to encourage development and marketing of value-added forest products (Abrams et al. 2015). In Vernonia, some family forest owners engage in commercial nontimber forest products production from their lands, and there is a tourism economy developing in association with a recent rails-to-trails project. In addition, the community is reinventing itself as a "green" community, with rural development projects revolving around rebuilding schools according to Leadership in Energy and Environmental Design-certified standards and heat from locally produced biomass energy, and a new rural sustainability center promoting forest sustainability and clean energy (Hibbard and Lurie 2013). Hibbard and Lurie (2013) discussed barriers to the development of new natural resource economies, and suggested policies and programs that might help; none pertain directly to federal forest management.

Communities in decline—
A third general trajectory of change identified for rural forest communities in the United States experiencing dwindling commodity production is decline (Morzillo et al. 2015). Such communities are unable to recover from significant job losses associated with traditional modes of production, and therefore experience population and employment declines. They are often remote, may have undesirable environmental legacies from former extractive industries such as forestry or mining, and often have high and growing poverty rates (Lawson et al. 2010, Morzillo et al. 2015). These communities have not attracted investors or wealthy, educated immigrants; have limited development options; and are economically and politically marginalized. Not only have they failed to attract new investments; the viability of traditional economic activities such as forestry, ranching, farming, and mining continues to dwindle (Lawson et al. 2010, Nelson 1997). An example is Happy Camp, California, which was heavily affected by cutbacks in timber harvesting associated with the NWFP (Charnley et al. 2008a). Nevertheless, these communities have latent potential for development associated with the availability of labor, land, natural resources, or infrastructure that may become valuable in the future (Morzillo et al. 2015).

Adaptation to change—
A common theme that crosscuts the discussion above is community adaptation to change. Community capacity and community resilience are important to well-being in forest communities, making them more resilient to change and disturbances (such as wildfire, climate variability, and declines in the wood products industry) (Berkes and Ross 2013, Folke et al. 2010). The elements, mechanisms, and determinants of community resilience are not necessarily the same across community contexts, implying a need to consider the various development pathways of rural communities over time and their particular relationships with nearby public forest lands (Donoghue and Sturtevant 2008).

As noted, our discussion of rural community development pathways above identifies archetypes. Rural communities that have strongly "multifunctional" characteristics are more likely to be resilient to social, economic, and ecological changes associated with federal forest management, and to mitigate their negative impacts, making them more resilient (Wilson 2010). Multifunctional rural landscapes are those that have a mix of uses, including commodity production (e.g., forest products, agriculture); amenity-driven development (e.g., recreation, tourism, services); and natural resource protection (e.g., forest restoration, jobs with land management agencies). Multifunctionality helps communities diversify their rural economies and contributes to both environmental and economic health (Hibbard and Lurie 2013). Not all communities are able to develop multifunctional characteristics, and doing so depends on their natural and social assets.

Research on NWFP impacts conducted in 17 communities around federal forests in the NWFP area following the first decade of the Plan's implementation (Charnley et al. 2006b, 2008b) found that different communities experienced the different trajectories of change described above in pursuing (or not pursuing) new opportunities. Owing to their proximity to natural amenities, several communities experienced an influx of retirees, commuters, mobile or self-employed workers, or second-home owners, and benefitted from being popular recreation or tourism destinations, although not all community residents viewed this as a positive change (Charnley et al. 2008c). Other communities reoriented around new forms of production such as agriculture; new industries or service sectors associated with proximity to a major transportation corridor in or near a regional center; or the growth of tribal businesses, administration, and services. And some were in decline—especially those that were remote, surrounded by federal lands, and previously highly dependent on the wood products industry. Regardless, all communities were making efforts to develop and diversify, which was easier for some than others, depending on community characteristics.

One study (Harrison et al. 2016) examined the role of social capital (defined as behavioral norms and social networks that facilitate collective action) in influencing the capacity of three Pacific Northwest communities affected by the decline of the wood products industry to adapt to change and take advantage of new opportunities. The study found that a community's ability to develop along new trajectories aligned with local goals was influenced by interactions between different forms of social capital (bonding, linking, bridging).[14] In particular, a combination of strong bridging and linking social capital was found to facilitate desirable community outcomes. This finding builds on earlier work from the 1990s that found social cohesion to be an important characteristic influencing rural community well-being (Beckley 1998, Doak and Kusel 1996, Harris et al. 2000). Local cultural context also plays an important role in influencing how communities respond and adapt to changes like mill closures (Lyon and Parkins 2013).

These observations suggest that there is no one pathway, or set of variables, that will make communities resilient in the face of change, ensure successful adaptation, or promote socioeconomic well-being. Individual communities draw on the assets and opportunities available to them, which differ depending on social, cultural, economic, and environmental conditions. Moreover, community well-being is based on a host of quality-of-life attributes, including health, safety, political participation, social equity, and access to social services as well as jobs and income. Federal forest management can contribute to socioeconomic well-being in multiple ways (Kusel 2001, Nadeau et al. 2003, Sturtevant and Donoghue 2008), but it is only one of many factors influencing community well-being.

[14] Bonding social capital refers to relations between individuals within a community who have similar social and economic backgrounds. Bridging social capital refers to relations between individuals having different backgrounds. Linking social capital refers to relations between community members and people outside the community who have the ability to affect community outcomes (Harrison et al. 2016).

How Goods, Services, and Opportunities from Federal Forests Contribute to Community Socioeconomic Well-Being

Federal forest management contributes to socioeconomic well-being in rural communities by providing timber and nontimber forest products, recreation opportunities, jobs, other ecosystem services, and backdrops for where people want to live and work. Charnley (2006c) and Grinspoon et al. (2016) detailed and quantified many of these contributions for NWFP-area national forests and BLM districts over the first 20 years of the Plan. Here we focus on jobs in forest restoration and firefighting, nontimber forest products (NTFPs), the economic effects of recreation on federal forests, and ecosystem services from federal forests. NTFPs are also addressed in chapters 10 and 11, and recreation is also addressed in chapter 9.

Summary—
Federal forest management contributes to socioeconomic well-being in rural communities in ways that go beyond providing timber and associated jobs in the wood products industries. This section discusses jobs in forest restoration and firefighting, biomass use, nontimber forest products (NTFP) gathering, the economic effects of recreation on federal forests, and other ecosystem services from federal forests.

Restoration of federal forest lands may benefit forest communities through associated economic

activities (e.g., in-woods work and processing of restoration byproducts) as well as by providing the ecosystem services associated with restored ecosystems. In the Pacific Northwest, the ability of local communities to compete for and obtain contracts for work on nearby federal forests, and to retain local dollars, is an important factor in the adaptive capacity of communities. The Pacific Northwest has a high concentration of both hand crew and equipment-based fire suppression contracting, many of which also engage in forest restoration contracting. In some regions of the Pacific Northwest, the restoration contracting industry has transitioned to lower skill jobs, and Forest Service contracting practices for such activities tend to favor mobile businesses that employ a high proportion of temporary and migrant laborers. Although in some places the type of forest-related contracting has changed, many nongovernmental organizations and private businesses still depend on these forest-based activities for economic and social benefits, and continue to build their business around meeting federal agency needs for forest activities. Biomass energy production presents one possible pathway for adding value to restoration byproducts; examples from across the West demonstrate its potential economic benefits and suggest its role in reconciling diverse interests in forest management.

Federal forests in the NWFP region are important sources of a wide variety of commercial and noncommercial nontimber forest products, such as moss, mushrooms, cones, grasses, and firewood. These products provide important safety net, buffering, and provisioning functions for rural and urban households, and activities surrounding their harvest, processing, and use often help build social capital and cultural identities, as well as strengthening human-nature connections. The retail value of NTFPs in the United States is estimated to be at least $1.4 billion, with much of that coming from the NWFP region. Studies that have measured NTFP employment in the Pacific Northwest have estimated that roughly 10,000 individuals work as harvesters, buyers, or processors in the floral greens/bough sector, and an equal number of people who earn income in the wild mushroom sector. State recreation surveys for Oregon and Washington suggest that the rate of participation in NTFP gathering and collecting activities (excluding hunting and fishing) exceeds that of many other outdoor activities. The 10- and 20-year socioeconomic assessments for the NWFP indicate that the Plan likely reduced physical access to NTFPs through road closures and restricted legal access to NTFPs owing to harvesting prohibitions in some late-successional and riparian reserves, and restrictions on the harvest of special-status plants. However, the most important impact of the NWFP on NTFP resources is likely to be the landscape-level changes in forest structure and composition brought about by the Plan's management provisions. Likely, these changes will bode well for NTFPs such as matsutake mushrooms and moss that do well in late-successional forests, but will lead to reduced supplies of NTFPs found in early-seral-stage forests, such as salal and boughs.

Recreation on federal forests supports economic activity in local forest communities as visitors spend money while on recreation trips, and federal agencies spend money maintaining recreation resources. In this synthesis we focus on the former. Recreation visitors to NWFP-area national forests spend about $612.6 million in the communities around those forests each year. That spending supports employees and proprietors of businesses that sell goods and services to recreationists, and generates additional economic activity through the multiplier effect. In general, the economic activity generated around federal forests from recreation visitor spending depends on (1) the amount of recreation use, (2) the types of trips (i.e., day or overnight, local or nonlocal) taken by recreationists, and (3) the size of the local economy. The activity of recreationists can influence some patterns in spending, but is less important than trip type. All else being equal, those visitors on overnight trips spend 5 to 8 times more in local federal forest communities than those on day trips.

In addition to providing the socioeconomic benefits identified above, federal forests also provide important ecosystem services both to local communities and more distant urban populations. These include fresh water, food and fiber, wildlife habitat, and outdoor recreation opportunities, among others. Federal agencies are beginning to develop methods and protocols for evaluating ecosystem services and how they are influenced by various federal actions. Within the NWFP area, efforts largely have focused on identifying and quantifying key ecosystem services produced on the region's national forests. Although these efforts have made significant progress in raising awareness and concern for these important forest benefits, formal methods for routinely including ecosystem services values into national forest management largely are still in development by the Forest Service.

Forest restoration and wildfire-suppression contracting— Despite the overall reduction in traditional timber management activity on national forest lands, in both the Forest Service and many rural communities there has been interest in and support for restoration and stewardship activities that generate both direct employment and byproducts of potential economic value (Nechodom et al. 2008). This opens the possibility for development of a "restoration economy" (Nielsen-Pincus and Moseley 2013) based on various activities, including "ecological forestry" (Franklin and Johnson 2012), associated with the restoration of structure or function to forest ecosystems. Such activities include stream rehabilitation, fish passage improvement, road decommissioning, riparian planting, forest fuel reduction treatments (designed to decrease fuel loads, break up fuel continuity, and reduce the risk of crown fire), and thinning projects designed to introduce structural heterogeneity to second-growth stands (fig. 8-16). All these activities entail employment in planning, implementation, oversight, monitoring, or other duties, and some of them produce byproducts that can be used for bioenergy, with associated economic benefits. Nielsen-Pincus and Moseley (2013) found that an average of 16.3 jobs, $589,000 in total wages, and $2.3 million in overall economic activity were associated with every $1 million of restoration grant spending in Oregon; and economic impacts were greater in rural counties than in metropolitan counties. Baker and Quinn-Davidson (2011) calculated that the restoration sector brought nearly $135 million into Humboldt County, California, between 1995 and 2007. Thus, restoration contracting now represents a potentially significant source of forest-based jobs in rural communities.

In the Pacific Northwest, restoration contracting includes a variety of forest-related management actions, such as reforestation, thinning, mastication and chipping, and other practices aimed at improving or restoring the health of the forest (see chapter 3). Forestry support work involves seasonal and labor-intensive activities including planting and maintaining tree seedlings, piling and burning brush, thinning trees, harvesting cones, and applying herbicides (Moseley 2006b) (fig. 8-17). These activities contribute to a variety of forest management goals, from forest and watershed restoration to timber management and wildfire mitigation (Moseley et al. 2014). Related wildland fire suppression work can include heavy-equipment operation and more manual tasks such as digging fire lines.

Relatively little scholarly research has focused on the forest management-related service-contracting sector. Past research suggests that these contractors operate in regional markets that involve working close to home as well as traveling relatively long distances, sometimes across state lines, to perform forest management services on federal lands (Nielsen-Pincus and Moseley 2013). Contractors are more likely to travel long distances if the work is manual and labor intensive, such as tree planting and hand thinning. Contractors that work on equipment-intensive activities such as stream restoration, road construction, and mechanical thinning tend to work closer to home (Moseley and Reyes 2008, Moseley and Shankle 2001, Moseley and Toth 2004).

Understanding where contractors are located has been an important component of the research on restoration contracting because it sheds light on where and how contracting

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Figure 8-17—Thinning to restore forest resilience to wildland fire can be equipment-intensive.

businesses create local community benefit. An intended outcome of the NWFP was for the Forest Service and BLM to offset job loss in the timber production, harvesting, and processing markets through public land restoration, including the use of contracting (Moseley 2006b). Both the Forest Service and BLM have transitioned away from intensive forest management for timber (e.g., replanting clearcuts) to more restoration-focused work (Moseley 2006b). Moseley (2006b) found that significant declines in Forest Service contract spending subsequently decreased the amount of contracting money flowing to rural communities. These trends have continued, as an increasing amount of the Forest Service budget is allocated to wildfire suppression (Calkin et al. 2011, Gebert and Black 2012, North et al. 2015).

In some regions of the Pacific Northwest, the restoration contracting industry has transitioned to lower skill jobs. Changes in federal policy and practice, and a refocus on reducing wildfire risk in drier, fire-prone forests in the early 2000s, led to a need for low-skill, labor-intensive fuels reduction work in federal forests (e.g., thinning trees and clearing brush). Forest Service contracting practices for these kinds of activities tend to favor mobile businesses that employ a high proportion of temporary and migrant laborers (Moseley et al. 2014; Sarathy 2008, 2012). The implications of these transitions and of contracting for lowest bid Forest Service work are further detailed in chapter 10. In northern California, for example, the availability and structuring of restoration contracts have put many smaller businesses based in rural communities at a disadvantage relative to larger, more mobile urban-based contractors (Baker and Quinn-Davidson 2011), which led a local, community-based nonprofit organization to begin training and hiring local residents to be able to contract with the Forest Service to perform this fuels-reduction work (Abrams et al. 2015). This example illustrates a shift by community organizations from other work into contracting, which is part of a growing trend in which organizations (nongovernmental and private businesses alike) are changing and adapting their roles to fit new or amplified needs emanating from changes in Forest Service forest restoration and fire-suppression contracting.

In the Pacific Northwest, the ability of local communities to compete for and obtain work contracts on federal forests, and retain local dollars, is an important factor in the adaptive capacity of communities. State or federal contracts for restoration or wildfire suppression services that are captured by local businesses can benefit local economies. In contrast, hiring contractors from outside local communities can reduce the amount of forest restoration dollars that circulate in the local economy.

Contracting for fire suppression purposes began in the 1970s, when loggers and other forest workers would fight fires as needed to protect their livelihoods—which were based on work in the forest. Fire suppression was conducted in the shoulder seasons for other forest work, or when forests were closed to forestry work in the hottest fire-prone months of the summer. Recent research exploring connections between restoration contracting capacity and fire suppression capacity found that the amount of money captured during a fire by community businesses located near the fire increases with the number of vendors involved in forest and watershed restoration prior to a fire, suggesting that local business restoration capacity might influence local fire suppression response (Moseley et al., n.d.). Similar to evidence about wildfire hazard mitigation (Moseley and Toth 2004), findings by Moseley et al. (n.d.) also suggest that counties containing more diversified urban economic centers may be more likely and prepared to capture wildfire suppression contracting work than smaller, less diversified, and moderately isolated counties.

Research on the effects of large wildfires in the Western United States by Nielsen-Pincus et al. (2013) found that wildfires generally improved county-level employment and wage growth while suppression efforts were active. However, following a wildfire, counties experienced increased economic volatility, though these effects differed by the type of county in which the wildfire occurred. Employment growth associated with fire-suppression spending suggests that developing community capacity could change how local economies experience wildfire, potentially facilitating more local community capacity to participate directly (fire crews or equipment), or indirectly (e.g., support services) in fire suppression, keeping wildfire suppression funds in the community longer (Nielsen-Pincus et al. 2013). Although these studies provide evidence of links between a community being engaged in forest management and restoration and local participation in fire suppression efforts, the lack of historical analysis of restoration and fire suppression contracting markets means that little is known about how these relationships have changed over time. However, recent related research on the location and diversity of fire suppression contractors and their equipment suggests that the two markets have become more complex as private wildfire contracting has become more nationalized and mobile (Huber-Stearns et al., n.d).

Changes in federal wildfire contracting policy, such as creating more nationalized dispatch systems, or the contracting award system, may unintentionally limit local contractors' ability to participate in local fire suppression efforts (Davis et al. 2014). In a time of increased focus on collaborative fire management and local workforce capacity development (e.g., the National Cohesive Wildland Fire Management Strategy), the finding that participation in federal contracting prior to a fire shapes suppression capacity can help focus policy and practice on these linkages.

The Pacific Northwest still has one of the highest concentrations in the United States of both hand crew and equipment-based fire suppression contracting (Huber-Stearns et al., n.d). In the past decade, fire-suppression contracting in the region has been experiencing a transition, as contracting processes have become more standardized, and more businesses have joined the industry. All the 48 regional and national hand crew businesses, and more than 600 of the 2,016 total equipment contractors active in

2015, were located in Oregon, Washington, and northern California (Moseley et al., n.d.).

Although many restoration businesses are still engaged in fire suppression contracting, there has been a shift in the past decade toward contracting companies entering the market primarily for fire contracting purposes (e.g., businesses purchasing equipment specialized for fire suppression, and hiring crews for fire suppression). This shift is in contrast to 20 years ago, when restoration contractors took on fire suppression work as needed and with the forestry equipment they had on hand (Moseley et al., n.d.). Recent research has also found that in several cases, these contracting companies come from other sectors, such as construction, heavy equipment, and services (e.g., portable showers, food, and housing units), and have now expanded their work into fire contracting. In many instances, restoration contracting is not the primary source of income for these businesses. Rather, it is fire suppression work, or the other sectors in which they operate during the rest of the year (e.g., construction) (Moseley et al., n.d.). As fire suppression needs differ year to year, some of the businesses that hire fire hand crews have faced critical challenges with employee retention, and looked to find other sources of income to extend the employment period for their seasonal hand crew employees. One option has been to enter the forest restoration contracting realm, using their fire suppression equipment and resources to conduct forest restoration work outside of fire season (Huber-Stearns et al., n.d).

As both Forest Service and BLM budgets and workforces decline, and are constricted further by a larger proportion of the budget going to wildfire suppression, agencies are contracting out an increasing amount of their land management work, which includes forest restoration and wildfire suppression (Moseley 2005). This suggests a continued (yet unpredictable) demand for forest-based restoration and fire contracting activities across the NWFP area. Although in some places the type of forest-related contracting has changed, many nongovernmental organizations and private businesses still depend on these forest-based activities for economic and social benefits, and continue to build their business around meeting federal agency needs for forest management and restoration work.

Biomass use—

In addition to the “in-woods” work associated with removing trees and other forest fuels, fuel reduction and thinning projects result in the production of restoration byproducts with potential economic benefit to forest communities. These include biomass materials such as tops, branches, and small-diameter trees as well as larger materials suitable for traditional commercial processing. The development of biomass-use infrastructure capable of adding value to otherwise unmarketable byproducts has been specifically supported through grant programs, targeted policies, and research efforts (Becker et al. 2009, 2011b). In particular, biomass energy production has been identified as a potential means of integrating forest restoration and rural community development while producing energy from renewable sources (Becker and Viers 2007, Hjerpe et al. 2009) (fig. 8-18).

It is extremely difficult for forest biomass energy production to be profitable as a stand-alone activity, owing to issues such as the dispersed nature of the raw material, long haul distances, the low energy density of wood, and low prices of other energy sources (Aguilar and Garrett 2009, Sundstrom et al. 2012). Development of forest biomass energy in areas with a large federal forest presence has been challenged by additional factors such as a lack of predictability in access to raw materials (Becker et al. 2011a, Stidham and Simon-Brown 2011). The cost of forest biomass harvesting is often greater than the value of resources removed (Evans and Finkral 2009); biomass treatments therefore tend to rely upon supportive public policies (e.g., direct subsidies, renewable energy mandates) to remain feasible. Biomass energy installations themselves can generate controversy regarding issues such as the possible effects of raw material demand on nearby forests (Stidham and Simon-Brown 2011). However, given appropriate public consultation and collaboration, the use of biomass can also represent an approach to reconciling diverse social, economic, and environmental restoration interests (Hjerpe et al. 2009).

The collection, transportation, and processing of biomass materials represents a potential economic opportunity


Susan Charnley

Figure 8-18—Forest restoration byproducts provide fuel for biomass energy production.

for forest communities. An analysis of 43 timber-producing counties in east Texas suggests that residue procurement and biomass energy production could collectively generate direct, indirect, and induced jobs equal to nearly one-third of current logging sector employment (Gan and Smith 2007). Using fiscal year 2005 data from five national forests in the Southwest, Hjerpe and Kim (2008) determined that fuel reduction expenditures (including prescribed fire) resulted in 337 direct full-time equivalent jobs and 151 indirect and induced jobs. Communities with installed biomass-use capacity may also benefit forests, as the presence of small-diameter processing facilities results in a greater ability to perform treatments on nearby forest land (Nielsen-Pincus et al. 2013). There is some evidence that development of local processing infrastructure can lower the per-acre cost of forest restoration activities, therefore allowing more area to be treated with a given level of funding (Becker et al. 2011a). Stakeholders in a number of communities have collaborated with one another and with Forest Service managers to design long-term, large-scale restoration projects capable of catalyzing this beneficial relationship between biomass-use capacity, forest restoration treatments, and associated economic benefits (Abrams 2011, Schultz et al. 2012). A key challenge in this context is aligning biomass-use infrastructure, state or federal policies regarding biomass utilization, and contracting mechanisms to stimulate investments that simultaneously support community economic development and forest restoration activities. An additional challenge is providing a long-term, reliable supply of biomass material from federal lands to incentivize infrastructure investments. Stewardship contracting is one mechanism the Forest Service and BLM can use to address this barrier (Nielsen-Pincus et al. 2013).

Nontimber forest products—

Nontimber forest products, or special forest products as they are known by the Forest Service and the BLM, include "bark, berries, boughs, bryophytes, bulbs, burls, Christmas trees, cones, epiphytes, fence material, ferns, firewood, forbs, fungi (including mushrooms), grasses, mine props,[15] mosses, nuts, pine straw, posts and poles, roots, sedge, seeds, shingles and shake bolts,[16] transplants, tree sap, rails, and wildflowers" (USDA FS 2001). These NTFPs are often grouped into broad functional categories, with common categories consisting of edibles, medicinals, arts and crafts, ornamental and decorative materials, fuel, transplants and other landscaping products, and construction materials (Alexander et al. 2011b). NTFP management and research are complicated by the extremely large number of species from which this broad array of products is derived. Vance et al.'s (2001) guide to commercial NTFPs in the Pacific Northwest describes products from 59 native species in detail, lists 60 additional native species that are commercially harvested, and emphasizes that many other species are bought and sold in markets. NTFP species harvested in the Pacific Northwest likely number in the hundreds (Jones and Lynch 2007). Table 8-1 lists some of the most common commercial NTFPs harvested in the Plan area. This chapter provides a broad overview of NTFP harvesting in the Plan area, whereas chapter 10 describes commercial NTFP harvesting by low-income and minority populations; and chapter 11 addresses the importance of specific NTFPs to American Indians.

It is difficult to characterize the contribution that NTFPs from federal forest lands in the Plan area make to community socioeconomic well-being because of the large number of products, variety of organism parts, and diversity of species that make up this category of forest products. No studies have systematically evaluated the relative importance of federal lands as a source of supply for NTFPs in the Plan area. Charnley (2006c) and Grinspoon et al. (2016) documented the quantities of special forest products sold from Plan-area Forest Service and BLM lands during the first two decades of the NWFP based on permits and contracts the agencies issue to members of the public. However, systems for tracking the quantities of NTFPs harvested on national forests and BLM lands are not structured in ways that would allow one to determine whether permittees have harvested more or less than the quantities indicated on their permits (Alexander et al. 2011b). And, no studies document the extent to which unauthorized NTFP harvesting takes place on federal lands in the NWFP region, although it is probable that a significant portion of NTFPs are harvested without authorization (Dobkins et al. 2016, McLain and Lynch 2010, Muir et al. 2006, NFWC 2015). Nevertheless, research suggests that federal forests are important sources of supply for a number of products, including wild mushrooms (McLain 2008, Pilz et al. 2007, Richards and Creasy 1996); beargrass (Charnley and Hummel 2011, Hummel et al. 2012); huckleberries (Kerns et al. 2004); firewood, Christmas trees, floral greens, limbs and boughs, moss, cones, and posts and poles (Charnley 2006c, Grinspoon et al. 2016) (fig. 8-19).

Table 8-1—Commonly harvested commercial nontimber forest product species in the Northwest Forest Plan area

Species	Scientific name
Floral greens:	
Salal	*Gaultheria shallon*
Evergreen huckleberry	*Vaccinium ovatum*
Beargrass	*Xerophyllum tenax*
Tall Oregon grape	*Berberis aquifolium*
Western redcedar	*Thuja plicata*
Noble fir boughs	*Abies procera*
Deer fern	*Blechnum spicant*
Western swordfern	*Polystichum munitum*
Mushrooms:	
Morel	*Morchella* spp.
Chanterelle	*Cantharellus cibarius*
Matsutake	*Tricholoma magnivelare*
Bolete	*Boletus* spp.

Sources: Blatner and Alexander 1998, Lynch and McLain 2003, Schlosser and Blatner 1995, Weigand 2002.

[15] Mine props are lengths of wood used to hold up a mine roof.
[16] Shake bolts are blocks of wood used for making shingles.

Rebecca McLain

Figure 8-19—Mushroom picking is an important commercial and recreational gathering activity on federal forest lands.

***Market context*—**Market demand for many NTFPs has increased over the past 20 years in response to growing consumer interest in wild-harvested and organically produced foods and medicines (Pilz et al. 2007, Smith et al. 2010) as well as shortages in supply in other parts of the world for products such as wild mushrooms that are traded primarily in international markets (McLain et al. 1998, Pilz et al. 2007). No reliable data exist for the amounts and values of NTFPs harvested in the United States or from the NWFP area. However, extrapolating from Forest Service and BLM permit and contract data, Alexander et al. (2011b) estimated that the retail value for NTFPs harvested from BLM and Forest Service lands in the United States in 2007 was at least $1.4 billion, with the majority attributable to NTFPs harvested in the Pacific Coast region. A similar analysis covering the years 2004 to 2013 found that the estimated retail value of NTFPs trended upward and was roughly $1.9 billion in 2013 (Chamberlain 2015). Nationwide, firewood, crafts and floral products, and Christmas trees—in that order—consistently had the highest total retail values (Alexander et al. 2011b, Chamberlain 2015). In both studies, the Pacific Coast region dominated in permitted harvest quantities (and therefore retail value) for arts, crafts, and floral products; edibles; grasses; nursery and landscape products; and regeneration and silviculture products. The region was second after the Rocky Mountain region in permitted harvest quantities of fuelwood and posts and poles. However, Alexander et al. (2011b) cautioned that it is unclear whether regional differences in permitted harvest quantities reflect differences in actual quantities harvested, or cross-regional differences in agency permitting and enforcement capacity. A 2014 survey of Forest Service employees in the agency's Pacific Northwest Region (Oregon and Washington) found that respondents most commonly labeled the following products as being among the "five most important" products gathered on the national forest where they worked: firewood (53 percent of respondents); boughs (14 percent); mushrooms (10 percent); beargrass (10 percent); Christmas trees (10 percent); and floral greens (5 percent) (Crandall 2016).

The only NTFP industries in the Pacific Northwest for which annual wholesale values have been calculated are floral greens and wild mushrooms. Schlosser et al. (1991) estimated the wholesale value of floral greens and boughs harvested in western Washington, western Oregon, and southwestern British Columbia during 1989 at $128.5 million. The wholesale value of wild edible mushrooms harvested in Oregon, Washington, and Idaho during 1992 was estimated at $41.1 million. Unfortunately, more recent valuations of NTFP industries in the Pacific Northwest (or elsewhere in the United States) do not exist. Many NTFPs harvested in the Pacific Northwest are sold in global markets (Alexander et al. 2002, 2011b), making them susceptible to demand and price fluctuations linked to economic and environmental conditions elsewhere. Although floral greens (including holiday greens for wreaths and swags), wild mushrooms, and huckleberries are commonly identified as the most economically important NTFPs in the Plan area (Schlosser and Blatner 1997), the values extrapolated from NTFP permit and contract data suggest that firewood and posts and poles are equally important economically, if not more so. No studies of the socioeconomic dimensions of either firewood or post and poles harvesting for the region exist.

The number of persons who currently earn a full or partial livelihood from NTFPs is unknown. However, Schlosser et al. (1991) estimated that, in 1989, processors in western Washington, western Oregon, and southwestern British Columbia bought floral greens and boughs from roughly 10,000 harvesters. In a later study, Schlosser and Blatner (1995) estimated that the wild mushroom industry provided income-earning opportunities for roughly 10,400 harvesters in Oregon, Washington, and Idaho. Whether and how much overlap there is between the two industries is unknown. Most of the processing facilities for NTFPs harvested in the Pacific Northwest were located west of the Cascade Range (Schlosser and Blatner 1997), but the number employed in those facilities is unknown. The NTFP sector offers income-earning opportunities that are easily accessible with little capital investment, but as described in chapter 10, working conditions for harvesters are sometimes poor, and it is likely that the more lucrative opportunities are in processing and marketing (Schlosser and Blatner 1997). As currently structured, the NTFP sector is "one piece of a larger mosaic of rural development options" (Schlosser and Blatner 1997: 2) rather than an economic driver. The NTFP sector contributes to the well-being of individuals, households, and firms located in both rural and urban areas. More than half of the harvesters interviewed during a study of beargrass harvesting on the Gifford Pinchot National Forest lived in the cities of Tacoma and Aberdeen, Washington (NFWC 2015). Many wild mushroom harvesters on the Deschutes National Forest in central Oregon also live in cities located west of the Cascades or in northern California (McLain 2008, Tsing 2015). However, the extent to which urban residents rely on NTFP-related work and the impacts that the NWFP has had on urban residents have not been the subjects of scientific studies.

Nonmarket contributions of NTFPs to socioeconomic well-being—The NTFP sector differs from most other natural resource sectors (i.e., mining, wood products, livestock production), in that much economic activity linked to the harvesting, processing, and exchange of NTFPs remains strongly rooted in the informal sector. Informal economic activity is defined as "economic activity that takes place outside of governmental regulatory and reporting systems" (McLain et al. 2008: 1), and as numerous studies attest (Brown et al. 1998, Carroll et al. 2003, Emery 1998, Hinrichs 1998, Levitan and Feldman 1991, Love et al. 1998, Nelson 1999, Richards and Alexander 2006), such activities are both ubiquitous and important contributors to community and household well-being. Assessments of the contribution of NTFPs to community well-being must therefore account for contributions from activities taking place at the edges and outside of the formal sector, as well as those tracked within the formal sector. Practically, this means that one cannot rely solely on standard economic measures, such as number of jobs created or the value of products sold in formal markets, to assess the contribution that NTFPs make to community well-being. In part this is because the number of jobs and market values associated with NTFPs are often not well captured in many of the standard economic activity accounting systems, such as the Harmonized Tariff Schedule that the U.S. government uses to track exports and imports (Alexander et al. 2011b), or the U.S. Census Bureau's County Business Patterns database, which tracks the number of businesses operating in each county, as well as how many people each business employs and the size of its payroll (Smith et al. 2010).

Ethnographic studies of NTFP harvesters and buyers indicate that NTFPs perform safety net, buffering, and provisioning functions for both rural and urban households (Emery 1998, Emery and Pierce 2005, Hinrichs 1998, Love et al. 1998, McLain et al. 2014, Poe et al. 2014). NTFP activities taking place outside of formal markets function as a type of "intergenerational and cultural glue," helping community members and families build and strengthen social ties and maintain cultural identities (Brown et al. 1998, Carroll et al. 2003, Love et al. 1998, McLain 2008, Richards and Alexander 2006, Poe et al. 2014). Unlike timber harvesting, which few people would categorize as a leisure activity, some commercial NTFP harvesting falls "somewhere in between" (Carroll et al. 2003, McLain 2008), with participants viewing harvesting as simultaneously work and leisure. A common theme among commercial and noncommercial harvesters alike is that NTFP harvesting is important to them in part because it provides an opportunity to strengthen their connections with the natural world and improve their physical and mental health (Emery and Ginger 2014, Love et al. 1998, McLain 2008, Poe et al. 2014, Tsing 2013).

Recent surveys of outdoor recreationists in Oregon and Washington show that "gathering/collecting things in a nature setting" is an activity practiced by a significant percentage of the population in the NWFP region. We are not aware of any comparable data for California. Washington state's Statewide Comprehensive Outdoor Recreation Plan (SCORP) survey results for 2012 analyzed the participation by residents from across Washington in four types of gathering/collecting activities (Responsive Management 2012). As indicated in table 8-2, slightly more than one-quarter of adult residents had participated in

Table 8-2—Percentage of Washington and Oregon SCORP survey respondents participating in specified outdoor activities during the 12 months preceding the survey

Outdoor activity	Washington respondents	Oregon respondents
	Percent	
Gathering/collecting things in nature setting:	27.2	21.9
Berries or mushrooms	14.9	—
Shells, rocks, vegetation	18.4	—
Firewood	6.7	—
Christmas trees	4.2	—
Selected outdoor activities:		
Bicycle riding (trails)	24.4	12.2
Camping (car/motorcyle with tent)	26.5	34.6[a]
Cross-country skiing	4.5	5
Downhill skiing	10.4	16.3
Hiking	53.9	48
Hunting (big game)	8.4	8.3
Off-roading (four-wheel drive)	9.5	9.8
Snowshoeing	6.7	8.5

SCORP = Statewide Comprehensive Outdoor Recreation Plan; — = No data.
[a] Car camping only.
Source: Responsive Management 2012 and Rosenberger and Lindberg 2012.

gathering or collecting in a nature setting in the previous 12 months, with participation rates in mushroom/berry picking and shell/rock/plant collecting being more than double the participation rates in harvesting firewood or Christmas trees. Table 8-2 shows that participation rates for gathering/collecting were greater than for many other outdoor activities, including downhill and cross-country skiing, hunting, off-road vehicle riding, and bicycling on forest or mountain trails. Residents of rural areas or small towns were somewhat more likely to participate in gathering or collecting than urban or suburban residents (29 percent and 24 percent of respondents, respectively). Table 8-3 shows that respondents gathered on diverse landownership types, with 18 percent gathering on national forests and only 1 percent on BLM lands. This difference is likely because very little BLM-managed land is located in Washington. Overall, the percentage of persons gathering or collecting on national forests or BLM-managed lands in Washington is relatively small compared with those who gather or collect on private or other types of public lands. However, these figures represent recreational gathering only; the bulk of commercial harvest likely takes place on federal and state forests and large private timber holdings.

Table 8-3—Percentage of Washington SCORP survey respondents who gather or collect things in nature settings on specified land ownerships

Land ownership category	Respondents
	Percent
National park or monument	8
State park	18
County/city/municipal park	8
National forest	18
State forest	8
National wildlife refuge	1
Bureau of Land Management land	1
Other public land	19
Own property	14
Someone else's private property	27

SCORP = Statewide Comprehensive Outdoor Recreation Plan.
Source: Responsive Management 2012.

The Oregon SCORP survey, which was also administered to residents statewide, collected data about gathering/collecting participation rates by Oregon residents during 2011, but did not break down the data by type of gathering activity (Rosenberger and Lindberg 2012). The percentage of Oregon residents who participated in gathering/collecting ranged from a low of 16.3 percent in the area around Portland to a high of 47 percent in northeastern Oregon, with an average of 22 percent for the entire state. Unfortunately, the authors lumped rock collecting in with plant, mushroom, and berry collecting, making it difficult to ascertain the percentage associated with NTFP gathering. The Oregon survey did not gather data about landownerships on which collecting took place. Table 8-2 shows how participation rates for gathering/collecting in Oregon compared with a selection of other activities.

A study by Starbuck et al. (2004) is the only example of research that has looked at the economic value of recreational NTFP harvesting in the Plan area. By using travel cost methods with 1996 permit data from the Gifford Pinchot National Forest, they estimated that one visitor day of berry and mushroom harvesting was worth \$30.02 (in 1996 U.S. dollars). This compared with roughly \$87/day for camping and \$53/day for picnicking (Alexander et al. 2011a). More studies using the travel cost method or other forms of non-market valuation are needed to understand how much different types of recreational NTFP harvesting contribute to local economies.

***How the NWFP affects NTFP supplies from federal lands*—**Permitted harvest quantities are currently the best data available for analyzing trends in the demand for NTFPs on federal lands. However, two important caveats limit the utility of permit data as an indicator of NTFP demand. Both the Forest Service and BLM lack the capacity to track with any accuracy the quantities of NTFPs actually being harvested, and permit data merely reflect the maximum amount that the permit holder hopes to be able to harvest. Additionally, other factors, such as price shifts, weather conditions, and changes in consumer preferences can and do affect how many permits are issued in any given year (Charnley 2006c). The NWFP 10-year socioeconomic monitoring report described trends in permitted quantities for BLM districts and national forests for the period 1994–2002 (Charnley 2006c); the NWFP 20-year socioeconomic monitoring report described these trends from 2004 through 2012 (Grinspoon et al. 2016). Table 8-4 shows the permit trends for NTFP products during these two periods. Unfortunately, the NTFP data in the 20-year report are presented in a format that does not permit a determination of the trends for a number of product categories. Nevertheless, the products for which a comparison across land ownerships and time is possible, some patterns do emerge. For both BLM lands and national forests, permitted harvest quantities of firewood initially declined and then increased, whereas greenery and foliage showed an upward trend for the entire period. Permitted harvest quantities for wild mushrooms increased on BLM lands through both periods, but on national forests they declined before trending upward between 2004 and 2012.

Based on interviews with specialists on three national forests and one BLM district, Charnley (2006c) identified several ways in which the Plan affected opportunities for the commercial harvest of NTFPs on national forests and BLM-managed lands between 1994 and 2006. Some provisions, such as road closures linked to the Plan's management guidelines, reduced the ability of harvesters to physically access resources. Other provisions, such as guidelines related to the management of late-successional reserves (LSRs) and riparian reserves, resulted in the closure of some areas to legally sanctioned commercial harvesting. Additionally, provisions prohibiting the harvest of special-status plants affected some commercially harvested species. The extent to which the standards and guidelines for LSRs and riparian reserves affected NTFP harvesting depended on how local Forest Service and BLM units interpreted them, and whether they were strictly applied. For example, some forests prohibited commercial harvesting of wild mushrooms in LSRs, while others did not (McLain 2000). Charnley (2006c) concluded that, during the first 10 years of implementation the Plan had the greatest negative impact on the harvesting of firewood and Christmas trees, both of which were previously closely linked to timber harvesting activities. Comparable interview data were not collected for the 20-year report, and consequently it is unclear what factors might account for the observed increases in permitted harvest quantities for firewood and stabilization in Christmas tree permits. Charnley (2006c) pointed out that, over the long term, the most important impact of the NWFP on NTFP resources is likely to be the landscape-scale

Table 8-4—Trends in permitted harvest quantities of nontimber forest products in the Northwest Forest Plan area (1994–2002 and 2004–2012)

	Bureau of Land Management districts		National forests	
Product	**1994–2002**	**2004–2012**	**1994–2002**	**2004–2012**
Fuelwood	-	+	-	+
Christmas trees	-	No data	-	Stable
Cones	-	No data	+	-
Moss	-	No data	Stable	-
Posts and poles	+	+	-	No data
Greenery and foliage	+	+	+	+
Boughs	+	-	Unclear	-
Mushrooms	+	+	-	+
Transplants	+	No data	-	No data

- = negative; + = positive.
Source: Charnley 2006c and Grinspoon et al. 2016.

changes it causes in forest structure and composition, changes that will affect the types, quantities, and qualities of NTFPs present in an area. Whether those impacts are negative or positive, however, depends on what changes in forest conditions have occurred in NTFP harvesting sites, as well as the types of products that are harvested there (Pilz and Molina 2002). The NWFP provisions and fire suppression are expected to encourage the development of older forest structure and processes, with a concomitant decrease in early-seral vegetation. Such conditions favor NTFPs such as matsutake mushrooms and moss, but will likely lead to reductions in the supply of products found in early-seral-stage forests, such as huckleberries, salal, and boughs (Charnley 2006c).

A promising avenue for enhancing the contribution of the NTFP sector to socioeconomic well-being is a forest management approach known as “compatible management” or “joint production.” In this approach, forest stands are managed simultaneously for timber and one or more NTFPs (Alexander et al. 2002, 2011a). For example, in a study comparing three scenarios of timber management, one using a timber management strategy that increased matsutake production, another using a timber management approach with a neutral effect on matsutake productivity, and the third with no timber harvest, Pilz et al. (1999) found that the most lucrative approach was to manage the forest for both timber and matsutake. A joint production approach to federal forest management would have the additional advantage of supporting other goals of the NWFP, including enhancing structural and biological diversity.

Recreation—

The Forest Service and BLM provide opportunities for urban and rural residents to recreate in a wide variety of settings and to participate in a wide variety of recreation activities. Current annual estimates are that 20 million visits take place each year to federal forests in the NWFP area—with 5.3 million to BLM lands and 14.6 (± 5.3 percent) million to Forest Service lands (Grinspoon et al. 2016, USDA FS 2016). Other federal agencies, state and local governments, and private businesses and organizations also provide places to recreate for many of the same individuals. Relative to other providers, the recreation opportunities provided by the Forest Service and BLM are typically farther from population centers and less intensively developed. Chapter 9 includes a detailed description of the amount of recreation use on NWFP-area national forests and common activities of those recreating. This chapter focuses on the economic contributions of recreation activity on federal forests in the NWFP area to local communities.

Recreation on federal forests drives economic activity in local communities, states, and across the NWFP region when recreation visitors spend money on recreation trips, and the agencies and their partners spend money to manage recreation sites. Recreation visitors also support economic activity when they purchase equipment and other durable goods (e.g., boots, binoculars, off-highway vehicles, skis) that they need for particular recreation activities. This spending is not attributable solely to a single recreation opportunity provider (e.g., a single NWFP-area federal forest or all of them combined), and is not discussed here. This section focuses instead on the effects of visitor spending during recreation trips.

The amount of recreation use, the types of trips visitors take, their activities (to a lesser extent), and the size of the local economy all combine to influence how and to what degree recreation visitation leads to private sector employment and business activity (Stynes and White 2006, White and Stynes 2008). The amount of recreation use determines the potential number of visitors who can spend money in an area. All else being equal, a national forest with more recreation use supports more visitor spending in local communities. The type of recreation trip (day trip, overnight trip, near or far from the visitor’s residence) is the key factor in determining recreation visitor spending (White and Stynes 2008). On average, spending by national forest recreation visitors nationwide ranges from $36 per party per trip for visitors on local day trips (trips within 50 miles of their residence), to $580 per party per trip for those on nonlocal (more than 50 miles between residence and destination) overnight trips where lodging is off the national forest (table 8-5). Average spending figures represent both those who spend money and those who do not spend money during the recreation trip. About 12 percent of visits to national forests involve no visitor spending; about 30 percent

Table 8-5—National forest visitor spending profiles for the United States by trip-type segment and spending category, dollars per party per trip[a]

Spending categories	Nonlocal Day	Nonlocal OVN-NF	Nonlocal OVN	Local Day	Local OVN-NF	Local OVN	Non primary	All visits[b]
	Dollars							
Motel	0	44.77	203.85	0	6.39	51.62	139.67	53.96
Camping	0	27.79	13.68	0	28.25	23.01	12.23	7.43
Restaurant	14.77	27.47	116.41	5.66	7.65	32.43	93.23	37.63
Groceries	10.67	55.09	72.52	6.62	71.54	59.62	49.85	29.68
Gas and oil	30.20	62.27	82.47	15.43	46.59	58.05	62.71	38.74
Other transportation	0.58	1.34	4.98	0.16	0.04	1.19	3.35	1.45
Entry fees	4.12	7.13	12.85	2.70	4.51	5.12	7.58	5.38
Recreation and entertainment	2.96	7.36	33.31	1.01	2.01	3.61	21.84	9.38
Sporting goods	3.15	10.77	13.75	3.83	11.78	9.48	7.91	6.62
Souvenirs and other expenses	1.93	7.73	25.87	0.60	1.10	11.48	23.74	8.62
Total	68.39	251.74	579.70	36.00	179.86	255.60	422.12	198.87
Sample size (unweighted)	2,112	3,600	2,289	9,225	1,388	295	3,955	22,864
Standard deviation of total	72	399	714	53	199	325	653	n/a

OVN = overnight, NF = national forest, n/a = not applicable.
[a] Outliers are excluded and exposure weights are applied in estimating spending averages. All figures are expressed in 2014 dollars. These averages exclude visitors who reported that their primary activity was downhill skiing/snowboarding. When completing analyses involving skiers/snowboarders, refer to subsequent tables. Local visitors are those who live within 50 miles of their recreation destination. Nonprimary visitors are those who were away from home to visit family, work, or recreate somewhere else. Their visit to the national forest was secondary to that other purpose.
[b] The all-visit averages are computed as a weighted average of the columns using the national trip segment shares for nondownhill skiing/nonsnowboarding as weights. Source: White 2017.

of visits involve spending of $20 or less. Because the spending averages include nonspenders and low spenders, some average values may appear low relative to typical costs.

Recreation activity has a secondary influence on visitor spending once trip type has been accounted for. For example, the spending of visitors who are downhill skiing or snowmobiling is systematically higher than average; and spending by visitors engaged in backcountry or primitive camping is lower than average (White and Stynes 2008). On average, spending by downhill skiers ranges from $60 per party per trip for local day trips (e.g., a couple who live in Bend, Oregon, and visit Mount Bachelor for morning skiing), to nearly $750 per party per trip for nonlocal overnight trips (table 8-6).

Following the processes outlined in White (2017), we calculate that, in total, recreation visitors to all the NWFP-area national forests combined spend roughly $612.6 million each year in the communities within about 50 miles of those national forests. About one quarter of that spending is generated by visitors engaged in downhill skiing and snowboarding ($156.8 million). Visitors who are hunting, fishing, or viewing wildlife on a national forest spend about $82.1 million in local communities; visitors engaged in other activities (excluding downhill skiing and snowboarding) spend about $374.8 million in local communities each year. Employees and proprietors of businesses that provide goods and services to recreationists receive direct benefits, in the form of income, from recreation visitor expenditures. The majority of expenditures by recreation visitors to NWFP-area forests are made for purchases of lodging and camping, food and beverages in grocery stores and restaurants, and fuel. The Mount Hood National Forest ($95 million), the Deschutes National Forest ($84 million), and the Siuslaw National Forest ($58 million) account for the greatest levels of spending at individual national forests. The presence of ski areas on the Mount Hood and

Table 8-6—Spending profiles of downhill skiers and snowboarders recreating on U.S. national forests, dollars per party per trip[a]

Spending category	Nonlocal segments		Local segments		Nonprimary	All visits[c]
	Day	Overnight	Day	Overnight[b]		
	- - - - - Dollars - - - - -					
Motel	0	193.53	0	88.83	146.10	95.76
Camping	0	0.43	0	0.20	4.23	0.37
Restaurant	20.53	158.80	9.83	72.89	129.36	85.48
Groceries	4.57	76.78	3.21	35.24	68.60	40.21
Gas and oil	24.43	64.96	13.44	29.82	55.28	40.73
Other transportation	0.28	1.89	0.24	0.87	9.78	1.39
Entry fees	37.68	90.73	17.93	41.65	107.20	58.39
Recreation and entertainment	18.62	107.74	11.13	49.45	52.21	58.79
Sporting goods	5.02	26.08	2.81	11.97	22.14	14.73
Souvenirs and other expenses	2.01	22.88	0.68	10.50	12.84	11.69
Total	113.15	743.81	59.26	341.41	607.74	407.54
Sample size (unweighted)	371	431	784	n/a	71	n/a
Standard deviation of total	96	825	81		772	n/a

n/a = not applicable.

[a] Outliers are excluded and exposure weights are applied in estimating spending averages. All figures are expressed in 2014 dollars. These averages are based on visitors who reported that their primary activity was downhill skiing or snowboarding. Analyses involving nonskier/nonsnowboarder visits should refer to previous tables on national forest visitor average spending. For downhill skiers and snowboarders, we have combined the overnight (OVN) national forest and OVN segments into a single OVN segment. Local visitors are those who live within 50 miles of their recreation destination. Nonprimary visitors were away from home to visit family, work, or recreate somewhere else. Their visit to the national forest was secondary to that other purpose.

[b] The sample size for local overnight visitors sampled at ski areas was insufficient, and here we calculate average spending as 46 percent of the nonlocal overnight average.

[c] The all-visit averages are computed as a weighted average of the columns using the national skier/snowboarder segment shares as weights.

Source: White 2017.

Deschutes National Forests helps explain the high levels of recreation expenditures there.

When a recreation visitor buys a good or service, economic activity that starts with the initial purchase spreads out to the broader economy in what is commonly referred to as the "multiplier effect" (e.g., Hjerpe et al. 2017). The size and diversity of other area businesses influence how that additional economic activity spreads within the local region, or leaves the area. Those areas with larger economies, such as Multnomah County near the Mount Hood National Forest or King County near the Mount Baker–Snoqualmie National Forest, will have greater multiplier effects from purchases by recreationists than places with smaller economies, such as Douglas County near the Umpqua National Forest or Skamania County near the Gifford Pinchot National Forest.

Recreation-related economic activity may be affected by climate change as wildfire and forest insect (e.g., bark beetle) activity are expected to increase with a warming climate, potentially leading to impacts on popular hiking and mountain biking areas (Hesseln et al. 2003, 2004; Loomis et al. 2001). Economic activity associated with forest recreation can be expected to decline when forests are closed because of high fire danger or active fire events (Starbuck et al. 2006), or trails or recreation sites are closed following fire events (Sánchez et al. 2016). Negative impacts on recreational quality can last for many years after a wildfire (Englin et al. 2001). However, research from southern California suggests that there can be positive economic effects when a fire creates opportunities for viewing postfire landscape processes (e.g., viewing flowers or new growth) (Sánchez et al. 2016).

Across mountainous regions of the world, alarm has also been expressed regarding possible climate change impacts on the ski industry and associated economic activity (Scott and McBoyle 2007). Potential concerns include a shortened ski season (Lal et al. 2011) as well as changes to avalanche conditions (Lazar and Williams 2008). Other recreational impacts may stem from heavy rainfall events that wash out access roads or otherwise result in flood-related damage (Sample et al. 2014). Climate change will affect multiple recreation-related variables, creating differential impacts depending on region, elevation, and other factors, with some areas potentially benefiting, for example, from longer snow-free seasons or fewer days of extreme cold (Irland et al. 2001, Richardson and Loomis 2004).

Ecosystem services—

In addition to providing the socioeconomic benefits previously discussed, federal forests also provide important ecosystem services both to local communities and more distant urban populations. These include contributions like fresh water, food and fiber, wildlife habitat, and outdoor recreation opportunities, to name a few (fig. 8-20). DellaSala et al. (2011), for example, noted substantial economic and ecological benefits associated with clean water that originates from national forests of the Western United States, and in particular from roadless areas, where timber harvest is prohibited. The importance of national forests for supplying surface drinking water in the NWFP area has been mapped,[17] but the economic value of this contribution

[17] https://www.fs.fed.us/ecosystemservices/FS_Efforts/forests-2faucets.shtml.

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Figure 8-20—Federal forests provide many ecosystem services, including clean water and fish and wildlife habitat.

has not been calculated. Brandt et al. (2014) identified several ecosystem services associated with Pacific Northwest forests, including timber harvesting, salmon populations, carbon storage in vegetation, soil organic matter, and landscape aesthetics. Many ecosystem services considered to be amenities (e.g., scenic views, recreation opportunities) contribute to rural residents' quality of life (e.g., Deller et al. 2001, Rudzitis and Johnson 2000), as well as attract inmigration of new residents (e.g., Gosnell and Abrams 2011, McGranahan 1999).

The past decade has seen significant and increasing effort among state and federal agencies, nongovernmental organizations, and others to identify and evaluate ecosystem services associated with various landscapes, including forests (e.g., Kline and Mazzotta 2012, Kline et al. 2013, Smith et al. 2011). There also has been increasing interest in developing and implementing policy instruments that provide monetary compensation to private forest landowners who produce particular ecosystem services, including direct payment programs, tax incentives, and ecosystem services markets, among others (e.g., Kline et al. 2000a, 2000b, 2009).

Within the Forest Service, the 2012 planning rule formally incorporated the concept of ecosystem services into national forest management and requires forest personnel to address ecosystem services as they prepare national forest plan revisions (USDA FS 2012). More recently, the Obama administration directed all federal agencies to consider ecosystem services values in federal planning and decision-making (Donovan et al. 2015), inducing agencies to develop methods and protocols for evaluating ecosystem services as outcomes of federal policies, programs, and agency performance. There also have been efforts to examine the potential for developing partnerships with nonfederal entities that may be willing to provide funding to assist in federal land management when it produces mutual benefits, such as restoration on federal lands that improve municipal watersheds (e.g., McCarthy 2014).

Within the NWFP area, federal efforts largely have focused on identifying and quantifying key ecosystem services produced from the region's national forests (e.g., Smith et al. 2011). In addition to characterizing biophysical ecosystem services such as water, habitat, food, and fiber, efforts also have included improving understanding of cultural ecosystem services associated with national forests and their importance to Pacific Northwest residents (e.g., Asah et al. 2012). Landscape modeling efforts have attempted to characterize tradeoffs among ecosystem services associated with alternative forest management regimes. For example, Kline et al. (2016) examined the potential for Pacific Northwest forests to store and sequester additional carbon, harvest timber, and retain/enhance habitat for seven focal wildlife species across an exhaustive array of management regimes for western Cascade Range forest landscapes. Results showed the levels of each ecosystem service produced under each management regime, as well as the tradeoffs among them from choosing one management regime over another. Northern spotted owl habitat was found to be complementary with stored carbon, with both generally increasing in older forests. Northern spotted owl habitat and timber harvest were found to range from largely competitive to neutral depending on the characteristics of the management regime examined. Joint production relationships involving northern spotted owl habitat and other wildlife species ranged from competitive for western bluebird to mostly neutral for Pacific marten, and complementary for the olive-sided flycatcher and red tree vole, depending on the differences or similarities in the forest conditions preferred by individual species (Kline et al. 2016).

Last, within the NWFP area there has been analysis of the willingness of nonindustrial private forest landowners to accept direct payments in return for agreeing to lengthen timber rotations to improve habitat for spotted owls (Kline et al. 2000b) and coho salmon (Kline et al. 2000a). Kline et al. (2000b), for example, suggested that many forest land owners would require little or no payment to forego harvest to improve habitat, while others would require a significant incentive.

Increasing recognition of ecosystem services by federal land management agencies can be viewed as an extension of the multiple-use approach toward more earnest consideration of the diversity of uses and values derived from national forests, and to a broader coalition of public parties interested in federal land management (Kline et al. 2013). Although efforts to identify and quantify key ecosystem services have made significant progress in raising awareness and concern for these important forest benefits, formal methods for routinely including ecosystem services values into federal forest management are being developed by the Forest Service and BLM. Formally incorporating ecosystem services concepts into federal land management processes generally requires information about: (1) current landscape conditions and how they are changing; (2) how management activities likely will affect ecosystem services; and (3) what people value about the landscape, how much they value those things, and how their values might be changing (Kline and Mazzotta 2012). Meeting these informational requirements depends on addressing various methodological challenges involving the availability of ecological data and analytical models for describing the responses of ecosystem services to management, as well as adequate staffing for conducting such analysis (Kline et al. 2013). Federal directives (e.g., Donovan et al. 2015, USDA FS 2012) suggest that efforts to develop and improve methods for evaluating ecosystem services and including them in federal land management will continue as policymakers and the public increasingly recognize the importance of addressing these benefits in federal decisionmaking.

How Rural Communities Contribute to Federal Forest Management

Summary—

Just as forest management can contribute to socioeconomic well-being in rural communities, so can rural communities contribute to federal forest management. Agency budgets have been reduced substantially since the NWFP was implemented, reducing agency capacity to accomplish forest management goals. In response, community-based groups and partner organizations have raised money and provided labor to help undertake forest work on federal lands. Wood processing infrastructure in communities has also declined throughout the Plan area since the 1980s, making timber sales less economical and creating a financial barrier to restoration. By working together, communities and federal land management agencies in the Plan area can develop strategies to support and maintain the business infrastructure needed for forest restoration while creating more local economic opportunities.

The community forestry literature from the United States emphasizes the reciprocal relationship between healthy forests and healthy communities (Baker and Kusel 2003, Kelly and Bliss 2009, Kusel and Adler 2003). Just as federal forest management can contribute to community well-being, so can communities contribute to federal forest management. For example, many communities and national forest units have begun to plan over large spatial scales and long time frames to create the consistency of work needed to attract investments in processing and contracting capacity (Schultz et al. 2012). Doing so provides both a more predictable employment base in local communities and the business capacity required to accomplish forest restoration.

Agency budgets, and the number of agency employees and field offices, have dropped substantially since the NWFP was implemented, particularly for the Forest Service and especially in its Pacific Northwest Region (Grinspoon et al. 2016, Stuart 2006). These declines have reduced agency capacity to undertake forest restoration and other forest management work. One way in which the Forest Service has dealt with declines in budget and personnel is through outsourcing work to contractors, partners, or volunteers. For example, Seekamp et al. (2011) identified 35 different types of recreation partnerships that the Forest Service engages in to help accomplish recreation-related work on national forests nationwide. Partners range from individual volunteers to service organizations, commercial outfitters, and other government agencies (fig. 8-21). Community-based organizations, local business partners, environmental and recreation organizations, and other groups have helped raise money and provide labor to accomplish forest management goals

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Figure 8-21—A partnership between the Six Rivers National Forest and the California Conservation Corps makes it possible to accomplish trail work on the national forest.

on federal lands in the face of declining agency capacity to do so, filling critical gaps. But communities must have an interest in and capacity to provide support, which is linked to their assets and overall community health and well-being.

There are several such examples from the NWFP area. On the Siuslaw National Forest in Oregon, local partner organizations formed the Siuslaw Stewardship Group in the early 2000s (Sundstrom and Sundstrum 2014). The group has worked with the Forest Service to facilitate forest restoration on private and public lands in the Siuslaw watershed by pooling resources, assisting with monitoring activities, and cooperating in work activities by using stewardship contracts and the Wyden Amendment Authority (which allows federal dollars to pay for work on private lands in shared watersheds to protect and restore resources or reduce natural disaster risk), while contributing to community economic health and avoiding legal conflict over treatments (Sundstrom and Sundstrom 2014). In California, the Trinity County Resource Conservation District has been managing a stewardship agreement on the "Weaverville Community Forest," comprised of 12,000 ac (4856.2 ha) of the Shasta-Trinity National Forest and 1,000 ac (404.7 ha) of the BLM's Redding Field Office lands (Frost 2014). Their objective is to develop and implement forest management activities that meet local objectives while addressing forest health concerns. The community plays a central management role, recruits skilled local workers to accomplish restoration activities, and contributes financial support by leveraging money from other federal and state partners to help fund new projects in the community forest (Frost 2014).

In another example on the Shasta-Trinity National Forest, the Watershed Resource and Training Center has filled a number of institutional voids to help accomplish forest management activities while creating local jobs (Abrams et al. 2015). These include job training to create a skilled local workforce to engage in ecosystem management and forest restoration activities, running a work crew to accomplish fuels reduction on federal and private lands, monitoring of projects, developing new local wood processing infrastructure, helping the Shasta-Trinity to develop stewardship projects, developing a community wildfire protection plan, and leading interdisciplinary project planning teams. Despite the fact that some community-based organizations such as these have innovated to fill in the gaps left by declining federal agency capacity, there are legal and economic limits to what these organizations can accomplish, and they may also be limited by their own internal organizational capacity (Abrams et al. 2015). In all these examples, external organizations help provide funding and labor to accomplish work on federal forests that the agencies do not have sufficient budgets or staffing to undertake.

An important way in which economically healthy communities contribute to ecologically healthy forests is by having a skilled workforce and the business infrastructure needed to help federal agencies accomplish their management goals. As noted previously, declines in local wood processing infrastructure accompanied declines in timber production from federal lands in the NWFP area. Not only did this decline adversely affect some Plan-area communities, lack of local infrastructure for processing timber and small-diameter wood make timber sales and removal of small-diameter material that constitutes hazardous fuels less economical, creating a financial barrier to forest restoration. For example, Nielsen-Pincus et al. (2013) found that national forest ranger districts in Oregon and Washington that were within a 40-minute drive to a sawmill or biomass facility treated more overall hectares, and more hectares in the WUI, for hazardous fuels reduction than did ranger districts that were farther away. Ranger districts that were close to these facilities also incorporated more biomass into their treatments. These findings underscore some of the interdependencies between healthy forests and healthy communities in the NWFP area.

The Implications of Land Use and Ownership Changes for Forest Management

Summary—
Changes in land use and ownership, particularly those that involve conversions of forest land to low-density and urban development, are likely to remain a significant factor affecting the NWFP area owing to population growth in the region. Loss of forest land to development, associated fragmentation of the remaining forest land base, and accompanying changes in how remaining private forest lands are managed suggest that policymakers and managers cannot assume that the forest land surrounding federal lands will be the same in coming decades and available to contribute to NWFP objectives.

In addition to its significant area of federal and other public lands, the NWFP area includes a notable private land base. Nonfederal lands totaled more than 11 million ac (4.45 million ha) in 2009 in western Oregon, or about 57 percent of all land in the region (Lettman 2011). Sixty-five percent of nonfederal land in western Oregon was forest, with the remainder divided between mixed forest and agriculture, agriculture, and low-density and urban development (fig. 8-22). In western Washington, nonfederal lands totaled more than 10 million ac (4.05 million ha) in 2006, or about 65 percent of all land (Gray et al. 2013). Seventy percent was forest, with the remainder in mixed forest and agriculture, agriculture, and low-density and urban development (fig. 8-23). Significant private forest lands also exist in northern California (Waddell and Bassett 1996, 1997), with nonfederal lands comprising 48 percent of all forest land in NWFP-area counties in California (Christensen et al. 2015). Private forest lands, including both industry- and nonindustry-owned, often augment federal and other public lands in providing ecosystem services (Kline et al. 2004a), including habitat for at-risk wildlife species (Stein et al. 2010; see also chapters 5 and 7). However, private lands also often differ from federal and other public lands in their forest structural

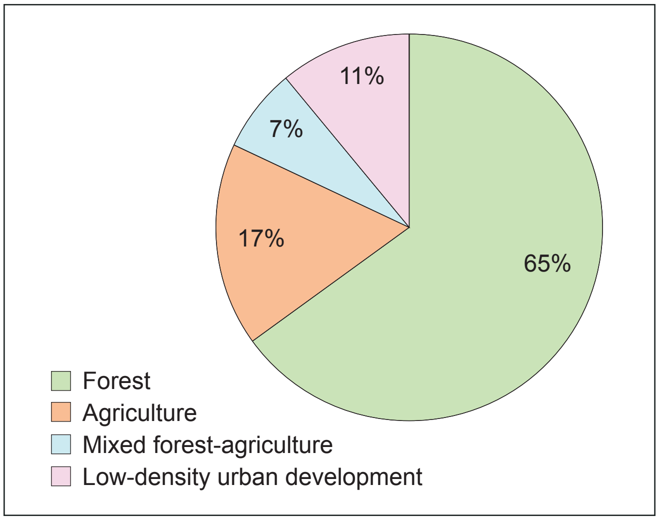

Figure 8-22—Land use of nonfederal lands in western Oregon (11 million ac [4.45 million ha]). Source: Lettman 2011.

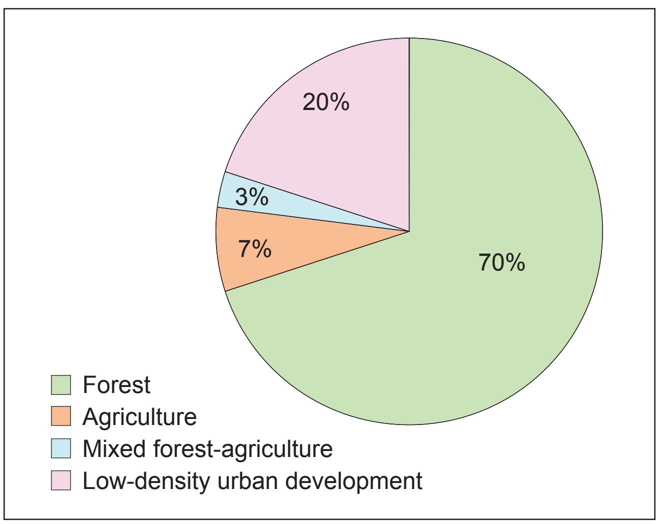

Figure 8-23—Land use of nonfederal lands in western Washington (more than 10 million ac [4.05 million ha]). Source: Gray et al. 2013.

attributes, with potential implications for habitat and other resource issues (Azuma et al. 2014). Although the public land area generally will remain constant for the foreseeable future, private forest lands are subject to possible conversions to other nonforest land uses, including agricultural, residential, commercial, and industrial development associated with population growth in the region. Federal and other public lands also can attract development on adjacent private lands, potentially leading to increased road densities, more human-caused wildfire ignitions, and greater demands for recreation, among other changes affecting federal lands (e.g., Azuma et al. 2013). The uneven distributions of ecosystems, ownerships and management activities across the NWFP area is one reason why it may be difficult to meet diverse biodiversity objectives on federal lands alone (Spies et al. 2007)

Forest land/agriculture conversions—
Within the NWFP area, actual conversions of private forest land to agriculture (and vice versa) are limited. Forest land conversions to agriculture totaled 9,000 ac (3642 ha) from 1974 to 2009 in the entire state of Oregon, relative to a nonfederal land base of nearly 29 million ac (11.74 million ha), while conversions from agriculture to forest land totaled 3,000 ac (1214 ha) (Lettman 2011). Similarly, net conversions from forest land to agriculture totaled just 1,761 ac (713 ha) in western Washington between 1976 and 2006, out of a nonfederal land base of more than 10 million ac (4.05 million ha) (Gray et al. 2013). This stability between forest and agricultural land uses stems largely from the unsuitability of existing forest land for agriculture because of soils and topography, and the high income-earning capacity of lands currently in agricultural uses relative to forestry.

Conversion of private forest land to more developed uses—
More prevalent are conversions of private forest land to residential, commercial, industrial, and other developed uses (fig. 8-24). Private forest land conversions to development in Oregon totaled 172,000 ac (69 606 ha) from 1974 to 2009, or about 2 percent of the nonfederal forest land statewide during this period, with 163,000 ac (65 964 ha) (95 percent of this total) involving conversions to low-density residential development, and the remaining 5 percent (9,000 ac) (3642 ha) involving urban development (Lettman 2011). These changes have been most prevalent in urbanizing regions along Oregon's Interstate 5 corridor (Lettman 2011).

Similarly, forest land development totaled 479,324 ac (193 976 ha) in western Washington between 1976 and 2006, or about 6 percent of the nonfederal forest land in western Washington. Of this total, 419,678 ac (169 838 ha) (88 percent) were converted to low-density residential development, and 59,646 ac (24 137 ha) (12 percent) to urban development (Gray et al. 2013). Population densities

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Figure 8-24—Conversion of private forest land to residential development, Oregon.

have more than doubled in the Puget Sound region in recent decades, contributing to significant urban expansion onto forest land (Alig and White 2007). In some northwestern Washington counties, population increase owing to net domestic migration was more than double the natural increase in population during the 1990s, with associated increases in forest land development (White and Mazza 2008). Land use data suggest that development has been increasing on private lands adjacent to federal and other public lands, particularly in selected counties of western Washington and on the eastern slope of the Cascade Range in Deschutes County, Oregon (Azuma et al. 2013).

National-level projections based on expected population growth suggest continued loss of forest land to development through 2030 in northern California and the Pacific Northwest, largely following national patterns of development near existing urban areas (Stein et al. 2005, 2009). Regional projections of future low-density residential and urban development on forest land in western Oregon through 2024 are fairly modest largely owing to Oregon's land use planning program, with most conversions involving the transition of low-density developed forest land to urban uses (Kline 2005b). In eastern Oregon, forest land development also is projected to be fairly modest through 2025, with most conversions involving low-density to largely urban transitions (Kline et al. 2007). In western Washington, forest land was projected to decline by 8 percent from 1997 to 2027, with most converting to urban development (Alig and White 2007). However, projections in western Washington do not consider the potential conservation influence of Washington's land use planning program (implemented in 1990), which early analysis is suggesting may be beginning to have some effect on slowing development on both forest and agricultural lands (Kline et al. 2014). Development is expected to be most prevalent in valleys near urban areas, based on analyses conducted for western Oregon (Kline at al. 2003) and western Washington (Kline et al. 2009). Similar patterns also are reflected in analysis of western Oregon and western Washington combined, with greater loss of forest land expected through 2040 in the Puget lowlands and

Willamette Valley relative to the Coast Range and Cascades regions (Lewis and Alig 2014). We are unaware of regional-level land use projections for northern California.

Forest land development largely results from market forces. Population growth and inmigration, rising incomes, and economic growth over time combine to increase demands for land in developed uses (Kline et al. 2004a). Demands also increase with people's lifestyle choices when, for example, people relocate to rural areas or desire second homes in scenic forest settings. When demands for developed land uses increase, forest landowners may be able to earn more by selling their land than they can by maintaining it as forest (Kline et al. 2004a). When these market forces are at play, some loss of forest land to development is inevitable. Research also suggests that these trends can influence the degree to which forest landowners continue to perceive forestry and forest ownership as a worthwhile endeavor (Creighton et al. 2016). The combined influence of various socioeconomic factors on land use change largely has been confirmed in the Pacific Northwest from econometric land use modeling and analysis conducted at the county level (e.g., Parks and Murray 1994) and at finer spatial scales (Kline 2003; Kline and Alig 2001; Kline et al. 2001, 2003, 2007, 2009). Additionally, fine-scaled models, based on geocoded point data (e.g., Gray et al. 2013, Lettman 2011), suggest that location and natural amenity factors also play a role. Land use modeling for western Oregon, for example, found a positive correlation between development and the proximity of land to the Interstate 5 corridor and the Pacific Coast (Kline and Alig 2001, Kline et al. 2001). Analysis for the eastern slope of the Oregon Cascades found a positive correlation between development and the presence of scenic mountain views (Kline et al. 2007).

In general, conversions of forest land to development in both Oregon and Washington have been more common on private nonindustrial lands than on industry-owned lands (Lettman 2013). The area of timber industry-owned forest land has remained fairly constant in both Oregon and Washington since the mid-1970s, while the area of forest land in each state owned by nonindustrial owners has declined by 6 percent and 10 percent, respectively (Lettman 2013). We are unaware of studies addressing forest land development in northern California. Analysis and projection of future changes in forest land ownership has been hampered by a lack of data describing land ownership over time that spatially and temporally aligns with land use data sets developed for the region (e.g., Gray et al. 2013). Thus, knowledge of anticipated changes in land ownership tends to derive from predictions about which land ownerships are most likely to be involved in projected future land use changes (e.g., development), rather than predictions about potential future changes in ownership. For example, landscape-level modeling and projections for the Coast Range physiographic province of Oregon has suggested that forest land development could reduce industry-owned forest land by 6 percent, and nonindustry-owned forest land by 35 percent by 2096, with the greatest reductions near urbanizing Portland, Oregon (Johnson et al. 2007). Such reductions generally are not as likely to involve the most commercially productive industry-owned timber lands in the region, largely because of their relative geographic isolation from urbanizing locations where development will be prevalent owing to greater proximity to urban areas and transportation corridors (Kline and Alig 2005).

In addition to concern about the loss of forest land to development and its potential ecological impact, are concerns about how development often brings greater numbers of homes into dry, fire-prone forest types, expanding the WUI. In addition to the various land-use projection efforts previously mentioned (e.g., Kline et al. 2003, 2007, 2009), which can be used to anticipate future expansion of the WUI within the Plan area, are other regional and national efforts to define the current WUI and anticipate its future growth (e.g., Hammer et al. 2007). Such expansion likely will present future challenges to public land managers who will need to consider how to expend limited wildfire management funds to meet potentially competing objectives, including managing for ecological integrity and resilience to climate change, and habitat for species such as the northern spotted owl versus mitigating wildfire risk to homes.

Timber investment management organizations and real estate investment management trusts—

A growing interest nationally in recent years involves the seeming rise in forest land ownership of timber investment management organizations (TIMOs) and real estate investment trusts (REITs), as they purchase forest parcels previously held by more traditional timber industry owners. Forest policymakers, for example, question whether TIMOs will continue to manage their holdings for long-term timber production versus eventual development (Lettman 2013). Whereas timber industry owners are perceived by policymakers as focused solely on securing an expected flow of timber revenue over the long term via active forest management, TIMOs and REITs are perceived as less committed to solely managing forests over the long term, and more amenable to other ways of generating income, including development (Lettman 2013). The NWFP area, however, has seen little research regarding how prevalent these forest land owners have become in recent years, their potential future trends, and whether and how their management of forest land holdings might change. Although TIMOs and REITs have been involved in several large acquisitions of previous industrial forest land in both Oregon and Washington (Lettman 2013), what this means for future management of such holdings as well as longer-term forest land ownership trends within the Plan area remains uncertain. Additionally, given that TIMOs and REITs typically do not own and operate wood processing facilities, it is conceivable that their increased forest land ownership in the Pacific Northwest could be accompanied by increases in log exports. Such changes potentially could increase the importance of federal timber harvests in supporting timber-related economic activity within the region.

Land use planning—
An additional and potentially significant influencing factor in both the pace and pattern of forest land development within the Plan area is land use planning, which restricts developed uses on private lands to promote efficient land use and secure various conservation benefits. Oregon's land use planning program—often cited as a national model for statewide planning (Kline and Alig 1999)—has provided a measurable degree of protection of forest and agricultural lands since its inception in 1973 (Gosnell et al. 2011), with an estimated 1.4 percent of the private forest land base saved from development by 1994 that otherwise would have been developed without land use planning in effect (Cathcart et al. 2007, Kline 2005a). Land use projections suggest that the Oregon land use planning program will continue to conserve forest land in the future, totaling 315,000 ac (127 476 ha) (4.4 percent) between 2004 and 2024 (Kline 2005b). Although less studied than Oregon's land use planning law, research suggests that Washington's land use planning program also has had some effect at reducing development of private forest land since its implementation in 1990 (Kline et al. 2014). To our knowledge, land-use planning effects on conserving forest land in California have not been examined. Additional public land use policies, including most notably preferential property tax assessment, also likely influence land use changes within the Plan area, but we are unaware of any studies addressing these.

Land use change and fragmented forests—
Secondary to the direct impact that development can have on reducing the total area of forest land is the role it plays in fragmenting remaining forest land. For example, as the area of forest land in western Washington has declined, it has become more fragmented, with greater edge to interior portions and smaller patch sizes (Gray 2013). Forest fragmentation can have implications for wildlife habitat and other ecosystem services, as well as influence how remaining forest lands are managed. For example, forest land development has been linked to loss of forest cover and associated declines in coho salmon populations in rivers feeding the northern Puget Sound (Bilby and Mollot 2008), as well as degradation of stream conditions and fisheries generally owing to declines in vegetation and increased area of impervious surfaces (Morley and Karr 2002). Azuma et al. (2014) suggested that even small amounts of development can lead to meaningful changes in forest conditions on both private lands and lands adjacent to federal and other public lands, including increases in invasive species.

Increased use of fine-scale spatial land use modeling (e.g., Kline et al. 2003) versus county-level models (e.g., Parks and Murray 1994) in recent years has enabled greater consideration of how future development is likely to affect specific ecosystems and habitats. For example, development in western Washington is expected to be more prevalent on level or moderately sloped lands and nearer to existing urban areas (Kline et al. 2009). Similar patterns are projected in western Oregon, with development expected to have a greater impact on oak woodland habitat along the Willamette Valley perimeter than on the coniferous forests of the western Cascades and Coast Ranges (Kline and Alig 2005). In the Coast Range physiographic province of Oregon, development is expected to occur more frequently on gently sloping valley bottoms (Spies et al. 2007), including high intrinsic-potential coho salmon streams (Burnett et al. 2007). On the eastern slope of the Oregon Cascades, projected development is expected to adversely affect habitat connectivity for mule deer, potentially impeding animal movement for winter foraging (Kline et al. 2010). National-level analysis has identified significant numbers of at-risk species on corporate-owned lands in select watersheds in coastal areas of northern California, southern Oregon, and Washington (Stein et al. 2010).

Forest fragmentation resulting from development also has been found to be accompanied by changes in how remaining private forest lands are managed. Research from western Oregon found that increasing building densities on private forest land were associated with lower forest stocking rates as well as reduced precommercial thinning and tree planting following harvest (Kline et al. 2004b). This contrasts with similar research conducted for eastern Oregon, which suggested that development had not significantly influenced private forest management owing largely to the relatively lower rates of development, among other factors (Kline and Azuma 2007). Modest rates of forest land development throughout western Oregon are projected to lead to additional reductions in active forest management for commercial purposes at least through 2054 (Kline and Alig 2005). Such changes are thought to arise, in part, from forest fragmentation (or parcelization), which breaks up large forest parcels into smaller parcels for development, thereby increasing the cost of active forest management. Additional research suggests that private landowners of smaller forest land parcels tend to manage less for commercial timber production and more for recreation, aesthetics, and other passive-use values (Kline et al. 2000a, 2000b). There also is emerging evidence suggesting that private forest landowners may have different perspectives and approaches to managing wildfire risk than do federal land managers (e.g., Charnley et al. 2017). Such changes in private landowner objectives and perspectives potentially offer opportunities for enlisting private landowners in landscape-level conservation and wildfire management efforts, possibly through financial incentives, education, and technical assistance (Fischer et al. 2014; Kline et al. 2000a, 2000b).

Research Needs, Uncertainties, Information Gaps, and Limitations

The science synthesis presented in this chapter is necessarily limited by information gaps stemming from lack of available science to adequately answer the guiding questions. Here we identify research needs that could help fill some of these gaps.

The Wood Products Industry

There is increasing recognition that federal forest management, especially forest and watershed restoration, should be done at the landscape scale and across land ownerships to ensure better outcomes. Concurrently, there is recognition that forest management and the production of ecosystem services take place within complex social-ecological systems (chapter 12) in which management outcomes are influenced by both social and ecological conditions, which are linked and which interact to influence one another. Further, these social-ecological systems are characterized by complexities such as time-lagged effects, tipping points that yield dramatic changes over short periods of time, and spatial connectivity. Much of the landscape-level modeling conducted within the Plan area is now decades old or has not fully accounted for the linked social-ecological system dynamics that influence forest management. New research that recognizes and quantifies these dynamics, and that simulates landscape-level management over long time frames, is needed to better understand potential futures and tradeoffs in the production of ecosystem services under alternative management regimes within the Plan area. Such research could provide insight into whether the availability of federal timber for harvest will continue to change in coming decades, and how federal timber production might affect other values associated with federal forests.

Global competition, technological change, consumer demand, and other factors unrelated to federal timber supply all influence wood products manufacturers in the Plan area. In Oregon, there has been recent interest among policymakers and the business community in mass-timber buildings as a potential new market for wood products manufacturers. Mass-timber buildings (which are often multistory and use large panels and columns constructed from wood rather than concrete or steel) are proposed, or under construction, in Portland, Oregon, and an Oregon manufacturer has begun producing mass-timber panels. Additional research is needed to identify products for which wood products manufacturers in the Plan area may have a competitive advantage, given the realities of global markets for commodity wood products such as dimension lumber and structural panels.

Community Socioeconomic Well-Being

Land managers have expressed interest in how socioeconomic well-being in the Plan area has changed since the NWFP was implemented. In this chapter, we have described general trajectories of change in forest communities, characterizing these trajectories according to certain archetypes. We do not know how many communities in the Plan area fall into each type, the geographical distribution of different community types, or the extent and nature of hybrid types ("multifunctional") communities, although typologies have been developed and mapped at the county scale. Existing studies that rely on a small handful of indicators from secondary data sources, such as the U.S. Census, are insufficient for fully understanding change in the region, and how it may be linked to federal forest management as one driver of change. An assessment of community types in the Plan area could help managers better understand how communities have been changing, and how management actions could be tailored in different places to provide different types of local community benefits. Adding to this, NWFP socioeconomic monitoring during the first decade provided a rich characterization of the impacts of the Plan on rural communities, and how they were adapting to changes in federal forest management. NWFP socioeconomic monitoring during the second decade focused on change at the county scale, and relied solely on secondary data from existing sources. Community studies that include primary data gathering directly from community residents would provide a much richer understanding of how socioeconomic well-being in the Plan area has changed over time, and its links to federal forest management. Currently, there is a paucity of community-level studies from NWFP-area communities.

Forest Service Contracting

Climate change promises to further complicate the relationships among wildfire, federal spending, and community benefits. On the one hand, communities with higher levels of fire suppression contracting infrastructure may benefit economically from increases in fire frequency and extent, owing to increased economic activity associated with more fire suppression. On the other hand, increasingly nationalized and mobile fire suppression response means that local fire suppression capacity (e.g. trained crews and equipment) may be elsewhere when a fire strikes, and therefore unable to support local suppression efforts (thus requiring dispatch to call upon crews from outside the local area). Additionally, communities may experience economic challenges in the months following a wildfire despite an initial increase in economic activity associated with firefighting (Davis et al. 2014, Nielsen-Pincus et al. 2014). Forest-specific climate adaptation strategies for the region identify the need for active management to make forests more resilient to wildfire and climate-change effects, and undertake other stewardship activities (chapter 2) (Spies et al. 2010, Whitely Binder et al. 2010), all of which imply potential contracting opportunities for local communities. The lack of historical analysis of forest restoration and fire suppression contracting leads to many uncertainties in understanding the future of such contracting work, or the linkages between restoration and fire suppression contracting. Much of the research to date has focused either on very specific geographies and case studies, or on more regional data and trends. In addition, the challenges facing restoration contractors and fire suppression contractors differ, not only in the contracting and dispatching protocols, but also in the scale at which the work is conducted.

Additional research focused specifically on understanding the businesses that engage with federal agency contracting (restoration service, timber sales, and fire suppression) would provide a more comprehensive understanding of the overlap and linkages between these businesses, as well as the communities to which they are connected and their local impacts.

Within the confines of timber sale and contracting requirements, the Forest Service has a number of innovative tools available to enter into partnerships, agreements, and stewardship contracts with private businesses and nongovernmental organizations. These innovative tools can be used to accomplish a variety of natural resource projects, produce a range of ecosystem goods and services, and bolster the performance of both the agency and the cooperating entity. Much of the recent research on the use of innovative tools in the Pacific Northwest has taken place in dry forests, east of the Cascades. Additional research is needed within the NWFP area on how the connections between the Forest Service and local communities can be strengthened through the use of such tools. In addition, the Plan area has been a source for experimentation with new models of natural resource governance (Montgomery 2013), including models in which community-based organizations fill in for gaps in federal capacity (Abrams et al. 2015). It remains to be seen how the evolution of these new institutional arrangements will affect contracting activities and the spatial distribution of benefits from Forest Service contracting.

Biomass

Much is still unknown regarding the potential for biomass energy production and related ecosystem service work to support rural communities in the future. Doing so will depend on the details of renewable energy, climate change, and ecosystem service-oriented policies and markets. Various climate change mitigation or adaptation initiatives may provide incentives and support for forest biomass production and use. For example, programs to increase the production of energy from non-fossil-fuel sources could increase demand for forest-based biomass materials and outputs. However, uncertainties remain regarding the carbon benefits of forest biomass energy (Hudiburg et al. 2011, Nechodom et al. 2008, Ter-Mikaelian et al. 2015), raising the possibility that biomass may not continue to be favored as part of a low-carbon energy portfolio. Further, the feasibility of biomass as a complement to forest stewardship and as a contributor to rural development is challenged by current harvest, transportation, and processing costs and the low demand for biomass materials; this scenario could change with new markets, subsidies, or biomass-based products (Crandall et al. 2017). Research is needed to better understand the full suite of costs and benefits associated with biomass energy development under different market and public policy scenarios, and to understand where and under what conditions biomass harvesting may help to complement other forest management activities or contribute to a low-carbon energy matrix. Additional research could also help to clarify how the interactions of various energy and non-energy policies influence the development of biomass businesses (Abrams et al. 2017, Becker et al. 2011b).

Nontimber Forest Products

Nontimber forest products on federal forests support community and household well-being by providing income-earning opportunities in the formal and informal economic sectors, strengthening individual and community social capital, facilitating intergenerational ecological knowledge transfer, and enabling NTFP practitioners to develop stronger connections with nature and improve their mental and physical health. Research conducted in the previous two decades has begun to reveal some of the diverse and complex ways in which NTFPs contribute to human well-being, but there is much more to be learned (fig. 8-25). Specifically, we know very little about even some of the most basic social, economic, and ecological aspects of NTFPs, such as:

1. Who is harvesting NTFPs and what are their motivations for harvesting these products? To what extent do urban, as well as rural, residents participate in NTFP-related activities?
2. Where are harvesters getting NTFPs from and how much are they actually harvesting?
3. How does the spatial and temporal distribution of NTFP activities vary within and across seasons?

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Figure 8-25—Much remains to be learned about the harvesting of even the most important nontimber forest products in the Northwest Forest Plan area, such as wild mushrooms and firewood.

4. What are the cumulative impacts of agency regulations such as large-scale area closures, permit requirements, seasonal restrictions, etc. on NTFP livelihoods?
5. What are the ecological impacts (positive and negative) of NTFP harvesting? And what are the impacts of different vegetation management and restoration practices on NTFP species and livelihoods? What active management approaches can be adopted to enhance the productivity of different NTFPs, while also producing timber?
6. How is climate change likely to affect the location, quantities, and qualities of NTFP species? What adaptive measures can be taken to ensure the viability of NTFP livelihoods in the face of changing climatic conditions?
7. What do informal and formal NTFP value chains look like, and how are benefits distributed along those value chains? How do permit prices align with the costs incurred by harvesters?
8. What methods exist or could be developed for measuring the contribution to community well-being of NTFP activities taking place outside the market place, and how can these be adapted for research on NTFP activities in the Plan region? How can the recreational, cultural, and provisioning values of NTFPs best be assessed?

Additionally, most of the research on NTFPs in the Plan region has focused on the "big three"—floral and holiday greens, wild edible fungi, and huckleberries. No studies have been done of firewood, which provides the bulk of NTFP revenues on many national forests and serves as a heating source for many rural residents. Little is known about the native seed and transplant industries, which play a major role in restoration on both federal and private lands. Likewise, little is known about the social and economic aspects of medicinal plant gathering on federal forests in the NWFP region, yet the medicinal plant industry is one of the largest and fastest growing NTFP sectors.

The biggest gains in knowledge about NTFPs in the NWFP region and the people who rely upon them for their livelihoods, enjoyment, and cultural traditions were made between 1990 and 2010, thanks in large part to the Pacific Northwest Research Station's interdisciplinary applied research program focused on improving understanding of the social, economic, and ecological aspects of NTFPs. A key take-home message from that experience is that building and strengthening partnerships, both across academic disciplines and among scientists, managers, and NTFP harvesters/buyers, is likely the key to the development of a program of NTFP research that can enhance socioecological resiliency and community well-being in the NWFP region.

Recreation

Recreation opportunities on federal forests support the well-being of local communities by providing leisure opportunities for local residents and by attracting visitors who spend money in local communities during their recreational trips. Research is generally clear on what communities can do to promote greater visitor spending, such as providing lodging opportunities, restaurants, and recreation services. There is limited research within the Plan area on how federal forest resource conditions and management influence recreation use and recreation behavior of local residents and visitors. More research is needed to understand how management actions across the landscape, and at important resource destinations, influence how people use forests for recreation.

Ecosystem Services

Given the degree of contentious debate that motivated the NWFP and that has been inspired by it over the years, it is surprising that little analysis has addressed the potential net co-benefits associated with the Plan. Specifically, what has the NWFP meant in terms of water quality, outdoor recreation, and habitat for species other than the spotted owl? Quantifying these possible net co-benefits, even approximately, might offer additional information with which to more fully evaluate the long-term effects of the Plan. Future research could be directed toward characterizing how the NWFP has influenced various ecosystem services, building on case studies and approaches in development (e.g., Kline and Mazzotta 2012, Smith et al. 2011).

Additional research could be directed toward further evaluating the degree to which various policy instruments, including direct payments, tax incentives, and ecosystem services markets, could be used to provide incentives to private landowners to conduct actions that pursue NWFP goals on private lands, augmenting current efforts on federal lands. In the early 2000s, for example, there was significant excitement about the expected development of markets for nontimber ecosystem goods and services that are produced from forests (e.g., carbon storage, water quality improvements) (e.g., Kline et al. 2009). However, achieving these expectations has been spotty within the NWFP area, in part because to effectively implement them, such markets require new or tighter environmental regulations restricting actions that damage ecosystem goods and services, making such markets difficult to establish (Kline et al. 2009). Despite limited success thus far, the presence of a carbon market in California and other cases in Oregon and Washington provide some promise that such markets can provide additional revenue streams from private forests. But how, and if, public forests can contribute to carbon markets and other ecosystem service markets remains largely unknown. Use of other landowner compensation mechanisms, such as direct payments and tax incentives, to advance NWFP goals on private lands arguably have received less attention by environmental advocates, but offer similar promise. Key research needs regarding compensation mechanisms of any type include evaluating the degree of difficulty in their implementation, and evaluating the potential returns in terms of the net ecosystem services benefits gained.

There also are opportunities for improving knowledge concerning the use of nonfederal funding to finance forest restoration on federal lands. Existing research demonstrates examples of supporting forest restoration projects that lead to watershed improvements (e.g., McCarthy 2014). The Pacific Northwest accounts for the majority of high-biomass forests nationwide, and federal lands account for nearly half of the regional total (Krankina et al. 2014), suggesting possible opportunities related to protection and stewardship of sequestered carbon should carbon markets be developed in the region and be open to participation by federal lands. The development of these potential financing opportunities will depend upon, among other factors, supportive public policies and organizational capacity at multiple scales (Davis et al. 2015, Kline et al. 2013). Exactly how such financing approaches can operate on public forest lands, how much additional revenue such approaches could provide toward forest restoration on federal lands, and how the revenue derived from these approaches should be distributed to benefit both people and forests are areas in need of further research.

Land Use Change

Given the impact that housing and other development could have on the amount and condition of remaining private forest land, analysis of the implications that such development could have for whether NWFP goals can be met in the future would seem warranted. In many cases, private lands likely augment public lands in providing various types of habitat, depending in part on the degree of development present. Most analyses have treated land use as an "either-or" proposition—land is considered either forest or developed. Increasingly, however, we are likely to see growing fragmentation of privately owned forest lands, with housing and other development interspersed "among the trees." Such development can have a variety of effects on habitat and ecosystem services, including effects on spotted owls, depending on how private landowners choose to manage their lands—whether for timber or largely for environmental amenities such as aesthetics, recreation, and habitat. For these reasons, development and its influence on landowner decisions could be a significant social process influencing the Plan area in the future. We see value in maintaining a research program that examines land use change and its effects on habitat and other NWFP goals, and that analyzes the effects of various policies that can be used to influence land use change.

Conclusions and Management Considerations

This chapter discusses how the NWFP, among other social and economic factors operating at multiple scales, has affected rural communities in the Plan area, and how they have changed since the Plan was implemented. It also highlights many of the ways in which federal forest management contributes to community socioeconomic well-being, and vice versa. The chapter is based on a set of guiding questions, several of which federal forest managers in the Plan area identified as being of interest. Given the statutory and policy foundation for considering socioeconomic well-being in federal forest management, a number of relevant management considerations based on the literature synthesized in the chapter are identified here.

Management Considerations

Wood products production remains important. Increased use of alternative silvicultural methods and expanded restoration treatments could increase federal timber production to maintain local wood processing infrastructure and the forestry workforce and support investments in new wood products markets. Historically, timber production was the central way in which federal forests in the NWFP area contributed to community socioeconomic well-being. The supply of timber from federal forests has dramatically declined post-NWFP. That decline, coupled with broadscale changes in the wood products industry, has altered this important connection between federal forests and communities. How to meet the NWFP goal of producing a predictable and sustainable supply of timber in the future to contribute to community socioeconomic well-being remains an important and continuing management challenge. Federal forests contribute roughly 10 percent of the regional timber supply today, reflecting current social acceptability and management approaches. Efforts and plans to pursue alternate management strategies focused on increased use of alternative silvicultural methods, and expanded restoration treatments could increase the volume of federal timber produced compared to recent outputs. How any increased federal forest harvest volume would influence the wood products industry and private forest land in the region is complex, however, and also is heavily affected by market and industry conditions outside of local control. Increased federal timber supply may be especially important in locations in which it provides the means to maintain local wood processing infrastructure and a forestry workforce, where federal agencies are the primary owner of local timberlands, or where the local forest products industry is attempting to expand into new wood products markets or to produce niche products.

Most timber harvested in the Plan area comes from private lands. Understanding how social, economic, and environmental variables influence timber production from private forests is important because it supports the business infrastructure needed for timber sales and restoration treatments on federal lands. In many places within the Plan area, the capacity to undertake forest restoration on federal lands depends on the presence of mills to buy timber products generated through restoration projects (which can help pay for restoration work through stewardship contracting), and the presence of a contract forestry workforce to do the work. The lack of mills to buy material is currently more of a challenge east of the Cascade Range, and the need to retain existing infrastructure west of the Cascades is critical for supporting forest restoration. With federal timber harvests declining in recent decades, forest managers and policymakers may want to consider the capacity of private forest lands to continue to supply the bulk of timber to mills within the NWFP area. Production from private forest lands is important because management of federal forests, in many cases, depends on having a market for logs to fund other restoration activities and on supporting the workforce to do that restoration. Challenges facing the productivity of private forest lands in some locations include reduced private investment in forestry, the potential for wildfire, insects, and disease, and the management goals and decisions of private forest owners. To what extent will private forest lands continue to be available for economically viable harvest in the future? Can private forest lands sustain current or increased timber harvest levels in a manner that is ecologically sustainable? Will the increasing number of more-urban-minded forest owners have any interest in harvesting? Answers to these questions will have implications for the ability of federal forests in the Plan area to meet their timber production and forest restoration goals.

Local communities could benefit more from jobs associated with forest restoration if the predictability and accessibility of restoration contracting opportunities improve and if stakeholders build social agreement on biomass harvesting and processing projects. Finding ways to create forest restoration jobs that local residents can capture will help build skills, capacity, and infrastructure needed to support management activities on federal forests, including fire suppression response, and will promote both healthy forests and healthy communities. The opportunities for local communities to benefit from forest management are strongly conditioned by factors such as the existing workforce, the processing capacity in the community, and the structure of work contracts. To promote more beneficial linkages between rural communities and their nearby public lands, agencies could consider structuring contracts in ways that make them more accessible to local communities. For example, they could consider the effect of restoration contract size and scope on local contracting capacity, and provide restoration contracts in a variety of sizes to support business diversity. Community capacity to participate in the restoration economy is not only a function of the structure of individual contracts but also of the consistency and predictability of contracts over time. Using a variety of tools may help build a predictable, sustainable program of restoration and biomass use work that will help support investments in contracting and processing capacity.

The harvesting and processing of biomass materials may also help deliver economic benefits from restoration work, but biomass production has often been controversial and economically challenging in the NWFP area. To improve the opportunities for positive outcomes, working closely with community members and other key stakeholders to build agreement on biomass harvesting and processing projects is important. Consideration of local benefits as a contributing factor to such projects may help build social agreement.

Forest management decisions affect access to and use of NTFPs and people's ability to benefit from harvesting them. Thus it is important to consider the social and ecological tradeoffs involved when making decisions that affect NTFP management. The key to supporting a robust and resilient NTFP sector in the Plan region is to recognize that many of the informal aspects of that sector enhance community and household well-being. By providing low-cost income-earning and provisioning opportunities, the NTFP sector can provide the flexibility that some individuals and households might need to survive times of crisis or improve their quality of life during better times. NTFP activities that take place outside the market also function as social-ecological glue, linking people to each other and strengthening human-nature connections. When developing forest management policies and regulatory frameworks, agencies may wish to consider how they will affect the informal economic activities associated with NTFPs, and weigh carefully how the ecological benefits of large-scale area closures for commercial NTFP harvesting and increased formalization stack up against the costs of decreased economic resiliency and a weakening of social connections.

Community economic benefits from federal forest-based recreation are greatest when visitors take overnight trips. Developing recreation opportunities that encourage overnight stays and align with visitors' desires will help local communities benefit from recreation spending. Recreation visitor spending is a significant driver of economic activity in many forest communities within the NWFP area. The key factor in explaining how much recreation visitors spend in local communities during their trip is whether the visitor spends the night (either in a public campground or private lodging). Visitors who spend the night away from home spend an average of 5 to 8 times as much as visitors who are in the area for the day only. Communities seeking to generate the greatest amounts of visitor spending locally would do well to focus on efforts that (1) increase the likelihood visitors will spend the night there, and (2) support businesses that supply the types of services, goods, and experiences that recreation visitors desire.

Policies and programs are needed to incentivize private forest landowners to produce desired ecosystem services and to help them benefit from doing so. Local communities, including private landowners, may stand to benefit from emerging markets in ecosystem services. Similarly to forestry and restoration work, however, the nature of these benefits will depend upon how market access is structured. To promote these benefits, managers and policymakers could consider local community needs in the development of ecosystem service markets, and provide opportunities for local businesses and landowners to benefit from restoration, carbon sequestration, and other stewardship activities. For example, habitat improvements on private forest lands likely could be enhanced by targeting incentive programs or technical assistance toward forest landowners whose own objectives include habitat protection.

Development of private forest land raises questions about society's ability to benefit from forests, and will affect ecological conditions and processes across land ownerships. Anticipating its implications is important for federal forest management decisionmaking. Private forest land development and accompanying changes in forest management are an inevitable outcome of social and economic forces. Forest land development raises three main concerns: (1) how does it affect our ability as a nation to produce sufficient forest commodities, (2) how does it affect the many ecological values (e.g., biodiversity) and ecosystem services we desire from forests as open space, and (3) how does it affect our capability to reduce wildfire risk in the WUI? Potential ecosystem services impacts from development are less certain. Low-density and urban development of forest lands undoubtedly have some adverse ecological consequences as forest lands are converted to residential and other developed uses. However, less intensive management of remaining private forest lands also could alter ecological characteristics in unanticipated ways, adversely affecting habitat for some species while improving habitat for others. Evaluating net ecosystem services impacts resulting from increasing development of forest landscapes will require anticipating how resulting changes in private forestry are likely to affect ecological conditions and processes, and their associated ecosystem services. Such studies have been fairly limited in the Pacific Northwest.

When developing communication and outreach strategies to help communities adapt to fire-prone landscapes, tailor them to community type; different community types will have different opportunities and challenges associated with wildfire adaptation. Timber harvesting is no longer the only focal federal forest management concern from a socioeconomic standpoint, as it was when the NWFP was developed. Two decades later, wildfire management has risen to become another important management concern for communities located near federal forests. A number of social scientists have conducted research about what factors drive community adaptation to fire-prone landscapes, and how to build community capacity to address wildfire risk (see McCaffrey et al. 2013). Paveglio et al. (2015b) suggested that strategies to build community capacity to address wildfire risk will depend on community type. They develop a four-part typology of WUI communities that includes formalized suburban communities, high-amenity/high-resource communities, rural lifestyle communities (these last two are consistent with the amenity trajectory), and working landscape/resource-dependent communities (consistent with the production trajectory). They suggest that communities sharing similar characteristics are likely to encounter similar challenges and opportunities in adapting to wildfire risk. Thus, agencies and others seeking to assist WUI communities become more resilient to wildfire could develop communication and outreach strategies tailored to each community type. Paveglio et al. (2015b) detailed what some of these might be.

When possible, drawing on local community resources to help fight wildfires (e.g., equipment, labor) could improve fire suppression response and help communities capture fire suppression dollars. Regarding fire-related jobs, given the erratic nature and small windows of demand for wildfire contracting, most businesses and workers need to perform other activities when they are not working on fire crews. As a consequence, local contracting capacity for fire suppression may be concentrated in particular regions, at least in part because there is other work for businesses to do when they are not fighting fires. This means that local capacity for fire suppression may be unequally distributed across the region, and concentrated in pockets where restoration work has historically existed. Related to this, the mobile and national nature of fire suppression means that local businesses trained in fire suppression will often be dispatched to fires outside their local community. Consequently, the ability of communities to capture fire suppression dollars locally may be reduced because firefighters (and fire camp support services) spend money on lodging, food, gas, and other supplies in the locale where they are fighting the fire. No matter where a fire occurs, firefighters will bring some of the income they earn back to their home areas. But, with such a necessarily mobile workforce, some firefighter earnings will be spent while on deployment to fires. This finding suggests that when fire resource needs and dispatch procedures allow for it, linking local fire suppression response capacity to less mobile resources (e.g., local fire districts, other fire suppression resources not signed up for national or regional deployment) might improve both local response and economic capture.

Working with communities to help mitigate negative climate change impacts will contribute to community well-being. Adaptation to climate change is another key concern for community socioeconomic well-being. This is not a purely technical exercise; it entails consideration of a multitude of social values and economic activities. Working with local community members to identify forest resources and economic activities potentially at risk from a changing climate, and considering management approaches that address these impacts, are ways that agency managers may help mitigate the impacts of climate change on communities.

Conclusions

Rural communities are not all alike, forest management policies and practices affect different communities differently, and the social and economic bases of many traditionally forest-dependent communities have changed in the years since the start of the NWFP. Better understanding and consideration of the economic development trajectories of different communities will help identify forest management activities that best contribute to their well-being. Providing a diverse set of benefits from federal forests may support communities in their efforts to diversify economically, and help build community resilience to future change.

Additionally, local relationships are important. Building constructive relationships with place-based nongovernmental organizations and other entities that are working to help communities become more resilient to external stressors can contribute to community resilience, for example by helping communities capture the economic benefits from forest management activities. The stressors affecting communities include changes in federal forest management policy, markets for forest products, development, wildland fire, and climate change. These same organizations may also be able to contribute resources and capacity to help address unmet needs on National Forest System lands, including (but not limited to) maintaining trails and other recreational infrastructure, filling gaps in planning capacity, building local business capacity to undertake forest restoration, raising funds to pay for forest management work, and leading collaborative forest planning efforts. Healthy forests and healthy communities are linked; thus it is in the interest of federal forest management agencies to contribute to community socioeconomic well-being, and it is in the interest of local communities to contribute to the capacity of agency managers to accomplish forest management work.

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Participants map their favorite destinations in the Mount Baker–Snoqualmie National Forest, Washington, during a human ecological mapping workshop.
Photo by Lee Cerveny.

Chapter 9: Understanding Our Changing Public Values, Resource Uses, and Engagement Processes and Practices

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Introduction

The Northwest Forest Plan (NWFP, or Plan) signified a movement away from intensive focus on timber management that was common through the 1980s and toward an ecosystem management approach, which aims to conserve ecological conditions and restore natural resources while meeting the social, cultural, and economic needs of present and future generations (Brussard et al. 1998). The NWFP emerged in response to expanded scientific knowledge about forests and shifting public values about resources and their management. An important goal of the NWFP was to protect forest values of late-successional, old-growth, and aquatic ecosystems. These may include amenity values (scenery, quality of life), environmental quality (clean air, soil, and water), ecological values (biodiversity), public-use values (outdoor recreation, education, subsistence use), and spiritual values (cultural ties, tribal histories) (Donoghue and Sutton 2006). This synthesis looks at the latest research on many of these forest values and adds to our thinking about how the NWFP has contributed to their protection.

Since the NWFP was instituted, the social context of the Plan area has changed. The social dimension of natural resource management in the NWFP is dynamic and inherently complex, resembling what some have referred to as "wicked problems" (Head 2008, Weber and Khademian 2008) or resource challenges that are unstructured (where it is difficult to identify causes and effects), crosscutting (multiple stakeholders, across jurisdictions, social complexity), and relentless (with no final solution). These wicked problems are often characterized by a high degree of uncertainty and potential for conflict, with little agreement on the solution (Weber and Kahdemian 2008). Effective management of wicked problems in the NWFP area requires significant resources, strong social networks, and collective engagement of actors (agencies, institutions, and individuals) in diverse policy arenas within the planning area (Weber and Kahdemian 2008).

At the same time, U.S. society has become polarized by both ideology and vocal partisanship, which have been linked to economic insecurity in the postindustrial era, and the potential for shifting power relations among sociocultural groups, including gender, ethnicity, and religion, referred to as "cultural backlash" (Inglehart and Norris 2006). Collaborative management and expanded emphasis on public processes that engage diverse stakeholders where objectives are transparent and sideboards are visible can help navigate the terrain of wicked problems. However, there is no guarantee that these efforts will result in an outcome that is widely embraced. Still, a process that generates mutual understanding, leads to informed decisions, incorporates new knowledge, and recognizes diverse uses and values would be a step forward.

Also since the NWFP was developed, scientists have explored and embraced new conceptualizations of ecosystems and ways to understand their benefits to people. Resource governance increasingly has adopted a framework of ecosystem services—the conditions, processes and components of the natural environment that provide tangible and intangible benefits to sustain and enhance human life (Daily 1997). Scientists and forest managers are updating their thinking about the variety of forest benefits

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that serve society and developing ways of measuring and comparing a diverse array of tangible and intangible benefits. As managers seek strategies for more integrated and holistic resource management using an ecosystem services approach, the importance of considering an array of public values (including aesthetic, recreational, spiritual, and heritage) becomes paramount.

Scientists increasingly recognize that conservation initiatives are more likely to lead to better informed decisions when ecological and social elements are integrated (Charnley 2006) (see chapter 12). Socioecological systems (SES) science recognizes the inextricable linkages between human societies and ecological systems (Berkes et al. 2000), and that ecosystems are embedded in levels of social organization (Brondizio et al. 2009). Halliday and Glaser (2011) considered an SES to be "a system composed of organized assemblages of humans and non-human life forms in a spatially determined geophysical setting" (2011: 2). Changes to social systems, such as population dynamics, market shifts, or changes in structural relations among natural resource institutions, can affect the natural environment. Conversely, changes to the ecological system, such as fire, flood, or diminished forest health, can affect human-nature interactions and settlement patterns (Gunderson and Holling 2002, Machlis et al. 1997). The social component of the SES refers broadly to property and access rights; land and resource tenure systems; resource knowledge systems, including local and traditional ecological knowledge; subsistence uses; worldviews; values; and perceptions about the environment (Berkes et al. 2000). An SES encompasses a variety of agencies and actors as they interact with the natural environment at multiple scales in ways that are dynamic, complex, and continuously adaptive (Folke et al. 2005, McGinnis and Ostrom 2014). An understanding of public values is essential to understand the complex influences of social values and choices on ecosystem uses and condition (Ives and Kendal 2014).

By thinking of the NWFP area as an integrated SES, with a complex web of interactions, forces, dynamics, and elements, we can begin to recognize and address major shifts in that system and understand their corresponding effects on the natural and social environment. This system includes public and private lands, governing agencies (federal, state, tribal), communities of place (municipalities, counties), and communities of interest (stakeholders, user groups). We recognize that the social dimensions of the Plan area influence how ecological goals are established, pursued, and met or not met (Lange 2016, Spies and Duncan 2012).

A science synthesis of the Plan area is not complete without a comprehensive understanding of the region's complex social ecology, particularly with regard to public values, citizen engagement, and governance of federally managed lands. Governance is a term widely used in political science and public administration to describe formal and informal processes, decisionmaking norms, and interactions among institutions involved in a collective problem (Hufty 2011). Governance may be undertaken by governments, tribes, legal corporations, multilateral commissions, collaborative groups, boards of directors, or social organizations. Governance explains how rules, norms, and decisions are structured, maintained, regulated, and monitored. Governance can be accomplished using a variety of tools, including laws, rules, markets, social norms, contracts, collaborative agreements, and public-private partnerships, as well as through symbols, maps, and language (Bevir 2013). In this chapter, we discuss governance as a formal process managed by government institutions like the U.S. Forest Service, primarily through laws and regulations. We also refer to "collaborative governance," which describes the contribution of collaborative groups, which engage federal, tribal, state, and municipal governments, citizen groups, and corporations in deliberation over common resource problems.

Public values, attitudes, and beliefs about forests and the management of forest resources are not fixed, but can shift over time, owing to a multitude of complex factors (e.g., economic, political, social, cultural) (Manfredo et al. 2003, Vaske et al. 2001). Changing demographics related to urbanization, amenity migration, or regional population shifts in response to economic opportunities all can alter the makeup of a population and result in a potential shift in environmental values, beliefs, and behaviors, as well as in the kinds of connections people have to place (Gosnell and Abrams 2011, Jones et al. 2003). Public uses and outdoor experiences in national forests and other federal lands also evolve in response to emerging consumer trends, economic factors, new technologies, or changes to geophysical or climate conditions (Cordell et al. 2002, Tuan 2013). In addition, the ways that citizens engage in natural resource management and share their views with land management agencies have changed, as people express a desire to be involved in decisionmaking about public lands (Stern and Dietz 2008). American politics since the 2000s has been characterized by increasing partisanship, identity politics, and ideological divides that have pulled people apart and presented mounting challenges to public lands management (Abramowitz and Saunders 2006, Iyengar and Hahn 2009). Emerging collaborative structures that attempt to bring together multiple agencies and stakeholders to deliberate and plan for resource management have become prevalent (Emerson and Nabatchi 2015).

Public land management agencies are finding new ways to measure and evaluate the variety of benefits that ecosystems provide. The concept of ecosystem services has developed more over the past 10 years in resource management as a useful framework. The ecosystem services framework assigns economic and noneconomic values to ecological functions, allowing policymakers to evaluate ecosystems using comparable metrics (Carpenter et al. 2009). The MEA (2005) framework describes four categories of ecosystem services: supporting, provisioning, regulating, and cultural. Ecosystem services featured prominently in the National Forest System land management planning rule, which guides how forest management plans for each national forest are developed (USDA FS 2012). The new planning rule is historically significant in that it signals a shift toward valuing resources more broadly (using the ecosystem services framework) as well as a greater emphasis on public engagement, which recognizes the importance of public values, attitudes, and beliefs. This is especially relevant for the NWFP, which exists as amendments to 17 forest plans that are due for revision.

One goal of the NWFP was to provide a "balanced and comprehensive strategy for the conservation and management of forest ecosystems, while maximizing economic and social benefits from forests." An updated understanding of these complex dynamics related to humans and their myriad interactions with public lands in the NWFP area is an essential component of this chapter, particularly with regard to public lands. This chapter illuminates how public perceptions, attitudes, beliefs, and values regarding forests and their economic and social benefits may have changed over the past 20 years. While chapter 8 speaks to socioeconomic ties between communities and forests, this chapter identifies what we know about shifting values, place meanings, outdoor recreation trends, and ways of public engagement. The focus of chapter 9 falls into the basket of "cultural ecosystem services," (also referred to as cultural services) (Costanza et al. 2014). Cultural services include benefits gained through spiritual enrichment, outdoor recreation, religious or spiritual value, reflection, learning, sensory enhancement, and socializing, as well as place-based benefits such as identity, cultural heritage, and sense of place (Chan et al. 2012, Klain and Chan 2012, MEA 2005, Satterfield et al. 2013) and often emerge as a result of enduring relations between people and a landscape over many generations (Fagerholm et al. 2012).

Several chapters in this volume address other aspects of the sociocultural aspects of the SES, with many points of articulation with chapter 9. Chapter 8 focuses on the socioeconomic well-being of rural communities, the role of forest industries, and implications for private landowners in the Plan area. The discussion of recreation's contributions to rural economies in chapter 8 can be considered alongside discussion of recreation trends in chapter 9. In addition, both chapters touch on notions of trust and its importance for effective resource governance. For an indepth discussion of challenges and opportunities related to environmental justice, poverty, and resource access in the NWFP area, see chapter 10; for discussion of tribal resource governance, resource use, and indigenous knowledge systems, refer to chapter 11. As we consider elements of public involvement and collaboration in this chapter, it may be useful to inquire whether existing governance mechanisms promote participation from underserved communities. These discussions can be considered alongside findings related to collaboration in this chapter. These points of overlap are intentional and desirable to fully understand the SES as an integrated whole.

Guiding Questions

A goal of SES science is to better understand the social context in which ecological goals are identified and achieved. The questions below were given to the chapter 9 science team by managers. The authors used these questions to frame chapter contents and relied on available literature to address and respond to these questions.

- What does social science tell us about how stakeholders' attitudes, beliefs, and values have changed over the past 20 years? How are these attitudes, beliefs, and values associated with resource management (recreation, resource use, protection)?
- How have stakeholders' relationships to the landscape and natural resources changed in the NWFP area?
- What value do people place on cultural ecosystem services from public lands, including recreation?
- What has been learned about the importance of valuing place?
- How have public uses and interactions with forests and grasslands changed over the past 20 years?
- What are the drivers that shape public uses of forest lands for recreation?
- How have recreation values and uses changed in the past 20 years?
- How does the body of science inform sustainable recreation?
- What strategies are effective in engaging communities and the public in the NWFP area?
- What kinds of collaborative groups and processes are engaged in the NWFP area?
- How is collaborative forest management changing?
- What elements contribute to successful collaboration in forest management? What examples exist of successful collaboration?
- How much has collaboration contributed to achieving objectives in resource management and socioeconomic well-being?

Two additional topics were added later by the science team to address the specific themes considered of importance to understanding the scientific basis of forest planning and management. These topics included a discussion of trust as well as social acceptability of various harvest practices. We structured the chapter into seven subsections: public values, attitudes, and beliefs; valuing place; cultural ecosystem services; outdoor recreation; trust; involving the public; and agency-citizen collaboration. Each subsection deals with a set of topics that contribute to the questions asked and concludes with a brief summary. The chapter concludes with an overview of research needs, uncertainties, and information gaps, as well as a discussion of management considerations.

The study team used standard social science perspectives rooted in geography, anthropology, sociology, environmental psychology, and public administration. It was not our intent to collect primary data, but rather to synthesize existing literature in these five topic areas assigned to this chapter. We relied on the best available social science to highlight current knowledge about these important topics. For some topics, there is little or no empirical research conducted in the Plan area. Authors drew from case studies, dissertations, or technical reports when peer-reviewed publications for a given topic were not available. We focused foremost on scientific findings relevant to the Plan area. However, we did include a few seminal works which offered theoretical or methodological contributions or relevant research results from other parts of North America to demonstrate a trajectory of inquiry with bearing on the Plan area. Data synthesized here are based on scientific publications and case studies that occurred since the previous NWFP science synthesis in 2006 (Haynes et al. 2006), except in cases when current research was not available.

Key Findings

Public Values, Attitudes, and Beliefs

Understanding values, attitudes, and beliefs has become increasingly important in environmental decisionmaking and natural resource governance (Allen et al. 2009). Recognizing how and why people value different aspects of ecological systems potentially can allow resource managers to gain awareness about how different forest management goals and strategies may be viewed by the public and potentially understand the roots of conflict among stakeholders. Values inform how people interact with the landscape and engage with conservation issues (Brown and Reed 2012). Values are known to predispose attitudes, management preferences, and behaviors. Thus, values can indicate whether proposed activities or goals in a plan would be socially acceptable and to whom (Allen et al. 2009, Fulton et al. 1996, Vaske and Donnelly 1999). By understanding public values, land managers will be better equipped to reach informed decisions (Tarrant et al. 2003).

Understanding values, attitudes, and beliefs— Values are most commonly understood as enduring beliefs about the world that are often formed in childhood and serve as guideposts for desirable actions (Rohan 2000, Rokeach 1973, Schwartz 1994). Values are "modes of conduct" or end-states of what is desirable (Manfredo et al. 2004). Two types of values are discussed by natural resource social scientists. "Held values" represent an embedded human characteristic that shapes the judgments people make about the world and the subsequent actions they take (Bengston and Xu 1995, Rokeach 1973). Held values are associated with desirable goals, standards, guidelines, or criteria that help people decide what is right or wrong, worthy, or undesirable (Schwartz et al. 2012). "Assigned values" can be attached to a specific object or physical place in the world, as well as to intangible concepts (i.e., an economic system or political institution) whereby a person attempts to denote relative worth to an object or place on the landscape (Bengston 1994, Brown 1984, Rokeach 1973). Both held and assigned values are important for land managers because they have been shown to predispose people to certain attitudes toward forest management practices and certain patterns of resource use and other environmental behaviors (Fulton et al. 1996).

The cognitive hierarchy model provides a logical structure for understanding the relationship between values, attitudes, and beliefs, and how these in turn influence human behaviors and actions (Dietz et al. 2005, Rokeach 1973, Vaske and Donnelly 1999) (fig. 9-1). Originally developed by Rokeach (1973), the model was fleshed out more fully by Fishbein and Ajzen (1975) as the "Theory of Reasoned Action" and later the "Theory of Planned Behavior" (Ajzen 1991, Ajzen and Fishbein 2005).

The cognitive hierarchy offers a reasoned conceptual framework that allows social scientists to explore the relationship between values, attitudes, and goals for forest management (Brown and Reed 2000). The components of the model include beliefs, value orientations, attitudes, intentions, and behaviors (box 1). Beliefs are statements of a person's understanding of the world; "they are facts as an individual perceives them" (Dietz et al. 2005: 346). Beliefs are a person's judgment about what they consider to be true or false. They can be shaped by science, feelings, experiences, intuition, or social norms (Zinn et al. 1998). Value orientations are the aggregation of beliefs about a particular issue or topic (Allen et al. 2009). Values are not directly measured, as they are often difficult to express, but social psychologists do measure value orientations as the basic set of beliefs (Fulton et al.

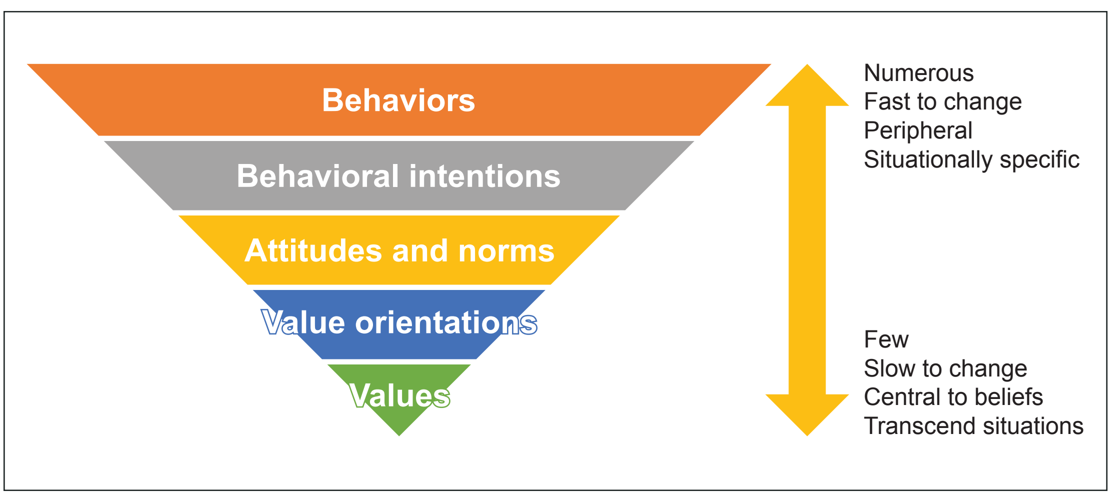

Figure 9-1—Cognitive hierarchy model of human behavior. Source: Adapted from Fulton et al. 1996.

Box 1—Key Definitions

Values: Enduring, consistent principles, often formed at an early age, about the important elements in life, including, what is good or bad; desirable or undesirable (Manfredo et al. 2009) (e.g., inclusiveness, justice, integrity, equality).

Value orientations: Set of beliefs about nature and the environment (Fulton et al. 1996). (e.g., orientations toward nature, human's role in the environment, public land management).

Beliefs: Judgments about what is true or false and what attributes are associated with someone or something, or the consequences of an action. (Ajzen 2002). (e.g., beliefs about land management agencies, forest conditions, or effects of actions).

Attitudes: Learned tendencies to react favorably or unfavorably to a situation, conditions, people, objects or ideas (e.g., level of support for an agency's actions; preferences for particular activities or actions).

Intentions: Convictions, aims to act in a particular way.

Behaviors: What people do, actions they take (e.g., participate in environmental activism, voting, stewardship behaviors, recycling, littering, outdoor recreation use, consumptive use of resources).

Norms: Implied or explicit rules or guidelines that regulate behavior and prescribe what people do (Stern et al. 2000). Norms can be individual (personal guidelines) or social (societal expectations).

1996, Rokeach 1973). Beliefs form the basis for attitudes. Attitudes are statements of people's positive or negative evaluations of a specific object or situation and are typically expressed as likes or dislikes, or preferences (Hoult 1977). Attitudes stem from values and also from lived experiences that shape a person's typical response or approach to something. They reflect one's dominant personality traits (e.g., optimistic vs. pessimistic; internal responsibility vs. external responsibility) (Ajzen and Fishbein 1975). Environmental attitudes have shown to be more predictive than values for understanding management preferences. The relationship between values, beliefs, and attitudes has been explored in many studies in natural resource settings (Bright et al. 2000, Fulton et al. 1996, Vaske and Donnelly 1999).

Fishbein and Ajzen (1975) set out to develop a framework that could predict intentions (the aim of a particular action) and behaviors (actions people take in nature, and may include stewardship, recreation, or consumption of forest resources). In their Theory of Reasoned Action, their focus is on antecedents to behavior, including beliefs about the consequences of a specific behavior and generalized attitudes (favorable or unfavorable) about a specific behavior (Fishbein and Ajzen 1975). For example, the behavior of riding motorized vehicles off developed roads would depend on a person's understanding of how that action affects the biophysical and social environment as well as overall attitudes about off-highway vehicles. They also introduce the concept of normative beliefs and subjective norms. The normative beliefs are judgments held by others about the appropriateness of a particular behavior. The subjective norm is a combination of beliefs about the existence of social norms and individual motivations to comply with norms (Ajzen 2000). The interaction among beliefs, attitudes, intentions, and behavior is shown as a feedback loop, whereas when a particular behavior (behavior X) is performed, this affects one's normative beliefs about what is appropriate, which is guided by social norms, which then shapes intentions (fig. 9-2). The next iteration of the model, the Theory of Planned Behavior, added a component of individual agency or power, noting how the role of an individual's perceived control over their behavior can affect behavioral intentions (Ajzen and Fishbein 2005).

Values, attitudes, and beliefs can affect human intentions and actions (behaviors), but other factors play a role, including norms. Another concept used commonly in natural resources settings, particularly with emphasis on understanding pro-environmental behavior, is the

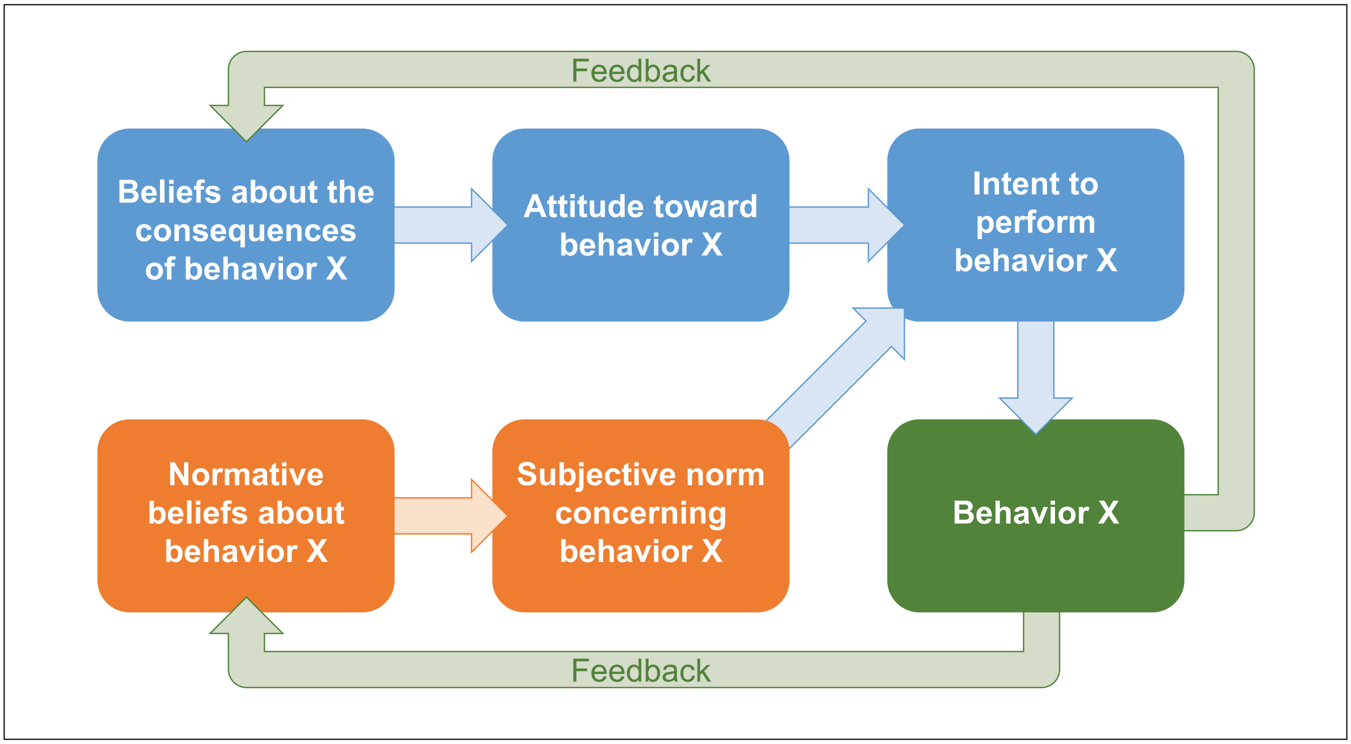

Figure 9-2—Theory of reasoned action (Fishbein and Ajzen 1975).

values-beliefs-norms (VBN) theory of environmentalism (Stern 2000). The VBN theory has been successful in explaining different types of environmental actions (Stern et al. 1999) and the acceptability of social or environmental policies or actions. This theory suggests that values do not directly predict behavior, but are indirectly implicated through beliefs and norms (de Groot et and Steg 2008, Steg and Vlek 2009). The idea is that values affect behavior indirectly by activating personal norms (moral obligations to perform a particular action). Personal norms are activated when someone acknowledges that (a) not acting pro-environmentally will lead to negative consequences, (b) when someone feels personally responsible for those negative outcomes, and (c) they believe their own efforts will help to mitigate the problem or minimize consequences (taking responsibility). One should first be aware of problems caused by the relevant behavior before considering to what extent one personally contributes to the problems and whether one could possibly be part of the solution, which in turn determines the extent to which personal norms are activated. Values thus influence the extent to which one is aware of the problem, but also may predict variables about how they respond to the problem (de Groot and Steg 2008). Our awareness of those norms influences or fine tunes our ultimate actions (Stern 2000).

Steg et al. (2014) discovered four value types important for understanding beliefs, norms, intentions, and behaviors: hedonic (concern for achieving personal needs or exerting minimal effort), egoistic (concern for costs and benefits for the individual), altruistic (concern for human welfare), and biospheric (concern for quality of nature and the environment). Biospheric and altruistic values were found to promote pro-environmental attitudes and behaviors (Stern and Dietz 1994). In contrast, egoistic and hedonic values were negatively related to pro-environmental attitudes and behaviors. Those with altruistic and biospheric values are likely to be more aware of the problem, while awareness is lower with those who have hedonic and egoistic values (de Groot and Steg 2008). Whitmarsh and O'Neill (2010) learned that self-identity as an environmentalist is a significant predictor of behavior, especially in combination with values, attitudes, and beliefs. Yet, others have shown that self-identity has a minimal effect (Rise et al. 2010). More research is needed to show whether self-identity as an environmentalist is a useful variable.

Recent studies of values have combined perspectives from cultural anthropology with systems theory. Anthropologists have long suggested that values are relatively stable and enduring and are developed through collective processes of socialization (schools, religious organizations, traditions, etc.) and that are shared with communities, cultural groups, or societies (Kenter et al. 2015, Kluckhohn 1951, Schwartz 2006). Values do not exist alone, but are deeply embedded in our social institutions, governments, collective behaviors (cultural practices), and the media (Schwartz 2006). Values exist at multiple levels and locations throughout our socioecological system and are mutually reinforced (Manfredo et al. 2017). One's individual values may guide one's actions or determine one's membership in a particular organization, but that organization reflects and reinforces the shared values deeply embedded in the social system. For values shift to occur, multiple entities at various levels of the socioecological system would need to be engaged (Manfredo et al. 2017). This systems theory framework views values as resistant to rapid change, but recognizes that major socioecological events, such as mass migrations resulting from changing environmental conditions (Kitayama et al. 2010), modernization (Inglehart 1997), or urbanization (Manfredo et al. 2009) can result in a gradual shift in values (Manfredo et al. 2009, 2017). New research, such as that offered by Dietsch et al. (2016), is needed to explore the influence of macro-level organizations on one's values and the ways that shared values emerge.

We know that values can evolve during processes of deliberation and discussion, where mutual learning takes place among people who have different backgrounds and experiences (Daniels and Walker 2001). Deliberation allows participants to consider their own arguments and the assumptions behind them, hear the perspectives and experiences of other participants and understand the reasoning behind their views, evaluate various positions, and reach informed decisions. Deliberation results in social learning (Cundill and Rodela 2012). Deliberation through organized workshops and stakeholder engagements can lead to exposure to different perspectives and result in new insights and knowledge about how people value natural resources (Steyaert et al. 2007). Deliberative processes are useful for identifying values that are difficult to pinpoint (Kenter et al. 2016a). Collaborative groups, public engagement opportunities, and other processes can result in individual and group learning. Efforts to engage citizens in collaborative and deliberative processes are discussed later in this chapter.

Exploring environmental values and attitudes— Environmental values have been measured in a variety of ways. Table 9-1 features several approaches in the literature that are the most common. This is not meant to be an exhaustive list, and there are new approaches to measuring environmental values, attitudes, and beliefs that are not included here, because they have not been widely used.

Many scholars measure "value orientations," which are sets of values that link together based on a common orientation to nature and the environment. Environmental value orientations are clusters of interrelated values that reflect an overall relationship between humans and the environment (Fulton et al. 1996, Vaske et al. 2001). Many classification systems have been used to explore value orientations. Xu and Bengston (1997) classified values into instrumental (the usefulness of forests as the means to a further end, such as logs for housing or recreation use for people); and noninstrumental (forests are valuable in themselves), which Moore (2007) also calls intrinsic values. Stern and Dietz (1994), Schultz et al. (2005), and others used three value categories: egocentric (self-oriented), altruistic (public good), and biospheric (for nature itself) to predict environmental behavior. A widely used framework of value orientations used by Vaske et al. (2001) identified anthropocentric (utilitarian) and biocentric (nature centered) (Steel et al. 1994). Later studies added a third orientation, moral/spiritual/aesthetic, which also encompasses sacred values and heritage values as well as bequest values (Bengston et al. 2004). This category of values includes both religious values as well as spirituality that relates to people's respect for natural forces, as well as a spirituality that exists without humans (Proctor 2009). Winter and Lockwood (2004) developed a natural area scale, which included intrinsic, use, and non-use values.

Table 9-1—Various approaches to exploring environmental values

Values approach	Goal	Tools or methods	Relevant studies
Value orientations	Quantifying values and classifying respondents into similar groups based on their orientations to the natural environment	Various Likert scales	de Groot and Steg 2008, Dietz et al. 2005, Fulton 1996, Steel et al. 1994, Stern and Dietz 1994, Stern et al. 1995, Xu and Bengston 1997
New Ecological Paradigm, also known as "New Environmental Paradigm"	Measures anthropocentric and biocentric orientations to the natural environment	15-item scaled survey	Cordano et al. 2003, Dunlap 2008, Dunlap and van Liere 1978, Dunlap et al. 2000, Stern et al. 1995
Natural area values scale	Measures values relevant to natural areas, including intrinsic, use, non-use, recreational, and aesthetic	20-item scaled survey	Ford et al. 2012, Winter and Lockwood 2004
Values suitability analysis Values compatibility analysis	Evaluates consistencies between land management prescriptions and public values	Numerical rating system	Brown and Reed 2012, Reed and Brown 2003
Public values of forest	Predicts public values based on forest outputs, amenities, and protection	12-point scaled survey	Tarrant et al. 2003
Landscape values mapping	Shows what values are associated with places on the landscape using maps	Maps and other spatial tools	Alessa et al. 2008; Brown and Kyttä 2014; Brown and Reed 2000, 2012
Valued attributes of landscape scale	Emphasis on the value of site attributes (natural, social, experiential, cultural, productive)	Measures 26 value attributes	Kendal et al. 2015
Deliberative Value Formation Model	Based on the notion of shared values. Use of a deliberative process to generate learning and values shift.	Value orientation scales and deliberative process	Kenter 2016; Kenter et al. 2016a, 2016b

Value orientations provide the foundation for specific attitudes toward forest management (McFarlane and Boxall 2000, Steel et al. 1994, Tarrant and Cordell 2002). Environmental value orientation scales approaches distinguish between anthropocentric (oriented to human well-being) and biocentric (oriented to ecological well-being). Others identified an ecocentric orientation that emphasizes ecosystems (Surmeli and Saka 2013). Value orientations can often predict attitudes toward forest management practices and support for natural resource policies, although particular circumstances can override values, such as a person's unique relationship to a particular setting or if their income depends on the decision outcome. Individual motivations sometimes override value preferences. While some of these studies have successfully shown that people hold shared sets of values, they have not been able to explain why distinct stakeholder groups (e.g., fishermen and biologists) holding the same set of value orientations exhibit divergent behaviors.

Studies have shown that some factors can predict attitudes toward management outcomes. One of the most consistent predictors of values is gender. Several independent studies have shown that women tend to favor biocentric (noneconomic) values more often than men, although the differences are small in most cases (Kellert and Berry 1987, Steel et al. 1994, Tarrant et al. 2003, Vaske et al. 2001). Other linkages have been found between value orientations and the visual impacts of resource management (Tindall 2003), perceived threats to forest health (Abrams et al. 2005), and participation in activism directed at the forest sector (McFarlane and Hunt 2006). Working in Australia, Ford et al. (2009) showed that the public's acceptance of clearcutting was related to their value orientation. Those with stronger "use values" (timber production) were more likely to find clearcutting acceptable than those with stronger "intrinsic values" for nature. In Canada, Tindall (2003) also found that those with a biocentric orientation tended to support policies aimed toward resource protection and view commercial forest practices as unsustainable, and its visual impacts unacceptable. Steel et al. (1994) conducted a study in Oregon about whether value orientations predict public attitudes toward various forest management practices. They observed that respondents with an anthropocentric values orientation support resource use for economic gain, and view forest management as sustainable and its visual impacts acceptable (Steel et al. 1994). From their work on public acceptance of clearcutting in Australia, Ford et al. (2012) learned that a person's aesthetic experience in nature is filtered by values and that this experience directly shapes their attitudes toward management actions.

Clement and Cheng (2011) used a random household survey (response rate 34 percent) in Colorado and Wyoming to explore values and attitudes toward forest management, and preferences for specific management activities (logging, oil and gas drilling, and off-highway vehicle use) in three national forests. Overall, respondents scored highest on the values "aesthetic," "recreation," and "biodiversity." Statistical analysis showed that both values and attitudes influenced management preferences. Specifically, they found that certain values were more prevalent in classifying respondents for each management issue. Understanding one's values was helpful in predicting responses to management preferences. Respondents that shared similar value orientations sometimes held different and even opposing policy preferences (Clement and Cheng 2011), which suggests that values and attitudes/preferences are most powerful when examined together. For example, those with stronger values in "recreation" and "economic" were positively correlated with oil and gas leasing. Interestingly, the same two values also correlated positively with sport hunting and fishing and negatively with wilderness designation. They also found that those who ranked recreation, economic, historical, and cultural values high are more comfortable with forest treatments to reduce wildfire risk. This approach is a useful example of how to tease out the relationship between values, attitudes, and management preferences. Results show that members of the public have a range of values and may share many in common, while holding different management preferences.

The New Ecological Paradigm (NEP), also referred to as the New Environmental Paradigm, is another widely used scale that measures environmental attitudes along a **biocentric** to **anthropocentric** continuum. The original scale developed in 1978 used 12 items and measured along three facets of internalized beliefs and values: beliefs about humans' ability to upset the balance of nature, recognition of limits to growth, and beliefs about humanity's rights to rule over nature (Dunlap and van Liere 1978). The scale was later updated and renamed to constitute a 15-item scale that measured values along five facets: beliefs that humans affect the balance of nature, beliefs that humans are causing harm to the environment, beliefs that humans are not exempt from constraints of nature, beliefs that the Earth's resources are limited, and beliefs that humans have the right to modify and control the environment (Dunlap 2008, Dunlap et al. 2000). Respondents agreed or disagreed with statements related to each facet to develop a score for each facet and an overall NEP score. In a meta-analysis conducted in 2009 (Hawcroft and Milfont 2010), the authors found 69 distinct studies (52 in the United States) that used NEP in 36 countries. Despite its widespread use, Hawcroft and Milfont (2010) found a lack of empirical and theoretical integration in studies that used NEP to measure environmental attitudes. Partly this is due to variations in the implementation of NEP (differences in sample size and context). Others who tested the validity of NEP found variation among the five facets, with the most reliable being the scale measuring the "balance of nature." Moreover, they found that the five subscales were more useful than the cumulative NEP score

(Amburgey and Thoman 2012). Still, it is not clear that the NEP scale is adequately measuring environmental attitudes.

The Natural Area Values Scale has been used primarily in Australia. This approach identifies five factors (intrinsic, use, non-use, recreational, and aesthetic) (Ford et al. 2012, Winter and Lockwood 2004). Values suitability analysis (also called values compatibility analysis) is an approach that evaluates the extent to which public values and attitudes are consistent with agency management actions. Brown and Reed (2012) determined where all-terrain vehicles (ATV) could occur within a national forests that would not compromise other values. These data helped land managers establish areas that were compatible with ATV use and areas that would require tradeoffs with other forest uses. Acknowledging distinct values sets and their compatibility with management actions is useful, but in many cases, resource users have conflicting values. Collaborative learning processes can raise awareness of values to the surface and acknowledge the tradeoffs among values that exist, achieving solutions that reflect a multiplicity of coexisting values (Daniels and Walker 2001).

The Public Values of Forest Scale (Tarrant et al. 2003) is a survey based on a 12-item scale that considers three factors: outputs (timber, roads, raw materials, range, recreation), amenities (quiet, education, aesthetics), and protection (clean water, fish and wildlife, endangered species). The survey was found to have predictive validity for discerning values among demographic variables and in predicting attitudes toward wilderness. Understanding attitudes is helpful, yet one's attitudes do not necessarily predict whether one accepts a particular management approach or outcome, which can be influenced by contextual conditions and learned behaviors (Oreg and Katz-Gerro 2006).

Landscape values mapping (LVM) is an approach used to understand what values people attach to places on the landscape (Brown and Reed 2000). The LVM approach is used to capture values across a landscape for use in planning and decisionmaking (Brown and Reed 2009, Raymond and Brown 2006). The approach understands humans as cognizant actors who experience the landscape directly through their senses, and assign meaning to places based on these experiences (Zube 1987). Brown and Reed (2000) built a landscape values typology derived from work of Rolston and Coufal (1991). They defined 13 landscape values: economic, learning, historic, cultural, future, intrinsic, spiritual, therapeutic, subsistence, life supporting, biodiversity, recreation, and aesthetic, and asked respondents to place colored dots on a map for each value. Brown and Reed (2000) validated their landscape values typology by demonstrating that each landscape value represented a discrete construct, and that the values could not be organized into higher order factors. The study also showed that respondents were as likely to select noncommodity values (aesthetic, spiritual) as commodity values (economic, subsistence). The assigning of landscape values to a map requires that the respondents recall their direct experiences or the images from stories told about these places and the meanings generated by these experiences, which are influenced by held values.

LVM has been applied in a wide variety of countries, spatial scales, and sociocultural settings and has achieved some level of standardization through replication (Alessa et al. 2008; Beverly et al. 2008; Brown 2006, 2012; Brown and Raymond 2007; Brown and Weber 2012; Clement and Cheng 2011; Fagerholm et al. 2012; Nielsen-Pincus 2011; Reed and Brown 2003, Reed et al. 2009; Sherrouse et al. 2011). The landscape values typology is commonly used in conjunction with spatial attributes mapping (Brown 2004) where participants have options to assign multiple values across a landscape (using points or drawing shapes). Across the studies, there has been fairly consistent application of the original 13 landscape values, with some customization to suit sociocultural or biophysical conditions. For example, in Alaska and Washington, the value "subsistence" was used because of the cultural, political, and economic importance of food gathering as a cultural practice (Alessa et al. 2008, Cerveny et al. 2017). Another value that has been sometimes added is "wilderness," which is appropriate in Euro-American settings, but is less meaningful in non-Western societies (Brown and Alessa 2005). Several studies employing the landscape values typology have included "special places" as an additional mapped feature, often designated with a special symbol ("X") and described using narrative description. (See Brown and Kyttä [2014] for a comprehensive overview of existing public participation geographic information systems studies).

The Valued Attributes of Landscape Scale represents a tool that measures the value of site attributes, features, or properties (Kendal et al. 2015). Site attributes may be understood as natural, social, experiential, cultural, or productive. This scale uses a standardized approach that can be compared across groups of people and in diverse landscapes (Kendal et al. 2015). The approach is an attempt to bridge held values (core values) and assigned values (attached to places or objects). More tests are required to establish the reliability of this approach.

Values are known to shift or change in response to new learning and deliberative engagement (Manfredo et al. 2017). The Deliberative Value Formation Model (DVF) is built upon the idea that group interactions and deliberative processes can result in new learning that results in a shift in values (Kenter et al. 2016a). Through deliberation, people can learn from each other and gain practice in forming reasoned opinions and evaluating arguments, resulting in new knowledge and insights (Steyaert et al. 2007). In group processes, members can express their views, reflect upon their own opinions as well as others, share experiences, and engage in meaningful debate (Kenter et al. 2016a). The DVF approach integrates deliberation with structured valuation to inform both individual values and group values (Kenter et al. 2016a). The model is based on an understanding of shared values, or those values held in common as communities, societies, and cultures (Kenter et al. 2015). The model has been tested in studies focused on monetary valuation and ecosystem services (Kenter 2016, Kenter et al. 2016b), as well as deliberative decisionmaking by communities for marine-protected areas (Ranger et al. 2016). Although different, these studies all showed the emergence of shared values among deliberative groups. DVF has not been tested in the NWFP area to date, but represents a promising approach, particularly given the preponderance of collaborative groups engaged in shared learning about resource management, discussed later in this chapter.

Each of the approaches described above and presented in table 9-1 have potential value or application to attempts to manage resources in the Plan area. Some approaches, such as LVM, which highlights the public's connection to landscape and places at various scales, have already been used extensively in the Plan area, as this chapter describes below. We note later in the chapter that longitudinal social values data for the Plan area would be useful for understanding if or how the social context may be changing since the inception of the NWFP. Approaches that use surveys to measure value orientations and a sampling scheme that allows for a representative sample would illuminate the range of value orientations throughout the Plan area and enable comparisons between urban and rural communities, among different counties, states, or subregions, or by demographic factors.

The diversity of stakeholder values, attitudes, and preferences associated with land management are a source of ongoing difficulty for resource managers. Assessing the range of social values orientations and attitudes toward forest management goals held by the public and how these values may be changing is important to inform resource management decisions. Yet, as studies have shown, stakeholders can share common attitudes or beliefs, but possess different sets of values, while some constituents who disagree about forest management practices may share common values. The relation to place can be a factor, as studies have shown differences in attitudes among stakeholders who have a specific knowledge or keen interest in a particular ecosystem, place, issue, or activity (Ford et al. 2009, Seymour et al. 2011). Understanding the relation between values and attitudes and behaviors will help resource managers understand the implications of actions and decisions on various stakeholders.

Changing relationships to the landscapes and resources in the NWFP area—

Over the past 12 years, very few studies have been conducted in the NWFP region that relate to environmental values, attitudes, or beliefs about forest management practices. National studies in value orientations and environmental attitudes demonstrate a shift away from commodity values and toward a mix of resource production and protection (commodity and noncommodity), sometimes referred to as "green drift" (Klyza and Sousa 2010, Sousa 2011). Few such studies have been conducted in the Plan area in recent years, and those that have been published are summarized below.

Beliefs about the ecological value of old-growth forests began to change in the 1970s as new science revealed important information about forest structure and composition (Spies and Duncan 2012). Steel et al. (1994) compared Oregon residents (n = 872; 75.7 percent response rate) with a national sample (n = 1,094; 68.4 percent response rate) to understand value orientations and attitudes toward forest management. The study found that respondents both in Oregon and the national sample held biocentric values more so than anthropocentric values. The study also found that respondents in the national sample held stronger biocentric views compared to Oregon residents. In other words, overall, the U.S. population leaned more toward valuing nature for the sake of nature than valuing the human use of nature. The study also found that the national sample universally opposed traditional resource management (regardless of values orientation), whereas in Oregon, primarily those with biocentric orientations opposed traditional forest management practices while those with anthropocentric values were more likely to favor policies that promote jobs and rural communities (Steel et al. 1994). These studies show that regional differences in value orientations exist.

Another study in the NWFP area showed that variations can exist at the county level. Dietsch et al. (2016) explored wildlife conservation values in Washington state (n = 4,183) in relation to wolf management. The goal of the study was to understand the relationship between modernization (urbanization, wealth, and education) and wildlife value orientations. Wildlife values were measured on a scale that examined degrees of mutualism (prioritizing the needs of wildlife) and domination (prioritizing human needs). The study found a positive association between modernization and mutualism and a negative association between modernization and domination at the county level, but variations existed among counties, with some areas exhibiting more domination values and others with a mix of values. This implies that setting influences values. In particular, counties in northwest Washington had a higher prevalence of mutualism than other regions, with the exception of one county (Shelton), which had a lower level of mutualism. Meanwhile, counties in eastern Washington had the lowest support for mutualism. Yet, one county in eastern Washington had strong support for mutualism, demonstrating that variation is not entirely based on regional setting. These results suggest that a variety of value orientations exist throughout the region.

In 2013, the Oregon Values Project, cosponsored by Oregon State University, surveyed more than 9,500 Oregonians about their beliefs related to various issues, including the environment (DHM Research 2014). Study results have not been published in peer-reviewed journals, and it should be noted that no response rate was reported and a quota sampling scheme was used (DHM Research 2014). Survey results indicated that 57 percent of Oregonians believe that environmental protection should be prioritized even at the risk of slowing economic growth, although there were variations statewide with 62 percent of metropolitan Portland respondents favoring environmental protection, compared to 50 to 54 percent in other parts of Oregon. Statewide, 35 percent said that economic growth should be given priority, even if the environment suffers. Responses also varied regionally, ranging from 30 percent in metropolitan Portland to 49 percent in eastern Oregon. Again, these results show variation in conservation attitudes among regions within a state. The study also inquired about support to increase timber harvests in forest stands that were described as "dense and overcrowded." Statewide, 53 percent were in favor of timber harvest in overcrowded stands, but responses ranged with less support from Portland (48 percent) and more support in other parts of Oregon (60 to 67 percent). It is possible that the wording of this question, framing the forests as "overcrowded" influenced responses. Still, findings suggest uniquely rural and urban patterns in values related to the environment.

One study in the NWFP area investigated public attitudes toward policies that favor environmental preservation or economic opportunity on public lands. Williams et al. (2017) explored public attitudes toward forest management in the Mount Baker–Snoqualmie National Forest, an urban-proximate forest in the northwest Cascade region. Respondents (n = 1,796) participated in an online survey and in community workshops, answering a series of questions about the importance of 26 forest management goals on a five-point scale. Water quality, wildlife habitat, clean air, aesthetics, and human-powered recreation were in the top five management goals, compared to wood (ranked 18th), energy (19th), and mining (24th). The study found few significant differences in management preferences between rural, suburban, and urban respondents. In a different study, which also featured a participatory mapping component identifying special places and resource interactions, responses of urban and rural residents were compared (McLain et al. 2017a). The study found that special places identified by urban residents were scattered throughout the entire national forest, while rural residents identified special places close to home. Resource uses among urban and rural residents were largely similar; however, rural residents were more likely to use the area for hunting and gathering foods, while urban residents were more likely to engage in active recreation (McLain et al. 2017a). While this study did not explore values, the results suggest different orientations to forests and their use.

Landscape values mapping and public participation geographic information systems (PPGIS) have been used to understand public values in the NWFP area. Brown and Reed (2009) used random household surveys of area residents to explore landscape values using a 13-item scale in three Oregon national forests in the Plan area: Deschutes/Ochoco (n = 1,916; 11.8 percent response rate), and Mount Hood (n = 1,350; 11.4 percent response rate). Based on the frequency of responses, they found the top five values to be consistent in all three forests: developed recreation, primitive recreation, aesthetic, wilderness, and biodiversity. Economic values were ranked seventh (Deschutes/Ochoco) and eighth (Mount Hood) (Brown and Reed 2009).

McLain et al. (2013a) studied landscape values for residents of Washington's Olympic Peninsula using a community workshop approach that included 169 respondents who were recruited using key informants and a snowball approach. Eight community workshops were held. Collectively, respondents identified 880 mapped places and labelled each with a primary landscape value from a list of 14. The most frequent "primary" landscape value assigned was **recreation** (56 percent), followed by **economic**, **aesthetic**, and **home**. When secondary values were combined with the primary values, recreation remained the most prominent value, followed by aesthetic and economic (Cerveny et al. 2017). These results suggest a balance of commodity and noncommodity values associated with this particular region.

We also looked at studies conducted in regions adjacent to the Plan area to understand values, attitudes, and beliefs. Hamilton et al. (2012) conducted a household survey of 1,585 northeast Oregonians and compared findings to a national sample (no response rates reported.) Although outside the Plan area, these results provide some insight into the views of rural residents in other parts of the state. This study asked respondents to rank management goals and found that northeastern Oregon residents were more likely than Americans nationwide to prioritize jobs and "use of forest resources" over resource conservation. Respondents also were more likely than the national population to believe that conservation practices and environmental rules that restrict development had negative effects on their local community. Moreover, in prioritizing a list of environmental problems facing their community, northeastern Oregonians identified "forest jobs" over a multitude of resource issues, including wildfire, insects, population growth, forest fragmentation, global warming, and overharvesting (Hamilton et al. 2012). Working in the Inland Northwest region, which includes eastern Oregon, Nielsen-Pincus (2011) conducted a household survey (n = 767) that also used an LVM approach to explore values attached to public lands. The study determined that the most important values were recreation, aesthetic, and economic. These results are similar to those found by McLain et al. (2013a) and demonstrate the mix of values that acknowledge forests for their recreation and scenic benefits, but also value income and employment opportunities associated with forests.

Changing relationships to the landscapes and resources outside the NWFP area—

Changes in environmental values in the NWFP area and the Pacific Northwest may be understood in the broader context of changes in American values. In the 1990s, scholars documented a paradigm shift in American public attitudes toward forest management away from a focus on economic values, outputs, and commodities, and toward more diverse values that include noneconomic values, especially protection of ecosystems and aesthetic values (Bengston and Fan 1999, Brown and Reed 2000, Brunson and Steel 1996, Manning et al. 1999, Rolston and Coufal 1991, Tarrant and Cordell 1997). These studies suggest that survey respondents favor a balance of protection and production in forest management. In a national study, Bengston et al. (2004) relied on computer-assisted media analysis between 1980 and 1990. The authors observed a decline in the expression of **anthropocentric** values and an increase in **biocentric** value expressions.

Shields et al. (2002) surveyed North American households and found that respondents were strongly oriented toward environmental protection, and nonconsumptive services were rated as more important than consumptive goods and services. Another study explored national forest policy decisions through the mid-1990s and noticed a shift toward greater ecological sensitivity, attributed to the success of environmental organizations disseminating information to legislators (Burnett and Davis 2002). Studies conducted in other regions of the United States, taken collectively, shed light on trends in the NWFP area, especially given the dearth of empirical studies in the NWFP area. Several studies in other parts of the country echo these national trends. Brown and Reed (2000) surveyed Alaskans and found that the most important values were aesthetic, recreation, life sustaining (ability to provide air and water), and biological. Manning (1999) found that rural Vermonters living near a national forest were more likely to identify aesthetic, ecological, and recreational values over economic values. Bliss et al. (1997) found that the public favored a balance of values but leaned heavily toward environmental protection. Collectively, these studies suggest a broader shift in American public values. Still, as Rentfrow (2010) noted, regional clusters of environmental values and beliefs exist, and caution should be exercised in conveying national trends.

A variety of studies conducted in rural, resource-dominated regions throughout the United States and Canada may shed light on value subsets of the NWFP area. It often is assumed that urban residents have a more biocentric values orientation, while residents of rural, resource-based communities are more anthropocentric. Recent studies have proven that these divisions are not clear cut. Racevskis and Lupi (2006) found that timber-dependent communities in Michigan did not uniformly fall into an anthropocentric orientation of commodity production and utilitarian use. Also, urban residents did not express a strong preference for resource protection. This diversity may be explained by inmigration of new residents with biocentric orientations into resource-dependent regions. McFarlane et al. (2011) studied forest-dependent communities in New Brunswick and uncovered a wide range of values in both rural and urban communities. Residents of forest-based communities did not always prioritize economic benefits over the natural environment, and urban communities did not always prioritize resource protection. Nadeau et al. (2008) found that urban residents in New Brunswick had strong ties to rural forest lands through family connections, woodlots, and second homes.

Amenity migration also may be associated with localized shifts in values. Jones et al. (2003) in a national study learned that urban residents are drawn to amenity-rich areas to improve their quality of life. This migration diversifies value orientations and increases potential for conflict. Smith and Krannich (2009) found more similarities than differences in environmental values among new and long-term residents in amenity-rich places in the Rockies. Fortmann and Kusel (1990) studied California communities and found that migrants to amenity-rich areas with biocentric orientations shared values with a subset of existing residents whose voices had been previously dominated by more anthropocentric views. The new arrival of urban residents led to increased conflict as long-time residents with biocentric views became more outspoken. These studies on amenity migration and shifting values present mixed results but reinforce the notion of regional variation in value orientations and attitudes. Although these studies occurred outside the NWFP area, several cities in the NWFP are facing growth in amenity migration, and results from these studies can inform our overall understanding.

Changing values around forest harvest practices— Over the past three decades, a number of studies have explored public response to forest treatments and the acceptability of various harvest practice (see Burchfield et al. 2003; Ford et al. 2009; Kearney 2001; Shindler et al. 2002, 2004). Social acceptability refers to public judgments about the appropriateness of a given management action, policy, practice, or resource condition (Allen et al. 2009, Brunson 1996). When there is a lack of public acceptance of a policy or management action, it is likely to fail or lead to conflict (Shindler et al. 2002, Wondolleck and Yaffee 2000). Social acceptability includes both individual beliefs about what is right and social norms of what is appropriate (Allen et al. 2009). Shindler et al. (2004) have identified several important themes associated with social acceptability. Social acceptability is (a) a dynamic process, (b) a result of multiple factors (ecological knowledge, prior experience, place attachment, risk perception), (c) context dependent (what is acceptable in a neighboring county may not be acceptable in my backyard), (d) process-dependent (if the process is more transparent, there is likely to be greater acceptance), and (e) based on the degree of trust among the public in land management agencies (Shindler et al. 2002, 2004).

An abundance of early research explored the scenic qualities associated with landscape treatments (See Ribe 1989 for a complete review.) This work continues with focus on alternative siviliculture treatments (Ribe 1989, Shelby et al. 2003) and scenic beauty as an indicator of social acceptability (Gobster 1996). Despite the power of visual images, judgments based on scenery can be influenced by the degree of ecological knowledge, environmental communication, and individual value orientations (Brunson and Reiter 1996). Acceptability judgments about forest harvest treatments were linked to how sites appear once practices have been implemented, how the natural characteristics of sites might change, the level of trust in information offered, perceived community benefits, and citizen engagement in the process (Olsen et al. 2012, Shindler and Collson 1998). Trust appears to be critical to social acceptability. Trust can be both broad based (trust in an agency to manage resources and serve public interests) and project based (trust that the project will not cause undue harm to the environment or change in resource use) (Ribe 2013). The public can be influenced by local political narratives and debates, perceptions of trust and justice, and fears about potentially adverse effects of management (Ford et al. 2009, Tindall 2003). Ribe (2013) emphasized that resource managers design forest treatments that express visible stewardship and public education in a way that broadens understanding of ecological aesthetics (naturalistic treatments). Existing studies about forest perceptions deal primarily with visual aesthetics and are not focused on social acceptability based on management goals, such as restoring ecosystem health.

In one NWFP study, perceptions of scenic beauty were compared among respondents grouped based on their orientation to resource conservation. Ribe (2002) used images of coastal mountain ranges to evaluate perceptions of scenic beauty as they corresponded to management acceptability among three groups: those favoring resource production, those favoring resource protection, and moderates. Respondents in Washington and Oregon (n = 1,035) rated photographic images for scenic beauty and acceptability, using fixed categories ranging from "very beautiful" to "unattractive" to label scenes based on their subjective perceptions. The authors found that all respondents (regardless of values) determined "very beautiful" scenes to be acceptable. Participants with views that favored resource production had lower standards for what is acceptable to them and what is beautiful, compared to those favoring resource protection. Those favoring resource production were more likely to perceive "unattractive" scenes as acceptable.

The potential effects of timber harvesting on ecosystems historically has been a focus of public attention and some contention in the NWFP area (Brunson et al. 1997). As Ribe (2006) observed, research on harvest practices has historically considered timber harvesting and forest preservation as two ends of a continuum (Manning et al. 1999) or positioned clearcutting against other types of forest treatments (Bliss 2000). A growing body of work has focused on what non-clearcut harvests look like and how the public responds to these treatments. New types of forestry, including ecological forestry (chapter 3 and described below) have gained momentum in the past 10 years, providing an array of options to harvest some trees in a stand in a way that sustains ecosystem function (Franklin et al. 2007). Ford et al. (2009) provided simulations of various harvest types along with information about logging plans and outcomes in Tasmania. They found environmental value orientations to be the most reliable predictor of perceptions of acceptability, with "protectionist" respondents finding clearcuts least acceptable, and selection harvests most acceptable and "productionist" respondents having the reverse pattern.

Research has shown that clearcutting is not an acceptable management strategy for a large portion of the public in the United States (Bliss 2000), and specifically in the Pacific Northwest (Ribe and Matteson 2002). This lack of support for clearcutting was also evident elsewhere (Clement and Cheng 2011). Hansis (1995) surveyed residents of northwest Oregon and southwest Washington and found general opposition to clearcutting practices, with particular opposition by women, urban residents, educated residents, and those with a liberal ideology. Meanwhile, Ribe (2006) used photographs of forest treatments to evaluate the social acceptability of various forest treatments (19 scenarios) that included combinations of age, harvest intensity, retention pattern, and down wood level. Respondents were shown four photographs per treatment type and asked to rate treatments for scenic beauty, service to humans, service to wildlife, and overall acceptability. A survey (n = 272) of western Oregon residents was conducted with the photo elicitation. The study revealed that 9 of the 19 forest treatments were of "conflicted acceptability," including all three treatments involving old-growth forests. Results also showed widespread opposition to clearcutting and some acceptance of retention harvests and forest thinning. This methodology, adapted from Ford et al. (2007), has been used in several other studies in the Pacific Northwest, with similar results (Ribe 2009, Ribe and Matteson 2002, Ribe et al. 2013).

Meanwhile, Abrams et al. (2005) conducted household surveys (stratified random sample) in Washington and Oregon studying the relationship between self-ascribed environmental or economic priorities and two variables: the acceptability of forest management practices and perceived threats to forest heath. They analyzed surveys from 492 respondents (51 percent response rate). They found that selective thinning was generally accepted by most respondents, regardless of their prioritization of policies in favor of environmental preservation or economic opportunity. Respondents with a pro-environmental viewpoint perceived human-caused factors (overharvesting, motorized vehicle use, road building, and fire suppression) as the greatest threats. Those who supported jobs and employment opportunities over environmental preservation saw naturally occurring processes (disease, wildfires) as the greatest threats.

Olsen et al. (2012) studied public opinions of alternative management strategies in the McKenzie River watershed of western Oregon, specifically disturbance-based management (DBM). The study included surveys (n = 230) of the "local attentive public" who had shown past interest in forest management issues based on attending public meetings or other events. Overall, support for DBM was mixed in the study population. The authors found that members of the public had varying degrees of knowledge about landscape-level disturbance processes or concepts, with most having low to moderate levels. In addition, they observed low levels of confidence in the information provided by agencies, and trust levels of local officials appeared to be higher than trust levels in the agency as a whole. Study participants worried that national level policies and directives would affect their communities. They also had fears about DBM being used to harvest old-growth forests. The authors suggested that transparent decision-making processes and public engagement opportunities that feature clear discussion of the risks may increase support for forest treatments.

Although outside the NWFP, a study of perceptions in the Rocky Mountains supports this trend (Clement and Cheng 2011). In a study of three national forests in Colorado and Wyoming, the researchers found that support for mechanical thinning treatments depended largely on management goals associated with those treatments. There was support for logging when it was done to protect human life and private property, to remove dead trees or insect-infested trees, or to improve wildlife habitat. However, there was less support for logging for commercial profit or for clearcutting as a management technique. These results are important because they emphasize the need for clear communication of management goals to public audiences.

Although few in number, these studies suggest that residents of the NWFP area embody a range of views related to the social acceptability of timber harvest and that these views are based on their values as well as connections to place. Although it appears that the public in the Plan area does not generally support clearcutting as a management strategy, there does appear to be potential for public support for alternative harvest strategies, such as DBM (Olsen et al. 2012), especially when efforts to expand public knowledge and share accurate information are included in the management effort. It also appears important that any harvest strategy avoid old-growth forests and old, large individual trees.

***Ecological forestry*—**Ecological forestry represents a recently emerging framework for attacking the "wicked problems" associated with forest conservation and management (Weber and Khademian 2008). The framework uses a systems approach that recognizes the interlinkages and mutually modifying processes among various entities to create a networked system. The framework also relies on ethical guidelines for managing forests around ecological objectives (Franklin et al. 2007). The approach recognizes forests as dynamic systems adaptive to new conditions and that exist as one part of a broader landscape that is managed (by multiple actors) to achieve various objectives (Batavia and Nelson 2016). This approach assumes a socioecological standpoint, acknowledging humans as part of the ecosystem and the need for integration of social and ecological elements.

The goal of ecological forestry is to sustain healthy and productive forests, retain native species, and provide a range of ecosystem services (Batavia and Nelson 2016). This goal is met by "managing forests in ways that bring them closer … in structure, function, and composition to healthy, natural forests at all stages of successional development" (Palik and D'Amato 2017: 51). Ecological forestry strives to mimic the effects of natural disturbance and succession processes, which includes retaining some elements of the existing stand (Batavia and Nelson 2016). Ecological forestry is based on (a) continuity of forest structure and function between pre- and postharvest systems; (b) structural and compositional complexity biodiversity, and spatial heterogeneity at a variety of scales; (c) carefully timed treatments based on understanding of ecological processes; and (d) planning forest management with understanding of the broader context at the landscape scale (Palik and D'Amato 2017).

Traditional forestry was based on utilitarian or anthropocentric views of forests as producing benefits for human use and consumption (Nocentini et al. 2017). Although research in this area is ongoing, presumably the ethic of ecological forestry would lean toward a biocentric orientation with timber output being a byproduct of more holistic landscape management. This approach also acknowledges humans as active ecosystem participants with specific wants and needs including a broad range of ecosystems services that forests provide. Batavia and Nelson (2016) argue that "ethics need to be institutionalized in the routine practice of natural resource management" (2016: 8). Ecological forestry also emphasizes the integration of social and ecological elements, which makes understanding of values, attitudes, and beliefs important. The framework recognizes the need for multiple actors to be coordinated and engaged around the task of integrating ecological, social, and economic sustainability and developing an ethical framework (Nocentini et al. 2017). Ecological forestry has been proposed in the NWFP area (Franklin et al. 2012); however, the practice has received limited testing, and few known studies, with the exception of Olsen et al. (2012), have evaluated the social acceptability or public attitudes toward these treatment practices. More work in this area is needed to understand the potential applications of ecological forestry in the Plan area.

Summary—

Differences in stakeholder values and attitudes are at the root of many forest management conflicts. Building consensus among stakeholders with different sets of values often is difficult and time-intensive. Values can change over time in response to major societal changes. Values and attitudes differ among geographic regions, residential classifications (urban or rural), and proximity to public lands. Moreover, national values are sometimes perceived to be in conflict with local interests, which suggests the importance of understanding the multitude of values and attitudes. Findings show that the United States has experienced a measureable values shift since the 1950s that is related to a wave of policies diminishing the importance of utilitarian values and increasing the importance of experiential, aesthetic, and biocentric values. Ongoing monitoring of public values will enhance our understanding of what is important to people vested in the NWFP area. Awareness of these values shifts allows resource managers to consider public needs in planning and decisionmaking and allows managers to anticipate conflict and consider diverse communication strategies. Land managers who acknowledge the diversity of values, attitudes, and beliefs among stakeholders and socioeconomic groups at the appropriate geographic scale will be better equipped to understand characteristics of the social system and anticipate the need for change. Growing understanding of human-resource connections can strengthen relations between agencies and communities and contribute toward trust building.

Valuing Place

The NWFP's signature characteristic is its focus on ecosystem management, a management approach that is fundamentally place based (Williams et al. 2013). Place has increasingly been used as a concept in national forest planning and public engagement efforts (Farnum et al. 2005, Kruger and Williams 2007, Williams et al. 2013). The term "place" embodies both biophysical characteristics and sociocultural meanings that are critical to quality of life and social identity. This section describes current research related to place and lessons learned of relevance to forest planning and plan implementation in general, and to forest plan revisions within the NWFP area.

Defining place—

Places are not merely geographic locations but rather are produced when individuals and groups assign value or meaning to undifferentiated space (Tuan 1977). Places and the meanings that one attaches to them help people to make sense of the world and motivate the actions they take with respect to particular locales (Sack 1992). Sense of place, or "the perception of what is most salient in a specific location" (Cantrill and Senecah 2001: 187), is manifested in our views about the kinds of activities and uses we consider acceptable in that location.

The tools and conceptual frameworks for assessing and inventorying place meanings in natural resource settings are still in the early stages of development. Studies about the roles that place plays in environmental and land use management have examined the factors that contribute to the production of place (Ardoin 2014), the role of place in the formation and maintenance of self and group identities (Twigger-Ross and Uzzell 1996), the ways in which place meanings connect people to particular landscapes or ways of life (Davenport and Anderson 2005, Kil et al. 2015), and how place meanings are mobilized to support or contest land management or economic development objectives (Stokowski 2002). Places are multidimensional and produced through a melding of the individual and group histories, memories, values, and beliefs associated with a locale and its biophysical attributes (Ardoin et al. 2012, Cheng et al. 2003, Jorgensen and Stedman 2001, Stedman 2003).

It is generally understood that sense of place has three major elements: (1) a biophysical setting (array of physical features and amenities embodied in a particular place); (2) the individual meanings associated with the location, produced through a combination of individual personality traits and lived experiences; (3) and the sociocultural or shared meanings linked to that location (Smaldone et al. 2005). Of these, only biophysical features are relatively straightforward for land managers to assess and integrate into planning. Yet, the individual and cultural meanings associated with specific locations are equally important to understand if politically viable environmental policies and management actions are to be implemented under the revised NWFP or other management plans.

A place meaning is the significance that people assign to places (Davenport et al. 2010). Place meanings can be positive or negative, specific to an individual, or shared within and across groups (Scannell and Gifford 2010). Place meanings are critically important to understanding people-place relationships, which in turn influence whether policies and management actions will have broad-based acceptance among interest groups and the general public. Assessing place meanings, however, is challenging in part because although relatively stable, they are not static as individuals and groups respond to changes in their social and physical surroundings (Smaldone et al. 2005: 397; Williams 2002: 17). Over the past decade, social scientists have developed conceptual frameworks and practical tools that managers can draw upon to understand the type and intensity of connections that different segments of the public have with places in their management jurisdiction (fig. 9-3).

Key concepts: place attachment, place dependence, and place identity—

Place researchers often distinguish between three concepts linked to the notion of place (box 2): place attachment, place dependence, and place identity, with place dependency and place identity considered subcomponents of place attachment (Anton and Lawrence 2014). Understanding the difference between these three concepts is important for resource managers because they shape how different segments of the public are likely to respond to proposed policy changes, such as revisions to the NWFP as well as proposed management actions.

Place attachment is the process by which individuals or groups become connected, whether emotionally or for instrumental purposes, to a specific geographical location

Lee Cerveny

Figure 9-3—Observing the surf from the Siuslaw National Forest.

Box 2—Key Concepts About Place

Place attachment—people develop strong connections to a place based on repeated experiences and in-depth knowledge of that place.

Place dependence—people have places they rely on to provide services and products that sustain their livelihoods or lifestyles or provide desired experiences.

Place identity—people sometimes have places that have such deep symbolic meaning (cultural, historical, spiritual) that those places help define who they are in the world.

(Scannell and Gifford 2010). For groups, place attachment is considered "a community process in which groups become attached to areas wherein they may practice, and thus preserve, their cultures" (Scannell and Gifford 2010: 2). Empirical research on place suggests that strong positive person-place bonds can motivate individuals and groups to care for, protect, and defend particular places or types of settings (Eisenhauer and Kra 2000, Kil et al. 2014, Stedman 2002). Place attachment also is important because it is sometimes associated with negative social and environmental outcomes (Lewicka 2011, Yung et al. 2003). For example, strong attachments to place may lead to intense conflict between long-time residents and newcomers who bring with them very different ideas about what uses and activities are desirable for that place (Hurley and Walker 2004). Indeed, the conflicts over timber that led up to the NWFP arguably were partly struggles between two groups with very different, but equally strong, attachments to place. Proctor (1995) described how disagreements in the early 1990s between loggers and environmentalists were linked to their very different experiences and connections with the Pacific Northwest forest landscape. A regional socioeconomic assessment of the impacts of the NWFP found that feelings of a loss of cultural identity were common among residents in rural Oregon and Washington communities that had been heavily affected by the shift in forest management practices brought about by the NWFP (Charnley et al. 2008).

Studies of place attachment among transient residents and tourists indicate that even short-term visitors can develop strong attachments to places such as seasonal homes, parks, or natural areas (Lewicka 2011). Evidence is mixed, but overall, it appears that visitors with stronger local social ties or who visit more frequently develop stronger place attachments (Lewicka 2011). People can become attached to places that they have only heard about or imagined, a concept that Kruger (2008: 2) refers to as "existence attachment." Just as people can have an "existence value" (a willingness to pay to ensure someplace exists even though they have never visited) for resources, so too can people develop attachments to places they have never visited (Kruger 2008). Attachment to places not visited has important management implications for NWFP implementation as it points to the need for land managers to take into account the place meanings of stakeholders who do not use an area, as well as those who do.

Place dependence has to do with the "importance of place in providing features and conditions that support specific goals or desired activities" (Ujang and Zakariya 2015: 712), and is related to how well the physical characteristics of a place fulfill an individual's goals and needs (Scannell and Gifford 2010). The better the conditions at a place meet a person's needs or goals, the more attached that person is likely to be to that particular location. The degree to which an individual is place dependent also hinges upon how well the quality of a place they are currently using compares with the quality of potential substitute places (Smaldone et al. 2005). However, the **meanings** associated with the physical features of a place may be what cause people to value that place rather than the features themselves (Stedman 2003). Changes in biophysical features as a result of forest management actions or policies may trigger strong negative reactions among those segments of the public for whom that particular suite of biophysical characteristics is imbued with deeper meaning.

In some circumstances, bonds to places or settings are so strong that those places become intimately bound up with the person's or group's core sense of self (i.e., personal or social identity), a phenomenon known as **place identity** (Proshansky et al. 1983). Place identity is closely linked with the symbolic meanings of place rather than its utilitarian values and "is based on the notion that places serve various functions in identity development that promote a sense of belongingness" (Davenport and Anderson 2005: 628). In situations where the symbolic meanings of place are spiritual in nature, such places are sometimes viewed as sacred in the eyes of those for whom they have spiritual meaning. In the late 20th century, so-called old-growth forests of the Pacific Northwest became imbued with religious meaning for many Americans (Lee 2009, Proctor 2009), and efforts to protect what many people had begun to see as sacred forests arguably contributed toward policies such as the NWFP.

Droseltis and Vignoles (2010) described the distinction between place attachment and place identity, a subset of place attachment, as the difference between a place where someone feels "at home" (place attachment) and a place that one feels is a fundamental part of one's self (place identity). When individuals identify with a place or have a particularly strong attachment to it, place disruptions, or changes in the fit between place meanings and its physical and social characteristics, may lead to feelings of severe anxiety and loss (Devine-Wright 2009, Proshansky et al. 1983, Stedman 2002, Twigger-Ross and Uzzell 1996). Denial, detachment, and taking part in place-protective actions, such as forming protest groups or signing petitions against proposed changes, are among the strategies used by individuals and groups to cope with threats to place meanings (Devine-Wright 2009). The resource conflicts associated with the development of the NWFP are just one example of the intense social tensions that can emerge when place identities are threatened. Proactively identifying which places (or types of places) are likely to trigger large-scale place identity crises if they are fundamentally changed through forest management actions is one strategy that managers could use to reduce the likelihood of major land use conflicts and intense polarization. Like social identities, which are generally relatively stable but which can change under some circumstances for some individuals (Amiot et al. 2015, Carlsson et al. 2015, Cohen and Sherman 2014, Miller and Caughlin 2013, Perozzo et al. 2016), place identities tend to be stable but can change as individuals and groups have new experiences or engage in dialogue with others for whom a place has different meanings (Coen et al. 2017, Wheeler 2017).

Salience, or the "probability that an identity will be activated in a situation" (Stets and Burke 2000: 229), is an important concept in social identity theory that has implications for how place identity can provide the seed for constructive collaboration as well as conflict (Bryan 2008). Social identity is "a person's knowledge that he or she belongs to a social category or group" (Stets and Burke 2000: 225). Characterization, another important social identity theory concept, is "what an individual or group perceives another individual or group to be" (Wondolleck et al. 2003). All individuals' social identities are derived from membership in multiple categories (Stets and Burke 2000). Which social identity is salient for an individual or group depends on the social context, or the degree to which an individual perceives that a social category they have characterized themselves as fits with reality (Turner 1987). As described earlier in this section, geographical context can serve as the basis for social identity, with place identity arising from the link between groups of individuals and specific locales (Proshansky 1983, Wondolleck et al. 2003).

Social identity theory further suggests that "conflict derives in part from social group comparisons in which in-groups portray themselves (identity) more positively and out-groups (characterization) more negatively" (Bryan 2008: 54), processes known respectively as identity or characterization framing (Wondolleck et al. 2003). Identity and characterization framing can be used to describe the roles that an individual plays without assigning judgment, to draw connections with others, or to distinguish one's self or one's group from others (Wondolleck et al. 2003). The Quincy Library Group is a place-based collaborative planning group that emerged in California's northern Sierras in the 1990s in response to a major reduction in timber harvested on federal lands. The Quincy Library Group helped shift participants' salient identities from the previously conflictual identities of "logger," "environmentalist," or "Forest Service employee" to a common identity linked to place, i.e., "resident of Plumas County" (Bryan 2008). A similar process of identity reframing where "us vs. them" moved toward "we" occurred in the Applegate Valley of southwestern Oregon during the same period (Rolle 2002).

Place-making is a political process (Manzo 2003, Yung et al. 2003) and some natural resource conflicts are as much struggles about place meanings as they are about how those resources should be allocated (Cheng et al. 2003). When place meanings are threatened by prospective land management actions, groups or individuals whose identities are tied to them may try to defend those meanings or create new ones (Hurley and Walker 2004, Manzo 2003). Through the process of place creation and maintenance, individuals and groups promote their values and beliefs about what landscapes should look like, what activities should take place where, and who belongs (or does not belong) in particular places (Cheng et al. 2003). Understanding the dynamics of the politics of place can provide managers with insights on the fundamental issues underlying natural resource conflicts and facilitate the development of natural resource decisionmaking processes that are less contested (Austin 2004, Kemmis and McKinney 2011, Yung et al. 2003).

People often use symbols, myths, and narratives as tools for supporting or resisting place claims (Cheng et al. 2003, Stokowski 2002). Such techniques typically rely on the "moral language of ecology or community" (Williams 2002: 21). To understand conflicts over place meanings—and take a step toward potentially finding solutions to those conflicts—it may be helpful to pay attention to the language and stories that different stakeholders use to create and maintain place meanings (Stokowski 2002, Yung et al. 2003). During the past two decades, collaborative forest management groups operating in the NWFP region have provided new venues where stakeholders with diverse interests can create shared meanings and common ground as to what activities are considered acceptable in particular locations (Moseley and Winkel 2014). However, Yung et al. (2003) point out that in contexts of intense resource conflict, multiple and incompatible senses of place often lie at the heart of the conflicts. In such contexts, creating shared meanings will be challenging, and in some cases, impossible. Managers may find it useful to develop the capacity to identify when collaborative management is likely to be a successful strategy for creating shared meanings and when other strategies are called for. A rich body of research on place-related concepts has emerged over the past 20 years. However, examples of how place-related concepts have informed the design and implementation of planning processes or how data regarding place meanings, attachment, identities, or dependence have been used in planning or management processes are rare.

Public participation GIS and how "place" connects to participatory mapping—

During the past decade, public participation GIS (PPGIS) has increasingly been used as an approach for collecting data about place attachment, place dependence, place identity and other place-related constructs (McLain et al. 2013b). PPGIS links computerized mapping technology with broad-based public participation processes to generate spatial data about human-environmental connections. The discussion of LVM studies earlier in this chapter focuses on how PPGIS has been used to study values. However, PPGIS can also help clarify understandings of place meanings (McLain et al. 2013b: 652). Maps created from these data show how place meanings are distributed across the landscape, and spatial analyses can help identify how place meanings are related to certain habitat types, landforms, or other biophysical features (Brown and Brabyn 2012). These tools could be useful to land managers to improve their understandings of the types and intensity of place meanings that different segments of the public associate with forested landscapes.

In the United States, PPGIS is typically structured as a data collection process, with the goal of expanding the opportunities the general public has for providing input into environmental planning processes (Brown et al. 2014). However, in some contexts—primarily in developing countries and among indigenous peoples in industrialized nations—PPGIS is structured so that mapping participants have an opportunity to design the mapping process, analyze alternatives, and empower individuals to have a voice in decisionmaking (Sieber 2006). Public participation GIS has been used to identify places where social and ecological hotspots are co-located (Alessa et al. 2008), measure changes in place values over time (Brown and Donovan 2014), and understand place meanings associated with forested ecosystems (Gunderson and Watson 2007, Lowery and Morse 2013). However, national forests have been slow to adopt PPGIS (Brown 2012). Brown (2012) attributed the lack of interest in PPGIS on the part of the U.S. Forest Service to organizational culture and regulatory barriers, including the lack of directives calling for the collection of data on place meanings, lack of capacity within the agency to collect and analyze such data, uncertainty about whether such data are considered scientifically valid, and the difficulty of getting approval from the Office of Management and Budget (OMB) for collecting such data.

Place-based planning—

Interest in place-based planning emerged in the late 20th century as resource management shifted from single-species or dominant-use management toward integrated and holistic systems approaches aimed at managing for a diverse set of ecological and human values (Potschin and Haines-Young 2013, Williams et al. 2013). Lowery and Morse (2013: 1423) defined place-based planning as "a process used to involve stakeholders by encouraging them to come together to collectively define place meanings and attachments." Other scholars view place-based planning as a process that fosters social learning and adaptive management at the scale of the place of interest to the community engaged in planning (Cheng and Mattor 2010, Farnum et al. 2008). The degree to which place-based planning tends more toward information gathering or more toward social learning and participatory adaptive management differs considerably. Most PPGIS efforts fall into the information-gathering category (McLain et al. 2013b); forest collaborative planning processes focus more on social learning (Davis et al. 2017).

Place-based planning is site-specific and takes into account both social and biophysical contexts (Potschin and Haines-Young 2013, Yung et al. 2003). Place-based planning differs from locally based participatory planning in that place-based planning focuses around a particular geographical area or place but may include nonlocal participants, such as members of regional or national interest groups (Moseley and Winkel 2014). Yet places do not exist in isolation from each other (Flint 2013). Consequently, place-based planning must factor in the socioecological connections that link bounded places to the broader realm in which they are situated (Flint 2013). This might take the form of establishment of a regional or national group composed of participants who are also active in planning at more local levels, and which therefore provides opportunities for the sharing of planning or management priorities and socioecological knowledge across scales (Flint 2013).

Place-based planning acknowledges "the multiple relationships people have with geographic locations, relationships that encompass livelihood and economics, **and** values, symbols, emotions, history, and identity" (Yung et al. 2003: 856). To identify these multiple uses, values, and meanings, placed-based planners purposefully set up opportunities for stakeholders coming from multiple perspectives to engage in constructive dialogue with each other (Kruger 2008). Through the conversations that take place between stakeholders, place-based planning reveals the diversity of meanings that people attach to different parts of the planning area. Moreover, through dialogue about those place meanings, participants can engage in place-making, which in some situations may enable them to create a "shared image of place" (Patriquin and Halpenny 2017: 5). Even when place-making is not the goal of place-based planning, knowledge of which meanings are associated with which geographic locations can help managers identify when proposed management actions are likely to be contentious and how management actions might be structured so as to minimize the likelihood or intensity of conflict (Yung et al. 2003). It is important to recognize that the participatory nature of place-based planning will likely create expectations among the public that their recommendations will be incorporated into decisions; these expectations need to be acknowledged and managed (Bruña-Garcia and Marey-Pérez 2014, McCall 2003).

Cheng and Kruger (2008) describe a place-based planning project on the Grand Mesa, Uncompahgre, and Gunnison National Forests in which a multi-stakeholder participatory mapping approach was used. The working groups first expanded the range of management options on the table by developing thematic landscape units, categories of land that included a much broader set of values and uses than were included in the forests' traditional management units. The thematic units were places that participants identified as being significant for a combination of social and ecological reasons, and which took into account the special or unique features of those areas as well as future conditions participants envisioned for those parts of the landscape. The themes varied from natural conditions only to permanently altered areas. Maps were used as a starting point for dialogue, and mapping exercises were structured around stakeholder-derived categories, which revealed interdependencies in uses and values at landscape scales (Cheng and Kruger 2008). Although the process provided opportunities for social learning, some stakeholders felt that meaningful participation was hindered by the management framework imposed by the U.S. Forest Service. Moreover, the use of technical language during the meetings functioned as a barrier to widespread participation. And, some stakeholders accustomed to issues-based planning resisted the idea of place-based planning (Cheng and Mattor 2010).

Issues-based planning focuses attention on outputs of individual uses (i.e., timber production, wilderness, recreation, wildlife habitat), and stakeholders organize their participation in planning around "protecting and increasing the output of their favored uses while opposing the output of other uses that are perceived to interfere with their own uses" (Cheng and Mattor 2010: 397). In contrast, place-based planning focuses on acquiring a broad-based understanding of the meanings associated with particular parts of the landscape and managing so as to maintain or create a particular sense of place (Cheng and Mattor 2010). Presumably through the process of place-based planning, participants revise their expectations as to what outputs can be derived from the planning area. However, a report on forest restoration occurring as a result of the Quincy Library Group planning process mentioned earlier in this chapter found that timber production goals fell short of what the timber industry participants in the group had hoped to achieve (Pinchot Institute for Conservation 2013). A more detailed discussion about the challenges of place-based collaborative planning is provided later in this chapter.

Another challenge associated with place-based planning is the difficulty in scaling locally successful planning processes up to regional and national scales (Potschin and Haines-Young 2013). Moreover, local-level data required for planning are often inadequate or unavailable (Potschin and Haines-Young 2013). Additionally, place-based planning can be costly in terms of the time and resources needed to involve a diverse set of stakeholders in deliberative planning processes over a sustained period (Cheng and Mattor 2010). In the NWFP area, the most salient examples of place-based planning are the forest-level collaborative planning groups that have emerged since the mid-1990s (Moseley and Winkel 2014). Many of these collaboratives emerged out of a desire to find common ground through creating a shared sense of place, partly as a means to reduce tensions perceived as unproductive. The collaboratives and their relationship to the NWFP are described in greater detail later in this chapter.

Studies about place in the NWFP region—
We located several studies that focused on or incorporated elements of place and place-based planning from the NWFP area conducted since 2003. Using a psychology-of-place approach, White et al. (2008) looked at the relationship between place identity and place dependence on visitor perceptions of ecological, social, and depreciative impacts (i.e., littering, vandalism, dumping garbage) linked to recreation activities in the Molalla River Corridor Recreation Area and Table Rock Wilderness in western Oregon. They also looked at the relationship between the length of time visitors had been coming to the area and the intensity of their place identity and place dependence. They found no association between place identity or place dependence and perceptions of recreation-linked social, ecological, or depreciative impacts. However, individuals who had been coming to the recreation area longer had higher levels of place identity and, to a lesser extent, place dependence. Specifically, White et al. (2008) found that visitors' sense of place identity increased by 7 percent on the five-point scale used in the interviews for every additional year they had been coming to the site. One important implication of this study for forests in

the NWFP area is that longer term recreation users (and likely other types of forest users as well) are likely to have stronger attachments to particular locations, and are likely to react negatively to any management actions that change those places unless they have a voice in the planning processes that lead to those changes.

Rudestam (2014) examined links between sense of place, regional identity, watershed perceptions, and water-use behavior in the Willamette River basin. She found that landowners consistently described the water supply as being limited and scarce, belying the region's reputation for excessive rain. Although most interviewees articulated deep connections to water in the Willamette basin, few were willing to change their water-use behavior. A take-home lesson for planners is that strong place attachments are not necessarily associated with actions that improve the ecological conditions at a particular location, and that other incentives may be required to encourage ecologically beneficial behaviors.

Cheng and Daniels (2003) looked at how geographic scale and ways of knowing about watersheds are linked in place-based collaborative planning venues in the McKenzie River valley. They found that participants in the watershed group working at a smaller geographic scale were much more place oriented than their counterparts that covered a larger area. They concluded that people know places in multifaceted ways, and the scale at which a collaborative group operates affects place knowledge. However, because participants differed between the two groups, the extent to which the study's observed differences in place orientation can be attributed to scalar differences rather than differences in participants is unclear.

One of the challenges of place-based planning is the mismatch between traditional administrative boundaries and the way in which people inhabit places. Farnum et al. (2008) describe an effort by the Willamette National Forest to develop a set of place-based planning units corresponding to three geographic scales: an overarching "social resource unit" made up of three "human resource units," each of which in turn was composed of several "community resource units." The project was undertaken as a proactive step toward identifying community priorities, but the data and analytical tools it produced were never integrated into the forest's planning or assessment processes. The authors attribute this to a combination of factors, including managers' reluctance to accept anthropological data as "scientific," loss of support for the project owing to leadership turnover, and the lack of planning directives calling for this type of analysis. Brown and Reed (2009) also identified a serious gap in the U.S. Forest Service's capacity to incorporate data about place meanings into its planning processes. Given that place meanings can significantly affect whether forest policies and management actions are viewed as socially acceptable, filling this gap in agency capacity would be one way to reduce controversy and build stronger partnerships and collaborations. The discussion of agency capacity in chapter 8 helps to illuminate the challenges and opportunities that exist to build partnerships.

McLain et al. (2013a, 2017b) conducted a study that mapped meaningful places on the Olympic Peninsula. The authors found that east-side residents on the Olympic Peninsula differed noticeably from west-side residents in how they mapped their meaningful places (McLain et al. 2013a). The west-side residents drew much larger polygons, often covering entire watersheds, while east-side residents typically used smaller polygons, points, or lines, to mark places. The authors speculate that the differences in the sizes and shapes of meaningful places reflect differences in how the two groups connect with and use the landscape, as well as topographical differences. The mapping study also revealed social identities linked to residents' relationship with place, particularly in the western part of the peninsula, which has historical roots in the timber industry (McLain et al. 2017b).

Todd (2014) collected data on meaningful places from Olympic Peninsula visitors. Intercepts were done at major trails, campgrounds, and visitor centers as well as on the ferries. Todd's research showed that the visitors' meaningful places tended to be located in Olympic National Park. In contrast, the places marked by residents in McLain et al.'s (2013a) study were heavily concentrated on the Olympic National Forest or on state trust lands. Todd also found that less-frequent visitors tended to map fewer places, and the places they mapped were generally limited to the major tourist destinations. More frequent visitors and locals mapped more places and covered a broader geographic range. These results suggest differences in stakeholder connections to the area based on visitation frequency and residency.

McLain et al. (2017a) explored special places and associated resource uses on the Mount Baker–Snoqualmie National Forest as part of a study in support of travel management planning (USDA FS 2015c). Among other findings, this study showed that special places for rural residents tend to be more concentrated close to home, while urban residents identified special places with more geographic diversity. Resource uses were similar between urban and rural residents, with hiking being the predominant activity; however, urban forest visitors were more likely to engage in strenuous recreation (mountain biking, backpacking, climbing) while rural residents were more likely to be involved in hunting and berry picking, which are important both for food, lifestyle, and recreation (McLain et al. 2017a).

The projects by Farnum et al. (2008), McLain et al. (2013a, 2017a), and Todd (2014) resulted in the development of methods useful for identifying the range of ways that people connect with particular landscapes, information that can help guide forest planning and management actions. However, the process by which this information is then considered and incorporated will ultimately determine whether tradeoffs are acceptable and conflict minimized. Todd (2014) showed that residents and visitors have very different relationships to their landscape, underlining the importance of ensuring that efforts to inventory place meanings are structured in ways that capture place meanings from a broad spectrum of forest users. Moreover, McLain et al. (2017a) noted differences in landscape connections between urban and rural stakeholders.

Brown and Reed (2009) observed that differences in locations of special places differed depending on familiarity with the forest, whether the respondent worked in the forest products industry, and membership in an environmental organization. They concluded that the location of special places differs by subgroups, and recommended the use of multiple data collection approaches (Internet, mail survey, meetings). Barriers they identified to the use of LVM in forest planning included (a) lack of directives specifically mentioning collecting data on landscape values and special places, (b) costs associated with conducting surveys, (c) difficulties with getting approval from the OMB to administer surveys, (d) unfamiliarity of Forest Service personnel with this approach, and (e) uncertainty about whether LVM data will stand up in court.

Regional studies of place and place-based planning— Given the small number of studies falling within the NWFP area, we also examined studies that took place in the broader region. These include one study from eastern Washington, one from the region where Idaho, Oregon, and Washington intersect, and one from the Sierra Nevada region of California.

Nielsen-Pincus et al. (2010) drew on psychology of place theory to examine whether place identity and place attachment differed between local and absentee property owners in three rural counties in northeastern Oregon and northern Idaho. They found the models could not distinguish between place dependence and place attachment and concluded that at landscape scales, the two may be indistinguishable. Their study also showed that place identity was slightly stronger among local landowners when compared with absentee landowners, but not enough to be meaningful. Findings suggest that place identity is likely more influenced by self and social identity than by day-to-day experiences. For place attachment, their analyses showed that the number of months spent in the place each year was more important than the amount of time spent in residence. This study points to the value of ensuring that planning processes are structured in ways that include long-term seasonal residents, as well as year-round residents.

Donovan et al. (2009) captured the full range of landowner and stakeholder views about the landscape in the Palouse region of eastern Washington, and overlaid the resulting maps on ecological and land cover GIS layers. They asked participants to assign one value to each mapped location, but found that participants resisted this restriction, wanting to assign multiple values, which is consistent with previous findings, that multiple factors draw people to a place (Cerveny et al. 2017, McLain et al. 2013a). The values mapped fell into two distinct clusters: (a) historical/cultural/agriculture/private land; and (b) outdoor recreation/natural diversity/scenic views. Donovan's study points to the importance of using methods that can capture and adequately describe a range of place meanings. Practically, this implies that few places have just one meaning, even for individuals, and that it may be the suite of meanings that needs to be maintained in order for management actions to be socially acceptable, rather than just a dominant meaning.

Brown (2013) piloted a Google Maps™ values mapping application on the Sierra, Sequoia, and Inyo National Forests in northern California, using both a LVM survey and volunteered data. Outreach targeted diverse stakeholders, including a conservation group, forest industries organization, and resource managers. Brown et al. (2014) also asked respondents to map acceptable and unacceptable forest uses. Comparing survey data from randomly selected households with Web link respondents, Brown et al. concluded that the volunteer-Web mappers had mobilized to ensure that their values were strongly represented, concluding that PPGIS practitioners should not assume that the data received through open Web links are representative of the general public's views (Brown 2013, Brown et al. 2014).

Collectively, these studies have important implications for forest plan revisions in the NWFP area and subsequent implementation: (1) place meanings are likely to differ for different subgroups of the public (i.e., visitors, residents, rural, urban), (2) methods used to collect place-related data differ in terms of the types of publics that they are likely to reach, (3) use of multiple data collection approaches can help to diversify participation, which allows a broader range of place meanings to emerge, (4) institutional barriers exist within the Forest Service (and likely within other land management agencies as well) to the collection and use, and long-term storage of social science data, and (5) challenges in the agency's ability to collect and use place-based data may hinder the agency's capacity to develop socially acceptable policies, plans, and management actions.

Summary—

People have the capacity to derive symbolic meanings and develop emotional ties with outdoor places. Place meanings, whether derived through stories, histories, or experiential knowledge, have implications for forest ecosystem management. The positive power of place motivates people to engage in forest stewardship projects, planning processes, and collaborative groups. The variety of place meanings held by diverse stakeholders suggests the need for broad-based public engagement processes. Because place meanings are dynamic and constantly being renegotiated, a public engagement process that emphasizes multiple ways of gathering information about place meanings and that is deliberately designed to reach out to a broad spectrum of the public is far more likely to capture the range of meanings than processes that rely on only one approach.

Cultural Ecosystem Services

Ecosystem services describe the wide range of benefits that forests and landscapes provide to people and that help to sustain human life (Brown et al. 2007). Ecosystem services provide a comprehensive and holistic framework for considering and evaluating multiple resource benefits (MEA 2005). The significance of ecosystem services for resource governance in the United States is becoming increasingly evident. A presidential memorandum issued in 2015 directs all federal land managers and regulatory agencies to use an ecosystem services framework for planning, policymaking, and decisionmaking (OMB 2015). Consideration of ecosystem services also is mandated in the national forest planning process under the 2012 forest planning rule (USDA FS 2012). Ecosystem services is a category for consideration in the forest assessment phase, although studies of early adopter forests demonstrate an uneven treatment of the ecosystem services principles (Ryan et al., in press). For more discussion of ecosystem services, see chapters 8 and 12.

Key concepts—

The Millennium Ecosystem Assessment (MEA) defined cultural ecosystem services as "the nonmaterial benefits people obtain from ecosystems through spiritual enrichment, cognitive development, reflection, recreation, and aesthetic experiences" (MEA 2005: 5). Many of these human benefits are intangible, such as spiritual benefits, cultural benefits, symbolic benefits, or heritage benefits (de Groot et al. 2010, MEA 2005). Cultural ecosystem services are the products of people's interactions with landscapes and ecosystems (Chan et al. 2011, Fish et al. 2016). They are shaped by direct human perceptions and senses (Braat and de Groot 2012) and are further shaped by human values, norms, and beliefs (Fish et al. 2016). Cultural ecosystem services can inspire "'deep attachment" between communities and landscapes (Chan et al. 2011) and serve as points of entry for public involvement processes related to ecosystem management (Daniel et al. 2012).

Cultural ecosystem services (also referred to as "cultural services") have proven to be challenging to operationalize and measure (Hernández-Morcillo et al. 2013). Analysis of cultural ecosystem service indicators has found that some are more readily captured, such as education and recreation, while others are more difficult to quantify or are often conceptualized inconsistently (de Groot et al. 2010, Hernández-Morcillo et al. 2013). Efforts to assign value to both tangible and intangible aspects of cultural services have been fraught with challenges; new methods of operationalizing cultural services are sought (Daniel et al. 2012, Hernández-Morcillo 2013, Plieninger et al. 2013). Cooper et al. (2016) observed that aesthetic and spiritual values are frequently mentioned in MEA reports as important, but there has been very little research to explore how these values may be best characterized, operationalized, quantified, or measured (fig. 9-4). Cultural services are rarely considered in ecosystem

Keith Routman

Figure 9-4—Dawn on the Hoh River, Washington.

services assessments, or if they are included, they often are given a cursory treatment (Feld et al. 2009). Some cultural services are considered vague and difficult to capture or quantify. And, cultural services are not always singular, but are intertwined or "bundled" with other services. As Klain et al., 2014) noted, Pacific salmon have cultural value as well as provisional value to Northwest coast indigenous people. They can be quantified based on price, but this ignores the spiritual value of salmon (Klain et al. 2014). Because of the lack of conformity of cultural services to a reliable metric, there has been a tendency to relegate cultural values to an afterthought, which has impacts for stakeholders who rely on ecosystems for a variety of cultural benefits (Chan et al. 2011). New studies are investigating ways to capture cultural services (Bryce et al. 2016, Daniel et al. 2012). Cultural services should not be overlooked because they play an important role in building public support for ecosystem management (Plieninger et al. 2013). An analysis of ecosystem services that does not fully maximize the measurement of cultural services is incomplete.

Managing for cultural services—
The MEA framework provides a useful template for land managers to consider the vast array of ecosystem benefits and to prioritize benefits for their management unit. The identification of cultural services as a critical component of that framework encourages even greater attention to the less "tangible" benefits associated with forest ecosystems, which often get overlooked in the planning process or when identifying forest management objectives and targets. The research on cultural services is emerging, and there have been some attempts to develop a management framework (see Fish et al. 2016).

Several studies have explored how PPGIS can be used to explore cultural services for use in land and resource planning (Brown and Fagerholm 2015, Brown et al. 2012, Bryan et al. 2010; Klain and Chan 2012, Plieninger et al. 2013, Raymond et al. 2009, Sherrouse et al. 2011). Mapping stakeholder preferences provides understanding of how cultural services attach to places on the landscape (Fagerholm et al. 2012). Klain et al. (2012) found that it was much more common to identify areas that were associated with tangible values (recreation, cultural heritage, aesthetics) than intangible values (spiritual, sense of place, identity). Recreation values are often associated with developed recreation facilities, just as cultural heritage values can be evaluated by the number of heritage sites in a landscape and scenic areas can be used as a proxy for aesthetics. However, it may be more difficult to operationalize concepts like "social identity" or "sense of place," which are typically measured through qualitative investigations. More research is needed to understand the distribution of cultural services across landscapes and implications for resource managers. Potentially, PPGIS would be useful to explore cultural services spatially.

A useful framework for investigating cultural ecosystem services was developed by Fish et al. (2016), who created four categories of cultural ecosystem services: **environmental spaces** (localities, places, landscapes where people and nature interact); **cultural practices** (symbols, signs, interpretation, and other expressions about the relations between people and nature); **cultural benefits** (areas where human health and well-being are linked to interactions between people and forests, such as spirituality, inspiration, freedom); and **cultural goods** (or services), where the interaction between people and nature result in market transactions or other exchange that results in income or other benefit (e.g., guiding, tourism, sporting events, festivals). This framework may be useful for exploring the diverse human connections of forests in the NWFP area.

The Forest Service has begun to use an ecosystem services framework to describe forest values (monetary and nonmonetary) provided by public lands (Deal et al. 2017). Forests using this framework have found it helpful to identify relevant ecosystem services for their forest, assess tradeoffs among services associated with proposed forest treatments and management activities, and engage partners who share mutual benefits from particular services (Deal et al. 2017). Several projects in the NWFP area incorporated an ecosystem services framework as a way to assess benefits, develop metrics, and monitor outcomes for a particular planning area (Deal et al. 2017, Smith 2014, Smith et al. 2011). One project in the Big Marsh area of the Deschutes National Forest emphasized the tradeoffs between water quality, aquatic habitat, recreation activity, and mushroom harvest, to name a few (Smith et al. 2011). Another project involving active forest management on the Williamette National Forest engaged tribes to identify values associated with timber harvest, cultural heritage, recreation, wildlife, water quality, and harvest of special products (i.e., huckleberries, beargrass).

Two empirical studies explored public perceptions of ecosystem services in the NWFP area. Asah et al. (2012) investigated how people identify and construct forest ecosystem services in Deschutes County, Oregon. Results revealed that the public view of ecosystem services is similar to the MEA framework, with some notable differences. Although this framework categorizes mushroom picking and Christmas tree harvest as "provisioning services," local residents view these both as provisioning and cultural services, providing opportunities to nurture social relationships and develop forest connections. The study also revealed that respondents viewed the national forest lands as both a source of affordable housing (temporary residence) and as a hedge against urban sprawl (Asah et al. 2012).

In a related project, Asah et al. (2014) investigated perceptions of ecosystems benefits by the Confederated Tribes of Warm Springs. Respondents emphasized both provisioning services (especially fish, game, and plants) and cultural services (especially spiritual, aesthetics, and place attachment), with less attention to regulating and supporting services (Asah et al. 2014). They also described direct and indirect connections between provisioning and cultural services, whereby the gathering of provisions provides an opportunity to solidify ties among tribal members and strengthen intergenerational connections. Tribal members emphasized items as cultural services that are not featured on the original MEA list, including sense of place, sense of community, and political license to exercise historical tribal rights (Asah et al. 2014). More research in the NWFP area is needed to understand public perceptions of cultural services. Chapter 11 addresses many of the cultural aspects of forests and landscapes for American Indian tribes in the NWFP area.

Finally, in a study described earlier, Williams et al. (2017) used principal component analysis to create bundles or clusters of management preferences for residents of northwest Washington (n = 1796). Respondents were asked to evaluate the importance of 26 management preferences for the Mount Baker–Snoqualmie National Forest. The authors revealed six preference bundles: environmental quality, utilitarian, heritage, general recreation (hiking, scenic viewing), specialized recreation (mountain biking, equestrian, winter), and access/roads. The bundles were fairly consistent across sociodemographic categories and residential classifications (rural, suburban, and urban). Notably, some management preferences did not bundle, including nature study and food/fuel gathering. These bundles roughly coincide with the ecosystem service items described in the MEA (MEA 2005).

Summary—

Ecosystem services, and cultural services in particular, could be a very useful framework for land managers in the NWFP area to consider the diversity of spiritual, aesthetic, recreation, heritage, discovery and learning, and therapeutic benefits associated with forest settings. Currently, the agency emphasizes one aspect of cultural services, recreation benefits, which are discussed below. Recreation is quantifiable and measurable within standard agency practices. Also commonly considered are scenic resources and heritage sites, although budgetary and personnel constraints limit these functions. Other aspects of cultural services, like spirituality, solitude, wilderness therapy, and education, are managed but not actively tallied, which is a missed opportunity. A growing emphasis on cultural ecosystem services will allow resource managers to recognize the various benefits associated with a forest and stakeholder attachment to sets of benefits. The ecosystem services framework can be useful in identifying and measuring a full range of benefits and values assigned to forests and landscapes.

Outdoor Recreation

This section focuses on how society uses forests generally, and specifically within the NWFP area, for outdoor recreation and leisure. It addresses trends in who is recreating on forests and what they are doing, how technology and changes in leisure time are changing recreation patterns, and recreation sustainability.

Recreation is viewed as an important forest benefit and is a critical component of the cultural services model. Hiking, camping, and nature study are important activities that allow people to experience the benefits of forests (de Groot et al. 2006). Additional forest benefits include mental health and well-being, aesthetic encounters, cognitive development, and others (Chan et al. 2011) (fig. 9-5). Recreation benefits of parks, forests, and public lands have been widely recognized (Nielson et al. 2007, Stein and Lee 1995). Numerous studies acknowledge the positive effects of nature exposure to human health and well-being (Bowler et al. 2010; Hartig et al. 2003, 2011; Karmanov and Hamel 2008; and others); and green spaces are important venues for promoting exercise that leads to improved health (Henderson and Bialeschki 2005). Recreation use is facilitated by the presence of built amenities (Donovan et al. 2016) and access, but also depends on ecological factors (Fuller et al. 2007). Various monetary and nonmonetary approaches have been used to characterize recreation values, most of which rely on knowing frequency of visitation, intensity of use, and visitor recreation spending (Stynes 2005).

Lee Cerveny

Figure 9-5—Hiker in the Mount Baker–Snoqualmie National Forest.

The 20th century was a prolific period for recreation research and assessment, especially in the latter half of the century, which saw the establishment of the Outdoor Recreation Resources Review Commission in 1958, as well as the establishment and findings of the President's Commission on Americans Outdoors in 1985. In the 21st century, the Federal Interagency Council on Outdoor Recreation, established in response to the America's Great Outdoors Report, continues the coordinated, multi-agency effort to better understand recreation and its management. Management issues and challenges faced by all of the federal land agencies have been the focus of recreation research over the past several decades. However, the National Park Service and the U.S. Forest Service's National Forest System (NFS) have received the greatest research attention. This section will draw primarily on research in the Pacific Northwest on national forest lands, but will also include broader studies of recreation trends and recreation behavior elsewhere in the United States and on other public lands.

Trends in outdoor recreation and visitation to national forests in the NWFP area—

The degree to which Americans are recreating outdoors generally, and on federal public lands specifically, has been the source of discussion in mainstream books, such as *Last Child in the Woods* (Louv 2005), as well as scientific literature (Pergams and Zaradic 2008, Stevens et al. 2014). Special attention has been given to the extent to which youth are recreating in nature and the implications for future attitudes about natural resources and recreation use. Although some (Pergams and Zaradic 2008, Stevens et al. 2014) contend that outdoor recreation on public lands has been declining, a number of researchers have disputed that notion, suggesting instead that visitation is flat to slightly increasing (Jacobs and Manfredo 2008; Larson et al. 2011; Siikamaki 2011; Warnick et al. 2010, 2013).

Visitation levels—

Studies based on data from the National Survey on Recreation and the Environment in the United States have found that the percentage of the population participating in outdoor recreation on public and private lands has remained relatively flat in recent years and is projected to remain that way in coming decades (Bowker et al. 2012, Cordell 2012, White et al. 2016). Future increases in the total U.S. population will overcome the steady, or even slightly declining, participation rates, so the total number of people recreating in the outdoors is projected to increase over time (Bowker et al. 2012). If potential climate changes are also considered in those projections, participation rates for undeveloped skiing and snowmobiling are projected to decline by 6 percent and 18 percent, respectively, but the general projection of greater number of participants in the future remains largely unchanged nationally. Within specific regions (e.g., the Northeastern United States), the effects of climate change on recreation use may be more pronounced, and the number of participants in some regions may decline markedly (Bowker et al. 2012).

Activities such as viewing nature, visiting developed sites (which includes developed-site camping and picnicking), and visiting interpretive centers are projected to have the greatest numbers of participants across the Nation (each having more than 200 million participants) in 2030 (Bowker et al. 2012, Cordell 2012, White et al. 2016). In addition, more than 100 million people are projected to participate separately in hiking, visiting primitive areas (primitive camping, backpacking, visiting wilderness areas), and birding. As is the case presently, most future participants in outdoor recreation are expected to be participating in general activities, such as hiking, picnicking, or viewing nature. Participation in specialized activities like undeveloped skiing (10 million participants), motorized snow use (11 million participants), horseback riding (16 million participants), and challenge activities (e.g., rock climbing—25 million participants) is projected to continue to be small in 2030 relative to participation in general activities (Bowker et al. 2012, White et al. 2016).

Long-term assessments of recreation use and activity patterns are made difficult by variations in measurement systems and the missions and monitoring resources of federal land management agencies. The National Visitor Use Monitoring (NVUM) program, used by the NFS to monitor recreation, has been in place since 2000, although pilot testing on some national forests started in 1996. Estimates of recreation use under NVUM are not comparable to estimates under prior recreation monitoring systems used by the NFS. Further, comparisons of NVUM results for individual national forests can only reliably be made between two periods, 2005–2009 and 2010–2014, because of refinements to its methods after the initial 2000 to 2004 monitoring period and the 5-year sampling cycle of NVUM. The most recent visitation estimate for the NFS using data collected between 2011 and 2015 was 149 million visits. This visitation has been trending upward since 2010 (the earliest comparable year for analysis) with 2015 estimates about 4 percent greater than 2010 (USDA FS 2016b) (table 9-2).

Forest Service recreation monitoring indicates that use has been relatively stable over the last 10 years in the NWFP national forests. National forests within the NWFP area have received about 15 million recreation visits per year in recent years (USDA FS 2016b) (table 9-3). Day-use developed sites and the undeveloped (but nonwilderness) portions of national forests account for the greatest numbers of recreation visits. Recreation use in wilderness areas of NWFP-area forests is about 1 million visits per year. The difference in visit estimates between 2006–2010 and 2011–2015 cannot yet be interpreted as a trend because it is based on only two points in time and they are not statistically different.

Recreation use on NWFP-area national forests is consistent with the pattern of high participation in outdoor recreation by residents of the three-state region (California State Parks 2014, Oregon Parks and Recreation Department 2013, Washington State Recreation and Conservation Office 2013). The most recent statewide comprehensive outdoor recreation plans for Oregon and Washington found that more than 90 percent of state residents participate in some form of outdoor recreation (including activities such as hiking/walking, picnicking, camping, outdoor sports, and general relaxing) at least once a year. The statewide comprehensive outdoor recreation plan for California focused specifically on outdoor recreation that took place in parks and public lands (i.e., open space provided for natural environments and/or leisure opportunities), unlike in Washington and Oregon, but more than 90 percent of California's population reported using an outdoor park at least once in the prior year.

Table 9-3—Recreation use at national forests in the Northwest Forest Plan (NWFP) area by forest/site type for two recent periods[a]

Forest/site type	2006–2010	2011–2015
	Millions of visits	
All NWFP-area national forests	15.6	14.6
Site visits:		
Day-use developed sites	7.5	8.5
Overnight-use developed sites	2.4	2.0
Undeveloped areas	10.0	8.4
Wilderness	0.9	1.4

[a] Visitors typically complete multiple site visits during their visit to the national forest so the sum of site visits is more than the "all NWFP-area national forests" value.
Source: USDA FS 2016b.

Recreation activities—

Hiking, downhill skiing, and nature-related pursuits (i.e., viewing natural features, visiting nature centers, and nature study) are the most common primary recreation activities on national forests in the NWFP area (table 9-4). A primary recreation activity is defined as the single activity that prompted the recreation visit to the national forest. The relative popularity of those three activities is generally consistent with patterns of use on other national forests throughout the NFS. More specialized activities, such as cross-country skiing, camping, hunting, off-highway-vehicle (OHV) use, boating,

Table 9-2—Trend in visits annually to the National Forest System[a]

Year	FY 2006–2010	FY 2007–2011	FY 2008–2012	FY 2009–2013	FY 2010–2014	FY 2011–2015
	Millions					
Visits	143.6	145.5	147.5	146.7	146.8	149.0

[a] The National Visitor Use Monitoring Program runs on 5-year cycles. National-level visit estimates are calculated for these 5-year periods.
Source: USDA FS 2016b.

Table 9-4—Participation in primary recreation activities in Northwest Forest Plan (NWFP)-area national forests for two recent periods

Primary activity	NWFP area 2006–2010	NWFP area 2011–2015	National averages 2011–2015
	Percent		
Hiking	18	25	24
Nature related	18	15	14
Downhill skiing	12	15	16
Hanging out/relaxing	7	6	5
Some other activity[a]	6	6	4
Fishing	7	5	6
Cross-country skiing	3	5	2
Hunting	4	3	5
Developed camping	4	3	3
Driving	3	3	5
OHV use	4	2	2
Boating	3	2	2
Biking	2	2	4
Other nonmotorized	2	2	2
Primitive camping/backpacking	2	2	1
Picnic	1	1	2
Snowmobile	1	1	1
No activity provided	2	< 1	1
Resort use	< 1	< 1	< 1
Horseback riding	< 1	< 1	1
Total	100	100	100

[a] Some outdoor recreation activities are not listed directly and would fall into the catagory of "some other activity" such as orienteering, geocaching, parasailing, and other forms of recreational aviation.
Source: USDA FS 2016b.

and bicycling are less common primary recreation activities on NWFP-area national forests. The patterns found for those specialized activities are also consistent with national-level patterns. The Plan-area forests differ slightly from national patterns in the share of visits that are nature related (a higher share of visits), and hunting and biking (smaller shares of visits). Within the Plan area, between the two NVUM periods, the share of visits with hiking or downhill skiing as the primary activity increased slightly, while the share of visits in nature-related activities decreased slightly. Those differences cannot yet be interpreted as trends because they represent only two points in time.

The patterns in recreation activities on the NWFP-area national forests are consistent with patterns in outdoor recreation activity of the general populations of California, Oregon, and Washington (California State Parks 2014, Oregon Parks and Recreation Department 2013, Washington State Recreation and Conservation Office 2013). Walking for pleasure is the most commonly reported activity in each state, with between 64 and 73 percent of residents of each state reporting walking for leisure at least once during the year. About 50 percent of the populations in each state report hiking on unpaved trails at least once during the previous year. More than half of each state's population reported participating in general, nature-based recreation activities, such as sightseeing or picnicking. About half of Oregon residents and 40 percent of California and Washington residents reported that they had camped in a developed camping site in the past year. Participation in more specialized nature-based outdoor recreation activities, such as hunting, fishing, backpacking, biking, and freshwater boating, was generally reported by less than half, and typically less than one-quarter, of residents in the three states.

Research conducted elsewhere shows that volunteers can be motivated by a variety of factors, including the desire to expand public access and recreation opportunities, social engagement, and commitment to the environment (Bruyere and Rappe 2007, Lu and Schuett 2014, Propst et al. 2003). Volunteer organizations in the NWFP area have sizeable memberships and work closely with public land managers to identify mutually desired projects. Nationwide, reliance on partners and volunteers has played an important role in bolstering the capacity of national resource agencies, which face maintenance backlogs on recreation infrastructure (Seekamp and Cerveny 2010, Seekamp et al. 2011). The National Trails Stewardship Act of 2016 (P.L. 144-225) directs the Forest Service to expand volunteerism and partnerships further in support of trail maintenance. Volunteering and stewardship have also been studied in relation to place attachment, with stewardship in a forest or park generating stronger feelings of connection (Caissie and Halpenny 2003, Dresner et al. 2015, Ryan 2005).

Population aging and implications for forest visitation— Most recreation visits to NWFP-area national forests are by those between the ages of 30 and 60 (table 9-5). Those less than 20 years old account for about 17 percent of visits. For comparison, those under age 18 represented about 23 percent of the U.S. population in 2014 (U.S. Census) (Colby and Ortman 2015). In most cases, those visits from someone under the age of 16 likely involve family recreation with children. The age distribution of those recreating at NWFP-area forests is consistent with patterns on all national forests, although there are slightly more visits in the 20 to 40 age group in the plan area compared to the national pattern

Table 9-5—Percentages of Northwest Forest Plan (NWFP) area and national recreation visits by age groups for two recent periods

Age group	2006–2010 NWFP	2011–2015 NWFP	2011–2015 National
	Percent		
Under 16	17	13	16
16–19	4	4	4
20–29	14	16	13
30–39	17	17	15
40–49	18	17	17
50–59	16	17	17
60–69	10	13	13
Over 70	3	4	5

Source: USDA FS 2016b.

(table 9-5). Compared to the national median age of 37.7 in 2014, Oregon's residents are slightly older, Washington's residents are about the same age, and California's residents are slightly younger.

The average ages of the populations of California, Oregon, and Washington are expected to continue to increase over time. Age is consistently found to be a factor in recreation participation and correlates with differing perceived barriers to participation in recreation (Bowker et al. 2006, Child et al. 2015). Considering outdoor recreation anywhere, not just on Forest Service land, those over 45 years of age participate in a smaller set of recreation activities than those who are younger and, as people age, they continue to reduce activity participation (Cordell 2012, White et al. 2016). Recreationists in age groups over 45 are most commonly participating in developed-site activities and viewing and photographing nature (table 9-6). Those over age 45 have moderate rates of participation in motorized activities, hunting, and fishing that decline steadily as they age. Older people are more likely to feel that personal health, safety, and disability are barriers to participating in outdoor recreation; younger people view the amount of leisure time, limited information about recreation opportunities, and lack of transportation as barriers to participating in outdoor recreation (Ghimire et al. 2014).

Table 9-6—Percentage of age groups 45 and older participating in outdoor recreation by site/activity type

Site/activity type	Age 45–54	Age 55–64	Age 65+
	Percent		
Visiting developed sites	81	75	62
Viewing and photographing nature	80	75	65
Backcountry activities (including hiking)	48	37	22
Motorized activities	37	27	17
Hunting and fishing	38	29	20
Nonmotorized winter activities	10	5	2
Nonmotorized water activities	22	15	7

Source: Cordell 2013, adapted from White et al. 2016.

Those under age 20 account for about 17 percent of the recreation visits on NWFP-area national forests (see table 9-5). That rate of outdoor recreation participation is generally consistent with what was found nationally. In a national study of the outdoor recreation behavior of those under age 20, Larson et al. (2011) found that the majority of children do spend time in outdoor recreation each week and that 62 percent spend at least 2 hours recreating outside daily. Of those under 20, those between 16 and 19 had the lowest rates of being outdoors for recreation: most of respondents at that age spent less than a half an hour outdoors daily (Larson et al. 2011). Hispanic youth had the highest rates of spending time in outdoor recreation. Across all groups, those under 20 were focused on general recreation in the outdoors, e.g., playing or hanging out (84 percent of participants); biking, walking, jogging (80 percent); and using electronic devices outdoors (65 percent). More specialized outdoor recreation activities such as wildlife viewing (31 percent), hiking/camping/ fishing (29 percent), and snow sports (9 percent) were reported by lesser shares of young participants (Larson et al. 2011). The greatest impediment to participating in outdoor recreation for those under 20 was interest in other activities, including using electronic media indoors. Issues with limited access, lack of transportation, or concerns about safety were cited as reasons for not recreating outdoors by less than one-fourth of those under 20 (Larson et al. 2011).

Work patterns and leisure time—

Lack of time has been identified as the key reason that some Oregon and Washington residents never visit national forests for recreation, or visit them less frequently than desired (Burns and Graefe 2007). Lack of time was also found to be a moderate impediment to youth participation in outdoor recreation generally (Larson et al. 2011). Time availability was identified as a much stronger factor in constraining recreation use of national forests than perceived recreation site characteristics or crowding (Burns and Graefe 2007). In the NWFP area, the median duration of a national forest visit is about 4 hours (table 9-7). However, that figure is influenced by the length of stay of those camping in national forest campgrounds. Excluding campground use, the median length of stay of visitors to Plan-area national forests is less than 3 hours for day-use sites and general forest areas, and less than 4 hours for those recreating in wilderness. The vast majority of recreation visits to Plan-area forests are short-duration

Table 9-7—Median duration of visits to NWFP-area national forests

Category	2005–2009	2010–2014
	Hours	
National forest visit (all sites)	4.5	4.1
Day-use developed sites	1.7	2.1
Overnight-use developed sites	44.2	41.8
Undeveloped areas	3.5	3.0
Wilderness	4.4	4.0

Source: USDA FS 2016b.

trips. The preponderance of short-visit durations is consistent with the patterns of high use in developed sites (where visits are likely focused on viewing natural features or a brief hike).

Sustainable recreation—

For natural resource management, broadly, sustainability is typically thought to relate to the capacity of the landscape (comprising human and natural systems) to provide desirable social, ecological, and economic outcomes now and into the future under current management. Research addressing the sustainability of recreation has largely focused on (1) the ability of the resource and managers to provide current recreation opportunities (especially winter recreation) in the face of a changing climate (e.g., Beaudin and Huang 2014, Buckley and Foushee 2012, Smith et al. 2016); (2) how alteration of environmental conditions through disturbance, recreation use, or resource management affects the conditions of recreation resources and user experiences (e.g., Brown et al. 2008, Cole 2013, Shelby et al. 2005, White et al. 2008); (3) how high use levels at recreation sites may change the behavior, experience, and satisfaction of visitors (e.g., Cole and Hall 2009, Fonner and Berrens 2014, Lawson et al. 2003); or (4) the social and economic conditions in recreation gateway communities and reliance of those communities on tourism for economic activity (e.g., Andereck et al. 2005, Frauman and Banks 2011, Kurtz 2010). Within the recreation scientific literature, perhaps the greatest attention has been paid to items 2 and 3. The scientific literature lacks a definition of "sustainable recreation," and integrated studies of recreation sustainability that look at a suite of sustainability factors. This lack of scientific research into sustainable recreation contrasts with the fairly extensive use of the term in management and policy directions in recent years. Unlike the focus of scientific literature, which is more broad, managers tend to view recreation sustainability in terms of capacity to provide desired recreation opportunities in the face of declining agency budget allocations and perceived greater recreation use.[2]

[2] U.S. Department of Agriculture, Forest Service [USDA FS]. 2016. Region 6 sustainable recreation strategy. Unpublished report. On file with: Lee K. Cerveny, U.S. Forest Service, Pacific Northwest Research Station, 400 N 34th Street, Suite 201, Seattle WA 98103, lcerveny@fs.fed.us. 20 p.

Visitor satisfaction with recreation site conditions and the recreation experience is a component of sustainable recreation. Oregon and Washington residents have rated recreation conditions on the national forests they visited most frequently at moderate to high quality (Burns and Graefe 2006). The highest quality rankings were given for the undeveloped characteristics of views, courteous and friendly staff, and safe sites with clearly posted rules and regulations. The lowest quality scores were given for availability of multilingual services, accessibility of uniformed Forest Service personnel, risk of vandalism and theft to vehicles, and assistance for people with special needs. However, even for those items, the most common quality ranking was "fair" (the second lowest rating on a scale from "awful" to "excellent"). In a separate study, Burns and Graefe (2007) found that 60 percent of households in Oregon and Washington with a person having a disability felt hampered in their ability to use national forests for recreation. However, 21 percent of those who felt national forests were not accessible for recreation also stated no interest in outdoor recreation (Burns and Graefe 2007). The conditions of roads and trails and conditions of facilities were rated as good to very good (Burns and Graefe 2006). Recreationists stated their perception of site quality was highest when there was (1) minimal litter, (2) a feeling of safety and security, (3) clearly posted rules and regulations, and 4) clean restrooms and toilets (Burns and Graefe 2006). The presence of litter, trash, or vandalism was the key factor in explaining recreationists' perceptions of recreation site quality and environmental condition at Bureau of Land Management recreation sites in the northwest Oregon Cascade foothills (White et al. 2008). Visitors who have visited those sites with litter over increasingly long time frames appear more sensitive to deterioration in site conditions (White et al. 2008).

Recreation and climate change—

Changing climate can change (increase or decrease) the availability and quality of recreation opportunities (Shaw and Loomis 2008). Changing environmental conditions that result from weather and climate patterns can affect the ability of people to participate in certain recreation activities with implications for quality of life and future public health

(White et al. 2016). Climate change models project warmer weather conditions for longer periods, which are expected to increase participation in summer and warm-weather recreation activities (Bowker et al. 2013, Farley et al. 2011). Temperature and precipitation changes directly change the availability and quality of recreation sites. Based on preliminary research conducted in the northern Rocky Mountains (Hand and Lawson 2018) and more generally (Shaw and Loomis 2008), it is understood that climate change can alter ecological conditions and may affect optimal recreation conditions. Recreation visitors are likely to engage in substitution as an adaptation strategy to climate change—substituting one location for another, changing the timing of their recreation visits, or shifting into new activities as opportunities for their favorite activities decline (Loomis and Crespi 2004). However, substitution may represent a net benefit loss, even when participation changes only subtly. For example, the substitute site may be more expensive to access, take more time to reach, or offer inferior quality. Studies conducted in central Oregon are underway and have identified certain recreation activities that may be more sensitive to a warming climate as well as implications associated with the possible expansion of shoulder seasons.

Summary—

Recreation visits are expected to grow in day-use settings and developed facilities. At the national level, the number of outdoor recreation visits will increase in the coming decades in accordance with population growth. The majority of outdoor recreation use is for general recreation activities, like hiking, viewing nature, visiting nature centers, viewing wildlife. Most recreation visits to national forests are relatively brief, lasting less than one-half day, and tend to occur at developed sites. These are important trends to consider when managers are asked to allocate resources to recreation facilities. The greatest barriers to outdoor recreation participation are lack of time and travel distance to national forests. Other barriers include concerns for personal safety, signage, and accurate information, all of which have positive effects on visitor perceptions of site conditions. Natural resource agencies like the Forest Service seek information about the ecological effects of recreation in efforts to promote sustainable recreation. Lack of conceptual development of what sustainable recreation means or tested sustainable recreation models or tools is inhibiting use of this concept in planning.

Trust

Trust is one of the key foundations of human social order and is viewed as critical for personal development, interpersonal relationships, mutual cooperation, and enduring institutions, such as governments, financial markets, and religious organizations (Lewicki et al. 1998). Humans operate in an environment often characterized by ambiguity, complexity, risk, and change (Lewicki et al. 1998). Trust and distrust are distinct emotional responses that allow individuals and entities to navigate uncertainty, manage efficiently, and survive.

Defining trust—

Trust is defined by early social psychologists as expressions of confidence in others' intentions and motives. Trust was understood as the sincerity of a person/institution's word (Mellinger 1956), and was seen as dependent upon the confidence that one's interests would be protected and promoted by another and with an agreement on full information sharing (Read 1962). Predictability was also seen as integral to the notion of trust (Deutsch 1958). Scholars later explored trust as an aspect of actual behavior, rather than as a primary motivation, understanding trust as one's hope of another's favorable behavior in a situation of vulnerability (Hosmer 1995). Regardless of their motivations, there is an expectation that, in a position of dependence, one will not injure or ignore the interest of another (Hosmer 1995). Lewicki et al. (1998) suggested that trust and distrust are best understood not as a binary construct in polar opposition, where trust is good and distrust is bad. Nor is trust/distrust viewed as an inverse relationship, where trust increases only when distrust decreases and vice versa. "There are elements that contribute to the growth and decline of trust, and there are elements that contribute to the growth and decline of distrust" (Lewicki et al. 1998: 440). These elements are repeatedly modified through frequent human encounters and transactions. Because of the many layers and facets of human interactions, it is possible to both have trust and distrust for a person or entity simultaneously—trusting some aspects of the relationship, but not others. Understanding that trust and distrust can coexist has important implications for public engagement in forest management, in particular the critical importance of creating processes that are trusted.

Trust also should be understood with both attention to social context and recognizing it as a dynamic process (Lewicki et al. 1998). A person can trust an individual or agency in one sociopolitical setting but be wary of their performance in another setting. For example, an environmental advocate can develop a trusting relationship with a timber industry representative in the context of a small collaborative group focused on forest restoration, but this level of trust may change when the organizations appear in a large public hearing to deliberate a proposed timber sale. And, trust is dynamic and inconsistent. Trust can build and subside with each short-term interaction, which can influence the long-term trajectory of a relationship. For natural resource agencies, which often make decisions in the context of wicked problems, conflicting ideologies, and high stakes, developing processes and protocols that can be trusted is essential, even when trust can be elusive among various actors involved in those processes.

Trust as a topic in natural resource management— Trust has been a topic of investigation in scholarship related to natural resource management (Beierle and Konisky 2000). Trust between stakeholders has been characterized as a factor that shapes natural resource management outcomes (Cvetkovich and Winter 2003, Davenport et al. 2007, Stern 2008a). At its core, trust is a fundamental component of human relationships that suggests a party's acceptance of vulnerability related to positive expectations of the behavioral intentions of another party (Rousseau and Tijoriwala 1999). In the context of natural resource governance, scholars distinguish between various types of trust. Davenport et al. (2007) delineated two kinds of trust: "institutional trust" (trust in agencies to represent and serve the public) and "interpersonal trust" (trust based on personal relationships). Some scholars have focused on "rational trust," calculated based on an entity's predictable behavior, accountability, and reliability of performance (Hardin 2002, Stern 2008b). Others emphasize "social (or affinitive) trust," which grows based on shared experiences and enduring interactions (Braithwaite 1998, Cvetkovich and Winter 2003). Trust in natural resource agencies has been discussed in the context of "broad-level" trust in governing agencies to achieve goals of resource conservation and meeting public needs, and as "project level" trust, which emphasizes whether the agency can be trusted to successfully implement the project goals and minimize harm to the social and natural environment (Ribe 2013). For an agency to craft a socially acceptable management strategy, trust is important both at the broad level and the project level (Olsen and Shindler 2010).

Community-based collaborative groups, which are discussed later in this chapter, have emerged partly in response to perceptions of distrust between communities and public land agencies. In the context of collaborative management, Stern and Coleman (2015) developed a conceptual framework that identified four types of trust: "dispositional" (the predisposition of individuals to trust), "rational" (based on likeliness of predicted behavior as judged by prior performance), "affinitive" (based on shared values and developed through positive interactions), and "systems based" (transparent process, fair and just procedures) (table 9-8). They posited that the diversity of these four trust types within natural resource management contexts is important for successful outcomes. Stern and Baird (2015) used this framework to study variation of degrees and proportions of the four types of trust. They found that explicit attention to the development of three types of trust (rational, affinitive, and systems based) can enhance the efficiency and resilience of natural resource management institutions. They also found that when one type of trust is damaged, having other types of trust can buffer the loss (Stern and Baird 2015). These studies emphasize the importance of trust to the success of collaborative management and suggest the need for deliberate

Table 9-8—Varying interpretations of trust

Types of trust	Definitions	Citations
Institutional trust	Trust in agencies to represent and serve the public	Davenport et al. 2007
Interpersonal trust	Trust based on personal relationships	
Social trust	Trust among people that grows based on shared experiences and enduring interactions	Braithwaite 1998, Cvetkovich and Winter 2003
Rational trust	Based on predictable behavior, accountability, and reliability of performance	Stern and Baird 2015, Stern and Coleman 2015
Affinitive trust (similar to social)	Trust among people that grows based on shared experiences and enduring interactions	
Dispositional trust	The predisposition of individuals to trust (based on one's natural inclinations, values, experiences	
Systems-based trust	Derived from presence of fair processes; just procedures	

attention to fostering all four types of trust to maximize institutional resilience.

Recent studies have explored the relationship between values and trust in forest management. Although some suggest that the degree of institutional trust can influence the extent to which someone supports forest management actions, we do know that trust expands when agencies make decisions that reinforce an individual's values (Vaske et al. 2008). Trust can be built (and in many cases conflict reduced) through fair participation processes or transparent decisionmaking (Webler and Tuler 2000, Webler et al. 2001). In a comparative study among national forests in northern California, northern Florida, and Michigan, Winter et al. (2004) found a relationship between shared values and social trust in a study of fuel management strategies. In California, Winter et al. (1999) learned that trust predicted attitudes in the public's willingness to pay recreation fees. In their study of prescribed fire burning in Colorado, Vaske et al. (2008) used an approach known as "shared values similarity," which measures the degree of similarity among a set of environmental values (Cvetkovich and Winter 2003). They found that when values were held in common between the public and the land management agency, there was a greater degree of trust. They also learned that when social trust was improved, there was more support for land manager policies of prescribed burning and mechanical thinning. A lack of trust in governing agencies is cited as a primary barrier in natural resource planning (Lachapelle and McCool 2012) and can potentially lead to litigation or noncompliance (Stern 2008b).

Achieving trust among multiple conflicting parties in resource management can be challenging; still, there is an increasing recognition that trust can be fostered by direct public engagement or participation in a collaborative decision processes where deliberation is encouraged. For trust to flourish, processes should be inclusive, representative, transparent, and predictable (Beierle and Konisky 2000). In addition, trust can be aided by groups having clear objectives, outlined roles and responsibilities, and a tangible and enduring commitment from key partners.

Summary—

Natural resource institutions like the Forest Service often make difficult decisions in uncertain environments in which science is evolving and public sentiment is conflicted. The degree of trust established between public agencies, stakeholders, and communities is an important factor in public support for resource management decisions. Clear objectives, consistent communication, transparent processes, reasonable timelines, maintained commitments, and opportunities for candid deliberation can enhance institutional trust both at the project level and at the national level. Developing processes and protocols that can be trusted is essential, even when trust can be elusive among various actors involved in those processes.

Involving the Public

Public participation in federal agency land management planning processes is required by various laws, regulations, and policies, including the Multiple-Use Sustained-Yield Act of 1960 (MUSYA), National Environmental Policy Act of 1969 (NEPA) and the National Forest Management Act of 1976 (NFMA) (box 3). A national planning rule for the USDA Forest Service stipulates that public participation efforts must "…have significant potential to reach and involve diverse segments of the population that historically have not played a large role in NFS (National Forest System) planning and management" (USDA FS 2012). This contemporary emphasis on robust public participation in land management planning suggests new innovations, strategies, and methods of encouraging diverse public participation, which can generate trust among stakeholders and land managers. This section will review recent trends in public participation, including institutional constraints and best practices. Because peer-reviewed research on this topic is limited in the U.S. Forest Service's Pacific Northwest Region, this section will also include information from federal agency reports, doctoral dissertations, and standard texts in the field of public administration, conflict management, and collaboration.

Box 3—MUSYA, NEPA & NFMA Requirements

MUSYA requires that management "best meet the needs of the American people," by identifying the public's values and desires (US Congress, OTA, p.78).

NEPA requires agencies to inform the public about the possible environmental impacts of their decisions, including the public as a participant in decisionmaking.

NFMA further reinforced the public's right to participate in Forest Service planning and decisionmaking.

Trends in public participation in natural resource management—
In 1969, Sherry Arnstein published the article, "A Ladder of Citizen Participation." Although dated, this article remains relevant as a way of describing different types of public involvement. At the core of Arnstein's argument is the premise that different types of public involvement are directly related to the different levels of power citizens have in determining outcomes (Arnstein 1969). The ladder is a metaphor for illustrating increasing levels of public influence in decisionmaking as one climbs each rung of the ladder. Lower rungs indicate nonparticipatory types of public involvement, such as education, while middle rungs allow participants to share information without assurance that a change in the outcome will occur (Arnstein 1969). The top rungs of the ladder provide increasing levels of influence in decisions affecting the outcome.

A key finding of Arnstein's work is the recognition that participation without a clearly defined public role (i.e., the type of participation, or identifying which "rung on the ladder") can lead to a meaningless or frustrating process for all involved. In 1999, the International Association of Public Participation (IAP2 2014) transformed Arnstein's "ladder" into a "spectrum." This decision-oriented, objective-driven, and values-based approach to public participation was designed to assist with selection of the appropriate level of public participation in any community engagement program (fig. 9-6). The spectrum seeks to "…legitimize differing levels of participation depending on the goals, time frames, resources and levels of concern in the decision to be made" (IAP2 2014). As described by the IAP2, the spectrum defines the **promise** being made to the public within each level of participation (2014). The arrow at the top of the diagram indicates that as one moves to the right, the level of participation and public influence in the decisionmaking process increases, similar to moving up the rungs of Arnstein's ladder. Note that while the spectrum covers the full range of public influence in a decisionmaking process, government agencies retain their decision-making authority in all instances and are ultimately

IAP2'S PUBLIC PARTICIPATION SPECTRUM

The IAP2 Federation has developed the Spectrum to help groups define the public's role in any public participation process.

INCREASING IMPACT ON THE DECISION

	INFORM	CONSULT	INVOLVE	COLLABORATE	EMPOWER
PUBLIC PARTICIPATION GOAL	To provide the public with balanced and objective information to assist them in understanding the problem, alternatives, opportunities, and/or solutions.	To obtain public feedback on analysis, alternatives, and/or decisions.	To work directly with the public throughout the process to ensure that public concerns and aspirations are consistently understood and considered.	To partner with the public in each aspect of the decision, including the development of alternatives and the identification of the preferred solution.	To place final decisionmaking in the hands of the public.
PROMISE TO THE PUBLIC	We will keep you informed.	We will keep you informed, listen to and acknowledge concerns and aspirations, and provide feedback on how public input influenced the decision. We will seek your feedback on drafts and proposals.	We will work with you to ensure that your concerns and aspirations are directly reflected in the alternatives developed and provide feedback on how public input influenced the decision.	We will work together with you to formulate solutions and incorporate your advice and recommendations into the decisions to the maximum extent possible.	We will implement what you decide.

Figure 9-6—The International Association of Public Participation's [IAP2] Public Participation Spectrum.

responsible for their actions (Wondolleck and Yaffee 2000). Thus, government agencies are not authorized to use the "empower" end of the IAP2 spectrum. Bryan (2004: 882) put this into perspective: "While participants may challenge the decisions authorities ultimately make, they do not challenge their authority to make those decisions."

The IAP2's Public Participation Spectrum can assist with the selection of the level(s) of participation that defines the public's role throughout a forest plan revision effort. Importantly, the amount of effort required among the different spectrum levels can vary widely for both the agency and the public. Imperial (2005: 312) emphasized the importance of judiciously identifying collaborative opportunities that add public value while minimizing transaction costs, and suggests that "public managers are…cautioned to use collaboration wisely. When used correctly, collaboration is an effective governance strategy. When used inappropriately, it can create more problems than it solves."

Thus, different phases of plan revision call for different levels of public involvement. For example, in the assessment phase of plan revision, a **collaborative** process could be designed to identify the benefits provided to people by a national forest. Here, the public works in conjunction with agency personnel to identify the unique places, roles, and contributions a national forest provides based on their preferences, interests, and values. English et al. (2004) emphasized the importance of eliciting values early on in public involvement processes and further acknowledged that to be effective, these processes "must be tailored to the place, the people, and the circumstances; there is no single recipe for success." Collaboratively identifying unique roles and contributions, early in the assessment phase of forest plan revision, can focus forest management on issues that people value most.

Alternatively, during the NEPA phase of plan revision, while the interdisciplinary team is conducting its analysis in compliance with the act, it may be appropriate to **inform** the public as a means of assisting them in understanding issues or alternatives. For example, following the 90-day comment period on the Inyo, Sierra, and Sequoia National Forest draft plans and draft environmental impact statements, the interdisciplinary team spent months analyzing comments, defining issues and resolutions, and preparing responses to comments in preparation for release of the final environmental impact statement and draft record of decision (USDA FS 2016a). During this time frame, little interaction with the public occurs. To fill this gap, a series of informational bulletins provided additional detail on topics of interest identified during the comment period (Long et al. 2014). In the case of the Sierra synthesis, the agency is not asking for public feedback, it is providing information to assist the public in understanding issues or alternatives. As Arnstein and others have found, the key is defining these various levels of public participation prior to initiating the plan revision, and being clear with the public about what their actual role will be, ensuring them a meaningful and robust participation process.

Newer research continues to support and refine Arnstein's work and that of the IAP2 (Carpini et al. 2004, Kelshaw and Gastil 2008, Lynam et al. 2007, Rowe and Frewer 2005). Rowe and Frewer (2005) developed a typology that further defines key concepts of public engagement within the IAP2 spectrum based on the direction information flows from the sponsor (i.e., Forest Service) to the public. This typology (Rowe and Frewer 2005) defines three key types of public engagement: public communication (e.g., **inform** on the IAP2 spectrum), public consultation (e.g., **consult** on the IAP2 spectrum), and public participation (e.g., **involve and collaborate** on the IAP2 spectrum). Specifically, Rowe and Frewer (2005) suggested that public communication characterizes information flowing from the agency to the public, public consultation from the public to the agency, and information flowing both directions as public participation. Another aspect of their research is the importance of aligning mechanisms, defined as processes, techniques, and instruments, to the appropriate level of engagement (Rowe and Frewer 2005). Carpini et al. (2004) focused their research on the mechanism of "face-to-face" meetings. They found that face-to-face communication is the single greatest factor in increasing the likelihood of cooperation among participants (Carpini et al. 2004). Kelshaw and Gastil (2008) differentiated face-to-face meetings among the different types of public engagement. For example, informational meetings fall into the "inform" level of the IAP2 spectrum, where the agency initiates conversation with the public. Alternatively, communication flows both directions in collaborative face-to-face meetings initiated by the agency and the public. Finally, Lynam et al. (2007:1) summed up the importance of applying the right mechanism to the right level of public engagement: "…picking the right tool does not guarantee that the data desired will be produced, but selecting the wrong tool does make success less likely."

As mentioned in the beginning of this section, NEPA requires federal land management agencies, including the Forest Service, to **involve** the public in agency planning processes (Brown and Donovan 2013, Hoover and Stern 2014). Hoover and Stern (2014: 174) argued that although "NEPA regulations do not specifically empower the public to directly influence the NEPA process," the public generally becomes involved in these efforts to have a genuine impact, "…or **influence** on decisions that affect them or the public resources they value." They also acknowledged, "While there are minimum standards related mostly to the timing of involvement and disclosure, the NEPA process grants the implementing agency broad discretion regarding the form and nature of the public involvement process" (Hoover and Stern 2014: 175).

Given this considerable level of discretion, scholars have argued that understanding what motivates the behaviors and actions of key personnel, such as interdisciplinary team leaders as well as the public, has the potential to improve the public participation experience for both agency personnel and the public (Hoover and Stern 2014, Lipsky 1980, Yang 2005). According to Cerveny et al. (2011: 202), "The ID team leader is responsible for managing group interactions, synthesizing scientific findings, and coordinating analysis of alternatives." Hoover and Stern (2014) found that agency team leaders of planning processes across the Forest Service expressed a desire for greater public influence in planning processes through improved "substantive" input to management decisions rather than through objections and litigation. Stern and Predmore (2011) have characterized substantive comment as information that can improve management decisions, as opposed to comments based on opinions or conjecture.

The literature describes four broad and interrelated behavioral factors of participating publics related to their ability to gain influence in decisionmaking (Hoover and Stern 2014). These factors include values and desires, time, trust and prior experience, and the skill to provide comments (Beierle and Konisky 2000, Cheng and Mattor 2006, Creighton 2005, Germain et al. 2001, Halvorsen 2006, Smiley et al. 2010, Smith and McDonough 2001, Whitall 2007, Yang and Pandey 2011). Thus, in understanding and accommodating these inherent behavioral factors, Forest Service team leaders and decisionmakers can improve the public's level of influence in decisionmaking earlier in the planning process through improved "substantive" comment processes.

To identify key factors that either motivate or constrain an interdisciplinary team leader, Hoover and Stern (2014) conducted a qualitative case study analysis of interviews with Forest Service employees. Through their research, they found that the following four factors influenced interdisciplinary team leaders' (IDTLs') ability "…to go above and beyond the minimum requirements to facilitate public influence: (1) the IDTLs' personal beliefs and norms; (2) past and present experiences with the public; (3) the IDTLs' workloads; and (4) the influence of the decision maker" (Hoover and Stern 2014: 181). To enhance motivation of IDTL's, Hoover and Stern (2014) suggested that the agency may be able to improve employees' ability to cope with stress, assist in maintaining reasonable workloads, and offer training to effectively respond to public concerns about resource management.

The complexity of public participation in the 21st century—

Creating effective public involvement strategies is challenged not only by varying levels of public influence, statutory ambiguity, and consequent agency discretion, but also by socially dynamic systems (Brown and Donovan 2013) (box 4). Changes in demographic patterns are occurring most rapidly in the southern and western regions of the United States, with increasing numbers of young people and immigrants (Colby and Ortman 2015). Along with these changing demographic patterns are changing values and user preferences (Brown and Donovan 2013). Incorporating traditional and emerging values necessitates new methodologies for creating public involvement processes as required by the 2012 planning rule. This section highlights new research in the fields of dispute resolution, stakeholder and social network analyses, as well as public participation GIS.

> ### Box 4—Whom Do You Ask?
>
> **Sample bias—**The answer you get depends on whom you ask.
>
> In a 2012 PPGIS case study in the southern Sierra Nevada, it was found that responses from a random sample of households preferred forest amenities over the stronger utilitarian values and consumptive use preferences of stakeholders who volunteered to participate in the study (Brown et al. 2013).

Seeking resolution through dialogue: The importance of framing and reframing—
As a science-based organization, the Forest Service has focused much of its attention on increasing the amount of technical information provided to the public as a means to increase understanding of complex environmental issues and associated risks. In other words, the agency uses a technical frame of reference to define and explain environmental issues. Nisbet (2009) and others have argued that this type of information is likely to reach a small audience of already informed and engaged citizens (Ho et al. 2008, Nisbet 2005, Popkin 1991). He further stated, "…the rest of the public either ignores the coverage or reinterprets competing claims based on partisanship or self-interest, a tendency confirmed across several decades by public opinion research" (Nisbet 2009: 14). Nisbet's argument illustrates how technical and lay populations frequently frame environmental issues differently. Framing involves "shaping, focusing, and organizing the world around us" (Gray 2003: 11). Gray (2003: 12) further explained that "through framing, we place ourselves in relation to the issues or events—that is, we take a stance with respect to them." Simply, a frame reflects what we believe is going on and how we see ourselves and others involved in what is happening. The process of framing then offers insights into why some environmental issues are difficult to resolve (Gray 2003).

Elliott et al. (2003) drew conclusions from eight case studies on how framing affects the potential for conflict resolution of intractable environmental disputes. They found that frames may not be permanent and can change through reframing activities (Elliott et al. 2003). In seven of the eight cases studied, they found that efforts were made to consciously reframe the conflict through public dialogue. Lengwiler (2008) found that the lay-technical divide could be transcended by reframing the dialogue within a wider socioeconomic context. In other words, by reframing the environmental issue within a wider socioeconomic context, laypersons have the potential to coalesce around a set of common concerns and effectively engage in problem-solving activities. Thus, they are not expected to become scientific and technical experts, nor are experts expected to compromise their role in solving environmental issues (McKinney and Harmon 2008). The goal, as stated by McKinney and Harmon (2008: 63), is "…to integrate expert and public knowledge and information to shape decisions that are scientifically credible, politically legitimate, and relevant to the problem at hand."

In another case study, Whitall et al. (2014) used interest-based problem-solving (IBPS) techniques to reframe environmental conflict in the Sierra Nevada ecoregion. Here "IBPS techniques were used to redefine the meaning ascribed to the ecological restoration of the Sierra Nevada ecoregion from two differing points of view. Techniques included focusing the conversation on **why** these individuals wanted something, as opposed to **what** they wanted or needed" (Whitall et al. 2014: 176). In so doing, common interests emerged from intractable positions. Yet Burton (1990) and Maiese (2004) provided a cautionary note when using IBPS techniques: "…while interest-based bargaining is effective in interest-based disputes, it should not be applied to disputes involving deep differences in values."

Thus, this research (Elliott et al. 2003, Lengwiler 2008, McKinney and Harmon 2008, Whitall et al. 2014) suggests that in at least some environmental conflicts, frames can change through intentional actions and interventions. Reframing environmental issues within a wider socioeconomic context has the potential to bridge the gap between technical experts and laypersons. Finally, by reframing dialogue from positions (**what** people want) to interests (**why** people want it) it is possible to render interest-based disputes more tractable.

Public participation and the identification of stakeholders—
Reed et al. (2009) found that the role of stakeholders is becoming increasingly embedded in environmental policy. Yet, they argued, "…stakeholders are often identified and selected on an ad hoc basis. This has the potential to marginalize important groups, bias results and jeopardize long-term viability and support for the process" (Reed et al. 2009: 1933). Thus, they discussed growing interest in a collection of systematic methods that can be used to identify

individuals, groups, and organizations who are affected by a decision and then prioritize these individuals and groups for involvement in the decisionmaking process (Reed et al. 2009). Stakeholder analysis is one way of systematically identifying groups implicated by an environmental policy or decision (Grimble and Wellard 1997, Prell et al. 2009, Reed 2008, Reed et al. 2009).

Reed et al. (2009) identified three critical, sequential steps of stakeholder analysis: (1) identifying stakeholders and their interest in the problem or decision, (2) differentiating between and categorizing stakeholders, and (3) exploring relationships among stakeholders. For each step, a variety of methods exist depending on the knowledge, skills, and resources available. For example, in step one where individuals and groups with a stake in the plan revision or amendment process are widely known, the stakeholder analysis can be conducted without active participation of the stakeholders themselves. Yet, Reed (2008) cautioned that stakeholder participation may be necessary if the agency has incomplete knowledge on the population that may have an interest in the outcome. Identifying stakeholders is an iterative process, where stakeholders are added as the analysis continues using different methods such as expert opinion, focus groups, semistructured interviews, or snowball sampling (Prell et al. 2009, Reed 2008, Reed et al. 2009).

Various methods also exist for step two: categorization of stakeholders. Here, methods may be either top-down or bottom-up. In the top-down approach, stakeholders are classified based on their observations as applied through a predetermined conceptual framework or theoretical perspective (Grimble and Wellard 1997, Reed 2008). The bottom-up approach allows categories to be defined by the stakeholders themselves, allowing the analysis to better reflect their perceptions (Dryzek and Berejikan 1993, Hare and Pahl-Wostl 2002).

Finally in step three, two principal methods are used to investigate the relationships among and between stakeholders (both as individuals and groups): social network analysis, which provides insights into patterns of communication, trust, and influence between stakeholders in social networks (Lienert et al. 2013, Prell et al. 2009, Whitall 2007); and knowledge mapping analysis, which examines the flow of information between these stakeholders (Reed et al. 2009). When used in conjunction with social network analysis, Reed argued that knowledge mapping may extend the "who knows who" of social network analysis by providing a visual representation of "who knows what" (Reed et al. 2009: 1940). Social network analysis has been used in the NWFP area to evaluate the structure of fire protection and restoration institutions in the eastern Cascade Range of Oregon (Fischer and Jasny 2017, Fischer et al. 2016).

The increasing use of stakeholder analysis in natural resource management reflects a growing recognition that stakeholders influence environmental decisionmaking (Prell et al. 2009). The literature also shows that stakeholder analysis can be used to minimize conflict, reduce marginalization of certain groups, and provide fair representation of diverse interests (Prell et al. 2009, Provan et al. 2005, Reed 2008, Reed et al. 2009, Whitall 2007).

Participatory mapping and geospatial approaches— A growing number of scholars and government agencies are interested in the integration of technology and spatial information into public participation strategies (Brown et al. 2013). As an example, the 2012 planning rule encourages the U.S. Forest Service to be proactive and use contemporary tools such as the Internet to engage the public in forest planning (USDA FS 2012). As noted earlier, public participation GIS has featured prominently in forest plan revision efforts in the past decade (Brown and Donovan 2013, Brown and Reed 2009, Brown et al. 2013).

The term "public participation geographic information systems" was conceived in 1996, during the National Center for Geographic Information and Analysis meeting (Sieber 2006). Brown and Reed (2009: 166–167) described the process as "…using GIS technologies to produce local knowledge with the goal of including and empowering marginalized populations." In 2012, they conducted a public participation GIS case study on the Chugach National Forest as part of the forest plan revision process under the 2012 planning rule. Results of their study indicate the potential utility of public participation GIS to assist forest planners in identifying areas suitable for various forest uses (Brown and Reed 2012). Essentially, Brown and Reed (2012) found that public participation GIS provides a systematic approach to identifying the social suitability of various forest uses to supplement traditional biophysical analyses and can assist the agency in determining whether particular activities or uses are consistent with desired conditions (box 4; fig. 9-7).

Effectiveness of public involvement approaches—
Reed (2008: 2417) suggested, “The complex and dynamic nature of environmental problems requires flexible and transparent decisionmaking that embraces a diversity of knowledge and values.” Studies suggest that public involvement can improve Forest Service analyses and provide information otherwise unavailable to the agency that may improve the quality of the decision (Creighton 2005, Hoover and Stern 2014). Scholars have identified additional benefits, including enhanced relationships, reduced conflict, public buy-in, and increasing compliance with agency regulations and removing barriers to project implementation (Koontz 1999, Stern 2008, Whitall 2007). Although these studies suggest that stakeholder participation can improve the quality of decisions, Reed (2008: 2421) asserted that they do so with one strong caveat: “…the quality of a decision is strongly dependent on the quality of the process that leads to it.” What follows is current research concerning institutional constraints as well as public involvement best practices that can either enhance or hinder the quality of public involvement and hence the quality of associated decisions. Another critical consideration affecting the quality of decisionmaking is the perception and effect of public involvement activities on indigenous peoples and nations.

Von der Porten and De Loë (2014) conducted a systematic review of collaboration literature that focused on environmental concerns and referred to indigenous peoples. Through this review they found that many collaborative processes are grounded in assumptions about the roles of

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Figure 9-7—Participatory mapping for travel management planning, Washington.

different members of society in decisionmaking that are incompatible with how indigenous peoples view themselves. While indigenous peoples have been portrayed as "stakeholders," "minorities," "groups," "participants," or as "nations," only the recognition of indigenous peoples as nations aligns with indigenous governance literature (Von der Porten and De Loë 2014: 1041). Thus, they affirm, "…how indigenous peoples are characterized and treated in collaborative processes is a sensitive and important issue" (Von der Porten and De Loë 2014: 1041). Chapter 11 discusses tribal governance and efforts to share traditional ecological knowledge and tribal perspectives as part of tribal engagement in resource management.

Institutional constraints—

Scholarly research concerning institutional constraints to public participation and the quality of decisions addresses three different levels within the U.S. Forest Service: agency, unit, and employee (Davenport et al. 2007, Kaufman 2006, Margerum and Whitall 2004, Stankey et al. 2003). Agency-level constraints have been attributed to diminished resources, organizational commitment, centralized power structure, and the statutory and regulatory environment (Davenport et al. 2007, Stankey et al. 2003).

Through their research, Davenport et al. (2007) identified the centralized system of decisionmaking as inhibiting the unit's ability to be responsive to the public and address their concerns in a timely manner. Unit-level constraints include increased division of labor, use of technical jargon in planning documents, and reliance on traditional forms of public involvement (Davenport et al. 2007). Here, an increasing division of labor, or specialization, among unit employees was found to reduce the unit's overall responsiveness to communities (Davenport et al. 2007). Additionally, Davenport et al. (2007) found that meeting minimum legal requirements for public involvement was not enough to stimulate local participation.

Employee-level constraints included staff turnover and long-distance commuting (Davenport et al. 2007, Margerum and Whitall 2004, Wondolleck and Yaffee 2000). Studies also acknowledge the difficulty in maintaining long-term relationships with local communities, private entities, and nongovernmental organizations from frequent turnover of personnel (Davenport et al. 2007, Margerum and Whitall 2004, Wondolleck and Yaffee 2000). Paradoxically, advancement within the Forest Service hierarchy is frequently dependent on personnel moving to different locations within the agency (Wondolleck and Yaffee 2000). Kaufman's study of the U.S. Forest Service (2006) found that by routinely moving field officers to different agency locations and levels, they formed allegiances to one another, the organization, and specific policies and procedures, allowing a large, dispersed organization with multiple objectives to successfully create a coherent, unified decisionmaking regime. Kaufman (2006) further acknowledged that this unified approach has not been without challenges, especially during times of social change. The 2012 planning rule's emphasis on collaborative development of land management plans represents a change from previous planning rule public involvement requirements by emphasizing the importance of building and maintaining relationships. Specifically, the planning rule final directives state that "public participation…helps build and maintain working relationships, trust, capacity, and commitment to the plan" (USDA FS 2015b: 3). Building and maintaining relationships takes time and requires access. Margerum and Whitall (2004) found that staff turnover slowed the momentum of collaboration efforts in southwest Oregon because of the time required for new participants to become involved and the different operating approaches that new managers held. Davenport et al. (2007: 47) emphasized these findings, noting, "Staff turnover has reduced the time communities and agency personnel have to get to know and trust one another. Long-distance commuting by agency employees has meant they are not actively participating in the community…"

Assessing success of public involvement and applying best practices—

Current research on the effectiveness of public involvement approaches is divided into two categories: (1) those that evaluate the success of **processes** and (2) those that evaluate the success of **outcomes** of processes (Chess and Purcell 1999, Cundill and Rodela 2012, Muro and Jeffrey 2008, Newig and Fritsch 2009, Renn 2006). Chess and Purcell (1999: 2685) acknowledged that "evaluating the outcome …is problematic because researchers cannot be sure if an effect is due to public participation efforts or to other variables." Yet they take a position in the middle of the process-outcome spectrum by arguing that "…neither "good" process nor "good" outcome is sufficient by itself." Cundill and Rodela (2012) agreed with this middle ground by suggesting that processes and outcomes work in tandem: improvements in processes such as sustained interaction, shared knowledge, and ongoing deliberation can lead to social outcomes of improved decisionmaking, better relationships, and improved problem-solving capacity. Muro and Jeffrey's (2008) research found additional social outcomes of participatory learning processes, including the generation of new knowledge, acquisition of technical and social skills, and increased trust. Finally, Newig and Fritsch (2009) supported Renn's (2006) argument that listening to the public and establishing a two-way communication stream is not alone sufficient: "Discursive processes need a structure that assures the integration of technical expertise, regulatory requirements, and public values" (Renn 2006: 9). In combining these processes effectively, Newig and Fritsch (2009) concluded that the ecological standard of decisions was positively influenced. Yet, Irvin and Stansbury (2004) argued for caution in deciding whether participatory processes achieve better outcomes on the ground. They found that certain situations precipitate "ideal" (low-cost/high benefit) conditions for public involvement, while other circumstances led to "ineffective and wasteful" (high-cost/low benefit) participatory processes (Irvin and Stansbury 2004: 62).

Finally, the literature shows broad consensus over key features of best practices in public involvement processes. Reed (2008) used qualitative methods and a systematic approach to derive key features from existing literature that includes the following:

- Stakeholder participation needs to be underpinned by a philosophy that emphasizes empowerment, equity, trust, and learning.
- Stakeholder participation should be considered as early as possible and throughout the process.
- Relevant stakeholders need to be analyzed and represented systematically.
- Clear objectives for the participatory process need to be agreed among stakeholders at the outset.
- Methods should be selected and tailored to the decisionmaking context, considering the objectives, type of participants, and appropriate level of engagement.
- Highly skilled facilitation is essential.
- Local and scientific knowledges should be integrated.
- Participation needs to be institutionalized, ensuring that decisionmakers are comfortable in committing to an unknown outcome of a participatory process, while understanding that ultimate decision authority resides with the agency.

Summary—
The quality of a resource management decision depends on the quality of the process that leads to it. A public involvement strategy that resonates with a dynamic and diverse range of interests helps to ensure sound resource decisionmaking. Best practices include a philosophy of empowerment, equity, and inclusiveness; systematic assessment of potentially relevant stakeholders and strategies to encourage participation; engaging stakeholders early in the process; iterative or frequent engagement throughout the process; clear objectives, timelines, and parameters; skilled facilitation; integration of local and scientific knowledge; and enduring agency commitment to the process. NEPA grants agencies broad discretion in the structure of public involvement; agencies engaged in resource planning are empowered to take advantage of the spectrum of public involvement approaches. Different planning phases may call for different levels of public involvement. Defining these various levels of engagement prior to initiation of plan revision promotes a robust participation process.

Agency-Citizen Collaboration

Contemporary natural resource management decisions present complex choices among interests and values, so that the choices are political, social, cultural, and economic, as much as they are scientific and technical (Dietz and Stern 2008). As a result, over the past several decades, communities, governments, private organizations, and individuals have increasingly turned to collaboration as a supplement to traditional planning and decisionmaking processes. By focusing on shared concerns and promoting problem-solving, the intent is to better address complex resource management issues such as watershed management, endangered species management, planning for climate change, or habitat restoration.

Collaboration is defined here as "a process through which parties who see different aspects of a problem can constructively explore their differences and solutions that go beyond their own limited version of what is possible" (Gray 1989: 5). Collaborative approaches are often place based, cooperative, involve multiple parties, and strive to create or improve relationships between individuals and groups, or develop solutions to specific issues or problems. The approach involves interactions with representatives of a variety of stakeholder groups and organizations, often over a period of months or years, depending on the scope and complexity of the group's efforts. Collaboration requires diverse stakeholder participants (private landowners, American Indian tribes, government organizations, nongovernmental organizations, businesses, and others) to work together over a period of time to identify and address resource management issues. The efforts often rely on outside neutral facilitators to help them work toward their common goals.

Why collaboration?—
The rise of collaborative approaches reflects a shift toward increased civic participation in agency planning and decisionmaking. This shift has occurred because resource management issues are not easily solved, are characterized by incomplete or contradictory information, and are subject to increasing interdependencies between management agencies, nongovernmental organizations, and citizens (Head 2008). Natural resource management also has become extremely complex and networked, as responsibility for many issues has shifted from the federal government to state and local governments as a result of shrinking federal government resources and programs (Emerson and Nabatchi 2015). Frustration with gridlock, declining budgets, and overall lack of trust in government decisionmaking processes have fueled interest in collaboration, as have challenges with the multiple jurisdictions and landowners needed to effectively manage resource issues across landscapes (Dukes and Firehock 2001, Wondolleck and Yaffee 2000).

Societal expectations and policy-driven requirements for public involvement in resource decisionmaking have also increased the use of collaborative approaches. For example, in the Forest Service, "…laws such as NEPA, NFMA, and the Endangered Species Act (ESA) provided important leverage to conservation groups and gave them an empowered seat in collaborative processes" (Nie and Metcalf 2015: 6). Nie and Metcalf (2015) summarized the evolution of collaboration in the Forest Service, noting that "collaboration was increasingly invoked to facilitate a more inclusive dialogue as part of a new focus on ecosystem management in the 1990s," and the two were linked together by the Forest Service's Committee of Scientists (1999), which recommended more ecosystem and collaborative-based approaches to forest planning (Committee of Scientists 1999). Collaboration was also called for in the 2003 Healthy Forests Restoration Act, the 2009 Collaborative Forest Landscape Restoration Act, and the 2012 NFMA regulations, which focus extensively on public participation in forest planning, with collaboration encouraged by the agency, and public participation required during plan development, revision, and amendment (Nie and Metcalf 2015).

Collaboration is touted as an appropriate approach because many resource management issues are local, site specific, and often cannot be easily resolved within legislatures, agencies, or courts (O'Leary and Bingham 2003). Proponents of collaboration argue that it is a logical response to policy gridlock and litigation (Susskind et al. 1999) and an alternative to centralized planning and command and control regulation. Collaboration can produce more creative and adaptive solutions to natural resource management problems, encourage shared ownership of the problem, and facilitate implementation of potential solutions (Bacow and Wheeler 1984, Susskind et al. 1999). Such efforts also can garner sufficient resources or expertise to achieve what cannot be accomplished by one single party or a smaller coalition, and is often less costly than litigation

(Dukes and Firehock 2001, Susskind and Ozawa 1984). In many cases, collaboration has proven to be a powerful tool for resolving conflict, building trust, addressing uncertainty, fostering cooperation and coordination, and developing capacity for addressing future resource management issues (Wondolleck and Yaffee 2000). Collaboration is often viewed as part of the solution to increasing trust and social license for forest management.

General critiques and concerns about collaboration—
Critiques of collaborative approaches argue that the process does not necessarily ensure "better" decisions, and that collaboration may reinforce existing power disparities rather than promote truly diverse stakeholder inclusion and meaningful dialogue (Burke 2013, Dukes and Firehock 2001). Not all stakeholders can or will participate; there may not be enough time to resolve the issues; the issues may not be "ripe" or ready for collaboration; and there are serious capacity concerns related to the time and other resources needed for participation (Amy 1987).

Other studies have raised concerns about the devolution of public lands management and suggested that collaboration could potentially weaken environmental protection (Hibbard and Madsen 2003, Kenney 2000). Questions have been raised about the nature and quality of the environmental outcomes from collaborative processes, which is an enduring question across all sectors. Layzer (2008: 5) suggests that "…the initiatives whose goals were set in collaboration with stakeholders have produced environmental policies and practices that are less likely to conserve and restore ecological health than those whose goals were set through conventional politics." More recently, efforts to understand links between collaboration and performance reveal that while there is a perceived positive link between the two, concerns remain about costs in terms of power, time, conflict, stress, process, suboptimal outcomes, and resources required (Mitchell et al. 2015).

Structures and functions of collaborative approaches—
In practice, collaboration is designed and implemented in a wide variety of ways. Differences in structures and functions across several key factors illustrate multiple interactions, each of which affect the processes and outcomes of a particular effort. Collaborative efforts can vary in several ways including:

- Who sponsors (funds all or part)
- Who convenes (plans and leads)
- Facilitation
- Scope (local, state, regional, national, international)
- Jurisdictions, authorities, and laws
- Geographic scale (related to scope)
- Participants and who they represent (more inclusive or less inclusive)
- Purpose and goals (e.g., policy or issue oriented; or site specific and focused on a specific issue related to a particular place)
- Drivers of the effort, such as direct conflict over an issue, or a perceived opportunity
- Urgency of issues and timeframe for decision
- Formal and informal rules (decisionmaking approaches—ranging from consensus to voting to agency maintaining decision authority, managing interactions over time)

Types of federal forest land collaboration and trends in the Pacific Northwest—
Research on federal forest land collaboration in the Pacific Northwest has covered collaboration at scales from local communities of place to larger landscapes and regions. It has tended to focus on collaboration for wildfire risk reduction and forest health restoration, and particularly on collaboration during planning (e.g., before and during NEPA analysis, or community wildfire protection planning). Although collaborative watershed management for fish habitat and other aquatic restoration goals has also become common practice, most of these efforts have focused on private landownerships and capacity to implement restoration projects on the ground, so science on this topic is not reviewed here.

Because there is no established baseline from which to begin, it is difficult to accurately and completely describe the status of collaborative forest management on national forests, and whether or how this has changed over time. Anecdotally, there are a plethora of types of collaboration in the region, yet there are few if any studies that comprehensively document this (censuses, statewide assessments, etc.). Some documentation can be found in policy or program reports (Bixler and Kittler 2015, Swezy et al. 2016, White et al. 2015), or student theses (Hughes 2015, Spaeth 2014, Summers 2014), but there is no single standard for defining, identifying, or studying collaboration. Although much of the existing science on collaboration consists of case studies, the research has identified common themes, challenges, lessons learned, and best practices. Two overall trends emerge from the available science:

- Collaboration takes place at a variety of spatial and political scales but is increasingly occurring over larger landscapes as federal policies and programs have focused at larger scales over time.
- Collaboration is increasingly occurring through collaborative groups ("forest collaboratives") that meet regularly and focus on a specified landscape, rather than individual processes or projects. Organization and leadership of these groups differ greatly by location and context.

Changes in the scale of collaboration over time— Collaboration often is thought of as occurring at the local community scale, referring to communities of place. This is a very fine-scale approach. Following the NWFP and listing of species such as the northern spotted owl (*Strix occidentalis caurina*) and Chinook salmon (*Oncorhynchus tshawytscha*), community-based collaborative efforts arose in several affected communities in the Pacific Northwest states. These are documented in previous science syntheses covering the period prior to 2003 and are often described as community-based forestry, community forestry, community-based conservation, or grassroots ecosystem management (Baker and Kusel 2003, Weber 2003). Early efforts were often spurred by local community members, typically working on a range of projects including, but not limited to, federal lands management, e.g., local business development or community multiparty monitoring. The goal of these efforts was to improve ecological and socioeconomic conditions in a given place, and leadership was local.

From 2001 onward, there has been a trend toward collaboration driven in part by state and national policies and programs. A primary focus of more community-scale collaboration since 2003 has been community wildfire protection planning, spurred by the National Fire Plan (2001) and Healthy Forests Restoration Act (2003). However, the scale of these processes and plans differed, as some plans covered subdivisions or neighborhoods, while others were for entire counties (Jakes et al. 2007). Currently, there are community wildfire protection plans (CWPPs) in nearly every county and at smaller community scales in the NWFP area, indicating that this form of collaboration has become widespread (Oregon Department of Forestry, Washington Department of Natural Resources, California Fire Safe Council). However, the nature of collaboration around the plans also differed; some were largely developed by consultants and others through extensive community engagement and collective action (Williams et al. 2012). Over time, there has been an increase in community-scale groups collaborating beyond this planning process under the "cohesive strategy" process (USDA FS 2014). Twelve communities (two of which are in the Plan area) are collaborating through the tools provided by the Strategy to become "fire-adapted communities." They also are participating in a larger nationwide network (Fire-Adapted Communities Network).

No scientific research has comprehensively reviewed these CWPPs, but there has been some case study research. The largest study included 13 CWPP cases at diverse scales, (including two cases from the Plan area), and found that successful CWPP processes emphasized problem framing, choosing tractable scales, and ensuring a path toward implementation (Williams et al. 2012). Other studies found that trust was an essential ingredient in two cases in west-central Montana (Lachapelle and McCool 2012). However, one study of two cases in Oregon (one in the Plan area and one in eastern Oregon) concluded that CWPP processes were not necessarily successful for future wildfire risk reduction, in part because communities could not or did not establish effective decisionmaking processes or have sufficient influence to induce change (Fleeger and Becker 2010).

Collaboration in the Pacific Northwest has more recently shifted to a focus on watershed and landscape-scale restoration. The term "landscape" has multiple definitions and expressions. For the Forest Service, the Forest Landscape Restoration Act (2009) and resulting Collaborative Forest Landscape Restoration Program (CFLRP 2010) allocated funds to national forest units and their collaborative partners to work on landscapes of at least 50,000 ac (20 234 ha) with a 10-year plan of prioritized restoration treatments (Schultz et al. 2012). Within the Pacific Northwest states, there are currently nine CFLRP projects (USDA FS 2015a). These are largely outside the NWFP area, given the program's wildfire focus. Portions of the Okanagan-Wenatchee and Deschutes National Forests, where two CLFRPs are active, are within the Plan area. Insights about the program, however, may be applicable to future collaborative efforts within the Plan area.

An initial study of the CFLRP suggested that collaborating at the landscape scale and requiring monitoring could result in more efficient future forest management (Schultz et al. 2012); but further research indicates that barriers such as lack of stakeholder and agency capacity may be emerging (Schultz et al. 2014). Another monitoring report identified inconsistent implementation of socioeconomic monitoring among the CFLRs (Swezy et al. 2016). Other recent research has dug more deeply into collaborative processes and collaboration during the implementation stage (Butler et al. 2015). Collaboration during implementation expands possible roles for collaborative groups, which in the past have been confined to planning and monitoring activities. Collaboration during the implementation phase also may strengthen accountability and stakeholder diversity (Butler et al. 2015). Increasing collaboration during implementation may pose new legal tensions; meanwhile, given that ultimate decisionmaking authority remains with the land management agency. Another related study finds that the program's mandate to collaborate may lead to increased stakeholder engagement and attention to designing effective collaborative processes (Monroe and Butler 2016). Bixler and Kittler (2015) conducted a meta-analysis of CFLR research to identify research gaps. The biggest needs in research were related to leadership, trust, and accountability. Finally, Urgenson et al. (2017) examined six CFLRP collaborative groups, identifying common challenges (meeting multiple objectives; collaborative capacity and trust; and integrating ecological science and social values in decisionmaking) and strategies used to overcome these challenges.

From "collaboration" to "collaborative"—
There is little scientific documentation of how the organization and structure of federal forest land collaboration may be changing (Davis 2015a, 2015b, 2017) and general knowledge of forest collaborative groups in the NWFP gained from discussions with colleagues in regional and national conferences as well as preliminary (unpublished) research in Washington and Idaho. Numerous forest collaborative **groups** or "collaboratives" are now active on national forests in the NWFP area—working together and with the Forest Service beyond individual processes or projects (fig. 9-8). These groups typically have a recognized name, mission, and a regular process for meeting, reviewing federal land management activities, and providing collective input to the Forest Service. But, there is no single definition of "collaborative" currently found in federal or state policy, and likely a great degree of variability in what these collaboratives do and how they are organized between and among states.

The trend of organized collaboratives groups has been extensive in Oregon, where an estimated 25 forest collaboratives are currently identified as active in all national

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Figure 9-8—South Santiam All-Lands Collaborative, Oregon.

forests (Davis et al. 2015a, 2017). Other Western states (California, Colorado, Idaho, Montana, and Washington) also have growing numbers of identified collaborative groups. Although some of these efforts date to the 1990s, particularly in California and Oregon, a majority appear to have originated more recently (post-2009). No comprehensive assessment or empirical research on these collaboratives has been conducted. Existing research suggests that "collaboratives with formalized structures and workgroups" tend to be more successful at attracting and using resources than less formalized groups (Cheng and Sturtevant 2012). The capacity of these groups to organize and accomplish their goals has not been tested.

Elements of successful collaboration—
Notions of success depend on the particular goals of the participants in the collaborative process, and evaluation is challenging because collaborative efforts produce a variety of different products. The large variation in the structure, function, goals, context, and terminology of collaborative processes poses challenges for researchers, making it difficult to operationalize and measure collaboration and collaborative performance, and there is little agreement about what constitutes effective performance in collaborative arrangements (Emerson and Nabatchi 2015). Further, a majority of existing science on collaborative evaluation examines single or few case studies, and often, only a few factors of success. Therefore, the amount of generalizable or comprehensive information about what constitutes successful collaboration is limited. In particular, little to no research explicitly evaluates collaboration in the NWFP area.

Evaluations of collaboration that do exist have tended to focus on combinations of process, social, environmental, or economic criteria (Conley and Moote 2003). For example, collaborative efforts are often deemed successful based on process outcomes such as whether the effort established a shared vision among participants, had diverse and inclusive participation, used an open and transparent process, made links to groups beyond those participating, and made decisions by consensus. Social outcomes could include relationships built or strengthened, increased trust, whether participants gained knowledge or understanding, increased capacity for dispute resolution, or changes in existing or new institutions. Process and social outcomes can be particularly confusing to analyze, because they can also be factors in success as well as evidence of success. A study conducted of six projects in national forests in eastern Oregon suggests that four of the six projects that had input from collaborative groups appeared less likely to be appealed (Summers 2014); more research is needed to understand potential intervening variables and to identify a clear definition of "collaboration" among these cases (box 5).

The environmental outcomes of collaboration are also difficult to evaluate owing to monitoring challenges, relatively long time frames between implementation of collaborative outputs and detection of environmental responses, and demonstrating that implementation of collaborative involvement (rather than other factors) changed environmental conditions (Koontz and Thomas 2006). Further, scientific studies of economic impacts of collaboration are also limited, although some policies and programs are increasingly requiring monitoring of job creation and economic activity. Similar to the environmental outcomes challenge, linking specific collaborative activities to economic outcomes is quite difficult.

More recent efforts to understand links between collaboration and performance reveal that while there is a perceived positive link between collaboration and performance, concerns remain about costs in terms of power, time, conflict, stress, process, suboptimal outcomes, and resources required (Mitchell et al. 2015). Margerum (2011) presented an overview of the principal elements of successful collaboration (see box 5), which can focus on inputs (information for decisionmaking); process (such as the equitability, diversity of participation, and other aspects of the decisionmaking process); outputs (assessing products, such as plans or agreements); performance (measuring plan and policy performance); outcomes (monitor actual results); and program logic (linking outputs to outcomes).

The available research on successful collaboration in federal forest lands is quite limited, and focuses on selected factors of success. No research is available that addresses all possible factors of success. From the available research, the following can be important ingredients for success:

Box 5— Factors Identified With Successful Collaboration

Inputs:
- Clear goals
- Available information
- Appropriate scope
- No fundamental value differences
- Issues are “ripe” for collaboration
- Appropriate scale
- Appropriate authority

Process:
- Shared information
- Trust/good faith in participation
- Decision rules
- Shared vision/goals
- Diverse participation
- Satisfaction
- Membership
- Facilitation
- Legitimacy
- Support for agreement/process

Outputs:
- Plan, agreement or project
- Implementation plan
- Monitoring or enforcement
- Clear communication
- Shared, high-quality fact base
- Intervention strategy

Outcomes/performance:
- Difficult to evaluate because of time lag associated with implementation
- Difficult to link to collaborative effort unless well documented and monitored
- Research needed linking outcomes to specific actions of collaborative groups

- Collaborative capacity, or the ability to “organize, coordinate, and manage people, resources, and tasks to achieve desired outcomes” (Cheng and Sturtevant 2012: 2).
- A shared culture and set of behaviors (Cheng 2006).
- Inclusion of all interests (Hibbard and Madsen 2003)
- Undertaking monitoring, joint fact-finding, or other information gathering and learning activities, which has been found to lead to shared understanding, social learning, a sense of community, and trust; however, research also finds that monitoring data are often not being used or rarely results in adaptive management (Fernandez-Gimenez et al. 2008).
- Agency participation, which can demonstrate commitment and bring technical knowledge and support (Butler 2013, Wondolleck Yaffee 2003).
- Genuine non-agency, community leadership (Cheng 2006).

Summary—

Collaboration has been widely accepted as a useful model for engaging diverse stakeholders in the process of deliberation over critical forest issues, including fire, forest restoration, and wildlife protection. Collaboration requires time, resources, and enduring commitment. Managers choosing to participate in collaborative efforts may want to ensure they have adequate resources and provide support to staff to enable them to build these relationships over time. Successful agency participation in collaboration includes open communication, clear expectations, and realistic information about internal priorities, plans, schedules, and decision points. Collaborative groups may be part of the solution to increasing trust and social license for forest management to meet NWFP and other goals. Yet, not every stakeholder is eager to participate in a collaborative. The scope, scale, and end goals of a given collaborative effort may make it more relevant and accessible to some stakeholders than others. Currently, collaboratives in the NWFP area exist in a variety of forms and are engaged in a diversity of activities, including stewardship contracting, environmental assessment for NEPA, and monitoring (Davis et al. 2015b). Federal land management agencies have committed resources to collaboration and have invested in its long-term success. Collaboration is often viewed as part of the solution to increasing trust and social license for forest management. Yet, we do not have consistent evidence that collaboratives are achieving the goals that proponents have espoused, including trust building, better results on the ground, a reduction in appeals or litigation, or improved forest health.

Research Needs, Uncertainties, Information Gaps, and Limitations

Socioecological systems science recognizes that human societies and ecological systems are interwoven and interdependent (Berkeset al. 2000). We understand that ecosystems are embedded in levels of social organization (Brondizio et al. 2009). Although contemporary resource management is built from a strong foundation of ecological information, knowledge of our social systems and how they integrate with ecological elements at the appropriate geographic scale is sorely lacking. The need is great for data that describe the socioeconomic, psychological, cultural, and political landscape in the NWFP. This chapter characterizes the current state of knowledge in the planning area and identifies the gaps.

In this section, we identify the most high-priority research needs and significant gaps in knowledge for each of the five key findings of this chapter. The dearth of social science research in the NWFP area creates a need for a wide variety of studies to understand more about changing values and relations with place, changing recreation patterns, and changing expectations for public involvement in resource management. The recent emergence of collaborative forms of governance is creating new opportunities for the public to engage in resource management, although the science has not yet captured the benefits and challenges of collaboration.

Public Values, Attitudes, and Beliefs

Values orientations of North Americans are said to be shifting from an emphasis on resource production to a balance of resource protection and production. However, little research has been conducted in the past 20 years to assess the current status of environmental values, either nationally or regionally. Longitudinal data is sorely lacking in the social sciences, as most research studies focus on a single case (Stidham et al. 2014). Longitudinal social values monitoring in the NWFP area would help to evaluate regional trends and identify any subregional variations or disparities among urban, amenity-migrant, and rural residents. Values research also would be useful to identify value sets held by sociocultural groups and stakeholders with a keen interest in management of the NWFP lands. A better understanding of stakeholder values will help land managers identify and predict attitudes toward resource management practices such as restoration.

Constraints to conducting this work are primarily budgetary and regulatory. First, the cost of random-sample survey work has escalated in recent years, and societal trends in favor of privacy have made it challenging to collect survey data with an adequate response rate from the study population. New ways to budget for this type of consistent social data might be considered as well as ties with existing agency monitoring programs. Second, any survey being conducted by federal government agencies or on behalf of federal agencies is required to undergo review by the OMB per the Paperwork Reduction Act of 1995. This OMB approval process must ensure that the public is not being unduly burdened or harmed by the study or that the project is not redundant. Interagency coordination of public values studies within the NWFP area may help to increase efficiencies associated with obtaining OMB approval.

If a process was established to monitor public values at reasonable iterations (every 10 years, for example), data collection and analysis could be standardized and institutionalized, with efficiencies gained. These data would benefit multiple resource agencies and provide the opportunity for public values and environmental beliefs to be considered by federal, state, and municipal land managers. Without these data at the regional or local scale, land managers are left to make decisions based on information gathered in public meetings or in small studies that are not coordinated regionally and may not accurately reflect the definitive views of a diverse public.

Finally, new research about the social acceptability of forest management practices is warranted, with emphasis on public knowledge and perceptions of ecological forestry as well as traditional forms, reactions to a diversity of treatment types and setting conditions, and with a focus on understanding the role of trust and communication on shaping public responses.

Valuing Place

Extensive literature exists on the concept of place and its relationship to natural resource management. However, very little of this research has taken place in the Pacific Northwest or California. The need for applied research that helps managers and policymakers capture and integrate into planning the place meanings associated with forest socio-ecological systems is therefore great. Findings from such studies can be applied to the NWFP area.

One area for place research that has immediate practical utility for land managers is improving understanding of the physical and social characteristics associated with places that tend to trigger place-protective behaviors. This knowledge would be useful for land managers making decisions about what places to emphasize for protection through management, where to focus management, or where to allow certain activities to occur. Places mean different things to different people. A related research topic has to do with identifying how place attachment differs within forest user groups. For example, boaters residing near a lake may resist proposed closures of picnic areas on the lakeshore, whereas boaters who come from a distance may be indifferent to those closures. More broadly, place research can also help managers better understand where to allow certain activities by forest users and where to focus vegetation management and other activities.

A gap exists in our knowledge about which public engagement methods are most effective for capturing place meanings for different segments of the public. Studies suggest that Internet and mail surveys as well as standard mapping workshops tend to be biased toward forest users who are older, better educated white men with relatively high incomes. A workshop format proved successful at reaching Latino forest product harvesters on the Olympic Peninsula (Biedenweg et al. 2014), forest users who were not reached through a broader mapping workshop process. However, only the tip of the iceberg has been seen so far, and more work needs to be done to determine which methods work for which segments of the population. Additionally, there is a mismatch between the techniques likely to be effective at reaching nontraditional forest users and the capacity of the land management agencies to conduct such outreach. A grassroots-driven mapping project on the Mount Baker–Snoqualmie National Forest offers a promising avenue for how land management agencies can work with partners to augment their capacity for developing place meaning inventories (McLain et al. 2017a).

Finally, a regional or forestwide sense of place assessment might usefully inform forest management decisions and public engagement processes. Key questions might be: What places matter to people? To whom do they matter, why, or for what? How much do they matter? And, under what social, economic, political, and ecological circumstances do they matter? Because the answers to these questions are likely to differ depending on the socioecological context, a clear need exists for research that moves beyond isolated case studies toward a coordinated set of regionwide applied research projects. There is a need for applied research on place meanings and related concepts, such as place attachment, place identity, and place dependence. By applied research, we mean the systematic collection and analysis of data that are deliberately structured so as to inform land management and policy.

Cultural Ecosystem Services

A 2015 presidential memorandum directs all federal land managers and regulatory agencies to use an ecosystem services framework for planning, policymaking, and decisionmaking (OMB 2015), and consideration of ecosystem services is also required by the 2012 forest planning rule (USDA FS 2012). New research on cultural ecosystem services has drawn attention to the less tangible benefits associated with forest ecosystems. However, there have been few empirical studies that explore how people perceive cultural services, how cultural services may be operationalized, measured, and monitored, and how to integrate cultural services into the planning process. Case studies in ecosystem services management with inclusion of cultural services are needed to build a foundation of knowledge. In addition, tools and applications are needed that reinforce concepts in cultural ecosystem services. Early studies and cases in the NWFP area offer new insights and promising prototypes. In particular, it would be helpful to learn more about how cultural services have been bundled in various ways and whether it makes sense to unbundle them.

Outdoor Recreation

Much of recent scientific literature on U.S. outdoor recreation is at the national level or from studies in other U.S. regions. Just because those studies were not completed in the NWFP area does not mean their findings are not transferable to the NWFP area. However, it would be useful to increase the amount of recreation research in the NWFP area to learn whether there are unique patterns locally. Much of the traditional scientific literature on national forest recreation focuses on what the authors thought were traditional national forest recreation activities, such as backpacking, primitive camping, and hunting. Contemporary recreation use is much more focused on short visits, often to developed recreation sites, and focused on generalist activities. New research is needed to understand how the Forest Service fits into current and future demands for the full suite of leisure patterns by Americans, and specific desired outcomes from national forest recreation. The recreation research literature lacks comprehensive studies of multiple, combined factors of sustainable recreation; most of the current literature has focused on one or two factors (e.g., economic impacts, social impacts) individually, and does not look at the whole array of factors in an integrated fashion. Comprehensive studies that consider integrated factors of sustainable recreation will inform managers as they respond to policy direction to manage for sustainable recreation.

Trust

Research on trust has identified and defined multiple types of trust in the context of natural resource governance. Stern and Coleman (2014) identified four types of trust: dispositional (one's natural inclination to trust); rational (stemming from predictable behavior, past performance, and reasoned logic); affinitive (based on personal relationships that develop through repeated encounters); and procedural (based on having processes that are viewed as fair, just, and open). A follow up study found that at least three types of trust needed to be present for broader institutional trust to be acknowledged (Stern and Baird 2015). Although this study was conducted in the context of natural resource collaborative groups, the trust typology is applicable to other forms of forest governance. More research is needed to understand the various types of trust and how they interact. For example, how does affinitive (interpersonal) trust relate to broader agency trust? What happens when rational trust declines, while procedural trust grows? What types of processes and protocols help to enhance procedural trust?

There are opportunities to explore how broad-based trust for a public agency affects project-level trust, and vice versa. In other words, what happens when the public distrusts the agency at the national policy level but has greater trust in the ability of local officials to manage a project? And, what happens when there is high broad-based trust in an agency's purpose, but lack of trust at the project level? In addition, more information is needed about the ways that public trust can influence social acceptability of forest management practices, such as active management and forest restoration as well as prescribed burns. Finally, trust can be enhanced through participation in various types of public engagement opportunities and in collaborative or comanagement groups. Yet, it is not clear what types of trust may be generated by these different types of processes.

Public Involvement

Peer-reviewed research on public involvement is limited in the NWFP area. Broad-scale questions exist concerning the disconnect between participatory requirements and Forest Service structures and cultural norms. Practical policy implications of public involvement in decisionmaking lead to questions of accountability, such as: To what degree does the agency give up some control and still maintain or improve social, economic, and ecological outcomes? Perhaps more fundamental is the question: What results are important to achieving socially sustainable outcomes? The application of stakeholder and social network analysis within the government sector remains largely untapped, and could provide a wealth of knowledge about building effective networks in support of the public good. Specific questions also exist concerning public involvement methods employed by the "early adopter" national forests engaged in planning under the 2012 planning rule.

Forest Service social scientists can play a much greater role in assisting the agency develop innovative public involvement strategies. Although the field of social science is diverse, expertise exists in the methods and practice of understanding values, attitudes, and beliefs; stakeholder and social network analysis; and place identity and attachment through tools like public participation GIS. Integration of these foundational methods and practices into programmatic forest planning can increase the likelihood that land management decisions better reflect the diverse range of public and tribal interests.

Agency Collaboration

Very little research has been conducted on forest collaboration in the NWFP area, and our synthesis only examines research on Forest Service-related collaborative efforts. Therefore, many gaps and research needs exist. First, large-scale, comprehensive studies of collaborative forest management and how it has changed are needed, including basic information about how many collaboratives exist and how they are defined and function. It is not possible to say whether communities are more engaged with the agency than before, or how, as there is no commonly accepted definition of "engagement" or what would indicate "more engagement." Research on collaboratives tends to occur via case studies, and single case studies are not generalizable to broader scales. Further, because much collaboration has occurred around wildfire in dry forests, few studies have focused on the NWFP area, which means there is even less clarity about the drivers, activities, outputs and outcomes of collaboration. Many NWFP forests differ from their east-side counterparts in terms of higher annual precipitation, longer historical fire-frequency intervals, diverse moist/wet forest types, presence of endangered species, different forest health challenges, increased population density, and proximity to urban areas; southern Oregon and northern California forests have more frequent fire and a mixture of east- and west-side characteristics.

Second, no scientific evaluations have been conducted on whether, or how, collaboratives are achieving resource management goals or social or economic objectives. No studies measure these goals or outcomes, or identify what can be attributed specifically to collaboratives as opposed to other variables such as economic change, agency or other organizational change, efforts from programs and activities occurring outside the collaboratives, etc. One study monitored changes in several indicators such as timber harvest, acres restored, and jobs created as a result of the state's Federal Forest Health Program in Oregon, but it did not clearly link these outcomes to specific activities of collaboratives (White et al. 2015). One master's thesis examined the question of whether collaboratives in eastern Oregon are decreasing appeals—a topic of great interest to managers and policymakers—but the evidence was largely inconclusive (Summers 2014).

Beyond these broad questions, additional questions about collaboratives remain unanswered and present rich opportunities for future research. Some of these relate to process: How do collaboratives build, codify, and use social agreement? Does the agency use the agreements that collaboratives make, and if so, how? Other questions relate to the roles of various stakeholder types (e.g., industry, conservation, American Indian tribes) and whether and how they participate. How collaboration relates to other processes such as consultation with tribes or the established public process used during environmental planning is not well understood. Moreover, the potential tradeoffs of forming an enduring collaborative group versus a collaborative process for specific issues or decisions have not been assessed. For example, do the investments in capacity and organizational development of collaborative groups pay off in outcomes? Do collaborative groups offer input that accurately reflects the spectrum of public values? In general, the need for collaborative groups or processes may differ and collaboration may not provide solutions for all issues in all places. Additional questions concern collaboratives and knowledge production, as many collaboratives focus on "science-based restoration." How compatible are scientific and collaborative processes? What role do collaboratives and agencies have in bringing science to bear on management? Finally, few if any studies have examined the ability of collaboratives to endure and adapt in the face of shocks and stressors, such as large wildfires, climate change, or community social discord.

Conclusions and Management Considerations

Conservation goals are most often met when ecological and social elements are integrated (Charnley 2006). Socioecological system (SES) science recognizes that changes to society, including demographic shifts, changes in human settlement patterns, new governance structures or regulations, can affect interactions with the natural environment and likewise, large-scale landscape and climate variations can affect human institutions, such as markets and communities. Greater awareness of how social and ecological systems intersect will help resource managers improve the quality of their decisionmaking. This includes greater understanding of human values and management preferences, place-relations, resource uses, and visitation patterns. New participatory strategies have attempted to democratize and deepen citizen engagement in environmental decisionmaking. The proliferation of collaborative institutions has the potential to influence future management of ecological systems. As agencies expand their conceptualizations of forest resources from "sustained yield" to "ecosystem management" to "ecosystem services" in the NWFP area, there are no doubt implications for what this means on the ground. A greater recognition of diverse stakeholder values and place attachments, of shifting visitation patterns across the forest landscape, and of the opportunities for public expression of these values will enhance efforts to understand the NWFP area as a dynamic and integrated system. This chapter notes several highlights of interest to resource managers.

Systematic and steady research and monitoring of public values will enhance our understanding of what is important to people living in and around the NWFP area, or who have a stake in the future of these lands. A greater understanding of environmental values associated with public lands can improve the ability of land managers to weigh public needs alongside the best available science to make management decisions. Information about public values, attitudes, and beliefs also allows managers to anticipate conflicts in values and develop strategies for communicating with stakeholders. As societal values shift, public responses to resource policies and decisions will also likely evolve. And, as population changes occur, new migrants can influence the existing composition of values and value orientations within a particular subregion. Land managers who have access to up-to-date information about the values, beliefs, and preferences of both the general public and a variety of stakeholders and socioeconomic groups at the appropriate geographic scale will be better equipped to understand characteristics of their social system and anticipate the need for change. Moreover, social system data gathered at the appropriate scale can be integrated with biophysical data about ecological conditions to expand understanding of the complex socioecological system and identify possible barriers and opportunities in implementing management plans.

Residents of the NWFP area embody a range of views related to the social acceptability of various land uses, including timber harvest and these views are based on their environmental values, connections to place, knowledge of harvest practices, awareness of goals and outcomes, and degrees of trust. Existing research suggests that stakeholders and citizens in the NWFP generally do not support clearcutting as a desirable silviculture strategy. There does appear to be potential public support for alternative harvest strategies, such as disturbance-based management and other practices that mimic natural processes, particularly when old-growth forests can be avoided.

Recognition of the symbolic meanings and emotional ties that people have with places is important for engaging people in stewardship efforts, encouraging collaboration, and engaging the public in resource management.

The scientific evidence about the importance of place is unequivocal. Places and the meanings that are bound up in them have real implications for forest ecosystem management. Managers can take advantage of the positive power of place as the bonds that people have with places can also motivate them to engage in forest stewardship projects. By recognizing and appealing to these bonds, managers may be able to attract volunteers with substantial knowledge of local conditions to accomplish objectives.

The variability that exists in place meanings, together with the strong feelings that are bound up in people-place relationships, suggests that broad-based public engagement processes are critical at an early stage of the planning process. There are great benefits in considering the dynamics of place in a systematic way. For example, a conflict over forest management that threatens the social identity of stakeholders may call for a very different outreach and involvement approach than another type of conflict where place is important, but not critical to social identity. Because place meanings are dynamic and constantly being renegotiated, a public engagement process that emphasizes multiple ways of collecting data about place meanings and that is deliberately designed to reach out to a broad spectrum of the public, is far more likely to capture the range of meanings than processes that rely on only one type of information gathering approach.

Cultural ecosystem services provides a framework for land managers in the NWFP area to consider the diversity of spiritual, aesthetic, recreation, heritage, discovery and learning, and therapeutic benefits associated with forest settings. Recreation is one example of a quantifiable and measurable benefit that currently is monitored by federal agencies, along with scenic resources and heritage sites. Other aspects of cultural ecosystem services, like spirituality, solitude, wilderness therapy and education, are managed, but not actively tallied, which is a missed opportunity. A growing emphasis on cultural ecosystem services will allow resource managers to recognize the variety of benefits associated with a forest and stakeholder attachment to sets of benefits. The ecosystem services framework may prove useful in identifying and measuring a full range of benefits and values assigned to forests and landscapes.

Recreation visits are expected to grow in day-use settings and developed facilities. At the national level, current scientific literature indicates that the numbers of people participating in outdoor recreation will increase in the coming decades with continued population growth, although participation rates will be relatively flat. From general public surveys and NFS visitor monitoring, we know that the vast majority of outdoor recreation use is for general recreation activities, like hiking, viewing nature, visiting nature centers, viewing wildlife. Further, NFS recreation monitoring indicates that the vast majority of recreation visits to national forests are relatively brief, lasting less than one-half day, and the majority are focused on recreation at developed sites (e.g., day-use areas, campgrounds, visitor centers). Recognition of that pattern of use is helpful when considering the amount of resources committed to managing general, common recreation activities versus more specialized, but perhaps higher profile, activities in which fewer people are engaged. The greatest barriers to participating in outdoor recreation identified in the literature are items over which Forest Service managers have limited control: lack of time and distance to national forests. For items the Forest Service can control, expanding personal safety at recreation sites, improving signage, and providing information, all have been found to have positive effects on user perceptions of site conditions.

Developing decisionmaking processes and protocols that are consistent, reliable, fair, and transparent can help to improve institutional trust. The degree of trust established between public agencies, stakeholders, and communities is an important factor in public support for resource management decisions. Trust is not fixed, but rather is dynamic and iterative—modified by each encounter and shared experience. Public trust of natural resource agencies may vary between the local (or project) level and the national (broad-based) level. Trust exists in multiple forms: dispositional (based on inclination), rational (based on predictability), affinitive (based on relationships), and procedural (based on process), and although trust levels between entities can vary among those four types, enduring trust cannot exist without at least three of these present. Having clearly stated objectives, consistent communication, transparent processes, reasonable timelines, honored commitments, and opportunities for candid deliberation can enhance trust. Enduring personal relationships also are important.

The quality of a resource management decision is strongly dependent on the quality of the process that leads to it. The necessity of designing public involvement strategies that resonate with a dynamic and diverse range of interests is imperative to ensuring sound decisionmaking affecting Forest Service lands. Best practices provide critical components of an effective public participation strategy, and offer useful guidance for incorporation into decision processes (Reed 2008), including: a philosophy of empowerment, equity, and inclusiveness; systematic assessment of potentially relevant stakeholders and strategies to encourage participation; engaging stakeholders early in the process; iterative or frequent engagement throughout the process; clear objectives, timelines, and parameters; skilled facilitation; integration of local and scientific knowledge; and enduring agency commitment to the process. The NEPA grants the Forest Service broad discretion in the public involvement process, and therefore this process should take full advantage of the spectrum of opportunities available.

Inconsistent use of "collaboration" as a catch-all term for public involvement has often led to conflicting expectations on the part of agency employees, stakeholders, and tribal entities. These different expectations can result in reduced trust, and more importantly, less willingness to participate in long-term planning. The IAP2 Public Participation Spectrum can assist in selecting the level(s) of participation that define(s) the public's role throughout a forest plan revision effort. Different phases of plan revision or any management decision process may call for different levels of public engagement. The key is defining these various levels of public participation prior to initiation of plan revision, ensuring the public a meaningful and robust participation process (Arnstein 1969).

Collaboration takes time, resources, and long-term commitment from all parties. Managers seeking to initiate or participate in a collaborative process or group may want to consider that collaboration, particularly through an organized group, is typically time consuming, requiring tremendous commitment and effort to build and maintain relationships. Managers choosing to participate in collaborative efforts may want to ensure they have adequate resources and provide support to staff who collaborate, so they can build these relationships over time.

Successful agency participation in collaboration includes open communication, clear expectations, and realistic information about internal priorities, plans, and timelines. If plans or timelines change, it helps to provide timely and transparent information. Managers can also aid collaborative efforts by notifying participants of future decision points, and how they may use any collaborative input that they receive. Collaborative groups may be part of the solution to increasing trust and social license for forest management to meet NWFP and other goals.

Collaboration is not the answer for every situation. Not every stakeholder for a management unit will be eager to participate in an organized collaborative group or process. If the alternatives to collaboration are expected to be better than participation, some stakeholders may opt out. In some cases, the outcome from an agency appeal or court decision process may be preferred over collaboration. Moreover, the scope, scale, and end goals of a given collaborative effort may make it more relevant and accessible to some stakeholders than others. Research about the effectiveness of collaborative groups for achieving social and ecological goals is still underway. Until we have definitive results, we do not really know yet whether collaboration is the ultimate answer.

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Inner City Youth Institute, High School Natural Resources Camp, Multnomah Falls, Oregon.
Photo by USDA Forest Service.

Chapter 10—Environmental Justice, Low-Income and Minority Populations, and Forest Management in the Northwest Forest Plan Area

Susan Charnley, Delilah Jaworski, Heidi Huber-Stearns, Eric M. White, Elisabeth Grinspoon, Rebecca J. McLain, and Lee Cerveny[1]

Introduction

This chapter synthesizes literature about the relation between federal forest management and low-income and minority populations, as defined by Executive Order (E.O.) 12898 (February 16, 1994)—“Federal Actions to Address Environmental Justice in Minority Populations and Low-Income Populations” (Clinton 1994). The order requires federal land managers to identify and address any disproportionately high and adverse human health and environmental effects of agency programs, policies, and actions on minority and low-income populations. In this chapter, we use the term “environmental justice populations” to refer to populations protected by E.O. 12898 in matters of environmental justice (defined below). The U.S. Department of Agriculture (USDA) Forest Service and U.S. Department of the Interior Bureau of Land Management (BLM) primarily address environmental justice in their land and resource management planning processes. For example, the Forest Service 2012 planning rule[2] requires responsible officials to “encourage participation by youth, low-income, and minority populations” (p. 21167) throughout all stages of the planning process, and, under the National Environmental Policy Act (NEPA) process, preparation of an environmental impact statement that includes impacts on low-income and minority populations.

Northwest Forest Plan (NWFP, or Plan) socioeconomic monitoring has not explicitly monitored low-income or minority populations other than American Indian tribes. Moreover, since 2006, NWFP socioeconomic monitoring has focused on status and trends in socioeconomic well-being in the Plan area, and has not examined how these trends might be linked to the NWFP. Thus, we are unable to specify how the NWFP has affected low-income and minority populations. However, federal land managers in the Plan area submitted several priority management questions pertaining to environmental justice and forest management for consideration in this science synthesis report. These serve as the guiding questions for this chapter. Chapter 8 discusses the economic impacts of the plan in Plan-area communities. American Indian tribes are the subject of chapter 11; this chapter focuses on other minority populations.

In the absence of monitoring data, we rely mainly on existing scholarly research studies. Existing environmental justice-related forestry research focuses mainly on urban issues; for example, the distribution of urban tree cover in relation to the social and economic characteristics of people living in different city neighborhoods (e.g., Schwarz et al. 2015). Nevertheless, some studies address how environmental justice populations use and value federal forests. Although none has directly investigated how the NWFP has affected minority populations, some include information about how federal forest management may affect them more broadly. A tendency to think about environmental justice as an urban issue challenges federal forest managers to consider how their actions may affect environmental justice populations in rural settings.

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[2] U.S. Department of Agriculture, Forest Service, 36 CFR Part 219, National Forest System Land Management Planning, Section 219.5. Federal Register 77(68): 21162–21276. April 9, 2012, Rules and Regulations.

Defining Environmental Justice

Most of the following section on defining environmental justice—including the references to other documents—is excerpted from Grinspoon et al. (2014: 3–8), a guidance document for Forest Service staff to help them comply with E.O. 12898 during the NEPA process. NEPA requires the agency to consider the potential social and economic effects of its proposed actions. There is no one universally agreed-upon definition of environmental justice in the scholarly literature; the Forest Service defines environmental justice in accordance with USDA departmental regulations (USDA 1997). Environmental justice includes the fair treatment and meaningful involvement of all people regardless of race, color, national origin, or income with respect to the development, implementation, and enforcement of environmental laws, regulations, and policies (USEPA 2013). An environmental justice population is a group of people that meets the criteria for low-income or minority status under E.O. 12898. An environmental justice population may be low income and/or minority.

Defining Minority Population

During the 1980s and 1990s, the U.S. Census Bureau (USDC CB 1999) enumerated population in four racial categories (White, Black, American Indian or Alaska Native, Asian or Pacific Islander), and two categories of ethnicity (Hispanic and non-Hispanic). Adopting the Census Bureau's categories, USDA regulations define a minority as "a person who is a member of the following population groups: American Indian or Alaska Native; Asian or Pacific Islander; Black, not of Hispanic origin; or Hispanic" (USDA 1997: 2). Following guidelines for federal data regarding race and ethnic categories issued by the Office of Management and Budget in 1997, the Census Bureau revised its racial categories for the 2000 and 2010 censuses (White; Black or African American; American Indian and Alaska Native; Asian; Native Hawaiian and other Pacific Islander; some other race; and two or more races). It also revised its ethnicity categories for the 2000 and 2010 censuses from Hispanic and non-Hispanic, to Hispanic or Latino and Not Hispanic or Latino. USDA regulations have not been updated to reflect these more recent Census Bureau categories; however, environmental justice guidance documents continue to treat all populations other than non-Hispanic or non-Latino Whites as minorities. Note that there are some White people who are non-Hispanic or non-Latino who may be considered to be minorities based on other national origins (e.g., people of Middle Eastern origin) who are excluded by this USDA definition. For purposes of this chapter, we adopt the terminology for minority populations used by the Census Bureau at the time of the study or data cited; or by the terminology used by the study we cite if it is different from the Census Bureau categories (to accurately represent research findings).

In its direction on environmental justice in NEPA, the Council on Environmental Quality (CEQ) defines a minority population as:

1. A readily identifiable group of people living in geographic proximity with a population that is 50 percent minority or greater. The population may be made up of one minority or a number of different minority groups; together the sum is 50 percent or more; or,
2. A minority population may be an identifiable group that has a meaningfully greater minority population than the adjacent geographic areas, or may also be a geographically dispersed/transient set of individuals such as migrant workers or Native Americans (CEQ 1997).

Defining Low-Income Population

According to the CEQ, a low-income population is a community or a group of individuals living in geographic proximity to one another, or a set of individuals such as migrant workers or American Indians, who meet the standards for low income and experience common conditions of environmental exposure or effect (CEQ 1997). USDA departmental regulations (USDA 1997: 2) state that low-income populations in an affected area should be identified by the annual statistical poverty thresholds from the Census Bureau's annual current population reports (Series P-60) on income and poverty. The official poverty measure was developed in the 1960s. The Census Bureau (USDC CB 2013) defines low-income populations by the percentage of people living below poverty in a given area, which is consistent with CEQ's environmental justice guidance. Low-income status is determined by comparing annual income to a set of dollar values called poverty thresholds that differ by family size, number of children, and age of householder. If a family's before-tax monetary income is less than the dollar value of their threshold, then that family and every individual in it are considered to be living in poverty. For people not living in families, poverty status is determined by comparing the individual's income to his or her poverty threshold.

For tables showing Department of Health and Human Services guidelines for poverty, see the Federal Register notice (USDHHS 2013).[3] For more information, see also "How poverty is calculated in the ACS [American Community Survey]" (USDC CB 2013). In 2013, the poverty guideline for the 48 contiguous states and the District of Columbia was $11,490 for a one-person household and $23,550 for a four-person household. The Census Bureau updates the poverty thresholds annually using the Consumer Price Index.

Guiding Questions

Regional federal land managers wished to know whether environmental justice populations in the NWFP area have been growing, and to understand the implications of trends in the size of these populations for federal forest management. Thus, this chapter addresses the following questions pertaining to environmental justice, low income and minority populations, and federal forest management:

1. What are the trends in the size of low-income and minority populations in the NWFP area since the Plan was adopted, and what is their current distribution?
2. How do low-income and minority populations interact with federal forests in the NWFP area?

We address the implications of these trends and interactions for forest management in the "Conclusions and Management Considerations" section of this chapter.

Key Findings

Trends in Low-Income and Minority Population Sizes and Current Distribution

The size and percentage of environmental justice populations in the Plan area have increased since the NWFP was adopted, consistently with national trends. This increase has occurred both in the size of low-income populations (measured here by number of people living below the poverty line), and the number of people belonging to minority groups specified by E.O. 12898. These trends are detailed below. We use 1990 as our baseline because of the availability of decennial U.S. Census data from 1990. For current status, we use U.S. Census data from 2012, consistent with the 20-year NWFP socioeconomic monitoring report (Grinspoon et al. 2016). The census data provide the best available information on low-income and minority populations across the Plan area. Although some low-income and minority populations may be missed by census takers, such as transient workers or undocumented immigrants, no other datasets are currently available that capture these populations for the Plan area as a whole in a statistically significant manner.

There are 72 counties—32 metropolitan, and 40 nonmetropolitan—in the Plan area (appendix). The population size data presented below are for the Plan area as a whole, and for metropolitan versus nonmetropolitan counties (in aggregate). There is no evidence to suggest that trends in the size and percentage of environmental justice populations since the NWFP was adopted are in any way linked to the Plan.

Low-income populations—

The poverty rate in the NWFP area as a whole increased from 11.2 to 14.7 percent of the region's population between 1990 and 2012 (table 10-1). Nevertheless, the poverty rate was lower overall than the national poverty rate during the three periods reported here. Although poverty rates fell in many subregions of the Plan area between 1990 and 2000, those improvements were more than offset by increases in poverty across the Plan area between 2000 and 2012. Poverty rates were uniformly higher in nonmetropolitan counties than in metropolitan counties during the analysis period, and they were also higher than the national average (which includes both metropolitan and nonmetropolitan

Table 10-1—County-level poverty rates in the Northwest Forest Plan (NWFP) area, 1990, 2000, and 2012

	1990 poverty rate	2000 poverty rate	2012 poverty rate
		Percent	
United States	13.5	11.3	15.0
All NWFP-area counties	11.2	10.0	14.7
All metropolitan counties	10.3	9.1	13.9
All nonmetropolitan counties	15.3	14.2	19.0
All California NWFP-area counties	11.4	11.1	15.4
All California metropolitan counties	9.6	9.0	13.1
All California nonmetropolitan counties	15.6	16.4	21.7
All Oregon NWFP-area counties	12.2	10.4	16.9
All Oregon metropolitan counties	11.4	9.7	16.4
All Oregon nonmetropolitan counties	15.2	13.2	19.2
All Washington NWFP-area counties	10.5	9.4	13.2
All Washington metropolitan counties	9.9	8.8	12.8
All Washington nonmetropolitan counties	15.1	13.3	16.7

Source: U.S. Census Bureau small-area income and poverty estimates.

counties) (table 10-1). Overall, poverty rates were highest in Oregon and lowest in Washington in both 1990 and 2012. However, in California, nonmetropolitan counties had the highest poverty rates within the Plan area during the period. These counties also experienced the biggest increase in poverty—rising from 15.6 percent in 1990 to 21.7 percent in 2012, with no dip in 2000, unlike the other subregions (table 10-1). Figure 10-1 shows poverty rates in the NWFP area by county in 2012. The highest poverty rates were concentrated in northern California and southern Oregon. Counties with the lowest poverty rates are concentrated around the greater San Francisco, Portland, and Seattle metropolitan areas.

Minority populations—

The percentage of the population identifying as a racial or ethnic minority grew in both metropolitan and nonmetropolitan counties within the Plan area between 1990 and 2012 (table 10-2). Most notably, the percentage of the population identifying as Hispanic or Latino doubled in nonmetropolitan counties, and nearly tripled in metropolitan counties in the Plan area. The percentage of the White population declined more in metropolitan counties than in nonmetropolitan counties. Plan-area counties with high concentrations of minority residents were clustered near California's Central Valley and east of the Cascade Range crest in Washington (fig. 10-2). This finding may be explained by evidence that about half of farm laborers and their supervisors in the United States are Hispanic or Latino (USDA ERS 2012), and these are areas of high agricultural activity.

American Indian and Alaska Native populations were higher in the NWFP area than in the nation as a whole (table 10-3). They were more prevalent in nonmetropolitan counties than in metropolitan counties of the Plan area throughout the period (table 10-2). In 2012, they accounted for a higher percentage of the population in nonmetropolitan counties in California (in aggregate) than in other subregions (tables 10-4 to 10-6; fig. 10-3). In Oregon and Washington, counties with high percentages of American Indian and Alaska Native populations reflect the presence of tribal reservation lands (e.g., the Warm Springs Indian Reservation in Oregon and the Colville Indian Reservation in Washington). In contrast, Black or African American, and Asian, Native Hawaiian, and other Pacific Islander populations formed a higher percentage of the population in metropolitan than in nonmetropolitan counties (table 10-2), and the highest percentage population for both was in metropolitan counties in Washington (tables 10-4 to 10-6). At the individual county level, Black or African American, and Asian, Native Hawaiian, and other Pacific Islander populations are concentrated around Seattle, Portland, and

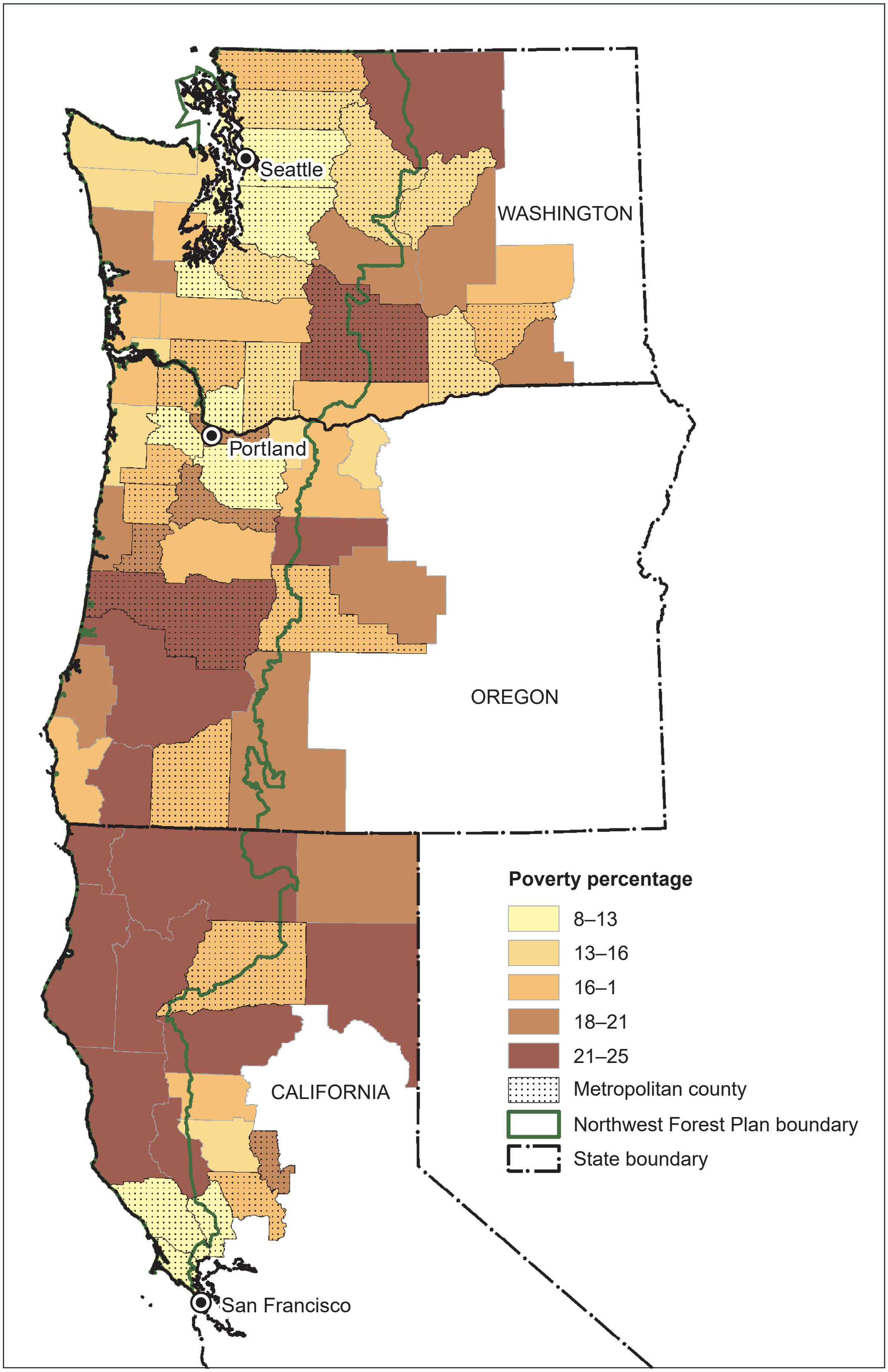

Figure 10-1—Percentage of people living in poverty in Northwest Forest Plan-area counties, 2012.

Table 10-2—Minority populations in the Northwest Forest Plan area, 1990, 2000, and 2012

	1990			2000			2012		
	Plan area	Nonmetro-politan	Metropol-itan	Plan area	Nonmetro-politan	Metropol-itan	Plan area	Nonmetro-politan	Metropol-itan
	Percent								
American Indian and Alaska Native	2	3	1	2	3	1	2	3	2
Asian, Native Hawaiian, other Pacific Islander[a]	4	1	4	5	1	6	7	2	9
Black or African American	3	1	3	3	1	3	3	1	4
White	92	95	92	88	93	86	84	90	82
Hispanic or Latino[b]	5	6	5	9	7	8	14	12	14
≥ two races[c]				3	2	3	4	4	4

Source: U.S. Census Bureau population estimates.
[a] The 1990 Census grouped Asians and Pacific Islanders into one category. In 2000, this category was divided into two: Asian, and Native Hawaiian and other Pacific Islander. For consistency across years, we have grouped these two back into one category.
[b] Hispanic or Latino is a category of ethnicity. Individuals may identify as Hispanic or Latino and any of the racial categories (e.g., Hispanic or Latino and White, Hispanic or Latino and Black). Therefore, table totals will not sum to 100 percent.
[c] This category was not available on the 1990 census form.

Table 10-3—Minority populations in the United States, 1990, 2000, and 2012

	1990	2000	2012
	Percent		
American Indian and Alaska Native	1	1	1
Asian, Native Hawaiian, other Pacific Islander[a]	3	4	5
Black or African American	12	13	13
White	84	81	78
Hispanic or Latino[b]	9	13	17
≥ two races[c]		1	2

Source: U.S. Census Bureau population estimates.
[a] The 1990 Census grouped Asians and Pacific Islanders into one category. In 2000, this category was divided into two: Asian, and Native Hawaiian and other Pacific Islander. For consistency across years, we have grouped these two back into one category.
[b] Hispanic or Latino is a category of ethnicity. Individuals may identify as Hispanic or Latino and any of the racial categories (e.g., Hispanic or Latino and White, Hispanic or Latino and Black). Therefore, table totals will not sum to 100 percent.
[c] This category was not available on the 1990 census form.

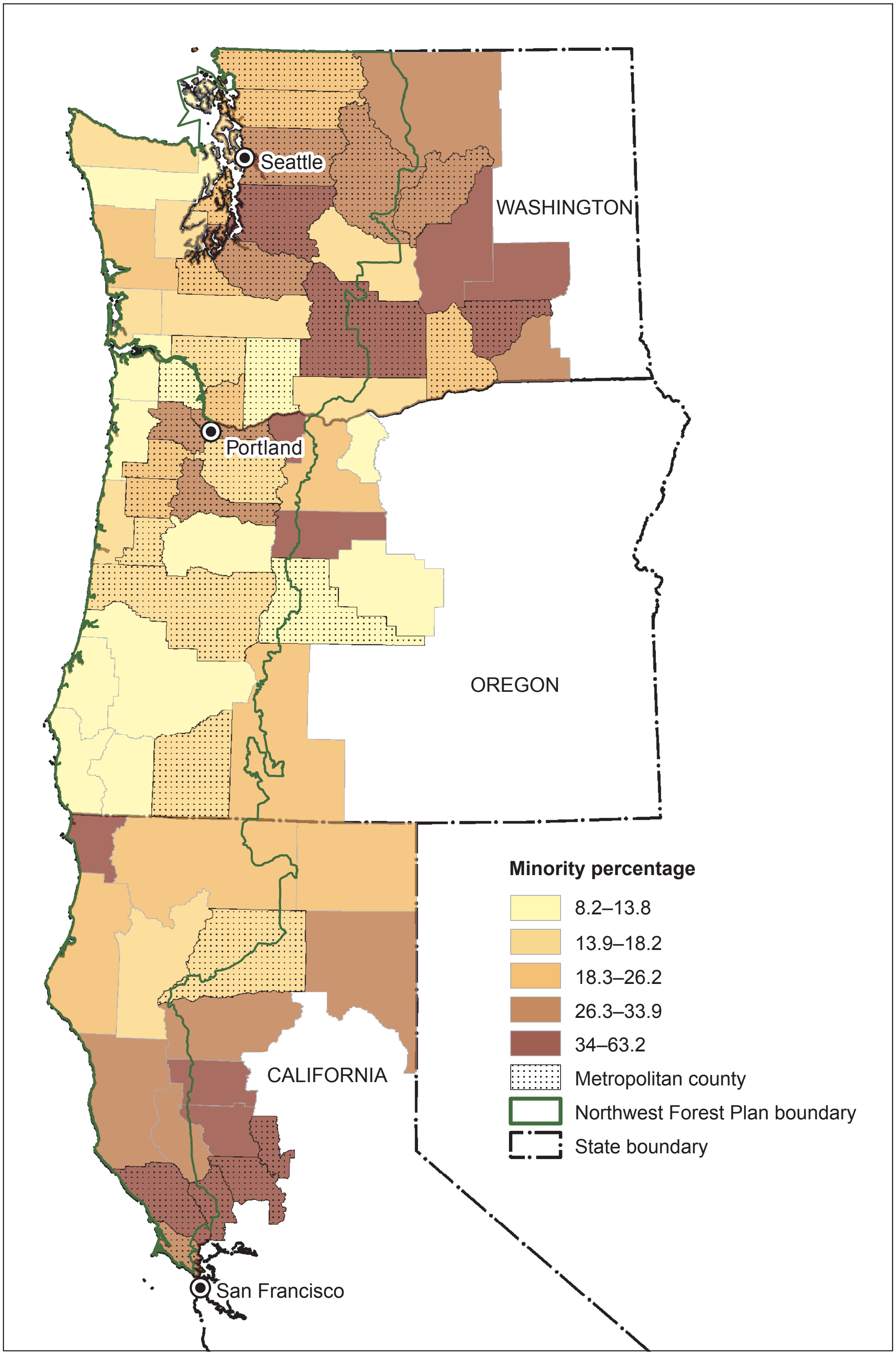

Figure 10-2—Minority percentage of populations (combined) in Northwest Forest Plan-area counties, 2012.

Table 10-4—Minority populations in California's Northwest Forest Plan area, 1990, 2000, and 2012

California	1990 Nonmetropolitan	1990 Metropolitan	2000 Nonmetropolitan	2000 Metropolitan	2012 Nonmetropolitan	2012 Metropolitan
	Percent					
American Indian and Alaska Native	4	1	4	1	5	2
Asian, Native Hawaiian, other Pacific Islander[a]	1	4	1	5	2	7
Black or African American	1	2	2	2	2	2
White	94	93	90	89	87	85
Hispanic or Latino[b]	9	11	14	17	19	23
≥ 2 races[c]			3	2	4	4

Source: U.S. Census Bureau population estimates.
[a]The 1990 Census grouped Asians and Pacific Islanders into one category. In 2000, this category was divided into two: Asian, and Native Hawaiian and other Pacific Islander. For consistency across years, we have grouped these two back into one category.
[b] Hispanic or Latino is a category of ethnicity. Individuals may identify as Hispanic or Latino and any of the racial categories (e.g., Hispanic or Latino and White, Hispanic or Latino and Black). Therefore, table totals will not sum to 100 percent.
[c]This category was not available on the 1990 census form.

Table 10-5—Minority populations in Oregon's Northwest Forest Plan area, 1990, 2000, and 2012

Oregon	1990 Nonmetropolitan	1990 Metropolitan	2000 Nonmetropolitan	2000 Metropolitan	2012 Nonmetropolitan	2012 Metropolitan
	Percent					
American Indian and Alaska Native	2	1	2	1	2	1
Asian, Native Hawaiian, other Pacific Islander[a]	1	3	1	1	2	6
Black or African American	<0.5	2	1	3	1	3
White	97	94	94	90	92	86
Hispanic or Latino[b]	3	4	6	9	9	15
≥ 2 races[c]			2	2	3	4

Source: U.S. Census Bureau population estimates.
[a] The 1990 Census grouped Asians and Pacific Islanders into one category. In 2000, this category was divided into two: Asian, and Native Hawaiian and other Pacific Islander. For consistency across years, we have grouped these two back into one category.
[b] Hispanic or Latino is a category of ethnicity. Individuals may identify as Hispanic or Latino and any of the racial categories (e.g., Hispanic or Latino and White, Hispanic or Latino and Black). Therefore, table totals will not sum to 100 percent.
[c] This category was not available on the 1990 census form.

Table 10-6—Minority populations in Washington's Northwest Forest Plan area, 1990, 2000, and 2012

	1990		2000		2012	
Washington	Non-metropolitan	Metropolitan	Non-metropolitan	Metropolitan	Non-metropolitan	Metropolitan
	Percent					
American Indian and Alaska Native	3	2	3	1	3	2
Asian, Native Hawaiian, other Pacific Islander[a]	2	5	2	8	2	10
Black or African American	1	4	1	4	1	5
White	94	89	92	84	90	79
Hispanic or Latino[b]	6	5	7	7	11	12
≥ 2 races[c]			2	3	3	5

Source: U.S. Census Bureau population estimates.
[a] The 1990 Census grouped Asians and Pacific Islanders into one category. In 2000, this category was divided into two: Asian, and Native Hawaiian and other Pacific Islander. For consistency across years, we have grouped these two back into one category.
[b] Hispanic or Latino is a category of ethnicity. Individuals may identify as Hispanic or Latino and any of the racial categories (e.g., Hispanic or Latino and White, Hispanic or Latino and Black). Therefore, table totals will not sum to 100 percent.
[c] This category was not available on the 1990 census form.

the San Francisco Bay area (figs. 10-4 and 10-5). The high percentage of the population that was Black or African American in northeastern California is attributable to the demographic composition of the prison population that resides in Lassen County. The percentage of the population identifying as Hispanic or Latino was high relative to other minority groups in the Plan area as a whole, and was similar between metropolitan and nonmetropolitan counties (table 10-2). The percentage of the population that was Hispanic or Latino was highest in California counties (in aggregate) (tables 10-4 to 10-6). Hispanic or Latino populations were highest in NWFP counties of eastern Washington and California's Central Valley, where farming is an important economic sector.

The percentage of American Indian and Alaska Native, and Black or African American populations did not increase between 1990 and 2012, while the percentage of Asian, Native Hawaiian, and Pacific Islander populations grew, and the percentage of Hispanic or Latino populations grew substantially (table 10-2). The NWFP area had a higher percentage of the total population that was American Indian or Alaska Native, and Asian, Native Hawaiian, or other Pacific Islander compared with the nation as a whole in 2012, but a substantially lower percentage of the total population that was Black or African American, or Hispanic or Latino, compared with the nation as a whole (tables 10-2 and 10-3).

Many poor counties in the Plan area also have large shares of minority residents (fig. 10-7). However, poverty is not limited to those areas having high concentrations of minorities. For example, Josephine, Douglas, and Lane counties in Oregon and Trinity County in California have some of the highest rates of poverty in the Plan area (all exceed 20 percent), yet their residents are predominantly White who are not of Hispanic/Latino origin. Similarly, low-poverty counties in the greater San Francisco, Portland, and Seattle metropolitan areas have relatively high concentrations of minorities. The coarseness of county-level data used for NWFP socioeconomic monitoring over the past decade, and the data presented here, prevent finer scale comparisons (e.g., community-level) of minority status, poverty, and the relationship between them. Examining how trends in minority group populations and poverty rates may be linked is beyond the scope of this chapter. Nevertheless, at the national level, Black/African American, American Indian, and Hispanic or Latino populations in the United States experience significantly higher rates of poverty than White and Asian populations (Macartney et al. 2013).

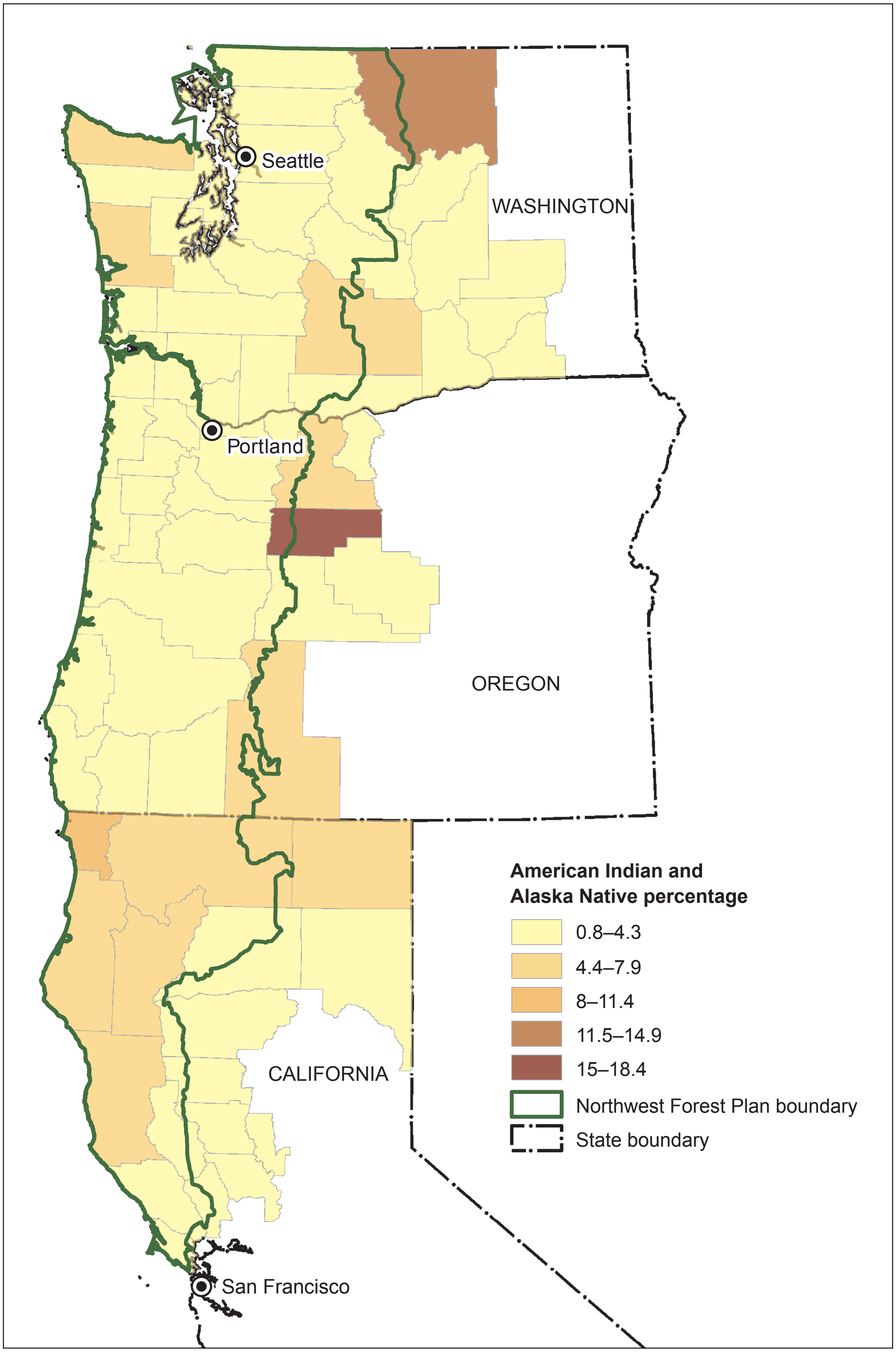

Figure 10-3—American Indian and Alaska Native percentage of populations in Northwest Forest Plan-area counties, 2012.

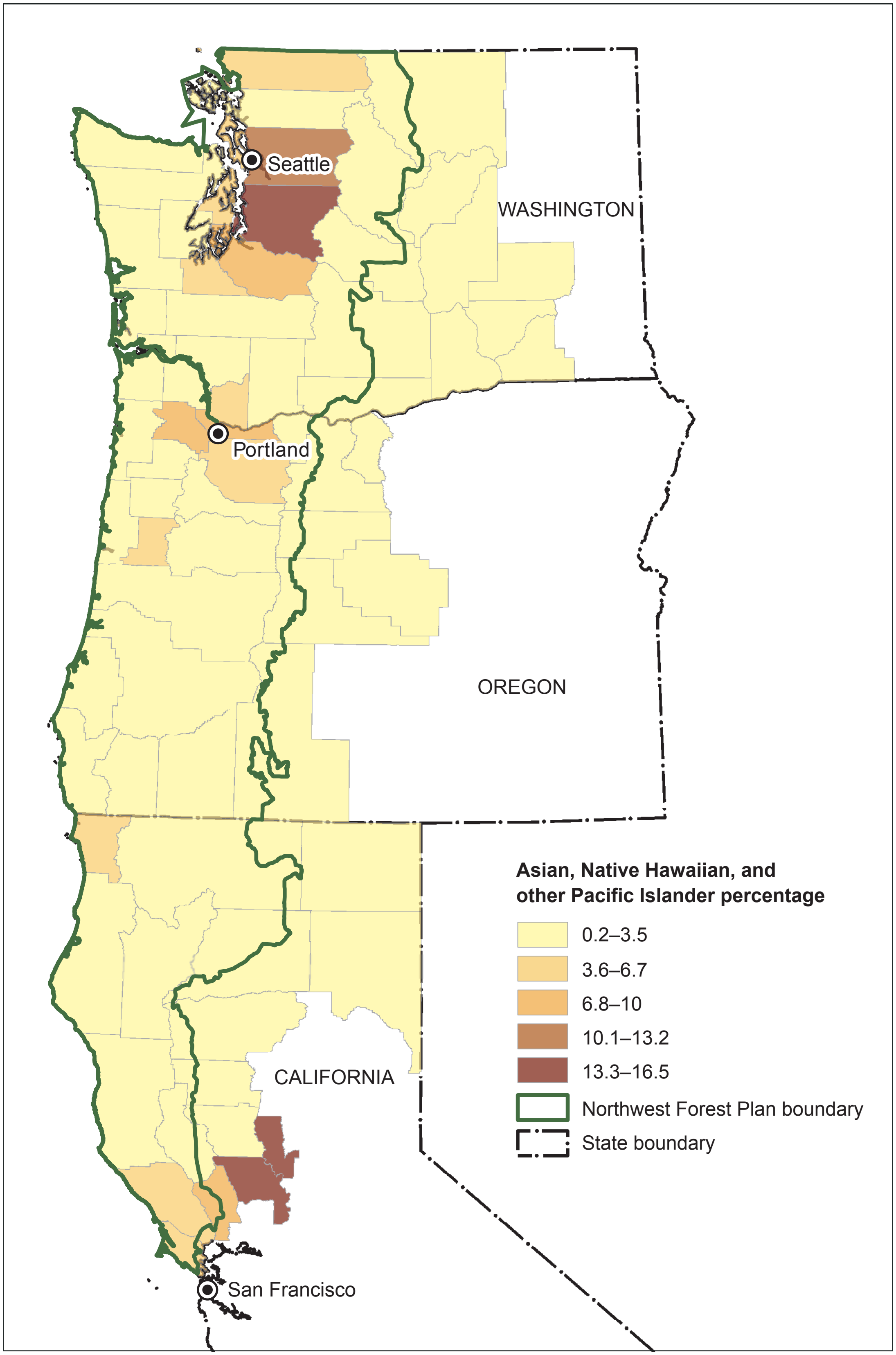

Figure 10-4—Asian, Native Hawaiian, and other Pacific Islander percentage of populations in Northwest Forest Plan-area counties, 2012.

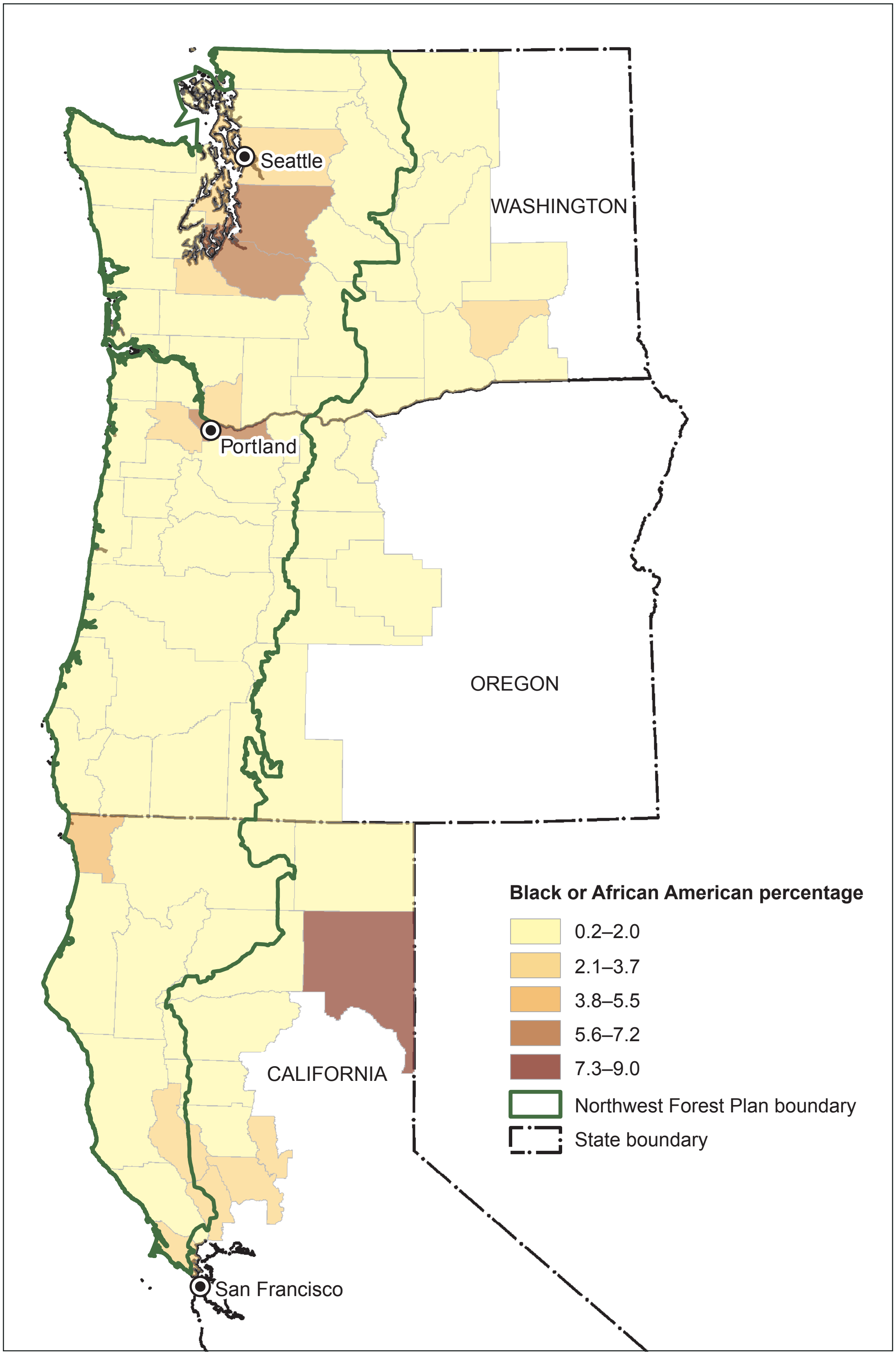

Figure 10-5—Black or African American percentage of populations in Northwest Forest Plan-area counties, 2012.

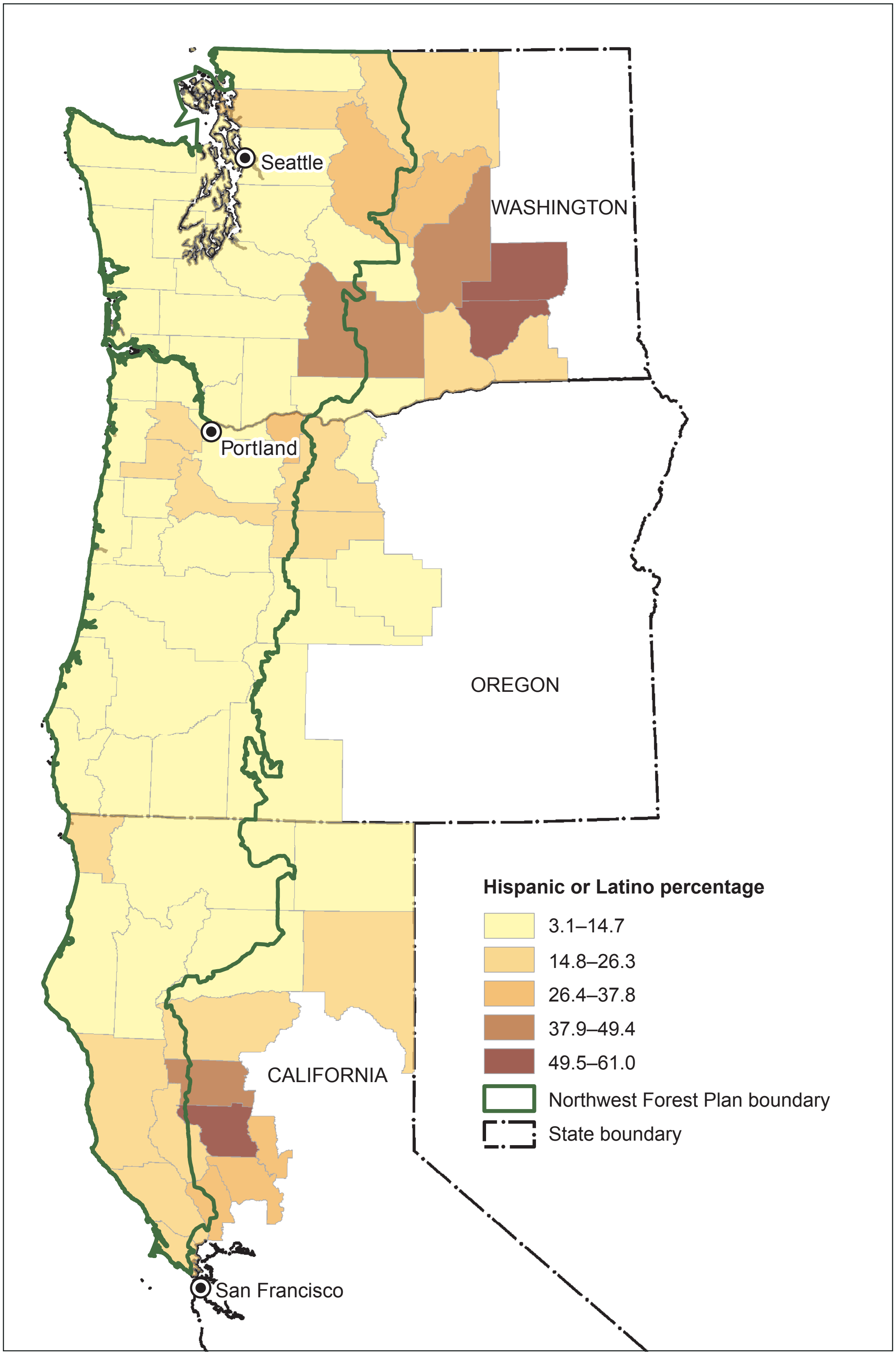

Figure 10-6—Hispanic or Latino Percentage of populations in Northwest Forest Plan-area counties, 2012.

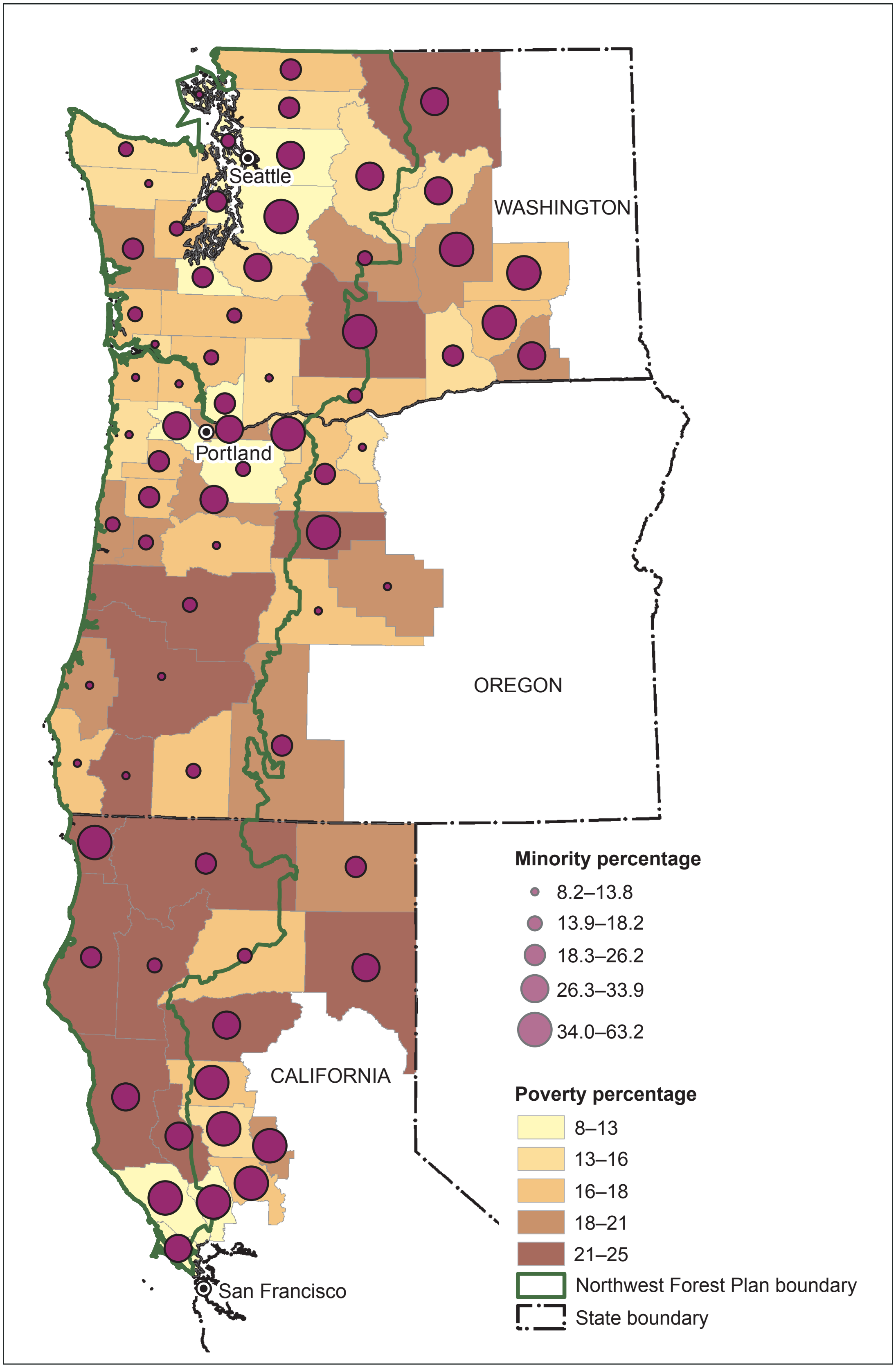

Figure 10-7—Minority populations by Northwest Forest Plan-area county poverty rates, 2012.

How Low-Income and Minority Populations Interact With Federal Forests in the Plan Area

The demographic composition of the NWFP area is changing: the percentage of the total population comprised of minority groups, especially Hispanic or Latino, is increasing, as are poverty rates. Research indicates that to some degree, different populations maintain different relations to federal forests, have different use preferences, and face different constraints that influence their use of federal forests, though variation within groups exists (as it does among all demographic groups). They may also have different views of the environment and resource management, and different environmental behaviors and values (see chapter 9). To comply with E.O. 12898 and to encourage use of federal forests by environmental justice populations, it is important to understand these differences and ways of overcoming constraints. The scientific literature from the Pacific Northwest is limited in this arena, and focuses mainly on participation by low-income and minority populations in the environmental work force, the gathering of nontimber forest products, and recreation. We address these topics below, and also note some emergent issues: the presence of temporary residents—including homeless populations—on national forests, many of whom are likely low income; and connections between wildfire management and environmental justice.

The environmental workforce—

Forest workers are employed by contracting businesses to conduct forest restoration and related work, often on federal lands. Forest workers in the Pacific Northwest perform a variety of labor-intensive tasks, such as planting and thinning trees, piling and burning brush, manual herbicide application, and digging firelines to fight wildland fires (Moseley 2006) (fig. 10-8). These tasks, which typically


Susan Charnley

Figure 10-8—Forest workers cut and pile brush on the Six Rivers National Forest in California.

require shoveling, planting, hand cutting, and similar manual labor, are more labor-intensive than other work performed by forest workers that entails less manual labor, such as operating heavy machinery and timber cruising. In 2014, there were an estimated 6,400 forest workers in Oregon alone during peak season (Wilmsen et al. 2015). This section focuses on the working conditions of workers hired by forestry and fire contracting businesses. To date there have been few systematic studies conducted on forest workers (Wilmsen et al. 2015). Most of the studies we draw on took place in western Oregon; information is much more limited for the California and Washington parts of the Plan area.

Since the late 1970s, forest workers in the Pacific Northwest have been predominantly Hispanic or Latino, marking a shift away from what was previously a White, not Hispanic or Latino, workforce toward one that is now primarily composed of low-income Hispanic or Latino immigrants and undocumented workers (Casanova and McDaniel 2005; Moseley et al. 2014; Sarathy 2006, 2012). Although the available scientific literature does not provide current statistics on the proportions of Hispanic or Latino workers now in the Northwest's forestry workforce, research by Sarathy (2008) found that, in the mid-2000s, Mexican immigrants constituted the largest proportion of immigrant forest workers on federal lands in the Pacific Northwest. Moseley (2006) found that at a time when the U.S. Census put Oregon's Hispanic or Latino population at 8 percent of the state total, a random sample of contractors interviewed from two national forests in Oregon during the high season were 45 to 60 percent Hispanic or Latino. A 2006 estimate found that between 2.2 and 3.1 million of the unauthorized immigrants residing in the United States were active in the labor force, particularly the lower skilled labor force (e.g., agri-food, crop production, forestry, and food services) (Sarathy and Casanova 2008).

What has been termed the "Latinization" of forestry work originated in a confluence of public policy (e.g., Small Business Administration set-asides for minority-owned businesses) and social networks (i.e., recent immigrants enter the sector because of relationships with earlier immigrants who work in forestry services) (Sarathy 2006). In contrast, minorities are underrepresented in the white-collar environmental workforce that offers higher job quality (e.g., less manual labor, more consistent oversight of safety); one possible explanation is their low participation in university environmental programs (Weintraub et al. 2011). In Oregon, the majority of ecological restoration businesses are small, family-owned seasonal businesses that fit the Small Business Administration's definition of a small business (Nielsen-Pincus and Moseley 2013). Research about the proportion of forestry and fire contracting businesses that are minority owned is lacking.

With this change in workforce composition came a series of working condition concerns that disproportionately affected immigrant forest workers, especially those without legal immigration status (Sarathy 2008). These forest workers are often Hispanic or Latino, and have been referred to as "pineros" by the U.S. media (a Spanish word meaning "man of the pines" or "someone who works in the woods") to describe the ethnicity of the workers and the type of work that they do (see Knudson and Amezuca 2005, Sarathy 2012). Job quality among forest workers is typically low, measured by lack of employment stability, low wages, no benefits, and distance from home. Although all such workers face low job quality, Hispanic or Latino workers are more likely to work far from home and seasonally, and less likely to receive health insurance through their employers (Moseley 2006). For example, Moseley (2006) found a statistically significant correlation between the ethnic composition of a company's workforce and the type of work performed. During high season, Hispanics or Latinos comprised 43 percent of the total workforce and 66 percent of the labor-intensive workforce. By contrast, those who were not Hispanics or Latinos accounted for 53 percent of the total workforce but 73 percent of the equipment-intensive workers. Labor-intensive workers often work seasonally and travel long distances (Moseley and Reyes 2008). Although there is limited research on equipment-intensive contractors, available data suggest that these contractors do not typically travel as far for work, and are less exposed to exploitative working conditions, compared to labor-intensive workers (Moseley and Reyes 2008). These findings suggest that there are job quality differences between Hispanics or Latinos, and non-Hispanics or Latinos.

Research on forest workers from Oregon also shows that Hispanics or Latinos often face poorer working conditions than their counterparts who are not Hispanic or Latino, including disrespectful treatment, uncompensated injuries, little opportunity for advancement, and retribution if they complain (Sarathy 2012). High injury and fatality rates; low, unpaid, or stolen wages; lack of training; decline of union protection; and dangerous work environments also characterize this sector (Campe et al. 2011, Moseley et al. 2014, Sarathy 2012, Wilmsen et al. 2015). Similar poor working conditions exist for immigrants who are agricultural workers. For example, low-wage immigrant workers in labor-intensive agricultural occupations often experience wage theft, unsafe working conditions, inadequate safety training, and fear of retaliation for reporting injuries or unfair working conditions (Wilmsen et al. 2015). Hispanic or Latino immigrants often face a particular disadvantage owing to language barriers and limited access to legal resources; fear of deportation makes it less likely that forest workers will report labor exploitation or dangerous working conditions (Campe et al. 2011; Sarathy 2008, 2012; Sarathy and Casanova 2008). Many of these concerns have been hidden from elected officials, the general public, and decisionmaking bodies, with scholarship, media, and public policy focusing disproportionately on the concerns of White, native-born loggers (Sarathy 2008).

Poor working conditions for forest workers have persisted over the past two decades. This problem is particularly prominent in the Pacific Northwest; for example, forest workers in Oregon were found to have rates of occupational injury, illness, and fatality three times higher than the workforce at large (Hayford 2013, Wilmsen et al. 2015). Moreover, documented rates are thought to be low estimates owing to historical underreporting of such problems by workers and employers alike (Azaroff et al. 2002, Ruser 2008, Sarathy 2012, Wilmsen et al. 2015). Two deaths of forestry services workers in on-the-job accidents in southern Oregon in 2011 were a reminder to the public and others of the dangers found in this sector (Wilmsen et al. 2015). Increased media attention on working conditions for forest workers has led to more Congressional oversight and labor law enforcement, but this political attention has been inconsistent as other issues arise (Moseley et al. 2014). Some groups representing forest workers such as the Northwest Worker Justice Project and the Northwest Forest Worker Center (formerly the Alliance of Forest Workers and Harvesters) advocate for better federal labor and contracting law enforcement to improve working conditions. But as Moseley et al. (2014) explained, the persistence of poor working conditions despite decades of political attention and advocacy suggests that changing labor laws alone will be insufficient for improving job quality. The vulnerability of immigrant workers, federal land management policy, and federal contracting regulations can also affect working conditions, and deserve attention (Moseley and Reyes 2008).

The debate regarding how to address poor working conditions, punctuated by political controversy and advocacy, is as of yet unresolved. Recent research from southern Oregon (Wilmsen et al. 2015)—a region having a high proportion of forest workers who are mainly Spanish-speaking Hispanic or Latino immigrants—still reported workplace practices that are inconsistent with labor laws. Workers' vulnerable economic status, lack of legal status, and fear of retaliation remain some of the largest drivers of marginalization for the increasing immigrant labor force in the Pacific Northwest (Campe et al. 2011, Moseley et al. 2014, Sarathy 2008, Wilmsen et al. 2015). This situation can cause the most marginal and vulnerable groups to shoulder a disproportionate level of risk and find ways to navigate the system invisibly (Moseley et al. 2014, Wilmsen et al. 2015).

Although there have been some job quality improvements for Hispanics or Latinos in recent years, these have mainly occurred in the arena of fire suppression work, including compensation for travel and training (Moseley et al. 2014). There is limited research on the impacts of federal contracting on businesses that engage in wildfire suppression and their employees, many of whom are forest workers (Caldwell et al. 2005, Lyon et al. 2017). Nevertheless, fire suppression work is historically more profitable and less price competitive than federal forestry work, in which contractors are pressured to cut costs to get contracts (Moseley et al. 2014). Additionally, firefighter safety and preparedness have become a high priority for federal land management agencies, and a culture of firefighter safety has been integrated into the incident command structure in which contractors operate (Moseley et al. 2014). Contract firefighters also work closely with federal, state, and local government employees on fire incidents, making it difficult to hide workplace safety issues. In contrast, working conditions for other forest workers have received inconsistent attention; labor and safety law enforcement is dispersed across a variety of state and federal labor and land management agencies, and workplace safety issues are less visible (Moseley et al. 2014).

Traditionally, most workplace health and safety strategies have focused on improving the physical safety of the workplace, which is particularly relevant for improving the safety of working in the woods (e.g., hard hats, correct equipment and gear). However, the broader well-being of workers is also important. Research suggests that, once basic physical safety conditions for forest workers are addressed, there should be more explicit consideration of employee well-being to improve retention, morale, and staff stability (Mylek and Schirmer 2015).

Research is lacking regarding the proportion of minority women who are forest workers. However, there has been increasing attention on issues of gender in the environmental workforce. To date, this attention has been more prevalent in the popular press than in the scientific literature, but the topic warrants attention in considering workforce conditions. A recent Washington Post article recounted women firefighters' experiences with harassment, discrimination, and sexual violence (Fears 2016), which was followed by Congressional oversight hearings. Although the hearings focused on federal employees, similar problems are experienced by those contracted by the federal government (Moseley et al. 2014, Sifuentez 2016). A 2016 Association for Fire Ecology survey found that 32 percent of firefighters have witnessed sexual harassment, and 54 percent have witnessed gender discrimination in the workplace (Association for Fire Ecology 2016). Similar to Hispanic or Latino environmental workers, women may be especially vulnerable to workplace safety and culture issues, an area that warrants future research attention.

In summary, forest workers in the NWFP area are predominantly Hispanic or Latino. They work as contractors who perform a variety of labor- and equipment-intensive forestry work on federal forests in addition to participating in fire suppression crews. Much of the published literature about forest workers draws attention to the low job quality and poor working conditions they have experienced over the past few decades. Low job quality includes low wages, lack of stable employment, no benefits, and long travel distances to work sites. Poor working conditions experienced by forest workers include disrespectful treatment, little opportunity to advance, unsafe working environments and high rates of injury and fatality, lack of training opportunities, and fear of retaliation or deportation if they complain. Although there have been some improvements in recent years, especially in the area of fire suppression work, debates over how to address these poor working conditions remain unresolved. To date, federal agencies have not notably changed their oversight of service contract crews or enforcement of labor law provisions (Moseley et al. 2014, Sarathy 2012, Wilmsen et al. 2015).

Nontimber forest products gathering—
The gathering of nontimber forest products (NTFPs) in the Pacific Northwest for subsistence, commercial, recreational, and cultural purposes is important and widespread, both in urban and rural areas (Alexander et al. 2001; Alexander and Fight 2003; Charnley et al. 2007; Jones and Lynch 2007; Love et al. 1998; Lynch and McLain 2003; McLain et al. 2012; Poe et al. 2013, 2014). National forests and BLM land are important sites for commercial NTFP harvesting (Charnley et al. 2008). Most commercial NTFP harvesting in the Pacific Northwest occurs in temperate forests from the Cascade Range crest west to the Pacific coast, owing to high concentrations of economically important species, more people, and infrastructure that makes access easier (Charnley et al. 2008). Chapter 8 provides an overview of NTFP gathering in the Plan area, including common species harvested. Our focus here is on commercial gathering owing to the scarcity of studies specific to environmental justice populations' participation in recreational gathering and subsistence gathering (apart from American Indians, see chapter 11) in the Plan area. One of the few studies that includes a substantive discussion of recreational harvesters found that the majority (83 percent) of the recreational

chanterelle mushroom (*Cantharellus* spp.) harvesters interviewed on the Olympic Peninsula were Euro-Americans, with the next most common ethnic group represented being Japanese Americans (6 percent) (Love et al. 1998). In that study, none of the Latinos or Southeast Asians categorized their harvesting activities as recreational. As elaborated in chapter 8, the distinction between work and leisure is blurred for many commercial NTFP harvesters.

Low-income and minority populations are often active in harvesting NTFPs for commercial purposes, although subsistence and cultural uses are also important. For instance, on Washington's Olympic Peninsula—a focal point for the Northwest's floral greens industry—the harvester workforce was originally Euro-American, but shifted in the late 1970s and early 1980s to being dominated by refugees from Southeast Asia,, then shifted again in the late 1980s to become dominated by immigrants from Mexico and Central America (McLain and Lynch 2010). Asians are also active participants in commercial wild mushroom harvesting, particularly matsutake (*Tricholoma magnivelare*) (Tsing 2015). Commercial NTFP harvesting for some people may be their primary source of income, but for most it fills gaps or provides supplemental income between other seasonal jobs such as agricultural or forestry services work, or jobs in cities (Love et al. 1998; McLain 2000, 2008; Tsing 2015).

A survey from the early 1990s—which provides the only regional-level data available—found that roughly half of the commercial mushroom harvesters in the Northwest were White, followed by 37 percent Asians and Pacific Islanders, and 9 percent American Indians (Schlosser and Blatner 1995). An ethnographic study of the Olympic Peninsula chanterelle harvest (Love et al. 1998) documented the presence of four major groups of pickers—Cambodian (and other Southeast Asian), White, Latino, and Native American—during 1994 and 1995, but did not provide percentages for each category. An analysis of wild mushroom permit data for 1996–1998 from the Sisters Ranger District on the Deschutes National Forest, which falls within the eastern margins of the NWFP area and is a popular morel (*Morchella esculenta*) and bolete (*Boletus edulis*) harvesting site during the spring, estimated that 62 percent of permit holders were White, 28 percent Southeast Asian, and 10 percent Latino (McLain 2000). The only study identified that examined the intersectionality between gender and ethnicity for NTFP harvesters found that, among commercial chanterelle harvesters on the Olympic Peninsula, women comprised roughly 30 percent of Euro-American pickers but few Latino and Southeast Asian pickers were women (Love et al. 1998). There are no more recent studies providing statistics on NTFP harvester sociodemographic characteristics, whether at the local or regional scale.

Beargrass (*Xerophyllum tenax*) is one example of an NTFP harvested from federal forests located within the Plan area. Commercial harvesting of beargrass for its flowers and leaves gained importance in the Pacific Northwest in the 1980s (Higgins et al. 2004, Lynch and McLain 2003), and it has since become one of the leading commercial NTFP species harvested in the region, and a multimillion dollar industry (Charnley and Hummel 2011). Most commercial beargrass harvesters in the Pacific Northwest are Southeast Asian and Latino immigrants (Hansis 1998). Despite the physical hardships, these groups may be drawn to gathering beargrass and other NTFPs because it is work that does not require English language skills; jobs in the forest may be more appealing than low-paying jobs in cities; the job can be performed by and with families; payment is in cash; and it may provide cultural continuity to gathering traditions from immigrants' home countries (Charnley and Hummel 2011, Hansis 1998).

Wild mushrooms are another example; matsutake, the most economically valuable mushroom in the world (Tsing 2015), is a case in point. Four distinct populations harvest matsutake in the Pacific Northwest. Japanese-Americans have been harvesting the mushroom in the region for a century and pick them as part of their cultural heritage; Oregon's Mount Hood area is a favorite spot (Tsing 2013a). These are largely recreational pickers who distribute mushrooms among their relatives and across the Japanese-American community, which reinforces social relations and their heritage. Matsutake gained commercial value for the export trade to Japan in the 1980s. At that time, a second group started picking it, White men, such as workers who had lost jobs in the timber industry and other rural residents. These pickers have since been largely displaced by a wave of Southeast Asian refugees to the United States who entered the woods in the thousands beginning in the late 1980s: the Khmer from Cambodia, and the Lao, Hmong, and Mien from Laos (Richards and Creasy 1996; Tsing 2013a, 2013b). Many of these pickers migrate to the Pacific Northwest seasonally from cities in California to harvest mushrooms between other seasonal or temporary jobs (Tsing 2015). Despite associated dangers such as the presence of hunters or the possibility of getting lost, mushroom harvesting offers these pickers, who often are poor, a sense of freedom and the ability to earn money as long as they have a permit, transport, and camping equipment (Tsing 2013b). Latino pickers, originating primarily from Mexico and Guatemala, also participate in the commercial matsutake harvest in central Oregon. Many are undocumented and thus are in a more precarious legal position than Southeast Asian refugees, most of whom either have permanent residency or U.S. citizenship (Tsing 2013c). Many Latino pickers use the matsutake harvest as a way to fill in gaps in the demand for work in the agricultural and horticultural sectors (Tsing 2013c). Tsing (2015) described the matsutake industry and the pickers who are part of it in detail (fig. 10-9).

Salal (*Gaultheria shallon*), a major commercial product in the floral greens industry, is a third example of an NTFP

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Figure 10-9—Southeast Asians play a prominent role in the matsutake industry.

harvested from federal forests in the Plan area. Most salal harvesters are undocumented migrant workers from Mexico and elsewhere in Latin America, and Southeast Asian immigrants (Ballard and Huntsinger 2006, McLain and Lynch 2010). Research about these harvesters finds that many have detailed local ecological knowledge related to stand conditions, canopy cover, soil conditions, and disturbances that affect salal (Ballard and Huntsinger 2006).

Other researchers have also found that NTFP harvesters may possess substantial local ecological knowledge about the species they harvest, though this varies with experience (Charnley et al. 2007, Love et al. 1998, McLain 2000, Tsing 2013a). These findings indicate the potential capacity of NTFP harvesters to contribute to sustainable forest management. However, environmental justice populations who engage in NTFP harvesting, and NTFP harvesters more broadly—regardless of ethnic or racial identity—have been underrepresented in the process of developing management guidelines and regulations for NTFPs (Charnley et al. 2007, Jones and Lynch 2007, McLain 2000, McLain 2002, McLain and Jones 2001, McLain and Lynch 2010). A variety of factors likely contributes toward NTFP harvester underrepresentation, including limited knowledge of English on the part of some harvesters, commercial harvesters' unfamiliarity with land management agency public input processes, and ineffective outreach on the part of federal and state land management agencies (Ballard and Sarathy 2008, McLain 2002, McLain and Lynch 2010).

Land tenure, and the formal and informal rules governing harvester access to commercially viable harvesting sites, further condition environmental justice populations' interactions with forests in the NWFP area (Charnley et al. 2007, Love et al. 1998, McLain 2000, McLain and Lynch 2010, Tsing 2015). Harvesters are highly dependent on public or large tracts of private lands for gathering, making them subject to access and use regulations imposed by landowners who typically grant access through the issuance of short-term permits or longer term leases (Ballard and Huntsinger 2006, McLain and Lynch 2010, Tsing 2015). Research on wild mushroom policies in central Oregon suggests that failure to incorporate or consider harvester input has sometimes resulted in the development of regulations, such as prohibitions on harvesting very small-sized matsutake (known as "babies") and fixed harvesting season starting and ending dates, that fit poorly with ecological conditions (McLain 2002, Tsing 2015). Moreover, other land uses (e.g., timber harvest, grazing) and management actions (e.g., fire suppression) have an impact on the productivity and diversity of NTFP species. Thus, NTFP harvesters have a strong interest and stake in federal forest management.

Little research has focused specifically on assessing the impacts of restrictions emanating from the Plan on NTFP harvesters, whether members of environmental justice populations or not. An exception is McLain's (2000, 2002, 2008) research on central Oregon's wild mushroom harvest, which documented how restrictions on the commercial harvest of NTFPs in late-successional reserves and the closure of thousands of miles of forest roads significantly reduced areas open to commercial wild mushroom harvest on national forests in that area. As discussed in chapter 8, the extent to which NTFP harvesters rely on late-successional forest ecosystems for products will vary, depending on the requirements of the species gathered. No studies about NTFP harvesting on lands administered by the BLM were identified in our literature search.

As commercial harvesting of NTFPs increases in response to market demand, tensions between commercial gatherers and gatherers primarily interested in recreational, subsistence, and cultural uses have emerged in some areas where there is competition over harvesting the same species (Charnley and Hummel 2011, Dobkins et al. 2016, Jones and Lynch 2007, Tsing 2015). For example, beargrass is highly valued for the floral greens industry, but it is also a culturally important plant to American Indian tribes in the NWFP area, especially for basketry (Charnley and Hummel 2011) (see chapter 11). Leaf properties desirable for commercial versus cultural purposes differ, as does forest stand management to promote the desired properties (detailed in Charnley and Hummel 2011). These competing interests and management requirements can cause conflict among users; some tribal members have expressed concern over the impact of commercial beargrass harvesting on the plant (Charnley and Hummel 2011). Tension also exists among participants within specific NTFP sectors, such as within the floral greens industry. For example, on the Olympic Peninsula, tensions have arisen among floral green harvesters when some participants follow harvest regulations and others do not (McLain and Lynch 2010). Moreover, some environmental groups do not support any gathering activities that they perceive as threatening forest health, even if only for subsistence use (Salazar 2009). A generalized lack of inventory and monitoring data collected in ways that would enable the impacts of harvesting on NTFP species to be evaluated makes it difficult to develop effective management guidelines (Jones and Lynch 2007).

For their part, harvesters have expressed a number of concerns related to NTFP gathering and management. For example, Latino harvesters from the Olympic Peninsula who participated in a natural resource values mapping exercise that included national forest lands stated that their main concerns were: the presence of hunters and target shooters who they perceived as acting irresponsibly in places where they harvest, making them feel unsafe; challenges associated with harvesters who do not comply with harvest regulations; and encounters with immigration and law enforcement officers looking for undocumented workers (Biedenweg et al. 2014). Racial profiling by Forest Service law enforcement officers is another concern expressed by floral greens harvesters on the Olympic Peninsula (Biedenweg et al. 2014) and the Gifford Pinchot National Forest (Northwest Forest Worker Center 2015), and by matsutake harvesters in central Oregon (Tsing 2015). Additional studies are needed to determine whether these concerns apply more generally across the Plan region, to harvesters of other NTFPs, or to groups other than Latino harvesters.

Other concerns revolve around the intersection between labor relations and land tenure. In the wild mushroom sector throughout the Plan area, most pickers, regardless of ethnicity, operate as independent or family-based entrepreneurs and gain access to harvesting sites through relatively affordable permits (McLain and Lynch 2010, Tsing 2015). They thus have some measure of independence from the firms to which they sell their mushrooms. Conditions for many pickers in the floral greens industry on the Olympic Peninsula are much less favorable. In that setting, most floral greens harvesters, most of whom are Latino, gain access to harvest sites through people who operate buying sheds. Buying sheds are the buildings where the greens are purchased from harvesters, sorted, quantified, and boxed for shipping to wholesale distributors and exporters. Shed owners on the Olympic Peninsula often obtain leases to large tracts of public or private forest where harvesting occurs, then give permission for harvesters to pick on those lands, often specifying informally (and illegally) that the harvesters must sell their product to them (Lynch and McLain 2003, McLain and Lynch 2010). One consequence of the pickers' economic position under such circumstances is that they are unable to take advantage of higher prices paid by competing sheds (McLain and Lynch 2010). Harvesters and small-scale buyers have expressed opposition to leases, which large-scale buyers have historically monopolized and which appear to facilitate the exploitation of harvesters by limiting their resource access options (McLain and Lynch 2010, Northwest Worker Center 2015).

Harvesters have also identified theft of floral greens from leased lands as a problem (McLain and Lynch 2010, Northwest Forest Worker Center 2015). In response to complaints during the early 2000s by shed operators who did not have leases, the Washington Department of Labor and Industries sought, unsuccessfully, to have floral greens harvesters who gained access to harvesting sites through sub-leasing arrangements categorized as employees rather than independent contractors. The debate over whether harvesters acquiring access to floral greens through sub-leases should be considered shed employees, rather than independent entrepreneurs, has implications for their rights as workers, their working conditions, and whether they receive fair prices for their products (McLain and Lynch 2010).

To summarize, environmental justice populations in the NWFP area—particularly Southeast Asians, Latinos, and low-income Whites—play an active role in the commercial NTFP industry, with Latinos especially prominent in the floral greens industry and Asians and Whites prominent in the wild mushroom industry. National forests and BLM lands are important harvesting sites, but there has been virtually no published research documenting the impact of the NWFP on environmental justice populations who harvest NTFPs there. Although these populations are affected by agency regulations associated with NTFP harvesting and management practices influencing the distribution and productivity of the species they target, they have been underrepresented in developing regulations and management guidelines for NTFPs on federal forests in the Plan area. Important issues for managers to be aware of include potential social tension between commercial gatherers and those primarily interested in recreational, subsistence, and cultural gathering; tenure arrangements governing access to NTPFs; physical safety of harvesters when they are out in the forest; fear of encounters between undocumented workers and immigration and law enforcement officers; challenges associated with illegal harvest activities (e.g., theft); and the rights to safe working conditions and fair employment practices for harvesters.

Recreation—

Research about recreational uses of federal forests in the NWFP area by environmental justice populations comes from national surveys and scholarly research. The NWFP socioeconomic monitoring reports (Charnley 2006, Grinspoon et al. 2016) contain data on recreation visitation in the NWFP area by national forest and BLM district, but these reports do not display recreation visitation data by income, racial, or ethnic group. In this section, we present recreation participation data for Plan-area national forests in aggregate by income, and minority group from the Forest Service National Visitor Use Monitoring Program. Comparable data are unavailable for BLM districts. We also briefly synthesize key areas of knowledge from the literature about outdoor recreation participation by environmental justice populations in the region and nationwide, and constraints to participation. Some of this literature is specific to Forest Service lands, but none is specific to BLM lands. See chapter 9 for a broader discussion of recreation in the NWFP area.

Low-income populations—
More than half of recreation visits to NWFP-area national forests are made by people whose household incomes are less than $75,000 per year (table 10-7). Households with incomes under $25,000 per year account for about 12 percent of all recreation visits in the NWFP area, slightly higher than what is found nationally. The only income group with a lower participation rate is households having incomes of more than $150,000 per year (table 10-7).

Research from the Pacific Northwest about recreational use of public lands among low-income populations focuses on income levels and cost as determinants of participation. Using a random sample of 2,005 adult Washington and Oregon residents, Burns and Graefe (2006) found lower interest and participation in outdoor recreation among those with the lowest personal incomes. One-quarter of those surveyed whose personal incomes were less than $10,000/year reported that they were "not at all" interested in outdoor recreation; and 13 percent of those with personal incomes between $10,000 and $30,000 reported the same low interest level. In contrast, only about 5 percent of respondents with incomes greater than $30,000 reported no interest in outdoor recreation. The vast majority (between 86 and 92 percent) of those making more than $30,000 per year had participated in an outdoor recreation activity during the preceding year, while about 56 percent of those making less than $10,000 had participated (Burns and Graefe 2006). On average, those earning less than $10,000 per year visited national forests about 2.6 times per year compared to about 8.5 times per year for other income groups (Burns and Graefe 2006). This pattern of visit frequency may be due, at least in part, to the ability of people with higher incomes to afford the cost of recreation trips to national forests (Ostergren et al. 2005).

Regardless of urban or rural residency, the cost of recreation on federal forests includes equipment and gear expenses, transportation costs to reach the recreation site, and in some places, recreation fees. Of these expenses, federal land managers have influence only over recreation fees. The Forest Service's Recreation Fee Demonstration program, initiated in the late 1990s, established recreation fees at many dispersed areas on national forests that previously had no site fees. Brown et al. (2008) examined permit data from 1991 through 2005 and found that recreation fees to park and access a wilderness area on Oregon's Willamette National Forest had a greater negative effect on recreation visitation than did high-severity fire within the wilderness area. In their previously referenced survey from Washington and Oregon, Burns and Graefe (2006) found that the lowest income respondents in their study (earning less than $10,000 per year) were the most likely to indicate that they could not afford to pay a hypothetical recreation-use fee on national forest lands (although more than half of the respondents in this income category indicated they could pay a hypothetical recreation use fee).

Table 10-7— Visits to national forests in the Northwest Forest Plan area and nationally of people age 16 and older by household income

Annual household income	Plan area 2006–2010	Plan area 2011–2015	National 2011–2015
		Percent	
Less than $25,000	10	12	10
$25,000–$49,000	24	18	18
$50,000–$74,999	25	23	22
$75,000–$99,999	18	20	18
$100,000–$149,999	15	17	16
$150,000 and up	8	11	16

Source: USDA FS 2016.

Minority populations—

Nearly all recreational visits to NWFP-area national forests are by White visitors (table 10-8). People of Hispanic or Latino ethnicity account for more recreation visits to NWFP-area national forests (4 percent) than people belonging to other minority groups. Across all national forests, the vast majority of visits are also from White visitors, and about 6 percent of visits nationally are from those of Hispanic or Latino ethnicity, again exceeding visits by other minority groups (table 10-8).

Data from the 2008 National Survey on Recreation and the Environment (NSRE) (Cordell 2012) indicate that, nationwide, American Indians have activity participation patterns that are similar to Whites, although American Indians have higher rates of participation in backcountry activities (like primitive camping, backpacking, visiting wilderness), and lower rates of nonmotorized winter recreation participation than Whites (table 10-9). Asian, Native Hawaiian, and other Pacific Islanders, like most other groups, have high rates of participation in activities at developed sites. A much higher percentage participate in viewing and photographing nature than in backcountry activities, hunting and fishing, motorized recreation (e.g., off-highway vehicles, motorized trail bikes, use of motorized play areas), and nonmotorized activities. Hispanic or Latino populations surveyed participate more in some activities than other minority groups, and less in others, but the relative popularity of different activities is generally similar between Hispanics or Latinos and other groups.

Table 10-8—Visits to national forests in the Northwest Forest Plan area and nationally of people age 16 and older by racial and ethnic group

	Plan area 2006–2010	Plan area 2011–2015	National (2011–2015)
	Percent		
American Indian and Alaska Native	3	3	2
Asian	2	3	2
Black or African American	1	1	1
White	96	95	95
Native Hawaiian and other Pacific Islander	1	1	1
Hispanic or Latino	4	4	6

Source: USDA FS 2017.

Table 10-9—Nationwide percentage of participation in outdoor recreation activities of people age 16 and older by racial and ethnic group

Activity	American Indian	Asian, Native Hawaiian, Pacific Islander	Black or African American	White	Hispanic or Latino
	Percent				
Visiting developed sites	84	82	69	80	75
Viewing and photographing nature	79	73	59	78	71
Backcountry activities	60	34	21	46	43
Motorized activities	42	24	15	41	35
Hunting and fishing	38	19	21	38	32
Nonmotorized winter activities	7	11	4	13	12
Nonmotorized water activities	21	21	7	24	19

Source: Adapted from White et al. (2014) and Cordell (2012).

Visiting developed sites, and viewing and photographing nature, were the most common activities. Adult Hispanic or Latino day visitors interviewed at urban national forests in southern California reported that they most often participated in picnicking and water recreation when visiting day-use sites (Chavez and Olson 2009). Blacks or African Americans have the lowest levels of participation in outdoor recreation relative to the other groups surveyed. However, more than half of respondents had visited developed sites and participated in viewing and photographing nature (table 10-9).

Floyd et al. (2008) provide a comprehensive review of research related to race/ethnicity and leisure, including the many factors that affect recreation participation by minority groups. One national-level study using the 2004 NSRE data found that, relative to other groups, ethnic minorities, older people, women, and those living in rural places perceived higher constraints to participating in outdoor recreation (Ghimire et al. 2014). The primary perceived barriers were lack of time or money, concerns about personal safety, lack of transportation, and lack of multilingual signage. Facility condition, perceived crowding, and environmental quality were infrequently seen as barriers to outdoor recreation by these groups. Distance and cost to access recreation opportunities on federal lands are key factors influencing outdoor recreation use (Cho et al. 2014, Stevens et al. 2014) (fig. 10-10). For example, Bowker et al. (2006), in a national

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Figure 10-10—Distance to primitive settings and the cost of recreation, including equipment expenses, are constraints to outdoor recreation participation by low-income and minority populations.

study using NSRE data, found evidence that a central factor in lower participation by minority group members age 16 and older is distance to primitive settings. Johnson et al. (2007), also using NSRE data, found that minority groups were less likely than other groups to support a fee for using a recreation site.

Chavez (2008) pointed out the importance of understanding the distinct preferences, expectations, and barriers to participation in outdoor recreation among Hispanic or Latino populations in the United States. As table 10-2 shows, the share of the Hispanic or Latino population is large and growing in metropolitan counties of the NWFP area (see also Johnson and Stewart 2007), making consideration of Hispanic or Latino preferences and barriers to access especially important for the management of urban national forests (e.g., the Mount Baker–Snoqualmie National Forest in Washington and the Mount Hood National Forest in Oregon). Some studies have found that Hispanics or Latinos tend to participate in outdoor recreation activities with extended family members in natural areas located close to urban centers (Burns et al. 2008, Chavez 2008). Constraints to participation include distance to recreation areas, lack of transportation, and lack of information (in Spanish and English) about where to recreate and who to contact to learn about recreation opportunities (Burns et al. 2008).

Burns et al. (2008) conducted four focus groups (small groups of select people who discuss questions pertaining to specific research topics) with adults belonging to different minority groups in several Oregon cities. Potential participants were identified through recreation managers (mostly from the Oregon Parks and Recreation Department) who worked in the communities where the focus groups were held. They found that Asian Americans in their study liked to recreate with their children and extended family, and preferred developed facilities having amenities over camping. Safety concerns loomed large, however, especially the safety of children. African Americans in the study disliked recreating in remote locations, preferring parks close to urban areas having well-managed, clean facilities, aesthetically pleasing views, and amenities such as picnic tables, places to barbeque, and areas to play sports. Both groups identified lack of information about opportunities to recreate in parks and on public lands, information in multiple languages as an additional constraint (fig. 10-11). Metcalf et al. (2013) surveyed 234 racial and ethnic minority groups visiting the Mount Baker–Snoqualmie National Forest about their perceived constraints to outdoor recreation participation, and strategies they use to overcome these constraints. The chief factors constraining recreation on national forests among these users were preferences for other recreation activities, limited time and other obligations, and weather conditions. Very few survey respondents reported that discrimination from other recreation users or Forest Service employees limited their outdoor recreation participation.

In sum, recreation visitation by environmental justice populations to national forests in the NWFP area is relatively low. Nationwide, different racial and ethnic groups exhibit different preferences for types of outdoor recreation activity, although visiting developed sites and viewing and photographing nature are the most popular activities among all groups, including Whites. A main barrier to recreating on national forests for low-income populations is cost of the trip. Among minorities, distance, cost, lack of transportation, safety concerns, lack of awareness about recreation opportunities, and lack of available information in languages other than English are barriers. Ways of overcoming these barriers are discussed under Management Considerations.

Nonrecreational camping and homelessness—
Camping is a common recreational use of Forest Service and BLM lands in the NWFP area. But many people camp on public lands for nonrecreational purposes, with these lands serving as a temporary residence. Some nonrecreational or long-term campers temporarily reside on public lands as a lifestyle choice or in response to local economic conditions. Others are homeless (with no permanent address).

Accurate estimates of homeless individuals in the United States are difficult to achieve. One study by Abt Associates for the U.S. Department of Housing and Urban Development estimated 549,928 homeless persons in a one-day count in 2016 (Henry et al. 2016). Yet, the National Law Center on Homelessness & Poverty suggests that these figures are grossly underestimated, and places the figure

USDA Forest Service

Figure 10-11—Information in multiple languages may encourage recreation use of national forests by minority populations whose primary language is not English.

between 2.5 to 3.5 million (NLCHP 2015). Some people who are chronically or episodically homeless choose to live on public lands. These represent vulnerable populations, both in terms of economic vulnerability and social vulnerability. Poverty is the primary risk factor for homelessness (Ji 2006). Other economic risk factors include high unemployment, lack of affordable housing, and a female-only head of household. Personal setbacks, such as an accident, divorce, natural disasters, unpaid medical bills, sudden job loss, or loss of a loved one can exacerbate these problems and increase a person's vulnerability (Elliott and Krivo 1991). Social vulnerabilities include lack of access to adequate health care, unmet mental health needs, domestic violence, and divorce (Elliot and Krivo 1991, Wasserman and Clair 2010). Untreated mental health issues such as depression, addiction, post-traumatic stress disorder, and others, can negatively affect personal resiliency and are associated with homelessness.

Federal land management agencies do not have accurate counts of how many people live on federal lands. A recent survey of 290 national forest law enforcement officers revealed that encounters with nonrecreational campers occur in every region of the United States, and that nonrecreational campers were most common in national forests near urban areas (Cerveny and Baur n.d.). For national forests and grasslands in California (Region 5) and Oregon and Washington (Region 6), 41 percent of law enforcement officers surveyed reported weekly encounters with nonrecreational campers, and 85 percent reported encounters at least monthly. These encounter rates were higher than for the nation as a whole (39 percent weekly and 75 percent monthly). In addition, 47 percent of officers in Region 5, and 49 percent in Region 6 reported that encounters with long-term nonrecreational campers had increased in the years since they had begun their current assignment (mirroring the national average of 47 percent).

Cerveny and Baur (n.d.) also identified 10 types of nonrecreational campers who were using the national forest as a residence. The most common type in Regions 5 and 6 were "separatists," who were alone and seeking solitude; "transient retirees" living in RVs and moving from place to place; and "families." The survey asked officers what they perceived as most often contributing to people living in national forests on a long-term basis. The most commonly mentioned factors associated with homelessness and long-term camping were substance abuse, mental health issues, lack of employment, and lack of available housing (Cerveny and Baur n.d.).

An unpublished master's project conducted in Oregon's Willamette National Forest by students from the University of Oregon explored the incidence of homelessness and long-term camping there (Bottorff et. al. 2012). The authors conducted interviews with staff from the national forest, local service agencies, law enforcement, and homeless individuals to gain a better understanding of homelessness. They learned that the homeless people on the Willamette were mostly seasonal, and that lack of services in nearby towns often drives the homeless to nearby forests. In addition, many homeless people were unwilling to stay in available shelters, which prohibited either children or pets. They also observed that homeless people living in the forest often struggled with addiction and mental health problems.

These results echo the risk factors mentioned above and suggest economic and social vulnerabilities. National forests and grasslands are serving as a temporary home for people who are suffering from health challenges or economic hardship. These results confirm a finding from the Deschutes National Forest (Asah et al. 2012) that one ecosystem service not commonly identified is the ability of national forests to serve as a temporary shelter for people who are marginalized by dominant economic, social, and health care systems. The magnitude of temporary residence as a phenomenon and management issue on federal forests, and the degree to which it represents a problem for federal forest managers in the NWFP area, are unknown; research on these topics is only beginning to emerge.

Wildfire management and environmental justice— Wildfire management is one of the many areas in which federal land management actions may affect adjacent and nearby residents and landowners, some of whom may be low income or minorities. Research about the relation between wildfire management and low-income and minority populations living in fire-prone forest ecosystems of the United States is limited; research on this topic from the Pacific Northwest is even more limited. Key findings from the studies that have been conducted include the following:

1. The rural poor living in fire-prone areas in the wildland-urban-interface (WUI) in Arizona's White Mountains, and low-income residents in a community in the Sierra-Cascades foothills of northern California, were found to have fewer resources for creating defensible space around their homes, investing in fire-resistant building materials, purchasing insurance, or adopting other wildfire mitigation strategies than middle- and high-income rural residents (though other variables also influence people's choices to mitigate fire on their properties) (Collins 2005, 2008).
2. In the southeastern United States, communities having high wildfire risk and high social vulnerability (e.g., below poverty line, non-White, low education) are less engaged in wildfire mitigation programs than communities having high wildfire risk and low social vulnerability (Johnson Gaither et al. 2011, Poudyal et al. 2012). Similarly, research from Arizona found that participation in wildfire mitigation programs is lower among socially vulnerable communities located in areas of high wildfire risk, than among communities with low social vulnerability located in high wildfire risk areas (Ojerio et al. 2011).

3. In Washington state, a higher percentage of poor households than non-poor households live in areas having few to no wildfire response resources that provide wildfire protection (Lynn and Gerlitz 2006). In Florida, Mercer and Prestemon (2005) found that the more poverty in a county, the lower the rate of wildfire ignitions but the larger the wildfire (acres burned) when an ignition occurs. They attribute the lower rate of ignitions to the fact that poorer counties have more rural WUI, and federal and state lands dominated by pine forests that are intensively managed for timber production, where prescribed fire is commonly applied. These management actions lower wildfire hazard. However, wildfires burn more acres when an ignition occurs because poorer counties have fewer firefighting resources available for initial attack.
4. Research from Utah (Roberts 2013) and Florida (Mercer and Prestemon 2005) found that people living in higher income WUI locations prefer dense forest stands for aesthetic reasons, increasing wildland fire risk; however, they are less vulnerable to wildfire because they can afford insurance policies and have better access to fire mitigation and suppression resources.
5. In the Northwest and elsewhere in the Western United States, poor households usually outnumber wealthier households near federal lands, but tend to be located in areas having low housing density that do not meet the threshold for WUI delineation (Lynn and Gerlitz 2006, Radeloff et al. 2005). Thus, they receive fewer benefits from fire hazard mitigation activities and suffer longer wildfire response times (Lynn and Gerlitz 2006).
6. Research from the Southern United States (Johnson Gaither et al. 2015) found that smoke plumes from wildfires and prescribed fires did not disproportionately adversely affect socially vulnerable populations (defined using an index of indicators including poverty, minority status, renters, and age- and education-related variables). These populations experienced no more smoke exposure than populations who are not socially vulnerable. Comparable research about the impacts of smoke on environmental justice populations from the Pacific Northwest is lacking.
7. Research about the location of hazardous fuels reduction treatments on national forests in relation to the distribution of nearby environmental justice populations in the Pacific Northwest is currently underway. Initial results from two national forests in central Oregon found no systematic evidence of disproportionate benefit or lack of benefit to environmental justice populations from fuels reduction treatments (Adams and Charnley 2018). However, localized areas of potential concern were identified where further inquiry is warranted.
8. Finally, decades of disaster research by social scientists reveal that the effects of natural hazards, such as wildfire, are not experienced equally within a community. The most socially vulnerable people have the most difficulty coping and recovering from the hazard event and adapting afterward (e.g., Oliver-Smith 1996).

These research findings indicate that wildfire management actions can have differential impacts on people living adjacent to or near federal forests because of differences in social vulnerability to wildfire that may be associated with low-income and minority status. Wildfire management is but one example of how the environmental effects of agency management actions such as timber harvesting and watershed management, and associated changes in ecosystem services, have environmental justice implications.

Research Needs, Uncertainties, Information Gaps, and Limitations

The vast majority of scholarly research on environmental justice has focused on unequal exposure to environmental toxins, largely in urban areas. There is only a small subset of research that focuses on environmental justice in the context of unequal access to environmental benefits, and that work mostly concerns parks, outdoor recreation opportunities, and street trees in urban areas (e.g., Landry and Chakraborty 2009, Montgomery et al. 2015). Furthermore, there is virtually no research or monitoring data that concern the specific impacts of the NWFP on low-income or minority populations apart from American Indians (see chapter 11). If federal forest managers wish to fill this information gap, perhaps environmental justice inquiry could be integrated into NWFP socioeconomic monitoring. However, this would require revising the current monitoring approach to explore links between federal forest management and socioeconomic well-being. This chapter provides information about general trends in environmental justice populations in the NWFP area between 1990 and 2012 using readily available county-level data. More recent, detailed, or geographically specific trends in low-income and minority populations could be identified using U.S. Census data as part of the socioeconomic assessment to support forest plan revisions for NWFP-area national forests.

Most of the research reported here about how environmental justice populations relate to federal forests comes from Washington and Oregon; this literature is more limited for the California portion of the NWFP area, except for American Indian tribes. Literature for BLM lands is also scarce. There is a reasonably substantive literature about how minority populations relate to national forests around work (e.g., forestry services work, commercial NTFP harvesting) and recreation. However, neither the complexity of forestry work impacts on forest worker vulnerability, nor the relationship between changing agency and contracting business employment structures and forest worker vulnerability, are well studied or understood. Also missing in the literature are explorations of how and which environmental justice populations have input into provisions in the NWFP, and associated regulations and management approaches regarding NTFPs. Most of the literature on NTFP harvesting is from the 1990s or early 2000s, and may not reflect current conditions. Little information is available about uses (recreational, subsistence, and cultural) of NTFPs by environmental justice populations apart from American Indians.

More broadly, apart from recreation, little information is available about noneconomic relations between environmental justice populations and federal forests, including cultural and spiritual connections, except for American Indians. This gap could potentially be filled through additional research, including using methods such as focus groups with populations of interest that include participatory values mapping exercises to document how different populations use and value federal forests (e.g., Biedenweg et al. 2014).

Regarding the impacts of forest management activities on environmental justice populations, research is beginning to fill the gap in knowledge about the environmental justice implications of Forest Service hazardous fuels reduction activities. However, there is a lack of information about how fire—managed, prescribed, or wild—and associated smoke affect low-income and minority populations in the Plan area. There is also a research void regarding how other federal forest management activities such as timber harvesting, travel management, and watershed management affect environmental justice populations. Finally, there is a void in the literature about the role of environmental justice populations in forest governance, particularly collaborative decisionmaking processes associated with federal forest management and planning.

It is uncertain whether the research findings presented here are relevant locally, and reflect the nature of interactions between environmental justice populations and federal forests on specific national forest and BLM units. Research pursued at finer scales would help address this uncertainty, as would research to better understand the variation within minority groups regarding their interactions with federal forests in particular places. Another large gap in the literature on environmental justice and forests from the NWFP area and nationwide is how low-income or minority status intersect with subgroup characteristics (i.e., gender, age, religion) to influence forest values, uses, and management impacts. The only related research we are aware of in this area is a handful of recreation studies conducted in urban and rural parks (e.g., Casper et al. 2013, Cronan et al. 2008, Larson et al. 2014, Perry et al. 2011). Growth in environmental justice populations throughout the NWFP area calls for reassessing earlier findings, and ongoing research into how these populations relate to federal forests and are affected by their management in order to address the information gaps and limitations of existing research identified in this chapter.

Conclusions and Management Considerations

Environmental justice populations in the NWFP area are growing. Census data reflect the changing demographics of the region, and research from within and outside the Plan area provides insight into how some members of low-income and minority populations interact with federal forests. When thinking about these relationships, it is important to avoid overgeneralizing and creating stereotypes about the values, uses, preferences, and behaviors of specific groups. Inevitably, there will be variation within groups, some of it influenced by gender, age, length of time in the United States, and other factors. The research synthesized here can be used to increase awareness and flag potentially relevant topics for agency staff to examine more closely at the local level. It also raises a number of issues that are relevant to federal forest management.

Management Considerations

The environmental workforce—

As the demographic composition of the NWFP area continues to change, and the forestry workforce is increasingly represented by Hispanics or Latinos and other environmental justice populations, it is important that federal forest managers address the issue of working conditions for forest workers. Doing so means considering contracting markets and contract oversight, which include bidding on, awarding, and monitoring compliance for projects. Based on the literature synthesized in this chapter, the following actions might help improve working conditions for forest workers.

1. The Forest Service and BLM already stipulate in service contracts that contractors must comply with all relevant labor laws. These agencies have the authority to enforce the provisions of their own contracts, which includes the labor law provisions, and could do so more rigorously.
2. Agencies could examine how the beneficial features of fire-suppression contracting could be incorporated into other, non-fire contracts (e.g., specific contract requirements and more oversight).
3. Agencies could strengthen policies to make labor law compliance inspection more consistent, combining these inspections with technical specification inspections, and increasing agency inspector training (Wilmsen et al. 2015).
4. The competitive low-cost bid process could be changed to reduce contractor incentives for cutting costs and explicitly incorporate the costs of safety trainings and daily safety briefings into contract awards (Moseley et al. 2014, Wilmsen et al. 2015).

Other considerations that emerge from the literature pertain to increasing the ability of forest communities to capture contracting opportunities on nearby federal forests, which would contribute to local economies. For example, agencies might structure contracts in ways that allow local communities to benefit by facilitating local training opportunities, or changing contracting guidelines. They might also consider using local restoration contracting service providers for fire suppression to support local forest contracting capacity, and the ability of local contractors to capture contracts during wildfires. Agencies could also identify how to address potential obstacles, such as wildfire contracting policies, that inhibit local contractors' participation. Having a trained local workforce with the capacity to respond to wildfire rapidly and perform forest restoration work could help increase community preparedness for wildfire.

NTFP harvesting—

Despite the long history and continued prevalence of NTFP gathering in the Pacific Northwest, federal forest managers have been slow to meaningfully consider NTFPs in management (Jones and Lynch 2007). Ballard and Huntsinger (2006), Biedenweg et al. (2014), Charnley et al. (2007), Jones and Lynch (2007), McLain (2008), McLain and Jones (2001), and McLain and Lynch (2010) offered numerous insights into how to address issues associated with NTFP gathering and management on public forest lands in the Pacific Northwest, and how to better engage harvesters in management and decisionmaking associated with NTFPs in the region. Many of these are relevant to all harvesters, regardless of race, ethnicity, or class (see chapter 8). Those pertaining specifically to issues raised by environmental justice populations, include addressing harvesters' safety concerns associated with NTFP gathering (for example, encouraging harvesters to wear blaze-orange vests during hunting season), and examining how policies, including tenure arrangements for NTFP harvesting on federal forests, affect the working conditions and earnings of harvesters. Consideration of how federal forest management activities affect the abundance, distribution, diversity, and quality of economically and culturally important NTFP species also warrants more attention in the planning process.

Recreation—

The growing ethnic and racial diversity of the American population, reflected in NWFP area statistics reported in tables 10-4 through 10-6, has important implications for recreational uses of federal forests because recreation patterns are shaped by cultural norms and preferences (Sheffield 2012). Minority and low-income populations are currently underrepresented among national forest visitors nationwide (Roberts et al. 2009). To ensure that all populations can enjoy federal forests, and to broaden the base of support for public lands, finding ways to increase recreation use by environmental justice populations is important. However, it is also important to recognize the diversity in values within individual ethnic and racial minority groups and to not view these groups as homogenous (Li et al. 2007). The management considerations discussed here focus on how to foster more recreation participation by environmental justice populations on federal forests in the Plan area.

Constraints to recreation participation by these populations that are important to address include a lack of information about available recreation opportunities; improving transportation options to urban national forests; and a shortage of recreation opportunities that match these users' preferences (Metcalf et al. 2013). For example, Spanish-language materials, developed recreation sites that accommodate large groups, and outreach to Hispanic or Latino communities related to volunteer and employment opportunities could strengthen the relationship between federal forests and Hispanic or Latino populations (Chavez 2008). Burns et al. (2008) make a number of suggestions for improving outreach to Latinos, Asian Americans, and African Americans to increase their recreation participation on national forests. Key among these are increasing information about available opportunities in multiple languages, and working with media outlets that target these populations in doing so. Improving facilities so that they accommodate user preferences is also important. For groups concerned about safety, safety concerns could be addressed by increasing the visibility of law enforcement officers and access to agency and emergency personnel (Ghimire et al. 2014). However, increasing the presence of law enforcement may create an environment in which some racial and ethnic minority groups feel threatened. Increasing the presence of Forest Service or BLM employees in uniform on federal forests could also be helpful.

Several strategies to help alleviate cost barriers to recreation participation on national forests by low-income visitors have been suggested: (1) offer people who cannot afford to pay visitor use fees the opportunity to do volunteer work on a national forest in exchange for a fee waiver; (2) set aside areas where visitor use fees are not required; (3) establish days or times when site fees are waived; and (4) provide financial assistance to low-income visitors, for example, by giving them free annual recreation passes (Burns and Graefe 2006, Scott 2013). Some of these practices are already in place in the Pacific Northwest (Burns and Graefe 2006).

Roberts et al. (2009) provided a resource guide to help land management agencies better serve culturally diverse populations in California by improving communication and outreach, providing appropriate facilities and services, developing partnerships and relationships with organizations that promote outdoor experiences for low-income and minority groups, and taking advantage of other available resources. For example, some specific suggestions include: (1) use international symbols for facilities that are easily understood across cultures; (2) hire multilingual field personnel with strong cultural competency; (3) cultivate a partner to sponsor a van or minibus to transport local diverse populations to recreation sites; and (4) engage with community centers in hard-to-reach communities (Roberts et al. 2009). The suggestions contained in the guide are relevant to the NWFP area as a whole.

Nonrecreational camping and homelessness—

U.S. Forest Service law enforcement officers surveyed by Cerveny and Baur (n.d.) reported that the frequency of homelessness and long-term camping on national forests is increasing and that the greater share of responsibility for addressing the issue seems to fall on patrol officers. The officers typically respond on a case-by-case basis by issuing citations for “stay violations,” “illegal residence” violations, or other violations (e.g., sanitation, litter, or drug possession). However, the same individuals repeatedly return to the forest, often to the same sites, or they may shift between national forest land and other nearby public lands. Recognition by agency management of the resource impacts and social effects associated with long-term camping would spotlight the concerns raised by law enforcement. Treating homelessness as a chronic and systemic phenomenon in which the agency plays a critical role would potentially lead to greater acceptance of responsibility and action. For example, law enforcement officers surveyed described creative solutions that involved partnerships with public health agencies, social services, municipal police, and citizen groups to identify safe housing options in local communities.

Wildfire management—
Whether reducing hazardous fuels or engaging in other forest management activities, managers are required to consider how their actions may adversely affect environmental justice populations disproportionately. It is also important to consider whether certain populations disproportionately benefit from wildfire risk mitigation and wildfire suppression activities and resources so that these benefits may be more equitably distributed. Poverty and minority status are among the social variables that researchers use as indicators of social vulnerability. Research indicates that socially vulnerable populations living in fire-prone forest ecosystems in which the fire hazard is high tend to be more vulnerable to wildfire than less socially vulnerable populations because they often have fewer resources to invest in wildfire mitigation actions, have lower participation rates in wildfire mitigation assistance programs, and have less access to wildfire response resources when a fire ignites. These findings suggest that not only is it important for fuels reduction treatments to be proportionately distributed to places where low-income and minority populations border or live near fire-prone federal forests characterized by high wildfire hazard; but treatments might target these locales because of higher social vulnerability to wildfire. Furthermore, given research that indicates that low-income and minority populations may have less access to assistance programs that support wildfire mitigation strategies, directing outreach as well as financial and technical assistance to these populations may help them increase fire-safe practices around their homes for greater protection from high-severity fire.

Conclusions

This chapter responds to federal forest managers’ request for information about trends in the size of environmental justice populations in the NWFP area, and the implications of these trends for federal forest management. We found that poverty rates grew in the Washington, Oregon, and California portions of the Plan area between 1990 and 2012, and were most pronounced in northern California and southern Oregon. Poverty rates were uniformly higher in nonmetropolitan counties than in metropolitan counties in the Plan area during the analysis period, and were also higher than the national average. Minority populations also increased in size and percentage of the regional total, and this increase was greatest among the Hispanic or Latino population. The percentage of the population identifying as Hispanic or Latino doubled in nonmetropolitan counties, and nearly tripled in metropolitan counties in the NWFP area.

The published literature about environmental justice populations and their relations with federal forests in the Plan area focuses primarily on the environmental workforce, commercial NTFP gathering, and recreation. Low-income and minority populations are prominent in the environmental workforce and in commercial NTFP gathering on federal forest lands. However, as forest workers, they often experience low job quality, and they are underrepresented when it comes to developing regulations and management guidelines for NTFP harvesting, suggesting a need for more oversight and outreach by forest managers. In contrast, low-income and minority populations have low participation rates in recreation on national forests in the Plan area. The literature addresses constraints to their participation and provides suggestions for how forest managers can overcome some of these constraints. Two emergent topics that are less well documented but where research is ongoing are nonrecreational camping on federal forests, particularly homelessness, and the impacts of wildfire management activities on environmental justice populations.

Important information gaps remain, however. There is virtually no information about how the NWFP or forest management activities more broadly have affected low-income or minority populations apart from American Indians. Aside from recreation, research gaps exist regarding noneconomic relations between environmental justice populations and federal forests. More research is needed to increase understanding about variation within minority groups regarding their interactions with federal forests in particular places, including how low-income or minority status intersects with subgroup characteristics (i.e., gender, age, religion) to influence forest values, uses, and management impacts. The growth in environmental justice populations throughout the NWFP area calls for ongoing investigation into how these populations relate to federal forests and are affected by their management.

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Appendix: Counties in the Northwest Forest Plan area (2012 designation)

State, county, designation	State, county, designation
California, Colusa County (nonmetropolitan)	Oregon, Polk County (metropolitan)
California, Del Norte County (nonmetropolitan)	Oregon, Sherman County (nonmetropolitan)
California, Glenn County (nonmetropolitan)	Oregon, Tillamook County (nonmetropolitan)
California, Humboldt County (nonmetropolitan)	Oregon, Wasco County (nonmetropolitan)
California, Lake County (nonmetropolitan)	Oregon, Washington County (metropolitan)
California, Lassen County (nonmetropolitan)	Oregon, Yamhill County (metropolitan)
California, Marin County (metropolitan)	Washington, Adams County (nonmetropolitan)
California, Mendocino County (nonmetropolitan)	Washington, Benton County (metropolitan)
California, Modoc County (nonmetropolitan)	Washington, Chelan County (metropolitan)
California, Napa County (metropolitan)	Washington, Clallam County (nonmetropolitan)
California, Shasta County (metropolitan)	Washington, Clark County (metropolitan)
California, Siskiyou County (nonmetropolitan)	Washington, Cowlitz County (metropolitan)
California, Sonoma County (metropolitan)	Washington, Douglas County (metropolitan)
California, Sutter County (metropolitan)	Washington, Franklin County (metropolitan)
California, Tehama County (nonmetropolitan)	Washington, Grant County (nonmetropolitan)
California, Trinity County (nonmetropolitan)	Washington, Grays Harbor County (nonmetropolitan)
California, Yolo County (metropolitan)	Washington, Island County (nonmetropolitan)
Oregon, Benton County (metropolitan)	Washington, Jefferson County (nonmetropolitan)
Oregon, Clackamas County (metropolitan)	Washington, King County (metropolitan)
Oregon, Clatsop County (nonmetropolitan)	Washington, Kitsap County (metropolitan)
Oregon, Columbia County (metropolitan)	Washington, Kittitas County (nonmetropolitan)
Oregon, Coos County (nonmetropolitan)	Washington, Klickitat County (nonmetropolitan)
Oregon, Crook County (nonmetropolitan)	Washington, Lewis County (nonmetropolitan)
Oregon, Curry County (nonmetropolitan)	Washington, Mason County (nonmetropolitan)
Oregon, Deschutes County (metropolitan)	Washington, Okanogan County (nonmetropolitan)
Oregon, Douglas County (nonmetropolitan)	Washington, Pacific County (nonmetropolitan)
Oregon, Hood River County (nonmetropolitan)	Washington, Pierce County (metropolitan)
Oregon, Jackson County (metropolitan)	Washington, San Juan County (nonmetropolitan)
Oregon, Jefferson County (nonmetropolitan)	Washington, Skagit County (metropolitan)
Oregon, Josephine County (nonmetropolitan)	Washington, Skamania County (metropolitan)
Oregon, Klamath County (nonmetropolitan)	Washington, Snohomish County (metropolitan)
Oregon, Lane County (metropolitan)	Washington, Thurston County (metropolitan)
Oregon, Lincoln County (nonmetropolitan)	Washington, Wahkiakum County (nonmetropolitan)
Oregon, Linn County (nonmetropolitan)	Washington, Walla Walla County (nonmetropolitan)
Oregon, Marion County (metropolitan)	Washington, Whatcom County (metropolitan)
Oregon, Multnomah County (metropolitan)	Washington, Yakima County (metropolitan)

Roasting salmon over an open fire.
Photo by Jon Ivy, Coquille Indian Tribe.

Chapter 11: Tribal Ecocultural Resources and Engagement

Jonathan Long, Frank K. Lake, Kathy Lynn, and Carson Viles[1]

Introduction

In this chapter, we review scientific information regarding the conservation and restoration of forest ecosystems on public lands within the Northwest Forest Plan (NWFP, or Plan) area that harbor special value for American Indian tribes and individuals. We highlight advances in understanding how changes in climate, fire, hydrology, vegetation, and resource management regimes have affected tribal ecocultural resources and how land management can promote ecocultural resources in the future. In particular, we examine how distinctive strategies for engaging tribes in restoring ecocultural resources can uphold both tribal rights and federal responsibilities, while supporting other federal land management goals.

An Integrative Perspective on the Term "Ecocultural Resources"

A key theme in this chapter is the interconnections among tribal communities and their environment within a larger socioecological system. When considering socioecological systems that have developed with indigenous people over millennia, dividing biophysical entities into "ecological" and "cultural" categories would be particularly problematic (Burger et al. 2008). Tribal worldviews in the Pacific Northwest emphasize that humans are an integral part of the natural world and their well-being depends upon maintaining reciprocal relationships with its inhabitants (Anderson 2005, Heyd and Brooks 2009). Based upon work by others who have addressed that issue, we adopt the more integrative term "ecocultural" in this chapter. Rogers-Martinez (1992) was an early advocate for recognizing the need for ecological and cultural integration in restoration in a tribal context: "In other words, what we aim to restore is not only the land, but our relationship with it" (p. 69). Similarly, Harris and Harper (2000) used the term "eco-cultural dependency webs" in characterizing interactions between tribal people and their environment. The term "ecocultural" has been featured by Tomblin (2009) and the Karuk Tribe (Lake et al. 2010) and many others to characterize goals of tribal restoration in recent years.

The term "resource" can help to describe physical assets for which the U.S. government has a particular responsibility to tribes to protect (see "The Federal-Tribal Relationship"), but it also suggests an emphasis on material uses. Tribes regard many places, waterbodies, animals, plants, and fungi for material uses as foods (figs. 11-1 through 11-3), medicines, and crafts, but also for nonmaterial values, including sense of place, sacredness, and other dimensions of cultural significance (Burger et al. 2008). In a similar vein, we use the term "ecosystem services" (see chapters 1, 9, and 12), but we emphasize the importance of "cultural ecosystem services" that encompass both subsistence values and nonmaterial values important to native peoples (Burger et al. 2008, Schröter et al. 2014).

Background on Tribes in the Northwest Forest Plan Area

Over 70 federally recognized American Indian tribes, and many more tribes that are not currently recognized, have tribal lands or ancestral territory within the NWFP boundary (Vinyeta and Lynn 2015). Between 1954 and 1964, Congress "terminated," or ended federal acknowledgment, for scores of tribes particularly in California and Oregon. This chapter uses the term "tribes" when describing the collectives recognized as sovereign governments by the U.S. government, as well as many tribes that have petitioned for such recognition (Koenig and Stein 2008).

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Frank K. Lake

Figure 11-1—A traditional meal of lamprey and Chinook salmon (*Oncorhynchus tshawytscha*) is prepared on coast redwood and western redcedar sticks over a madrone wood fire along the Salmon River, California, April 2016.

Frank K. Lake

Figure 11-2—Tanoak acorns, lion's mane (*Hericium erinaceus*) mushrooms, and evergreen huckleberries collected on the Six Rivers National Forest near Orleans, California, October 2005.

Frank K. Lake

Figure 11-3—Preparing a fall dinner plate of mushrooms (lion's mane, chanterelles, and oyster) with a leg of black-tailed deer, served in Orleans, California, October 2005.

Much of the ancestral territory of tribes was transferred to the U.S. Forest Service, Bureau of Land Management (BLM), and National Park Service by the early 20th century; however, that process of land transfer continued even into the 1960s, when the U.S. government terminated its relationship with the Klamath Tribes and transferred their reservation to form much of the current Fremont-Winema National Forest in Oregon (Catton 2016). Many tribes that were re-recognized starting in the last quarter of the 20th century did not regain control over their former lands (Slagle 1989). However, the U.S. government has transferred some public lands back to tribal control in recent decades (Catton 2016). Several returns were made to correct for survey errors, including transferring part of the Gifford Pinchot National Forest to the Yakima Indian Reservation, parts of the Mount Hood and Willamette National Forests to the Warm Springs Reservation, and parts of the Olympic National Forest to the Quinault Tribe. Congress also transferred public lands to the Coquille Tribe in 1996 after it was re-recognized (see “Coquille Indian Tribe” on p. 882).

Each tribe has a unique history and relationship with the U.S. government, as well as unique environmental, economic, and cultural ties that influence how they are affected by public land management in the NWFP area. Federal land management and policy affects tribal ancestral lands and resources that remain critical to the well-being of tribal communities. The U.S. government has a legal responsibility to consult with federally recognized tribes regarding their interests in public lands and potential impacts to tribal trust resources and rights (see “The Federal-Tribal Relationship” on p. 854), as articulated in the Presidential Memorandum on Government-to-Government Relations with Native American Tribal Governments (Clinton 1994). The Record of Decision for the NWFP restates that responsibility and calls for resolving conflicts collaboratively with affected tribes because of the potential to affect tribal activities in areas subject to tribal treaty off-reservation rights (USDA and USDI 1994).

The chapter also uses the term “American Indians” to refer to individuals of Native American ancestry and especially in a historical context before the United States assumed control over the lands of the NWFP. In addition to laws and policies that deal with tribes as sovereign nations, the U.S. government has policies that deal with American Indians as individuals (Catton 2016). For example, the new 2012 forest planning rule accords both tribes and American Indians special consideration (USDA FS 2012). The rule highlights environmental justice, for which Executive Order 12898 directs agencies to evaluate whether federal activities have disproportionately high and adverse human health or environmental effects on minority and low-income populations, which includes American Indians (see chapter 10).

Guiding Questions

Managers from the Forest Service requested that the synthesis report address the two-part question of “What is the capacity of the Northwest Plan area to provide for Native American first foods (e.g., salmon, elk, huckleberry, camas, etc.), and is active management called for?” “First foods” is a term that some tribes have applied to traditional foods that have been and remain very significant in their diet and culture (Lynn et al. 2013). This chapter addresses that question as part of a larger examination of opportunities to promote tribal ecocultural resources and engagement in management of federal forest lands. In particular, we consider the effects of historical changes in the relationships between tribes and forests in the NWFP area, and how restoring tribal cultural practices would affect sustainability of those socioecosystems. After first considering the general context for land management and restoration to support values important to tribes, we delve into recent science to address several questions in more detail:

1. What resources within the NWFP area have special value to tribes, and what factors are influencing the quality and availability of those resources, as well as the ecosystems that produce them? In particular, how has the reduction in tribal influences since Euro-American colonization affected those resources and ecosystems?

The Federal-Tribal Relationship

A brief overview of the distinctive relationship between the U.S. government and 567 federally recognized American Indian and Alaska Native tribes is important to understanding the issues considered in this chapter. All federal agencies have a trust responsibility to protect tribal rights, lands, assets, and resources, which collectively constitute tribal trust rights and resources (Clinton 1994, Wood 1995). Federal recognition acknowledges tribes as political sovereigns with inherent rights to self-governance. When the U.S. government entered into treaties with American Indian tribes, it made commitments to provide tribes with goods and services and to protect their ability to harvest natural resources. For example, the Superintendents of Indian Affairs in Washington and Oregon, Isaac Stevens and Joel Palmer, respectively, negotiated 10 treaties involving tribes in the Pacific Northwest between 1853 and 1856. These treaties included provisions to protect specific activities on lands beyond the reservations such as harvesting fish (fig. 11-4) and shellfish, hunting, gathering plants such as roots and berries and erecting temporary buildings to cure them, and pasturing horses and cattle (Bernholz and Weiner 2008, Woods 2005). Court decisions have recognized that tribes reserved rights to harvest resources in ways that encompass trapping, camping, and other activities on public lands that are not necessarily referenced in a given treaty (Catton 2016, Goodman 2000, Wilkinson 1997). Figure 11-5A shows the locations of present-day reservations and the much larger cessions of lands from tribes to the U.S. government under those treaties. The U.S. government had negotiated 18 treaties with many tribes in California from 1851 to 1852, totaling one seventh of its land area, but the Senate refused to ratify them (Wood 2008). Instead, through executive orders and Congressional authorizations over subsequent decades, the U.S. government established a number of small reservations across the Pacific Northwest, and even smaller "Rancherias" for many tribes in California (fig. 11-5B) (Wood 2008).

Tribes have other claims that influence off-reservation land management even in the absence of ratified treaties of cession. For example, tribes have fishing and water rights for their reservations; legal defenses of those rights have prompted restrictions on upstream water withdrawals, notably in the Klamath River basin (Gosnell and Kelly 2010). Some tribes, such as the Klamath Tribes, have retained rights in former reservation lands that were acquired by the United States following termination (Goodman 2000). The Forest Service has established agreements with many tribes that do not have formal treaty rights that allow traditional harvesting within their ancestral lands (Catton 2016). Therefore, the cessions mapped in figures 11-5A and 11-5B present a very incomplete picture of tribes' ancestral connections to lands in the NWFP area, but they nevertheless illustrate particular connections between tribes and public lands that are enshrined in federal law. Given that federal public lands agencies control so much tribal ancestral land, and many tribes have only small land areas under their direct control, federal land management actions profoundly affect tribal access to resources (Dobkins et al. 2016).

The unique status of federally recognized tribes requires that U.S. government entities consult directly with these tribal governments when addressing issues that may affect trust resources and the welfare of their tribal members. Consultation is a cornerstone of the government-to-government relationship and clearly distinguishes the tribes from other entities (Nie 2008). Executive Order 13175, Consultation and Coordination with Indian Tribal Governments, sets requirements for the consultation process to ensure meaningful and timely input by tribal officials when federal action may affect tribal lands and resources. In addition, consultation obligations are found in numerous statutes (Galanda 2011). For example, the Native American Graves and Repatriation Act (P.L. 101-601) of 1990 imposed requirements for consultation with tribal officials or lineal descendants when officials anticipate or discover that activities on federal lands will affect American Indian burials.

U.S. Army Corps of Engineers

Figure 11-4—Tribal members fishing with dipnets at Celilo Falls, which was submerged by the construction of The Dalles Dam in the 1950s. Several tribes have rights to fish associated with this historic location on the Columbia River on the border of Washington and Oregon.

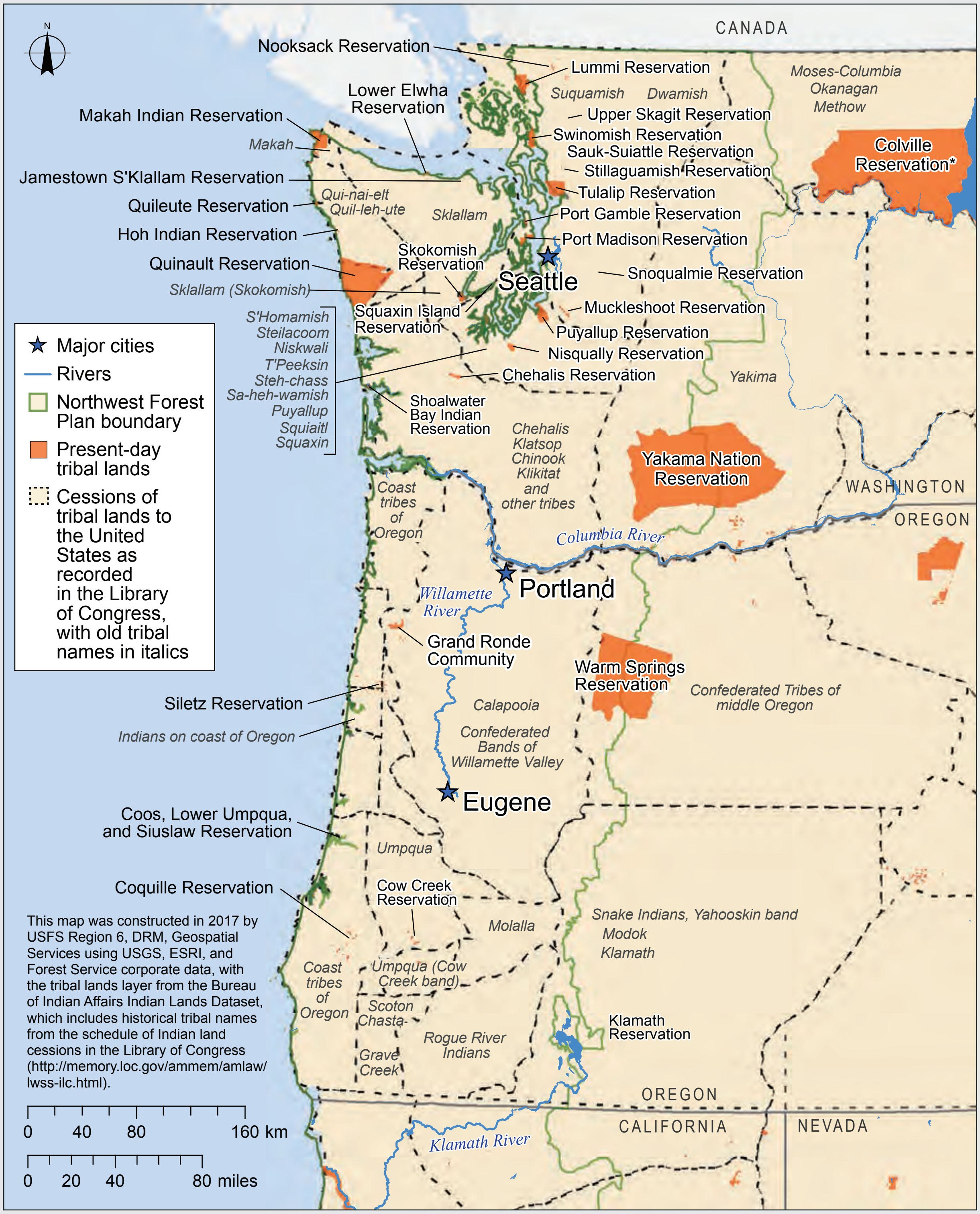

Figure 11-5A—Present-day tribal reservations, and ancestral lands mapped as cessions to the U.S. government in Library of Congress records, within the Northwest Forest Plan area in Washington and Oregon. The asterisk denotes the Colville Reservation, which lies outside the boundary but belongs to tribes that have ancestral lands and reserved treaty rights within the boundary.

Figure 11-5B— Present-day tribal reservations, and ancestral lands mapped as cessions to the U.S. government in Library of Congress records, within the Northwest Forest Plan area in California. Asterisks denote tribal lands outside the boundary belonging to tribes that have ancestral lands within the boundary.

2. What land management strategies can promote tribal ecocultural resources, and how do those strategies relate to management, research, and monitoring for economic, social, cultural, terrestrial and aquatic systems more broadly?
3. What strategies for engaging tribes in forest planning and management have been effective in addressing tribal concerns over how federal land management affects tribal ecocultural resources and rights?

This chapter focuses on issues for which federal land management entities such as the U.S. Forest Service and BLM have primary influence, such as managing vegetation, fire, roads, and trails. Because of that focus, this chapter mentions but does not examine in depth many other issues that have important effects on tribal ecocultural resources, including reintroduction of extirpated species, human population growth, urban development, and management of nonfederal lands. The intent of the science synthesis is to inform land management planning but not to make policy recommendations (see chapter 1). However, the periodic monitoring reports under the NWFP (Harris 2011, Stuart and Martine 2005, Vinyeta and Lynn 2015) were guided by a Tribal Monitoring Advisory Group to complete tribal surveys and case studies that informed recommendations for strengthening federal-tribal relationships under the NWFP.

Source Materials

The current land management planning rule requires decisionmakers to use best available science and also to request information about tribal traditional ecological knowledge (referred to as "native knowledge," see Glossary), land ethics, cultural issues, and sacred and culturally significant sites (USDA FS 2012). This chapter, as do others in this science synthesis, draws primarily from peer-reviewed scientific publications, focusing on those published since the NWFP was adopted. This chapter also draws upon findings from related chapters in this report to highlight how broader strategies being considered in forest management and planning may affect tribal ecocultural resources. Because considerable information regarding particular tribal resources and federal-tribal relationships has been documented in other kinds of publications, including theses, dissertations, and agency and tribal reports, this chapter refers to some of these publications to help fill gaps in peer-reviewed literature. However, tribal knowledge is often passed down orally in native languages rather than specialized, technical terminology (Ellis 2005). Therefore, relying on published information excludes traditional tribal knowledge that has not been referenced in such publications. Such exclusion risks perpetuating long-standing power imbalances (Gavin et al. 2015) as well as reinforcing barriers to integrating traditional knowledge into land management. Managers may discount traditional knowledge that does not seem to fit with their framing or understandings of particular issues (Bussey et al. 2016). Furthermore, tribal knowledge may be distorted or diminished as it is "scientized," or translated into Western scientific syntheses written in nonnative English (Agrawal 2002). Publication and institutionalization of traditional knowledge risks transforming it into "non-living knowledge for which no one has specific responsibility to pass on" (Gamborg et al. 2012: 542). The section on "Integrating traditional ecological knowledge in collaborations" (p. 900) identifies safeguards that have been recommended to avoid such outcomes.

Despite these concerns, it is important to recognize that many tribes have become forerunners in producing scientific knowledge in the Western tradition (Breslow 2014), and the participatory approaches used with tribes to prepare many of the articles, theses, dissertations, and scientific reports considered in this synthesis afford some protections against misuse. Nevertheless, readers of this synthesis are advised to consider the implications of relying exclusively on scientific publications. For example, published science may not well reflect tribal concerns over practices that are widely used in nontribal institutions, such as permitting, herbicide use, and burning outside of customary seasons (Halpern 2016, LeCompte-Mastenbrook 2016). Consistent with the planning rule, planners can elicit such information through a variety of pathways in addition to formal consultation, including collaborative partnerships as discussed within this chapter.

Key Findings

Our synthesis starts by considering important concepts that help to frame the context for forest management to promote tribal well-being.

What Is the Context for Promoting Tribal Well-Being Through Forest Management?

The forest planning rule requires that land management plans promote ecological sustainability and contribute to social and economic sustainability, in particular by managing areas of tribal importance (USDA FS 2012). Scientific research has recognized the deeply interwoven relationships between American Indians and the nonhuman elements of ecological systems in the Pacific Northwest region. These relationships remain critical to sustaining tribal food and health security; economic prosperity; recreation and tourism; spiritual and ceremonial practices and observances; heritage and cultural identity; and traditional knowledge systems, beliefs, and intergenerational exchange (Burger et al. 2008, De Groot et al. 2002, Fisher et al. 2008, Tengberg et al. 2012). For example, tribal material well-being continues to depend on material from forests for food, water, medicines, fuel, crafts, arts, and other creations. Tribal well-being also depends upon forest environments for sense of place and the ability to practice and pass on cultural traditions (Satterfield et al. 2013), including ceremonies for world renewal, coming of age, and first foods (Willette et al. 2015). Various species represent "cultural keystones" because of their prominent roles in maintaining tribal economies, identity, and cultural traditions (Garibaldi and Turner 2004). For example, first food ceremonies held by many tribes feature huckleberries (*Vaccinium* spp.), salmon (*Oncorhynchus* spp.), venison, and edible roots (Mack and McClure 2002), while salmon and tanoak (*Notholithocarpus densiflorus)* may have provided half of the traditional diet among members of the Karuk Tribe in California (Norgaard 2014a). The inability of many tribal members to harvest such foods has been linked to a host of social ills (LeCompte-Mastenbrook 2016, Norgaard et al. 2017). Many tribes are working to increase their access to traditional foods (figs. 11-1, 11-2, and 11-3) as part of a food "security" or "sovereignty" movement, which is part of broader efforts to sustain and enhance the well-being of tribal communities (Daniel et al. 2012, Hernández-Morcillo et al. 2013, LeCompte-Mastenbrook 2016). Researchers have extended the cultural keystone concept to "cultural keystone places," where cultural keystone species often occur, and which also have particularly great cultural, historical, social, ecological, and economic values (Cuerrier et al. 2015). Tribal cultural revitalization efforts depend heavily on having influence over management of public lands (MacKendrick 2009, Turner and Turner 2008).

The new land management planning rule focuses on ecosystem services (see chapter 9), encompassing "provisioning services" that support tribal harvesting of wild plants, animals, and materials, as well as less tangible "cultural ecosystem services" that are distinctively important to tribes and often underaccounted in conventional analyses (Asah et al. 2014). However, Raymond et al. (2013) and others have criticized the implicit emphasis of ecosystem services on economic production and associated markets. In contrast, they suggest that other metaphors such as "ecocultural community" invoke values that are important to indigenous peoples, such as reciprocity and relationships with past and future human generations and nonhuman entities. Upholding such values traditionally limited resource harvest in ways that promote sustainability, as highlighted in studies of harvesting plants and wildlife (Deur 2009, Jordan 2015). Such traditional principles are important in modulating societal demand for ecosystem services, which is a key challenge in applying the concept to public lands management (Patterson 2014).

Vulnerability and risk assessments for tribal communities need to be specialized to properly consider risks to tribes and their members who have traditionally relied more heavily upon wild fish, game, and wild plant foods, medicines, and other natural materials that are processed, stored, and used in homes (Burger 2008; Donatuto et al. 2014, 2011; Kerns and Ager 2007). For example, in a study of members of the Confederated Tribes of the Umatilla Reservation (within the Columbia River watershed east of the NWFP area), Harris and Harper (1997) reported that exposures to various contaminants for an average American Indian engaged in a traditional subsistence lifestyle may be 2 to 100 times greater than for an average suburban resident owing to greater ingestion of fish and other products that could bear contaminants. These findings are likely relevant to American Indians throughout the NWFP area who engage in lifestyles that similarly involve high consumption and handling of resources from wildlands. These factors increase the need for both protective standards and management that account for the distinctive characteristics of tribal communities.

Cross-boundary and broad-scale perspectives—
Tribes in the NWFP area are connected to a diverse range of ecosystems from the mountains to the sea, encompassing marine, estuarine, riverine, valley, wetland, grassland, foothill, montane, and alpine environments that collectively offer a wide range of places and resources valued by tribes (Suttles 1990, Turner et al. 2011). This synthesis focuses on forested ecosystems while considering other interconnected ecosystems, including grasslands, meadows, wetlands, estuaries, bays, and the Pacific Ocean that collectively sustain many species of special concern to tribes. Tribal well-being is strongly connected to the condition of entire terrestrial and aquatic ecosystems across federal, tribal, state, county, and private lands. Development and environmental degradation of areas and waterbodies outside of present-day tribal lands has limited the ability of tribal communities to access desired resources (Donatuto et al. 2014, Norgaard et al. 2017). Consequently, working across broad scales and boundaries is critical for sustaining tribal ecocultural resources. A focus on watershed processes is particularly important because many of those resources depend on flows from mountain peaks to coastal zones and because many tribes in the NWFP area reside in coastal areas and river valleys (fig. 11-5A and 5B). Federal land management planning emphasizes such a watershed perspective, which helps to consider how forest management may affect downstream aquatic systems and related uses that are important to tribes. There are also important cross-boundary issues involved in terrestrial systems, especially because tribes have treaty harvesting rights and interests in ancestral lands beyond their present-day reservations, opportunities to treat adjacent national forest lands under the Tribal Forest Protection Act of 2004, and concerns for transboundary ecological processes such as wildfire. There are also complex land management situations such as the Quinault Special Management Area, which is managed by the Forest Service with 45 percent of proceeds from the sale of forest products to be provided to the Quinault Indian Nation (Vinyeta and Lynn 2015). Tribes that have been displaced from their ancestral homelands often have strong interest in lands that are distant from their current residences (Cronin and Ostergren 2007). In particular, some reservations are governed by confederated tribes whose members originated from broad territories and held a wide range of traditions and cultural practices. For example, descendants from the Rogue River tribes are now members of the Confederated Tribes of the Siletz Indians and the Confederate Tribes of the Grande Ronde Community who currently reside in northwestern Oregon, but they retain interest in forest management activities in their ancestral territory on the Rogue-Siskiyou National Forest in southwestern Oregon. As another example, the Nez Perce Tribe, whose reservation is in Idaho, retained rights to fish within the NWFP area. These examples demonstrate how maps of both contemporary tribal lands and ceded territories, such as in figures 11-5A and 11-5B, underrepresent tribal interests across the region.

What Ecocultural Resources and Associated Ecosystems Have Special Value to Tribes in the NWFP Area?

In this section, we highlight resources and associated ecosystems that emerged in our review as particularly important to tribes across the NWFP region. Land management agencies have long focused on archaeological sites and artifacts as the subjects of cultural resource protection, but increasingly there has been a recognition that living resources are critical cultural resources (Catton 2016). Tribes generally hold that all elements of the natural world have cultural significance, or as described by one Pacific Northwest tribal leader, “The Creator made all things one. All things are related and interconnected. All things are sacred. All things are therefore to be respected” (Turner and Berkes 2006: 499). The chapter provides only examples of the profound and varied relationships between tribes and nonhuman entities that have been especially prominent in scientific literature. To characterize the significance of all species, ecosystems, and places from the perspective of dozens of tribes would require a far more extensive report than can be provided here. However, the chapter includes citations that offer more breadth and depth.

Water and waterbodies—
Water has tremendous material value that can be measured in terms of quality, quantity, and availability, as well as nonmaterial values that are discussed further below. Tribes and federal land management agencies have been involved in conflicts regarding water rights, dams, diversions and instream flows to sustain fisheries (Gosnell and Kelly 2010). Because the construction of large dams in river basins of the Pacific Northwest has greatly reduced anadromous fish populations and availability of traditional fishing sites (Gosnell and Kelly 2010, Hamilton et al. 2005, McClure et al. 2003), reservoir dam removal is an important issue discussed further below.

Ancestral and sacred places—
Like streams of water, tribal ancestral ties permeate and connect the diverse landscapes of the Pacific Northwest. The antiquity of resource uses is evident in sites across the NWFP area, including camas roasting pits dating to more than 7,000 years ago, berry processing camps dating back 3,000 years, scars in cedar trees that are hundreds of years old, and many other features that are discernable to experienced observers (Turner 2014). Lands and bodies of water support a variety of tribal values beyond their importance as sustenance and habitat for people, plants, and animals, including historical and spiritual values (Colombi 2012, Russo 2011, Russo and Zubalik 1992). Such values are recognized as cultural ecosystem services under the planning rule (USDA FS 2012). American Indians commonly place high priority on the cultural and spiritual values of public lands and in maintaining undeveloped conditions, while still recognizing that human activities such as maintaining roads and resource management are important to sustaining traditional relationships to the land (Flood and McAvoy 2007). Many areas considered sacred by tribes are likely to have a history of caretaking, productivity, and diversity (Hughes and Jim 1986), which could render them high priorities for conservation and restoration.

Focus on keystone species—
Several groups of organisms represent prominent tribal ecocultural resources across the NWFP area, including anadromous fish; ungulates; geophytes; fungi and lichens; trees that provide nuts, foliage, bark, and wood; berry-bearing shrubs; and many other plants and animals used for food, medicine, regalia, and crafts. Many of the plants and animals discussed below are likely to qualify as cultural keystone species for multiple tribes (Garibaldi and Turner 2004) because of their important roles in maintaining cultures and because they were widely used and traded by tribes in the NWFP area (Turner and Loewen 1998). These species can also be ecological keystones owing to their importance in maintaining important ecological processes. Consequently, many of these species warrant consideration as potential focal species under the new forest planning rule, and they would also be important to consider as keystones in an integrated ecocultural context.

Mammals, including ungulates and furbearers—
Columbian black-tailed deer (*Odocoileus hemionus columbianus*), Columbian white-tailed deer (*Odocoileus virginianus leucurus*), elk (*Cervus elaphus*), and antelope (*Antilocapra americana*) are large animals valued for food, hides, and nonmaterial cultural values in the NWFP area. These species depend on forest openings and nonforest communities that were maintained with former tribal burning practices (Anderson 2009, Boyd 1999, Turner et al. 2011). Managers of private forest lands have argued that populations of elk and black-tailed deer have declined without regeneration harvests (Burns et al. 2011). Fuels reduction can enhance the quantity and quality of elk forage (Long et al. 2008). Deer browse the new shoots or branch-tip growth of many of the berry-producing shrubs that are also important to tribes, including salal (*Gaultheria shallon*) (Stockton et al. 2005). In some areas within the NWFP, such as the Gulf and San Juan Islands, black-tailed deer have increased, leading to declines in many understory plants as well as birds (Martin et al. 2011). However, in many other parts of the NWFP area, a decline in elk and deer populations associated with fire exclusion and suppression and forest succession has reduced hunting opportunities and diminished tribal food security (LeCompte-Mastenbrook

2016, MacDougall 2008, MacKendrick 2009). Collaborative landscape efforts designed to restore habitats (e.g., winter range associated with lower elevation oak woodlands, or higher elevation forests) can help address tribal interests in increasing these wild ungulate populations. For example, under a settlement of a lawsuit by the Muckleshoot Indian Tribe, the Mount Baker–Snoqualmie National Forest designated two special management areas for elk forage (LeCompte-Mastenbrook 2016). That action was in part a response to impacts of late-successional reserve designations under the NWFP on elk habitat, which has also been highlighted as a tribal concern in NWFP monitoring reports (Stuart and Martine 2005).

Tribes use many mammals such as river otter (*Lontra canadensis*), American beaver (*Castor canadensis*), mountain beaver (*Aplodontia rufa*), Pacific marten (*Martes caurina*), fisher (*Pekania pennanti*), mink (*Neovison vison*), and porcupine (*Erethizon dorsatus*) in making regalia and other cultural items (Dobkins 2009, Matthews et al. 2013). Many of these species have prominent symbolic roles in tribal cultural traditions as well. Ecological implications of the decline or extirpation of some species, such as wolf (*Canis lupus*) and beaver, are discussed further below under "Species losses," while chapter 6 provides additional discussion of ecology and management of wildlife.

Birds important for food, regalia, and ceremonies— Various birds are important as sources of food and materials for tribal regalia, and many species have special cultural significance in ceremonies, stories, and songs. Turner and Bhattacharyya (2016) provide an extensive review of the cultural significance of birds from the Pacific Northwest, recounting the deeply rooted connections among tribal people, plants, and birds in both corporeal and spiritual realms. They reported common connections among important bird species and plants harvested for fruits and roots. For example, they noted that many tribes identify the Swainson's thrush (*Catharus ustulatus*) as the "salmonberry bird," an important indicator of the ripening of salmonberries (*Rubus spectabilis*) in coastal forests of the Pacific Northwest. Jordan (2015) provides a detailed examination of how the Hupa people have woven the pileated woodpecker (*Dryocopus pileatus*) into their material and spiritual culture by using the feathered scalps to make dance regalia (figs. 11-6 through 11-8) and maintaining a reciprocal relationship with the bird. For example, the Hoopa Valley Tribe has engaged in research to study how forest disturbances influence the species (see "Tribal Ecosystem Services From Dead Trees and Forest Gaps" on p. 864). Other birds that are prominently featured in tribal featherwork include mallard duck (*Anas platyrynchos*), bald eagle (*Haliaeetus leucocephalus*), golden eagle (*Aquila chrysaetos*), mountain quail (*Oreortyx pictus*), California quail (*Callipepla californica*), band-tailed pigeon (*Patagioenas fasciata*), acorn woodpecker (*Melanerpes formicivorus*), and northern flicker (*Colaptes auratus*) (Gleeson et al. 2012). Some species, such as various owls, have cultural

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Figure 11-6—A pileated woodpecker head mounted on a handle made of madrone "curly" wood (with disfigured growth from a honeysuckle [*Lonicera hispidula*] vine) and adorned with woodpecker tail feathers and shells from dentalium (*Dentalium* sp.) and abalone (*Haliotis* sp.). This regalia item, photographed June 2007, was made and used in contemporary tribal (Karuk and Yurok) brush dance and war dance ceremonies.

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Figure 11-7—Lake family regalia photographed August 2016, including a quiver made from fisher adorned with red abalone shells surrounded by men's ceremonial headbands composed of acorn woodpecker scalps sewn on tanned deer hide.

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Figure 11-8—Hupa men dressed in brush dance regalia in 2015, adorned with pileated woodpecker scalps along with a variety of other products derived from forest and ocean wildlife.

significance even though members of some tribes in the NWFP area avoid physically interacting with them and their feathers (Gleeson et al. 2012). California condor (*Gymnogyps californianus*) was historically significant, with feathers used in regalia items, and it remains a species of interest for some tribes in the NWFP area (Gleeson et al. 2012).

Forest management and fires affect bird habitat in complex ways, but, in general, increasing forest heterogeneity to include a variety of successional stages can increase avian diversity (Burger et al. 2013). Tribes often emphasize the importance of food webs and habitat to support the range of species on which they depend (Turner and Bhattacharyya 2016). For example, they call attention to the importance of tree cavities and production of nuts, berries, and other foods not only for their own use, but also for wildlife (Long et al. 2016a). Riparian areas are particularly important as harbors for many bird species of special importance to tribes (Turner and Bhattacharyya 2016). Turner and Bhattacharyya (2016) suggested that traditional tribal practices helped to sustain the diversity and productivity of habitats for many important bird species.

Anadromous fish—

Many tribes in the NWFP area value anadromous fish such as salmon and trout (*Oncorhynchus* spp.) (fig. 11-1) and sturgeon (*Acipenser* spp.) as cultural keystones (Benson et al. 2007, Crozier and Zabel 2006, Richter and Kolmes 2005). Lamprey (*Lampetra tridentata*) is another anadromous fish of special value to tribes (figs. 11-1, 11-10, and 11-11) (Close et al. 2002, Larson and Belchik 1998, Petersen Lewis 2009, Sheoships 2014). Eulachon or candlefish (*Thaleicthys pacificus*) (fig. 11-11) is an important traditional food and trade good when smoke-dried or processed

Tribal Ecosystem Services From Dead Trees and Forest Gaps

The Hoopa Valley Tribe recently partnered with Humboldt State University researchers to examine the effects of tree damage caused by black bears (*Ursus americanus*) (Mendia 2016). They found that bear damage in 40- to 60-year-old stands of Douglas-fir (*Pseudotsuga menziesii* var. *menziesii*) was significantly correlated with dead and decaying trees larger than 10 in (25.4 cm) diameter at breast height. While the damage to trees negatively affected the lumber value of the stand, it created dead wood that would normally be found in older stands and was associated with increased observations of pileated woodpecker, a culturally important species used by tribal members for regalia, as well as red-breasted sapsucker (*Sphyrapicus ruber*) and other cavity-nesting birds. The researcher also observed deer browse on new growth of western swordfern (*Polystichum munitum*) in the canopy gaps resulting from killed trees (fig. 11-9). Consequently, this study found that the small-scale disturbance caused by bears promoted provisioning and cultural ecosystem services associated with biodiversity and tribal spiritual values.

Shannon Mendia

Figure 11-9—Canopy gap resulting from black bear damage to trees in a second-growth redwood stand on the Hoopa Valley Indian Reservation.

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Figure 11-10—Alme Allen (left) and Eugene Coleman hold lampreys caught with a modern wire and rim basket trap along the Klamath River, near Orleans, California, May 2005.

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Figure 11-11—Lamprey (top), candlefish (bottom), and night smelt (*Spirinchus starksi*) (center) harvested by Yurok Tribal members on a basket tray made from sandbar willow (*Salix exigua*), March 2014.

into oil (Larson and Belchik 1998, Mitchell and Donald 2001); the species was listed as threatened in the NWFP area in 2010 (chapter 7). These anadromous species, and their safety for human consumption, have been affected by increasing freshwater temperatures, drought, parasites, and toxins (Benson et al. 2007, Crozier and Zabel 2006, Richter and Kolmes 2005). Norgaard et al. (2013) studied trace metals in three species used by the Karuk Tribe in the Klamath River (salmon, steelhead trout, and freshwater mussels) and found that the foods were deemed safe even at the comparatively higher levels of consumption in traditional tribal diets. A recent Environmental Protection Agency (EPA) study found that 91 percent of lakes in Washington, Oregon, and Idaho had mercury levels in fish tissue that were dangerous to people who consumed high levels of fish (about six fish meals/week) (Herger and Edmond 2012). An earlier EPA study (USEPA, n.d.) conducted with the Columbia Intertribal Fisheries Commission also found high levels of toxins. They found that levels were higher in resident fish than many of the anadromous fish species listed above, except for white sturgeon (*Acipenser transmontanus*), which had some of the most hazardous levels of contamination. They also reported that health risks were far greater to American Indians than to the general public because their fish consumption was 6 to 11 times greater. This study demonstrated the importance in tailoring risk assessments to particular tribal contexts, as well as to consider the potential impacts of releases of toxic substances in sediments stored behind reservoir dams.

Amphibians and mollusks—

Frogs have tribal cultural significance, as portrayed on totem poles and in traditional stories, where they are often represented as supernatural beings that carry important messages and should not be harmed (Barbeau 1930, Turner and Berkes 2006, Wassen 1934). Freshwater mussels (e.g., *Margaritifera falcata, Gonidea angulata,* and *Anodonta californiensis*) are important tribal sources of food (Davis et al. 2013), and they provide other important ecosystem services, including sustaining water quality and food webs (Vaughn et al. 2008). They have a very patchy and reduced abundance in the region particularly resulting from declines in host fish species associated with degraded physical habitats, nonnative fishes, and reduced connectivity resulting from dams on the Klamath, Columbia, and other large rivers (Box et al. 2006, Davis et al. 2013, Howard 2010). Other mollusks, including terrestrial snails and slugs (see chapter 6), have special values to tribes.

Nut-bearing trees—

Tree species that were traditionally valued for nut production include hardwood species such as tanoak (figs. 11-2 and 11-12) (Bowcutt 2013), California black oak (*Quercus kelloggii*) (Long et al. 2016a), Oregon white oak (*Q. garryana*) (Hosten et al. 2006), and California hazel (*Corylus cornuta* var. *californica*) as well as conifer species such as sugar pine (*Pinus lambertiana*) (Anderson 2005) and whitebark pine (*P. albicaulis*) (Mack and McClure 2002).

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Figure 11-12—Chris Peters harvesting acorns, near Orleans, California, November 2012.

Many of the hardwood species are capable of resprouting following fires, but the loss of mature crowns retards nut production for long periods in several species (see chapter 3). There is greater potential for lost nut production in many of these species because fire exclusion, conifer encroachment, and increased fuel loading have increased the potential for high-severity fire (Cocking et al. 2012, Devine and Harrington 2006). However, Sadler's oak (*Q. sadleriana*) is a shrubby oak also valued for nut production, but which can respond to fire with vigorous acorn production. Sudden oak death is a fungal disease that threatens many of the hardwood species (Cobb et al. 2012, Ortiz 2008) (see chapter 3), while white pine blister rust threatens sugar pine and other white pines (Samman et al. 2003). Strategies to promote forests that are more resilient to mortality agents, especially in more frequent-fire forest types, include reducing fuel loads, restoring fire regimes, reducing tree density, and shifting composition toward more fire-adapted native plants (see chapter 3 and Long et al. 2014a).

Trees used for material and medicine—

Many other tree species have special values to tribes for materials, medicines, and other traditional cultural purposes, including various pines (*Pinus* spp.), spruces (*Picea* spp.) (fig. 11-13), Pacific yew (*Taxus brevifolia*), Port Orford cedar (*Chamaecyparis lawsoniana*), coast redwood (*Sequoia sempervirens*), Alaska yellow-cedar (*Callitropsis nootkatensis*), red alder (*Alnus rubra*), bigleaf maple (*Acer macrophyllum*), bitter cherry (*Prunus emarginata*), cascara (*Rhamnus purshiana*), black cottonwood (*Populus trichocarpa* ssp. *trichocarpa*), and many other species (Turner and Hebda 1990, Turner and Loewen 1998). The Pacific crabapple (*Malus fusca*) is a native pome-bearing tree that grows in riparian wetlands and was an important traditional source of food, medicine, and wood for tribes across the coastal range of the NWFP area (Turner and Turner 2008). Western redcedar (*Thuja plicata* Donn ex D. Don) has been particularly highlighted as a cultural keystone species, reflecting its many uses, including canoes (fig. 11-14), totem poles, hats, clothing,

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Figure 11-13—Deanna Marshall (right) with her mother Laverne Glaze, harvesting Sitka spruce (*Picea sitchensis*) roots for basketry material, July 2006. This rain forest species is used by tribes in the coastal zone of the Northwest Forest Plan area.

Dale Northup

Figure 11-14—Carvers Frank Harlow and his nephew Ben Harlow carved four canoes from a large western redcedar tree near Queets, Washington, circa 1932.

baskets, and other crafts (Barbeau 1930, Garibaldi and Turner 2004, Stewart 1995). Western redcedar has been the subject of restoration partnerships involving the Forest Service and tribes (Smith and Farque 2001). The use of some conifers for material, such as cedar trees peeled for bark, have produced culturally modified trees that retain evidence of intentional alteration by American Indians. Because such trees have scientific and cultural value as records of activity by past generations of American Indians, they are important to consider when planning harvest and fire management (Eldridge 1997, Turner et al. 2009). Populations of both hardwoods and conifers are threatened by diseases, a rapidly changing climate, and associated disturbances (see “Climate change” on p. 873).

Understory plants for material items, floral greens, medicines, berries, and other foods—

A wide variety of understory plants are important for maintaining the health, diet, lifeways, and cultural traditions of tribal communities (Lynn et al. 2013, Rogers-Martinez 1992, Turner 2014). Many of these plants produce berries, including huckleberries (fig. 11-2), cane fruits and brambles (*Rubus* spp.), elderberries (*Sambucus* spp.), buffaloberries (*Shepherdia* spp.), strawberries (*Fragaria* spp.), and serviceberry/saskatoon berries (*Amelanchier alnifolia*) (Kellogg et al. 2009, Turner and Turner 2007).

Several species of huckleberries, especially *Vaccinium membranaceum, V. deliciosum*, and *V. ovatum*, have historically been and today remain a prominent first food and trade item for many tribes across the NWFP area (Deur 2009, LeCompte-Mastenbrook 2016, Mack and McClure 2002). Some of these huckleberry species have yielded substantial market values for their berries or foliage. The production of huckleberries from a good site near Mount Adams in Washington state was reported to be as much as 100 gal/ac (935 L/ha), with a value of $11/gal ($2.90/L) suggesting an estimated value of $1,100/ac ($2,700/ha) in 1977 (Minore and Dubrasich 1978). Arnette and Crawford (2007) reported that wholesale prices in 2007 were about $18/gal ($4.76/L) (which is within the range of prices in the U.S. Forest Service Pacific Northwest (Region 6) special forest products appraisal system database). These figures indicate that huckleberry production can be valued at several thousand dollars per acre or hectare. For many decades, the high socioeconomic value of these berries to tribal members has been recognized, along with conflict with commercial harvest by non-American Indians (Carroll et al. 2003, Hansis 1998, Richards and Alexander 2006). However, there has been untapped potential for land management to enhance the productivity of such resources to support multiple benefits (Von Hagen and Fight 1999), including enhanced suitability for tribal harvest.

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Figure 11-15—Lillian Rentz (left) harvesting cluster-lilies (*Brodiaea coronaria*) with LaVerne Glaze near Somes Bar, California, July 2006.

A variety of understory plants provide important material for making baskets and many other traditional items, including willows (*Salix* spp.), sedges (*Carex barbarae* and *C. obnupta*), cattails (*Typha latifolia*), tule (*Schoenoplectus* spp.), dogbane (*Apocynum cannabinum*), and many others. Salal is an important shrub harvested by tribal members for edible berries and medicine, and workers from many ethnic groups also harvest it commercially for the floral greens industry (Ballard et al. 2008). Many geophytes, including camas (*Camassia* spp.), cluster-lilies (*Brodiaea* spp.) (fig. 11-15), biscuit roots (*Lomatium* spp.), onions (*Allium* spp.), and lilies (*Lilium* spp.), are important traditional foods. Improving camas production was the goal for prescribed burning as part of the Camas Prairie Restoration Project in prairie habitat on the Willamette National Forest, Oregon (Nabhan et al. 2010, Smith and Farque 2001). Tribal harvesters use the leaves of another important geophyte, beargrass (*Xerophyllum tenax*) (fig. 11-16), to make baskets and tribal regalia items; treatments to promote those uses have been the subject of joint Forest Service and tribal partnerships (Hummel et al. 2012, Shebitz et al. 2009a).

Many understory plants are associated with disturbances, such as fire, that create or maintain canopy gaps and open understory environments. Canopy gaps allow light to reach the understory, and burning often promotes characteristics desired by harvesters, such as long, supple stems, larger roots, and increased fruit production, as well as ease of access for harvesting. For example, research indicates that tribal harvesters prefer beargrass from stands with fewer, larger trees and less down wood, which are conditions that can be promoted through thinning and frequent fire (Hummel and Lake 2015). However, such

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Figure 11-16—LaVerne Glaze holding harvested beargrass (*Xerophyllum tenax*), July 2005.

relationships can vary greatly among closely related species. For example, Kerns et al. (2004) found that red huckleberry (*Vaccinium parvifolium*) foliage growth (not necessarily fruiting) would likely benefit from thinning young conifer stands. Similarly, Keyes and Teraoka (2014) found red huckleberry to be more dominant in second-growth than old-growth redwood stands in northern California. On the other hand, the more shade-tolerant evergreen huckleberry (*V. ovatum*) appeared more abundant in closed-canopy forests along the Oregon coast in a study by Kerns et al. (2004). In addition, Halpern and Spies (1995) had found that cover and frequency of big huckleberry (*V. membranaceum*) were greater in old-growth or mature forest stands in the Cascade Range of Washington. While speculating that thinning in such stands might cause declines in vegetative growth, Kerns et al. (2004) cautioned that they were unable to determine a relationship between stand condition and fruiting patterns, and they concluded that more site-specific investigations informed by tribal harvesters would improve understanding of favorable management practices. While forestry and botanical research typically evaluate vegetative abundance, tribal harvesters evaluate additional characteristics that affect harvest suitability, such as fruit abundance, size, and taste, when recommending management strategies for particular stands.

Wetland plants—

Tribes have valued and tended several kinds of plants found in wetlands, including tules, cattails, sedges, willows, and wapato (*Sagittaria* spp.). Tribes in Oregon and on the Olympic Peninsula in Washington have long harvested small cranberry (*Vaccinium oxycoccos*) from bogs and used fires to deter encroaching trees and shrubs and to stimulate the plants to produce more fruit (Anderson 2009). Bog Labrador tea (*Rhododendron groenlandicum*) is another species used for tea and medicine that has a similar history of tribal burning; the plant resprouts from stems following low-intensity fires and from deep rhizomes following more severe fires (Anderson 2009). Klamath tribal members cultivated marsh edge areas to harvest seeds from the yellow pond lily (*Nuphar polysepala*) (Deur 2009). Another example is the rare western lily (*Lilium occidentale*), a threatened species. As described in a 5-year status report (USFWS 2009), this plant is endemic to the coast of northern California and southern Oregon, where it occupies freshwater wetlands, coastal prairie and scrub, and the edges of Sitka spruce forests. Declines in habitat quality for the species have been linked to reductions in tribal burning and ungulate grazing, which historically was provided by elk but for which cattle can be a useful surrogate (USFWS 2009). Imper (2016) asserted that within the coastal region, disturbances such as grazing and burning are important to deter encroachment by sedges and conifers into open wetland habitats that support populations of the rare lily, along with many other rare plants.

Fungi and lichens—

Many species of fungi are important sources of food, medicine, and income for tribal members, including matsutake (*Tricholoma magnivelare*), morels (*Morchella* spp.), chanterelles (*Cantharellus* spp.); hedgehogs (*Hydnum* spp.), boletes (*Boletus* spp.), *Hericium* spp., and oyster mushrooms (*Pleurotus* spp.) (fig. 11-3) (Anderson and Lake 2013). Many of these species produce fruiting bodies following fire and other disturbances to trees and soils (Anderson and Lake 2013). For example, recent research outside the NWFP area found that profuse morel production in the first year following the Rim Fire in the Sierra Nevada mountains could sustainably support "relatively liberal harvest limits" (more than 4 L/day) by recreational and subsistence harvesters (Larson et al. 2016). Wila or horsehair lichen (*Bryoria fremontii*), which scientists recently described as a "macrolichen symbiosis" of a fungus, algae, and yeast (Spribille et al. 2016) is a "first food" for tribes particularly in the interior of the Pacific Northwest, where edible varieties have been harvested from forests and distinguished from the inedible ones using tribal traditional knowledge (Crawford 2007). Chapter 6 includes more information about responses of fungi and lichens to fire and management.

Tribal ecological knowledge systems—
Tribal cultures across the NWFP area constitute a great diversity of languages, knowledge systems, practices, and traditions that reflect the ecological diversity of their ancestral territorial homelands (Kroeber 1920, Suttles 1990, Turner 2014). Many parts of the region, such as the Klamath Mountains, have distinctive mixes of species and ecosystems that also occur in the Coast Range, Cascade Range, and California provinces. Tribal knowledges reflect similar mixes, as tribes of northern California have cultural knowledge and practices of species that extend from the Great Basin, Pacific Northwest, and California floristic biomes (Kroeber 1920). Meanwhile, tribes along the lower Columbia River depend upon and have knowledge of plants and animals found both in the Pacific Northwest and on the Columbia Plateau. Tribal knowledge systems have evolved with an understanding of conditions across bioregions and habitats, which makes them particularly valuable for informing adaptation. Maintaining these knowledge systems requires managing resource conditions and accessibility through applications across large geographic areas (Dobkins et al. 2016, Trosper 2003, Turner et al. 2003).

Although tribes living in similar environments may represent different language groups, they tend to exhibit similarities in cultural practices. For example, tribes along the coast from British Columbia to northern California used and still depend upon many similar resources (Suttles 1990). Although many tribes use the same species for similar purposes, their stewardship methods differ based on culturally specific knowledge and customs, as illustrated in the case of Pacific lamprey (Close et al. 2004, Petersen Lewis 2009). Similarly, all the tribes have rich basket weaving traditions, and many use primarily the same few species, such as hazel and beargrass as central components. However, just as tribes have distinctive weaving techniques and designs, they also have distinctive cultivation and harvesting practices (Hummel et al. 2012, Hummel and Lake 2015). The distinctions in how tribes use and manage forest resources are important for planning, prioritizing, and implementing strategies for managing large landscapes (Stumpff 2006), as each interested tribe may have specific values attributed to particular places. Tribal knowledge can guide and inform resource management for a suite of similar habitats and species, but specific prescriptions and treatments may be needed to promote desired conditions for specific sites. For example, many tribes may want to use fire within a landscape, but they may have different approaches regarding the timing of burning in particular habitats (see "Reestablishing fire regimes" on p. 885). Consequently, consultation, coordination, and communication by federal agencies with individual tribes is important to address landscapes, habitats, and species of interest, rather than expecting that generalized prescriptions will serve the needs of all tribes in an area (Raish et al. 2007).

What Factors Are Influencing the Quality and Availability of Tribal Ecocultural Resources?

Factors influencing the availability of ecocultural resources range from harvesting rights to biophysical factors that influence the quality and quantity of production. The periodic monitoring reports under the NWFP (Harris 2011, Stuart and Martine 2005, Vinyeta and Lynn 2015) considered how tribes evaluated the accessibility and condition of important resources and places on public lands. While devoting much attention to legal and bureaucratic constraints, the reports also discuss how competition with non-Indians affects their capacity to obtain desired resources. While those reports note that some tribal respondents regard the NWFP as having improved the condition of terrestrial and aquatic ecosystems by providing protections for old-growth forest and aquatic habitats, many also note that fire suppression and strict preservation approaches linked to the NWFP have inhibited restoration of conditions desired by tribes. Similarly, research by LeCompte-Mastenbrook (2016) recounts how members of the Muckleshoot Tribe regard the institutionalization of "minimal disturbance" under the NWFP as having had negative effects on tribal ecocultural resources such as huckleberries and elk. Such perceptions are consistent with trends discussed in chapter 12, namely, that the Plan has encouraged managers to limit intentional disturbance while extending the legacy of fire suppression, which has led to reduced composition and productivity of many resources that are not favored by dense conifer forests. Beyond the accessibility and productivity of ecocultural resources, tribal members are also concerned about obstacles to applying tribal stewardship practices themselves on their ancestral lands. Having such opportunities enables them to enhance not only resources, but also traditional ecological knowledge and community capacity.

Changes in tribal socioeconomic conditions and resulting effects—
A broad historical perspective is helpful for understanding how changes in the lands and waters are associated with changes in the well-being of the indigenous peoples of the Pacific Northwest. While tribes throughout the region have maintained close connections to land, many of them underwent a shift from subsistence to market-based economies by the start of the 20th century. During that shift, many tribal members sought employment in regional fisheries as well as agriculture- and timber-based industries (Mondou 1997). Tribes and their members have long faced challenges in attempting to maintain both economic security and traditional cultural practices. Many tribal economies remain strongly linked to forest industries and management through activities such as harvesting timber and nontimber forest products, firefighting, and positions with land management agencies. As employment in timber and fishing industries have declined, tribal members have relied on more restoration-based jobs or harvesting of nontimber forest products (MacKendrick 2009). The Jobs in the Woods Program, set up to mitigate socioeconomic impacts of the NWFP by providing restoration-based jobs for workers from timber-based communities, appeared particularly effective in tribal contexts by supporting effective retraining, valuable jobs, increased economic security, aquatic habitat improvement, and cultural capacity through projects on tribal lands (Harris 2011, Middleton and Kusel 2007).

During much of the 20th century, local tribes had little influence over resource management on federally managed lands for a variety of reasons, including less developed tribal institutions, dismissal of tribal traditional knowledge and concerns, and inconsistent federal recognition and policies (Catton 2016, Record 2008). As noted by many tribes, public lands management during that era, including suppression and punishment for tribal burning and harvesting, engendered considerable distrust of land management agencies, while degrading the quality and quantity of important tribal ecocultural resources (Dobkins et al. 2016, Lake 2013, Norgaard 2014a). Various land management policies, including removal of tribal stewardship, fire exclusion, commercial timber harvest, and protections for threatened species and wilderness areas, have contributed to denying tribes the benefits they derived from ancestral lands, which in turn has depressed tribal community well-being and engagement in forest management (Freedman 2002, LeCompte-Mastenbrook 2016, Norgaard 2014b).

Access for harvesting forest products—
As discussed in the "Federal-Tribal Relationship" on p. 854, some tribes have legal rights to harvest various forest products from public land areas. More generally, the Farm Bill of 2008 authorized the Secretary of Agriculture to provide any trees, portions of trees, or forest products to Indian tribes free of charge for "traditional and cultural purposes," for which the Forest Service adopted a final rule on September 26, 2016 (USDA FS 2016). Previously, the requirements for such collections widely varied through time and across the different national forest districts and other

jurisdictions in the NWFP area (Catton 2016). A recent study of the Northwest Native American Basketweavers Association found that American Indian harvesters of special forest products encountered a range of obstacles to harvest on public lands, including gates, closed or poorly maintained roads, requirements for obtaining permits, fees for access, and insufficient support in agreements (Dobkins et al. 2016). Similarly, access to suitable logs to construct river- and ocean-going canoes (fig. 11-14) has been a limiting factor for larger contemporary tribal traditions and celebrations (Johansen 2012). The Quinault Indian Nation reported difficulty in procuring logs from adjacent national forest lands to use in river restoration efforts (Harris 2011). Tribes have faced obstacles in obtaining logs from national forests across the NWFP area owing to limited availability, constraints associated with late-successional reserves and special status and sensitive species, disputes over fees, and other procedural hurdles (Catton 2016, Harris 2011, Vinyeta and Lynn 2015). Some tribal members have criticized various bureaucratic processes associated with obtaining information and approvals or permits to harvest forest products as being unduly burdensome, and some have described the expectation of having to obtain permits as an affront to religious freedom, tribal rights, and other values (Dobkins et al. 2016, Flood and McAvoy 2007). The economic impacts of fees on low-income and minority populations are also discussed in chapter 10. Strategies to address tribal concerns over policies that constrain resource access are discussed further below.

Competition for harvesting nontimber forest products—
Harvesting of nontimber forest products (also known as special forest products) represents a substantial socioeconomic activity in the Pacific Northwest (see chapter 10), with commercial harvest of products such as floral greens and mushrooms valued in hundreds of millions of dollars (Alexander et al. 2011, Von Hagen and Fight 1999). An important practical constraint on tribal resource use has been a limited supply to meet tribal needs (Findley et al. 2001), which reflects environmental degradation as well as competition for that production especially from nontribal commercial harvesters (Dobkins et al. 2016). Competition and outright conflict over nontimber forest products on public lands has occurred between tribal members and nonlocal groups from nontribal minority and low-income populations, especially immigrants from Southeast Asia and Latin America (Charnley et al. 2008a, Hansis 1998). During the early 1990s, tribal concerns over non-American Indian harvest of matsutake mushrooms, particularly by Southeast Asian immigrants from distant urban areas, triggered protests of national forest management of commercial harvest on the Happy Camp district (Richards and Creasy 1996). The researchers explained that such groups had strong incentives to overharvest the resource as they were not likely to recoup the benefit of leaving it, while the tribal harvesters had cultural practices that were more likely to favor sustainability. Hansis (1998) similarly reported that nontribal itinerant groups had disincentives to harvest various resources sustainably across other parts of the NWFP area. As noted in chapter 10, management designed to support commercial harvest and tribal cultural harvest may differ for a number of resources, including beargrass, as the qualities preferred by those groups may differ. Furthermore, the fact that some tribal members harvest products for sale as well as subsistence adds complexity to issues regarding permits and competition. In addition to impacts of nontribal harvesters, recreationalists can also affect tribal hunting, fishing, trapping, plant harvesting, and ceremonies. Various strategies to address nontribal impacts to tribal resource use through seasonal closures or special-use areas are discussed below.

Illegal marijuana cultivation—
Marijuana cultivation on national forests and other public lands has proliferated since the 1990s, especially in northwestern California (Bauer et al. 2015), but increases have also occurred in Oregon and Washington (National Drug Intelligence Center 2007). This activity is merely a subset of a larger problem of illegal activities on public lands that poses concerns for public safety, access, and resources; for example, methamphetamine labs and dump sites also significantly increased since the late 1990s (Tynon et al. 2001). However, the particularly rapid and extensive growth of marijuana cultivation has had widespread social and ecological impacts, including harm to culturally important wildlife species. For example, illness and deaths in fisher

populations in southern Oregon and northern California, including on and around the Hoopa Valley Reservation, have recently been linked to the use of rodenticides in marijuana cultivation (Gabriel et al. 2012). Other researchers found that the rodenticides cause direct or indirect mortality to wildlife species of cultural significance such as black bear, fisher, bobcat (Serieys et al. 2015), owls, and other predators or scavengers that consume rodents laced with the toxic compounds (Hosea 2000, Stone et al. 1999). Additionally, Bauer et al. (2015) found that water diversion associated with illegal marijuana cultivation in several California watersheds negatively affected the health of salmonids and amphibians. Finally, these operations pose safety concerns for forest users and land managers responsible for treating, monitoring, and protecting forests (Tynon and Chavez 2006). Some tribes have expressed safety concerns for tribal harvesters who encounter illegal marijuana cultivation sites on federal and tribal lands.

Climate change—
Changes in climate can potentially jeopardize tribal ecocultural resources, and the well-being of tribal communities more generally, by exacerbating droughts, extreme storms and runoff events, wildfires, and outbreaks of insect pests and plant pathogens (see chapter 2). In addition, rising seas, melting glaciers, and associated flood hazards are affecting tribes in low-lying and coastal areas (Papiez 2009), which increases the importance of federal lands for sustaining tribal communities. As discussed in chapter 2, there is considerable uncertainty regarding how climate, fire, invasive species, and other influences will affect species composition and habitat at fine scales, but climate trends such as reduced water availability in soils and streams are expected to have greater impacts within inland and southern portions of the Pacific Northwest region. Such changes threaten the availability of traditional foods, medicines, and materials to tribes, which in turn can harm diets, health, and other important dimensions of community well-being (Bennett et al. 2014, Lynn et al. 2013). Because tribal communities in the Pacific Northwest are so strongly associated with large rivers and the Pacific Ocean, they can be affected by climate change even well outside of their current lands. Impacts of changing climate are compounded by other stressors, including insect pests, plant pathogens, hydrologic alterations, changes in fire regimes, and increases in tree densities and fuel loads (Pfeiffer and Voeks 2008, Spies et al. 2010). For example, Turner and Clifton (2009) identified examples of declines in amphibians, fishes, forest health, and tribal ecosystem services in British Columbia, adjacent to the NWFP area, which they attributed to changes in climate, intensifying droughts, and outbreaks of insect pests and plant diseases.

When assessing vulnerability to climate change and other stressors, focusing attention on tribal values helps to evaluate threats and identify stressors and needs for adaptation. Tribes have been engaged in a number of initiatives to evaluate vulnerability to climate change and support adaptation actions (see "Tribal Engagement in Climate Change Initiatives" on p. 885). MacKendrick (2009) worked with the Hoopa Valley and Coquille Indian Tribes to evaluate priority concerns regarding vulnerability to climate change, many of which involve transboundary issues with public lands such as wildfire hazard and water quality in shared streams. In cases where Western scientific knowledge of climate-habitat-species relationships is available for species of significance to tribes, they can be crosslinked with tribal knowledge to better forecast and anticipate threats to tribal uses (Turner et al. 2011) and to identify possible refugia (Carroll et al. 2010a, Olson et al. 2012). Various tree species that have special tribal importance have been studied to assess their vulnerability to projected changes in climate. For example, Alaska yellow-cedar and Oregon white oak both rank as particularly vulnerable species (Case and Lawler 2016, Coops and Waring 2011, Hennon et al. 2012). Conversely, California black oak, tanoak, bigleaf maple, and western redcedar appear highly adapted and more likely to expand their ranges under the warmer and more fire-prone conditions that have been commonly predicted (Case and Lawler 2016, Coops and Waring 2011). Tribal members often depend upon large, long-lived trees with particular characteristics to obtain nuts and special wood products. Consequently, predictions of range expansion for important species do not sufficiently gauge the sustainability of ecosystem services for tribal communities.

Species invasions—

Invasive species are affecting the condition of ecosystems within the NWFP area (see chapter 3), and they are also degrading the ability of American Indians to harvest ecocultural resources. Although there are too many to list in this report, specific examples of invasive plants that have degraded tribal gathering areas include Scotch broom (*Cytisus scoparius*), yellow starthistle (*Centaurea solstitialis*), and Himalayan blackberry (*Rubus armeniacus*) (Pfeiffer and Ortiz 2007, Pfeiffer and Voeks 2008, Senos et al. 2006). Tribes have undertaken restoration efforts to combat exotic knotweeds (*Fallopia* spp.) (Harris 2011); those invasive plants can have profound and persistent effects on the structure, functioning, and diversity of riparian forests by displacing native species (Urgenson et al. 2009). Furthermore, legions of invasive fishes, snails, and plants such as purple loosestrife (*Lythrum salicaria*) also negatively affect native salmonids and other native aquatic resources (Sanderson et al. 2009).

The spread of the sudden oak death pathogen (*Phytophthora ramorum*) is having profound implications for ecological processes (see chapter 3) and tribal ecocultural resources in the northern California and western Oregon coastal region. The disease has killed many large tanoak and black oak trees, and it infects many other species of special value to tribes, including California bay laurel (*Umbellularia californica*), California hazel (*Corylus cornuta*), huckleberries (*Vaccinium* spp.), and salmonberry (*Rubus spectabilis*) (Cobb et al. 2012, Ortiz 2008). Although infection does not necessarily kill those understory plants, it reduces their suitability for tribal use owing to lesions and may prompt land managers to remove infected plants, especially California bay laurel, to protect tanoak stands (Swiecki and Bernhardt 2013).

The spread of a closely related pathogen, *Phytophthora lateralis*, has affected populations of the Port-Orford cedar within its range in northwestern California and southwestern Oregon. This riparian species not only holds special ecocultural value but also has high market values and plays an important ecological role, especially on ultramafic soil areas (Hansen 2008). Because roads are an important vector for the spread of the pathogen, road closures have been used to restrict its spread (Hansen et al. 2000). Although intended to benefit forest sustainability, such closures can also affect tribes' ability to access resources.

Species losses—

When cultural keystone species are reduced or eliminated from a tribe's ancestral territory, then the associated cultural traditions, knowledge systems, and material well-being of tribal communities suffer in turn (Colombi 2012). California condor is a tribally important species for which reintroduction within the Pacific Northwest has been considered (Walters et al. 2010). In general, federal land management agencies such as the Forest Service and BLM do not have primary roles in wildlife reintroductions, but they are often cooperators in such efforts by addressing habitat needs for those species.

Some species losses have altered ecosystem functions in ways that land managers consider in designing treatments. For example, recent decades have seen growing interest in the reintroduction of beaver. Ponds formed by beavers provide important habitat for coho salmon (Pollock et al. 2004). Structural treatments designed to mimic beaver dams and facilitate beaver recolonization known as "beaver dam analogues" have been undertaken within the three states of the NWFP area (Pollock et al. 2015). One recent study from Oregon's John Day watershed reported enhancements in steelhead habitat and juvenile growth following placement of such structures (Bouwes et al. 2016). Another example of the potential impacts of species losses and reintroductions involves top predators such as wolves. The gray wolf was extirpated in the Pacific Northwest, but populations have returned to parts of the region owing to efforts led by the Nez Perce Tribe (Donoghue et al. 2010). Beschta and Ripple (2008) suggested that reintroduction of wolves could have cascading influences on ecosystems in the Pacific Northwest. Their work built upon extensive research in Yellowstone National Park's Lamar Valley, where they contend that removal of wolves triggered an increase in elk herbivory on woody riparian plants, which in turn contributed to streambank erosion, channel incision and widening, and loss of wetlands and beaver habitat (Ripple and Beschta 2004). Along several rivers of Olympic National Park, where elk hunting is prohibited, they found reduced recruitment of black cottonwood and bigleaf maple, as well as greater channel braiding and bank erosion, as compared to riverine sites within the Quinault Indian Reservation where humans have continued to hunt elk. As a result, their analysis not only suggests possible ecological effects of removing or reestablishing wolves, but also suggests that predation by American Indians had important effects on the dynamics of those riverine systems. Reinforcing that point, Hutchings and Campbell (2005) contended that American Indian hunters influenced the vegetation and morphology of riparian-aquatic environments such as deltas of large rivers such as the Nooksack in Washington by altering ungulate and beaver populations. While hunting and management of wildlife populations are generally not under the purview of national forest managers, an understanding of these dynamics is important for understanding historical conditions and restoration strategies.

Alterations of hydrologic regimes—
Changes in hydrologic regimes resulting from past land use practices include decreases in low flow, increases in peak flow, and increases in water temperature (Beechie et al. 2013). Under warming climates, reduced snowpack, loss of glaciers, and increased rain-on-snow are expected to intensify those impacts, with negative consequences for coldwater fishes such as salmon and trout (Abdul-Aziz et al. 2011). Habitat fragmentation and elevated water temperatures have had a great impact on salmon fisheries (Coates 2012). Tribes are concerned about the threats such impacts pose to anadromous fishes that are critically important to many tribes' traditions and livelihoods (Dittmer 2013). Because reservoir dams are a leading cause of altered hydrology throughout the NWFP area, removal of such dams has become an important restoration strategy and subject of research (see "Removing reservoir dams" section on p. 890). Other hydrological alterations include intentional draining of wetlands that formerly sustained important ecocultural resources (Deur 2009).

Alterations of fire regimes—
Wildland fire affects the physical, biological, and sociocultural components of landscapes in ways that can benefit or damage tribal ecocultural resources. Fire has cascading effects, beginning with direct combustion and heating that can damage sites or resources, and extending to second-order physical effects such as soil erosion following severe fires, as well as third-order impacts to cultural values, which can result from tangible and intangible resource change, loss, or damage (Ryan et al. 2012). Tribal members often have strong concerns about the threat of wildfire to their lands (MacKendrick 2009). Fire management activities themselves, such as fireline construction (mechanically and manually) that results in physical removal or modification of vegetation and soil, can also degrade tribally valued resources (Timmons et al. 2012, Welch 2012). Tribal members have also cited instances when fire retardant applied aerially during wildfire fighting has affected harvesting areas (Norgaard 2014a). Retardants contain fertilizing chemicals that can cause eutrophication and fish toxicity when entering waterbodies; studies have suggested that they have very low toxicity to human firefighters and birds but can irritate eyes, skin, and respiratory tracts (Giménez et al. 2004, Kalabokidis 2000, Vyas et al. 2009). Although impacts from fire management are important concerns to tribes, advance planning in consultation and collaboration with tribes to prevent and manage wildfires can reduce the potential for harm to tribal ecocultural values by identifying favorable control strategies and tactics within particular landscapes (Ryan et al. 2012). Such efforts are currently the focus of the Western Klamath Restoration Partnership (see box on p. 888) in the southern portion of the NWFP area.

Fire regimes in many regions, especially dry forests but also in some wetter coastal environments, have been altered by frequent suppression of lightning fires and reductions in aboriginal burning (Boyd 1999, Kimmerer and Lake 2001, Skinner et al. 2009) (see also chapter 3). Tribal members also have stated that their ability to harvest forest products such as acorns, berries, beargrass, and hazel has declined owing to reduced resource quality, quantity, and accessibility, which they often attribute to lack of frequent fire and tribal stewardship as well as other changes in forest management, such as establishment of tree plantations (Charnley et al. 2008a, Dobkins et al. 2016, Halpern 2016, Long et al. 2016a). Lack of fire-associated forest products has reduced the quality of life for American Indians who depend on those resources (Norgaard 2014a).

Fire exclusion along with changing climate appears to be increasing the likelihood of very large fires (Stavros et al. 2014), which tend to have large stand-replacing burn patches (Miller et al. 2012, Reilly et al. 2017). Severe burns in turn threaten tribal ecocultural resources associated with mature trees and archaeological sites (such as rock art and obsidian artifacts) that can be particularly sensitive to high-intensity fire (Ryan et al. 2012). Fuel accumulations under fire exclusion have complicated efforts to reintroduce fire without risking such losses.

Changes in stewardship regimes—

Historical tribal stewardship practices that include plant harvesting, tilling, weeding, pruning, moving plant propagules, burning, raking debris, removing fuels, and hunting have been displaced and altered throughout ancestral tribal lands of the NWFP area (Anderson 2005, 2009; Deur 2009, LeCompte-Mastenbrook 2016). These practices affected ecosystems from patches to landscapes, and they evolved into a complex agroforestry system that tribes have used to maintain the quality and availability of ecocultural resources (Anderson 2005, Rossier and Lake 2014, Turner and Bhattacharyya 2016, Turner et al. 2013). Consequently, the disruption of traditional practices has perpetuated a cycle of degradation with various elements:

- Displacement of tribes from ancestral lands through confinement onto reservations was followed by land allotment and termination, which limited tribes' ability to practice land-tending traditions such as burning.
- Resource quality and quantity has declined.
- Areas are no longer suitable for harvesting desired foods.
- Community members suffer poorer health as well as food and economic insecurities.
- Intergenerational transmission of traditional ecological knowledge is impeded as elders have fewer opportunities to practice the traditions and teach them to youth, as well as reduced incentive to do so.
- Lands become feral and inhospitable "wilderness" (Anderson 2005).
- People's understanding of reference conditions becomes distorted as experience with past conditions is replaced by exposure to present degraded conditions, or "shifting baseline syndrome" (Papworth et al. 2009).

These effects further deter tribal members from reestablishing traditional practices. The elements of this cycle of degradation are described in several published studies that refer in particular to public lands within various parts of the NWFP area (Anderson 2005, Deur 2009, LeCompte-Mastenbrook 2016, MacKendrick 2009, Norgaard 2014c, Richards and Alexander 2006, Shebitz 2005, Wray and Anderson 2003). Understanding these patterns is important to avoid falsely assuming that a lack of present-day attempts to harvest resources indicates a lack of interest. All the other stressors discussed in this section have exacerbated this cycle by reducing the availability of ecocultural resources or constraining access by tribal members, as noted in tribal vulnerability assessments across the NWFP area (Donatuto et al. 2014, MacKendrick 2009, Sloan and Hostler 2014).

Implementation of policies since the Northwest Forest Plan—

During the initial development of the NWFP, many tribes did not contribute directly to the preparation of the alternatives, and the Bureau of Indian Affairs (BIA) represented tribal interests to the Forest Service. However, federal-tribal collaboration on land and resource management has evolved considerably in recent decades as laws and policies have developed; as tribes' political, economic, and sociocultural capacity has burgeoned; as agencies have increasingly appreciated tribes' knowledge about forest management; and as agencies have invested more in tribal liaison positions (Breslow 2014, Catton 2016, Record 2008). Tribes have increased the capacity of their natural resource institutions, in many cases using authorities provided by the 1975 Indian Self-Determination and Education Assistance Act (Pub. L. 93-638) and the Tribal Self-Governance Act of 1994 to assume control over natural resource programs that were previously overseen by the BIA (Catton 2016, Strommer and Osborne 2014). In addition, significant progress has been made in developing institutional platforms to address sensitive issues regarding resource management on federal lands (Jurney and Hoagland 2015).

Despite such advances, tribes have criticized some federal attempts at consultation since the NWFP as little more than notification of planned federal actions, followed by unilateral decisionmaking and inadequate attention to resolving disputes (Harris 2011, Vinyeta and Lynn 2015). In addition, tribes have expressed concerns that special designations have limited forest thinning, and that public lands management has inhibited use of fire more generally. Tribal members have contended that management under the NWFP has allowed declines in important tribal ecocultural resources (e.g., elk, huckleberries, beargrass, and black oaks) as a consequence of measures to avoid possible harm to late-successional forests, riparian reserves, the northern spotted owl, and various survey and manage species (Harris 2011, LeCompte-Mastenbrook 2016, Vinyeta and Lynn 2015). That concern appears generally consistent with findings described in chapter 12 and elsewhere in this report. Researchers studying public lands management in the United States have noted the tensions between addressing specific statutory requirements under the Endangered Species Act with strategies designed to promote landscape-scale resilience (Benson and Garmestani 2011) or tribal self-determination (Schmidt and Peterson 2009). A special case of this general issue is the Quinault Special Management Area, a 5,460-ac (2210 ha) area of forest land managed by the Forest Service that was established as partial compensation for the loss of territory that was supposed to have been included in the Quinault Reservation. The tribe has a right to 45 percent of the revenue generated in this special area, but constraints for Survey and Manage species have reduced harvests and revenues below what the tribe expected under this arrangement (Vinyeta and Lynn 2015). Another special case is the Coquille Indian Tribe, to whom Congress transferred lands but with the requirement that NWFP rules be applied to forest management (see “Coquille Indian Tribe” on p. 882).

How Has the Diminishment of Tribal Influence Affected Ecocultural Resources and Associated Ecosystems?

Understanding historical tribal practices for stewarding ecosystems is important for restoring conditions that sustain biophysical and cultural ecological services important to American Indians and tribes (Turner et al. 2013). In the sections below, we highlight how diminishment of tribal influences within the NWFP area has reduced the frequency and extent of low-intensity fire and, consequently, the availability of many species of high cultural-use value. Such shifts have far-reaching implications, yet we must also consider uncertainties in our understanding. Complex dynamics within coupled human-ecological systems make it difficult to understand and study the myriad potential effects of these influences over millennia. Much past research relied upon single-disciplinary approaches in ecology or ethnography, with or without tribal perspectives or information, which can lead to findings that appear inconsistent or conflicting. Interdisciplinary approaches that integrated multiple lines of evidence have led to greater consilience about where indigenous influences were most profound and where current conditions have deviated most sharply from conditions prior to Euro-American colonization (Crawford et al. 2015, Lightfoot et al. 2013). Furthermore, engaging tribes in research efforts has helped in our understanding of historical cultural influences on ecosystems (Lepofsky and Lertzman 2008).

Broad-scale fire history studies in the Pacific Northwest region have found American Indian influence on fire to be associated with climate and population density. For example, Agee (1993) concluded that evidence for large-scale American Indian burning was greater in inland areas, with much patchier burning in wetter coastal environments. Perry et al. (2011) found that American Indian burning likely shifted mixed-severity fire regimes to more frequent, low-severity fire regimes in areas with dense populations of American Indians, such as northern California and the Umpqua National Forest. Many sampling methodologies lack the resolution to recognize or distinguish human influence on fire regimes (Conedera et al. 2009). Consequently, studies of fire history sometimes subsume American Indian influences under the natural regime (e.g., Halofsky et al. 2011). The analysis used to develop the map of fire regimes in chapter 3 revealed that the fire frequencies in coastal forests of northern California before Euro-American settlement were higher than expected based upon temperature and moisture factors across the NWFP area. This finding indicated that historical American Indian influence on fire regimes was particularly significant within that region.

Scientists have published extensive evidence regarding how tribal burning and other practices modified vegetation within small patches; however, larger scale, longer term effects are more difficult to elucidate (Lepofsky and Lertzman 2008, Turner et al. 2013). Lewis and Ferguson (1988) described both areal "yards" burned by American Indians as well as linear "corridors" associated with streams, trails, and ridges. The maintenance of such corridors and yards would have promoted heterogeneity and connectivity for access by humans, ungulates, and other species at multiple scales (Lake 2013, Storm and Shebitz 2006, Turner et al. 2011). However, there remain questions regarding how much human influence modified fire regimes and vegetative communities beyond areas of intensive activity such as village sites, camps, harvesting and processing sites, and major trails (Lake 2007, 2013). Evidence of past caretaking by American Indians, including fire scars, culturally modified trees with bark selectively removed for use, and artifacts and features associated with resource processing serves to identify culturally modified landscapes (Turner et al. 2009). However, many decades of displacement and land use by Euro-Americans have obscured much of the evidence of such activities (Turner et al. 2013), in particular by developing the areas of greatest influence by American Indians. For example, Zybach (2003) in his dissertation concluded that areas of the Oregon Coast Range that were most likely subjected to regular burning by American Indians have been extensively developed, while areas that burned less frequently and more intensely have been maintained as forests by corporations, states, and federal agencies.

Hardwood communities and old trees—
American Indians have cultivated a variety of hardwood communities, including California black oak (Long et al. 2016a), Oregon white oak (Lepofsky and Lertzman 2008), Pacific madrone (*Arbutus menziesii*), and tanoak (Bowcutt 2013). Areas near hardwood woodlands have long been favored for human settlements in the Pacific Northwest, but these areas have been reduced in extent and degraded in quality by fire exclusion, land development, conifer encroachment, and exotic invasive species, in addition to reductions in tending and burning by American Indians (Hosten et al. 2006). Stands of old-growth hardwoods have similarly declined within conifer-dominated forests, owing especially to the lack of low-intensity fire (see chapter 3) (Cocking et al. 2012, Devine and Harrington 2006). Traditional tribal activities in many woodlands and forests include frequent use of low-intensity fire to support harvest of nuts and desired understory species (Huntsinger and McCaffrey 1995, Long et al. 2016a). By reducing fuels and stand densities, such practices may have extended the longevity of trees, especially oaks and sugar pines, which were key resources (Anderson 2005). Genetic study of the Pacific crabapple suggests that American Indians may have had a key role in distributing it across the region (Routson et al. 2012), and tribal elders have recounted how Euro-American colonization reduced tribal orchards of the species (Turner and Turner 2008).

Grasslands, meadows, wetlands, and forest gaps—
Nonforest communities that are dependent on fire to persist are important to sustaining tribal ecocultural resources. Even regions dominated by wet forests with an infrequent, high-severity fire regime had areas that were burned by American Indians more frequently than what occurs today (Boyd 1999). For example, burning by American Indians maintained bogs, prairies, and balds within areas otherwise dominated by high- and mixed-severity fire regimes, including the northwestern (Anderson 2009, Wray and Anderson 2003) and southeastern parts of the Olympic Peninsula in Washington (Peter and Shebitz 2006), redwood forests in northwestern California (Underwood et al. 2003), and the Coast Range in Oregon (Zald 2009).

Similarly, within the Willamette Valley, researchers have found that evidence of increased fire was positively associated with periods and areas of greater American Indian habitation, including more open environments that support key resources such as oaks, berries, and camas (Walsh et al. 2010). Grasslands and meadows have been declining across the region owing to reduction of aboriginal burning, changing climate, and other factors (Zald 2009) (also see chapter 3). Evidence such as a lack of biological legacies (i.e., large woody debris, stumps, snags, and remnant trees), dominance by graminoids rather than shrubs, and presence of disjunct and endemic plant species suggests that many of these communities were persistent, not an ephemeral, early-successional stage (Zald 2009). A description of practices by the Tolowa, Yurok, Karuk, Tututni, and Wiyot within redwood-dominated forests in northern California and southern Oregon indicated that human-created forest clearings were small, with the largest being only 0.25 mi (0.4 km) wide, and located in resource-poor parts of the landscape (Lewis and Ferguson 1988). Similarly, the abstract for Wills and Stuart (1994) summarized pre-Euro-American conditions in Douglas-fir-dominated stands within the Klamath National Forest as "exceptionally patchy, containing complex mosaics of different age and size." This patchy configuration was actively maintained through frequent fire. One forest surveyor described the entire Klamath River reservation belonging to the Yurok Tribe as being "over-run by fire" in 1912, when the U.S. government authorized rewards for stopping "incendiarists" responsible for setting those fires (Huntsinger and McCaffrey 1995). The ensuing era of fire suppression has reduced the occurrence of high-severity, stand-replacing fire, especially in moist forests, as well as low-severity fires, especially in dry forests (Miller et al. 2012, Reilly et al. 2017); these changes in fire regime have inhibited both the establishment and maintenance of early-successional or nonforest communities (see also chapter 3). For example, research by Peter and Shebitz (2006) within the southeastern Olympic Peninsula (Skokomish River Basin) indicated that ecosystems there had openings ranging from about 0.1 ha to many hectares, with few snags or down logs, in aerial photos from 1929, prior to any timber harvest. These conditions suggested that these openings had been maintained by tribal burning, and that lodgepole pine (*Pinus contorta*) and Douglas-fir had encroached into them starting over a century ago as a result of fire exclusion. Anzinger (2002) similarly described lodgepole pine encroachment into huckleberry meadows that had previously been maintained by tribal burning on the Mount Hood National Forest in the Oregon Cascade Range.

These nonforest communities support a range of tribally valued resources, including elk (*Cervus elaphus*) and deer (*Odocoileus* spp.); berries; edible geophytes; brackenfern (*Pteridium aquilinum*); and many other plant, fungi, and wildlife species (Huntsinger and McCaffrey 1995, Lepofsky and Lertzman 2008, Lewis and Ferguson 1988, Norton 1979). Wildfire and tribal burning have supported biodiversity by deterring homogenization through encroachment by dominant species, facilitating reproduction and vegetative persistence of rarer species, and maintaining hydrologic and nutrient cycling (Anderson 2009, Turner et al. 2011, Wray and Anderson 2003, Zald 2009). For example, tribal burning deterred trees from encroaching on open bog habitat that support cranberries and swamp gentian (*Gentiana douglasiana*); those plants in turn are key foods for the rare Makah copper butterfly (*Lycaena mariposa charlottensis*) (Larsen et al. 1995, Wray and Anderson 2003). Similarly, the range of the Puget blue butterfly (*Icaricia icarioides blackmorei*) has declined with losses of forest gaps and lowland prairies that support its host, sickle-keeled lupine (*Lupinus albicaulis*) (Larsen et al. 1995). Regular burning of meadows maintained the abundance and desired qualities of culturally important species, including various berries (*Vaccinium* spp., *Rubus* spp., etc.) and beargrass for traditional food and basketry uses (Peter and Shebitz 2006, Turner et al. 2011). The steep reduction in burning has caused conversion of grasslands to forested environments (Peter and Shebitz 2006, Zald 2009). The combined losses of former grassland areas owing to forest encroachment and land development have greatly diminished their socioecological benefits to tribal communities (Breslow 2014).

What Strategies Can Promote Tribal Ecocultural Resources and Effectively Engage Tribes in Forest Planning and Management?

Developing institutional capacity and agreements—
Tribes have had increased opportunities to influence management on national forests through agreements, compacts, and stewardship contracts under the Tribal Forest Protection Act and related authorities (McAvoy et al. 2005, Murphy et al. 2007). Examples of some of these agreements are featured in the "Promoting collaboration" section below. Donoghue et al. (2010) characterized different types of tribal-federal collaborative agreements, ranging from less formal working agreements to mutually dependent comanagement in which tribes participate in management decisions. Through these institutional arrangements, many tribes have greater capacity to actively engage in research, planning, and management to support collaborative landscape restoration efforts (Catton 2016, Vinyeta and Lynn 2015).

Addressing sacred sites protection and access—
Progress in federal-tribal relations has occurred despite several major disputes in recent decades in which federal land and water management decisions supported roads, mountaintop developments, and reservoirs. Such decisions were made despite tribal protests and lawsuits under the American Indian Religious Freedom Act of 1978 (P.L. 95-341) regarding the impacts of such developments on tribal sacred sites and religious values (Erickson 2009, Welch 1997). In 1996, Executive Order 13007, "Indian Sacred Sites," directed federal agencies to accommodate tribal access to and ceremonial use of sacred sites. Since then, Congress has passed legislation for specific areas to protect tribal access for traditional religious and cultural purposes through measures such as temporary closures to exclude nontribal visitors and restrictions on land use (Nie 2008). An example is the Northern California Coastal Wild Heritage Wilderness Act (P.L. 109–362) of 2006, which designated wilderness areas on the Mendocino and Six Rivers National Forests within the NWFP area with such stipulations. In addition, the departments of Agriculture, Energy, the Interior, and Defense, along with the Advisory Council on Historic Preservation, jointly adopted a memorandum of understanding (MOU) in December 2012 to improve the protection of and tribal access to American Indian sacred sites (USDA Office of Communications 2012).

Ensuring meaningful consultation—
The NWFP federal-tribal monitoring reports illustrate the importance of MOUs and memorandums of agreement (MOAs) to formalize consultation protocols and strengthen government-to-government relationships. For example, Vinyeta and Lynn (2015) found that such agreements clarify expectations and result in greater accountability in consultations by specifying how often federal-tribal meetings would occur, and who is to be involved in the meetings. They also found that such agreements provide opportunities for greater tribal participation in agency planning and decisions. Drawing on interviews with 27 tribal natural resources staff from within the NWFP boundary, they found that consultation is more effective when it includes formal protocols that are individualized to each tribe's unique needs, laws, practices, policies, and responsibilities to membership. That report includes recommendations for strengthening consultation, addressing tribal rights and access to cultural resources, and improving the compatibility of federal and tribal approaches to forest management, including the development of protocols for projects that involve traditional knowledge.

Promoting collaboration—
National forest planning has increasingly emphasized collaborative approaches, and experts have emphasized the value of participatory approaches throughout the life of projects, including research, monitoring, planning, implementation, maintenance, and review (Charnley et al. 2014). These trends generally complement tribal interests, while recognizing that tribes have a unique relationship with federal land management agencies. Intentions to promote collaborative relationships between federal agencies and communities that have been historically marginalized, including tribes, need to consider legacies of mistrust and inequity (Cronin and Ostergren 2007). Encouraging tribal participation in the full life cycle of projects can facilitate cooperation, trust, knowledge reciprocity, and

accountability. Facilitating development and retention of staff with good understandings of tribal relations is also important, because staff turnover is commonly cited as an obstacle to encouraging vibrant partnerships (Bussey et al. 2016, Vinyeta and Lynn 2015). The success of several tribal programs supported by Jobs in the Woods funding demonstrates the opportunities to jointly address social, ecological, cultural, and institutional objectives in forest and watershed restoration (Middleton and Kusel 2007).

Tribes have expanded efforts to influence ecosystem conditions through a variety of formal partnerships to address climate change, watershed and fisheries restoration, hazardous fuels reduction/forest thinning, and landscape forest restoration (Senos et al. 2006). Federal policies, authorities and directives, including the National Fire Plan (2000), Tribal Forest Protection Act (2004), Healthy Forest Restoration Act (2005), and Federal Land Assistance, Management and Enhancement Act (2010), have encouraged tribal participation in Forest Service land management activities. Concurrently, several notable community-based efforts, such as watershed and fire safe councils in northern California and southern Oregon (Senos et al. 2006), and nongovernmental organizational programs (e.g., The Natural Conservancy's Fire Learning Network), have supported tribal participation in restoration- and conservation-based efforts in the Pacific Northwest. Many of these efforts started as habitat or species-specific projects but grew into larger collaborative restoration partnerships with tribes as co-leaders (Cronin and Ostergren 2007). Some collaborative efforts have guided management and policy based upon the integration of tribal traditional knowledge and Western science (see "Coquille Indian Tribe" on p. 882). Another example is the Tapash Sustainable Forest Collaborative, in which the Yakama Nation has collaborated with the Okanogan-Wenatchee National Forest, Washington Department of Natural Resources, Washington Department of Fish and Wildlife, and The Nature Conservancy. The collaborative has planned and undertaken a variety of restoration projects on portions of forest land within 1.63 million ac (660 000 ha) managed by various entities (including tribes) in central Washington (Schultz et al. 2012, Urgenson et al. 2017).

Fostering cooperative management—
An important pathway for upholding and respecting tribal sovereignty, treaty rights, and culture is cooperative management of off-reservation lands and resources, which may also be described as "concurrent" or "collaborative" management or "co-management" (Diver 2016). These terms apply to varying degrees of tribal and federal influence on land management in an area (Nie 2008); however, a recent definition of co-management adopted by the U.S. Fish and Wildlife Service (see "Glossary") requires each entity to have legally established management responsibilities. For example, treaties that reserve the right to manage or control access to natural resources constitute a legal authority for co-management (Goodman 2000). A strong legal basis has been important in making co-management initiatives between tribes and state agencies focused on salmon particularly successful in conserving resources in the Pacific Northwest (Kellert et al. 2000).

Proposals for co-management between the Forest Service and tribes have had to address legal requirements for federal agencies to have final decisionmaking power over federal lands (Nie 2008). Federal decisionmakers have been concerned about creating expectations that collaborators will have a say in management decisions while retaining responsibility for those decisions, as well as in negotiating procedural requirements associated with advisory groups (Butler 2013). In Canada, and especially in British Columbia, there have been examples of devolving some management authority over public lands to local communities under the umbrella of "community forestry," and many of those involved co-management with indigenous communities (Charnley and Poe 2007, McCarthy 2006). There are also examples of community forests established by tribes through acquisition of private lands, such as the Yurok Tribe's acquisition of ancestral tribal lands along Blue Creek from the Green Diamond Resource Company in 2011. However, such designations have not been adopted for Forest Service lands (Charnley and Poe 2007, McCarthy 2006). Some environmental groups have resisted community forestry initiatives on public lands in the United States over concerns that such efforts would favor local timber industries, undermine environmental protections, and limit public input (McCarthy 2006).

Coquille Indian Tribe

The Coquille Indian Tribe has reacquired forest lands that were originally reserved for them and other tribes in an 1855 treaty that was never ratified. Following termination in 1954 and rerecognition in 1989, the tribe sought the return of its ancestral lands. They received 5,400 ac (2185 ha) of forested land from the BLM, which were placed into trust status in 1989 with the requirement that the lands meet the standards and guidelines of adjacent federal forests under the NWFP (MacKendrick 2009). The tribe (fig. 11-17) has adopted a forest management plan that upholds traditional values through the conservation of large trees, snags, and nesting sites of culturally important birds, and management practices that regenerate habitat for culturally significant wildlife following timber harvest (Vinyeta and Lynn 2013). The tribe proposed to extend approaches applied on its tribal lands through its Coos Bay Wagon Road Lands proposal, a collaborative effort with the BLM that incorporated silvicultural principles recommended by forestry experts Jerry Franklin and Norm Johnson (Franklin and Johnson 2012). For this coastal wet-forest environment, the proposed plan included provisions for new riparian management approaches; harvesting biofuels; retention of biological legacies such as large trees, coarse woody debris, and snags; variable-density thinning; long rotations; and regeneration harvest to maintain early-successional conditions (USDI BLM 2012).

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Figure 11-17—Coquille tribal members at Euphoria Ridge near North Bend, Oregon, spring 2003. Chief Don Ivy (left, with hat) addresses the group on a field trip to discuss tribal forest management and restoration strategies.

United States government policies, including the government-to-government relationship, constitute a distinctive legal basis for cooperation with tribes that recognizes their unique relationships to their ancestral lands. The U.S. Congress and presidents have established important laws and policies authorizing tribes to provide specific guidance to public lands management, which undergird some of the most substantive co-management arrangements on federal lands (Nie 2008). For example, the Santa Rosa and San Jacinto Mountains National Monument Act of 2000 stipulated that the U.S. Secretaries of the Interior and Agriculture "shall make a special effort to consult with representatives of the Agua Caliente Band of Cahuilla Indians regarding the management plan during the preparation and implementation of the plan" and authorized the use of "cooperative agreements and shared management arrangements with any person, including the Agua Caliente Band of Cahuilla Indians, for the purposes of management, interpretation, and research and education regarding the resources of the National Monument" (114 Stat. 1362 Public law 106-351). The Tribal Forest Protection Act of 2004 advanced such distinctive efforts by authorizing the U.S. Secretaries of Agriculture and the Interior to give special consideration to tribally proposed stewardship contracts, agreements, compacts or other arrangements on Forest Service or BLM land bordering or adjacent to Indian trust land to protect tribal trust resources from fire, disease, or other threats. A recent presidential proclamation established the Bears Ears National Monument in Utah to be managed jointly by the Forest Service and BLM while considering and integrating formal guidance and recommendations, which may be based upon tribal traditional and historical knowledge, from a commission made up of elected officers from five tribes (https://www.whitehouse.gov/the-press-office/2016/12/28/proclamation-establishment-bears-ears-national-monument).

In accordance with laws and policies cited above, the Forest Service has entered into landmark agreements that embody important principles of cooperative management and have recognized the unique stewardship role of tribes on their ancestral lands:

- In the late 1990s, the Lake Tahoe Basin Management Unit established various agreements with and issued special-use permits to the Washoe Tribe of Nevada and California to address tribal interests in managing ancestral lands at Lake Tahoe (Adelzadeh 2006).
- In 2004, the Plumas National Forest awarded a 10-year stewardship contract to the Maidu Culture and Development Group, a native nonprofit dedicated to strengthening Maidu culture and people, to apply traditional land management practices to 2,100 ac (850 ha) of national forest land in the Sierra Nevada (Donoghue et al. 2010).
- The Mount Baker–Snoqualmie National Forest entered into a MOA with the Tulalip Tribes in 2007 that supported cooperative efforts to sustain and enhance areas for treaty harvesting and other cultural practices, focusing on redcedar and huckleberries (LeCompte-Mastenbrook 2016). One particular outcome was establishment of a 1,280-ac (518-ha) "co-stewardship" area in the Skykomish watershed in 2011 to support mountain meadow restoration and huckleberry enhancement. The project has involved (1) removal of small conifers, (2) tribal youth involvement, and (3) maintenance of a road to provide tribal access.
- In 2011, the Fremont-Winema National Forest entered into a master stewardship agreement with the Klamath Tribes of Oregon, along with The Nature Conservancy and the Lomakatsi Restoration Project, in an effort to restore forests, reduce risks of severe wildfires, train the tribal workforce, and enhance wood product processing capacity (Hatcher et al. 2017).
- In 2015, the Forest Service entered into a 10-year master stewardship agreement with the Pit River Tribe and Lomakatsi Restoration Project to conduct treatments on more than 2 million ac within the Lassen, Modoc, and Shasta-Trinity National Forests in northern California (https://www.fs.fed.us/spf/tribalrelations/documents/agreements/15-SA-11052000-056.pdf).

Such cooperative arrangements have not only helped serve tribal communities, but they also can bring added expertise to public land management efforts, including better understanding of reference conditions and financial resources.

Integrating traditional ecological knowledge in collaborations—
Collaborative projects involving traditional ecological knowledge or native knowledge provide unique opportunities to enhance research and management, while also posing unique challenges for tribes and tribal-knowledge holders (Mason et al. 2012). There are many examples in which tribes and their members have seen benefits from working with researchers and land managers to inform restoration with traditional ecological knowledge, including burning to promote beargrass (Shebitz 2005) and land management planning (Clayoquot Sound Scientific Panel 1995). It is important to recognize also that tribal capacities and interest in conventional Western science have been critical in protecting vital resources such as salmon (Breslow 2014). Some tribes have suggested that agencies pursue collaborations that facilitate tribal application of traditional ecological knowledge to off-reservation lands within the respective tribes' ancestral territories without seeking to transfer or relinquish such knowledge (Norgaard 2014c). The latter is particularly important because many tribal knowledge specialists have expressed concerns that sharing cultural knowledge with nontribal entities could lead to its cooptation or misuse, such as loss of control by tribes or profiting by nontribal entities, as explained by Brewer II and Warner (2014) and the CTKW or Climate and Traditional Knowledges Workgroup (CTKW 2014). These authors, along with tribal representatives contributing to the NWFP 20-year monitoring report (Vinyeta and Lynn 2015), recommended taking steps to ensure that collaboration with tribes provides reciprocal benefits, minimizes risks to tribes, and recognizes inherent tribal rights and responsibilities to their communities. In particular, they suggested adopting agreements and principles such as "cause-no-harm;" ensuring "free-prior-and-informed-consent;" and protecting sacred, sensitive, or confidential information such as the locations of particular sacred sites, or specialized uses of fungi, plant, and animal species. Another approach is to establish stewardship agreements or compacts in which tribes can apply traditional ecological knowledge and applicable cultural practices on federal lands, such as the agreements between the Klamath Tribes and the Fremont-Winema National Forest (Hatcher et al. 2017).

Promoting tribal adaptive capacity—
Forest planning presents opportunities to support the continuity of traditional ecological knowledge across generations by maintaining culturally vital resources and tribal communities. In turn, tribal knowledge of historical and current ecological processes (Colombi and Smith 2012), and the seasonality of natural patterns, can help predict and prepare for future changes in habitats and species' distributions. Because traditional tribal cultures emphasize the interconnected nature of the human and nonhuman systems of the Earth, they are particularly well-adapted for addressing climate change (Heyd and Brooks 2009). Maintaining cultural keystone species such as salmon and safeguarding cultural keystone places are important for maintaining adaptive capacity, including memory and practices (Colombi 2012, Cuerrier et al. 2015). Maintaining cultural diversity in the form of tribal worldviews and languages regarding the natural world is also important for sustaining ecosystems and human communities (Pretty et al. 2009). Tribes continue to rely on historical intertribal networks that facilitate exchange of resources, cultural practices, and knowledge systems as a source of adaptive capacity (Papiez 2009, Trosper 2003, Turner and Cocksedge 2001). Many tribes across the region maintain such networks through summits, ceremonies, conferences, intertribal councils, and annual "canoe journeys" that support environmental governance and ecocultural revitalization (Norman 2012, Tveskov 2007). Federal land management agencies can support adaptive capacity by forming partnerships with tribes that value traditional tribal knowledge (see "Promoting collaboration" on p. 880), supporting monitoring and restoration of ecocultural resources, and engaging with intertribal resource management organizations (Whyte 2013).

Tribal Engagement in Climate Change Initiatives

In 2009, the Secretary of the Interior issued Order 3289, "Addressing the Impacts of Climate Change on America's Water, Land, and Other Natural and Cultural Resources," which established climate science centers (CSCs) and landscape conservation cooperatives (LCCs). The CSCs provide scientific information, tools, and techniques that resource managers and others can use to anticipate, monitor, and adapt to climate change impacts. The LCCs are landscape-scale conservation science partnerships that disseminate applied science, tools, and resources that support the management of cultural and natural resources. Within the NWFP area, the Northwest CSC, the North Pacific (NP) LCC and the Great Northern LCC have taken steps to facilitate tribal involvement. The Northwest CSC has a Tribal Engagement Strategy that provides opportunities for tribal engagement in each of its five core elements: executive services, science services, data services, communication services, and education and training services. The North Pacific Landscape Conservation Cooperative has tribal participation on the NPLCC steering committee, a tribal/first nation committee, and a subcommittee on science and traditional knowledge. The U.S. Department of Agriculture has also established regional climate hubs (https://www.climatehubs.oce.usda.gov/) to develop and deliver scientific information and technologies regarding climate to natural resource managers, including tribes. The science and resources developed by the LCCs, CSCs, and climate hubs can inform the management of culturally important tribal resources. Through funding support from Northwest CSC and the North Pacific LCC, tribes are fostering partnerships to bridge traditional knowledges and Western scientific knowledge (a complete list of tribal engagement projects is included on the Northwest CSC website and the NPLCC website). An example from within the NWFP is "Utilizing Yurok traditional ecological knowledge to inform climate change priorities" (Sloan and Hostler 2014).

Promoting multiscale temporal and spatial diversity in terrestrial habitats—

From stand to landscape scales, maintaining a diversity of plant communities that support tribal ecocultural resources is important for increasing resilience to wildfire, drought, pathogens, and insect pests (Churchill et al. 2013, Kauffman and Jules 2006). Efforts to promote heterogeneity within stands and across larger landscapes are likely to promote ecological diversity (see chapter 12), which in turn is important for maintaining traditional tribal livelihoods and lifeways (Lake 2013, Turner and Cocksedge 2001, Turner et al. 2011, Underwood et al. 2003). Traditional tribal burning practices that maintained nonforested habitats in both areal and linear arrangements were important for promoting diversity at different scales (Lewis 1982, Underwood et al. 2003). Tribal management has long accentuated transitional habitats, such as the edges between forest and nonforest habitats (Turner et al. 2003), to promote opportunities to obtain diverse resources. Although early-successional and nonforest communities are highly valued, maintaining large areas of old-growth forest is also important for sustaining tribal ecocultural values (Russo 2011, Yazzie 2007). Some wildlife species of special tribal value, including marten and pileated woodpecker, are associated with older forests, large decadent or dead trees, and dense tree canopies (Aubry and Raley 2002) (see chapter 6). Others are associated with young forests and more open forests that support vibrant understory plant communities and associated animals (e.g., porcupine and many Neotropical birds) (Carey 1996). Furthermore, arranging early successional patches in proximity to mature or old-growth patches is also important to promoting tribal uses (Rogers-Martinez 1992, Swanson et al. 2011). Thomas et al. (2006) recognized the importance of maintaining all structural stages across the landscapes of the NWFP area, which is a theme featured in chapters 3 and 12.

Reestablishing fire regimes—

A key principle for restoring landscapes in the NWFP area is the reestablishment of fire regimes in fire-adapted forest types through burns to accomplish resource objectives (Odion and Sarr 2007, Ryan et al. 2013) (see chapters 3 and 12). This approach reflects the strategy of managing or emulating "natural" disturbance regimes to promote ecological resilience (North and Keeton 2008, Odion and Sarr 2007). Restoration of fire regimes also remains one of the central elements of a strategy to promote tribal ecocultural resources and opportunities for ecocultural revitalization across the NWFP area. The importance of restoring fire is particularly prominent in the large areas marked by a frequent fire regime from northern California to central Washington (see chapter 3), but it is also important for sustaining woodlands, forests, prairies, and wetlands within regions characterized by less frequent fire regimes (Boyd 1999, Hamman et al. 2011). Because treatments to maintain tribal ecocultural resources often require more frequent and targeted applications of fire than would be expected through lightning ignitions alone, they depend on intentional burning (Turner 2014, Turner et al. 2011). Alterations of fire regime can be somewhat mitigated through harvest disturbances that emulate some fire effects (Anzinger 2002), but those surrogates cannot replicate all of the beneficial effects (see "Beneficial Effects of Fire for Ecocultural Resources" on next page).

In particular, frequent fires combined with other tending practices perpetuate ecocultural resources such as large hardwoods, camas, beargrass, and huckleberries in conditions that support tribal use (Hummel et al. 2015, Long et al. 2016a, Minore and Dubrasich 1978). More severe, stand-replacing fires also create or rejuvenate tribally eco-culturally important hardwood stands (Cocking et al. 2012), huckleberry fields (Anzinger 2002), riparian areas (chapter 7), and other early-successional plant communities. Such severe burns therefore provide opportunities to reinitiate tribal caretaking regimes; however, for many decades they also reduce important ecosystem services such as providing nuts and habitat for many species (Long et al. 2016a). Large and severe burns also pose serious threats to human lives, health, and property, and they can negatively affect down-stream aquatic resources (see chapter 7). Applying managed fire for resource objectives in concert with other silvicultural treatments helps to promote the desired fine-scale patchwork of successional conditions rather than a hands-off strategy that relies on unmanaged wildfires for disturbance. For example, treatments that reduce the likelihood of high-severity fire can mitigate threats to riparian areas and their associated fauna (Stephens and Alexander 2011). Efforts to maintain and restore tribal ecocultural resources will depend upon understanding how different spatial arrangements, frequencies, seasonalities, and severities of fire are likely to produce a favorable range of resources and ecosystem services (Storm and Shebitz 2006). Furthermore, understanding those fire effect patterns can help to predict which tribally valued resources will occur at specific places on the landscape over time (Lake 2013).

Strategies that involve greater use of fire will have to overcome a range of constraints, including air quality constraints, concerns for wildlife impacts, funding, crew availability, cross-boundary management, and public acceptance (Ryan et al. 2013). Chapter 12 considers these challenges given their relevance throughout this report. Weisshaupt et al. (2005) found that tribal members from central and eastern Washington were more likely to support prescribed burning than several nontribal groups because of their experience and cultural traditions of using fire. However, some tribal members and leaders share concerns about the risks of wildland fire with the general public. Such attitudes in part likely reflect lack of exposure to its traditional use (Carroll et al. 2010b, Norgaard 2014a). In addition, tribes with large reservations and extensive forestry operations have incentives to treat forests using harvest, which has historically supported jobs and other economic benefits.

Incorporating cultural burning—
Many tribes emphasize distinctions between cultural burning and prescribed burning, the latter of which is often practiced by public land management agencies. Cultural burning is planned to promote an array of ecocultural resources over time, often through relatively frequent applications (Burr 2013). Agency prescribed burning has often had a strong emphasis on reducing fuels, including residues from timber harvest or thinning, with frequent use of pile burning, cooler out-of-season burning, and other deviations from traditional fire regimes (Ryan et al. 2013). Such strategies can support restoration by phasing such fuels reduction activities prior to reintroducing more traditional use of fire (Lake and Long 2014, Long et al. 2016a); however, nontraditional treatments, which may include spring burning, may conflict with some tribal values and concerns for wildlife, as documented in the Klamath region of northern California (Halpern 2016). Furthermore, Anzinger (2002) suggested that restoring

Beneficial Effects of Fire for Ecocultural Resources

- Reducing the accumulation of forest fuels, which in turn can moderate the effects of stand-replacing wildfires without damaging large and old trees (Stevens et al. 2014, Waltz et al. 2014) (see chapter 12 for further discussion).
- Promoting understory diversity (Perry et al. 2011).
- Smoke-induced germination of species such as beargrass (Shebitz et al. 2009b).
- Reduction of pests such as filbert worms and weevils (Halpern 2016).
- Stimulation of fire-following fungi such as some morels to produce mushrooms (Larson et al. 2016, Pilz et al. 2004).
- Curbing the encroachment of conifers (Engber et al. 2011) and other more shade-tolerant or dominant plants into other desired and diverse vegetative communities.
- Producing plant structures and ground conditions that facilitate tribal harvesting and use (Lake and Long 2014).
- Lowering summer stream temperatures to sustain salmonids in particular areas through shading by smoke during critical summer periods (Lake and Long 2014). Robock (1991) previously demonstrated that smoke from wildfires lowered summer surface temperatures in the valley of the Klamath River.

huckleberry patches on the Mount Hood National Forest would require stand-replacing disturbance, such as high-severity burns or large patch cuts applied in conjunction with broadcast burns. Such examples demonstrate how strategies to promote ecocultural resources using fire will differ across the diverse array of tribal ancestral lands.

Many tribes desire a more active role in the implementation of cultural prescribed burns rather than leaving stewardship solely to the federal agencies and nontribal organizations (Eriksen and Hankins 2014). In 2003 and 2004, the Skokomish Indian Tribe worked with Olympic National Forest to restore beargrass and other native species using thinning and burning (Shebitz et al. 2009a). In 2006, the Quinault Indian Nation performed its own burn modeled after this project (Charnley et al. 2008b). Within the area of the Western Klamath Restoration Project (see box on p. 888), Karuk and Yurok tribal members and employees conducted prescribed burns in 2014 through the Klamath River Prescribed Fire Training Exchange (TREX) program (which was initiated by The Nature Conservancy and several federal agencies in 2002), in order to reduce hazardous fuels along an important road in the wildland-urban interface, increase tribal access to traditional food resources (e.g., acorns), and support research treatments; however, the project was limited to private and tribal lands rather than including Forest Service lands owing to a temporary agency ban on burning that summer (Harling 2015). Other projects have continued in the area (fig. 11-18), representing contemporary applications of traditional burning to achieve multiple tribal resource objectives (Lake et al. 2017).

Managing fires across jurisdictions—
To plan and implement fire-focused restoration treatments at the landscape scale requires cross-jurisdictional coordination (Lake et al. 2017). Revision of national forest plans provide new opportunities to coordinate with tribal communities developing community wildfire protection plans (often through Fire Safe Councils) and tribes developing integrated resource management plans. Fire management policy is allowing land managers to pursue more flexible approaches to use fire for resource objectives through managed natural ignitions and prescribed fire, including cultural burns. The U.S. Forest Service Pacific Southwest Region and The Nature Conservancy established an MOU to facilitate burning across public and private boundaries to achieve goals of the National Cohesive Wildland Fire Management Strategy (Harling 2015). Building upon such cooperative instruments, tribal groups are leading efforts to restore fire regimes through coordinated, landscape-scale burning, such as the Western Klamath Restoration Partnership. Such proactive coordination is important when allowing or curbing the spread of wildland fires across boundaries to meet the resource objectives of different landowners.

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Figure 11-18—Klamath River Training Exchange prescribed burn on a privately owned area for experimental research and tribal harvesting near Orleans, California, October 2015. Yurok and Karuk tribal members ignited an area under tanoak trees that had previously been treated (manually thinned in 2011, pile burned in 2012, and prescribed burned in spring 2013) to reduce hazardous fuels and improve acorn accessibility and quality (by reducing pests).

Western Klamath Restoration Partnership

The Orleans-Somes Bar Fire Safe Council, Karuk Indigenous Basketweavers, and Karuk Tribe initially partnered in 2001 (with funding through the National Fire Plan) to integrate tribal knowledge with hazardous fuels reduction and prescribed fire treatments on private and tribal lands between portions of the Six Rivers and Klamath National Forests in northern California (Senos et al. 2006). Building upon that foundation in recent years, the Karuk Tribe and Mid-Klamath Watershed Council have co-led the Western Klamath Restoration Partnership in designing and implementing landscape-scale integrated restoration strategies to reduce vulnerability of the environment and human communities to climate change, as well as to support tribal ecocultural revitalization efforts. The project area encompasses approximately 1.2 million ac (480 000 ha). Since 2013, the partnership has brought together tribes (Yurok and Karuk), tribal community groups (e.g., Indigenous Peoples Burning Network, Karuk Indigenous Basketweavers, Yurok Cultural Fire Management Council, and California Indian Basketweavers Association), The Nature Conservancy, federal and state agencies, and local fire safe/watershed councils to conduct hazardous fuels treatments and prescribed burns in and around several communities. As a demonstration project under the National Cohesive Wildland Fire Management Strategy, treatments have been designed to reduce fuel loading around homes, along critical emergency road routes, and ridges to facilitate use of fire across larger landscapes, as well as to enhance access to tribal basketry and food resources (Harling 2015, Senos et al. 2006). These projects have featured tribal workforce training and incorporated traditional ecological knowledge into prescriptions to promote tribal values. Early implementation steps for the partnership include prescribed burns under the TREX program (fig. 11-18) and the “Roots and Shoots” burn on September 29, 2016, to promote ecocultural resources, which was authorized under a fire management agreement between the Six Rivers National Forest and the Karuk Tribe.

Integrating tribal objectives into silvicultural approaches—
Many tribes harvest trees on their own lands as an economic activity and as a means of promoting desired resources, as a complement to fire to create canopy gaps and shift fuel conditions that facilitate the return of a more natural fire regime (Healey et al. 2008). Naturally formed canopy gaps from tree mortality create distinctive heterogeneity by forming pit and mound topography, broken tops and branches, and downed logs, which in turn stimulate understory diversity (Pollock and Beechie 2014) and associated wildlife communities (see "Tribal Ecosystem Services From Dead Trees and Forest Gaps" on p. 864). Various silvicultural approaches, including variable-density thinning treatments, can be important for recreating such natural stand heterogeneity and facilitating return of fire when restoring and maintaining woodlands and other nonforest areas encroached by trees (Carey 2003a, Devine and Harrington 2006, Hummel and Lake 2015). Chapter 3 features more discussion of restoration silviculture.

Many tribal silvicultural and related forest management approaches address sociocultural, economic, and ecological values with integrated management plans (Gordon et al. 2013), and these contribute to landscape diversity (see "Coquille Indian Tribe" on p. 882). The Pacific Northwest is particularly fertile ground for placing greater emphasis on the joint production of forest products in order to enhance community well-being, while also supporting biological diversity, recreation value, and aesthetic appeal (Von Hagen and Fight 1999). However, it is important to reconsider how constraints on harvest and thinning treatments imposed under the NWFP on forests over 80 years old in late-successional reserves, in addition to other restricted areas, have limited the opportunities for such treatments (Nelson 2015, Vinyeta and Lynn 2015).

Across many national forests of the NWFP area, historical logging has replaced mature forests with plantations (Healey et al. 2008). Many tribes have concerns over the effects of such plantings in terms of effects of chemical herbicides and alteration of successional pathways away from valued early-successional plant communities. Strategies for managing plantations have often focused on growth of commercial tree species, but strategies are increasingly directed toward promoting resilience to climate change, fire and pests, while concurrently providing services, including wildlife habitat, forest products, and tribal subsistence (Carey 2003b, Charnley et al. 2007, Franklin and Johnson 2012). Chapter 3 discusses these strategies for postfire management in more detail; Long et al. (2014b) also discussed reseeding for emergency erosion control, which could potentially affect understory plants used by tribes.

Proactively addressing use of chemicals in forest management—
National forest management uses herbicides, pesticides, fire retardant, and other chemicals for forest and resource management objectives, including accelerating growth of planted conifers and control of invasive species (Shepard et al. 2004). Many American Indians and tribes have registered concern over such use of chemicals because tribal harvesters are profoundly concerned about potential for exposure to environmental toxins (Huntsinger and McCaffrey 1995, Norgaard 2007, Segawa et al. 1997). These concerns are particularly strong for terrestrial and aquatic food resources and nonforest products such as foods and basketry materials that people place in their mouths. Researchers collaborated with the California Indian Basketweavers Association to study potential exposure to several common herbicides (glyphosate, hexazinone, and triclopyr) used to promote conifer growth on four national forests in the Sierra Nevada region of California (Ando et al. 2003, Segawa et al. 1997). They found that herbicides were detectable on several plant species that are likely to be gathered by American Indians for many months (a range of 4 to 130 weeks), and in some cases beyond the targeted treatment areas owing to drift or precipitation. In addition to ensuring that risk assessments properly consider the special vulnerabilities of tribal members (Burger et al. 2008), strong working relationships, including effective consultation, with tribes and harvesters are important to proactively understand and avoid potential for exposure of tribal members to harmful chemicals.

Actively managing riparian areas—
Promotion of tribal ecocultural resources within riparian areas depends on periodic disturbance to maintain gaps and understory production. Especially in drier areas with more frequent fire regimes, disturbances such as managed fire and removal of trees can be important for restoring desired conditions. Streams in mid-successional forests often can be more productive than those in old-growth forests under certain circumstances; therefore, active management may be important to sustain productivity of aquatic ecocultural resources such as fish (Reeves et al. 2006). On the other hand, researchers have suggested that removing trees from riparian areas could reduce suitability of associated streams for coldwater fishes (McClure et al. 2013). Considering regional and site-specific contexts, such as current temperature regimes, can often reconcile such potential tradeoffs, as discussed further in chapter 7.

Some tribes have expressed concern that restrictions in riparian reserves, which were intended to protect those sensitive areas adjacent to streams (Naiman et al. 2000), would impede their ability to maintain traditional harvesting and burning practices. For example, members of the Karuk Tribe expressed concerns that the Aquatic Conservation Strategy of the NWFP would impose restrictions on cutting willows in riparian reserves (Charnley et al. 2008a). However, several projects have included cutting and burning willows in riparian zones along the Klamath River (Lake 2007). Nevertheless, the tendency to leave riparian areas untreated, as discussed in chapter 7, can chafe tribal interests in promoting understory plants or shade-intolerant trees, such as large oaks and pines growing on river terraces adjacent to historic village sites (Hosten et al. 2006).

Restoring aquatic systems—
Given the importance of anadromous fish species such as salmon, lamprey, and sturgeon to tribes, a very broad approach is important to address their complex life stages that depend on diverse and interconnected habitats (Close et al. 2002, Miller 2012, Wang and Schaller 2015). Free-flowing stream networks from forested headwaters are also important for supplying driftwood to tribal riverine and coastal communities (Lepofsky et al. 2003). Recovery of tribally valued fish, waterfowl, and aquatic plant species heavily depends on restoration of hydrologic regimes and physical habitats through removal of reservoir dams that impede fish migration; restoration of degraded meadows; removal or relocation of roads, levees, and diversions; and other actions to restore the eco-hydrological system through more natural flows of water, sediment, wood, and organisms (Beechie et al. 2013, Nehlsen et al. 1991) (see chapter 7). Treatment of invasive exotic plants in wetlands and riparian areas may also be a priority for restoration of ecocultural resources. Such active measures can help to ameliorate the predicted effects of climate change (Wade et al. 2013). In particular, enhancing connectivity among native fish populations is important for increasing the potential for wildfire to benefit them rather than cause extirpations (Falke et al. 2014, Flitcroft et al. 2016).

Removing reservoir dams—
Although large reservoirs are an important part of infrastructure in the Pacific Northwest, removal of dams that form such reservoirs has become increasingly common as many aging dams require expensive modifications. In the last decade, several major dams have been intentionally breached within the NWFP area, notably the Elwha and Glines Canyon Dams in the ancestral lands of the Lower Elwha Klallam Tribe on the Olympic Peninsula (Pess et al. 2008), and the Condit Dam on the White Salmon River. More removals are anticipated, with the 2016 Klamath Power and Facilities Agreement set to remove four dams on the Klamath River. Such efforts will affect national forest lands and tribal ecocultural resources, and they are likely to increase the importance of upstream watershed conditions as stream reaches are reopened to migratory fish (Pess et al. 2008). Existing research points to a variety of anticipated benefits for migratory fish and associated mollusks; however, dam removals can also release accumulated sediments, nutrients, toxins, and other pollutants (Pess et al. 2008, Poff and Hart 2002, Stanley and Doyle 2003), which can temporarily disrupt downstream habitats of sensitive organisms such as freshwater mussels. An additional concern is the potential spread of invasive species upstream (Hart et al. 2002). Dam removal could also affect tribal concerns by exposing traditional sites, burials, and artifacts.

Therefore, although dam removal is expected to be critically important in restoring aquatic organisms of special significance, its potential for both beneficial and harmful effects should be considered. Because of the diversity of watershed settings, the recency of large dam removal, and short duration of post-removal studies, scientists are working to understand the longer term benefits and possible impacts of such actions (Hart et al. 2002, Poff and Hart 2002). In the meantime, large dam removal provides opportunities for integrated restoration of tribally valued riparian plants such as willows, berry plants (e.g., *Rubus parviflorus*) (Michel et al. 2011), and birds (Gelarden and McLaughlin 2013). As one example of how forest management can complement dam removal, McLaughlin (2013) recommended maintaining or increasing large woody debris within the riparian zones to encourage use by birds, which in turn disperses seeds across the bare sediments.

Managing roads—
Roads and associated water crossings can degrade aquatic resources by increasing erosion and creating barriers to movement as discussed in chapter 7. Tribes have successfully sued the state of Washington to demand remediation of culvert impacts on fish passage to support their treaty fishing rights (Breslow 2014). This lawsuit not only demonstrated the legal power of tribal treaty rights to shape environmental management across jurisdictions, but it also highlighted the importance of road management on tribal ecocultural resources. Tribes have partnered with national forests and BLM districts to implement and study road decommissioning to restore habitat for native salmonids (Burnson and Chapman 2000). One study that involved the Nez Perce Tribe found that road recontouring, rather than passive recovery following road abandonment, accelerated recovery of ecological and hydrological properties, including carbon storage (Lloyd et al. 2013).

Although roads can exact a toll on terrestrial and aquatic resources, aesthetics, and other values, they also provide access for tending forests, managing fire, hunting, fishing, plant harvesting, and other activities that are important to tribal members. Tribes have emphasized their interests in both access and watershed management (Vinyeta and Lynn 2015), so consultation is particularly important when making plans regarding roads. In particular, tribal members have noted that a lack of road maintenance and road closures can limit access to desired resources, especially for many elders and families with young children who rely upon vehicle access (Dobkins et al. 2016, LeCompte-Mastenbrook 2016). Consequently, intergenerational transmission of knowledge in part depends on suitable road systems. Because roads also offer access to nontribal members, they also have potential to exacerbate resource competition in preferred gathering areas.

Facilitating tribal access to forest products—
National forests have adopted various policies regarding regulation of harvesting by tribal members on ancestral lands (Catton 2016). Within the Sawtooth Berry Fields on the Gifford Pinchot National Forest, Hansis (1998) stated that "American Indians do not need to obtain permits to harvest huckleberries as part of their treaty rights" (Hansis 1998: 78). In a northern California example, the Six Rivers and Klamath National Forests established an MOU with the Karuk Tribe under which tribal members were not required to obtain permits from the Forest Service to harvest special forest products for subsistence (Stuart and Martine 2005). Many national forests provide fee waivers for tribal members to gather firewood on national forests; for example, the Fremont-Winema National Forest established an MOA with the Klamath Tribe that allowed tribal members to camp and gather firewood within former reservation lands (Catton 2016). Other remedies proposed to lessen the burden from permitting requirements include using tribal identification cards in lieu of permits (Wrobel 2015) or having tribal organizations rather than the Forest Service issue the permits (Dobkins et al. 2016). For example, as outlined in an MOU with several national forests in Michigan, Wisconsin, and Minnesota, the Great Lakes Indian Fish and Wildlife Commission has issued permits to members of several tribes to harvest wild plants and nontimber forest products, as well as to camp, on national forests (Wrobel 2015). The permitting system allows the commission to monitor and report on tribal harvest of various forest products.

Addressing conflicts with nontribal communities over access and use—
Public managers have implemented various strategies to address conflicts between tribal members and nontribal people, including recreationists and nontribal harvesters of forest products, regarding impacts to ecocultural resources. Forest Service policy (FSH 2409.18.80) restricts issuance of commercial permits when there are shortages to ensure that tribal use can be accommodated. As Alexander et al. (2011) pointed out, most collectors of nontimber forest products gather for personal or subsistence use, so records from commercial permits provide a very incomplete picture of demand. The Forest Service's National Tribal Relations Program Task Force recommended a variety of measures to improve tribal management of lands under federal care, including providing the Forest Service with the authority to close federal lands to the public for tribal traditional uses (Nie 2008). When supplies of desired resources are limited, land managers can regulate access through seasonal area closures that do not restrict access and harvest by tribal members. Numerous examples suggest that successful resolution of conflicts over access depends upon a strong and proactive working relationship between land managers and tribes that recognizes their unique status (Catton 2016). In an important precedent, the Gifford Pinchot National Forest designated a long-standing berry-harvesting area for exclusive use by American Indians under its land management plan (see box on next page). A similar approach was formalized under an MOU between the Mount Hood National Forest and the Confederated Tribes of the Warm Springs Reservation (Catton 2016, Wang et al. 2002).

Sustaining timber harvest and mill capacity—
Tribes with interest in commercial timber harvest from their lands, such as the Quinault Indian Nation, have expressed concern that cutbacks in harvest on federal lands have caused declines in mill capacity and other resources needed to allow them to manage and receive income from their working forest lands, as well as to protect their homelands from hazardous buildup of fuels (Vinyeta and Lynn 2015). In some parts of the NWFP area, such as the mid-Klamath region, declines in the timber industry have been partially offset by tribal leadership in economic development (Charnley et al. 2008a). These examples demonstrate interconnections among federal forest management and tribal and local economies, as well as opportunities for federal-tribal partnerships to promote mutual interests (Corrao and Andringa 2017, Mason et al. 2012). For example, the Yakama Nation's milling facility has processed logs resulting from the Tapash Sustainable Forest Collaborative forest restoration project.

Considering effects of special designations—
A variety of special designations, such as experimental forests, research natural areas, wild and scenic rivers, and wilderness areas can constrain activities on federal lands. As mentioned earlier in this chapter, sites recommended for special designations based upon distinctive qualities and limited degradation are likely to be significant to tribes (Hughes and Jim 1986). Consequently, proposals for special land management designations, including reserves, can impede tribal access to important resources and culturally important places (Freedman 2002) as well as the use of tools that could aid restoration. Past efforts to impose designations such a wilderness areas without tribal support have been a source of much consternation to the affected tribes (Catton 2016). Recent wilderness legislation has included special provisions to protect tribal religious concerns (see "Addressing sacred sites protection and access" on p. 880). Nevertheless, concerns persist among tribal communities that special designations for conservation purposes may limit their access and use (Baldy 2013, Nie 2008, Papiez 2009). For example, an analysis reported by Nelson (2015) noted that 47 percent of Mount Baker–Snoqualmie National Forest lands have wilderness status, 5 percent more are administratively withdrawn, and 36 percent are allocated to late-successional and riparian reserves, so only 10 percent remain as matrix or adaptive management areas where timber harvest is less constrained. These designations could limit active silvicultural management to enhance

Forest Service-Yakama Nation Handshake Agreement to Access Huckleberries

An important historical instance of federal-tribal collaboration is the 1932 Handshake Agreement between the Yakama Nation and the U.S. Forest Service. In response to growing pressure on wild huckleberries from the unemployed migrant workers during the Great Depression, J.R Burkhardt, then Gifford Pinchot National Forest supervisor, met with tribal representatives and agreed to reserve 2,800 ac (1130 ha) of off-reservation huckleberry patches for exclusive use by the Yakama Nation during huckleberry season (Richards and Alexander 2006). This agreement has been honored since, although it was only put into writing as recently as 1990, prior to the adoption of the Northwest Forest Plan (Fisher 1997, Richards and Alexander 2006). This case set an important precedent for upholding the federal trust responsibility and the rights of the Yakama Nation to harvest on public lands. However, there have still been conflicts when non-Indians have harvested in the exclusive area, which is signed and bounded by a road (fig. 11-19) (Hansis 1998).

Leslie Seaton

Figure 11-19—The Handshake Agreement sign denoting area set aside for tribal harvest of huckleberries in the Indian Heaven Wilderness, Gifford Pinchot National Forest, Washington, August 2012.

huckleberry, elk, and other tribal ecocultural resources to a very small percentage of their potential habitats (LeCompte-Mastenbrook 2016). Excluding wilderness and reserve areas from harvest both constrains and increases the importance of fire to sustain these resources.

An alternative type of special designation is contemporary tribal use or stewardship areas. Several national forests have designated landscape areas as tribal heritage districts, zones, or areas. These areas have a documented history of tribal uses and are conceptually similar to traditional cultural properties designated under the authority of the National Historic Preservation Act of 1966. Such tribal landscape area designations are linked to federal policies that facilitate consultation and coordination for heritage management (Wang et al. 2002). Agreements can guide permissible management actions, protect heritage or cultural resources, and foster tribal care and use of ecocultural resources for traditional and cultural purposes. As explained above, these approaches can address not only the ecological condition of forests, but also help to sustain tribal knowledge and social capacity. The concept of tribal stewardship areas bears some resemblance to previously mentioned "community forests," which are managed for the benefit of particular communities (see "Fostering cooperative management" on p. 881). There have been several examples of such designations in the NWFP area:

- Nearly two decades ago, the Mount Baker–Snoqualmie National Forest settled a dispute with the Muckleshoot Indian Tribe regarding its exchange of culturally significant tribal ancestral territory to a private corporation, by designating "special management areas" for protection of cultural and historical features and for promotion of elk forage, portions of which were subsequently targeted for huckleberry enhancement (LeCompte-Mastenbrook 2016).
- In recent decades, the Mount Hood National Forest has set aside huckleberry tracts for exclusive tribal use and cooperatively managed the areas with the Confederated Tribes of the Grande Ronde Community of Oregon and Confederated Tribes of the Warm Springs Reservation of Oregon using prescribed fire and thinning on competing vegetation (Anzinger 2002, Gerwing 2011, Wang et al. 2002).
- A 2012 agreement between the Klamath and Six Rivers National Forests and the Karuk Tribe supported restoration of the Katimiin Cultural Management Area through application of cultural practices, including reintroduction of fire (Lake and Long 2014). Revisions to the Klamath National Forest Land and Resource Management Plan had provided for such special designations (Diver 2016).

Supporting adaptive management—

Researchers have recommended greater use of adaptive management frameworks as a way to better understand the complex responses of socio-ecological systems to management strategies (Franklin and Johnson 2012, Gray 2000). The NWFP called for using adaptive management areas (AMAs) to allow land managers the flexibility to try new and innovative management strategies and treatment practices as experiments to reduce uncertainty of subsequent management actions (Bormann et al. 2007, McClure et al. 2013). Many tribal practitioners believed that such approaches shared a common perspective with traditional tribal systems (Catton 2016), which have been described as an aboriginal form of adaptive management (Berkes et al. 2000). Adaptive management efforts can support tribal engagement in monitoring, assessment, implementation, and evaluation of treatments to promote desired conditions (Stein et al. 2013). Such efforts can complement and propel larger landscape restoration strategies (Berkes 2009), as well as build capacity among tribes, stakeholders, and agencies (Fernandez-Gimenez et al. 2008). In an example from the NWFP, land managers of the Northern Coast Range AMA established agreements with the Confederated Tribes of the Grande Ronde Community of Oregon to facilitate cohesive management of a watershed that included 10,900 ac (4400 ha) of federal land (Gray 2000). Some projects in AMAs specifically addressed tribal ecocultural resources; for example, the Cispus AMA in Washington included a project to study beargrass production under different forest canopy levels (Blatner et al. 2004). However, many of the bureaucratic challenges that appear to have limited implementation of adaptive management, including limited staff and funds, cumbersome environmental reviews, and institutional momentum (Gray 2000), have frustrated tribal partnerships as well (Catton 2016). Such challenges, including reduced support for monitoring and review of proposed management changes, were specifically cited by tribal respondents in the 20-year monitoring report as inhibiting adaptive management (Vinyeta and Lynn 2015). The challenges in making formal adaptive management projects successful have encouraged less formal approaches that emphasize observation, communication, and explicit review of ecological changes and adaptation actions (Peterson et al. 2011).

Research Needs

There are many topics regarding tribal ecocultural resources and engagement that warrant research, and collaborating with tribes to identify cultural keystones could be especially helpful in setting priorities. There are valuable examples of collaborative research regarding tribal ecocultural resources in the NWFP area (e.g., beargrass, pileated woodpecker, huckleberries, and black oaks as mentioned previously), but more studies and expanded monitoring are needed to address the many interests of diverse tribal communities. Although many of these species have been studied, research designed by ecologists may not target the conditions used by harvesters, as explained by Kerns et al. (2004) in a study of huckleberries. Beatty and Leighton (2012) highlighted several common themes in tribal research priorities based upon a survey of tribal forest resource managers and decisionmakers, including (1) research related to water, fisheries, and other nontimber values from forests; (2) collaboration and cooperation, especially concerning the integration of traditional knowledge with Western science; and (3) adaptation of research projects to address local tribal concerns. More specifically, there is considerable need for monitoring and research in cooperation with tribes on the suitability and availability of ecocultural resources for tribal use. Norton-Smith et al. (2016) identified a need to research whether and why cultural keystone species are moving beyond tribal access. Research on reintroducing possible cultural and ecological keystone species such as condors, wolves, and beavers can evaluate not only the ecological effects within the NWFP area but also the effects on tribal cultural values.

A particularly important need is for research that is collaborative and integrative in evaluating the benefits of active forest management (Hummel and Lake 2015). In a report by the Karuk Tribe, Norgaard (2014b) prioritized the need for such socioeconomic research, in addition to research on the effects of climate change on tribal sovereignty, identification of effective contracting and agreement mechanisms, and study of carbon implications of tribal burning. Considering vulnerability and developing adaptation strategies in cooperation with tribal entities is important for understanding the effects of ecological change on tribal communities (Dittmer 2013, Norgaard 2014c, Petersen et al. 2014). Chapter 3 discusses the need to better understand the effects of applying ecological forestry strategies designed to reestablish or emulate natural disturbance regimes. It is particularly important to consider how a lack of active management is likely to affect tribes given current and expected future disturbances, including forest densification, dieback, and wildfire (Norton-Smith et al. 2016). Although this synthesis demonstrates such impact in qualitative terms, more precise understanding of the magnitude of those impacts would help to make better investments.

In many cases, information to quantify reference conditions, such as the abundance of particular resources and forest structure in pre-Euro-American times, is lacking, particularly at fine scales. Expected declines in both ecocultural resources and harvester knowledge of those resources increases the likelihood of "shifting baselines syndrome," mentioned above, under which current generations of harvesters and decisionmakers may no longer have an accurate understanding of past conditions. Collaborative partnerships in planning, research, and monitoring provide opportunities to better quantify and achieve appropriate desired conditions (Hummel et al. 2015, Long et al. 2016a).

Research is also needed on the socioeconomic, cultural, and ecological effects of resource harvests (potentially both recreational and commercial), road closures, and permitting systems on tribal ecocultural resources and the communities that harvest them (LeCompte-Mastenbrook 2016). Monitoring is important to help answer these questions. For example, Nelson (2015) suggested that a nonobstructive permit system would be useful in quantifying recreational harvest of huckleberries, while monitoring of resources such as cedar logs on public lands would help track inventories and supply tribal needs (Vinyeta and Lynn 2015).

Attention to the ethics of participatory research, including consideration of appropriate roles and relationships, open and transparent communication and decisionmaking, and facilitating opportunities for engagement in all stages of an effort, is important in encouraging community participation and promoting the likelihood of mutually beneficial outcomes that build capacity to solve problems (Fernandez-Gimenez et al. 2006, Long et al. 2016b, Walker et al. 2002). Tribes may support collaborative efforts that engage members, from youth to long-term harvesters, in monitoring, research, and restoration (LeCompte-Mastenbrook 2016). Through such efforts, tribal practices based upon traditional knowledge, such as cultural burning, can be studied, implemented, and evaluated for their effects on valued species, ecological integrity, and ecosystem services.

Conclusions and Management Considerations

Based upon the literature reviewed in response to the guiding questions for this chapter, including the original question posed by managers regarding the sustaining of first foods, we found the following conclusions for consideration by land managers:

1. Ecocultural resources and causes of degradation
    - Ecosystems of the NWFP area support a wide array of tribal ecocultural resources, including various foods, medicines, materials, and nonmaterial values.
    - Both social and biophysical factors detract from the ability of tribes to obtain ecocultural resources from public lands in the desired quality and quantity.
    - Degradation of important tribal resources, including a variety of "first foods," is attributable to shifts in fire regimes away from frequent fire, conifer encroachment and densification, invasions by exotic species, alterations of hydrologic systems, species extirpations, reductions in tribal tending, and other historical legacies.
    - Examples of highly desired biological resources that depend on restoration of disturbance regimes include numerous trees and shrubs that produce edible nuts and fruits, geophytes that produce edible roots, fungi that produce edible mushrooms, and grasses that produce nutritious seeds and forage for ungulates. Many important plants and fungi used for medicine, foods, and crafts are associated with nonforest communities and forest gaps, some of which constitute short phases of succession, and others which can be persistent. Other important resources came from woodlands and forests that were dominated by old trees but often maintained with fire.
    - Historical displacement of tribal influence in the region has contributed to the reduction in frequency of fire in many parts of the region, particularly in relatively drier inland areas and locations near historical tribal settlements, trade and travel routes, and harvesting and hunting areas. Many of these locations were in ecological transition areas, such as edges between forests and grasslands or wetlands, which were maintained by tribal use.

2. Land management approaches to promote tribal ecocultural resources
    - In general, ideas to promote tribal ecocultural resources are consistent with emerging directions in forest management, including seven core principles for restoring fire-prone inland Pacific landscapes suggested by Hessburg et al. (2015).
    - Restoring large landscape areas that span traditional areas still used by tribes can help to ensure long-term sustainability and availability of resources, with important socioeconomic benefits such as food security and restoration-related work opportunities (see "Western Klamath Restoration Partnership" on p. 888).

- Remediation of forest road systems and culverts constitutes a priority for restoring aquatic systems where forest management activities have impeded fish passage and flows of wood, water, and sediment. However, road systems are important for maintaining tribal access to resources and intergenerational transmission of knowledge.
- Active forest management, including understory and variable overstory thinning and greater use of fire, is vital to improve the productivity and availability of many tribal ecocultural resources. Active management strategies can be integrated with efforts to conserve large, old trees, cultural sites, and other ecocultural resources that might be vulnerable to severe disturbances.
- Reintroduction of ecocultural keystone species that have been extirpated, in conjunction with restoration of their habitat, is also important for sustaining tribal material uses, cultural values, biological diversity, and ecological processes.
- Development of burn strategies and prescriptions in cooperation with tribes can help to reestablish traditional cultural burning and produce desired fire effects. Such an emphasis is a greater need in drier ecosystem types that evolved with more frequent fire, but it is also important at fine scales within wetter ecosystem types. This finding is consistent with the principals suggested by Hessburg et al. (2015) to emulate disturbance regimes.

3. Engaging tribes in forest planning and management
   - Given the widespread interests of tribal communities in forest ecosystems of the NWFP area, tribal engagement, including formal consultation as well as broader partnerships, is important to achieve land management objectives set forth in the forest planning rule, to uphold tribal rights and federal responsibilities, and to recognize the importance of tribal ecocultural resources on ancestral lands.
   - The concepts and principles of adaptive management and restoration forestry are consistent with efforts to promote tribal interests.
   - Collaborative partnerships with tribes, encompassing consideration of native knowledge, in planning, researching, implementing, and monitoring treatments within an adaptive ecosystem management framework fosters adaptive capacity of tribes and the partnering institutions.
   - Such partnerships can build upon the legal foundations that provide for explicit tribal engagement and cooperative management.
   - In particular, designation of special tribal stewardship areas of cultural importance to tribes can achieve both social and ecological objectives of both tribes and federal land management agencies.

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A field tour with the Lakeview Forest Landscape Collaborative in the Fremont-Winema National Forest. Photo by Tom Spies, USDA Forest Service.

Chapter 12: Integrating Ecological and Social Science to Inform Land Management in the Area of the Northwest Forest Plan

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> "We are drowning in information, while starving for wisdom. The world henceforth will be run by synthesizers, people able to put together the right information at the right time, think critically about it, and make important choices wisely."
>
> —E.O. Wilson, *Consilience: The Unity of Knowledge* (1988)

Introduction

Long-term monitoring programs and research related to Northwest Forest Plan (NWFP, or Plan) goals, strategies, and outcomes provide an unprecedented opportunity to examine how the scientific basis and socioecological context of the Plan may have changed during the 23 years since its implementation. We also have a prime opportunity to reassess how well the goals and strategies of the Plan are positioned to address new issues.

The NWFP was developed in 1993 through a political process involving scientists in an unusual and controversial role: assessing conditions and developing plan options directly for President Bill Clinton to consider with little involvement of senior Forest Service managers. The role of Forest Service scientists in this planning effort is different—scientists are now limited to producing a state-of-the-science report in support of plan revision and management (USDA FS 2012a), and managers will conduct the assessments and develop plan alternatives.

Implementation of the NWFP was followed by monitoring, research, and expectations for learning and adaptive management; however, little formal adaptive management actually occurred, and the program was defunded after a few years. The goals of the NWFP were daunting and set within the policy and ecological context of the time. President Clinton's question to the Forest Ecosystem Management Assessment Team (FEMAT) was "How can we achieve a balanced and comprehensive policy that recognizes the importance of the forest and timber to the economy and jobs in this region, and how can we preserve our precious old-growth forests, which are part of our national heritage and that, once destroyed, can never be replaced?" (FEMAT 1993). The 1982 planning rule guided land management planning on National Forest System lands, emphasizing conservation based in part on maintaining population viability of native species.

Although many conservation concerns have not changed, new science and challenges have emerged. For example, since the Plan was developed in the early 1990s, the invasive barred owl (*Strix varia*) has become a major threat to populations of the northern spotted owl (*S. occidentalis caurina*) (chapter 4), the number of Endangered Species Act (ESA)-listed fish species has gone from 3 to more than 20, and the frequency and extent of wildfires in dry forest portions of the Plan area have increased substantially in response to climate warming (chapter 2) (Reilly et al. 2017a, Westerling et al. 2006).

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The policy context and social dimensions of the NWFP have also changed. For example, the 2012 planning rule (USDA FS 2012a) places more weight on managing for ecological integrity (an ecosystem or coarse-filter approach) and less weight on population viability of individual species (a species or "fine-filter approach) (Schultz et al. 2013) than did the 1982 rule. The Plan's evaluation of societal influences did not address the emergence and expansion of collaborative processes throughout the NWFP region (Skillen 2015), and the FEMAT assessment itself (1994) largely focused on commodity-based economic development and support for maintaining stability of local and regional economies (Charnley 2006a). In addition, many but not all local economies of the region have diversified away from dependence on federal timber, and the forest products industry has largely moved away from using and valuing large logs, favoring instead the use of small-diameter trees (Haynes 2009).

Scientists in the Plan region also now more fully understand that the social and political context of the NWFP had a strong influence on the setting and attaining of the ecological goals of the Plan—opinions and debates about federal forest management in the region were as much about social values and conflict resolution as they were about science (Lange 2016, Spies and Duncan 2009). Given this context, it is important to have realistic expectations for how this science synthesis might contribute to the assessments and subsequent revision of individual forest plans and forest management. Scientific findings alone will not resolve political debates about the use of natural resources. Reducing scientific uncertainty will not necessarily reduce political uncertainty; and politics will always outweigh science because "science does not compel action" (Pielke 2007). However, providing the latest scientific information and reducing scientific uncertainty are expected to lead to better management decisions within the context of social and political constraints.

There is also an increased emphasis on the social dimension of planning today compared to when the NWFP was developed. Federal managers increasingly use collaboratives, stewardship contracts, and local participation in decisionmaking (Leach 2006, Urgenson et al. 2017). The 2012 planning rule also emphasized that plans must provide for "social, economic and ecological sustainability." This increased emphasis on integrating social and ecological aspects of forest planning coincides with the developing science of coupled human and natural systems or "social-ecological systems" (Liu et al. 2007) (fig. 12-1).

This socioecological perspective goes well beyond the ecosystem management framework that guided development of the NWFP by accounting for interactions between social and ecological systems to help deal with system complexity (fig. 12-1), surprises, and unintended outcomes from policies (Spies et al. 2014). For example, the relationship of federal forests to community well-being has changed since initiation of the Plan. Many communities no longer depend on the economic contributions of wood products as they once did (Charnley 2006a). There is growing recognition of the economic benefits of public lands to communities from recreation and tourism (White et al. 2016a) and nontimber forest products (Alexander et al. 2011), and recognition that ecosystems provide many benefits to human communities beyond timber and nontimber resources. Many studies indicate that the impact of humans on the environment in the NWFP area is much broader than the effects of natural resource extraction. Furthermore, it is clear that the timber industry has also experienced changes throughout the NWFP region, many of which are independent of management decisions on federal lands (e.g., fluctuations in national and global markets for wood products, transformations in how forest products companies are structured, and adoption of new technologies for wood processing) (chapter 8). At the same time, researchers and managers better understand connections between the organizational capacity of agencies, mill infrastructure, and business capacity in the private sector (e.g., a skilled workforce) in achieving forest restoration goals (chapter 8).

The fundamental assumption of the NWFP was that the breadth of the biological and socioeconomic strategies would achieve its biodiversity conservation and socioeconomic goals, and that those goals were also compatible with each other. Scientists and managers now have the perspective afforded by 23 years of research, monitoring, and field experience to suggest that these assumptions were

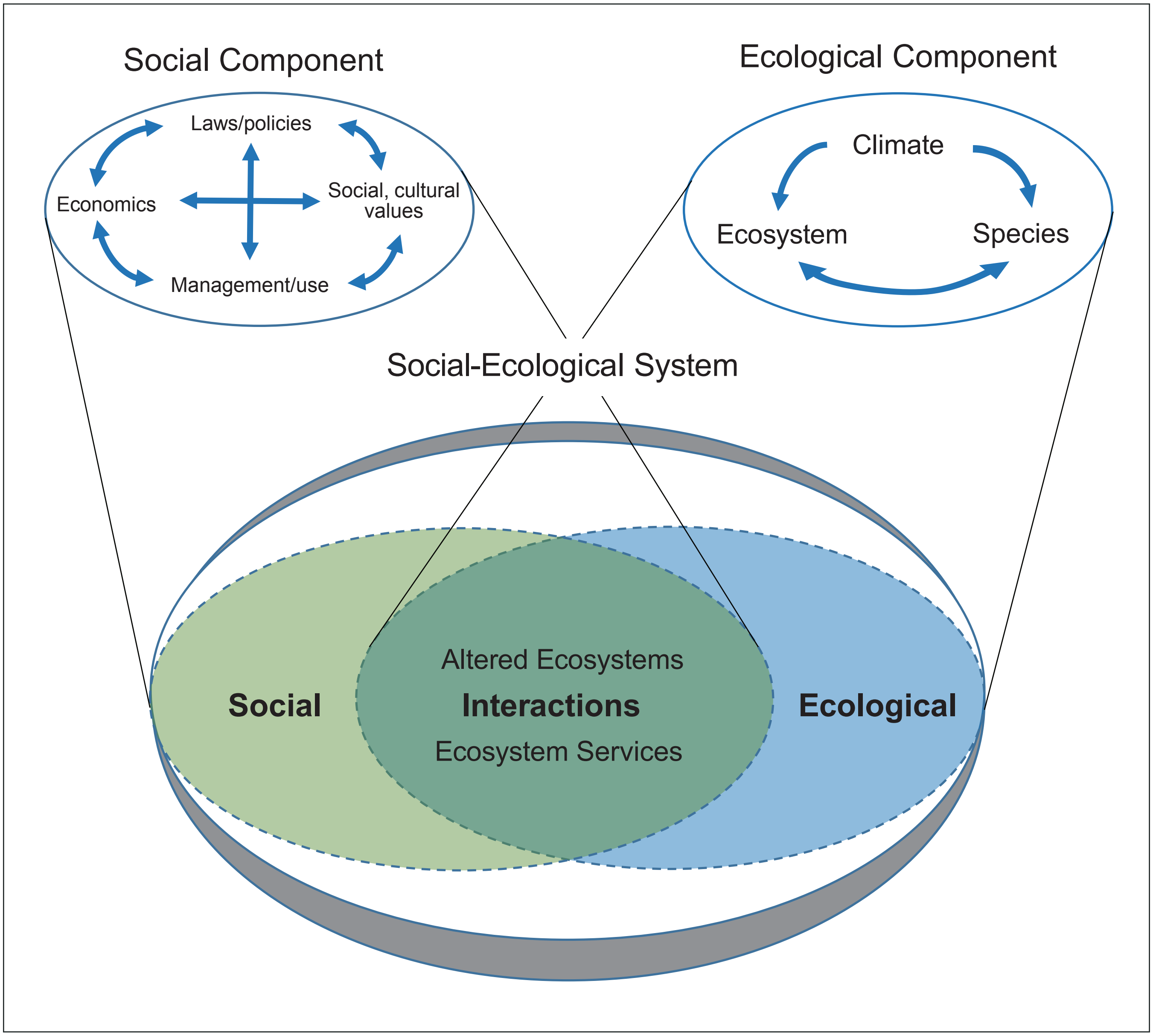

Figure 12-1—Major components and interactions in the Northwest Forest Plan social-ecological system.

only partially correct. In this chapter, we explore these assumptions in depth, using the lens of socialecological systems, and we identify new issues and concerns. We have four major objectives in this chapter:

1. Set the broader context of the NWFP goals and conservation approaches in terms of the science of socialecological systems.
2. Increase awareness of the diversity of ways that humans have influenced forest ecosystems, landscapes, and species of the Pacific Northwest.
3. Characterize how the conservation, restoration, and socioeconomic strategies of the NWFP interact, and how well they meet the original goals and new issues that have arisen since the Plan was established.
4. Identify key scientific uncertainties, research needs, and management considerations.

Guiding Questions

The guiding questions for this chapter are partly based on the questions from the managers (chapter 1), which are addressed more directly in individual chapters, and on cross-cutting questions and issues identified by the authors. The guiding questions for this chapter are:

1. What are the latest findings and perspectives on how global environmental change (including climate, land use, and invasive species changes) is altering forest and aquatic-riparian ecosystems, and their disturbance processes, and how relevant is this science to the NWFP area?
2. What are the latest scientific perspectives on reserve management for species conservation, given new understanding of ecosystem dynamics, and the influences of global environmental change?
3. What are key social components and drivers of the social-ecological systems in the NWFP area?
4. How compatible are the goals and strategies of the NWFP, and how well have the goals been met?
5. How compatible are coarse- and fine-filter approaches that simultaneously guide management for forest ecological resilience and single species viability across the range of disturbance regimes in the NWFP area?
6. What are new concerns within the social-ecological system of the NWFP area, and how well are the original Plan goals and strategies positioned to deal with them?
7. What is known about the tradeoffs of restoration actions across a range of conservation and community socioeconomic well-being goals?
8. What are the current and projected regional-scale issues and challenges associated with the goals of the NWFP?
9. What planning and management approaches are available for dealing with uncertainty in complex-social-ecological systems?
10. What are uncertainties, research needs, and management considerations related to plan revision in the area of the NWFP?

Key Findings

Perspectives on Conservation in an Era of Global Environmental Change

Overview of human influences on Northwest Forest Plan forests and aquatic-riparian ecosystems—
The effects of humans on forest ecosystems in the Plan area go well beyond timber management impacts and often originate from Earth system processes outside the region. The impacts of human activity to the global environment have become so pervasive that many scientists are beginning to argue that we are in a new geological epoch called the "Anthropocene" (Crutzen 2006, Steffen et al. 2007). Beginning in the early 1800s, this period of rapid industrialization, population growth, and global trade and transportation led to dramatic increases in atmospheric carbon, land use change, altered disturbance regimes, and introduction of nonnative species. (Carey 2016, Corlett 2015, Creed et al. 2016, Lewis and Maslin 2015, Lugo 2015, Sun and Vose 2016, Wohl 2013).

Americans Indians had managed landscapes in the NWFP area for 10,000 years to create conditions that favored food resources and other cultural values; fire was their most important environmental management tool (Charnley et al. 2007, Robbins 1999, White 1993). However, human activity since development of industrial society in the 19th century has brought many additional large changes to species, forests, streams, and landscapes of the Plan area. Although the ecosystems of the NWFP area are relatively unaltered by recent human activity compared to much of the United States, little if any area of the Plan area could be considered uninfluenced by humans. Forests and landscapes have been altered from pre-Euro-American conditions by human activity including logging, plantation management, building roads and trails, dam and levee construction, and fire exclusion. Even forests and watersheds in designated wilderness areas and in large unroaded areas (Strittholt and DellaSala 2001) have been influenced by humans, climate change, introduced diseases, fire suppression, and other factors (chapter 3) (Hessburg et al. 2016).

Nearly all forests within the NWFP area depend on fire to different degrees. Fire exclusion in dry forests, which occupy 43 percent of the Plan area, has had a profound effect on forest structure and composition, native biodiversity, and resilience to fire and climate change (chapter 3). Although fire activity has increased since the NWFP was implemented, most fire-prone forest landscapes are still running a fire deficit in comparison with conditions prior to the mid to late 1800s when fire frequency declined across the dry-forest zone (chapter 3) (Parks et al. 2015, Reilly et al. 2017a). Burned area is also less than would be expected under the current warming climate (chapter 2), for both moist and dry forests, as a result of fire suppression (chapter 3). The decline or elimination of intentional burning by American Indians is also part of altered disturbance regimes and ecosystems in many areas (chapter 11). The wildland-urban interface is also expanding rapidly in the Plan area. This expansion creates challenges to conservation and management including balancing fire protection and fire restoration goals (Hammer et al. 2007, Paveglio et al. 2009), both of which have implications for biodiversity conservation (McKinney 2002).

Biotic changes are also altering the ecosystems of the NWFP area. The extirpation of top predators and invasions by other species have altered food webs and the trophic structure and dynamics of terrestrial and aquatic ecosystems (Beschta and Ripple 2008, 2009; Wallach et al. 2015) across the region. Invasive species such as the barred owl are having significant effects on the northern spotted owl, and the sudden oak death pathogen (*Phytophthora ramorum*) is altering community structure and fire behavior across large areas of northern California and southern Oregon (Metz et al. 2011). Many of these biotic changes are challenging to deal with in a forest-management context because they are rooted in biological processes (e.g., demography, dispersal, and competition), whose control is often beyond the scope of federal forest land managers.

Finally, climate change is increasingly warming all parts of the NWFP region to levels that may exceed climate conditions experienced in the past 1,000 years (chapter 2). These conditions will continue to alter disturbances, ecological processes, plant and animal community structure, and biotic diversity (chapter 2) (Watts et al. 2016), and they will change the expected outcomes of NWFP conservation strategies (chapters 2, 3, 6, and 7).

In summary, forests, watersheds, and biotic communities in the Plan area have been influenced by native peoples for millennia, while human activities during the past 150 years have not merely altered them but reduced their resilience to natural disturbances. This reality has at least three major implications:

1. Some ecological conditions, even in old-growth forests, that are perceived as “natural” have been influenced by human activity.
2. Restorative actions may be needed to achieve goals for desired species and levels of resilience of forests and aquatic ecosystems to climate change and disturbances.
3. Knowledge of historical ecology can help guide us to the future, but management cannot recreate historical conditions.

Conservation in the Anthropocene

Unprecedented ecological shifts or alterations that have occurred across the globe are also described by an emerging concept of “novel” ecosystems, which describes systems that have “departed entirely and irreversibly from their historical analogs” (Hobbs et al. 2009, 2014; Radeloff et al. 2015). One implication of this perspective is that society may have to accept and manage for some of these novel or “hybrid” (seminatural) states, where it is impractical to change existing conditions. Pressures to maintain the status quo of altered conditions will most likely occur where current conditions provide values (supporting local livelihoods, quality of life, or habitats of desired species) that may not have occurred there historically.[2] This perspective does not mean that maintenance or restoration of native communities or historical dynamics could **not** be a goal—only that many scientists increasingly recognize that restoring and maintaining ecosystem integrity based on the historical range of variation of ecosystem attributes may not be attainable in some places, for ecological or social reasons. Sayer et al.

[2] There is a precedent for this in the National Forest Management Act: “…fish and wildlife habitat shall be managed to maintain viable populations of existing native and desired non-native vertebrate species in the planning area” (36 Code of Federal Regulations, sec. 219.19, app. 13).

(2013) and Hobbs et al. (2014) recommended using landscape approaches (e.g., spatially based planning over large and heterogeneous areas and long time frames) that recognize the social dimensions of the problem (e.g., see Cissel et al. 1999, Hessburg et al. 2015, 2016) to identify where it is possible to retain or restore native biodiversity, and where acceptance or management for some novel or "hybrid" (seminatural) qualities or ecosystems might be desirable.

Recognizing the realities of altered ecosystems in the current era has implications for using the 2012 planning rule (USDA FS 2012a). The rule is based on managing for ecological integrity—ecosystems that "…occur within their "natural range of variation[3] and can withstand and recover from most perturbations." The rule also includes the concept of resilience[4] as related to ecological integrity, in the sense that ecosystems with integrity are resilient and able to recover from disturbances (Bone et al. 2016). Given the pace and scale of environmental change, it may be tempting to assume that history or the historical range of variation are no longer relevant to conservation and management; however, this is not necessarily the case (Higgs et al. 2014, Keane et al. 2009, Safford et al. 2012). In conservation and management, the question is not the fundamental value of history, but how it is used (Keane et al. 2009, Safford et al. 2012). Knowledge of the past can inform management in several ways: (1) history as information for how ecosystems function, or as a reference, (2) enriching cultural connections to the land, and (3) revealing possible futures (Higgs et al. 2014). Using history to set precise reference information and targets may become less important and even have negative consequences (in the case of precise targets) as climate and landscape changes continue, but other types of historical information may become more valuable (Hiers et al. 2016, Higgs et al. 2014). Information about the historical range of variation may be derived from simulation and statistical models and from empirical reconstructions of ecological history and its variations (Hessburg and Povak 2015). Safford et al. (2012) provided several recommendations on the use of history in restoration and conservation including the following:

- Do not ignore history; to understand where an ecosystem is going, you must understand where it has been.
- Do not uncritically set management objectives based on historical conditions and avoid aiming for a single, static target.
- Historical conditions may be a useful short-term or medium-term "waypoint" for management, but they will rarely suffice to prepare an ecosystem for an altered future.
- Plan for the future, but do not forget that the past provides our only empirical glimpse into the range of possible futures.

Our advances in understanding the role of ecological history in a time of global change, notwithstanding the development of guiding principles, clear ecological goals, and metrics, is still a significant challenge and must increasingly consider the social dimensions of environmental problems. Managing for ecological integrity rather than more narrowly for the historical range of variation is considered a more realistic approach, but it is not without its own limitations. Managing for ecological integrity includes significant effort to conserve native biodiversity and promote resilience of species and ecosystems to climate change and invasive species (chapter 3) (Hessburg et al. 2016, Wurtzebach and Schultz 2016). But more importantly, managing for ecological integrity recognizes the importance of ecological processes such as natural disturbance agents that control the dynamics of ecosystems. Managing for ecological integrity and using it to guide monitoring and restoration efforts is a relatively new idea that has yet to be widely applied and evaluated in a land management context (Wurtzebach and Schultz 2016). Ecological integrity also includes managing for ecological resilience, which is the capacity to "reorganize

[3] For our purposes in this chapter, we use "historical range of variability" and consider it synonymous with "natural range of variability." See Romme et al. 2012 for comparisons of the definitions of historical range of variability and natural range of variability.

[4] Resilience is the capacity of a system to absorb disturbance and reorganize (or return to its previous organization) so as to still retain essentially the same function, structure, identity, and feedbacks (see Forest Service Manual Chapter 2020 and see also "socioecological resilience" in the glossary). Broad conceptions of resilience may encompass "resistance" (see glossary), while narrower definitions emphasize the capacity of a system or its constituent entities to respond or regrow after mortality induced by a disturbance event.

while undergoing change so as to essentially maintain the same function, structure, identity, and feedbacks" (Walker et al. 2004). The concept of ecological resilience is increasingly used by the Forest Service, but its use has been ambiguous and open to local interpretation (Bone et al. 2016). "Resilience" can be a useful term and goal only when clarified in terms of "resilience of what, to what?" (Carpenter et al. 2001). A major challenge of managing for ecological integrity or resilience, which are both based on understanding ecological history, is the lack of historical knowledge of ecosystems and their variability in many ecological components and processes. A second challenge is knowing future states: there may be multiple possible alternative states of ecological integrity based on certain realities of climate change, invasive species, and changing social values (Duncan et al. 2010, Romme et al. 2012).

Given changing anthropogenic climate change, land use changes, and changes in societal preferences, it is necessary to acknowledge the critical importance of social systems as both drivers of ecological change and as drivers of policy goals and expectations for forests. The importance of the social system suggests that the concept of resilience or integrity should be broadened to focus on managing for social-ecological resilience to global changes within the inherent capacities of earth life-support systems (Carpenter et al. 2001, Folke 2006). Managing for a broader concept of resilience may be more realistic than managing for a specific range of historical variation (Safford et al. 2012, Stine et al. 2014, Wurtzebach and Schultz 2016) or only a biophysical condition. It means focusing on both ecological and social systems and their interactions, and defining resilience not just in terms of recovery of desired ecological or social conditions (which may not be possible) but also adaptation, transformation, learning, and innovation that may lead to new systems that are better adapted to the current biophysical and social environments. Using social-ecological systems frameworks may provide a pathway toward better recognition of how federal forest management is influenced by the interplay of these two systems and where opportunities and barriers lie to reaching federal land management goals, which typically include both ecological and social outcomes. However, managing specifically with social-ecological resilience in mind is still in an exploratory, conceptual stage (Folke 2006), and it remains to be seen how using this framework could improve the effectiveness of federal management.

Fire exclusion—

Although clearcutting of moist old forests had a major effect on ecosystems in the area of the NWFP, altered fire regimes have also affected species and ecosystems. Fire is a critical ecological process in most of the forests of the Plan area, and this chapter devotes considerable attention to complex and sometimes controversial (see chapter 3) fire-related issues. This emphasis on fire is motivated by several factors: (1) fire is a fundamental process that affects most forest ecosystems, species, and human communities of the region; (2) the scientific understanding of the role of fire in both moist and dry forests has increased significantly since the Plan was developed; (3) the 2012 planning rule emphasized ecological integrity and restoration, which are grounded in disturbance ecology—and fire is generally the most significant and altered disturbance in the region; (4) managers have relatively more influence on fire, through suppression policies and management of vegetation, than do most other disturbance processes (e.g., wind or diseases) in the Plan area, and (5) prescribed fire and fire suppression have become a major component of federal land management efforts in policy and budgets in recent years.

The area of the NWFP encompasses a wide range of forest environments and can be broken into two major forest zones (dry and moist) and four different historical fire regimes (chapter 3; fig. 12-2).

One of the most pervasive anthropogenic effects within the drier forest zone, which makes up almost half of the NWFP area, is a major shift in fire regimes as a consequence of fire exclusion and suppression[5] (chapter 3). Lack of fire in dry forests and moist mixed-conifer forests, which historically experienced frequent to moderately frequent wildfire, altered forest structure and composition, and had cascading ecological effects on ecosystems and species.

[5] Fire exclusion is the minimizing or removal of wildfire as a keystone ecological process, either indirectly as a result of livestock grazing, roads, railroads, agriculture, and development, or directly via intentional fire suppression and prevention activities. Fire suppression is the act of putting out wildfires.

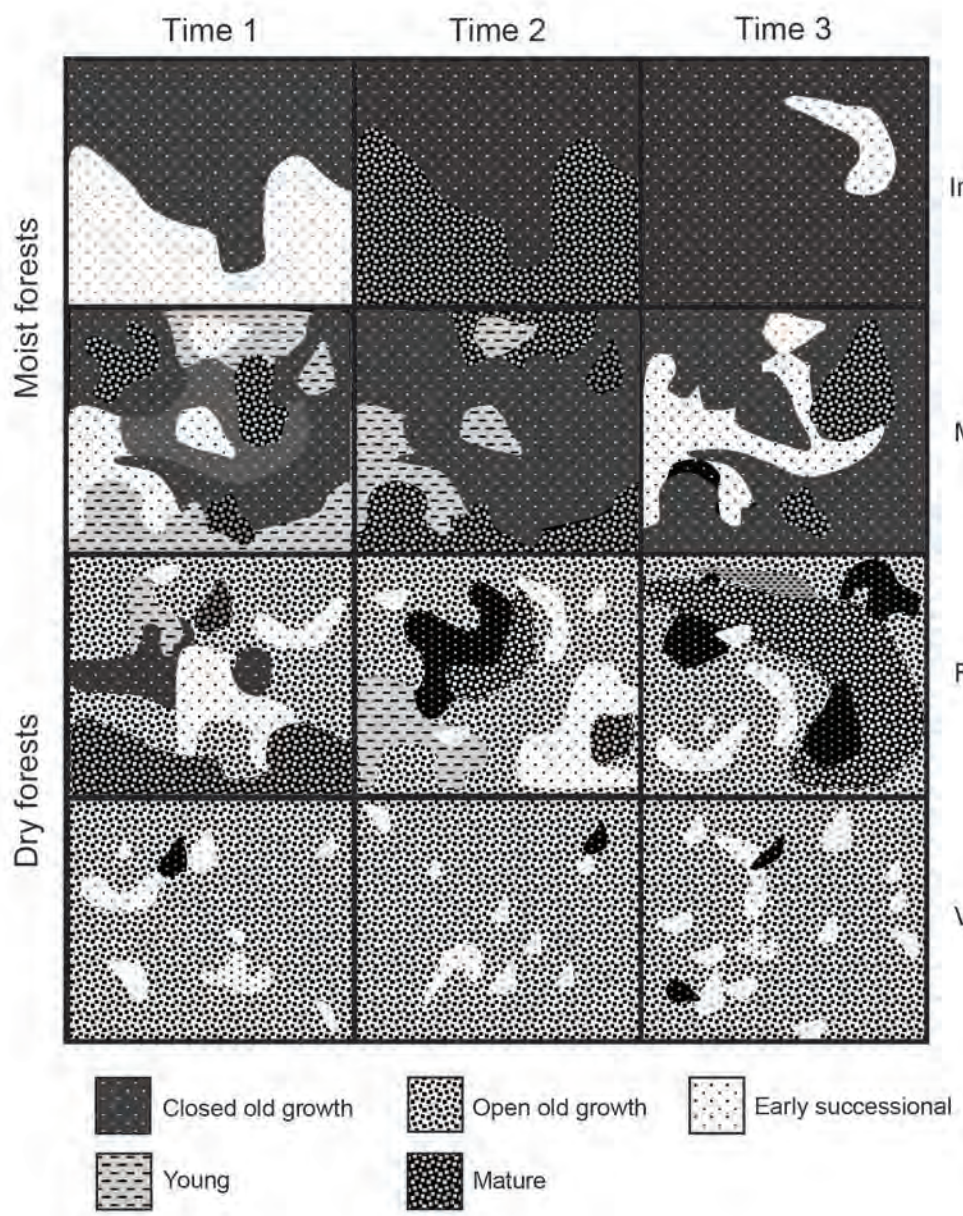

Figure 12-2—Idealized spatial patterns of forest successional stages in the two major forest zones and the four historical disturbance regimes of the Northwest Forest Plan area at three arbitrary points in time. Time 1 and 2 are separated by about 100 years; time 3 is at least 400 years later so that patterns from time 1 and 2 are not evident. See chapter 3 for more information. Illustration adapted from Agee 1998.

These effects include:

- Increased forest density and abundance of shade-tolerant tree species.
- Loss of early-successional, open-canopy young, and open old-growth forest types, and altered successional pathways.
- Increased area of dense, young, multistoried forest vegetation that is used by the northern spotted owl and other late-successional species.
- Decline in habitats for species that use open, fire-frequent forests or early-successional vegetation.
- Less frequent fire, but when fires occur under extreme weather conditions, they can result in uncharacteristically large, high-severity patches of fire.

Larger patches of high-severity fire in this historical regime may have undesirable short- and long-term effects in terms of accelerated upland erosion, loss of forest cover to continuous shrubfields, chronic stream sedimentation, chronically elevated bark beetle populations, and reduction of services from forests of all seral stages (chapter 3). Large patches of high-severity fire in forest ecosystems that historically burned with frequent but low-severity fire can kill many of the large, old, fire-resistant trees that survived fires in the past. Such trees are considered a regionally and globally significant "keystone ecological structure" in a wide range of ecosystem types (Lindenmayer et al. 2014). Extremely large and unusually severe fires also have major social and economic impacts through heavy smoke, evacuations, greenhouse gas emissions, costs of firefighting, lost productivity, and threats to and loss of lives, income, and property. Such social and economic impacts are expected to increase, particularly in the NWFP area, as climate change results in more hazardous fire and smoke conditions (Liu et al. 2016). The landscapes left following extremely large and uncharacteristically severe fires can pose significant management challenges too, as reforestation treatments can be costly and often dangerous in many burned areas. Planting may be needed to avoid persistent loss of forest cover in some areas, yet reintroducing fires while protecting the investment in young, fire-susceptible trees is particularly challenging.

Fire exclusion has also had an effect in moist forests that historically experienced long fire-return intervals. The effects are different than in dry forests, and relate mainly to decreased occurrence of diverse early and mid-successional and nonforest (meadow) vegetation. High levels of fuel accumulation at stand scales and landscape connectivity of fuels are characteristic of moist productive forests that grow for many decades or centuries without fire. However, lack of fire in drier parts of moist forests may lead to more homogeneous stand structures and fuel beds than occurred historically, when infrequent fire created a mosaic of seral stages. The broader ecological implications (e.g., ecosystem function and fire behavior) of these changes are not clear and are in need of further research (Tepley et al. 2013).

Social perspectives on altered forests—
The challenges to managing for ecological integrity, resilience, and desired species in the NWFP area are both ecological and social. In moist forests, where fire was and continues to occur infrequently, uniform plantations, the time required for succession to old growth (centuries), and fragmentation of older forests are key ecological concerns. In dry forests, which historically experienced very frequent and moderately frequent fire-regimes (chapter 3), the ecological constraints on management include the fact that, with build-up of fuels in historical fire frequent regimes, fire often cannot be reintroduced as prescribed fire without first reducing fuels via mechanical means. And, more significantly, climate change and invasive species will continue to alter fire regimes and vegetation dynamics, making these increases in fuels even more consequential.

The social and economic constraints to widespread restoration of fire in fire-frequent ecosystems are large and include agency budgets, limited workforce capacity, air quality regulations, social acceptability of prescribed fire, lack of markets for restoration byproducts, and the risk of losing other values (Charnley et al. 2015; Collins et al. 2010; North et al. 2012, 2015; Ryan et al. 2013) (chapter 8). Public support for restoring fire to the landscape will be required to make progress (North et al. 2015). In addition, the costs of restoring fire through mechanical treatments and prescribed fire are high (Houtman et al. 2013), and to be fully funded by Congress would require significant re-investment in public forest lands at levels beyond current annual wildfire suppression and preparedness funding. For example, the recent Forest Service budget appropriations for hazardous fuels reduction are less than one-fifth what they are for fire suppression (Charnley et al. 2015), and current rates of restoration treatments in many areas of the Western United States are well below what is needed for restoration (North et al. 2012, Reilly et al. 2017a, Spies et al. 2017, Vaillant and Reinhardt 2017). This deficit has led some to call for more use of managed natural ignitions (North et al. 2012). Some initial studies indicate that managed and some unmanaged wildfires have the potential to increase the scale of restoration benefits (Meyer 2015, Reilly et al. 2017b), though the relative benefits and costs of this approach (table 12-1) are not yet fully understood and will likely differ across the fire regimes of the Plan area (chapter 3).

Table 12-1—Summary of possible (known and hypothesized) major tradeoffs (effects) associated with current management activities and original ecological and socioeconomic goals

Management activity	Closed-canopy old-growth structure and function	Northern spotted owls	Marbled murrelets	Other late-successional old-growth species	Aquatic habitats	Timber and nontimber supply	Local economies	Tribal ecocultural resources
Suppression of wildfire	Increases shade-tolerant tree species and canopy cover Protects existing old-growth trees from loss **Reduces open old-growth types and landscape diversity** **Reduces area of diverse early-successional forest** **Reduces resiliency in some fire regimes**	Protects habitat in northern part of range Increases area of nesting and roosting habitat in fire- prone areas **Can increase landscape-scale risk to owl habitat by promoting larger high-severity patches of wildfire** **Can reduce habitat quality in southern part of range where habitat includes smaller patches of early-successional and nonforest vegetation**	Protects habitat	Protects habitat for species that prefer dense multilayered canopies Variable and poorly understood effects	Increases density of trees in riparian areas Can increase shade on streams	Protects forests scheduled for timber harvest Can increase risk of loss of timber resources where forest fuels accumulate Encourages growth of nontimber forest products that favor closed, mature forest conditions	Can provide short-term economic benefit to communities near wildfire Can promote recreation in unburned forests Protects homes and structures	**Reduces quantity and quality of ecocultural resources associated with early-successional forests, nonforest communities, and open old-growth forests**
Variable thinning plantations in uplands and riparian areas	Accelerate development of large trees Increases vegetation heterogeneity, diversity, and understory layers **Can reduce dead wood if trees removed**	Accelerate development of large nest trees and multiple canopy layers May reduce populations of red tree voles, a prey species, in the short term	Accelerates development of crowns with thick limbs **May create habitat for predator species**	Variable effects; some positive and negative short-term and longer term effects	Accelerate large trees Increases vegetation heterogeneity, diversity, and understory layers Effects likely variable with location in stream network **Can reduce dead wood inputs to streams if trees removed** **Can reduce shading and increase stream temperatures**	Can provide wood products and bioenergy and help support local processing infrastructure May favor nontimber forest products (NTFPs) associated with less dense forests **Not sustainable in long run because plantations may become too old or no longer benefit from restoration management**	Creates jobs for local communities	Can enhance ecocultural resources associated with less dense forests

Table 12-1—Summary of possible (known and hypothesized) major tradeoffs (effects) associated with current management activities and original ecological and socioeconomic goals (continued)

Management activity	Closed-canopy old-growth structure and function	Northern spotted owls	Marbled murrelets	Other late-successional old-growth species	Aquatic habitats	Timber and nontimber supply	Local economies	Tribal ecocultural resources
Thinning to restore resilience to fire-suppressed forests	Can help to maintain large, old, fire-resistant trees Can increase old-forest diversity in fire-dependent disturbance regimes Can reduce fire spread rates and reduce sizes of high-severity fire patches	Can protect patches of nesting and roosting habitat from large fires Reduces habitat quality at site level	May reduce loss of large nest trees to wildfire May create habitat for predator species	Same as variable-density thinning	Same as variable-density thinning	Can provide wood for local communities May not be sustainable in long run because restoration may shift toward prescribed fire as stands are repeatedly treated	Creates jobs for local communities Open forests may be preferred by some recreationists Reduces risk of wildfire to wildland-urban interface communities Open, managed forests may not be preferred by some recreationists	Can enhance ecocultural resources associated with less dense forests, including large nut-bearing trees
Prescribed fire	Same as above	Same as above	Unknown	Unknown	Unknown		Can reduce fuel loads and fire risk to local communities Could potentially support local crews dedicated to using managed fire Smoke can cause health and safety concerns	Can enhance eco-cultural resources, including fungi, plants whose germination is enhanced by smoke, and plants affected by insect pests
Early-seral creation in closed-canopy forests over 80 years in matrix to mimic wildfire effects	Can increase habitat diversity and provide habitat for species dependent on open, habitats and dead trees. **Not compatible with dense old-growth forest structure at stand scales** **Cannot fully replace wildfire effects if does not include prescribed fire or occur in older stands**	May not be compatible with habitat in part of the range Can be compatible at landscape scales in southern parts of range	**Not compatible with habitat at any scale**	Likely reduces habitat, but some species may benefit from juxtaposition of old and young habitats	Can increase light to streams and stream productivity Can increase habitat diversity and promote longer term integrity of stream ecosystems **Can decrease shade and increase stream temperatures in some contexts**	Can provide wood products and help support local mill infrastructure Favors NTFPs associated with early-seral forest	Can benefit local economies	Can enhance eco-cultural resources associated with less dense forests, including various understory plants and game animals

Table 12-1—Summary of possible (known and hypothesized) major tradeoffs (effects) associated with current management activities and original ecological and socioeconomic goals (continued)

Management activity	Closed-canopy old-growth structure and function	Northern spotted owls	Marbled murrelets	Other late-successional old-growth species	Aquatic habitats	Timber and nontimber supply	Local economies	Tribal ecocultural resources
Early-seral creation in closed-canopy plantations to mimic wildfire effects	Can increase habitat diversity and provide habitat for species dependent on open habitats and dead trees Does not reduce current area of old growth **Will reduce future amounts of dense old-growth forest structure at stand scales and landscape scales** **Cannot fully replace wildfire effects if does not include prescribed fire or occur in older stands**	**Could have some negative effects on dispersal habitat** May be compatible at landscape scales in southern parts of range	**Does not affect current habitat although it could increase edge and occurrence of nest predators at landscape scale**	Does not likely affect habitat for these species, and some species may benefit from juxtaposition of old and open canopy conditions	Can increase light to streams and stream productivity Can increase habitat diversity and promote longer term integrity of stream ecosystems **Can decrease shade and increase stream temperatures in some contexts**	Can provide wood products and help support local mill infrastructure Favors NTFPs associated with early-seral vegetation	Can benefit local economies	Can enhance ecocultural resources associated with less dense forests, including various understory plants and game animals
Planting after wildfire	Planting of key tree species may benefit forest recovery	Planting of key tree species may benefit longer term recovery of habitat	May benefit long-term recovery of habitat	May benefit long-term recovery of habitat	Planting of key tree species may benefit longer term recovery of habitat	Can benefit long-term recovery of forest resources	Some work for local communities	Recovery of conifer forest may benefit some resources, but recovery of nonconifer species (e.g., hardwood trees and shrubs) are also important concerns, as well as ability to restore frequent fire regime
Salvage after wildfire	**Removal of large dead wood reduces habitat for many wildlife species** **Logging of dead trees can kill regeneration and increase erosion** Removal of small dead trees in fire-excluded forests may reduce impacts on soil if reburn occurs	**Removal of dead trees likely not compatible with habitat**	NA (does not use postfire environments or dead trees)	**Removal of dead wood likely not compatible with habitat for some of these species**	**Removal of dead trees (especially large ones) may work against natural riparan/aquatic recovery processes**	Can provide timber and support local mills	Can provide jobs for local communities	

Table 12-1—Summary of possible (known and hypothesized) major tradeoffs (effects) associated with current management activities and original ecological and socioeconomic goals (continued)

Management activity	Closed-canopy old-growth structure and function	Northern spotted owls	Marbled murrelets	Other late-successional old-growth species	Aquatic habitats	Timber and nontimber supply	Local economies	Tribal ecocultural resources
Road removal	Can reduce spread of invasive species to older forest blocks Can reduce edge effects **Can reduce access for restoration management**	Unknown	May reduce corvid populations that prey on nests	Unknown	Reduces erosion potential Reduces risk of landslides and debris flows Increases fish passages through stream networks	**Can reduce access for timber management and NTFP gathering**	**Can reduce access for recreation and other forest uses** Can improve water quality	Can benefit desired aquatic resources, **but also can limit access to desired ecocultural resources**
Managing wildfire for ecological benefits	Can increase diversity of old-forest types Can increase landscape resilience to future fire **Can destroy old-growth forests and large old trees**	May reduce risk of loss from fire to surviving patches of habitat **Can eliminate habitat**	May reduce risk of loss from fire to surviving patches of habitat **Can eliminate habitat**	Not well known, but likely similar to spotted owl response	Can increase habitat diversity and promote longer term integrity of stream ecosystems **Can decrease shade and increase stream temperatures**	May damage and reduce value of trees that were scheduled for wood production Can reduce fuels and lower risk of loss of unburned forests	Can increase area of habitat for game species and increase hunting use Can reduce future risk of large high-severity fires that threaten local communities	Can promote ecocultural resources by restoring fire and more open structure as above, **but there may also be concerns about effects of large high-severity patches in untended areas on desired resources (e.g., mature oaks)**

NA = not applicable.

Note: Effects in regular type are generally consistent with a goal; effects in boldfaced type are generally not consistent with a goal or have negative effects on other goals not emphasized in the Northwest Forest Plan. Effects may differ with spatial and temporal scale and with geography. The effects are generalized, so they may not apply in all contexts and there may be considerable uncertainty, especially regarding the effects of extreme fires. For detailed discussions of these effects, see individual chapters of this synthesis.

Another social challenge is that some altered conditions of ecosystems in the NWFP area may be desirable to some people, despite being highly departed from historical conditions, and at greater risk to loss from wildfire and drought. For example, the denser forests that have developed in forests with very frequent and moderately frequent fire regimes now support more area of habitat for northern spotted owls and other dense forest species such as goshawks (*Accipiter gentilis*) (chapters 3 and 4) than they did under the historical fire regime. Some groups may favor maintaining some dense stands; for example, the Klamath Tribes expressed a concern for promoting mule deer (*Odocoileus hemionus*) habitat by retaining dense tree patches as deer hiding cover within ponderosa pine (*Pinus ponderosa*) forests that were historically open in their ancestral lands on the Fremont-Winema National Forest (Johnson et al. 2008). Based on discussions with stakeholders who participate in central Oregon forest collaborative groups, we have observed that some stakeholders value the aesthetic and wildlife values of the fire-excluded, multilayered grand fir (*Abies grandis*) and white fir (*A. Concolor*) forests, which appear to fit an idealized old-growth forest based on wetter old-growth types. A study from moist forests (moderately frequent, mixed-severity fire regime) in the western Cascade Range of Oregon indicates that tall, multilayered forests that develop in the absence of fire may buffer climate change effects on the microclimate for wildlife (Frey et al. 2016a, 2016b). It is unknown if that finding applies to fire-excluded dry forests. Finally, such forests may be more desirable to some people simply because they occur without active management (except for suppression), which may be simply mistrusted (e.g., see DellaSala et al. 2013 and "Trust and collaboration" section below).

Although some people see benefits in dense fire-excluded forests, many see the risks (see discussion in Brown 2009). For example, many stakeholders who participate in the central Oregon forest collaboratives mentioned above are concerned about the increased risk of widespread tree mortality resulting from severe fire, drought, and insects, and some see opportunity for economically feasible restoration treatments that would remove established grand fir/white fir established over the past 100 years in favor of fire-tolerant and drought-tolerant tree species.[6]

Invasive species—

Species invasions or range-expansion species native to North America have also affected the native biota of the NWFP region (chapter 6). Invasive species are widespread—more than 50 percent of inventory plots in almost all physiographic provinces of the Plan area contain nonnative plant species (Gray 2008), but most of them do not get much attention. An exception is the barred owl, which is an example of an invasive species (Peterson and Robins 2003) (some have called it a "native invader species") (Carey et al. 2012) that has become a major threat to the viability of northern spotted owl populations (chapter 4). Although the barred owl may be the most prominent example, there are many other examples in the NWFP area of species that may have been exotic or native to the region but are having undesirable effects on other species and ecosystems as a result of landscape and other anthropogenic changes. For example, native corvid (the crow/raven family) populations have expanded as a result of human food waste and human disturbance of vegetation (Marzluff and Neatherlin 2006, Peterson and Colwell 2014), and corvids prey on the nests of marbled murrelets (*Brachyramphus marmoratus*) (chapter 5).

The widespread expansion of true firs into pine forests, where fire has been excluded, could also be termed "native invader" (Carey et al. 2012, Simberloff 2011) species that were once rare or uncommon in a landscape, but now have become so abundant that they are altering community (e.g., through competition) and ecosystem dynamics (disturbance regimes) in undesirable ways. In the case of true firs in dry forests, their expansion has altered forest composition, structure, and fire regimes, and they are difficult to control by virtue of their copious seed rain (Hessburg et al. 2016, Stine et al. 2014), which can lead to rapid recolonization of disturbed areas.

The impact of barred owls on northern spotted owl populations is profound; it is not known if this impact can be reversed or at least stabilized across the spotted

[6] Merschel, A. 2017. Personal communication. Graduate student, Oregon State University, Department of Forest Ecosystems and Society, 321 Richardson Hall, Corvallis, OR 97331.

owl's range through efforts to remove them. An ongoing, large-scale experiment will shed more light on this future (USFWS 2013, Wiens et al. 2016). A proposal to remove an established species to protect another is a major challenge to society from ecological, economic, and ethical perspectives (Carey et al. 2012, Livezey 2010), but it is not unprecedented (e.g., Wilsey et al. 2014). Multiple approaches to northern spotted owl conservation, including large-scale experiments and landscape-scale forest restoration experiments, can provide more learning opportunities and more understanding of ways to promote resilience of the subspecies. In the long run, the northern spotted owl may be locally or completely displaced by the barred owl. From an ecosystem perspective (e.g., productivity, food webs, trophic cascades), the effect of loss of northern spotted owls on the forests and vertebrate communities is unknown, but it is hypothesized that prey species and other competing native predators may experience changes in behavior, abundance, and distribution as a result of predation by the barred owls, which has a broader prey base and occurs at higher densities than the northern spotted owl (Wiens et al. 2014).

Invasive species occur in aquatic and riparian ecosystems as well. Across the Plan area, 63 nonnative species and species groups are identified as regional aquatic-riparian invasive or nuisance species priorities (chapter 7). Of these, 31 (49 percent) species or species groups were designated as "high concern" and inventoried by the NWFP's Aquatic Riparian Effectiveness Monitoring Program (AREMP) in 2016. Nonnative species are not always harmful to native fishes or their habitats, but in many instances they can (1) compete with, prey upon, hybridize with, or infect native species with novel pathogens; (2) greatly alter the structure of food webs; or (3) cause habitat changes that reduce the productivity of desirable aquatic organisms. Climate change will likely influence the expansion of nonnative plant and animal species in the NWFP area, while at the same time either reducing or even extirpating native species (Dale et al. 2001, Garcia et al. 2014, Urban 2015).

Other disturbance agents—
Novel ecological conditions are also a concern where ecosystems are subject to multiple disturbance agents. For example, stands infested by the sudden oak death pathogen have increased potential for high burn severity (chapter 3), while rodenticides used in illegal marijuana cultivation and the spread of barred owls may tax populations of sensitive fishers (*Martes pennanti*) and northern spotted owls, respectively, so that they become more sensitive to other disturbances (Gabriel et al. 2012, 2013) (chapter 6). As an example from aquatic systems, the combination of climate change, severe fire, tree mortality, and floods may increase the potential for debris flows (Cannon and DeGraff 2009) and ensuing debris jams at culverts and bridges. Such flood impacts can threaten life, property, and access; damage expensive infrastructure; and impair stream functions by causing stream bank erosion and channel incision. The challenges to restoring fire and geomorphic disturbances to these ecosystems are daunting. Landscape and social-ecological systems perspectives are needed to meet the broad Forest Service goal (http://www.fs.fed.us/strategicplan) of increasing the resilience of forests and aquatic ecosystems to fire and climate change while meeting the specific late-successional forest goals of the NWFP (Fischer et al. 2016, Hessburg et al. 2015, 2016; Reeves et al. 1995, 2016; Stephens et al. 2013).

Perspectives on Reserves in an Era of Global Environmental Change

Views of the conservation community—
The scientific community's response to the cumulative effects of climate change, land use change, and invasive species has led some to call for new approaches to conservation (Millar et al. 2007, Wiens 2016). Some researchers have affirmed that "tomorrow's landscapes may become so altered by human actions that current management philosophies and policies of managing for healthy ecosystems, wilderness conditions, or historical analogs will no longer be feasible" and will require a new land ethic (Keane et al. 2009). Others have advocated for a new science of conservation rooted in the integrated nature of social-ecological systems (as mentioned above) and designed to promote human well-being as well as biodiversity conservation, particularly where poverty is pervasive, through judicious and sustainable use of ecosystems rather than strict preservation (Kareiva and Marvier 2012). In the conservation ethics

literature, the contrast is often made between humanism, emphasizing the importance of productive human use of natural resources, and biocentrism, emphasizing a primary goal of maintaining ecological integrity (Stanley 1995). These new perspectives have received pushback from some conservation biologists. For example, Miller et al. (2014) and Doak et al. (2014) argued that conservation centering on human values, now often organized using the framework of ecosystem services, is an "ideology" that (1) is not new (e.g., it reflects ideas advocated by Gifford Pinchot a century ago), and (2) does not address the root causes of lost biodiversity, which they described as "unabated consumption and increasing human populations." Instead, they emphasized preservation of biodiversity through large networks of protected lands arranged to foster connectivity and some sense of permanence. They devoted little attention, however, to what such protection means in disturbance-dependent and highly dynamic systems with a strong history of human impacts, or in systems in which invasive species are widespread, or where permanence of certain vegetation, habitat conditions, or biotic communities is simply unattainable.

These debates notwithstanding, nature reserves (also termed "protected areas") including wilderness areas, remain key components of conservation strategies and forest planning around the world (Simončič et al. 2015, Watson et al. 2014). E.O. Wilson, in his book *Half-Earth, Our Planet's Fight for Life* (Wilson 2016), challenged society to set aside half of the Earth's lands and seas to conserve biodiversity in reserves equivalent to World Heritage sites. Other scientists have echoed a similar call in advocating for an extensive reserve network focused on riparian areas across the United States (Fremier et al. 2015). Although we are a long way from these goals (e.g., 10 percent of U.S. land is in a protected area (Aycrigg et al. 2013), the area of wildland reserves or protected areas is growing (Götmark 2013) and have made essential contributions to maintaining populations of threatened species, or have slowed their rate of loss. In the NWFP area, reserves[7] on federal lands constitute about 80 percent of the federal forest area and 28 percent of the total forest area on public and private forest lands (chapter 3). Conservation biologists have argued that protected areas are necessary but not sufficient to meet conservation objectives (Margules and Pressey 2000, Noss et al. 1997, Rayner et al. 2014). Governance and management of reserves are as important as the designation of the reserve on a map. For example, ineffective governance of protected areas in many countries has not kept out detrimental land uses such as development, intensive logging for timber, degradation from invasive species, and illegal hunting (Watson et al. 2014). In addition, reserves may need active management to meet biodiversity goals (Lemieux et al. 2011, Lindenmayer et al. 2000) or to meet needs of local communities that are compatible with biodiversity goals (Watson et al. 2014). Pressey et al. (2007) suggested that appropriate actions within or outside reserves may include "control of invasive species, management of disturbance regimes, quarantine against disease, restrictions on harvesting, and restoration." In summary, the literature provides overwhelming support for the idea that reserves have an essential role to play in conservation (e.g., slowing rates of losses of native biodiversity), if they are effectively managed (Watson et al. 2014).

Many types of reserves—

Globally, there are many types of reserves, depending on a variety of existing conditions and long-term intentions. For example, the International Union for the Conservation of Nature (IUCN) defines seven categories that encapsulate the variety of purposes and specific contexts for a reserve (Spies 2006) (chapter 3). These range from category 1a, "strict nature reserve," which still allows some light human uses, to category 6, which allows sustainable use of natural resources, such as agroforestry. Biosphere reserves defined by the IUCN can include "core areas" or **sanctum sanctorum** which are open only to those with special scientific permits, and are bordered or surrounded by buffer zones with various allowances for ingression and resource use and extraction (e.g., Cumming et al. 2015, Peine 1998, Taylor 2004). These categories of reserve designs differ depending on the amount of human activity and use that is considered compatible with the primary conservation objectives of the reserve (Lausche and Burhenne-Guilmin 2011), although many of the IUCN reserve design architectures, including the core/buffer design, are not implemented as such in the United States.

[7] Designated wilderness areas account for about 42 percent of federal reserves, not including riparian reserves, and encompass roughly 7.1 million ac (including some national parks like Olympic and Mount Rainier National Parks).

In general, reserves are defined in terms of objectives and management actions that are needed or allowed, and in terms of actions that cannot be allowed in order to achieve primary conservation objectives, that is, by specifying human activities that are permitted or excluded. As a result, reserves exhibit a hierarchy of conservation goals, as demonstrated in the NWFP area, in which conservation of functional older forest and northern spotted owl habitat are the top priorities in late-successional reserves (LSRs), at least in the wetter provinces. In the drier provinces, according to the latest U.S. Fish and Wildlife Service recovery plan for the northern spotted owl, restoration becomes an "overlapping goal" with northern spotted owl habitat that must be reconciled (USFWS 2011). In addition, the 2012 planning rule emphasizes managing forests for ecological integrity and resilience to climate change, a goal that is not mentioned in the standards and guidelines for the LSRs (USDA and USDI 1994b). Thus, reserves as they have been conceived and implemented globally and regionally exist along a continuum of uses and management approaches, based on goals and cultural context.

Social controversies around reserves—
Although reserves are a cornerstone of conservation biology, they exist in a larger social context in which they may not be viewed so favorably. The idea of a nature "reserve" is a cultural construct associated with Euro-American notions of humans as distinct from nature (Cronon 1996) (see chapter 11). Rules governing permissible activities in protected areas or reserves differ across the globe (Simončič et al. 2014) and can be controversial (Brockington and Wilkie 2015). Reserves, with strict rules concerning management or resource extraction, have been criticized for threatening livelihoods by denying access to resources, and for not recognizing that nature changes as a result of disturbance and succession (Bengtsson et al. 2003); tribes, in particular, have expressed such concerns about NWFP reserves (see chapter 11). Often, the costs of reserves are experienced by local people, while benefits disproportionately accrue to people some distance away (Brockington et al. 2008). Controversies about reserves have several dimensions:

1. They are often written into the founding stories of a nation or culture (e.g., old-growth forests in the Pacific Northwest (Spies and Duncan 2009) and therefore touch deep emotions.
2. The local effects on people can be beneficial (e.g., amenity values) (Hjerpe et al. 2017, Holmes et al. 2016) or negative (e.g., reserves that restrict access to commodities or subsistence goods and can increase poverty in rural areas (Adams 2004, West et al. 2006).
3. The goals for nature in the reserves can be ambiguous or difficult to achieve given that nature is multidimensional, dynamic, and often influenced directly or indirectly by human activity.
4. Achieving biodiversity goals often requires management, especially given effects of past land use change, invasive species, and climate change, which can be controversial if stakeholders hold different values for reserves.
5. Reserves, which typically occupy a small part of most landscapes, are not sufficient by themselves to provide for biodiversity (Franklin and Lindemayer 2009).
6. They are flash points for politics of conservation related to land use and national and regional debates about values expressed through different interest groups (Brockington and Wilkie 2015).

Reserves in dynamic ecosystems—
Some conservation biologists and legal experts (e.g., see Craig 2010) recognize the problem of conserving biodiversity in fixed reserves, where vegetation structure and composition, disturbances, climatic influences, and plant and animal communities are highly dynamic. Approaches to reserves in dynamic systems fall along a gradient in terms of size and objectives. At one end of this gradient are relatively small fine-filter or coarse-filter (e.g., static vegetation states) reserves that some (Alagador et al. 2014, Bengtsson et al. 2003, Bisson et al. 2003, Lemieux et al. 2011) suggest could be moved in response to changing environmental conditions (e.g., disturbance, invasive species, climate change). Some of the late-successional reserves (LSRs) in the Plan area are small and would fit into this category in terms of size and objective. At the other end of the gradient are large (coarse-filter) reserves that are managed to accommodate dynamic ecosystem processes (e.g., disturbance and succession) (Bengtsson et al. 2003,

Pickett and Thompson 1978). Some of the large LSRs may meet this size criterion relative to fire sizes (chapter 3), but are primarily focused on maintaining or increasing one successional state—dense old-growth forests. The first type of reserve approach—in which new protected areas are established and old ones decommissioned in response to changing environmental conditions—has received little formal evaluation, and we are not aware of any publications that document where a reserve was decommissioned and replaced with a new one or an alternative approach in the United States. However, dynamic habitat conservation approaches (which do not use the term "reserve") are being used for two endangered forest species in fire-prone forests: the red-cockaded woodpecker (*Picoides borealis*), which depends on fire to maintain old-growth pine (*Pinus* sp.) forests of the Southeastern United States, and the Kirtland's warbler (*Dendroica kirtlandii*), which depends on dense young jack pine (*Pinus banksiana*) forests that regenerate following wildfire or logging in Michigan (Moore and Conroy 2006, Spaulding and Rothstein 2009). These cases indicate that alternatives to fixed no-management reserves for conservation of listed species of fire-prone landscapes exist, but examples do not exist for old-growth forests and northern spotted owls.

A simulation study in Quebec (Rayfield et al. 2008) evaluated static and dynamic habitat reserve strategies for American marten (*Martes americana*), a species that uses mature coniferous forests. The results indicated that the dynamic reserve strategy supported more high-quality habitat over a 200-year simulation than did static reserves. The locations of new reserves were constrained by fragmented forest patterns created through logging and wildfires in surrounding non-reserve areas. These findings have two major implications: (1) if reserves are focused on just one successional stage or habitat for a single species, they may not be effective in the long run in fire-prone landscapes; (2) if dynamic conservation strategies are to be successful in the long term, the surrounding nonreserved areas must be managed in a way such that habitat replacement options for target species are available when reserved areas are no longer functioning as intended. They also highlight the importance of investing in and supporting private lands conservation to enable possible future replacement options associated with private lands, and to provide habitat functions for species that are not restricted to reserves, or other species that were not the focus of the reserve.

In contrast to the above species-centric reserves or conservation areas, large reserves based on dynamic coarse-filter objectives (e.g., ecosystem patterns and processes) will more likely meet conservation goals than fixed-area reserves for particular species or vegetation conditions. Large protected areas (e.g., larger than 25,000 ac) (more than 100 of the existing LSRs are larger than 25,000 ac) could better support the full range of natural disturbances within their boundaries than could small reserves (see chapter 3 for evaluation of the dynamics of LSRs as a function of their size). In such cases, it may be more possible to capture inherent ecosystems dynamics—natural and intentional management disturbances used to change the vegetation in ways that match the biophysical and topographic template and contribute to overall successional diversity and resilience. Management may still be needed to achieve specific goals (e.g., creation of fire-resistant forest structures and heterogeneous fuel beds) and could promote resilience of some components of ecosystems components to climate change, drought, and fire.

Challenges to management of small and large reserves are significant. For small reserves with a narrow species or vegetation state objectives, moving reserves dynamically to deal with climate change, disturbance, and other changes may be more effective at maintaining biodiversity than fixed reserves (Bengtsson et al. 2003). However, a dynamic reserve in which adjustments to standards, guidelines, and reserve boundaries would be more difficult to implement, monitor, and govern than one in which reserves are fixed in perpetuity in location and management guidelines. Moving reserves would likely require an ongoing and robust decisionmaking process that involved diverse stakeholders and a high level of trust. In large reserves, with both ecosystem and species goals, there would likely be less need or motivation to move reserve boundaries because there would be fewer options for reserve placement in the larger landscape and because overall vegetation conditions in large reserves would be less likely to change as a result of disturbances.

The management of large reserves for ecological integrity and species goals would require development of standards and guidelines for dealing with natural disturbance events and restoration activities intended to restore ecological processes (e.g., fire and hydrological disturbances) while providing for any other goals (e.g., particular species or vegetation states). In addition, standards and guidelines would need to be flexible enough to deal with unforeseen future issues, such as invasive species or climate change effects that might require different types of intervention to meet ecological goals. Changes to reserve boundaries or to standards and guidelines in both large and small reserves would also involve consideration of environmental justice and equity, especially for people living and working near the reserve.

Although the idea of dynamic reserves, or reserves for dynamic ecosystems, may be relatively new in the literature (e.g., Harrison et al. 2008), the literature also lacks studies of the conservation of late-successional forests (i.e., dense older forests) in reserves within dynamic fire-prone ecosystems, which is the situation in the dry forests of the NWFP area. The NWFP was meant to be adaptive, and changes to reserve standards and guidelines might be considered given climate change, fire occurrence, invasive species, and species movements or other relatively new ecological concerns. See “Reserves” on p. 952 for more discussion of NWFP reserves and challenges of implementing reserves in dynamic ecosystems.

Key Social Components of the Social-Ecological Systems of the Northwest Forest Plan Area

Ecosystem services—

The ecosystem services concept, largely developed since the NWFP was initiated, recognizes that forests and other natural systems support many benefits to human communities beyond timber and water supply that were emphasized at the creation of national forests. The recognition of these diverse benefits is not new (Kline et al. 2013); however, efforts to explicitly recognize them within a broader “ecosystem services” framework is somewhat new, and in the process of being incorporated into federal forest management (Brandt et al. 2014; Bruins et al. 2017; Deal et al. 2017a, 2017b; Long et al. 2014; Moore et al. 2017; Penaluna et al. 2017; Smith et al. 2011). Categories of ecosystem services recognized by the Millennium Ecosystem Assessment are **provisioning services** (e.g., food and fiber), **supporting services** (e.g., pollination, soil formation, and nutrient cycling), **regulating services** (e.g., carbon sequestration and water purification), and **cultural services** (e.g., spiritual, symbolic, educational, heritage, and recreational services) (Wallace 2007). Many resource management systems in the United States took such services for granted until relatively recently, as the limits and vulnerabilities of ecosystems in supporting these benefits have become more apparent. However, ecosystem valuation is often difficult owing to the lack of markets for many collective goods. Forest managers often have difficulty assigning value to many features of the forests they manage in ways that appropriately inform decisionmaking (Smith et al. 2011). Kline et al. (2013) indicated that full development of ecosystem services frameworks for public lands will be constrained by lack of ecological data for planning units and economic capacity in terms of models and staffing. They argue that, given these limitations, efforts to apply ecosystem services concepts should include qualitative methods that can be used with stakeholders even without more detailed quantitative information.

Critics of the ecosystem service concept have argued that it has constrained thought and conservation of nature by focusing on “monetization and financialization of nature” that actually devalues nature by ignoring other values that cannot be monetized, and it creates “make-believe markets” that are not effective in conserving nature (Silvertown 2015). These other values include aesthetic, spiritual values and intrinsic values that might come under the title of “cultural services” but are not suited to an instrumental thinking approach (Batavia and Nelson 2017, Cooper et al. 2016, Winthrop 2014). Others have responded by saying that the ecosystem services concept has value beyond market and monetization, can take many forms (Schröter and van Oudenhoven 2016, Wilson and Law 2016), and is strongly rooted in intrinsic values that include spiritual fulfillment and sacred natural sites. Chapter 11 briefly discusses some of these issues, while Winthrop (2014) reflects on tribal contexts in proposing "culturally reflexive stewardship" as a useful framework for understanding motivations for conservation based upon knowledge of local ecosystems, a world view that humans are a part of nature, and cultural practices that reflect residence and use over many generations.

Deal et al. (2017b) suggested that the Forest Service is well positioned to make ecosystem services the "central and unifying concept in federal land management." A 2015 presidential memorandum (OMB 2015) directed all federal agencies to develop and institutionalize policies to promote consideration of ecosystem services in planning, investments, and regulatory policy (table 12-2). However, it has been challenging for the Forest Service to describe and value all the potential ecosystem services that public lands provide. No published full accounting of ecosystem services has been conducted for the NWFP area, but some localized efforts have been made (Deal et al. 2017a, 2017b; Kline et al. 2016; Smith et al. 2011,) (see also chapter 9), and a framework as has been proposed (Deal et al. 2017b). This framework includes describing the ecosystem services provided by forest landscapes, examining the potential tradeoffs among services associated with proposed management activities, and attracting and building partnerships with stakeholders who benefit from particular services that the forest provides. According to Deal et al. (2017a), the common needs for advancing ecosystem services as a central framework for the Forest Service include:

- Building staff capacity for the concept and application of ecosystem services.
- Creating and publishing ecosystem service resource and reference materials.
- Aligning agency staffing, funding, and program structures with ecosystem service priorities.
- Integrating and managing data.
- Identifying inventory metrics; defining outcome-based performance indicators; and organizing and linking data.
- Valuing and mapping ecosystem services using current tools and methodologies.
- Communication.
- Policy including leadership support of using ecosystem services as part of a governance framework.

A review of several project-level applications of ecosystem services in Oregon found that place-based applications can highlight the connections between ecosystem conditions and public benefits (Deal et al. 2017b). The review hypothesized that using this approach could help transform the agency into a more effective and relevant organization and will strengthen public investment in Forest Service activities. Key ecosystem services provided by federal forests in the Plan area include water, recreation, wildlife and plant habitat, wood products, and carbon sequestration. The contribution of Forest Service lands to water yield in streams differs regionally and is especially significant in streams that originate in the western Cascade Range and northern California (fig. 12-3). The water supply from many watersheds in the Plan area originates on national forests (Watts et al. 2016), and water from undisturbed old-growth forests can be especially high in quality as a result of high nutrient retention and low erosion (Franklin and Spies 1991). Streamflow in summer, which is typically quite low, is nevertheless higher from old-growth forest watersheds in the western Oregon Cascades than in watersheds dominated by maturing forest plantations (Perry and Jones 2016). Forested streamside buffers have been shown to protect water quality in many parts of the world (Sweeney and Newbold 2014).

The carbon sequestration potential of old-growth forest ecosystems in the NWFP area has received special attention (DellaSala et al. 2015, Hudiburg et al. 2009, Kline et al. 2016, Smith et al. 2013, Wilson et al. 2013). When the forests and soils of this region develop for long periods (hundreds of years) without natural or human disturbances, they can store some of the highest levels of carbon of any region in the United States and the world (fig. 12-4).

The expanded understanding of ecosystem services also reveals that synergies and tradeoffs can occur between and among biocentric and anthropocentric values (Hunter et al. 2014, Kline et al. 2016). For example, certain conservation approaches (e.g., protecting old growth and restoring watersheds) may have the added benefits of increasing carbon sequestration and water quality and providing economic benefits in the form of scenic quality/aesthetics, recreation, or restoration jobs (Brandt et al.

Table 12-2—U.S. natural resource legislation with examples of federal agency responses and applications of ecosystem services for agencies

Legislation	Intent of legislation	Examples of U.S. federal agency responses
Multiple Use–Sustained Yield Act (1960)	Promote sustainable management of natural resources to meet the growing needs of an increasing population and expanding economy	U.S. Forest Service (USFS) and Bureau of Land Management (BLM) directed to manage timber, range, water, recreation and wildlife with equal importance
National Environmental Policy Act (1969)	Encourage harmony between people and the environment, enrich the understanding of the ecological systems and natural resources important to the Nation, and establish a Council on Environmental Quality	Any federal, state, or local project that involves federal funding, work performed by the federal government, or permits issued by a federal agency must take a multidisciplinary approach to decisionmaking, including consideration of alternatives
Federal Land Policy and Management Act (1976) and National Forest Management Act	Establish policy of inventory and planning in accordance with the Multiple-Use Sustainable Yield Act	USFS and BLM develop land management plans in collaboration with the public to determine appropriate multiple uses, develop strategies for resource management and protection, and establish systems for inventory and monitoring to evaluate the status of resources and management effectiveness
National Forest System Land Management Planning Rule 2012	Regulation developed by the USFS to implement planning required by the National Forest Management Act	Rule explicitly requires USFS managers to address ecosystem services in planning to ensure that forests have the capacity to provide people and communities with a range of social, economic, and ecological benefits for the present and into the future. Staff across the agency develop and apply tools to address ecosystem services in land-management efforts.
Presidential Memorandum: Incorporating Ecosystem Services into Federal Decision-Making (OMB2015)	Directs federal agencies to incorporate natural infrastructure and ecosystem services into decision frameworks	National Oceanic and Atmospheric Administration uses ecosystem service valuation to assess benefits of dam removal and coastal rehabilitation, among other projects
		Natural Resources Conservation Service applies ecosystem service quantification tools to its programs, including watershed rehabilitation and flood mitigation
		U.S. Fish and Wildlife Service incorporates consideration of ecosystem services into wildlife refuge management
		Environmental Protection Agency makes ecosystem services the focus of determining adversity to public welfare in review of air quality standards
		BLM and U.S. Geological Survey collaboratively assess alternative methods and quantification tools for evaluating ecosystem services through a case study in the San Pedro River watershed

Source: Deal et al. 2017b.

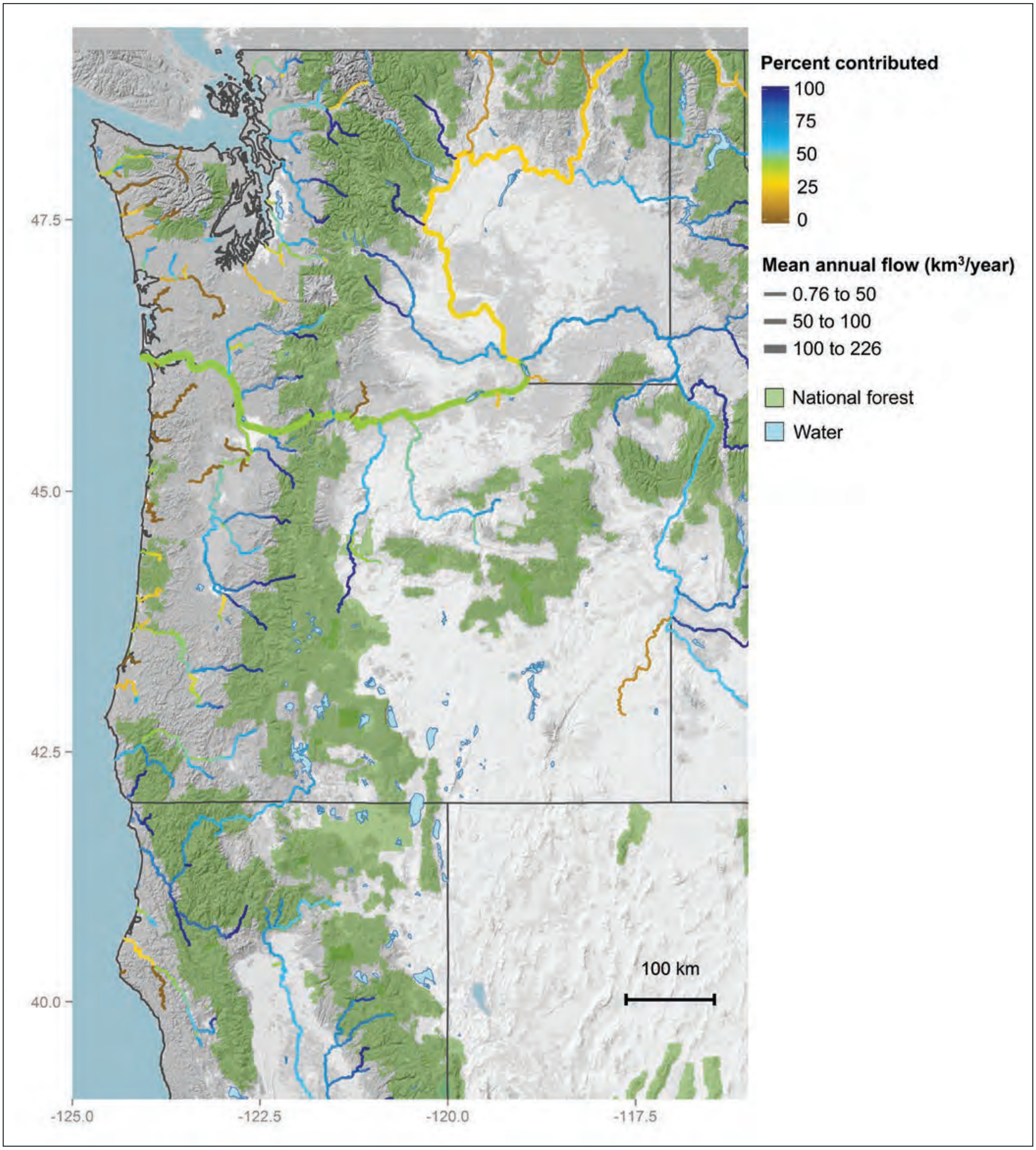

Figure 12-3—Percentage of annual streamflow from U.S. Forest Service lands in Washington, Oregon, and northern California. Data from Luce et al. 2017 (https://www.fs.usda.gov/rds/archive/Product/RDS-2017-0046/.) and https://www.fs.fed.us/rmrs/national-forest-contributions-streamflow-pacific-northwest-region-region-6.

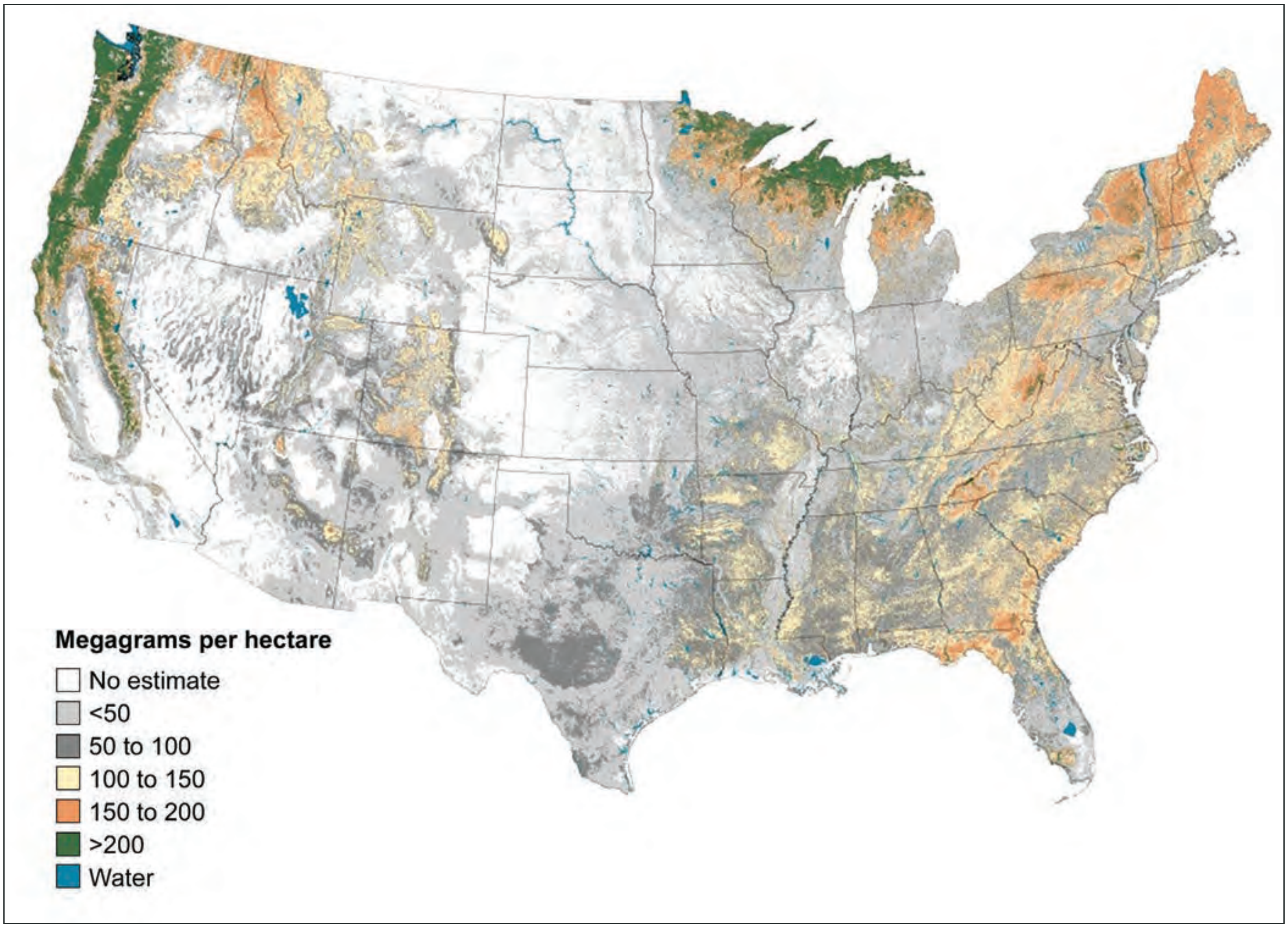

Figure 12-4—Total forest ecosystem carbon density in the United States, 2000–2009. Includes above- and belowground live trees, downed dead wood, forest floor, soil organic carbon, standing dead trees, and understory above- and belowground pools. From Wilson et al. 2013.

2014). In some cases, recreation and restoration benefits may help to offset job losses associated with declines in timber production. However, the economic systems and accounting for federal lands do not yet fully consider the values of carbon sequestration and water supply, and newer economies based on amenity values may not make up for job losses associated with protection of late-successional old-growth habitats and other economic factors in the timber industry (Charnley 2006a) (chapter 8). These variable effects and measures make it difficult to generalize about the ecosystem service impacts of the NWFP or conservation approaches in general. In addition, market forces external to NWFP communities and wood products manufacturing have also transformed since the NWFP was implemented, making it difficult to tease apart the role of federal lands management from other drivers of economic change in influencing community socioeconomic well-being.

Despite its limitations, many scientists consider the ecosystem services framework useful for managing the broad array of benefits that forests provide to people (Deal et al. 2017a, 2017b). Although there are challenges in operationalizing and measuring the entire set of ecosystem services outlined by the Millennium Assessment, the framework gives managers a more diverse set of possible objectives, including managing forests and rangelands for water, pollination potential, carbon, firewood/fuel, cultural heritage, spirituality, solitude, scenery, and many other values.

Institutional capacity—
A key interaction in the social-ecological system lies between the desire to restore forest dynamics and create more resilient forests and the limited capacity of human communities and federal agencies for active management. Although forest management on federal lands was often seen in the past (and still is by some) as a threat to native biodiversity, it is now seen by many ecologists and managers as critical to restoration and conservation of terrestrial ecosystems (Johnson and Swanson 2009). Interestingly, this view is not widely held for aquatic ecosystems (chapter 7). In the past, revenues from timber harvest often subsidized forest management, yet those revenues have declined with reductions in harvesting (chapter 8). Trends of declining agency budgets, increased fire suppression costs, and reduced agency staffing pose challenges to achieving forest management objectives such as ecological restoration, reducing wildfire risk to human communities, promoting habitat for wildlife (chapter 8), and providing diverse opportunities and settings for recreation (chapter 9). Federal agencies lacked the institutional capacity (staff with the required skills, financial resources, management flexibility, and incentives) to fully implement the NWFP's ecosystem management goals (Charnley 2006a). Efforts to maintain species and habitats and restore desired ecological conditions (e.g., old growth) and processes (e.g., succession fire and natural flows) require funding, forest management capacity (e.g., workforce and wood products infrastructure), and public support. The budgets for restoration and the annual rates of treatment are well below what is needed to restore fire to the historical levels found in frequent-fire landscapes (North et al. 2012, Reilly et al. 2017b, Spies et al. 2017). Limited budget and agency capacity has led to innovative approaches to accomplishing restoration, such as stewardship contracting and partnerships with nongovernmental organizations or other government agencies (chapter 8). However, wood processing mills needed to support forest restoration are closing in some regions (especially in less-productive dry forests), where timber supply from both private and public lands is insufficient to keep them in business (chapter 8).

The NWFP represented a dramatic shift in social priorities, from commodity production toward biodiversity conservation, which has been part of a larger national process that has been called "green drift" (Klyza and Sousa 2010) in environmental policymaking in the United States. However, the idea that "working forest landscapes" or "anchor forests" (multi-ownership landscapes that support sustainable timber and biomass production) can provide conservation values, funding for restoration, and support for rural communities has also gained much traction in recent years (Charnley et al. 2014, Corrao and Andringa 2017). Nevertheless, working forest landscapes are subject to the same concerns that have been raised about the balance between conservation and incorporation of human needs—how to reconcile different world views and values. This tension can only be resolved through social processes including public engagement and collaborative efforts that take into account social, ecological, and economic considerations and legislative actions (chapter 9).

Trust and collaboration—
Trust among federal land management agencies and the public is key to restoration and landscape-scale management for multiple goals, but trust is often lacking and difficult to cultivate (chapter 9). Trust among interested parties is essential for developing adaptive management strategies that can nimbly and effectively respond to changing climate, species, disturbances, human values, and markets. Trust can be lost in many ways on federal lands, especially when local-level agreements or collaborative processes are overridden by national-level political decisions (Daniels and Walker 1995), or when local decisions are seen as circumventing federal laws or policies. Researchers and practitioners have characterized public trust as integral to effective natural resources decisionmaking and implementation (Davenport et al. 2007, Pretty and Ward 2001, Shindler and Cramer 1999, Wondolleck and Yaffee 2000). Meanwhile, distrust can be a precursor for natural resource conflict (Nie 2003). Trust and distrust are not inversely related, but rather, trust is multidimensional and can coexist with distrust. Moreover, trust is contextual (depending on the setting or issue) and dynamic (changing based on each encounter or experience) (Lewicki et al. 1998). Trust in

natural resource institutions stems from creating trust in both processes and outcomes, whereas interpersonal trust depends on promoting trusting relationships between the public and agency personnel. For natural resource agencies, some factors shown to constrain the development of trust include unclear communication, limited public involvement opportunities, historical resentments, conflicting values, lack of progress in meeting objectives, lack of community awareness, and high turnover of personnel (Davenport et al. 2007). Trust among conflicting parties in resource management can be elusive, but it can be positively influenced through transparency, having clear processes, stated objectives, clarity of roles, and commitment to engagement (see chapter 9). A desire to build or expand trust is an important motivator for collaboration and conflict resolution (Wondolleck and Yaffee 2000), but "common ground will be elusive in conflicts involving fundamental value differences" (Wondolleck 2009). Frequent turnover among local forest management staff has been cited as a constraint on productive collaborations, particularly within tribal communities (see chapter 11).

Current efforts to enhance trust and generate social learning around restoration and other efforts to meet NWFP and other ecological goals are focused on collaboration among multiple agencies, and stakeholders around projects at various scales, from the watershed level to entire landscapes (chapter 9). Collaboration is touted as a means to achieve ecological goals as well as social benefits, which include conflict resolution, trust, and improved decisionmaking (Wondolleck and Yaffee 2000). Many of these collaborations are occurring in the fire-prone regions of the Western United States, and they are supported by funding related to forest restoration and fire-risk reduction programs. The Collaborative Forest Landscape Restoration Program is having some success in encouraging stakeholders to work together to help plan and implement forest restoration treatments, particularly in dry forests at the landscape scale (Butler et al. 2015, Urgenson et al. 2017).

Two well-established collaboratives fall within or immediately adjacent to the NWFP area: the Deschutes Forest Collaborative in central Oregon and Tapash Forest Sustainable Collaborative in eastern Washington. The Western Klamath Restoration Partnership is another example that builds upon years of collaboration in northern California. In addition to large-scale collaboration, there has been a proliferation of community-based collaborative groups in the Plan area that are engaged in National Environmental Policy Act planning, stewardship contracting, and multiparty monitoring, on both sides of the Cascades (Davis et al. 2015a) and in northern California. Other types of collaboratives in the NWFP area have formed around specific resource concerns, such as California Fire-Safe Councils (Everett and Fuller 2011) and the U.S. Fire Learning Networks (Butler and Goldstein 2010).

Collaborative processes are viewed by natural resource agencies as an effective way to engage stakeholders, provide an opportunity for dialogue and deliberation, and build trust and foster relations among groups that historically have worked in opposition (Butler et al. 2015, Urgenson et al. 2017). For example, the threat of high-severity wildfire in forests of the NWFP area that historically burned frequently may be a "common enemy" that can enable environmental and timber groups to work together with the Forest Service to advance restoration projects on the ground (Urgenson et al. 2017). This approach has emerged in some places such as the Western Klamath Restoration Project on the Klamath and Six Rivers National Forests in northwestern California, where a broad partnership of interests, including tribal communities (chapter 11) are coalescing around landscape-level restoration efforts rooted in returning fire to the system. Efforts like this will potentially be a model in some forest types for making meaningful progress on large-scale forest restoration. Collaboration appears promising, and studies to date have identified positive outcomes associated with social interactional concepts such as trust, social capital, learning, and process (Davis et al. 2017). There has been less emphasis on evaluating outcomes such as improved social and ecological conditions. The tremendous investment in collaborative processes may yield enhanced trust and improved ecological and social conditions. Although the landscape collaborative program in the United States has provided better community engagement in decisionmaking, the long-term benefits of the program have not yet been documented (Butler et al. 2015).

Forest collaboratives have been designed to distinguish the roles of agency staff as decisionmakers who consider input from stakeholder collaborators, rather than devolving decisionmaking to local communities or coopting the process to meet predetermined objectives (Butler 2013) (fig. 12-5). In other words, collaboratives are not engaged in true power sharing, because ultimately the federal agency's line officer makes the final decision. Agency participation in collaborative efforts often takes place at an "arm's length" with agency participants playing the role of "technical advisor" and often not holding roles as voting members of collaborative groups (Butler 2013). In fact, agency (Forest Service) participants in collaborative groups are more often motivated by the need to build social trust, whereas non-agency participants are motivated by the desire to achieve social and ecological outcomes (Davis et al. 2017). Greater decentralization of authority has arisen through co-management or community-based natural resource management efforts, particularly outside of the United States; however, there have been relatively few examples of such efforts in which both resource utilization and biodiversity conservation goals have been achieved (Kellert et al. 2000). Strong legal foundations, institutions, and investments in monitoring may have contributed to these successes, as demonstrated in some examples of tribes and state governments conserving salmon in the Pacific Northwest (Kellert et al. 2000) (chapter 11).

Team Hymas

Figure 12-5—The Forest Service has built upon precedents such as the Handshake Agreement of 1932 by establishing areas that are specially managed to support resources important to tribes within ancestral lands that are now national forests. Many of these approaches embody principles of cooperative management that go beyond collaboration, yet maintain the agency's decisionmaking authority. An area in the Sawtooth Berry Fields was reserved in 1932 by a handshake agreement between Yakama Indian Chief William Yallup and Gifford Pinchot National Forest Supervisor J.R. Bruckart for use by Indians.

Tribal perspectives—
Chapter 11, which addresses American Indian tribal values, vividly describes the integrated social and ecological values of ecosystems in the NWFP area. Tribes value a vast diversity of animals and plants for utilitarian values that include the use of timber, as well as intangible cultural values. The perspectives held by native peoples of the Pacific Northwest, informed by thousands of years of place-based experience, help to internalize many of the tradeoffs between use and preservation, as well as provide a long-term, broad spatial perspective about system dynamics. For example, many tribes want to sustain the legacy of old trees and associated biological diversity while also promoting the productivity and diversity of early-successional communities, nonforest communities, and hardwood communities, and also generating timber and nontimber forest products (fig. 12-6). To achieve such multifaceted goals, some tribes have developed innovative forest management plans that many consider to be fulfilling the promise of the NWFP for addressing both social and ecological goals (e.g., Baker 2003, Hatcher et al. 2017, Johnson et al. 2008). Chapter 11 highlights the critical role of fire in dry and some moist forest types for maintaining desired ecosystem conditions.

Thomas Dunklin

Figure 12-6—Clarence Hostler gathering matsutake mushrooms under tanoak trees on the Six Rivers National Forest, near Orleans, California, November 2013.

What Have We Learned About the Components of the Northwest Forest Plan and Their Compatibilities?

Coarse- and fine-filter approaches to conservation— Both coarse- and fine-filter strategies for conserving biodiversity (Hunter 2005, Noss 1987) are a part of the NWFP and the 2012 planning rule, and the relative importance of the two appears to have shifted toward coarse-filter approaches under the current planning rule. Earlier scientific debate on the pros and cons of single species (e.g., fine-filter) vs. ecosystem (coarse-filter) approaches to management (Casazza et al. 2016, Simberloff 1998, White et al. 2013) have been replaced by recognition that these approaches are complementary, and both are a valuable part of conservation strategies (chapter 6) (DellaSala et al. 2015, Hunter 2005, Noon et al. 2009, Reilly and Spies 2015, Simberloff 1998, Tingley et al. 2014). Meso-filter approaches (e.g., habitat elements like snags and large old trees) also have been included in a conservation approach hierarchy (Hunter 2005). The challenge now, and the source of some debate, is to find an appropriate level or balance of coarse-, meso-, and fine-filter approaches (Schultz et al. 2013). If a plan is weighted too much toward single species, or a particular successional stage, the strategy may succeed "in protecting a few of the actors at the expense of the majority of the cast" (Tingley et al. 2014). If weighted too much to the overarching ecosystem goals, the "stage" may be conserved but the "star actors may not show up" (Tingley et al. 2014).

Although the NWFP was based on coarse- and fine-filter strategies, the "star actor," i.e., providing enough suitable habitat to sustain northern spotted owl populations, had a very large influence on the Plan. The approach of using the northern spotted owl as a surrogate or umbrella for old-forest ecosystems developed "unintentionally," driven mainly by the need to meet the mandates of the ESA and other federal policies (Meslow 1993). The emphasis on the northern spotted owl carried through the development of the Plan, despite the fact that the NWFP was intended to be an "ecosystem management" plan. The single-species focus had unintended consequences for other biodiversity conservation and for management of resilience to fire and climate change across an ecologically diverse region. For example, in dry forests within the range of the northern spotted owl, large portions of the forest conditions that support this species are the result of 100 or more years of fire exclusion that has altered forest ecosystems and their resilience to drought and fire (chapter 3). The emphasis on the fine-filter aspect of the Plan—focusing on the northern spotted owl—challenges the Plan's ability to meet other ecosystem goals under the 2012 planning rule, including ecosystem integrity and resilience to climate change and other stressors.

The congruence of coarse- and fine-filter goals and management approaches varies by disturbance regime (chapter 3). The most congruence between managing for historical range of variation or ecological resilience (i.e., a coarse-filter approach based on ecosystem dynamics) and for species that use dense older forests is in moist forests, where fire was infrequent (frequencies of 200 to >1,000 years), and forests would often grow for centuries without major disturbance. However, in regimes where fire was frequent or very frequent (less than 50 years) and landscapes were dominated by open-canopy forests, it is challenging to manage for both a coarse-filter approach based on landscape-scale ecological integrity, and the fine-filter approach of the NWFP based on maintaining or increasing the area of dense older forests. That is not to say that the two goals cannot be integrated in dry forests, only that the current NWFP strategy in dry forests does not guide management toward ecological integrity, which would emphasize management for the ecosystem-regulating role of fire.

Congruence between the two approaches (ecological integrity and coarse filter based on prioritizing dense, multilayered forests) is intermediate in moderately frequent to somewhat infrequent fire regimes (50 to 200 years) of the drier part of the moist forests where fire exclusion has had somewhat less effect. Here, historical fire regimes created a highly dynamic mosaic of high-, moderate-, and low-severity fire and higher diversity of early, mid- and late-successional stages than in the infrequent fire regime areas (fig. 12-2) (chapter 3). The relative abundances and spatial patterns of different forest states in the fire regimes of the NWFP area create inherently different biodiversity and ecosystem process conditions in the NWFP region. This ecological and geographic variability means that weighting the plan too much in favor of a single successional stage (e.g., dense older forest) will not likely succeed in maintaining a broader set of goals related to ecological integrity or resilience to climate change and drought.

Northern spotted owl—
The northern spotted owl was listed as threatened under the ESA in 1990. Despite extensive efforts of federal agencies to protect northern spotted owls, conserve remaining habitat, and set aside areas as future habitat, populations have continued to decline (chapter 4). When the NWFP was implemented, northern spotted owl populations were predicted to continue declining for as long as 50 years owing to lingering impacts of previous habitat loss before populations would recover while sufficient area of younger forests grew into conditions that supported the owl (chapter 4). Unknown at the time were the effects that competitive pressure by barred owls would have on spotted owl populations, which have further compounded the challenges faced by northern spotted owls and accelerated their rate of population decline. Without the protections afforded by the NWFP and ESA, northern spotted owl populations would likely have experienced even steeper declines (chapter 4). Clearly, efforts to recover the subspecies are facing multiple challenges related to both habitat management and the barred owl invasion (USFWS 2011). With the continued population expansion of the barred owl within the range of spotted owls, the long-term prospects for spotted owls are not good and remain uncertain.

Although structural definitions of old-growth forests and northern spotted owl habitat are similar in many ways, they are not synonymous (Davis et al. 2016), and strategies to conserve them may differ (fig. 12-7). Additionally, northern spotted owls do not function as an umbrella for **all** or even most other species within the full range of vegetation conditions in the NWFP area (Burnett and Roberts 2015, Carroll et al. 2010), a fact that was recognized at the time of the development of the NWFP and which led to the development of the Aquatic Conservation Strategy (ACS) and additional species protections in the form of the Survey and Manage program (chapter 6) (Carroll 2010, Molina et al. 2006, Raphael and Marcot 1994, Thomas et al. 2006).

Marbled murrelet—
The marbled murrelet has habitat needs that overlap those of the northern spotted owl and that are compatible with many definitions of old-growth forests (fig. 12-7). Thus, plans and strategies that focus on northern spotted owls and old-growth forests are likely to benefit to a large degree the marbled murrelet within its range. However, there are some distinctive habitat differences between marbled murrelets and northern spotted owls that require special conservation considerations (chapter 5). The most obvious difference is that the murrelet is a diving seabird whose foraging habitat is in the coastal marine environment, thus marine conditions must be considered in murrelet habitat needs. Murrelet nesting habitat occurs in coastal forests that typically experienced infrequent, high-severity fire regimes. Within that environment, marbled murrelets preferentially select larger, more contiguous patches of forest throughout their range and tend to avoid edge habitats where risk of nest depredation is greater (Raphael et al. 2015) (chapter 5); therefore, unlike for the northern spotted owl, proximity of early-seral forest is undesirable because it can increase abundance of birds that prey on murrelet nests. Extensive efforts to restore fire-resilient open old-growth forests in the somewhat infrequent to moderately frequent, mixed-severity regimes in the range of the murrelet may reduce habitat quality by increasing the exposure of nests to predators.

Aquatic ecosystems—
Goals of aquatic ecosystems partly overlap with characteristics of old-growth forests, and with habitats for northern spotted owls and marbled murrelets (fig. 12-7). For example, large dead trees and shading from dense patches of streamside conifer forests contribute to habitat quality in stream channels and cool stream temperatures that support salmonid populations (chapter 7). In coastal areas, tall, multilayered conifer canopies can intercept fog and deliver more moisture to streams than can shorter dense forests, mitigating some of the effects of climate change (chapter 7). However, the absence of disturbance for extended periods can result in the decrease in suitable substrates, reducing habitat quality (Reeves et al. 1995) (chapter 7). Riparian and stream environments are also dependent on geomorphic and

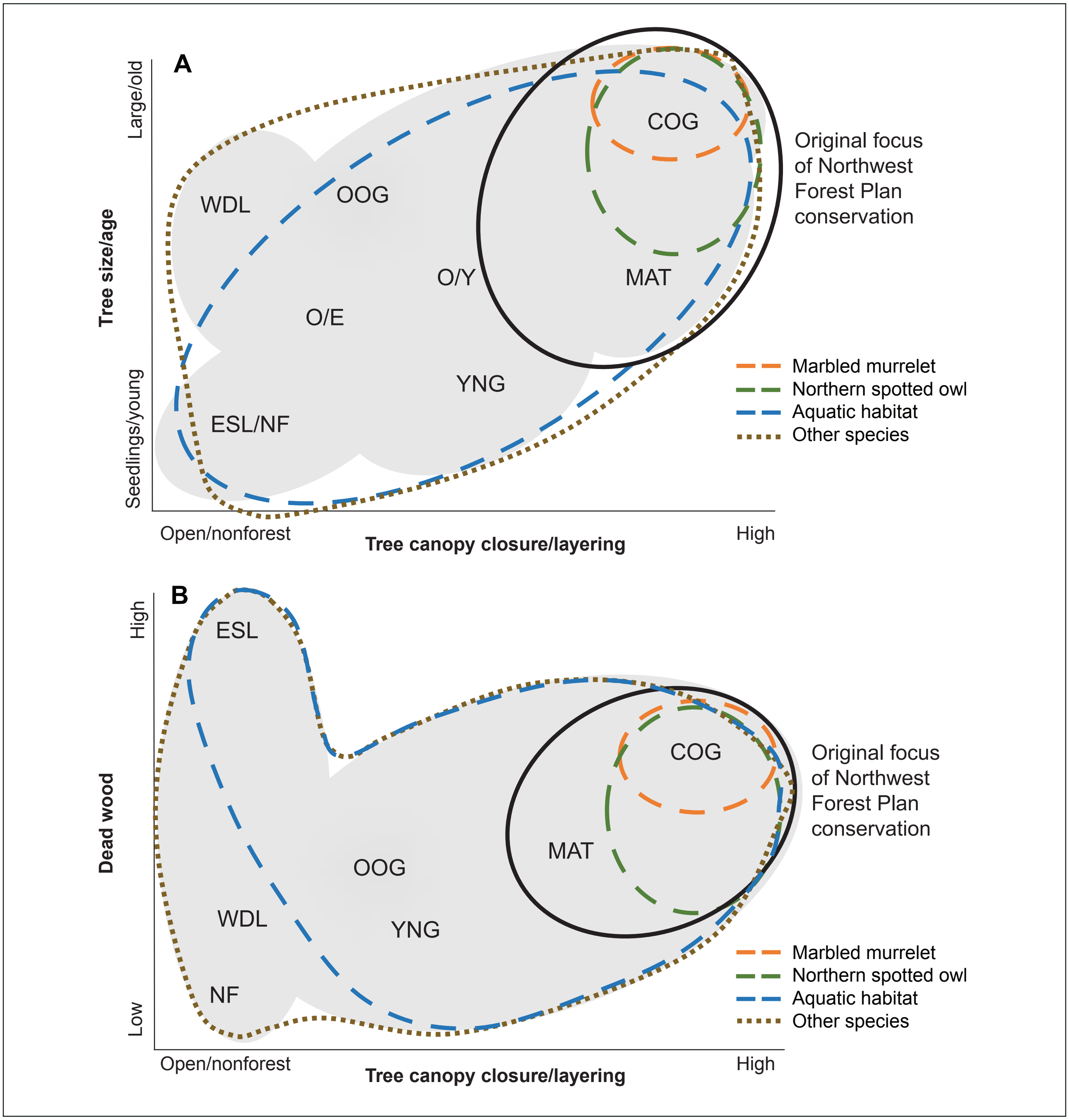

Figure 12-7—Distribution of habitat (dotted line ellipses) in relation to (A) tree canopy closure and tree size and (B) tree canopy closure and dead wood for different biodiversity components in the area of the Northwest Forest Plan. Northern spotted owl habitat refers to forests that are suitable for nesting and roosting. Gray ellipses refer to selected vegetation structure classes: COG—closed-canopy old growth; OOG—open-canopy old growth; YNG—young forest; MAT—mature forest; O/E—early successional with old live trees; O/Y—young forest with old trees; WDL—woodland; ESL/NF—early-seral/nonforest (shrubland, grassland). Conserving and restoring aquatic ecosystems requires a range of vegetation states, including older forest through time, but is not restricted to old growth (chapter 7). Many terrestrial species, including some tribal ecocultural resources, require early-successional and nonforest vegetation. Similarly, salmonid community assemblages differ between recently disturbed streams and undisturbed streams in old-growth forests.

hydrological disturbances that make many riparian areas a mosaic of older conifers, younger conifers, hardwoods, and shrubfields. This mosaic and the disturbance and successional dynamics that drive it means that the range of variation in riparian vegetation habitats may include conditions that do not qualify as old-growth forests (e.g., a lack of old conifer trees) or meet the habitat needs for northern spotted owls and marbled murrelets (fig. 12-8).

Fires burning through riparian areas and surrounding uplands may have reduced some stream qualities in the short term, but these events often improve conditions as large dead trees fall into streams, and as postfire floods, landslides, and debris torrents reorganize streams into more complex habitats (chapter 7) (Bisson et al. 2003). The absence of fire results in the lack of large influxes of sediments and wood, the basic building blocks of habitat for native fish, to the valley floors (Bisson et al. 2003, Flitcroft et al. 2016, Reeves et al. 1995) (chapter 7). Active management will continue to be used to reduce fuels and vegetation that make the forests susceptible to uncharacteristically large and severe wildfires. Such management often strives to prevent disturbances to streams, which can reduce or eliminate the occurrence of periodic disturbances that deliver sediment to the valley bottoms and stream channels. The lack of these disturbances and sediment can have serious unintended consequences to riparian-dependent wildlife and aquatic organisms (chapter 7).

Disturbances such as floods, landslides, and debris flows, which are essential for aquatic ecosystem functions, can be affected by roads that alter disturbance flow pathways and disconnect streams from uplands (Jones et al. 2000). These changes can reduce the resilience of these

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Figure 12-8—Mosaic of vegetation and substrate conditions along the North Fork of the Elk River, which occurs in an unlogged and largely unroaded watershed on the Rogue-Siskiyou National Forest in coastal Oregon.

ecosystems to these natural disturbance events. Decommissioning of roads can also improve passage for fish and other species and help reconnect streams and floodplains and improve water quality. Not all roads are the same, however, in terms of their ecological effects, and knowledge of how road networks are distributed relative to geomorphic processes can aid in the design of more effective road systems and restoration of watershed processes.

The potential of federal lands to contribute to the recovery of listed fish, particularly Pacific salmon, in many parts of the NWFP area is likely more limited than was recognized when the ACS was developed (chapter 7). The primary reason for this difference is that, in many situations, federal lands have a limited capacity to provide high-quality habitat for some of the listed fish. Federally managed lands are generally located in the middle to upper portions of watersheds, which tend to have steeper gradients and more confined valleys and floodplains, making them inherently less productive for some fish (Burnett et al. 2007, Lunetta et al. 1997, Reeves et al. 2016). Federal lands may, however, be major sources of wood, sediment (Reeves et al. 2016), and water (Brown and Froemke 2010, 2012) for downstream nonfederal lands, and will be important for the potential recovery of most populations. Nevertheless, their contribution to recovery may in many cases be insufficient without parallel contributions from nonfederal land ownerships elsewhere in the basin (Grantham et al. 2017).

Other species of late-successional and old-growth forest— The Survey and Manage program (chapter 6) identified and listed many fungi, lichens, bryophytes, invertebrates, and other species groups that were deemed to require specific surveying to help ensure their conservation under the NWFP. Although the NWFP protects 80 percent of the remaining old-growth forest in the region, this amount of old growth may represent only about 15 percent of the historical amounts of old growth that occurred in the moist forests across all lands in the NWFP area (chapter 3). The Survey and Manage program helped reduce the number of species on the list that were originally ranked as having low potential for persistence. The program also helped evaluate other species for potential addition to the lists and to make adjustments to surveys and site protection as needed for conservation of those species. Reduction in survey status or removal from the Survey and Manage species lists resulted from efforts to locate species during "predisturbance surveys" before harvests or other management activities. Since the 2006 synthesis (Haynes et al. 2006), no species have been added to the Survey and Manage species list; any additions would occur through a renewed annual species review process, and none was added the last three times the review process took place in 2001, 2002, and 2003.

The approach of the Survey and Manage program represented a fine-filter strategy applied to hundreds of species, which created a nearly impossible administrative and financial challenge to land management agencies (Molina 2006). This approach may not be consistent with the goal of having "a few species of special concern" under the new planning rule, although the rule also calls for creating lists of "species of conservation concern." At present, we recognize that alternative strategies to applying a fine-filter approach to large numbers of species include a meso-filter approach that is based on functional groups and habitat elements (chapter 6). As levels of intensive timber management from late-successional and old-growth forests continue to be low, as has been the case in recent years (fig. 12-9), and all such forests are excluded from timber management, the original motivation for the program—logging of unreserved older forest in the matrix (Molina et al. 2006)—would seem to have weakened. Most of the logging that has occurred under the NWFP appears to have been associated with restoration in plantations in moist forests and fuel reduction activities in dry fire-excluded late-successional and old-growth forests. The situation in dry forests raises the question of how to reconcile the goals of dense-forest species with those of ecological integrity and species that use more open fire-dependent forests? Fire exclusion has dramatically altered the habitats of both types of native species in these regimes (chapter 3) (Dodson et al. 2008; Keane et al. 2002, 2009); however, effects on biodiversity have received little empirical study in the NWFP area (Lehmkuhl et al. 2007), and broader evaluations of other dimensions of biodiversity (e.g., population genetics, food webs, and ecological functions) have generally not been made.

Forest carnivores, particularly those associated with old forest conditions, were not a primary focus of the original

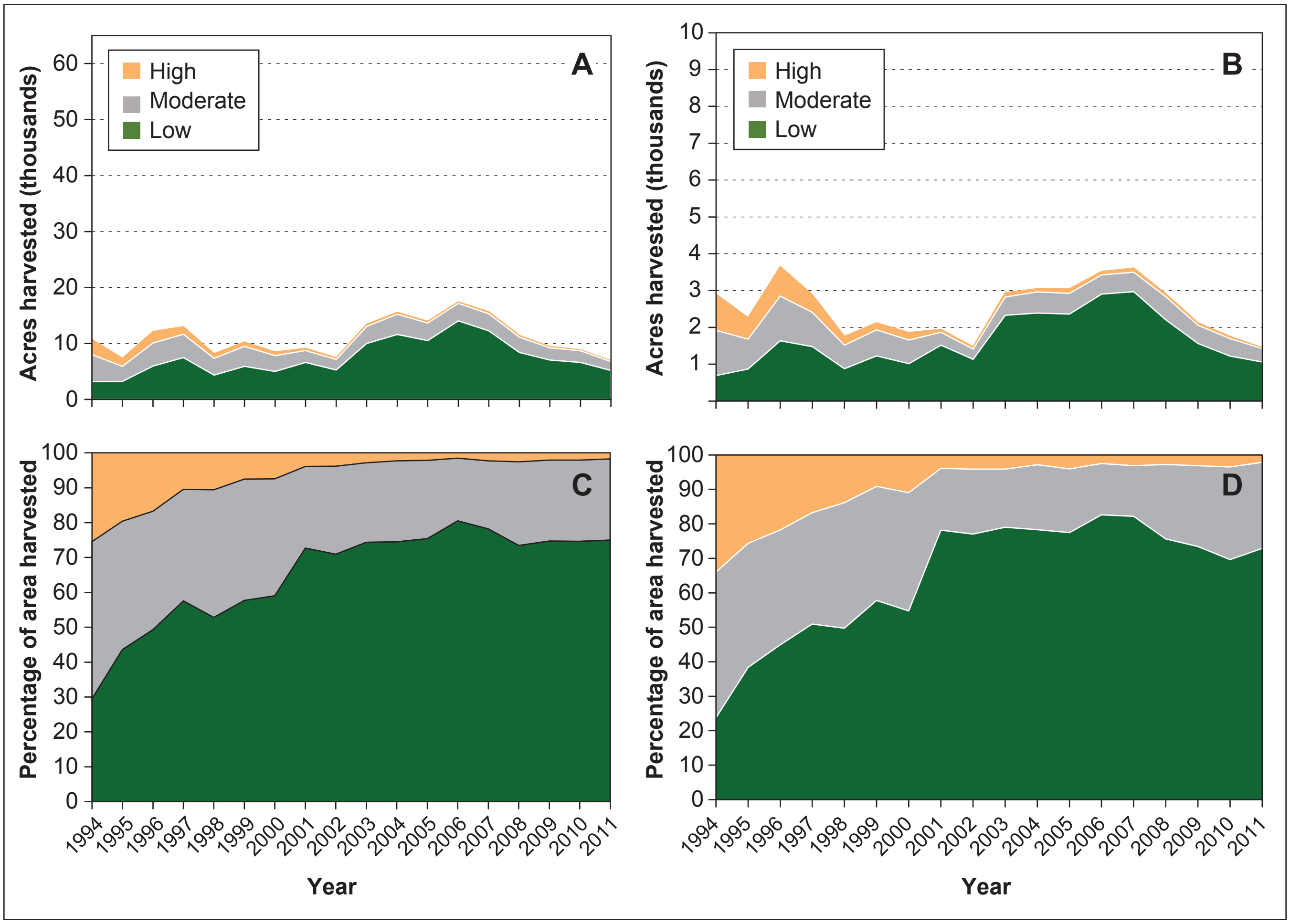

Figure 12-9—Trends in area of (A) old-growth structure index (OGSI) 80 harvested and (B) OGSI 200 harvested by intensity class (low, medium, and high) and percentage of harvest of (C) OGSI 80 and (D) OGSI 200 by intensity class on all federal lands between 1994 and 2011. OGSI is an index of stand structure based on live and dead tree characteristics that can be used to map the degree of old-growth development across a landscape as an alternative to classifications that simply define forests as old-growth or not. OGSI 80 and OGSI 200 represent the index at 80 and 200 years, respectively. Low = 0 to 33 percent loss of vegetation cover (all life forms); moderate = 33 to 66 percent loss, high = >66 percent loss. Note difference in scale between acres harvested in OGSI 80 and OGSI 200. Based on analysis of annual thematic mapper satellite imagery. Data are from Davis et al. 2015b. See Davis et al. 2015b for more information about OSGI.

NWFP. Fishers, marten, and lynx (*Lynx canadensis)* were addressed in the Forest Ecosystem Management Assessment Team report (FEMAT 1993) to a limited degree, with suggestions for conservation actions including closure to trapping of marten on federal land, evaluation of the effects of poisoning porcupines (*Erethizon dorsatum*), completion and implementation of habitat capability models for fishers and martens in California, and conducting more thorough surveys for both marten and fisher. Concern for the status of these species and for the wolverine (*Gulo gulo*) (which uses higher elevation, alpine, and subalpine habitats) has increased significantly in the past 23 years, and recent findings have identified new populations, new threats, and even new taxonomic species (see chapter 6). The Forest Service has increased measures to conserve habitat for these species, particularly in northwest California, where an extant population of fisher remains at risk. Increases in populations of carnivores would potentially have benefits to these ecosystems that cascade through trophic levels (Beschta and Ripple 2009), but the broader ecological effects of the further reduction or loss of these carnivores or their return in the NWFP area are not well understood.

Old-growth forest ecosystems—

The goal of the NWFP was to create “a functional, interactive, late-successional and old-growth ecosystem” (USDA and USDI 1994b: 6). As mentioned above, the congruence of the old-growth forest with the other conservation goals varies by location within the NWFP region, and by the definitions of old growth, and objectives. In general, the NWFP goals, which are yet to be fully achieved (e.g., in terms of area) (chapter 3) (Davis et al. 2015b), will provide a foundation for reaching many of the biodiversity goals of moist forests. But these goals are not consistent with managing for native biodiversity of dry forests and will not lead to long-term resilience of those ecosystems to wildfires and drought, or the broad diversity of successional and fuel patterns that support the natural fire regime (chapter 3) (fig. 12-7). Moreover, meeting NWFP goals has consequences for other components of forest biodiversity (e.g., early-seral species) not considered in the original NWFP (chapter 3) (Hessburg et al. 2016), especially those dependent on fire of different frequencies and severities, including aquatic ecosystems (chapter 7). In addition, new studies and increased recognition of the historical role of moderately frequent fire in drier parts of the moist forest zone, suggest that the Plan goal of conserving biodiversity associated with older forests may need to be revisited even in these relatively moist forests (chapter 3). Management for ecological integrity in this fire regime likely would seek to have a range of old-forest structural types (e.g., with and without tree age cohorts created by partial stand-replacement fires) and other successional conditions across landscapes. Fire in the moist forest zone sustains old forests and other successional stages, and contributes to hillslope processes (e.g., landslides and debris flows) that are fundamental to creating diverse and essential fish habitats (see below).

Reserves—

Late-successional forest and riparian reserves were major and controversial components of the NWFP. Based on the monitoring results and the original goals of the NWFP, the reserve strategy can be considered a success from the standpoint of halting old-growth logging (Davis et al. 2015b, 2016; Raphael et al. 2015). In addition, although late-successional and old-growth forests have continued to decline across the NWFP area owing to wildfire and logging (in the first few years of the NWFP), trends are in line with the Plan’s expectation of losses (Davis et al. 2015b, 2016), but new concerns have emerged about fire and climate change. Similarly, clearcutting of riparian forests on federal lands has also come to a halt, contributing to improvements in watershed health (chapter 7).

Although trends in the amount of dense old growth are in line with expectations at a regional scale, there are reasons for concern (chapter 3). First, as mentioned above, maintaining or increasing current amounts of dense older forests in the dry forest zone is not consistent with managing for ecological integrity, as defined under the 2012 planning rule. Second, the Plan did not consider climate change effects that are already significant in dry forests (chapters 2 and 3). Managing for large areas of dense older forest (e.g., current LSR design) will not promote resilience to fire and drought, both of which are increasing under climate change. We explore these concerns in more depth below.

The standards and guidelines for the reserves specifically called out a need for active management to restore ecological diversity to plantations in both moist and dry forest types. Restoration activity has occurred in plantations in LSRs in moist forests, where innovative approaches to thinning have been developed and widely applied (chapter 3). The standards and guidelines for dry, fire-frequent forests (east of the Cascades and in the Oregon and California Klamath provinces) were different (USDA and USDI 1994b). There, the focus of management was on accelerating older forest development in younger forests and reducing risk of loss to high-severity fire in older forests. This concern was the impetus for designating some LSRs under the NWFP as “managed LSRs,” in which silvicultural treatments were permitted to reduce risk of loss of stands around some northern spotted owl activity centers. However, the area of this type of LSR was small (about 102,000 ac) compared to the millions of acres of LSRs in dry forests (USDA and USDI 1994a). It is not clear how much restoration activity has actually occurred in older forests in LSR’s or in riparian reserves in fire-prone forests because the implementation monitoring program was not continued. However, indications are that between

1993 and 2012 (20 years) less than 2 percent of older forest (OGSI 80) in the dry forest zone had treatments (Davis et al. 2015b) that would reduce total canopy cover, and surface and ladder fuels and risk of loss of older forest to large high-severity fires.

The issue of the need for restoration management also applies to riparian reserves, where relatively few restoration treatments have occurred (chapter 7). Primary reasons for the limited amount of restoration activity include (1) differing perspectives about the characterization of reference conditions, conservation, and management; (2) concerns about the potential effects of mechanical treatments on stream temperature and wood recruitment; (3) concerns about rare and little-known organisms (Reeves 2006); and (4) lack of trust in managers to undertake actions primarily for ecological benefits (chapter 7).

The LSR strategy of the NWFP was not designed or implemented in a way that promotes or restores ecological integrity or resilience in frequent or moderately frequent fire regimes (Spies et al. 2006, 2012). The initial identification of LSRs used a triage-based methodology that identified remaining concentrations of dense older forests after a history of fire suppression and aggressive harvesting. These areas were intended to provide habitat for northern spotted owls with adequate size and spacing of late-successional and old-growth forests to support the owl's recolonization. But this delineation was done without consideration for topographic and environmental setting and historical fire regimes of the forests. The standards and guidelines for silviculture in fire-prone forests (USDA and USDI 1994b) place many restrictions on restoration in dry forests in LSRs, and emphasize stand-level treatments to accelerate development of late-successional (i.e., dense multilayered) forests in younger forests that do not "degenerate suitable [northern spotted] owl habitat." They also suggest that treatment in older forests "may be considered" where they "will clearly result" in reduced risks. The standards and guidelines also lack a landscape perspective for fire and dry forest dynamics (e.g., see Hessburg et al. 2015, 2016; Stine et al. 2014) that is now understood to be critical to achieving a mix of ecological goals in fire-prone landscapes. The main reason for the low level of restoration in older forests in LSRs mentioned above may be lack of social license including the threat of litigation (Charnley et al. 2015), which occurs much more frequently in the Forest Service's Pacific Northwest Region (Oregon and Washington) than any other region in the country (Miner et al. 2014). Other reasons may include valuing multistoried forests, the burden of protocols under the Survey and Manage Program, lack of trust in managers (Olsen et al. 2012), the perception of some that mixed-conifer forests do not need restoration (Urgenson et al. 2017), or that reserves mean no-touch areas. Nevertheless, a review of the literature conducted for the 10-year socioeconomic monitoring report, combined with interviews held with forest managers and community members in four case-study locations across the NWFP area, found that most people (84 percent) believe that active forest management is needed to maintain forest health, as long as it does not include harvesting old-growth or clearcutting (Charnley and Donoghue 2006). Most interviewees did not believe that enough active management had occurred during the first decade of the Plan, expressing concerns about fire, insects, and disease.

If the broader goal of managers is to build resilience to fire and climate change across fire-prone landscapes, our evaluation of recent science indicates that the current NWFP conservation strategy (e.g., LSRs, matrix, survey and manage species) in fire-prone forests would not increase ecological integrity or resilience of terrestrial and aquatic ecosystems in these landscapes (chapter 3). This is because the current approaches focus on maintaining current levels or even increasing the amount of dense older forest. Although some treatments are permitted in older forests to reduce risk of loss of northern spotted owl habitat to wildfire, insects and disease, the current strategy does not appear to have a goal of landscape-level resilience to fire and climate change as indicated under the 2012 planning rule. Landscape-level strategies that restore fire as an ecological process based on topography, vegetation heterogeneity, successional dynamics, fire behavior, and other factors would be more in line with the latest scientific thinking (Cissel et al. 1999, Hessburg et al. 2015). Such an approach would also be more in line with the most recent northern spotted owl recovery plan (USFWS 2011, 2012), which provides broad guidelines for navigating diverse ecological goals in these regions and states:

> …we recommend that dynamic, disturbance-prone forests of the eastern Cascades, California Cascades and Klamath Provinces should be actively managed in a way that reconciles the overlapping goals of spotted owl conservation, responding to climate change and restoring dry forest ecological structure, composition and processes, including wildfire and other disturbances… . Vegetation management of fire-prone forests can retain spotted owl habitat on the landscape by altering fire behavior and severity and, if carefully and strategically applied, it could be part of a larger disturbance management regime for landscapes that attempts to reintegrate the relationship between forest vegetation and disturbance regimes, while also anticipating likely shifts in future ecosystem processes due to climate… .

Modeling studies suggest that landscape approaches could reduce conflicts between restoration of fire-excluded ponderosa pine forests and conservation of the Mexican spotted owl (*Strix occidentalis lucida*) in Arizona (Prather et al. 2008); meanwhile, for the Sierra Nevada of California, Stephens et al. (2017) suggested that more comprehensive restoration treatments were needed to reduce wildfire risk to California spotted owls. Within the NWFP area, Spies et al. (2017) and Ager et al. (2017) modeled landscape scenarios in the eastern Cascade Range of Oregon and found that most of the existing area of spotted owl habitat could be maintained for 50 years despite the occurrence of wildfire (at recent rates) and restoration activities designed to create open, more resilient forests. Projected losses of owl habitat from wildfire were significantly more than from relatively limited restoration activities, but these losses were made up for by gains in habitat from growth and succession of small-diameter or relatively open forests. The value of examining both losses to fire and succession together has also been highlighted in a study by Reilly et al. (2017b), who found that in the eastern Cascades of Washington, Oregon, and California, losses of closed-canopy forests to high-severity fire between 1985 and 2010 were mostly balanced by gains from succession, though higher elevation forests showed significant declines and LSRs showed a small net decline in old, closed-canopy forests. These studies suggests that landscape-scale assessments of northern spotted owl habitat dynamics and fire need to take into account the age and structure distribution of all forests in a landscape and account for potential increases in northern owl habitat from succession. These trends may not hold in the future, however. Ager et al. (2017) found that if the rate of wildfire were to increase 2 to 3 times over current rates (e.g., moving from fire-return intervals of 250 years to 100 and 63 years, respectively), as some climate change studies suggest could happen (chapter 2), then the amount of northern spotted owl nesting and roosting habitat across the Deschutes National Forest could decrease by 25 to 40 percent in 30 years. Climate change projections also suggest decreased tree growth in the future (Restaino et al. 2016), which may affect the rate at which forest structure can regrow following fire.

The only explicit strategy that implements this vision for high-frequency fire forests is the Okanogan-Wenatchee National Forest Restoration Strategy (USDA FS 2012b). This strategy places a priority on restoring fire as an ecological process while maintaining adequate areas of spotted owl habitat that will shift across the landscape as fire and successional processes operate. Dynamic landscape approaches to reserves (as described above) or habitat conservation would have some similarities with recovery plans used for other listed bird species that find habitat in dynamic fire-prone landscapes (e.g., Kirkland's warbler and red-cockaded woodpecker). However, the habitats of these species are threatened by fire suppression rather than being promoted by it in the case of the northern spotted owl. The literature indicates that a dynamic landscape approach could still fit the broader definition of a "reserve" (e.g., exclusion of industrial level logging).

The current LSR-Matrix approach for dry zone forests does not appear to have or meet goals related to ecosystem integrity and management for resilience to climate change and fire. Managers may want to consider reevaluating and redesigning the NWFP conservation strategy for dry forests based on new scientific knowledge of climate change effects, knowledge of restoration strategies for dry forest landscapes (Hessburg et al. 2016), and the new 2012 planning rule, which emphasizes ecosystem approaches to conserving biodiversity. The science and experience with proposed changes to the NWFP conservation strategies indicate that design and implementation of such approaches would be facilitated by a transparent and inclusive decision-making processes (Olsen et al. 2012).

There may also be ecological benefits for alternative approaches for terrestrial and aquatic goals in dry parts of the moist zone forests (Cissel et al. 1999, Reeves et al. 1995). Management based on the historical disturbance regimes can benefit aquatic habitats (Reeves et al. 1995) in these fire regimes. For example, Cissel et al. (1999) found ecological benefits from changing the spatial distribution of reserves and standards and guidelines for LSRs and the matrix to better approximate the mixed-severity fire regime dynamics of the western Cascades of Oregon. Experiments were started in older stands to evaluate the management alternatives that included using timber harvest and prescribed fire as surrogates for partial stand-replacement fire. However, the effort was abandoned because stakeholders were skeptical of cutting older trees in the matrix lands, and they lacked trust in the agency to implement such approaches to achieve restoration goals (Olsen et al. 2012).

Thomas et al. (2006) suggested changing the NWFP allocations to protect all remaining older forest, whether located in reserves or the matrix. The U.S. Fish and Wildlife Service critical habitat designation recommends conserving spotted owl sites (recovery action 10) and protecting high-quality habitat (recovery action 32) whether it occurred in LSRs or the matrix (USFWS 2011, 2012). The science suggests that these actions will have ecological and social benefits, but there will be tradeoffs associated with timber production and needs of species that use other successional stages, although none of those species has been identified as threatened, endangered, or at risk because of conversion of their habitat to late-successional or old-growth forest conditions.

The NWFP was intended to adapt to new knowledge and changes in the environment (USDA and USDI 1994b), which is consistent with the idea that conservation should be adaptive and iterative (Carroll et. al. 2010, Walters 1986), but this goal has not been fully achieved for various reasons (see below). Although lines are drawn on maps, and standards and guidelines are developed for reserves and other land allocations, findings from conservation and ecosystem sciences suggest that these should not be seen as immutable. Ecological and social science research, adaptive management experiments at landscape scales, and monitoring are critical to learning and meeting the conservation goals of the NWFP. These tools are also critical to addressing other species and habitat concerns, along with other human values across the wide range of forest environments within the range of the northern spotted owl.

Socioeconomic goals—

The NWFP had four main socioeconomic goals (Charnley 2006b): (1) produce a predictable and sustainable level of timber and nontimber resources, (2) maintain the stability of local and regional economies on a predictable, long-term basis, (3) assist with long-term economic development and diversification in communities most affected by cutbacks in timber harvesting to minimize the adverse impacts associated with job loss (USDA and USDI 1994b), and (4) promote interagency collaboration and agency and citizen collaboration in forest management (Tuchmann et al. 1996). Regarding the first goal, 20 years of monitoring data indicate that the probable sale quantity of timber identified by the Plan was never met, meaning that timber sales have not been predictable or at the level envisioned (chapter 8). The probable sale quantity established by the Plan was based on a number of assumptions: (1) harvesting unreserved older forest in the matrix with novel silviculture would contribute roughly 90 percent of the volume during the first three to five decades of the Plan, (2) about half of the harvest during the first decade would come from forests more than 200 years old, and (3) the main harvest method would be regeneration harvest, using retention harvesting approaches (chapter 3) rather than clearcutting (Charnley 2006a). The area of regeneration harvest in OGSI 80 and OGSI 200 (fig. 12-9) was 1,000 to 2,000 ac annually in the first 5 years of the Plan, but it declined to near zero by 2000 and has stayed very low since then. Most of the harvest since 2000 has been in the form of thinning and partial canopy removal (figs. 12-9C and 12-9D), which generate less volume than intensive (regeneration) harvest. The early levels of regeneration harvest may have also included sales awarded before the Plan was implemented.

Appeals and litigation over timber sales that included large, older trees, and lack of public support for clearcutting and old-growth harvesting, were major factors preventing the agencies from cutting OGSI 80 and OGSI 200 to meet probable sale quantity (Charnley 2006a, Thomas et al. 2006). The need to protect more habitat for the northern spotted owl (given the threat from the barred owl), and the need to protect late-seral habitat for other species associated with older forest also limited harvest in mature and old-growth forests (chapter 6).

Thus, the main source of timber supply shifted from the intended ecological retention harvesting from older unreserved forests in the matrix in the first few years of the Plan to restoration thinning of smaller trees from plantations and forests less than 80 years old in LSRs and the matrix. Timber as a byproduct of thinning in plantations and restoration in dry older forests is compatible with several conservation goals as discussed above, and it is less controversial. However, such thinning in LSRs cannot be sustained, because in 10 to 20 years most of the plantations will have been thinned once, and most of them in the moist provinces will become too old (80 years) to be treated again according to the record of decision (USDA and USDI 1994b) (chapter 8). Likewise, the thinning and restoration of resilience in fire-prone older forests may not produce a sustainable supply of wood as restoration eventually shifts from mechanical removal of understory trees to using wildfire and prescribed fire to maintain resilience (Spies et al. 2007). The sale of wood products generated may not offset the costs of treatments.

One way that restoration might provide for more economically viable and longer term production of wood from federal lands is through the use of ecological forestry[8] approaches (Franklin and Johnson 2012) to create diverse early-successional habitats (chapter 3). Such habitats are created naturally by wildfires and other natural disturbance agents, but in most areas in the NWFP region these fires are suppressed to protect a variety of human and forest values (see chapter 3). Fire exclusion means that diverse early-seral conditions will develop from fire at lower rates than would have occurred historically. Restoration treatments (mechanical and prescribed fire) could be used to create diverse early-seral vegetation to help achieve biodiversity goals in contexts in which they do not conflict with goals for older forests. Such actions would typically remove some larger trees and could thereby provide timber for local economies, while helping to fund removal of small trees and biomass. Franklin and Johnson (2012) suggested that such actions be focused on existing plantations, outside of LSRs and in places where other late-successional goals are not compromised. This type of management could provide a niche for federal timber production that is something of a win-win for a diverse set of ecological and socioeconomic goals. In addition, the fact that federal timber cannot be exported could also provide a supply of timber for local mills that would not have to compete with export markets that are currently strong.

Ecological forestry principles could also be used in riparian forests to restore the diverse forest structure and composition that occurred under historical disturbance regimes. Since development of the ACS, there has been support in the scientific literature for discretion in setting site-specific activities (Kuglerová et al. 2014, Lee et al. 2004, Richardson et al. 2012), which can be economically beneficial (Tiwari et al. 2016). Greater flexibility in the management of riparian areas would depend on the “context” of the area of interest (Kondolf et al. 2006, Montgomery 2004) and the primary management objective for the specific area (Burnett and Miller 2007). However, development of such an approach has been limited because of the reliance on “off-the-shelf” and one-size-fits-all concepts and designs, rather than on an understanding of specific features and capabilities of the location of interest (Kondolf et al. 2003, Naiman et al. 2012). A mix of approaches could be undertaken, recognizing ecological and other goals such as timber harvest, especially if applied over larger spatial scales (Burnett and Miller 2007, Miller and Burnett 2008, Olson and Rugger 2007), and if consideration is given to the distribution of populations of concern and connectivity among them

[8] Ecological forestry uses silviculture based on knowledge of natural disturbance regimes and succession to manage forests for ecological goals or a mixture of ecological and socioeconomic goals. See chapter 3 for more information.

(Olson and Burnett 2009, Olson and Kluber 2014, Olson et al. 2007). Reeves et al. (2016) provided an example of such an approach and showed that small adjustments in the amount of area in which active management may occur results in substantial increases in wood production while still meeting ecological goals.

We now have a new understanding of the relations between federal forest management and community socioeconomic well-being (chapter 8) that helps us understand the ability of the NWFP to achieve goal 2 (maintain stability of local and regional economies). For example, private forests currently contribute the vast majority of logs processed by mills in the Plan area. Greater timber harvest on federal forests would increase the number of logs available to mills and likely create additional work opportunities for loggers, at least in the short term. Generally, increased federal harvest would reduce the prices paid for logs by mills, which in turn would make wood products producers better off, while making private landowners worse off because their logs will be worth less. However, there are exceptions where mills need to maintain capacity for processing but timber resources are in limited supply, including in forest regions with few mills. In these cases, increased federal harvests can help keep mills from closing, benefiting both wood products producers and private landowners.

Federal forest management can contribute to community well-being in other ways, through the production of a variety of commodities, natural amenity values, other ecosystem services, and employment opportunities, but it cannot ensure the stability of local communities and economies (chapter 8). Not only is community well-being a product of multiple influences at multiple scales; social systems, like ecological systems, are dynamic. Today a more relevant question for managers is how federal forest management can contribute to community sustainability and increase community resilience in the face of social and environmental change. Social, economic, and ecological sustainability are linked, and community resilience contributes to resilient social-ecological systems.

Regarding long-term economic development and diversification (socioeconomic goal 3), the Northwest Economic Adjustment Initiative and Jobs in the Woods programs had mixed results (see chapters 8 and 11). However, alternate formulas for payments to counties embedded in the Secure Rural Schools Act have made important economic contributions to NWFP-area counties and communities, although the future of these payments remains uncertain because the Secure Rural Schools Act expired in 2017.

As to the fourth goal—increased collaboration in forest management—the NWFP was perceived by many people who were interviewed as part of the socioeconomic monitoring program during the first decade of the Plan as moving forest management decisionmaking from the local to the regional level (Charnley 2006b). Since that time, however, the number of forest collaborative groups has grown in the Plan area (from 8 to 25), and the agencies have emphasized the importance of local-level collaboration as a way of doing business (chapter 9).

One way of reducing tradeoffs between the social and biodiversity goals of the NWFP would be to increase activities that contribute to community well-being while fostering the engagement of local communities in conservation. One clear example is to continue attempts to create quality jobs that employ local community residents in ecosystem restoration, research, monitoring, fire suppression, and other activities that contribute to forest stewardship (Charnley 2006a. Although such jobs are unlikely to replace the number of jobs lost over the past few decades in the wood products industry, and may not pay as well, they nevertheless can make a significant economic contribution in local communities and be a source of economic diversification.

Adaptive management and monitoring—

The NWFP was founded on the concept of adaptive management and learning, based on monitoring, adaptive management areas (AMAs), and other forms of reactive, active, and passive adaptive management. Adaptive management, social learning, and landscape-level experiments are key components of increasing social-ecological resilience (Tompkins and Adger 2004). Strategies to promote this type of resilience would include engagement of collaborative groups in management experiments, demonstration projects, and landscape restoration projects. Social networks may be able to help spread adaptive forest management ideas and practices to deal with fire and climate change in the area of the NWFP (Fischer and Jasny 2017, Jacobs and Cramer 2017).

Bormann et al. (2006) provided an indepth evaluation of the adaptive management and regional monitoring program for the NWFP; here we highlight a few key findings. First, the adaptive management program as embodied in the AMAs was generally not successful, as funding for the AMAs declined after 1998, and adaptive management protocols were not widely integrated into agency missions at local scales. However, some successes in active adaptive management did occur. For example, the Central Cascades AMA was the location of efforts to develop and implement alternative landscape-scale approaches to meeting NWFP goals based on mixed-severity fire regimes (Cissel et al. 1999). Other AMAs may have implemented valuable experiments, but we could not find published or unpublished reports that document these actions. Four obstacles to adaptive management in the NWFP area were identified by Bormann et al. 2006: (1) perceived or real latitude to try different approaches on AMAs was too limited; (2) adaptive management was perceived as only a public participation process and there was a lack of consensus on implementing ideas on the ground; (3) precautionary, risk-averse approaches dominated and eventually overshadowed efforts to learn by doing, limiting the ability to increase understanding of systems; and (4) sufficient resources for management activities and the attending followup monitoring and research were not available. The lack of adaptive management activity and restoration activity in general may be a consequence of the fact that federal forest management increasingly takes place in a "vetocratic" setting in which non-Forest Service stakeholders reduce the decision space of managers and make the agency less autonomous than it was previously (Maier and Abrams 2018). According to Maier and Abrams (2018), this situation developed as a way for managers to reduce likelihood of litigation and to provide funding for nontimber objectives that is tied to collaboration.

It also should be noted that the Plan was not implemented as written, as managers responded to various social, economic, and administrative constraints. The implementation of the Plan has occurred through a more reactive or passive adaptive management approach based on resource limitations, social influences, and different interpretations at the ground level. The changes made in implementation of the NWFP include avoiding timber production from older forests in the matrix, ending of surveying for rare species, limited restoration activities in LSRs in fire-prone forests and riparian zones, and, of course, adaptive management itself. Because the NWFP has not been formally changed, it can be confusing to discuss the "Plan" without qualifying whether one is referring to the NWFP as written or as applied.

Obstacles to learning and adaptive management and maintaining an effective monitoring program are not easily overcome (Bormann et al. 2006). Some key principles for more effective adaptive management and monitoring include (1) engaging multi-agency regional executives in guiding learning, (2) involving regulatory agencies, (3) accommodating reasonable disagreement among stakeholders, (4) committing to quality, standardized record keeping by managers, (5) developing long-term funding strategies and maintaining a critical mass of agency expertise, (6) reinterpreting the burden of proof and the precautionary principle so that passive management is not the default and different management approaches can be applied, and (7) allowing for scientifically credible and relevant management experiments to take place even if they do not have total social license.

Although the adaptive management component of the NWFP fell quite short of expectations, the effectiveness monitoring program has been a relative success as evidenced by the valuable and insightful information obtained by 20 years of monitoring of old-growth forest, northern spotted owls, marbled murrelets, aquatic systems, socioeconomic conditions, and tribal relations. Monitoring moved the implementation of the Plan from opinion to evidence-based decisionmaking, helped institutionalize some adaptive management at regional scales, provided evidence of measurement error and variance in key Plan indicators, and demonstrated that agencies can work together effectively.

Plan Goals and Strategies in Relation to New Concerns

Since the development of the NWFP in the early 1990s, several new conservation concerns and issues have emerged that are directly related to meeting its original goals. Perhaps the most significant new concern is the spread of the invasive barred owl and its strong effect on populations of northern spotted owls, as noted above. Here we highlight two other major concerns: (1) the exclusion of wildfire as a keystone ecological process in many NWFP-area forest ecosystems and (2) the role of climate change in profoundly affecting species, wildfire size and severity, and reducing the resilience of dense forests that have accumulated in dry forest zones in the absence of fire.

Fire exclusion and successional diversity—
We have already discussed at length the effects of fire exclusion on forest structure and composition and resilience of dry forests to fire and drought. Here we focus on a somewhat different aspect of that problem, the loss of other successional stages (which contribute to resilience) that are dependent on both low- and high-severity fire. Although not part of the original focus of conservation in the NWFP area, fire-dependent vegetation states are ecologically interdependent with dense old-growth forest in the sense that policies that promote these conditions (e.g., fire suppression) will reduce other vegetation types (Spies et al. 2006). Chapter 3 highlights the ecological significance of open, fire-dependent old-growth forests, including providing habitat for species such as the white-headed woodpecker (*Picoides albolarvatus*), a species that is on Bureau of Land Management (BLM) and Forest Service sensitive species lists for Oregon and Washington as a result of loss of open ponderosa pine forests to logging, and fire exclusion (Buchanan et al. 2003, Mellen-McLean et al. 2013).

Another fire-dependent state is early-successional vegetation (which can also arise from other disturbance agents). The lack of diverse early-successional ecosystems[9] has also become a major conservation concern (DellaSala et al. 2014, Franklin and Johnson 2012, Hessburg et al. 2016, Reilly and Spies 2015, Swanson et al. 2011). Many plant and animal species, including state-listed species, specialize in these early-successional conditions (Swanson et al. 2011, 2014). Some components of these ecosystems can persist for many decades (e.g., snags, dead wood, and open canopies) (Reilly and Spies 2015), but certain conditions within them (snag decay stages and environments for establishment of annual plants) are ephemeral, lasting just a few years. Whereas older forests can take centuries to develop, early-seral vegetation may be initiated in a few hours from a disturbance event and then further develop over many decades before tree canopy closure (chapter 3) (Raphael et al., in press). Maintaining occurrence of these episodic and dynamic ecosystems depends upon relatively frequent disturbance (of either natural or human origin) distributed across large landscapes (Reilly and Spies 2015). Clearcutting on private lands can produce open-canopy conditions that support some early-seral plant and animals species but lack dead and down wood, and active control of herbs, grasses, and shrubs to favor tree establishment and growth greatly limit the ecological diversity and function of clearcuts as surrogates for early-seral ecosystems (Spies et al. 2007, Swanson et al. 2011. Thus, early-successional stages, especially structurally and compositionally diverse ones, are important sources of biological diversity in the NWFP area, but their biodiversity has not been monitored or studied as well as later successional stages.

Despite increasing wildfire activity over the past 25 years, the occurrence of high-severity fire across all NWFP fire regimes has been low: rotations of 1,628 to 2,398 years in moist forest fire regimes and 333 to 690 years in dry forest fire regimes (chapter 3). Although area burned has increased with drought in the past 25 years in the area of the NWFP (chapter 2) (Reilly et al. 2017a), the amount of high-severity fire in moist forest may still be within the full historical range (over the past few thousand years) given the large amount of historical climate and fire variability in the region (chapter 3) (Reilly et al. 2017a, Walsh et al. 2015). However, when climate is taken into account, the recent (past 25 years) amount of high-severity fire and early-seral vegetation

[9] These are ecosystems dominated by shrubs, herbs, and grasses that have little or no tree canopy. They develop after stand-replacing disturbances (see chapter 3) and often contain dead legacies of the previous forest. Site conditions are such that they have the potential to develop into closed-canopy forests that can eventually develop into old-growth forests.

in moist forest regimes is probably low given that we are currently experiencing a warming climate. In addition, we know that more than 6,000 lightning-caused fires have been suppressed in moist forests during the past 20 years (chapter 3) within the Plan area. Thus, it is likely that the amount of early-seral post-wildfire vegetation within moist forest regimes is deficient relative to the historical range of variation, especially for the drier parts of the moist forests. In the historical very frequent fire regimes of the dry forests, large patches of high-severity fire that create early-successional vegetation would not have been common, and early-seral conditions would have occurred as a fine-grained mosaic within a matrix of open older forest (fig. 12-2).

Although early-seral post-wildfire vegetation on sites capable of growing forests appeared to be historically uncommon in most areas of high-frequency, low-severity fire (chapter 3), large patches of nonforest areas, such as savannas, grasslands, shrublands, and even some wetlands would have been relatively common and maintained by fire (chapter 3). These nonforest environments, which have been decreasing in many dry forest landscapes (Hessburg et al. 2007, Skinner 1995), are known to support unique biodiversity based on global-scale studies (Veldman et al. 2015) and may be more reduced than dense old-growth forests in the Pacific Northwest region. However, relatively little attention has been paid to the conservation needs of these nonforest and low-tree-density vegetation types in the literature from the NWFP region.

Climate change—
The effects of climate change have become a major concern and focus of research since the NWFP was developed and implemented (chapter 2). The effects and magnitude of climate change are still uncertain and will differ among species, ecosystem processes, and geographic area. In general, climate change adaptation goals can be congruent or compatible with many of the original goals and strategies of the NWFP, including large reserves in which commodity management and roads are excluded or minimized (Spies et al. 2010a). However, the degree of congruence varies with geography and spatial and temporal scale. For example, efforts to reduce tree density within forest stands and to increase resilience to drought conflict with development of dense, multilayer forest habitat at stand or patch scales (e.g., less than 100 ac). Early-seral vegetation created by wildfire or through restoration management could provide opportunity to plant or naturally establish more drought-resistant genotypes of native tree species (Spies et al. 2010a).

Addressing fish responses to climate change will be especially challenging because of the prominent role of ocean conditions and the importance of nonfederal lands for fish that move through large watersheds (chapter 7). The conservation and restoration strategies of the NWFP can benefit native fish, but there are inherent limits given the complex life histories of anadromous fish and ownership patterns. Populations of introduced or reintroduced fish species may expand under a warming climate and affect native species. Terrestrial and aquatic species responses to climate change will be variable, as mentioned above, or essentially unknown, as with most of the lichens, bryophytes, and invertebrates. We lack scientific assessments of which and how many species may respond negatively to climate change and how management strategies, including protection of climate refugia, silviculture to promote forest resilience, and possibly even managed relocation of organisms might benefit at-risk species (Schwartz et. al. 2012).

Mitigation efforts to limit releases of greenhouse gases and increase carbon storage can be compatible with many NWFP goals. For example, protecting and developing old-growth forests will contribute toward carbon sequestration in forest stands and landscapes (chapter 2). On the other hand, maximizing carbon sequestration will not be compatible with habitat creation for early-successional species (Kline et al. 2016), and may not be consistent with reducing stand density in dry forests to increase resilience to drought, fire, and insects. The tradeoffs between carbon emissions related to thinning and the carbon emissions that are avoided because forests are more resilient to fire- or climate-induced mortality (after thinning) will vary with scale of observation of fire, and forest type (McKinley et al. 2011, Ryan et al. 2010) (chapter 2). Carbon calculators are now available for exploring how different forest management and fire regimes might affect carbon sequestration in the forest ecosystem and in forest products (Zald et al. 2016).

Fire and climate change will also have an impact on some of the NWFP socioeconomic goals. For example, the ability of federal agencies to produce a predictable and sustainable supply of timber, recreation opportunities, nontimber resources such as mushrooms, and fish and game will be challenged as climate change alters weather, ecosystem productivity, and species distributions. Winter recreation associated with snow is already being affected by warmer winters, particularly at lower elevations. And, high-severity fire affects timber stocks and availability of nontimber forest products. As mentioned above, local job creation associated with forest restoration to increase resilience to wildfire, and for fire suppression, can support the Plan goal of contributing to economic development and diversification in communities (chapter 8).

Regional-Scale Issues and Challenges

The regional-scale concerns related to the NWFP goals include (1) the limited ability of federal forest lands to meet some conservation objectives, (2) the need for coordination among management units (e.g., national forests) to provide for population conservation goals and develop standards and guidelines that take regional ecological variability into account, (3) the connectivity and distribution of federal lands as they relate to the capacity of organisms to respond to changing climate and vegetation dynamics, and (4) coordination among ownerships to deal with cross-boundary and regional-scale issues such as wildfire and smoke, watershed processes, populations of sensitive species, and road systems.

The limits of federal lands to meet conservation goals for species and ecosystems were recognized at the time the NWFP was developed. These limits are particularly relevant to the marbled murrelet and the ACS. The marbled murrelet (as well as the northern spotted owl) occur in coastal forests in southwestern Washington, Oregon, and northern California, where the proportion of nonfederal forest land is relatively high (chapter 5). In these areas, continuing loss of marbled murrelet nesting habitat may eventually lead to a large gap in distribution of nesting habitat and thus a potential gap in the marbled murrelet distribution, leading to genetic isolation of northern and southern populations (Raphael et al. 2016). Habitat for six salmonid species is not well provided solely on federal lands because these species find high-quality habitat in lower reaches where most habitat is on private lands (chapter 7). With divergence of forest management intensity between federal and private forest lands, the landscapes may become more "black and white" with old forest on public lands and plantation forests on private lands (Spies et al. 2007). The implications of this landscape change in terms of edge effects and lack of diverse early- and mid-successional stages in the landscape as a whole are not well understood but may result in a reduction in regional biodiversity.

The need for coordination among management units (e.g., national forests, districts) for conservation of populations of listed species and recognition of variability in ecosystems and disturbance regimes was recognized in the development of the NWFP (USDA and USDI 1994b). The need for a regional-scale strategy still exists for the listed species (chapters 5, 4, and 7) (USFWS 2008). Recent science indicates that the regional-scale stratification of disturbance regimes into just two regimes (wet and dry) for purposes of standards and guidelines for management under the NWFP (USDA and USDI 1994b) was too simplistic because it lumped drier, more fire-frequent ecosystems in parts of western Oregon and Washington into one infrequent fire regime, and drier types into a single frequent regime with low- to moderate-severity fire (chapter 3).

Another limitation of the regional perspective that underlies the strategy and implementation of the NWFP is the lack of characterization of regional variability in socioeconomic conditions and aggregation of local-level variability at the human community scale, including community types and their contexts (e.g., proximity to and dependence on federal lands). For example, it might be possible to map regional or local variation in the availability of ecosystem services and well-being of communities (chapter 8) and community dependence on ecosystem services from federal lands. That information could be used to set priorities for meeting socioeconomic objectives and finding areas where restoration needs and socioeconomic needs line up.

The importance of regional connectivity of federal forest lands to provide for movements of plants and animals in response to climate change has been recognized (chapter 3) (Carroll et al. 2010, McRae et al. 2016, Spies et al. 2010a). The distribution of federal lands and reserves appears generally favorable for species that will likely need to move upslope and northward (DellaSala et al. 2016, Spies et al. 2010a). In general, areas occupied by federal lands have a relatively high topo-climatic diversity. Their permeability to movement of vagile vertebrates may be relatively high based on general land cover and use types (fig. 12-10), but it is not known how the distribution and condition of federal lands affects more sessile terrestrial organisms or benefits aquatic organisms.

Quantitative analysis of the effectiveness of the NWFP reserves and federal lands in providing for most species ecological processes, and other aspects of biodiversity under climate change, has been very limited. Carroll et al. (2010) found that "the current reserve system will face challenges conserving its current suite of species under future climates." They suggested that to address climate change for all species revisions to reserve networks designs may be needed. More research is needed to address this issue using updated models of climate, vegetation dynamics, species habitats, population dynamics, and landscape genetics.

The NWFP had a federal lands focus, but it is increasingly acknowledged that an all-lands or a multi-ownership perspective would be beneficial in dealing with issues such as fire, climate change, watersheds, and recovery of listed and at-risk species (chapters 4 and 7) (Bone et al. 2016; Charnley et al. 2017; Spies et al. 2007, 2010b). All-lands approaches can be promoted in several ways including prioritizing actions on federal lands based on conditions (context) in nearby nonfederal lands; providing funding mechanisms to support restoration on public, private, and tribal lands within shared landscapes; and coordinating management actions within watersheds and landscapes, where social and administrative processes enable such actions (Charnley et al. 2017, Knight and Landres 1998).

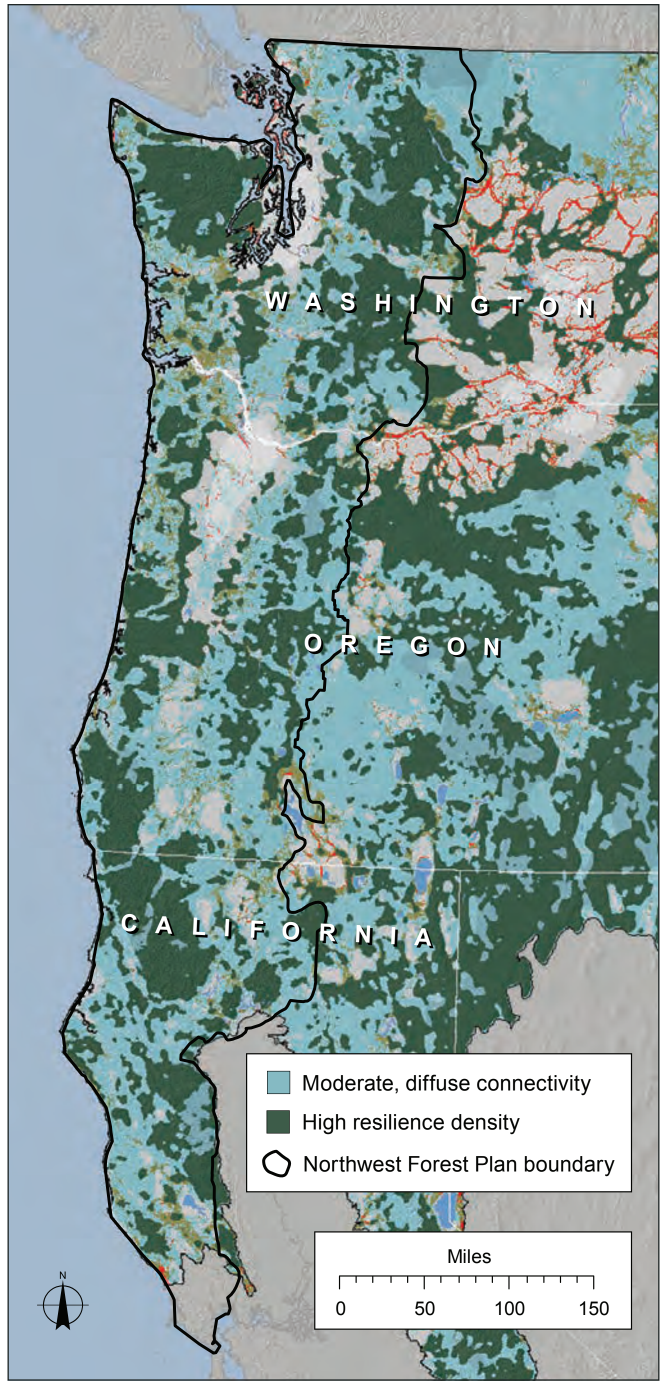

Figure 12-10—Regional connectivity and terrestrial resilience to climate change effects based on land cover types (connectivity) and topoclimatic conditions (resilience). Illustration adapted from McRae et al. 2016. Blue represents moderate levels of diffuse connectivity (movement is largely unrestricted); dark green represents areas of high resilience density (topoclimatic diversity).

Tradeoffs Associated With Restoration

Because the ecological goals of the NWFP are not necessarily consistent with addressing new conservation issues (e.g., the tension between managing for dense old-forest species versus open old-forest or early-seral species), it should not be a surprise that forest management activities for specific restoration goals would have variable effects across a spectrum of ecological and socioeconomic goals. We have touched on some of these in the previous section; here we summarize these in more detail in terms of specific management actions and how they might affect different management goals (table 12-1). Most of these effects are discussed in greater detail in other chapters of this report.

Variable-density thinning in plantations in moist and dry forests—

Variable-density thinning in plantations in uplands and riparian areas to immediately increase vegetation diversity and accelerate future development of large tree boles and crowns has a variety of effects across all fire regimes, as noted elsewhere in this document (chapters 3, 4, 5, and 7). Thinning can have immediate positive effects on several species; e.g., some lichens and bryophytes (chapter 6), and can accelerate growth of larger trees, but it reduces dead wood amounts compared to the unthinned state unless some thinned trees are left on the site. Studies of effects of variable-density thinning on invertebrates in western Washington indicate that the effects can be positive, especially in the short term, or negative depending on time since thinning, forest structure, and environment (Schowalter et al. 2003). Increasing spatial heterogeneity of the tree layer in plantations creates discontinuous fuel beds, increases structural and compositional diversity, and restores some of the heterogeneity that would have occurred in young post-wildfire stands. Similarly, thinning in riparian plantations can accelerate growth of large trees that occurred in variable densities near streams. Dense, uniform plantations are an altered ecosystem that may not serve as a good reference for management in riparian zones, many of which were historically a mosaic of older conifers, hardwoods, and shrub patches, especially near larger streams (chapter 7) (fig. 12-10). Thinning in plantations in riparian areas can also increase spatial heterogeneity of trees and shrubs and increase overall biotic community diversity, but reduce shading, which can increase stream temperatures (chapter 7). The role of thinning in increasing resilience of forests climate change has received only limited empirical study globally (chapter 2) (D'Amato et al. 2013, Elkin et al. 2015, Seidl et al. 2017).

Restoration of fire-excluded forests—

Thinning and prescribed fire to restore structure, composition, and resilience to older forests that historically experienced frequent fire can have numerous site- and landscape-level benefits (chapter 3; table 12-1) (Hessburg et al. 2016) that are both ecological and social. Restoration for ecological integrity and conservation of listed species can improve resilience to climate change and fire, and habitat for open old-growth species. Reducing fuel loads and increasing the heterogeneity of amounts and types of fuel can also reduce the potential extent of large patches of high-severity fire that result in losses of denser forest habitat. This practice can have adverse effects on northern spotted owls (but see North et al. [2017] for a different perspective) and some species such as fisher and marten that use dead wood as sites for foraging, resting, and denning. Little published science exists about blending the goals of conservation of northern spotted owl habitat and restoration of fire-dependent forest ecosystems at landscape scales. As experience with the Blue River plan (Cissel et al. 1999) indicates, this is both an ecological and socioeconomic problem that requires more research and evaluation through adaptive management and collaborative landscape efforts that try new approaches to the problem.

Restoration of fire-excluded forests also has social and economic benefits, particularly by reducing the risk of loss of property, structures, and lives to high-severity wildfire in the wildland-urban interface; by producing wood products and biomass that can be utilized; and by creating jobs. Tradeoffs include the impacts of smoke from prescribed fire treatments, the risk of escaped prescribed fire, and the cost of restoration treatments in areas where there are insufficient larger trees to provide revenue to offset restoration costs

Early-seral vegetation in moist forests—
Given that fire suppression has reduced the occurrence of early-seral vegetation, innovative silviculture including prescribed fire (such as ecological forestry) (Franklin et al. 2007), could be used to create large enough patches of early-seral conditions that are minimally influenced (e.g., by shade and belowground effects) from adjacent forest areas. To reduce impacts on existing older forests, such actions would be best focused on existing plantations, especially in matrix areas. Such activities would allow for establishment and persistence of early-successional species, including shrubs, and would contain large-diameter dead and some live trees that would be characteristic of higher severity post-wildfire environments (Franklin and Johnson 2012, Franklin et al. 2007) and that would serve as "legacy" elements of the previous stand conditions. The amount of retention of live trees would be variable to match variation in fire effects and site capacity at patch and landscape scales. Prescribed fire could be used in conjunction with this action to approximate some of effects of wildfire, especially on soil surface layers and understory plant and animal communities. This type of silviculture could meet diverse ecological and socioeconomic goals in both regimes of the moist forests and could target stands of any age because wildfire would occur across the full range of successional stages. However, when applied in older forests in the matrix, there are some tradeoffs (table 12-1). Large early-seral and nonforest patches do not provide habitat for late-successional species unless those species use early-successional and edge environments for some facet of their life history requirements. Cutting larger or older trees to create early-seral patches can provide larger volumes of wood for local mills, but it may not be socially acceptable because the focus and expectations of the Plan are currently to protect all remaining older forests from logging, and such harvest may conflict with the need to protect owl habitat given the threat of the barred owl. Recognizing these concerns, Franklin and Johnson (2012) have proposed that this type of habitat creation focus on stands less than 80 years old in the matrix. When applied in older plantations, this activity could produce significant amounts of wood and be a potential win-win for biodiversity and socioeconomic values. It should be noted, however, that there is little research and management experience in this type of restoration. In addition, using mechanical treatments to create early-successional habitat in younger forests and plantations will not provide large dead trees and other vegetation structures of late-successional and old-growth forests, nor some of the fire effects of naturally created early-successional vegetation (e.g., very large patches of early-seral ecosystems).

Post-wildfire management—
Post-wildfire management typically includes both salvage logging and planting of trees, which may or may not occur together in management. The ecological effects of postfire salvage logging can differ depending on treatment, fire severity, and biophysical setting (Peterson et al. 2009), but, in general, much existing research indicates that salvage logging does not have beneficial ecological effects on terrestrial or aquatic ecosystems (chapter 3) (table 12-3). However, there may be some exceptions to this rule. Peterson et al. (2015) and Hessburg et al. (2016) identified situations, e.g., concerns about lack of seed sources or reburns that maintain undesirable shrub fields, in which postfire wood removal might meet ecological goals. These include (1) fuel reduction treatments that reducs levels of large woody fuels derived from shade-tolerant species that may have accumulated under fire suppression and may pose a risk to soil fertility were the area to reburn; and (2) fuel treatments to reduce potential for high-severity reburns, and planting of trees to speed rate of forest succession where the potential for large semistable patches of shrubs is high and regeneration is lacking (Coppoletta et al. 2016, Dodson and Root 2013, Lauvaux et al. 2016, Meng et al. 2015); and (3) to reduce surface fuels that may impede establishment of trees. Sudden oak death also is likely contributing to ecologically novel configurations of dead trees and high fuels that may warrant interventions to reduce the potential for undesirable effects of reburn on soils.

Where timber salvage is conducted, reserving dense patches of snags adjacent to salvaged stands, rather than uniformly retaining small numbers of snags across a landscape, may be essential for sustaining populations of

Table 12-3—Summary of socioecological impacts of postfire management (salvage or planting)

Issue	Cons	Pros
Carbon	Carbon in dead trees may be slowly released as wood decays, and some may enter long-term pools in soils or in streams	Burned trees can be used as harvested wood products or can offset energy from more carbon-intensive energy sources when burned in biomass facilities; replanting of trees has potential to accelerate long-term carbon storage in areas where natural regeneration is poor
Wildlife habitat	Negative impacts on wildlife communities of removing biological "legacies" such as standing and down wood, particularly "early-successional" species that depend on standing snags	Planting of trees can accelerate forest development and reestablishment of late-successional habitat
Erosion	Mechanical activity can pose risks of increased erosion and runoff	Residual materials can be used as source of ground cover
Wood loading to streams	Removal can interrupt important process for storing sediments and reforming aquatic habitats	Reducing excessive wood loading could lessen risk of debris jams and downstream culvert/bridge failures
Fuel loading/fire hazard	Salvage can increase loading of fine fuels, leading to increased fire severity upon reburn; planted stands are highly vulnerable to fire for decades	Removal of excessive fuel load can moderate future fire severity and fire behavior in some contexts; can reduce risk to firefighters
Forest development	Salvage has potential to affect natural revegetation by trees and shrubs	Salvage plus replanting can accelerate return to forest conditions in areas
Economic returns	Investments in planted stands may be lost, especially as climatic conditions become less favorable to tree establishment and more favorable to frequent reburns, and they may also complicate use of fire at landscape scales	Timber from burned areas has high economic value, and returns can be used to offset costs of hazard reduction and long-term restoration; replanting can accelerate regrowth of timber-producing forests

early-successional species such as black-backed woodpecker (*Picoides arcticus*) (White et al. 2016b). Within riparian areas, more research is needed to understand variation in wood loading and whether there are loads that are detrimental to stream function, as well as the effects of riparian snag patches of different densities and sizes. As with terrestrial systems, retaining large snags that are likely to remain standing longer, and which are more likely to form persistent elements of aquatic ecosystems, could help to extend and moderate the input of large wood. Fuel hazard reduction might be achieved in part by removing smaller dead trees for biomass utilization or masticating them into ground cover where soils are severely burned and lack protective cover.

Roads—

The ecological effects of roads have been extensively reviewed in the literature (chapter 7) (Fahrig and Rytwinski 2009, Jones et al. 2000, Trombulak and Frissell 2000). The ecological effects of roads affect both terrestrial and aquatic ecosystems but are especially pronounced for aquatic ecosystems and species as the following list of impacts (chapter 7) indicates:

1. Accelerating erosion and increasing sediment loading.
2. Imposing barriers to the migration of aquatic organisms, including access to floodplains and off-channel habitats.
3. Increasing stream temperatures.
4. Causing changes in channel morphology.

5. Introducing exotic species.
6. Increasing harvest and poaching pressure.
7. Changing hillslope hydrology and resulting peak flows.

In the case of hydrological processes, the majority of roads have negligible effects, suggesting the need for a landscape approach to identity problem roads and prioritize road decommissioning. Hydrologically problematic roads constrain floodplains or have direct hydrologic connectivity with fish-bearing streams, but most streams in a network are not fish bearing.

On the other hand, roads are needed for forest restoration management, recreation, access to tribal resources and nontimber forest products, timber harvesting, and fire suppression. Roads are the primary way for people to access public lands, including private inholdings and historical tribal use areas. Decommissioning roads can help both reduce ecological impacts and reduce maintenance costs, which can be significant, but some road systems area still needed to meet other objectives. For example, roads provide access to forests and wilderness areas and are the pathways to special places to which people form strong attachments through repeated use. Roads also provide access to areas of the forest that generate incomes and provide jobs, as well as access to food and forage used by the public for everyday sustenance and survival. The costs associated with road decommissioning, which involves regrading, removing culverts, and revegetation, often make this option impractical. Roads that may be decommissioned by default through neglect may become safety hazards and sources of public conflict. Roads and road decommissioning are a prime example of tradeoffs associated with meeting competing goals for federal forests, including ecological restoration.

Uncertainty and Risk in Forest Planning and Management

Uncertainty and risk have long been a part of forest management and planning. However, as management objectives have shifted from commodity production to a broader range of ecological and social values from complex ecological and social systems (Moore and Conroy 2006, Rose and Chapman 2003), it has become even more crucial to consider ways of dealing with uncertainty, risk, and tradeoffs (Spies et al. 2010a). In addition, the threats from climate change, undesirable fire effects, invasive species, and social change introduce new drivers of forest ecosystems and management goals that are difficult to predict, control, and have variable effects on ecosystems and forest values. Uncertainty is defined as lack of information that falls on a continuum between absolute determinism and total ignorance (Walker et al. 2003). Risk can be defined as the probability (often not well known) of some, often undesirable, occurrence.

Uncertainty and risk pervade our understanding of the species, ecosystems, and social systems of the NWFP area. We know a great deal, of course, as the chapters of this synthesis demonstrate, but we also know that our knowledge in some key areas (e.g., persistence of the northern spotted owl and climate change effects, suitability of conditions other than old growth being favorable for fish and other aquatic organisms) is uncertain, and that the ability of management to achieve particular outcomes can be quite unsure. We also know that many forest values are at risk from influences that are both internal and external to the NWFP area and outside the control of forest managers (e.g., climate change and markets for wood products). Although concepts of uncertainty and risk are well known from the forest planning literature, the practical applications of this theory in decision support models and management are rare (Pasalodos-Tato et al. 2013). Managers and scientists may not be comfortable in admitting to the public that they are unsure of outcomes of proposed actions, but ignoring or not acknowledging uncertainties, risks, and tradeoffs can lead to poor decisions and bad planning alternatives (Pasalodos-Tato (2013). Although uncertainty is pervasive, it should not necessarily be seen as a reason for inaction (Dessai and Hulme 2004).

Several strategies exist for incorporating uncertainty and risk in forest management or biodiversity conservation. For example, Lindenmayer et al. (2000) suggested four approaches: (1) establish biodiversity priority areas (e.g., reserves) managed primarily for the conservation of biological diversity; (2) within production forests, apply structure-based indicators including structural complexity, connectivity, and heterogeneity; (3) use multiple conservation strategies at multiple spatial scales, spreading out risk in wood-production forests; and (4) adopt an adaptive management approach to test the validity of structure-based indices of biological diversity by treating management practices as experiments. Lindenmayer et al. (2000) also noted that "a biodiversity priority area should not imply a lack of need for active management regimes inside that area…such as the restoration of burning regimes that may be required by taxa dependent on particular seral stages or vegetation mosaics." Others have also called for risk spreading by creating heterogeneous systems at stand and landscape scales (Hessburg et al. 2016, O'Hara and Ramage 2013). In general, adaptive management (including monitoring) is considered one of the most important strategies for dealing with uncertainty (e.g., acknowledging it and reducing it) in forest planning and management (Keenan 2015, Moore and Conroy 2006, USDA FS 2012). Although more passive learning approaches can be successful, active and intentional adaptive management is much more likely to reduce uncertainty (McCarthy and Possingham 2007). It should be reiterated that active adaptive management is expensive and time consuming, however, how often have scientists and managers looked back 10 years and lamented lack of action to pursue such work?

Other approaches for dealing with uncertainty, risk and tradeoffs involve governance systems and interactions with stakeholders in plan development and implementation. The goals here are not so much to reduce uncertainty but to incorporate it into decisionmaking and communications with the public to provide more flexibility to change plans and management approaches to meet new challenges. Strategies include communication by managers with communities (in the case of natural hazards like fire), collaboratives, partnerships with nongovernmental organizations and planning boards (Calkin et al. 2011), and engaging stakeholders to improve plans and decisionmaking (Bizikova and Krcmar 2015, Keenan 2015).

Scenario analysis can help deal with and communicate to stakeholders the reality that social-ecological system complexity and stochasticity preclude prediction and certainty about management effects. Scenario analysis was used to inform forest management and policy across 13 states in the Southeastern United States (Wear and Greis 2012). In scenario analysis, a range of plausible futures is identified, and the consequences of different management strategies are evaluated with models (e.g., discussion/decision support tools) or expert opinion. This approach can help identify management alternatives that are likely to fail under certain futures and other alternatives that may provide some level of desired outcomes across a range of possible futures. Such efforts may help communicate sources of uncertainty and the idea that plans need to be flexible and adaptive to respond to unexpected and undesirable future outcomes and tradeoffs. However, this approach is also very labor intensive, involving much up-front work before engaging with stakeholders to develop and evaluate scenarios (Bizikova and Krcmar 2015). The challenges are many, including designing the social process of stakeholder engagement and interactions of stakeholders with data and models.

Policy research indicates that in our current biophysical and socioeconomic environment, forest plans must not only meet ecological and socioeconomic goals but also be robust and adaptable over time (Walker et al. 2013). Walker et al. (2013) listed three key principles to guide development of robust forest plans:

- Explore a wide variety of relevant uncertainties including natural variability, external changes, and policy responses.
- Connect short-term targets with long-term goals.
- Commit to short-term actions that keep options open for the future.

The NWFP was designed to be adaptable (e.g., through research, monitoring, and adaptive management), but as we described above, the adaptive management component of the Plan and some of the monitoring components did not survive for various social and economic reasons. Nevertheless, the idea that forest plans should be adaptable and underpinned by adaptive management is still considered the best way forward in a dynamic and uncertain world.

Information Gaps, Research Needs, and Limitations

Monitoring—

We lack information about the amount, pattern, and type of restoration activities that have occurred in upland and riparian forests. Implementation monitoring has not occurred to a degree that we can know the rate, pattern, and type of restoration actions across the NWFP area. Effectiveness monitoring has provided useful information (e.g., about the northern spotted owl and marbled murrelet), but disinvestment in some aspects of NWFP monitoring over time (e.g., socioeconomic, implementation, Survey and Manage species) has limited the amount and usefulness of the monitoring information produced. Research is needed to determine how well the current set of monitoring metrics (e.g., old-growth index) address issues related to fire exclusion (e.g., metrics for open canopy, old-growth forests) and climate change, and how effectiveness monitoring can be better linked with validation monitoring and research. Research is also needed to better understand what is causing the monitoring trends observed and how to address undesirable trends.

Climate change—

Uncertainties about the effects of climate change on ecosystems, including fire activity, remain large owing to regional variability, complex interactions, and the coarse spatial scale of projections. Having large areas dedicated to promoting biodiversity and resilience to climate change is a foundational strategy, but we lack quantitative analyses of how different management approaches to biodiversity conservation affect vulnerability to climate change. Silviculture, including innovative tree planting strategies, may help improve resilience of forests to climate change impacts (e.g., large patches of high-severity fire). However, we lack information on how future vegetation communities might form and adapt to different climate scenarios to fully understand the interactions and tradeoffs. We also are challenged to estimate how vegetation might change across time and landscapes under different climate scenarios and the degree to which various measures of and objectives for "forest resilience" may be met. This lack of information also tempers our confidence in climate change adaptation strategies for human communities. Landscape-scale models and tools are needed to analyze scenarios and the effects of alternative landscape designs on species, ecosystems, and human communities. New monitoring field studies and assessment tools are needed to evaluate stress and mortality in forests at landscape scales and to test hypotheses from landscape simulation models that are a major source of information about possible future climate change effects.

Species and ecosystems—

We have virtually no published information about how northern spotted owls respond to wildfires, including increased frequency and severity of fire. We also need to improve our understanding about interactions between northern spotted owls and barred owls and their niche separations to help identify key areas for northern spotted owl conservation.

Effects of fire suppression (e.g., increased forest density and increased proportion of shade-tolerant trees) on ecosystem processes and population responses of plants and animals are not well understood in the area of the NWFP. More research has been conducted on how changes in stand structure and composition affect fire behavior than on how those altered forest conditions affect resilience to drought, biodiversity and ecosystem function, and successional trajectories.

Conservation and restoration strategies—

The limits (ecological and social) to restoring forest ecological integrity (per the 2012 planning rule) and resilience with fire (both prescribed and wildfire managed to achieve resource objectives) across diverse landscapes are not well understood. More fundamentally, we need research to help develop definitions and metrics of integrity and resilience so that managers can operationalize them at different scales. It is unclear if we have passed tipping points (e.g., crossed ecological and socioecological thresholds that make it difficult to restore desired conditions) in some landscapes that have been transformed by the cumulative effects of altered disturbance regimes and climate change. In addition, the ecological and social impacts of using surrogates (e.g., mechanical fuels treatments) for fire are also not well understood across the fire regimes of the NWFP area, especially for biodiversity (most work has focused on forest structure and composition change, and fire behavior); previous work suggested that such surrogates may not serve well if they do not pay attention to biological legacies (Franklin et al. 2000). For example, research is needed to help us understand how well mechanical methods and prescribed fire create diverse early-successional habitat and functions, especially when applied to forest plantations. Although theory supports the hypothesis that biodiversity and ecosystem function associated with post-clearcutting environments and young plantations (e.g., on private lands) are different from post-wildfire or post-windthrow environments, no empirical research has been conducted.

Relatively little published research has focused on how well the regional NWFP strategy of reserves and associated management guidelines will meet biodiversity goals under changing climate and fire regimes. Research is needed to understand the ecological tradeoffs associated with alternative conservation land allocations and designs based on different ecological priorities (e.g., single species versus multiple species and processes).

Tradeoffs associated with alternative management strategies—

Although we have some knowledge of the tradeoffs associated with restoration and conservation strategies to meet ecological and socioeconomic goals, we generally lack knowledge of how those tradeoffs and interactions differ across the region, with scale, and over time. Reliance on precautionary approaches that avoid interventions may produce unintended outcomes because no action (e.g., not thinning a plantation or not using fire) may have undesirable effects (e.g., less biotic community diversity). In such cases, rigorous adaptive management approaches (e.g., learning by doing) are considered the best way to address uncertainty and complexity (Walters 1986). Research is needed for understanding the long-term and landscape-scale effects of restoration on terrestrial and aquatic species, biodiversity elements, and ecosystems and how these actions interact with social systems.

Scientific literature has been fairly clear in indicating that the benefits from salvage logging are generally economic, in the form of wood products, rather than ecological. However, we lack information on the long-term effects of salvage logging in burned forests whose density and composition have been heavily altered by fire exclusion before the fire. As the likelihood of reburn in immature forests increases with climate change, the rationale for such interventions may grow. In addition, we lack information on when and where planting might be needed and what kind of salvage might be appropriate, if at all, to facilitate recovery of desired forest conditions following large high-severity wildfire events. Finally, where salvage logging is conducted for economic objectives, we lack studies that quantify the ecological effects of salvage logging when managers seek to meet both ecological and economic goals through carefully planned approaches to post-wildfire management.

Social-ecological interactions and collaboration—

Although ecosystem services are now widely recognized as a framework for characterizing the range of values on federal forests, relatively little quantification and application have occurred on federal lands. Some ecosystem services, particularly cultural services such as support for spirituality or solitude, are important to many, but difficult to quantify or monetize. In addition, the potential for tradeoffs among ecosystem services (e.g., carbon sequestration, habitat for some species of wildlife, water supply, and regulation of fire), particularly across long periods and large areas, is not well understood. Research is needed to determine the best methods for quantifying ecosystem services, understanding tradeoffs, and using qualitative approaches in planning and management when quantification of ecosystem services does not exist. In addition, research is needed to determine the costs and benefits (e.g., providing more public support for investment in public lands) of using an ecosystem management framework compared to alternative ways of valuing and communicating the benefits that public lands provide.

Low income and minority populations protected by the 1994 Executive Order on Environmental Justice have increased throughout the NWFP area over the past two decades. This trend increases the need for ongoing research into how these populations relate to federal forests and are affected by their management. There is a fairly substantive literature about how minority populations relate to national forests in terms of work (e.g., forestry services work, commercial NTFP harvesting). However, apart from recreation, little information is available about noneconomic relations between federal forests and low-income or minority populations (other than American Indians). Furthermore, research is only beginning to fill the gap in knowledge about the environmental justice implications of Forest Service management actions. For example, there remains a lack of information about how fire—managed, prescribed, or wild—and associated smoke affect low income and minority populations in the Plan area. There is also little information about how management activities that influence forest structure and composition affect uses and values of associated species that are valued by these populations.

The ability to undertake active management to achieve diverse ecological and socioeconomic goals is constrained by many factors, but limited public trust in federal managers is among the most critical, especially when it comes to working in forests with larger or older trees in frequent and moderately frequent fire regimes. Forest landscape collaboratives provide socioecological laboratories for studying how interactions among stakeholders and federal managers affect the ability to achieve restoration and resilience to fire and climate change. These collaboratives are relatively new, and study results are still unfolding. However, findings thus far suggest that collaboratives have not been a cure-all for resolving conflicts about public values and minimizing litigation, but in some cases participants have suggested that progress has been made on those measures (Schultz et al. 2012, Urgenson et al. 2017). A contributing factor to those trends has been social learning by agency staff in managing their roles (Butler 2013), adopting new approaches such as multiparty monitoring and use of stewardship contracts, as well as picking collaborative projects that have a high likelihood of success. More information is needed about public responses to restoration management efforts, especially in complex contexts such as mixed-severity fire regimes (Urgenson et al. 2017), and addressing socioecological objectives including timber production while applying nonindustrial, ecological forestry methods.

We lack understanding of how trust at different organizational scales (individual, district, forest, national) affects public understanding of and support for various types of active forest management strategies. Finally, although research suggests that the efforts required for collaboration can be taxing on both agency staff and community stakeholders (Urgenson et al. 2017), we lack information on appropriate forms and levels of support to bolster the capacity of both for long-term engagement in collaborative processes.

Conclusions and Management Considerations

The goals of the NWFP for federal forests occur within a diverse, dynamic, and complex social-ecological system that has changed in significant ways since the Plan was implemented. For example, the capacity of the agency and of the forest industry to conduct restoration efforts across landscapes has declined significantly; budgets for managing resources are greatly diminished, and wildfire suppression programs and budgets overshadow most other work. The contributions of public forest lands to ecosystem services (e.g., carbon sequestration and water supply) are now more widely recognized than ever, but the ecosystem services framework has only just begun to be implemented at forest and project scales and not been applied yet in assessments and forest plan revision (Deal et al. 2017b). A major change in biodiversity conservation policy has also occurred for the Forest Service in the form of the 2012 planning rule, which emphasizes whole ecosystem approaches to conservation in contrast to previous planning rules, which emphasized population viability of individual species, and which the agency considered "procedurally burdensome to implement" (Schultz et al. 2013). NWFP monitoring indicates that progress is being made toward meeting several of the original long-term goals, namely maintenance of vegetation conditions that support northern spotted owls and marbled murrelets, protecting dense old-growth forests, providing habitat for aquatic and riparian-associated organisms, and reducing the loss of mature and old forests to logging, (Bormann et al. 2006, DellaSala et al. 2015). Other goals, such as providing for a predictable timber harvest to support rural communities, road decommissioning, adaptation, learning through adaptive management (Bormann et al. 2006, Burns et al. 2011) (chapter 8), and effectiveness and validation monitoring of old-forest species and biodiversity (chapter 6) have not been realized. Finally, Congressional legislation that provided alternative formulas for payments to counties most affected by the Plan to mitigate the financial impacts of reduced timber harvesting were realized in the short to mid term, but their long-term viability remains uncertain (Phillips 2006). In addition, new concerns have emerged that were not part of the original Plan, including a major threat to populations of the northern spotted owl from the native invasive barred owl, widespread loss of fire-dependent ecosystems including open old-growth, early-seral forests, nonforest communities, increased influence of exotic invasive species, and climate change.

Over the past 150 years, timber harvest, fire exclusion, and the loss of American Indian burning have profoundly changed both moist and dry forests of the NWFP area. Although the motivation for the Plan arose from halting 20th century clearcutting of old growth, moist forests and the associated loss of habitat for the spotted owl habitat and other old-growth forest species, the dry forests, which occupy about 43 percent of the Plan area, probably have experienced much more pervasive ecological changes as a result of human activity (chapter 3). Key changes in dry forests are loss of large, typically open grown, fire-resistant trees to logging; large increases in surface and canopy fuels and their connectivity; widespread shifts in seral-stage dominance; and changes in the patch size distributions of those seral stages. These changes have affected all species and processes; some in favorable ways (e.g., more habitat for dense, young multistory forest associates) and others in unfavorable ways (e.g., loss of open old-growth and early-seral forests, and associated resilience to fire and drought). Changes in moist forests are also significant, but they have been affected differently by logging and fire exclusion. Here, intensive timber harvest has been the primary impact on biodiversity by dramatically fragmenting and reducing the amount of closed-canopy old-growth forests, and habitats for the associated species. Fire exclusion in moist forests has also had important effects as well; historical fires created a highly diverse seral-stage patchwork with many patches of early- and mid-seral-aged forest. This patchwork is now highly altered.

Strategies are available to move these ecosystems, forests, landscapes, and species toward conditions that appear better aligned with policy direction (e.g., ecological integrity under the 2012 planning rule) and with current social values, both utilitarian (e.g., clean water, sustainable production of wood and special forest products, recreation) and intrinsic (nature for its own sake). The challenge will be to determine how to prioritize restoration goals and distribution actions across landscapes. Ecological history can be a valuable guide for restoration, but land managers, in reality, cannot restore ecosystems to any particular historical period or condition, or meet all management objectives in one area of land. However, they can learn from the historical conditions about the kinds of patterns and patch size distributions that offered the best hedging strategies against large wildfires and climate warming. Managers can take actions that increase the likelihood of retaining desired ecosystem services, species, intrinsic values of forests, and resilience to climate change and disturbances, even if their actions produce forest conditions that are altered relative to the pre-Euro-American period. Ecological and social history demonstrates that change is inherent in these forests, and we appear to be entering a new period of rapid change with uncertain outcomes.

Species and ecosystems—

The current outlook for widespread persistence of the northern spotted owls is not good. It appears unlikely that the northern spotted owl can persist without significant reduction in barred owl populations. However, without the implementation of the NWFP (e.g., if the pace of old-growth logging from the 1970s and 1980s had continued for 23 years), northern spotted owl populations would likely have already become moribund. Forests capable of supporting interconnected populations of northern spotted owls have increased or stayed relatively stable at the Plan scale. However, the rapid pace of climate and fire regime change suggests that recent trends may not continue. Continued success at conservation of northern spotted owls under the NWFP rests on understanding how to minimize the impacts of barred owls and on how to manage dry and moist zone forests in ways that increase rather than reduce future resilience to wildfire and climate change effects.

Under the original NWFP goals, an emphasis on multilayered old-growth forest conservation was critical given its relationship to owl habitat occurrence and its reduced abundance through harvesting. However, the 2012 planning rule emphasizes ecological integrity and resilience (ecosystem goals that were not part of the NWFP goals), and deemphasizes species viability approaches, a policy change that could significantly affect the conservation goals for biodiversity in the NWFP area. Managing to maintain current levels and patterns of multilayered old forests in dry forest zones (the NWFP goal) will not promote resilience of those dry forests to climate change, fire, and other stressors, and it will not restore more natural ecosystem dynamics. The new rule also has implications for supporting human communities, including tribes with protected treaty rights. Finally, the using ecological integrity as a guide means that conserving biodiversity in this region is more than just conserving dense old-growth forests—other stages are valuable, including open old growth, diverse early- and mid-successional post-wildfire vegetation, wetlands, oak-dominated forest patches and woodlands, and shrub-lands and grasslands.

Conservation and restoration—

The contribution of federal lands to the conservation and recovery of ESA-listed fish, northern spotted owl, and marbled murrelet populations continues to be essential, but it is likely insufficient to reach the comprehensive goals of the NWFP, or the newer goals of the 2012 planning rule. Contributions from streams and forests on nonfederal lands are important to achieving NWFP conservation goals, especially under climate change, which may shift species distributions. Transboundary collaborative efforts can help to address challenges such as restoration of fire regimes, and can enhance conservation efforts, especially when supported with innovative arrangements to share funding, resources, information, or liability, such as the Fire Learning Network and Training Exchange (TREX) program to support prescribed burning (fig. 12-11) (Goldstein and Butler 2010). These efforts have supported collaborations that have engaged tribes, including the Western Klamath Restoration Partnership (see chapter 11). Such approaches combined with other incentives can help to increase conservation on nonfederal lands, but further research to evaluate the impact of particular approaches within the NWFP context is needed.

Under current goals, a restoration strategy would likely combine efforts to ameliorate anthropogenic impacts, such as culverts that are likely to fail in priority watershed areas, as well as some dams and diversions used for irrigation water withdrawal, while also directing active management interventions, such as intensive thinning and use of fire, to restore degraded systems or at least increase their resilience to climate change and fire. Such active management may be particularly valuable in areas where both fire regimes and forest structure have been dramatically altered, because it can increase the likelihood that wildfires will help promote rather than erode resilience.

With congressional reserves, LSRs and riparian reserves, and administratively withdrawn areas occupying more than 80 percent of the Forest Service and BLM land base in the NWFP area, rates of additional fragmentation of older forests outside of reserves from management activities on federal lands will be very low. Landscape-level change will be dominated by succession of young and mid-seral forests, with increasing area of disturbance from wildfire. Concerns over connectivity among old-growth forests and LSRs have shifted to climate change effects and access to climate refugia, although the effects of past logging on connectivity remain. The widespread effects of roads on species and ecosystem processes also remain a conservation concern, especially those that constrain full floodplain functioning or contribute high sediment loads.

The small amount of logging within nonreserved northern spotted owl habitat or mature and old-growth forests over the past 15 years of NWFP implementation does not reflect the original provisions of the Plan as written, but it does mean that the major historical threat to biodiversity (commercial logging of old-growth forests) has been greatly reduced on federal lands. This outcome may have been largely a result of the Survey and Manage program and changes in the social acceptability of cutting old growth in the matrix. The lack of harvesting of older forests outside the reserves means that a major motivation for adding hun-dreds of species to the Survey and Manage lists no longer

Figure 12-11—2015 Klamath River Training Exchange prescribed fire at night.

exists (i.e., the older forest habitat needs of the northern spotted owl do not necessarily cover the needs of other late-successional species). The Survey and Manage program was abolished under its economic weight and because fewer older forests were being logged than originally projected.

Fire suppression in some parts of the moist forest region has reduced the amount of structurally diverse early-seral vegetation over the past several decades. It also has likely reduced the diversity of older forest structural and composition conditions and landscape diversity in the drier parts of the moist forest zone. Managers could explore opportunities to restore fire effects in these systems through combinations of thinning, prescribed burning, and managing wildfires. In theory, such restoration actions could occur in the matrix in forests with old trees (e.g., greater than 80 years old), but the ecological and social acceptability of this activity are unknown. The issue is well suited for adaptive management studies. Possible win-win (wood production and biodiversity) alternatives are to create early-seral vegetation in plantations in the matrix or to do more active management in plantations in riparian reserves using principles of ecological forestry or restoration silviculture.

A major challenge to management for resilience to fire and climate change exists in landscapes that historically experienced frequent fire in northern California, southern Oregon, and the eastern Cascade Range of Oregon and Washington. Fires in these areas have been much less frequent in recent decades than historically. However, some recent fires have created larger patches of high-severity fire compared to the historical regime, likely as a result of fuel continuity. The denser forests and more shade-tolerant tree species have increased the area of northern spotted owl habitat despite losses to fire in recent years (chapter 3). Landscapes that include northern spotted owl habitat reserves, in which little or no restoration or management to restore fire and successional dynamics occurs, likely will not provide for resilient forest ecosystems in the face of climate change and increasing fire. Prioritizing conservation of dense forest habitats that have increased in area with fire exclusion is not congruent with managing forests for ecological integrity or resilience to fire and climate change. Management strategies that promote resilience in fire-prone forest landscapes include restoring fire and the patchwork of open and closed-canopy forests, and tailoring these conditions to topography. Landscape-level strategies are needed to provide for dense forest conditions, where they would typically occur, and would be more likely to persist in the face of coming wildfires and a steadily warming climate. Finding and implementing these strategies is both a technical and social problem that is perhaps the most difficult challenge that land managers will face in the near term.

Scientists are becoming more aware that active management within reserves or redesign of reserves may be needed to conserve biodiversity in fire-frequent landscapes, where human activities have excluded fire and decreased resilience of forests to fire, insects, disease, and drought. Invasive species such as the barred owl and the sudden oak death pathogen are also motivators for interventions within reserves. Many studies suggest that conservation strategies (and reserve design) should periodically be reevaluated to determine how well they are meeting original and any new goals, and to make possible changes to standards and guidelines and reserve or habitat conservation area boundaries. This may include expanding reserves, increasing connectivity of reserves, shifting locations of reserves (e.g., for small reserves), or using dynamic landscape approaches based on historical disturbance regimes to guide management. Ideally, meeting ecosystem goals for reserves would require areas that are large enough to support fire and other key natural disturbance processes. Meeting both fine- and coarse-filter objectives in these dry forests requires landscape-scale approaches that can integrate potentially competing ecological goals over large areas and long time frames. Using disturbance-based management approaches to conservation is likely to require robust social engagement to increase transparency, public understanding, and trust in managers.

Social-ecological interactions—
For much of the 20th century, timber production was the central way in which federal forests in the NWFP area contributed to community socioeconomic well-being. Although timber production remains important today in some Plan-area communities, the economies of many communities have shifted or diversified their focus over the past two decades. Rural communities are not all alike, forest management policies affect different communities differently, and the social and economic bases of many traditionally forest-dependent communities have changed. Better understanding and consideration of the economic development trajectories of different communities will help to identify forest management activities that best contribute to their well-being. Providing for a diverse set of community benefits from public lands may be the best way to support communities in their efforts to diversify economically, and contribute to building community resilience to future changes in federal forest management and policy.

The forests of the NWFP area provide many ecosystems services to people of the region, in addition to wood. Carbon sequestration, water supply, and recreation are among some of the most valuable of these services. Several policies (table 12-2) direct the agency to use ecosystem management frameworks in planning. However, efforts to quantify and communicate ecosystem services and characterize the associated tradeoffs have yet to be applied in forest plan revision, and there is much to be learned about the most effective ways to use ecosystem services at project and forest scales, though some examples are beginning to appear.

The ability to sustain ecosystem services, conserve species, and promote ecosystem resilience to climate change and fire is highly dependent on socioeconomic factors. Declines in wood processing infrastructure throughout the Plan area have made vegetation management less economical and thus created a financial barrier to fully accomplishing forest restoration. With declining agency capacity, it will be difficult to impossible to maximize all of these objectives, and prioritization likely would be necessary for making progress or goals. Nongovernmental organizations (NGOs) and other government agencies may help managers meet their social and ecological goals. As outlined in chapter 11, an emphasis on engaging with tribes to promote tribal ecocultural resources, in part as a means of upholding the federal trust responsibility, would likely also align with other objectives for ecological restoration, while also providing additional tools and resources for accomplishing those objectives. Approaches such as disturbance-based management or "ecological forestry" may provide a way for federal forests to contribute to local timber-based economies, while providing early-successional habitat and vegetation dependent on fire that has been excluded by fire suppression to meet other management objectives.

Collaborative groups may be part of the solution to increasing trust and social license for forest management. However, collaborative processes are a relatively recent phenomenon and continued learning and adaptive management will be needed to determine the best way forward into an uncertain future. In addition, efforts to collaborate with neighboring landowners in planning and implementing management activities for landscape-level treatments can contribute to increasing forest resilience to climate change, invasive species, and wildfire, and to provide desired ecosystem services (e.g., owl and fish habitat) in mixed-ownership landscapes. Any strategies to promote resilience will need to recognize complex ecological and social system dynamics operating across land ownerships, as well as tensions that arise among competing goals, by adopting long-term and landscape-scale perspectives that include transparent accountability for all involved.

Major disturbances such as large wildfires can promote desired conditions and reestablish key ecosystem processes and species over larger areas of land than can be accomplished through prescribed fire or mechanical treatments. Institutional and social systems may need to evolve to take advantage of such opportunities; for example, by designing postfire management interventions based upon long-term restoration goals as well as more short-term considerations such as safety and timber salvage. Institutional capacity to take advantage of these opportunities is severely limited by an agency-wide decline in staffing, a decades-long history of budget cuts in non-wildfire areas, limited or absent infrastructure for wood processing of forest products, and monetary resource shifts toward fighting wildfires rather than restoring forests. Currently, nearly 55 to 60 percent of the total Forest Service budget each year goes to fighting fires, up from 17 percent 25 years ago.

The challenges ahead for public lands may well require new staffing and partnerships to get work done and new approaches to the problem of restoration. For example, managing natural ignitions for resource benefit may be a particularly cost-effective means of treating landscapes, but prior, large-scale, and widespread fire use planning is likely needed to make these methods effective.

Nevertheless, these opportunities for managing wildfire for resource benefit will pose difficult challenges for managers. Careful assessment of risk to life and property is paramount.

Tradeoffs associated with management—
All management choices involve some social and ecological tradeoffs among the goals of the NWFP. For example,

1. Variable-density thinning can accelerate the development of large live trees and habitat diversity that will benefit northern spotted owls and other species in the future, and produce wood products for the market. However, within the range of the murrelet, these actions may have a short-term negative impact on habitat quality, by creating diverse understory species that benefit murrelet predators, and can reduce amounts of dead wood that are habitat of other species.
2. Thinning and restoring fire to fire-dependent forests will increase habitat for species that use more open older forests and increase forest resilience to fire and drought while creating restoration jobs and reducing wildfire risk in the wildland-urban interface, but these actions can reduce habitat quality for species that use dense older forests.
3. Maintaining road systems to conduct landscape-scale restoration and support recreation will negatively affect some species and ecosystem processes. Many of the potential negative impacts can be ameliorated through landscape-scale planning and using best practices for decisionmaking.

In the long run, thinning in plantations less than 80 years old in LSRs to promote old-growth forest development will not sustain wood production for local communities (chapter 8). Future wood production depends on management in the matrix, where the NWFP allows timber harvest even from older forests. There is no new science that specifically indicates that timber management using retention silviculture in forests over 80 years old in the matrix is inconsistent with the original goals of the NWFP. In addition, partial stand-replacement fires were part of the historical dynamics of some older forests of the moist zone, and the ecological effects of excluding this type of disturbance are not well understood but might convey some resilience to climate and future fire. Given the social pressure to avoid logging of older trees, management in existing plantations for wood in the matrix would appear to be the most socially acceptable way to provide economic returns to support local communities while promoting biodiversity associated with early-seral ecosystems. In addition, it will be valuable to demonstrate how other ecosystems services (e.g., water, recreation) contribute to the mix of values of federal forests, and how effectively active management can meet ecological and social goals.

Monitoring and adaptive management—
The long-term NWFP monitoring program and complementary research efforts of countless agency, university, tribal, and NGO scientists have provided managers, researchers, and stakeholders with an enormous amount of information on how species, ecosystems, and social systems in the NWFP area interact, and have changed over the past 23 years. There will be a need for sustained technical and scientific capacity in the management agencies to keep up with and help translate the large volumes of rapidly expanding scientific knowledge and tools into guidance for planning and management. However, the capacity of agencies to generate new knowledge has precipitously declined, threatening their ability to sustain the flow of information that can lead to more effective management and policies. Scientific uncertainties and debates will continue. Although they may be frustrating to managers, scientists, and the public, the debates also spur research that can lead to new understanding and discovery of knowledge that challenges assumptions, and improve our ability to set and meet attainable goals for forests and aquatic and riparian ecosystems. Further, areas of scientific uncertainty, highlighted by risk analysis, can be clearly articulated to managers and decisionmakers who engage in risk management. Development, evaluation, and testing of new, highly integrated conservation strategies are encouraged to deal with changing knowledge, new perspectives on fire regimes, climate change, invasive species, and recognition of tradeoffs in pursuing biodiversity goals (e.g., coarse filter and fine filter), and other ecological and social dimensions of forest ecosystem management. These forest and social systems will undoubtedly change in the next 23 years. Continuation of monitoring, research, public engagement, and adaptive management will help managers and society adapt to these changes and to meet old and new goals.

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Scientific and common names of plant species identified in this report

Scientific name	Common name
Abies amabilis (Douglas ex Loudon) Douglas ex Forbes	Pacific silver fir
Abies concolor (Gord. & Glend.) Lindl. ex Hildebr.	White fir
Abies grandis (Douglas ex D. Don) Lindl.	Grand fir
Abies lasiocarpa (Hook.) Nutt.	Subalpine pine
Abies magnifica A. Murray bis	California red fir
Abies procera Rehder	Noble fir
Acer circinatum Pursh	Vine maple
Acer macrophyllum Pursh	Bigleaf maple
Achlys triphylla (Sm.) DC.	Sweet after death
Adenocaulon bicolor Hook.	American trailplant
Alliaria petiolata (M. Bieb.) Cavara & Grande	Garlic mustard
Alnus rubra Bong.	Red alder
Amelanchier alnifolia (Nutt.) Nutt. ex M. Roem.	Saskatoon serviceberry
Anemone oregana A. Gray	Blue windflower
Apocynum cannabinum L.	Dogbane
Arbutus menziesii Pursh)	Madrone
Arceuthobium M. Bieb.	Dwarf mistletoe
Arceuthobium occidentale Engelm.	Gray pine dwarf mistletoe
Arceuthobium tsugense Rosendahl	Hemlock dwarf mistletoe
Arctostaphylos nevadensis A. Gray	Pinemat manzanita
Brachypodium sylvaticum (Huds.) P. Beauv.	False brome
Brodiaea coronaria (Salisb.) Engl.	Cluster-lilies
Callitropsis nootkatensis (D. Don) Oerst. ex D.P. Little	Alaska yellow-cedar
Calocedrus decurrens (Torr.) Florin	Incense cedar
Cannabis L.	Marijuana
Carex barbarae Dewey and *C. obnupta* L.H. Bailey	Sedges
Centaurea solstitialis L.	Yellow starthistle
Chamaecyparis lawsoniana (A. Murray bis) Parl.	Port Orford cedar
Chimaphila menziesii (R. Br. ex D. Don) Spreng.	Little prince's pine
Chimaphila umbellata (L.) W.P.C. Barton	Pipsissewa
Clematis vitalba L.	Old man's beard
Clintonia uniflora Menzies ex Schult. & Schult. f.) Kunth	Bride's bonnet
Coptis laciniata A. Gray	Oregon goldthread
Corylus cornuta Marshall var. *californica* (A. DC.) Sharp	California hazel
Cornus canadensis L.	Bunchberry dogwood
Cytisus scoparius (L.) Link	Scotch broom
Disporum hookeri (Torr.) G. Nicholson var. *hookeri*	Drops-of-gold
Fallopia japonica (Houtt.) Ronse Decr. var. *japonica*	Japanese knotweed
Gaultheria ovatifolia A. Gray	Western teaberry
Gaultheria shallon Pursh	Salal

Scientific name	Common name
Gentiana douglasiana Bong.	Swamp gentian
Geranium lucidum L.	Shining geranium
Geranium robertianum L.	Robert geranium
Goodyera oblongifolia Raf.	Western rattlesnake plantain
Hedera helix L.	English ivy
Heracleum mantegazzianum Sommier & Levier	Giant hogweed
Hesperocyparis sargentii (Jeps.) Bartel	Sargent's cypress
Hieracium aurantiacum L.	Orange hawkweed
Ilex aquifolium L.	English holly
Iris pseudacorus L.	Paleyellow iris
Juniperus occidentalis Hook.	Western juniper
Lamiastrum galeobdolon (L.) Ehrend. & Polatschek	Yellow archangel
Lilium occidentale Purdy	Western lily
Linnaea borealis L.	Twinflower
Lithocarpus densiflorus (Hook. & Arn.) Rehder	Tanoak
Lonicera hispidula Pursh	Honeysuckle
Lupinus albicaulis Douglas	Sickle-keeled lupine
Lycopodium clavatum L.	Running clubmoss
Lythrum salicaria L.	Purple loosestrife
Mahonia nervosa (Pursh) Nutt.	Cascade barberry
Malus fusca (Raf.) C.K. Schneid.	Pacific crabapple
Notholithocarpus densiflorus (Hook. & Arn.) P.S. Manos, C.H. Cannon, & S.H. Oh	Tanoak
Notholithocarpus densiflorus (Hook. & Arn.) P.S. Manos, C.H. Cannon, & S.H. Oh var. *echinoides* (R.Br. ter) P.S. Manos, C.H. Cannon & S.H. Oh	Shrub form of tanoak
Nuphar polysepala (Engelm.)	Yellow pond lily
Nymphoides peltata (S.G. Gmel.) Kuntze	Yellow floating heart
Osmorhiza chilensis Hook. & Arn.	Sweetcicely
Phalaris arundinacea L.	Reed canarygrass
Picea engelmannii Parry ex Engelm.	Engelmann spruce
Picea sitchensis (Bong.) Carrière	Sitka spruce
Pinus albicaulis Engelm.	Whitebark pine
Pinus attenuata Lemmon	Knobcone pine
Pinus contorta Douglas ex Loudon	Lodgepole pine
Pinus contorta Douglas ex Loudon var. *contorta*	Beach pine, shore pine
Pinus jeffreyi Balf.	Jeffrey pine
Pinus lambertiana Douglas	Sugar pine
Pinus monticola Douglas ex D. Don)	Western white pine
Pinus ponderosa Lawson & C. Lawson	Ponderosa pine
Populus trichocarpa L. ssp. *trichocarpa* (Torr. & A. Gray ex Hook) Brayshaw	Black cottonwood
Potamogeton crispus L.	Curly pondweed
Potentilla recta L.	Sulphur cinquefoil

Scientific name	Common name
Prunus emarginata (Douglas ex Hook. D. Dietr.)	Bitter cherry
Pseudotsuga menziesii (Mirb.) Franco	Douglas-fir
Pteridium aquilinum (L. Kuhn)	Brackenfern
Pueraria montana (Lour.) Merr. var. *lobata* (Willd.) Maesen & S.M. Almeida ex Sanjappa & Predeep	Kudzu
Pyrola asarifolia Sweet	American wintergreen
Quercus agrifolia Née var. *oxyadenia* (Torr.) J.T. Howell	Coastal live oak
Quercus berberidifolia Liebm.	Scrub oak
Quercus chrysolepis Liebm.	Canyon live oak
Quercus douglasii Hook. & Arn.	Blue oak
Quercus garryana Douglas ex hook.	Oregon white oak
Quercus kelloggi Newberry	California black oak
Quercus lobata Née	Valley oak
Rhamnus purshiana (DC.) A. Gray	Cascara
Rhododendron groenlandicum Oeder	Bog Labrador tea
Rhododendron macrophyllum D. Don ex G. Don	Pacific rhododendron
Ribes lacustre (Pers.) Poir.	Prickly currant
Rubus armeniacus Focke	Himalayan blackberry
Salix exigua Nutt.	Sandbar willow
Senecio bolanderi A. Gray	Bolander's ragwort
Sequoia sempervirens (Lamb. ex D. Don) Endl.	Redwood
Smilacina stellata (L.) Desf.	Starry false Solomon's seal
Synthyris reniformis (Douglas ex Benth.) Benth.	Snowqueen
Taxus brevifolia Nutt.	Pacific yew
Thuja plicata Donn ex D. Don	Western redcedar
Tiarella trifoliate L.	Threeleaf foamflower
Trapa natans L.	Water chestnut
Trillium ovatum Pursh	Pacific trillium
Tsuga heterophylla (Raf.) Sarg.	Western hemlock
Tsuga mertensiana (Bong.) Carrière	Mountain hemlock
Typha latifolia L.	Cattails
Umbellularia californica (Hook. & Arn.) Nutt.	California bay laurel
Vaccinium alaskaense Howell	Alaska blueberry
Vaccinium membranaceum Douglas ex Torr.	Thinleaf huckleberry, big huckleberry
Vaccinium ovatum Pursh	Evergreen huckleberry
Vaccinium oxycoccos L.	Small cranberry
Vaccinium parvifolium Sm.	Red huckleberry
Vancouveria hexandra (Hook.) C. Morren & Decne.	White insideout flower
Xerophyllum tenax (Pursh) Nutt.	Beargrass

Glossary

This glossary is provided to help readers understand various terms used in the Northwest Forest Plan (NWFP) science synthesis. Sources include the Forest Service Handbook (FSH), the Code of Federal Regulations (CFR), executive orders, the Federal Register (FR), and various scientific publications (see "Glossary Literature Cited"). The authors have added working definitions of terms used in the synthesis and its source materials, especially when formal definitions may be lacking or when they differ across sources.

active management—Direct interventions to achieve desired outcomes, which may include harvesting and planting of vegetation and the intentional use of fire, among other activities (Carey 2003).

adaptive capacity—The ability of ecosystems and social systems to respond to, cope with, or adapt to disturbances and stressors, including environmental change, to maintain options for future generations (FSH 1909.12.5).

adaptive management—A structured, cyclical process for planning and decisionmaking in the face of uncertainty and changing conditions with feedback from monitoring, which includes using the planning process to actively test assumptions, track relevant conditions over time, and measure management effectiveness (FSH 1909.12.5). Additionally, adaptive management includes iterative decisionmaking, through which results are evaluated and actions are adjusted based on what has been learned.

adaptive management area (AMA)—A portion of the federal land area within the NWFP area that was specifically allocated for scientific monitoring and research to explore new forestry methods and other activities related to meeting the goals and objectives of the Plan. Ten AMAs were established in the NWFP area, covering about 1.5 million ac (600 000 ha), or 6 percent of the planning area (Stankey et al. 2003).

alien species—Any species, including its seeds, eggs, spores, or other biological material capable of propagating that species, that is not native to a particular ecosystem (Executive Order 13112). The term is synonymous with exotic species, nonindigenous, and nonnative species (see also "invasive species").

allochthonous inputs—Material, specifically food resources, that originates from outside a stream, typically in the form of leaf litter.

amenity communities—Communities located near lands with high amenity values.

amenity migration—Movement of people based on the draw of natural or cultural amenities (Gosnell and Abrams 2011).

amenity value—A noncommodity or "unpriced" value of a place or environment, typically encompassing aesthetic, social, cultural, and recreational values.

ancestral lands (of American Indian tribes)—Lands that historically were inhabited by the ancestors of American Indian tribes.

annual species review—A procedure established under the NWFP in which panels of managers and biologists evaluate new scientific and monitoring information on species to potentially support the recommendation of changes in their conservation status.

Anthropocene—The current period (or geological epoch) in which humans have become a dominant influence on the Earth's climate and environment, generally dating from the period of rapid growth in industrialization, population, and global trade and transportation in the early 1800s (Steffen et al. 2007).

Aquatic Conservation Strategy (ACS) —A regional strategy applied to aquatic and riparian ecosystems across the area covered by the NWFP) (Espy and Babbit 1994) (see chapter 7 for more details).

at-risk species—Federally recognized threatened, endangered, proposed, and candidate species and species of conservation concern. These species are considered at risk of low viability as a result of changing environmental conditions or human-caused stressors.

best management practices (BMPs) (for water quality)—Methods, measures, or practices used to reduce or eliminate the introduction of pollutants and other detrimental impacts to water quality, including but not limited to structural and nonstructural controls and to operation and maintenance procedures.

biodiversity—In general, the variety of life forms and their processes and ecological functions, at all levels of biological organization from genes to populations, species, assemblages, communities, and ecosystems.

breeding inhibition—Prevention of reproduction in healthy adult individuals.

bryophytes—Mosses and liverworts.

canopy cover—The downward vertical projection from the outside profile of the canopy (crown) of a plant measured in percentage of land area covered.

carrying capacity—The maximum population size a specific environment can sustain.

ceded areas—Lands that particular tribes ceded to the United States government by treaties, which have been catalogued in the Library of Congress.

climate adaptation—Management actions to reduce vulnerabilities to climate change and related disturbances.

climate change—Changes in average weather conditions (including temperature, precipitation, and risk of certain types of severe weather events) that persist over multiple decades or longer, and that result from both natural factors and human activities such as increased emissions of greenhouse gases (U.S. Global Change Research Program 2017).

coarse filter—A conservation approach that focuses on conserving ecosystems, in contrast to a "fine filter" approach that focuses on conserving specific species. These two approaches are generally viewed as complementary, with fine-filtered strategies tailored to fit particular species that "fall through the pores" of the coarse filter (Hunter 2005). See also "mesofilter."

co-management—Two or more entities, each having legally established management responsibilities, working collaboratively to achieve mutually agreed upon, compatible objectives to protect, conserve, use, enhance, or restore natural and cultural resources (81 FR 4638).

collaborative management—Two or more entities working together to actively protect, conserve, use, enhance, or restore natural and cultural resources (81 FR 4638).

collaboration or collaborative process—A structured manner in which a collection of people with diverse interests share knowledge, ideas, and resources, while working together in an inclusive and cooperative manner toward a common purpose (FSH 1909.12.05).

community (plant and animal)—A naturally occurring assemblage of plant and animal species living within a defined area or habitat (36 CFR 219.19).

community forest—A general definition is forest land that is managed by local communities to provide local benefits (Teitelbaum et al. 2006). The federal government has specifically defined community forest as "forest land owned in fee simple by an eligible entity [local government, nonprofit organization, or federally recognized tribe] that provides public access and is managed to provide community benefits pursuant to a community forest plan" (36 CFR 230.2).

community of place or place-based community—A group of people who are bound together because of where they reside, work, visit, or otherwise spend a continuous portion of their time.

community resilience—The capacity of a community to return to its initial function and structure when initially altered under disturbance.

community resistance—The capacity of a community to withstand a disturbance without changing its function and structure.

composition—The biological elements within the various levels of biological organization, from genes and species to communities and ecosystems (FSM 2020).

congeneric—Organisms that belong to the same taxonomic genus, usually belonging to different species.

connectivity (of habitats)—Environmental conditions that exist at several spatial and temporal scales that provide landscape linkages that permit (a) the exchange of flow, sediments, and nutrients; (b) genetic interchange of genes among individuals between populations; and (c) the long-distance range shifts of species, such as in response to climate change (36 CFR 219.19).

consultation (tribal)—A formal government-to-government process that enables American Indian tribes and Alaska Native Corporations to provide meaningful, timely input, and, as appropriate, exchange views, information, and recommendations on proposed policies or actions that may affect their rights or interests prior to a decision. Consultation is a unique form of communication characterized by trust and respect (FSM 1509.05).

corticosterone—A steroid hormone produced by many species of animals, often as the result of stress.

cryptogam—An organism that reproduces by spores and that does not produce true flowers and seeds; includes fungi, algae, lichens, mosses, liverworts, and ferns.

cultural keystone species—A species that significantly shapes the cultural identity of a people, as reflected in diet, materials, medicine, or spiritual practice (Garibaldi and Turner 2004).

cultural services—A type of ecosystem service that includes the nonmaterial benefits that people obtain from ecosystems through spiritual enrichment, cognitive development, reflection, recreation, and aesthetic experiences (Sarukhán and Whyte 2005).

desired conditions—A description of specific social, economic, or ecological characteristics toward which management of the land and resources should be directed.

disturbance regime—A description of the characteristic types of disturbance on a given landscape; the frequency, severity, and size distribution of these characteristic disturbance types and their interactions (36 CFR 219.19).

disturbance—Any relatively discrete event in time that disrupts ecosystem, watershed, community, or species population structure or function, and that changes resources, substrate availability, or the physical environment (36 CFR 219.19).

dynamic reserves—A conservation approach in which protected areas are relocated following changes in environmental conditions, especially owing to disturbance.

early-seral vegetation—Vegetation conditions in the early stages of succession following an event that removes the forest canopy (e.g., timber harvest, wildfire, windstorm), on sites that are capable of developing a closed canopy (Swanson et al. 2014). A nonforest or "pre-forest" condition occurs first, followed by an "early-seral forest" as young shade-intolerant trees form a closed canopy.

ecocultural resources—Valued elements of the biophysical environment, including plants, fungi, wildlife, water, and places, and the social and cultural relationships of people with those elements.

ecological conditions—The biological and physical environment that can affect the diversity of plant and animal communities, the persistence of native species, invasibility, and productive capacity of ecological systems. Ecological conditions include habitat and other influences on species and the environment. Examples of ecological conditions include the abundance and distribution of aquatic and terrestrial habitats, connectivity, roads and other structural developments, human uses, and occurrence of other species (36 CFR 219.19).

ecological forestry—A ecosystem management approach designed to achieve multiple objectives that may include conservation goals and sustainable forest management and which emphasizes disturbance-based management and retention of "legacy" elements such as old trees and dead wood (Franklin et al. 2007).

ecological integrity—The quality or condition of an ecosystem when its dominant ecological characteristics (e.g., 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 (36 CFR 219.19).

ecological keystone species—A species whose ecological functions have extensive and disproportionately large effects on ecosystems relative to its abundance (Power et al. 1996).

ecological sustainability—The capability of ecosystems to maintain ecological integrity (36 CFR 219.19).

economic sustainability—The capability of society to produce and consume or otherwise benefit from goods and services, including contributions to jobs and market and nonmarket benefits (36 CFR 219.19).

ecoregion—A geographic area containing distinctive ecological assemblages, topographic and climatic gradients, and historical land uses.

ecosystem—A spatially explicit, relatively homogeneous unit of the Earth that includes all interacting organisms and elements of the abiotic environment within its boundaries (36 CFR 219.19).

ecosystem diversity—The variety and relative extent of ecosystems (36 CFR 219.19).

ecosystem integrity—See "ecological integrity."

ecosystem management—Management across broad spatial and long temporal scales for a suite of goals, including maintaining populations of multiple species and ecosystem services.

ecosystem services—Benefits that people obtain from ecosystems (see also "provisioning services," "regulating services," "supporting services," and "cultural services").

ectomycorrhizal fungi—Fungal species that form symbiotic relationships with vascular plants through roots, typically aiding their uptake of nutrients. Although other mycorrhizal fungi penetrate their host's cell walls, ectomycorrhizal fungi do not.

endangered species—Any species or subspecies that the Secretary of the Interior or the Secretary of Commerce has deemed in danger of extinction throughout all or a significant portion of its range (16 U.S.C. Section 1532).

endemic—Native and restricted to a specific geographical area.

El Niño Southern Oscillation (ENSO)—A band of anomalously warm ocean water temperatures that occasionally develops off the western coast of South America and can cause climatic changes across the Pacific Ocean. The extremes of this climate pattern's oscillations cause extreme weather (such as floods and droughts) in many regions of the world.

environmental DNA (eDNA)—Genetic material (DNA) contained within small biological and tissue fragments that can be collected from aquatic, terrestrial, and even atmospheric environments, linked to an individual species, and used to indicate the presence of that species.

environmental justice populations—Groups of people who have low incomes or who identify themselves as African American, Asian or Pacific Islander, American Indian or Alaskan Native, or of Hispanic origin.

ephemeral stream—A stream that flows only in direct response to precipitation in the immediate locality (watershed or catchment basin), and whose channel is at all other times above the zone of saturation.

epicormic—Literally, "of a shoot or branch," this term implies growth from a previously dormant bud on the trunk or a limb of a tree.

epiphyte—A plant or plant ally (including mosses and lichens) that grows on the surface of another plant such as a tree, but is not a parasite.

even-aged stand—A stand of trees composed of a single age class (36 CFR 219.19).

fecundity—The reproductive rate of an organism or population.

federally recognized Indian tribe—An Indian tribe or Alaska Native Corporation, band, nation, pueblo, village, or community that the Secretary of the Interior acknowledges

to exist as an Indian tribe under the Federally Recognized Indian Tribe List Act of 1994, 25 U.S.C. 479a (36 CFR 219.19).

fine filter—A conservation approach that focuses on conserving individual species in contrast to a "coarse filter" approach that focuses on conserving ecosystems; these approaches are generally viewed as complementary with fine-filtered strategies tailored to fit particular species that "fall through the pores" of the coarse filter (Hunter 2005). See also "mesofilter."

fire-dependent vegetation types—A vegetative community that evolved with fire as a necessary contributor to its vitality and to the renewal of habitat for its member species.

fire exclusion—Curtailment of wildland fire because of deliberate suppression of ignitions, as well as unintentional effects of human activities such as intensive grazing that removes grasses and other fuels that carry fire (Keane et al. 2002).

fire intensity—The amount of energy or heat release during fire.

fire regime—A characterization of long-term patterns of fire in a given ecosystem over a specified and relatively long period of time, based on multiple attributes, including frequency, severity, extent, spatial complexity, and seasonality of fire occurrence.

fire regime, low frequency, high severity—A fire regime with long return intervals (>200 years) and high levels of vegetation mortality (e.g., ~70 percent basal area mortality in forested ecosystems), often occurring in large patches (>10,000 ac [4047 ha]) (see chapter 3 for more details).

fire regime, moderate frequency, mixed severity—A fire regime with moderate return intervals between 50 and 200 years and mixtures of low, moderate, and high severity; high-severity patches would have been common and frequently large (>1,000 ac [>405 ha]) (see chapter 3 for more details).

fire regime, very frequent, low severity—A fire regime with short return intervals (5 to 25 years) dominated by surface fires that result in low levels of vegetation mortality (e.g., <20 percent basal area mortality in forested ecosystems), with high-severity fire generally limited to small patches (<2.5 ac [1 ha]) (see chapter 3 for more details).

fire regime, frequent, mixed severity—A fire regime with return intervals between 15 and 50 years that burns with a mosaic of low-, moderate-, and high-severity patches (Perry et al. 2011) (see chapter 3 for more details).

fire rotation—Length of time expected for a specific amount of land to burn (some parts might burn more than once or some not at all) based upon the study of past fire records in a large landscape (Turner and Romme 1994).

fire severity—The magnitude of the effects of fire on ecosystem components, including vegetation or soils.

fire suppression—The human act of extinguishing wildfires (Keane et al. 2002).

floodplain restoration—Ecological restoration of a stream or river's floodplain, which may involve setback or removal of levees or other structural constraints.

focal species—A small set of species whose status is assumed to infer the integrity of the larger ecological system to which it belongs, and thus to provide meaningful information regarding the effectiveness of a resource management plan in maintaining or restoring the ecological conditions to maintain the broader diversity of plant and animal communities in the NWPF area. Focal species would be commonly selected on the basis of their functional role in ecosystems (36 CFR 219.19).

food web—Interconnecting chains between organisms in an ecological community based upon what they consume.

Forest Ecosystem Management Assessment Team (FEMAT)—An interdisciplinary team that included expert ecological and social scientists, analysts, and managers assembled in 1993 by President Bill Clinton to develop options for ecosystem management of federal forests within the range of the northern spotted owl (FEMAT 1993).

forest fragmentation—The patterns of dispersion and connectivity of nonhomogeneous forest cover (Riitters et al. 2002). See also "landscape fragmentation" and "habitat fragmentation" for specific meanings related to habitat loss and isolation.

frequency distribution—A depiction, often appearing in the form of a curve or graph, of the abundance of possible values of a variable. In this synthesis report, we speak of the frequency of wildfire patches of various sizes.

fuels (wildland)—Combustible material in wildland areas, including live and dead plant biomass such as trees, shrub, grass, leaves, litter, snags, and logs.

fuels management—Manipulation of wildland fuels through mechanical, chemical, biological, or manual means, or by fire, in support of land management objectives to control or mitigate the effects of future wildland fire.

function (ecological)—Ecological processes, such as energy flow; nutrient cycling and retention; soil development and retention; predation and herbivory; and natural disturbances such as wind, fire, and floods that sustain composition and structure (FSM 2020). See also "key ecological function."

future range of variation (FRV)—The natural fluctuation of pattern components of healthy ecosystems that might occur in the future, primarily affected by climate change, human infrastructure, invasive species, and other anticipated disturbances.

gaps (forest)—Small openings in a forest canopy that are naturally formed when one or a few canopy trees die (Yamamoto 2000).

genotype—The genetic makeup of an individual organism.

glucocorticoid—A class of steroid hormones produced by many species of animals, often as the result of stress.

goals (in land management plans)—Broad statements of intent, other than desired conditions, that do not include expected completion dates (36 CFR part 219.7(e)(2)).

guideline—A constraint on project and activity decision-making that allows for departure from its terms, so long as the purpose of the guideline is met (36 CFR section 219.15(d)(3)). Guidelines are established to help achieve or maintain a desired condition or conditions, to avoid or mitigate undesirable effects, or to meet applicable legal requirements.

habitat—An area with the environmental conditions and resources that are necessary for occupancy by a species and for individuals of that species to survive and reproduce.

habitat fragmentation—Discontinuity in the spatial distribution of resources and conditions present in an area at a given scale that affects occupancy, reproduction, and survival in a particular species (see "landscape fragmentation").

heterogeneity (forest)—Diversity, often applied to variation in forest structure within stands in two dimensions: horizontal (e.g., single trees, clumps of trees, and gaps of no trees), and vertical (e.g., vegetation at different heights from the forest floor to the top of the forest canopy), or across large landscapes (North et al. 2009).

hierarchy theory—A theory that describes ecosystems at multiple levels of organization (e.g., organisms, populations, and communities) in a nested hierarchy.

high-severity burn patch—A contiguous area of high-severity or stand-replacing fire.

historical range of variation (HRV)—Past fluctuation or range of conditions in the pattern of components of ecosystems over a specified period of time.

hybrid ecosystem—An ecosystem that has been modified from a historical state such that it has novel attributes while retaining some original characteristics (see "novel ecosystem").

hybrid—Offspring resulting from the breeding of two different species.

inbreeding depression—Reduced fitness in a population that occurs as the result of breeding between related individuals, leading to increased homogeneity and simplification of the gene pool.

in-channel restoration—Ecological restoration of the channel of a stream or river, often through placement of materials (rocks and wood) or other structural modifications.

individuals, clumps, and openings (ICO) method—A method that incorporates reference spatial pattern targets based upon individual trees, clumps of trees, and canopy openings into silvicultural prescriptions and tree-marking guidelines (Churchill et al. 2013).

Interagency Special Status and Sensitive Species Program (ISSSSP)—A federal agency program, established under the U.S. Forest Service Pacific Northwest Region and Bureau of Land Management Oregon/Washington state office. The ISSSSP superseded the Survey and Manage standards and guidelines under the NWFP and also addresses other species of conservation focus, coordinates development and revision of management recommendations and survey protocols, coordinates data management between the agencies, develops summaries of species biology, and conducts other tasks.

intermittent stream—A stream or reach of stream channel that flows, in its natural condition, only during certain times of the year or in several years, and is characterized by interspersed, permanent surface water areas containing aquatic flora and fauna adapted to the relatively harsh environmental conditions found in these types of environments.

invasive species—An alien species (or subspecies) whose deliberate, accidental, or self-introduction is likely to cause economic or environmental harm or harm to human health (Executive Order 13112).

key ecological function—The main behaviors performed by an organism that can influence environmental conditions or habitats of other species.

key watersheds—Watersheds that are expected to serve as refugia for aquatic organisms, particularly in the short term, for at-risk fish populations that have the greatest potential for restoration, or to provide sources of high-quality water.

land and resource management plan (Forest Service)—A document or set of documents that provides management direction for an administrative unit of the National Forest System (FSH 1909.12.5).

landform—A specific geomorphic feature on the surface of the Earth, such as a mountain, plateau, canyon, or valley.

landscape—A defined area irrespective of ownership or other artificial boundaries, such as a spatial mosaic of terrestrial and aquatic ecosystems, landforms, and plant communities, repeated in similar form throughout such a defined area (36 CFR 219.19).

landscape fragmentation—Breaking up of continuous habitats into patches as a result of human land use and thereby generating habitat loss, isolation, and edge effects (see “habitat fragmentation”).

landscape genetics—An interdisciplinary field of study that combines population genetics and landscape ecology to explore how genetic relatedness among individuals and subpopulations of a species is influenced by landscape-level conditions.

landscape hierarchy—Organization of land areas based upon a hierarchy of nested geographic (i.e., different-sized) units, which provides a guide for defining the functional components of a system and how components at different scales are related to one another.

late-successional forest—Forests that have developed after long periods of time (typically at least 100 to 200 years) following major disturbances, and that contain a major component of shade-tolerant tree species that can regenerate beneath a canopy and eventually grow into the canopy in which small canopy gaps occur (see chapter 3 for more details). Note that FEMAT (1993) and the NWFP also applied this term to older (at least 80 years) forest types, including both old-growth and mature forests, regardless of the shade tolerance of the dominant tree species (e.g., 90-year-old forests dominated by Douglas-fir were termed late successional).

leading edge—The boundary of a species’ range at which the population is geographically expanding through colonization of new sites.

legacy trees—Individual trees that survive a major disturbance and persist as components of early-seral stands (Franklin 1990).

legacies (biological)—Live trees, seed and seedling banks, remnant populations and individuals, snags, large soil aggregates, hyphal mats, logs, uprooted trees, and other biotic features that survive a major disturbance and persist as components of early-seral stands (Franklin 1990, Franklin et al. 2002).

lentic—Still-water environments, including lakes, ponds, and wet meadows.

longitudinal studies—Studies that include repeated observations on the same response variable over time.

lotic—Freshwater environments with running water, including rivers, streams, and springs.

low-income population—A community or a group of individuals living in geographic proximity to one another, or a set of individuals, such as migrant workers or American Indians, who meet the standards for low income and experience common conditions of environmental exposure or effect (CEQ 1997).

managing wildfire for resource objectives—Managing wildfires to promote multiple objectives such as reducing fire danger or restoring forest health and ecological processes rather than attempting full suppression. The terms "managed wildfire" or "resource objective wildfire" have also been used to describe such events (Long et al. 2017). However, fire managers note that many unplanned ignitions are managed using a combination of tactics, including direct suppression, indirect containment, monitoring of fire spread, and even accelerating fire spread, across their perimeters and over their full duration. Therefore, terms that separate "managed" wildfires from fully "suppressed" wildfires do not convey that complexity. (See "Use of wildland fire," which also includes prescribed burning).

matrix—Federal and other lands outside of specifically designated reserve areas, particularly the late-successional reserves under the NWFP, that are managed for timber production and other objectives.

mature forest—An older forest stage (>80 years) prior to old-growth in which trees begin attaining maximum heights and developing some characteristic, for example, 80 to 200 years in the case of old-growth Douglas-fir/western hemlock forests, often (but not always) including big trees (>50 cm diameter at breast height), establishment of late-seral species (i.e., shade-tolerant trees), and initiation of decadence in early species (i.e., shade-intolerant trees).

mesofilter—A conservation approach that "focuses on conserving critical elements of ecosystems that are important to many species, especially those likely to be overlooked by fine-filter approaches, such as invertebrates, fungi, and nonvascular plants" (Hunter 2005).

meta-analysis—A study that combines the results of multiple studies**.**

minority population—A readily identifiable group of people living in geographic proximity with a population that is at least 50 percent minority; or, an identifiable group that has a meaningfully greater minority population than the adjacent geographic areas, or may also be a geographically dispersed/transient set of individuals such as migrant workers or Americans Indians (CEQ 1997).

mitigation (climate change)—Efforts to reduce anthropogenic alteration of climate, in particular by increasing carbon sequestration.

monitoring—A systematic process of collecting information to track implementation (implementation monitoring), to evaluate effects of actions or changes in conditions or relationships (effectiveness monitoring), or to test underlying assumptions (validation monitoring) (see 36 CFR 219.19).

mosaic—The contiguous spatial arrangement of elements within an area. In regions, this is typically the upland vegetation patches, large urban areas, large bodies of water, and large areas of barren ground or rock. However, regional mosaics can also be described in terms of land ownership, habitat

patches, land use patches, or other elements. For landscapes, this is typically the spatial arrangement of landscape elements.

multiaged stands—Forest stands having two or more age classes of trees; this includes stands resulting from variable-retention silvicultural systems or other traditionally even-aged systems that leave residual or reserve (legacy) trees.

multiple use—The management of all the various renewable surface resources of the National Forest System so that they are used in the combination that will best meet the needs of the American people; making the most judicious use of the land for some or all of these resources or related services over areas large enough to provide sufficient latitude for periodic adjustments in use to conform to changing needs and conditions; that some land will be used for less than all of the resources; and harmonious and coordinated management of the various resources, each with the other, without impairment of the productivity of the land, with consideration being given to the relative values of the various resources, and not necessarily the combination of uses that will give the greatest dollar return or the greatest unit output, consistent with the Multiple-Use Sustained-Yield Act of 1960 (16 U.S.C. 528–531) (36 CFR 219.19).

natal site—Location of birth.

native knowledge—A way of knowing or understanding the world, including traditional ecological, and social knowledge of the environment derived from multiple generations of indigenous peoples' interactions, observations, and experiences with their ecological systems. This knowledge is accumulated over successive generations and is expressed through oral traditions, ceremonies, stories, dances, songs, art, and other means within a cultural context (36 CFR 219.19).

native species—A species historically or currently present in a particular ecosystem as a result of natural migratory or evolutionary processes and not as a result of an accidental or deliberate introduction or invasion into that ecosystem (see 36 CFR 219.19).

natural range of variation (NRV)—The variation of ecological characteristics and processes over specified scales of time and space that are appropriate for a given management application (FSH 1909.12.5).

nested hierarchy—The name given to the hierarchical structure of groups within groups used to classify organisms.

nontimber forest products (also known as "special forest products")—Various products from forests that do not include logs from trees but do include bark, berries, boughs, bryophytes, bulbs, burls, Christmas trees, cones, ferns, firewood, forbs, fungi (including mushrooms), grasses, mosses, nuts, pine straw, roots, sedges, seeds, transplants, tree sap, wildflowers, fence material, mine props, posts and poles, shingle and shake bolts, and rails (36 CFR part 223 Subpart G).

novel ecosystem—An ecosystem that has experienced large and potentially irreversibly modifications to abiotic conditions or biotic composition in ways that result in a composition of species, ecological communities, and functions that have never before existed, and that depart from historical analogs (Hobbs et al. 2009). See "hybrid ecosystem" for comparison.

old-growth forest—A forest distinguished by old trees (>200 years) and related structural attributes that often (but not always) include large trees, high biomass of dead wood (i.e., snags, down coarse wood), multiple canopy layers, distinctive species composition and functions, and vertical and horizontal diversity in the tree canopy (see chapter 3). In dry, fire-frequent forests, old growth is characterized by large, old fire-resistant trees and relatively open stands without canopy layering.

palustrine—Inland, nontidal wetlands that may be permanently or temporarily flooded and are characterized by the presence of emergent vegetation such as swamps, marshes, vernal pools, and lakeshores.

passive management—A management approach in which natural processes are allowed to occur without human intervention to reach desired outcomes.

patch—A relatively small area with similar environmental conditions, such as vegetative structure and composition. Sometimes used interchangeably with vegetation or forest stand.

Pacific Decadal Oscillation (PDO)—A recurring (approximately decadal-scale) pattern of ocean-atmosphere —a stream or reach of a channel that flows continuously or nearly so throughout the year and whose upper surface is generally lower than the top of the zone of saturation in areas adjacent to the stream.

perennial stream—A stream or reach of a channel that flows continuously or nearly so throughout the year and whose upper surface is generally lower than the top of the zone of saturation in areas adjacent to the stream.

phenotype—Physical manifestation of the genetic makeup of an individual and its interaction with the environment.

place attachment—The "positive bond that develops between groups or individuals and their environment" (Jorgensen and Stedman 2001: 234).

place dependence— "The strength of an individual's subjective attachment to specific places" (Stokols and Shumaker 1982: 157).

place identity—Dimensions of self that define an individual's [or group's] identity in relation to the physical environment through ideas, beliefs, preferences, feelings, values, goals, and behavioral tendencies and skills (Proshansky 1978).

place-based planning—"A process used to involve stakeholders by encouraging them to come together to collectively define place meanings and attachments" (Lowery and Morse 2013: 1423).

plant association—A fine level of classification in a hierarchy of potential vegetation that is defined in terms of a climax-dominant overstory tree species and typical understory herb or shrub species.

population bottleneck—An abrupt decline in the size of a population from an event, which often results in deleterious effects such as reduced genetic diversity and increased probability of local or global extirpation.

potential vegetation type (PVT)—Native, late-successional (or "climax") plant community that reflects the regional climate, and dominant plant species that would occur on a site in absence of disturbances (Pfister and Arno 1980).

poverty rate—A measure of financial income below a threshold that differs by family size and composition.

precautionary principle—A principle that if an action, policy, or decision has a suspected risk of causing harm to the public or to the environment, and there is no scientific consensus that it is not harmful, then the burden of proof that it is not harmful falls on those making that decision. Particular definitions of the principle differ, and some applications use the less formal term, "precautionary approach." Important qualifications associated with many definitions include (1) the perceived harm is likely to be serious, (2) some scientific analysis suggests a significant but uncertain potential for harm, and (3) applications of the principle emphasize generally constraining an activity to mitigate it rather than "resisting" it entirely (Doremus 2007).

prescribed fire—A wildland fire originating from a planned ignition to meet specific objectives identified in a written and approved prescribed fire plan for which National Environmental Policy Act requirements (where applicable) have been met prior to ignition (synonymous with controlled burn).

primary recreation activity—A single activity that caused a recreation visit to a national forest.

probable sale quantity—An estimate of the average amount of timber likely to be awarded for sale for a given area (such as the NWFP area) during a specified period.

provisioning services—A type of ecosystem service that includes clean air and fresh water, energy, food, fuel, forage, wood products or fiber, and minerals.

public participation geographic information system (PPGIS)—Using spatial decisionmaking and mapping tools to produce local knowledge with the goal of including and empowering marginalized populations (Brown and Reed 2009).

public values—Amenity values (scenery, quality of life); environmental quality (clean air, soil, and water); ecological

values (biodiversity); public use values (outdoor recreation, education, subsistence use); and spiritual or religious values (cultural ties, tribal history).

record of decision (ROD)—The final decision document that amended the planning documents of 19 national forests and seven Bureau of Land Management districts within the range of the northern spotted owl (the NWFP area) in April 1994 (Espy and Babbit 1994).

recreation opportunity—An opportunity to participate in a specific recreation activity in a particular recreation setting to enjoy desired recreation experiences and other benefits that accrue. Recreation opportunities include non-motorized, motorized, developed, and dispersed recreation on land, water, and in the air (36 CFR 219.19).

redundancy—The presence of multiple occurrences of ecological conditions, including key ecological functions (functional redundancy), such that not all occurrences may be eliminated by a catastrophic event.

refugia—An area that remains less altered by climatic and environmental change (including disturbances such as wind and fire) affecting surrounding regions and that therefore forms a haven for relict fauna and flora.

regalia—Dress and special elements made from a variety of items, including various plant and animal materials, and worn for tribal dances and ceremonies.

regulating services—A type of ecosystem service that includes long-term storage of carbon; climate regulation; water filtration, purification, and storage; soil stabilization; flood and drought control; and disease regulation.

representativeness—The presence of a full array of ecosystem types and successional states, based on the physical environment and characteristic disturbance processes.

reserve—An area of land designated and managed for a special purpose, often to conserve or protect ecosystems, species, or other natural and cultural resources from particular human activities that are detrimental to achieving the goals of the area.

resilience—The capacity of a system to absorb disturbance and reorganize (or return to its previous organization) so as to still retain essentially the same function, structure, identity, and feedbacks (see FSM Chapter 2020 and see also "socioecological resilience"). Definitions emphasize the capacity of a system or its constituent entities to respond or regrow after mortality induced by a disturbance event, although broad definitions of resilience may also encompass "resistance" (see below), under which such mortality may be averted.

resistance—The capacity of a system or an entity to withstand a disturbance event without much change.

restoration economy—Diverse economic activities associated with the restoration of structure or function to terrestrial and aquatic ecosystems (Nielsen-Pincus and Moseley 2013).

restoration, ecological—The process of assisting the recovery of an ecosystem that has been degraded, damaged, or destroyed. 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 (36 CFR 219.19).

restoration, functional—Restoration of dynamic abiotic and biotic processes in degraded ecosystems, without necessarily a focus on structural condition and composition.

riparian areas—Three-dimensional ecotones (the transition zone between two adjoining communities) 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 (36 CFR 219.19).

riparian management zone—Portions of a watershed in which riparian-dependent resources receive primary emphasis, and for which plans include Plan components to maintain or restore riparian functions and ecological functions (36 CFR 219.19).

riparian reserves—Reserves established along streams and rivers to protect riparian ecological functions and processes

necessary to create and maintain habitat for aquatic and riparian-dependent organisms over time and ensure connectivity within and between watersheds. The Aquatic Conservation Strategy in the NWFP record of decision included standards and guidelines that delineated riparian reserves.

risk—A combination of the probability that a negative outcome will occur and the severity of the subsequent negative consequences (36 CFR 219.19).

rural restructuring—Changes in demographic and economic conditions owing to declines in natural resource production and agriculture (Nelson 2001).

scale—In ecological terms, the extent and resolution in spatial and temporal terms of a phenomenon or analysis, which differs from the definition in cartography regarding the ratio of map distance to Earth surface distance (Jenerette and Wu 2000).

scenic character—A combination of the physical, biological, and cultural images that gives an area its scenic identity and contributes to its sense of place. Scenic character provides a frame of reference from which to determine scenic attractiveness and to measure scenic integrity (36 CFR 219.19).

science synthesis—A narrative review of scientific information from a defined pool of sources that compiles and integrates and interprets findings and describes uncertainty, including the boundaries of what is known and what is not known.

sense of place—The collection of meanings, beliefs, symbols, values, and feelings that individuals or groups associate with a particular locality (Williams and Stewart 1998).

sensitive species—Plant or animal species that receive special conservation attention because of threats to their populations or habitats, but which do not have special status as listed or candidates for listing under the Endangered Species Act.

sensitivity—In ecological contexts, the propensity of communities or populations to change when subject to disturbance, or the opposite of resistance (see "community resistance").

sink population—A population in which reproductive rates are lower than mortality rates but that is maintained by immigration of individuals from outside of that population (see also "source population").

social sustainability—"The capability of society to support the network of relationships, traditions, culture, and activities that connect people to the land and to one another, and support vibrant communities" (36 CFR 219.19). The term is commonly invoked as one of the three parts of a "triple-bottom line" alongside environmental and economic considerations. The concept is an umbrella term for various topics such as quality of life, security, social capital, rights, sense of place, environmental justice, and community resilience, among others discussed in this synthesis.

socioecological resilience—The capacity of socioecological systems (see "socioecological system") to cope with, adapt to, and influence change; to persist and develop in the face of change; and to innovate and transform into new, more desirable configurations in response to disturbance.

socioecological system (or social-ecological system)—A coherent system of biophysical and social factors defined at several spatial, temporal, and organizational scales that regularly interact, continuously adapt, and regulate critical natural, socioeconomic, and cultural resources (Redman et al. 2004); also described as a coupled-human and natural system (Liu et al. 2007).

source population—A population in which reproductive rates exceed those of mortality rates so that the population has the capacity to increase in size. The term is also often used to denote when such a population contributes emigrants (dispersing individuals) that move outside the population, particularly when feeding a sink population.

special forest products—See "nontimber forest products."

special status species—Species that have been listed or proposed for listing as threatened or endangered under the Endangered Species Act.

species of conservation concern—A species, other than federally recognized as a threatened, endangered, proposed,

or candidate species, that is known to occur in the NWFP area and for which the regional forester has determined that the best available scientific information indicates substantial concern about the species' capability to persist over the long term in the Plan area (36 CFR 219.9(c)).

stand—A descriptor of a land management unit consisting of a contiguous group of trees sufficiently uniform in age-class distribution, composition, and structure, and growing on a site of sufficiently uniform quality, to be a distinguishable unit.

standard—A mandatory constraint on project and activity decisionmaking, established to help achieve or maintain the desired condition or conditions, to avoid or mitigate undesirable effects, or to meet applicable legal requirements.

stationarity—In statistics, a process that, while randomly determined, is not experiencing a change in the probability of outcomes.

stewardship contract—A contract designed to achieve land management goals while meeting local and rural community needs, including contributing to the sustainability of rural communities and providing a continuing source of local income and employment.

strategic surveys—One type of field survey, specified under the NWFP, designed to fill key information gaps on species distributions and ecologies by which to determine if species should be included under the Plan's Survey and Manage species list.

stressors—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 (36 CFR 219.19).

structure (ecosystem)—The organization and physical arrangement of biological elements such as snags and down woody debris, vertical and horizontal distribution of vegetation, stream habitat complexity, landscape pattern, and connectivity (FSM 2020).

supporting services—A type of ecosystem service that includes pollination, seed dispersal, soil formation, and nutrient cycling.

Survey and Manage program—A formal part of the NWFP that established protocols for conducting various types of species surveys, identified old-forest-associated species warranting additional consideration for monitoring and protection (see "Survey and Manage species"), and instituted an annual species review procedure that evaluated new scientific and monitoring information on species for potentially recommending changes in their conservation status, including potential removal from the Survey and Manage species list.

Survey and Manage species—A list of species, compiled under the Survey and Manage program of the NWFP, that were deemed to warrant particular attention for monitoring and protection beyond the guidelines for establishing late-successional forest reserves.

sustainability—The capability to meet the needs of the present generation without compromising the ability of future generations to meet their needs (36 CFR 219.19).

sustainable recreation—The set of recreation settings and opportunities in the National Forest System that is ecologically, economically, and socially sustainable for present and future generations (36 CFR 219.19).

sympatric—Two species or populations that share a common geographic range and coexist.

threatened species—Any species that the Secretary of the Interior or the Secretary of Commerce has determined is likely to become an endangered species within the foreseeable future throughout all or a significant portion of its range. Threatened species are listed at 50 CFR sections 17.11, 17.12, and 223.102.

timber harvest—The removal of trees for wood fiber use and other multiple-use purposes (36 CFR 219.19).

timber production—The purposeful growing, tending, harvesting, and regeneration of regulated crops of trees to be cut into logs, bolts, or other round sections for industrial or consumer use (36 CFR 219.19).

topo-edaphic—Related to or caused by particular soil conditions, as of texture or drainage, rather than by physiographic or climatic factors within a defined region or area.

traditional ecological knowledge—"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" (Berkes et al. 2000: 1252). See also "native knowledge."

trailing edge—When describing the range of a species, the boundary at which the species' population is geographically contracting through local extinction at occupied sites.

trophic cascade—Changes in the relative populations of producers, herbivores, and carnivores following the addition or removal of top predators and the resulting disruption of the food web.

uncertainty—Amount or degree of confidence as a result of imperfect or incomplete information.

understory—Vegetation growing below the tree canopy in a forest, including shrubs and herbs that grow on the forest floor.

use of wildland fire—Management of either wildfire or prescribed fire to meet resource objectives specified in land or resource management plans (see "Managing wildfire for resource objectives" and "Prescribed fire").

variable-density thinning—The method of thinning some areas within a stand to a different density (including leaving dense, unthinned areas) than other parts of the stand, which is typically done to promote ecological diversity in a relatively uniform stand.

vegetation series (plant community)—The highest level of the fine-scale component (plant associations) of potential vegetation hierarchy based on the dominant plant species that would occur in late-successional conditions in the absence of disturbance.

vegetation type—A general term for a combination or community of plants (including grasses, forbs, shrubs, or trees), typically applied to existing vegetation rather than potential vegetation.

viable population—A group of breeding individuals of a species capable of perpetuating itself over a given time scale.

vital rates—Statistics describing population dynamics such as reproduction, mortality, survival, and recruitment.

watershed—A region or land area drained by a single stream, river, or drainage network; a drainage basin (36 CFR 219.19).

watershed analysis—An analytical process that characterizes watersheds and identifies potential actions for addressing problems and concerns, along with possible management options. It assembles information necessary to determine the ecological characteristics and behavior of the watershed and to develop options to guide management in the watershed, including adjusting riparian reserve boundaries.

watershed condition assessment—A national approach used by the U.S. Forest Service to evaluate condition of hydrologic units based on 12 indicators, each composed of various attributes (USDA FS 2011).

watershed condition—The state of a watershed based on physical and biogeochemical characteristics and processes (36 CFR 219.19).

watershed restoration—Restoration activities that focus on restoring the key ecological processes required to create and maintain favorable environmental conditions for aquatic and riparian-dependent organisms.

well-being—The condition of an individual or group in social, economic, psychological, spiritual, or medical terms.

wilderness—Any area of land designated by Congress as part of the National Wilderness Preservation System that was established by the Wilderness Act of 1964 (16 U.S.C. 1131–1136) (36 CFR 219.19).

wildlife—Undomesticated animal species, including amphibians, reptiles, birds, mammals, fish, and invertebrates or even all biota, that live wild in an area without being introduced by humans.

wildfire—Unplanned ignition of a wildland fire (such as a fire caused by lightning, volcanoes, unauthorized and accidental human-caused fires), and escaped prescribed fires.

wildland-urban interface (WUI)—The line, area, or zone where structures and other human development meet or intermingle with undeveloped wildland or vegetation fuels.

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