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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 20$^{th}$ 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.

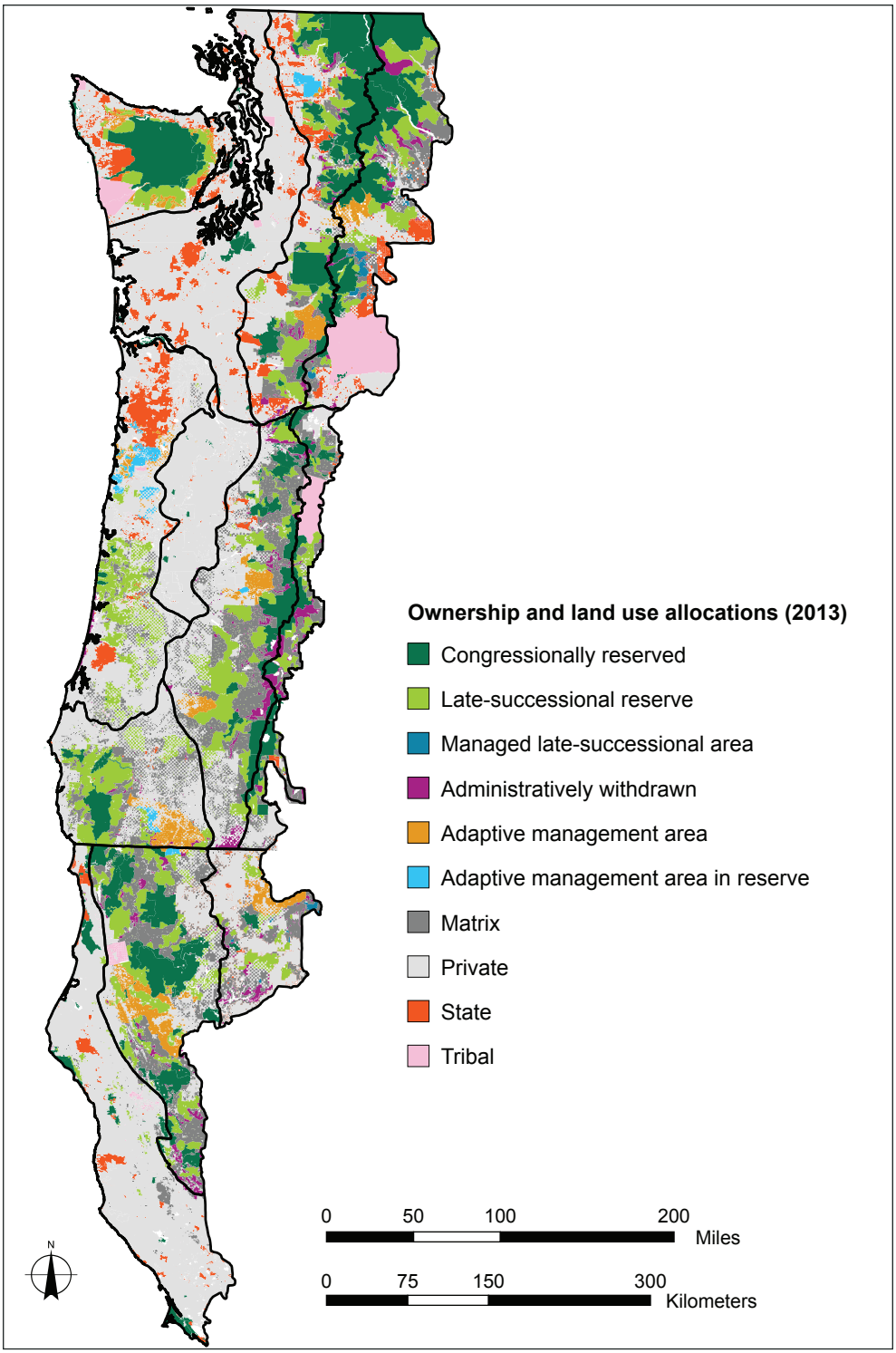


Figure 1-1—Land allocation categories and original 12 physiographic provinces (outlined in black) for the Northwest Forest Plan area. Note that "matrix" includes riparian reserves and other unmapped buffers (e.g., Survey and Manage).

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, socioeconomics, 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.

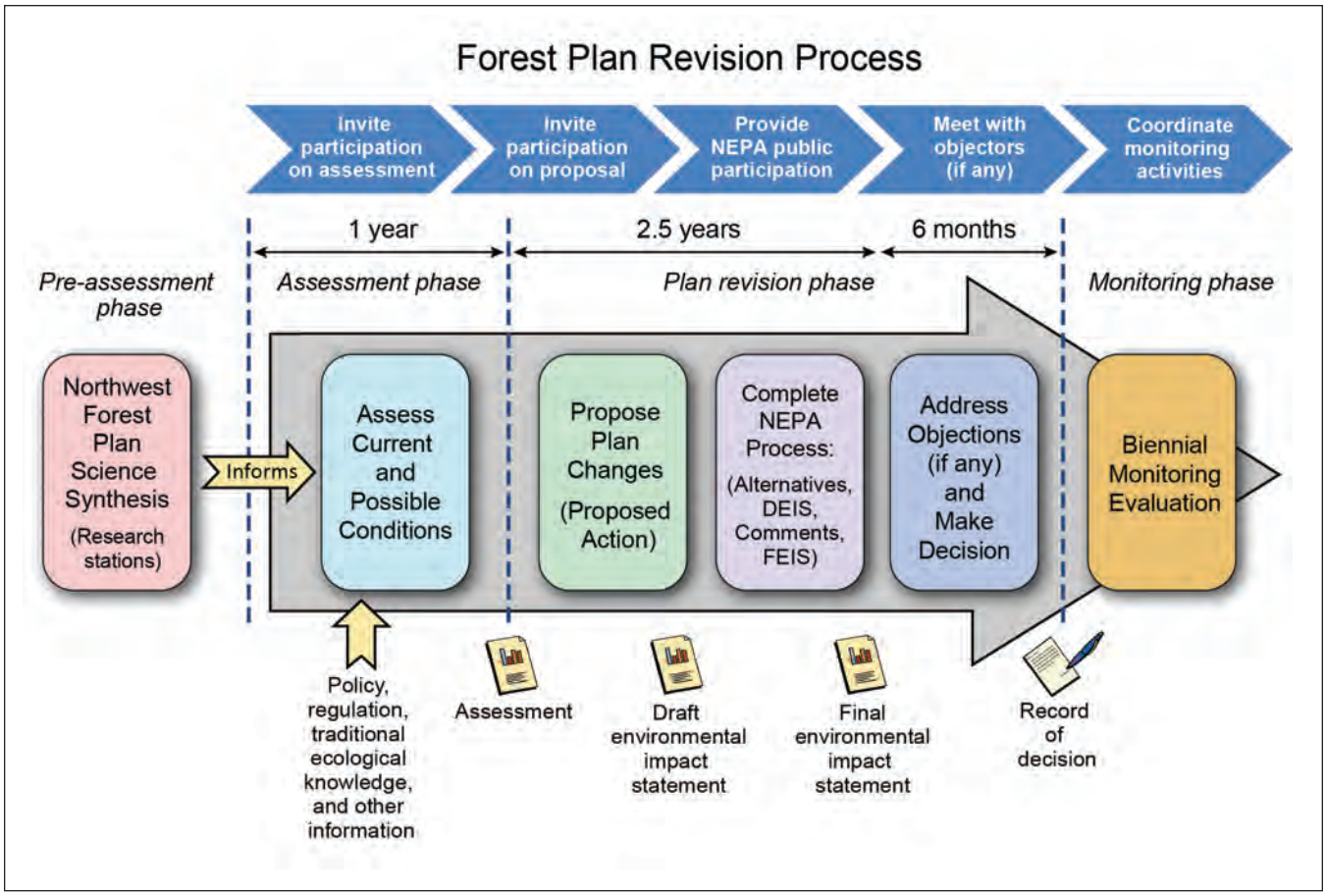


Figure 1-2—The science synthesis is part of the preassessment phase in forest plan revision and will inform the assessment phase of the planning process. NEPA = National Environmental Policy Act.

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.

## Literature Cited

**Bailey, R.G. 2009.** Ecosystem geography: from ecoregions to sites. 2nd ed. New York: Springer. 251 p.

**Barbour, M.G., ed. 2007.** Terrestrial vegetation of California. 3rd ed. Berkeley, CA: University of California Press. 712 p.

**Charnley, S. 2006.** The Northwest Forest Plan as a model for broad-scale ecosystem management: a social perspective. Conservation Biology. 20(2): 330–340. doi:10.1111/j.1523-1739.2006.00388.x.

**Cissel, J.H.; Swanson, F.J.; McKee, W.A.; Burditt, A.L. 1994.** Using the past to plan the future in the Pacific Northwest. Journal of Forestry. 92(8): 30–31.

**Corlett, R.T. 2015.** The Anthropocene concept in ecology and conservation. Trends in Ecology & Evolution. 30(1): 36–41. doi:10.1016/j.tree.2014.10.007.

**Davis, R.J.; Hollen, B.; Hobson, J.; Gower, J.E.; Keenum, D. 2016.** Northwest Forest Plan—the first 20 years (1994–2013): status and trends of northern spotted owl habitats. Gen. Tech. Rep. PNW-GTR-929. Portland, OR: U.S. Department of Agriculture, Forest Service, Pacific Northwest Research Station. 54 p.

**Davis, R.J.; Ohmann, J.L.; Kennedy, R.E.; Cohen, W.B.; Gregory, M.J.; Yang, Z.; Roberts, H.M.; Gray, A.N.; Spies, T.A. 2015.** Northwest Forest Plan—the first 20 years (1994–2013): status and trends of late-successional and old-growth forests. Gen. Tech. Rep. PNW-GTR-911. Portland, OR: U.S. Department of Agriculture, Forest Service, Pacific Northwest Research Station. 112 p. doi:10.2737/pnw-gtr-911.

**Dugger, K.M.; Forsman, E.D.; Franklin, A.B.; Davis, R.J.; White, G.C.; Schwarz, C.J.; Burnham, K.P.; Nichols, J.D.; Hines, J.E.; Yackulic, C.B.; Doherty, P.F.; Bailey, L.; Clark, D.A.; Ackers, S.H.; Andrews, L.S.; Augustine, B.; Biswell, B.L.; Blakesley, J.; Carlson, P.C.; Clement, M.J.; Diller, L.V.; Glenn, E.M.; Green, A.; Gremel, S.A.; Herter, D.R.; Higley, J.M.; Hobson, J.; Horn, R.B.; Huyvaert, K.P.; McCafferty, C.; McDonald, T.; McDonnell, K.; Olson, G.S.; Reid, J.A.; Rockweit, J.; Ruiz, V.; Saenz, J.; Sovern, S.G. 2016.** The effects of habitat, climate, and barred owls on long-term demography of northern spotted owls. Condor. 118(1): 57–116. doi:10.1650/condor-15-24.1.

**Falxa, G.A.; Raphael, M.G., tech. coords. 2016.** Northwest Forest Plan—the first 20 years (1994–2013): status and trend of marbled murrelet populations and nesting habitat. Gen. Tech. Rep. PNW-GTR-933. Portland, OR: U.S. Department of Agriculture, Forest Service, Pacific Northwest Research Station. 132 p.

**Forest Ecosystem Management Assessment Team [FEMAT]. 1993.** Forest ecosystem management: an ecological, economic, and social assessment. Portland, OR: U.S. Department of Agriculture; U.S. Department of the Interior [and others]. [Irregular pagination].

**Franklin, J.F.; Dyrness, C.T. 1973.** Natural vegetation of Oregon and Washington. Gen. Tech. Rep. PNW-8. Portland, OR: U.S. Department of Agriculture, Forest Service, Pacific Northwest Forest and Range Experiment Station. 427 p.

**Grossman, D.H.; Faber-Langendoen, D.; Weakley, A.S.; Anderson, M.; Bourgeron, P.; Crawford, R.; Goodin, K.; Landaal, S.; Metzler, K.; Patterson, K.D.; Pyne, M.; Reid, M. 1998.** International classification of ecological communities: terrestrial vegetation of the United States. The National Vegetation Classification System: development, status, and applications. Arlington, VA: The Nature Conservancy. Vol. I. http://www.natureserve.org/biodiversity-science/publications/terrestrial-vegetation-united-states-volume-i. (20 September 2017).

**Grumbine, R.E. 1994.** What is ecosystem management? Conservation Biology. 8(1): 27–38. doi:10.1046/j.1523-1739.1994.08010027.x.

**Hargrove, W.W.; Hoffman, F.M. 2004.** Potential of multivariate quantitative methods for delineation and visualization of ecoregions. Environmental Management. 34(1): S39–S60.

**Haynes, R.W.; Bormann, B.T.; Lee, D.C.; Martin, J.R. 2006.** Northwest Forest Plan—the first 10 years (1994–2003): synthesis of monitoring and research results. Gen. Tech. Rep. PNW-GTR-651. Portland, OR: U.S. Department of Agriculture, Forest Service, Pacific Northwest Research Station. 292 p. doi:10.2737/pnw-gtr-651.

**Hessburg, P.F.; Salter, R.B.; Richmond, M.B.; Smith, B.G. 2000.** Ecological subregions of the interior Columbia Basin, USA. Applied Vegetation Science. 3(2): 163–180. doi:10.2307/1478995.

**Johnson, K.N. 1997.** Science-based assessments of the forests of the Pacific Northwest. In: Kohm, K.A.; Franklin, J.F., eds. Creating a forestry for the 21st century. Washington, DC: Island Press: 397–409.

**Johnson, K.N.; Franklin, J.F.; Thomas, J.W.; Gordon, J. 1991.** Alternatives for management of late-successional forests of the Pacific Northwest. Report to the Agricultural Committee and the Merchant Marine Committee of the U.S. House of Representatives. Corvallis, OR: Oregon State University, Department of Forest Resources. 59 p.

**Johnson, K.N.; Swanson, F.J. 2009.** Historical context of old growth forests in the Pacific Northwest—policy, practices, and competing world views. In: Spies, T.A.; Duncan, S.L., eds. Old growth in a new world: a Pacific Northwest icon reexamined. Washington, DC: Island Press: 12–28.

**Kennedy, R.E.; Yang, Z.; Cohen, W.B.; Pfaff, E.; Braaten, J.; Nelson, P. 2012.** Spatial and temporal patterns of forest disturbance and regrowth within the area of the Northwest Forest Plan. Remote Sensing of Environment. 122: 117–133. doi:10.1016/j.rse.2011.09.024.

**Lugo, A.E.; Brown, S.L.; Dodson, R.; Smith, T.S.; Shugart, H.H. 1999.** The Holdridge life zones of the conterminous United States in relation to ecosystem mapping. Journal of Biogeography. 26(5): 1025–1038. doi:10.1046/j.1365-2699.1999.00329.x.

**Millennium Ecosystem Assessment [MEA]. 2005.** Ecosystems and human well-being: synthesis. Washington, DC: Island Press. 137 p.

**Miller, S.A.; Gordon, S.N.; Eldred, P.; Beloin, R.M.; Wilcox, S.; Raggon, M.; Andersen, H.; Muldoon, A. 2017.** Northwest Forest Plan—the first 20 years (1994–2013): watershed condition status and trend. Gen. Tech. Rep. PNW-GTR-932. Portland, OR: U.S. Department of Agriculture, Forest Service, Pacific Northwest Research Station. 74 p.

**Ohmann, J.L.; Spies, T.A. 1998.** Regional gradient analysis and spatial pattern of woody plant communities of Oregon forests. Ecological Monographs. 68(2): 151–182. doi:10.1890/0012-9615(1998)068[0151:rgaasp]2.0.co;2.

**Phillips, S.J.; Anderson, R.P.; Schapire, R.E. 2006.** Maximum entropy modeling of species geographic distributions. Ecological Modelling. 190(3–4): 231–259. doi:10.1016/j.ecolmodel.2005.03.026.

**Pipkin, J. 1998.** The Northwest Forest Plan revisited. Washington, DC: U.S. Department of the Interior, Office of Policy Analysis. 117 p. https://reo.gov/documents/reports/NFP_revisited.htm. (20 September 2017).

**Rapp, V. 2008.** Northwest Forest Plan—the first 10 years (1994–2003): first-decade results of the Northwest Forest Plan. Gen. Tech. Rep. PNW-GTR-720. Portland, OR: U.S. Department of Agriculture, Forest Service, Pacific Northwest Research Station. 42 p. doi:10.2737/pnw-gtr-720.

**Schultz, C.A.; Sisk, T.D.; Noon, B.R.; Nie, M.A. 2013.** Wildlife conservation planning under the United States Forest Service's 2012 planning rule. The Journal of Wildlife Management. 77(3): 428–444. doi:10.1002/jwmg.513.

**Spies, T.A.; Duncan, S.L. 2009.** Old growth in a new world: a Pacific Northwest icon reexamined. Washington, DC: Island Press. 360 p.

**Steen, H.K. 2004.** The U.S. Forest Service: a history. Seattle, WA: Forest History Society in association with University of Washington Press. 356 p.

**Stine, P.; Hessburg, P.; Spies, T.; Kramer, M.; Fettig, C.J.; Hansen, A.; Lehmkuhl, J.; O'Hara, K.; Polivka, K.; Singleton, P.; Charnley, S.; Merschel, A.; White, R. 2014.** 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. Gen. Tech. Rep. PNW-GTR-897. Portland, OR: U.S. Department of Agriculture, Forest Service, Pacific Northwest Research Station. 254 p. doi:10.2737/pnw-gtr-897.

**Thomas, J.W.; Franklin, J.F.; Gordon, J.; Johnson, K.N. 2006.** The Northwest Forest Plan: origins, components, implementation experience, and suggestions for change. Conservation Biology. 20(2): 277–287. doi:10.1111/j.1523-1739.2006.00385.x.

**Thomas, J.W.; Raphael, M.G.; Anthony, M.G.; Forsman, E.D.; Gunderson, A.G.; Holthausen, R.S.; Marcot, B.G.; Reeves, G.H.; Sedell, J.R.; Solis, D.M. 1993.** Viability assessments and management considerations for species associated with late-succession and old-growth forests of the Pacific Northwest—the report of the Scientific Analysis Team. Washington, DC: U.S. Department of Agriculture, Forest Service, National Forest System. 530 p.

**U.S. Department of Agriculture, Forest Service [USDA FS]. 2012.** National forest system land management planning. Subpart A. 36 CFR Part 219. Federal Register. 77(68): 21260–21276.

**U.S. Department of Agriculture, Forest Service [USDA FS]. 2016.** Ecosystem restoration. Forest Service Manual. Chapter 2020. Washington, DC.

**U.S. Department of Agriculture, Forest Service; U.S. Department of the Interior, Bureau of Land Management [USDA and USDI]. 1994.** Record of decision for amendments to Forest Service and Bureau of Land Management planning documents within the range of the northern spotted owl. [Place of publication unknown]. 74 p.

**U.S. Department of the Interior, Fish and Wildlife Service [USFWS]. 2011.** Revised recovery plan for the northern spotted owl (*Strix occidentalis caurina*). Portland, OR. 258 p.

**U.S. Office of Management and Budget [USOMB]. 2003.** Ensuring the quality of data disseminated by the federal government. Federal Register. 67(36): 8452–8460. https://www.whitehouse.gov/omb/fedreg_final_information_quality_guidelines. (20 September 2017).

## 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).

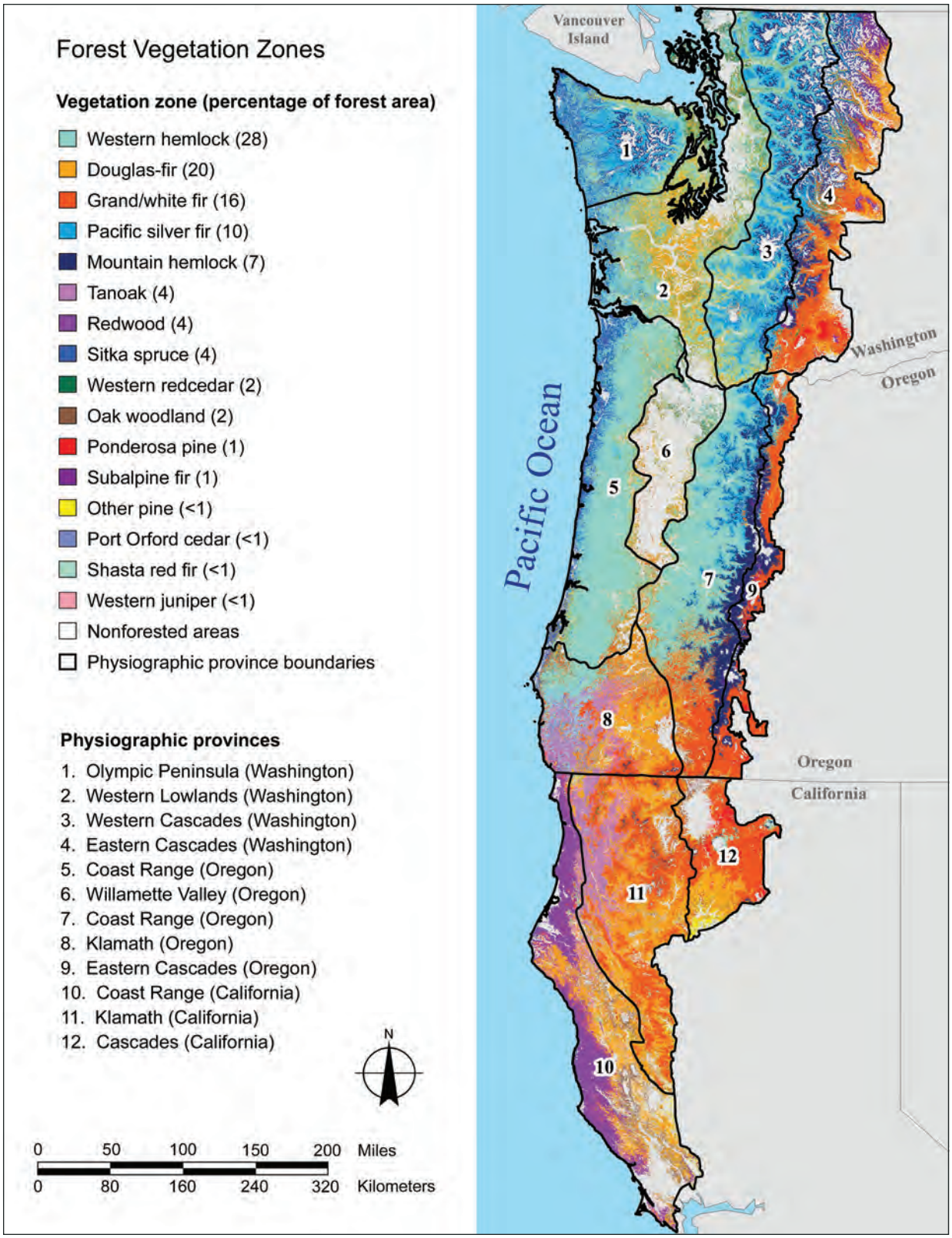



Figure 2-1—Geographic distribution of potential vegetation zones (Simpson 2013) and physiographic provinces within the Northwest Forest Plan area.

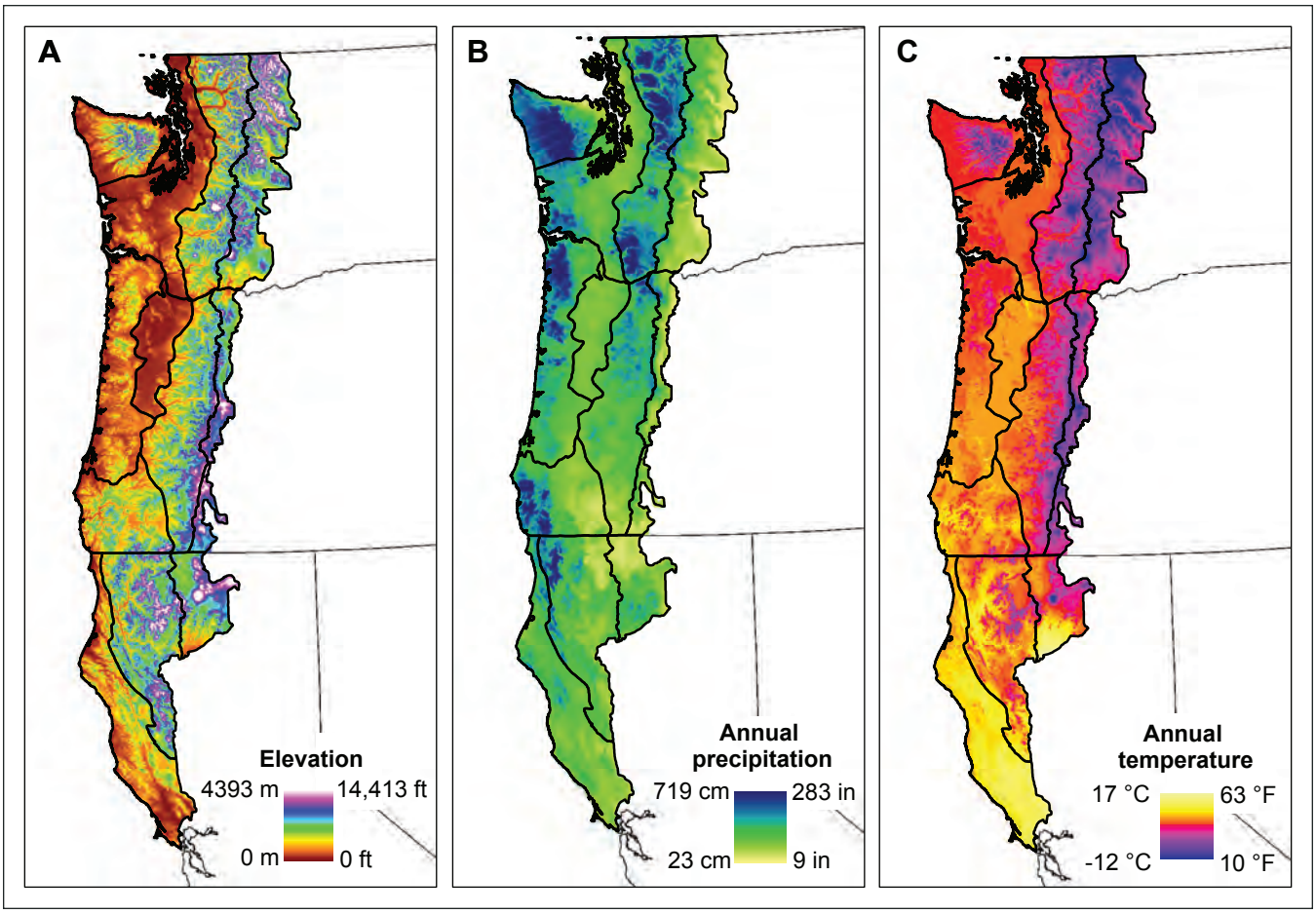



Figure 2-2—Maps of (A) elevation, (B) annual precipitation, and (C) annual temperature in the Northwest Forest Plan area. Temperature and precipitation are derived from 30 arc-second (~800 m) PRISM (parameter-elevation regressions on independent slopes model) (Daly et al. 2008) grids averaged from 1971 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).

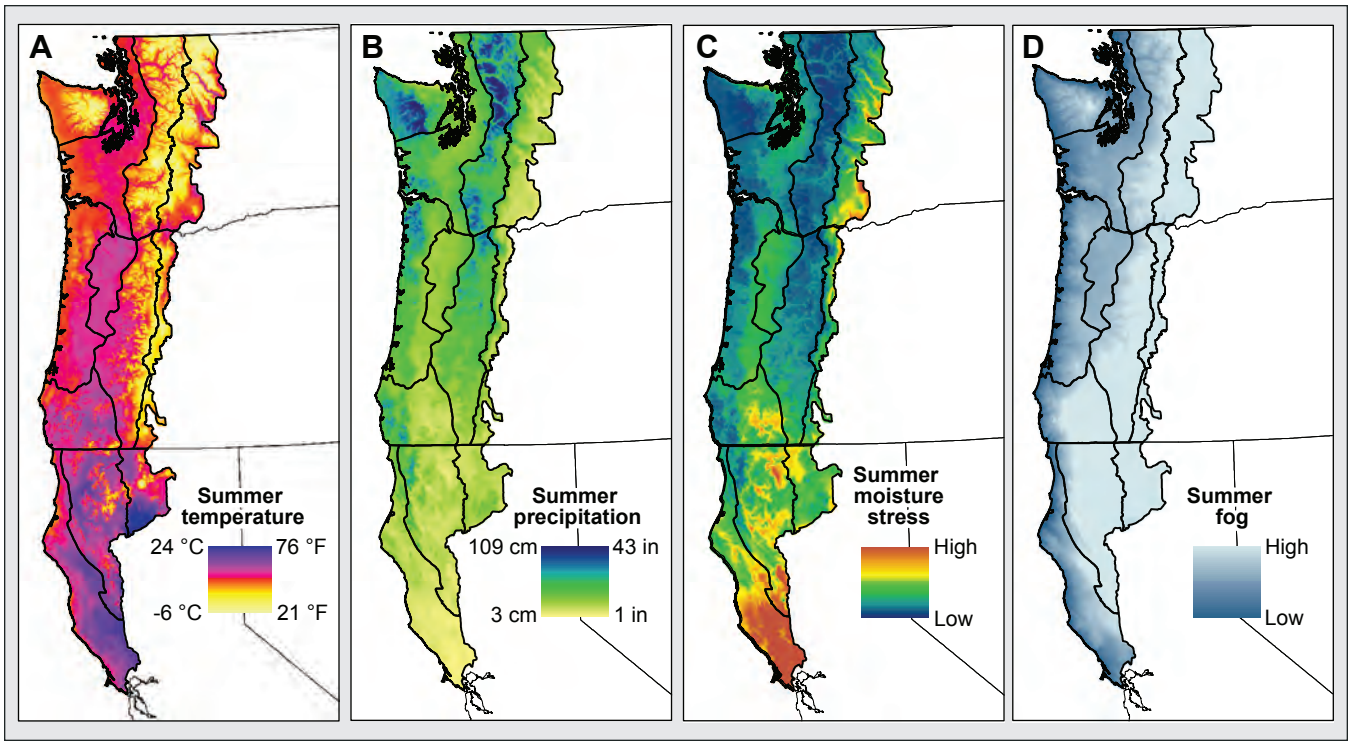



Figure 2-3—Maps of (A) mean summer temperature, (B) total summer precipitation, (C) summer moisture stress, and (D) summer fog in the Northwest Forest Plan area. Temperature and precipitation are derived from 30 arc-second (~800 m) PRISM (parameter-elevation regressions on independent slopes model) (Daly 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.

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 19$^{th}$ and early 20$^{th}$ 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 20$^{th}$-century range of variability (Kerns et al. 2016).

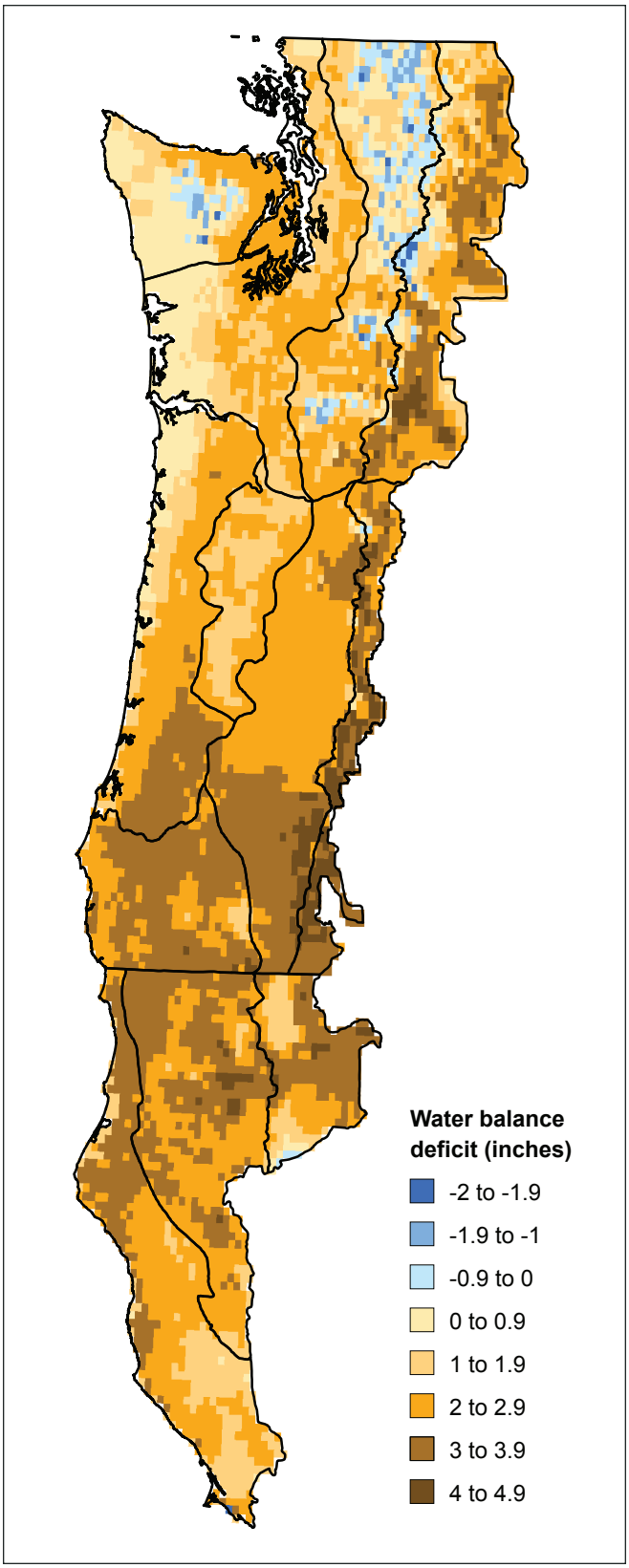
In northern California, under the mitigation-oriented B1 scenario, annual temperature is projected to increase by 2.7 °F (1.5 °C) by 2100, and, under the high-growth A2 scenario, the increase is projected to be 8.1 °F (4.5 °C) (Cayan et al. 2008). Simulations depict drier futures under the B1 and A2 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 20$^{th}$-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 20$^{th}$-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 20$^{th}$-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 21$^{st}$ 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.

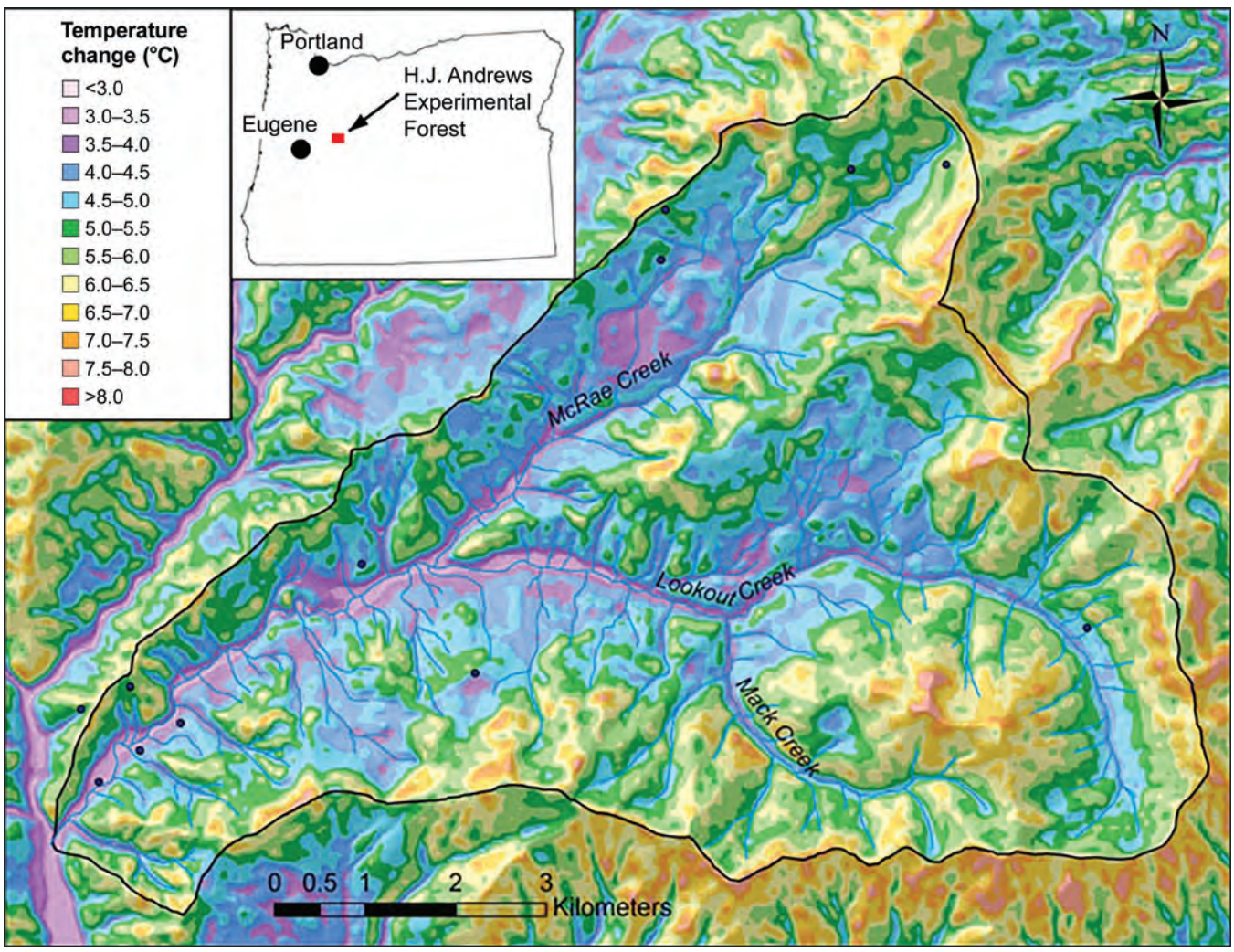



Figure 2-6—Projected changes in maximum December temperatures in response to a 2.5 °C regional temperature increase and changes in atmospheric circulation patterns in the western Cascade Range of Oregon. Source: Daly et al. (2010).

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 20$^{th}$ 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 20$^{th}$ 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-20$^{th}$ 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.

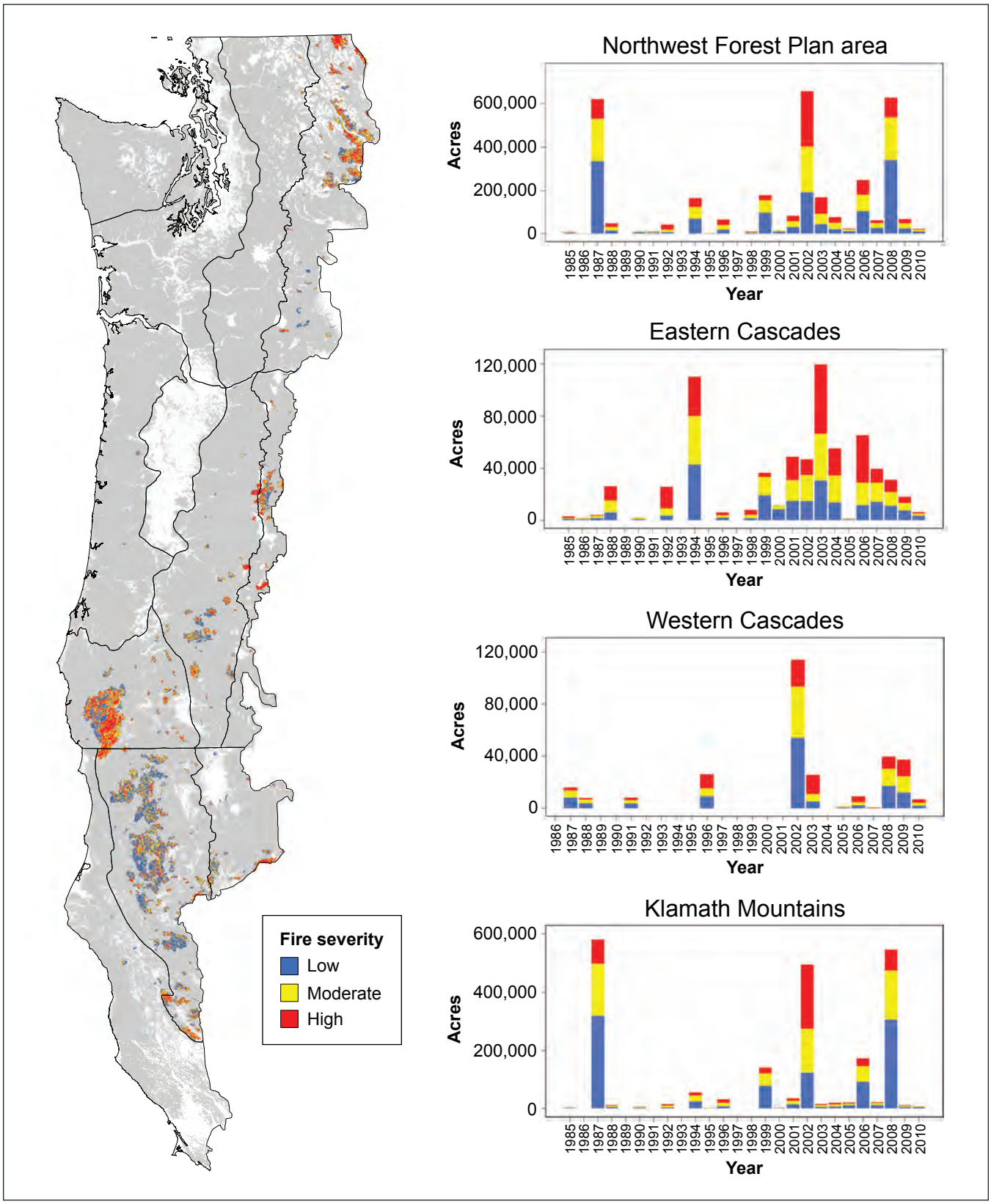



Figure 2-7—Geographic patterns of burn severity from 1985 to 2010 in the Northwest Forest Plan area. Burn severity is derived from the relativized version of the difference in the normalized burn ratio and is based on the percentage of basal area mortality as follows: low (<25 percent), moderate (25 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 HilleRis-Lambers 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-21$^{st}$ 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 21$^{st}$ 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<br>Favor drought-resistant species/genotypes |
| | Foster genetic and phenotypic diversity | Protect trees adapted to water stress<br>Collect seed for future<br>Maintain connectivity for natural species migration |
| Increasing area affected by fire, insects, and pathogens | Increase stand resilience | Thinning and prescribed fire<br>Increase stand heterogeneity<br>Favor fire-tolerant species |
| | Increase landscape resilience | Increase landscape heterogeneity<br>Increase diversity of patch sizes<br>Use topography to guide treatments |
| Loss of forest cover | Monitoring of change | Use existing data and add more where needed<br>Planting/assisted migration<br>Maintain connectivity for natural species migration |
| Exotic species | Increase control efforts | Early detection/rapid response/frequent inventory<br>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

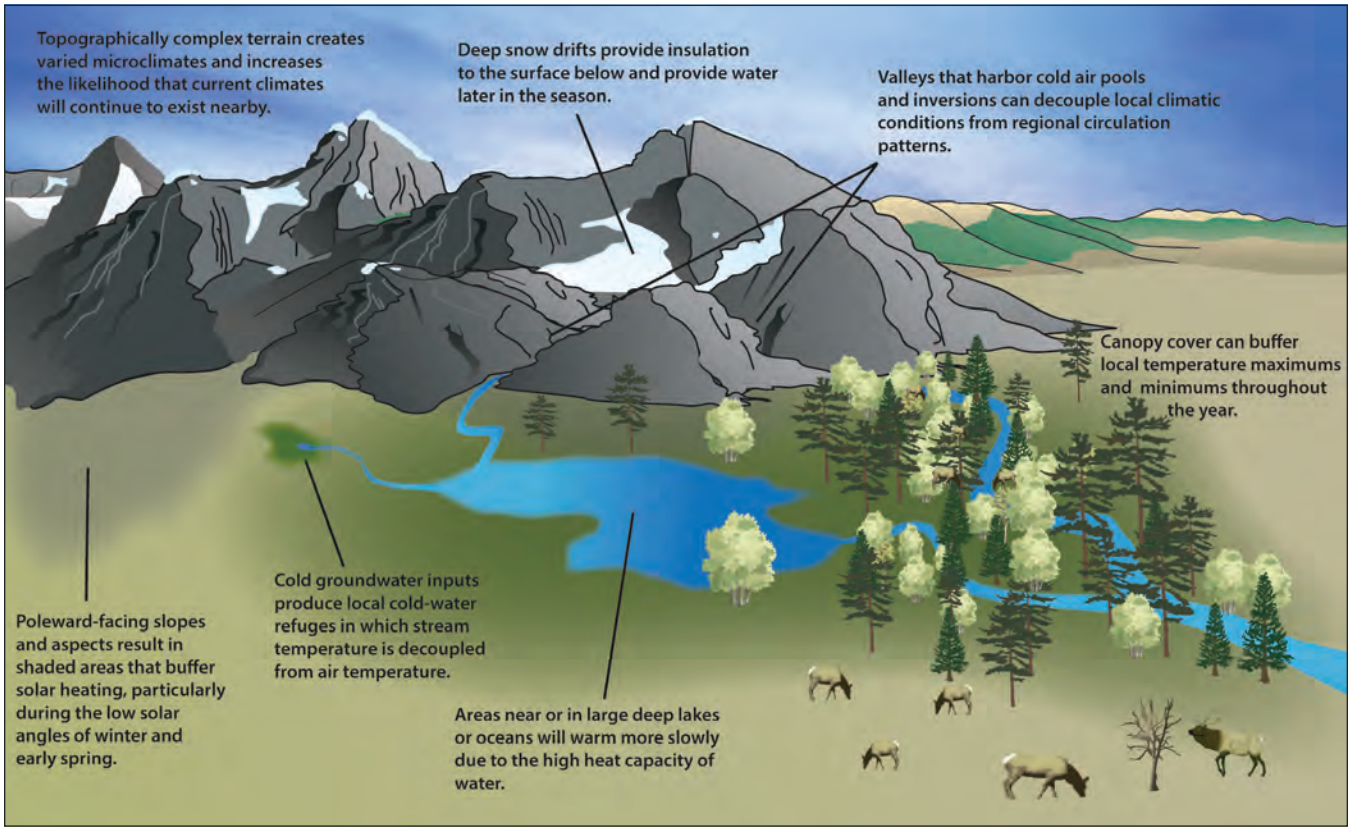



Figure 2-8—Examples of the physiographic and vegetation-based refugia that may experience reduced rates of climate change. Source: Morelli et al. (2016).

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

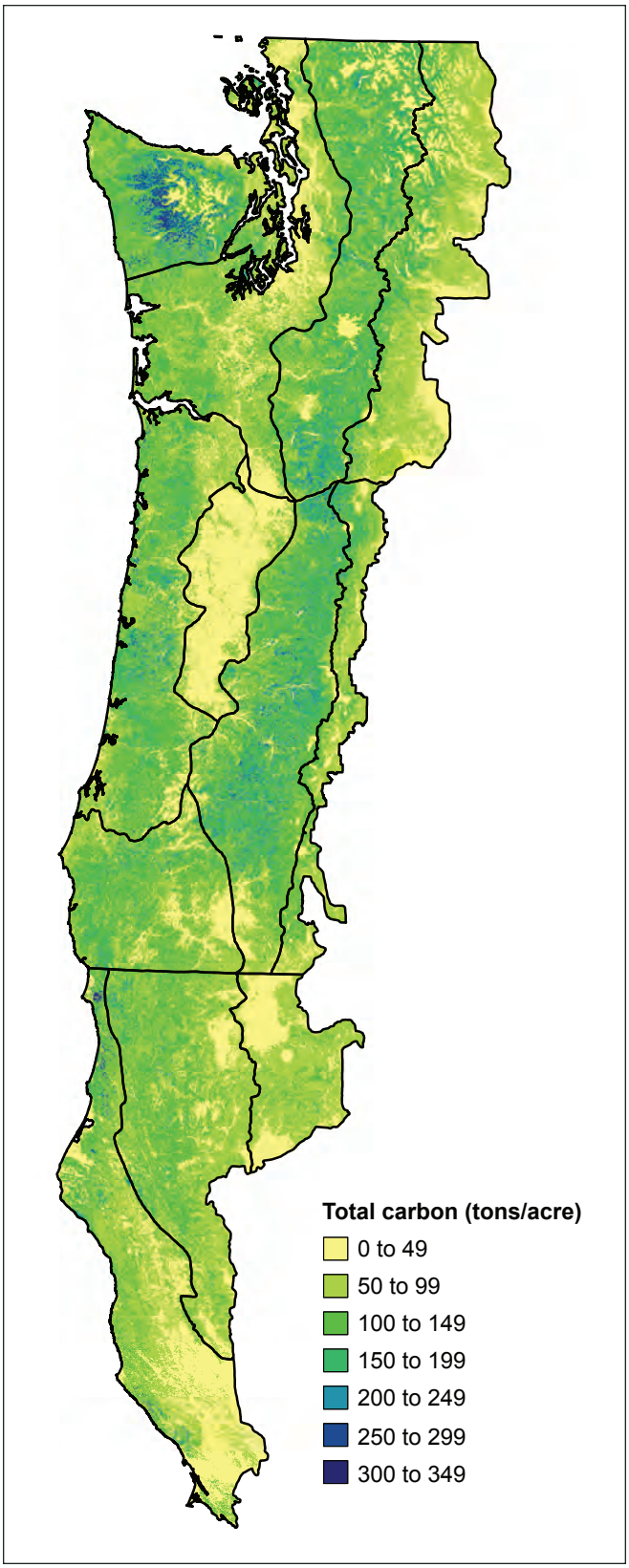



Figure 2-9—Total forest carbon density in the Northwest Forest Plan area (2000–2009). Carbon estimates are from Wilson et al. (2013). Map boundaries correspond with the physiographic provinces in figure 2-1.

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 20$^{th}$-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 21$^{st}$ 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.

## Literature Cited

**Abatzoglou, J.T.; Kolden, C.A.; Williams, A.P.; Lutz, J.A.; Smith, A.M.S. 2017.** Climatic influences on interannual variability in regional burn severity across western US forests. International Journal of Wildland Fire. 26(4): 269. doi:10.1071/wf16165.

**Abatzoglou, J.T.; Rupp, D.E.; Mote, P.W. 2014a.** Questionable evidence of natural warming of the northwestern United States. Proceedings of the National Academy of Sciences of the United States of America. 111(52): E5605–E5606. doi:10.1073/pnas.1421311112.

**Abatzoglou, J.T.; Rupp, D.E.; Mote, P.W. 2014b.** Seasonal climate variability and change in the Pacific Northwest of the United States. Journal of Climate. 27(5): 2125–2142. doi:10.1175/jcli-d-13-00218.1.

**Abatzoglou, J.T.; Williams, A.P. 2016.** Impact of anthropogenic climate change on wildfire across western US forests. Proceedings of the National Academy of Sciences of the United States of America. 113(42): 11770–11775. doi:10.1073/pnas.1607171113.

**Acker, S.A.; Boetsch, J.R.; Bivin, M.; Whiteaker, L.; Cole, C.; Philippi, T. 2015.** Recent tree mortality and recruitment in mature and old-growth forests in western Washington. Forest Ecology and Management. 336: 109–118. doi:10.1016/j.foreco.2014.10.008.

**Acosta, M.; Corral, S. 2017.** Multicriteria decision analysis and participatory decision support systems in forest management. Forests. 8(4): 116. doi:10.3390/f8040116.

**Adams, H.D.; Guardiola-Claramonte, M.; Barron-Gafford, G.A.; Villegas, J.C.; Breshears, D.D.; Zou, C.B.; Troch, P.A.; Huxman, T.E. 2009.** Temperature sensitivity of drought-induced tree mortality portends increased regional die-off under global-change-type drought. Proceedings of the National Academy of Sciences of the United States of America. 106(17): 7063–7066. doi:10.1073/pnas.0901438106.

**Agee, J.K. 1991.** Fire history along an elevational gradient in the Siskiyou Mountains, Oregon. Northwest Science. 65: 88–199.

**Agee, J.K. 1993.** Fire ecology of Pacific Northwest forests. Washington, DC: Island Press. 505 p.

**Agee, J.K.; Finney, M.; Gouvenain, R.D. 1990.** Forest fire history of Desolation Peak, Washington. Canadian Journal of Forest Research. 20(3): 350–356. doi:10.1139/x90-051.

**Agee, J.K.; Krusemark, F. 2001.** Forest fire regime of the Bull Run Watershed, Oregon. Northwest Science. 75: 292–306.

**Agee, J.K.; Skinner, C.N. 2005.** Basic principles of forest fuel reduction treatments. Forest Ecology and Management. 211(1–2): 83–96. doi:10.1016/j.foreco.2005.01.034.

**Ager, A.A.; Finney, M.A.; McMahan, A.; Cathcart, J. 2010.** Measuring the effect of fuel treatments on forest carbon using landscape risk analysis. Natural Hazards and Earth System Science. 10(12): 2515–2526. doi:10.5194/nhess-10-2515-2010.

**Agne, M.C.; Woolley, T.; Fitzgerald, S. 2016.** Fire severity and cumulative disturbance effects in the post-mountain pine beetle lodgepole pine forests of the pole creek fire. Forest Ecology and Management. 366: 73–86. doi:10.1016/j.foreco.2016.02.004.

**Albright, W.L.; Peterson, D.L. 2013.** Tree growth and climate in the Pacific Northwest, North America: a broad-scale analysis of changing growth environments. Journal of Biogeography. 40(11): 2119–2133. doi:10.1111/jbi.12157.

**Allen, C.D.; Breshears, D.D. 1998.** Drought-Induced shift of a forest-woodland ecotone: rapid landscape response to climate variation. Proceedings of the National Academy of Sciences of the United States of America. 95(25): 14839–14842. doi:10.1073/pnas.95.25.14839.

**Allen, C.D.; Breshears, D.D.; McDowell, N.G. 2015.** On underestimation of global vulnerability to tree mortality and forest die-off from hotter drought in the anthropocene. Ecosphere. 6(8): art129. doi:10.1890/es15-00203.1.

**Allen, C.D.; Macalady, A.K.; Chenchouni, H.; Bachelet, D.; McDowell, N.; Vennetier, M.; Kitzberger, T.; Rigling, A.; Breshears, D.D.; Hogg, E.H.; Gonzalez, P.; Fensham, R.; Zhang, Z.; Castro, J.; Demidova, N.; Lim, J-H; Allard, G.; Running, S.W.; Semerci, A.; Cobb, N. 2010.** A global overview of drought and heat-induced tree mortality reveals emerging climate change risks for forests. Forest Ecology and Management. 259(4): 660–684.

**Anderegg, W.R.L.; Hicke, J.A.; Fisher, R.A.; Allen, C.D.; Aukema, J.; Bentz, B.; Hood, S.; Lichstein, J.W.; Macalady, A.K.; McDowell, N.; Pan, Y.; Raffa, K.; Sala, A.; Shaw, J.D.; Stephenson, N.L.; Tague, C.; Zeppel, M. 2015.** Tree mortality from drought, insects, and their interactions in a changing climate. New Phytologist. 208(3): 674–683. doi:10.1111/nph.13477.

**Anderegg, W.R.L.; Kane, J.M.; Anderegg, L.D.L. 2012.** Consequences of widespread tree mortality triggered by drought and temperature stress. Nature Climate Change. 3(1): 30–36. doi:10.1038/nclimate1635.

**Asner, G.P.; Brodrick, P.G.; Anderson, C.B.; Vaughn, N.; Knapp, D.E.; Martin, R.E. 2016.** Progressive forest canopy water loss during the 2012–2015 California drought. Proceedings of the National Academy of Sciences of the United States of America. 113(2): E249–E255. doi:10.1073/pnas.1523397113.

**Baker, W.L. 2012.** Implications of spatially extensive historical data from surveys for restoring dry forests of Oregon's eastern Cascades. Ecosphere. 3(3): art23. doi:10.1890/es11-00320.1.

**Bansal, S.; Harrington, C.A.; Gould, P.J.; St. Clair, J.B. 2015.** Climate-related genetic variation in drought-resistance of Douglas-fir (*Pseudotsuga menziesii*). Global Change Biology. 21(2): 947–958. doi:10.1111/gcb.12719.

**Bansal, S.; Harrington, C.A.; St. Clair, J.B. 2016.** Tolerance to multiple climate stressors: a case study of Douglas-fir drought and cold hardiness. Ecology and Evolution. 6(7): 2074–2083. doi:10.1002/ece3.2007.

**Barnett, K.; Parks, S.; Miller, C.; Naughton, H. 2016.** Beyond fuel treatment effectiveness: characterizing interactions between fire and treatments in the US. Forests. 7(10): 237. doi:10.3390/f7100237.

**Barr, B.R.; Koopman, M.E.; Williams, C.D.; Vynne, S.J.; Hamilton, R.; Doppelt, B. 2010.** Preparing for climate change in the Klamath basin. Ashland, OR: National Center for Conservation Science & Policy and the Climate Leadership Initiative. http://hdl.handle.net/1794/10722. (13 December 2017).

**Bartlein, P.J.; Anderson, K.H.; Anderson, P.M.; Edwards, M.E.; Mock, C.J.; Thompson, R.S.; Webb, R.S.; Webb Iii, T.; Whitlock, C. 1998.** Paleoclimate simulations for North America over the past 21,000 years. Quaternary Science Reviews. 17(6–7): 549–585. doi:10.1016/s0277-3791(98)00012-2.

**Bell, D.M.; Schlaepfer, D.R. 2016.** On the dangers of model complexity without ecological justification in species distribution modeling. Ecological Modelling. 330: 50–59. doi:10.1016/j.ecolmodel.2016.03.012.

**Bell, J.L.; Sloan, L.C.; Snyder, M.A. 2004.** Regional changes in extreme climatic events: a future climate scenario. Journal of Climate. 17(1): 81–87. doi:10.1175/1520-0442(2004)017<0081:rciece>2.0.co;2.

**Bennett, K.; Provan, J. 2008.** What do we mean by 'refugia'? Quaternary Science Reviews. 27(27–28): 2449–2455. doi:10.1016/j.quascirev.2008.08.019.

**Bentz, B.J.; Duncan, J.P.; Powell, J.A. 2016.** Elevational shifts in thermal suitability for mountain pine beetle population growth in a changing climate. Forestry. 89(3): 271–283. doi:10.1093/forestry/cpv054.

**Bentz, B.J.; Régnière, J.; Fettig, C.J.; Hansen, E.M.; Hayes, J.L.; Hicke, J.A.; Kelsey, R.G.; Negrón, J.F.; Seybold, S.J. 2010.** Climate change and bark beetles of the western United States and Canada: direct and indirect effects. BioScience. 60(8): 602–613. doi:10.1525/bio.2010.60.8.6.

**Berg, N.; Hall, A. 2015.** Increased interannual precipitation extremes over California under climate change. Journal of Climate. 28(16): 6324–6334. doi:10.1175/jcli-d-14-00624.1.

**Berner, L.T.; Law, B.E.; Meddens, A.J.H.; Hicke, J.A. 2017.** Tree mortality from fires, bark beetles, and timber harvest during a hot and dry decade in the western United States (2003–2012). Environmental Research Letters. 12(6): 065005. doi:10.1088/1748-9326/aa6f94.

**Betts, M.G.; Fahrig, L.; Hadley, A.S.; Halstead, K.E.; Bowman, J.; Robinson, W.D.; Wiens, J.A.; Lindenmayer, D.B. 2014.** A species-centered approach for uncovering generalities in organism responses to habitat loss and fragmentation. Ecography. 37(6): 517–527. doi:10.1111/ecog.00740.

**Bizikova, L.; Krcmar, E. 2015.** Integrated scenario planning and multi-criteria decision analysis framework with application to forest planning. Open Journal of Forestry. 05(02): 139–153. doi:10.4236/ojf.2015.52014.

**Black, B.A.; Shaw, D.C.; Stone, J.K. 2010.** Impacts of Swiss needle cast on overstory Douglas-fir forests of the western Oregon Coast Range. Forest Ecology and Management. 259(8): 1673–1680. doi:10.1016/j.foreco.2010.01.047.

**Black, S.H.; Kulakowski, D.; Noon, B.R.; DellaSala, D.A. 2013.** Do bark beetle outbreaks increase wildfire risks in the central U.S. Rocky Mountains? Implications from recent research. Natural Areas Journal. 33(1): 59–65. doi:10.3375/043.033.0107.

**Bond, M.L.; Lee, D.E.; Bradley, C.M.; Hanson, C.T. 2009.** Influence of pre-fire tree mortality on fire severity in conifer forests of the San Bernardino Mountains, California. The Open Forest Science Journal. 2(1): 41–47. doi:10.2174/1874398600902010041.

**Bork, J.L. 1984.** Fire history in three vegetation types on the eastern side of the Oregon Cascades. Corvallis, OR: Oregon State University. 94 p. Ph.D. dissertation.

**Bradford, J.B.; Bell, D.M. 2017.** A window of opportunity for climate-change adaptation: easing tree mortality by reducing forest basal area. Frontiers in Ecology and the Environment. 15(1): 11–17. doi:10.1002/fee.1445.

**Briles, C.E.; Whitlock, C.; Bartlein, P.J. 2005.** Postglacial vegetation, fire, and climate history of the Siskiyou Mountains, Oregon, USA. Quaternary Research. 64(01): 44–56. doi:10.1016/j.yqres.2005.03.001.

**Brooks, M.L.; D'Antonio, C.M.; Richardson, D.M.; Grace, J.B.; Keeley, J.E.; Ditomaso, J.M.; Hobbs, R.J.; Pellant, M.; Pyke, D. 2004.** Effects of invasive alien plants on fire regimes. BioScience. 54(7): 677. doi:10.1641/0006-3568(2004)054[0677:eoiapo]2.0.co;2.

**Brown, P.M.; Baxter, W.T. 2003.** Fire history in coast redwood forests of the Mendocino Coast California. Northwest Science. 77(2): 147–158.

**Brown, P.M.; Kaye, M.W.; Buckley, D. 1999.** Fire history in Douglas-fir and coast redwood forests at Point Reyes National Seashore, California. Northwest Science. 73: 205–216.

**Brown, P.M.; Swetnam, T.W. 1994.** A cross-dated fire history from coast redwood near Redwood National Park, California. Canadian Journal of Forest Research. 24(1): 21–31. doi:10.1139/x94-004.

**Buma, B. 2015.** Disturbance interactions: characterization, prediction, and the potential for cascading effects. Ecosphere. 6(4): art70. doi:10.1890/es15-00058.1.

**Caldwell, B.A. 2006.** Effects of invasive Scotch broom on soil properties in a Pacific coastal prairie soil. Applied Soil Ecology. 32(1): 149–152. doi:10.1016/j.apsoil.2004.11.008.

**Callaway, R.M.; Brooker, R.W.; Choler, P.; Kikvidze, Z.; Lortie, C.J.; Michalet, R.; Paolini, L.; Pugnaire, F.I.; Newingham, B.; Aschehoug, E.T.; Armas, C.; Kikodze, D.; Cook, B.J. 2002.** Positive interactions among alpine plants increase with stress. Nature. 417(6891): 844–848. doi:10.1038/nature00812.

**Campbell, J.L.; Harmon, M.E.; Mitchell, S.R. 2012.** Can fuel-reduction treatments really increase forest carbon storage in the western US by reducing future fire emissions? Frontiers in Ecology and the Environment. 10(2): 83–90. doi:10.1890/110057.

**Carroll, C.; Dunk, J.R.; Moilanen, A. 2010.** Optimizing resiliency of reserve networks to climate change: multispecies conservation planning in the Pacific Northwest, USA. Global Change Biology. 16(3): 891–904. doi:10.1111/j.1365-2486.2009.01965.x.

**Case, M.J.; Lawler, J.J. 2016.** Relative vulnerability to climate change of trees in western North America. Climatic Change. 136(2): 367–379. doi:10.1007/s10584-016-1608-2.

**Case, M.J.; Lawler, J.J.; Tomasevic, J.A. 2015.** Relative sensitivity to climate change of species in northwestern North America. Biological Conservation. 187: 127–133. doi:10.1016/j.biocon.2015.04.013.

**Cayan, D.; Dettinger, M.; Pierce, D.; Das, T.; Knowles, N.; Ralph, F.; Sumargo, E. 2016.** Natural variability, anthropogenic climate change, and impacts on water availability and flood extremes in the western United States. In: Miller, K.A.; Hamlet, A.F.; Kenney, D.S.; Redmond, K.T., eds. Water policy and planning in a variable and changing climate. Boca Raton, FL: CRC Press: 17–42. doi:10.1201/b19534-4.

**Cayan, D.R.; Dettinger, M.D.; Kammerdiener, S.A.; Caprio, J.M.; Peterson, D.H. 2001.** Changes in the onset of spring in the western United States. Bulletin of the American Meteorological Society. 82(3): 399–415. doi:10.1175/1520-0477(2001)082<0399:citoos>2.3.co;2.

**Cayan, D.R.; Maurer, E.P.; Dettinger, M.D.; Tyree, M.; Hayhoe, K. 2008.** Climate change scenarios for the California region. Climatic Change. 87(S1): 21–42. doi:10.1007/s10584-007-9377-6.

**Chen, J.; Franklin, J.F.; Spies, T.A. 1993.** Contrasting microclimates among clearcut, edge, and interior of old-growth Douglas-fir forest. Agricultural and Forest Meteorology. 63(3–4): 219–237. doi:10.1016/0168-1923(93)90061-l.

**Chmura, D.J.; Anderson, P.D.; Howe, G.T.; Harrington, C.A.; Halofsky, J.E.; Peterson, D.L.; Shaw, D.C.; St. Clair, J.B. 2011.** Forest responses to climate change in the northwestern United States: ecophysiological foundations for adaptive management. Forest Ecology and Management. 261(7): 1121–1142. doi:10.1016/j.foreco.2010.12.040.

**Cohen, W.B.; Yang, Z.; Stehman, S.V.; Schroeder, T.A.; Bell, D.M.; Masek, J.G.; Huang, C.; Meigs, G.W. 2016.** Forest disturbance across the conterminous United States from 1985–2012: the emerging dominance of forest decline. Forest Ecology and Management. 360: 242–252. doi:10.1016/j.foreco.2015.10.042.

**Collins, M.; Knutti, R.; Arblaster, J.; Dufresne, J.L.; Fichefet, T.; Friedlingstein, P.; Gao, X.; Gutowski, W.J.; Johns, T.; Krinner, G.; Shongwe, M.; Tebaldi, C.; Weaver, A.J.; Wehner, M. 2014.** Long-term climate change: projections, commitments and irreversibility. In: Stocker, T.F.; Qin, D.; Plattner, G.K.; Tignor, M.; Allen, S.K.; Boschung, J.; Nauels, A.; Xia, Y.; Bex, V.; Midgley, P.M., eds. Climate change 2013—the physical science basis. Cambridge, United Kingdom: Cambridge University Press: 953–1028. doi:10.1017/cbo9781107415324.023.

**Cook, E.R.; Woodhouse, C.A.; Eakin, C.M.; Meko, D.M.; Stahle, D.W. 2004.** Long-term aridity changes in the western United States. Science. 306(5698): 1015–1018. doi:10.1126/science.1102586.

**Coops, N.C.; Waring, R.H. 2011a.** A process-based approach to estimate lodgepole pine (*Pinus contorta* dougl.) distribution in the Pacific Northwest under climate change. Climatic Change. 105(1–2): 313–328. doi:10.1007/s10584-010-9861-2.

**Coops, N.C.; Waring, R.H. 2011b.** Estimating the vulnerability of fifteen tree species under changing climate in northwest North America. Ecological Modelling. 222(13): 2119–2129. doi:10.1016/j.ecolmodel.2011.03.033.

**Coops, N.C.; Waring, R.H.; Law, B.E. 2005.** Assessing the past and future distribution and productivity of ponderosa pine in the Pacific Northwest using a process model, 3-PG. Ecological Modelling. 183(1): 107–124. doi:10.1016/j.ecolmodel.2004.08.002.

**Cordero, E.C.; Kessomkiat, W.; Abatzoglou, J.; Mauget, S.A. 2011.** The identification of distinct patterns in California temperature trends. Climatic Change. 108(1–2): 357–382. doi:10.1007/s10584-011-0023-y.

**Creutzburg, M.K.; Scheller, R.M.; Lucash, M.S.; Evers, L.B.; Leduc, S.D.; Johnson, M.G. 2016.** Bioenergy harvest, climate change, and forest carbon in the Oregon Coast Range. GCB Bioenergy. 8(2): 357–370. doi:10.1111/gcbb.12255.

**Creutzburg, M.K.; Scheller, R.M.; Lucash, M.S.; Leduc, S.D.; Johnson, M.G. 2017.** Forest management scenarios in a changing climate: trade-offs between carbon, timber, and old forest. Ecological Applications. 27(2): 503–518. doi:10.1002/eap.1460.

**Crookston, N.L.; Rehfeldt, G.E.; Dixon, G.E.; Weiskittel, A.R. 2010.** Addressing climate change in the forest vegetation simulator to assess impacts on landscape forest dynamics. Forest Ecology and Management. 260(7): 1198–1211. doi:10.1016/j.foreco.2010.07.013.

**Cwynar, L.C. 1987.** Fire and the forest history of the North Cascade Range. Ecology. 68: 971–802.

**Dale, V.H.; Joyce, L.A.; McNulty, S.; Neilson, R.P. 2000.** The interplay between climate change, forests, and disturbances. Science of the Total Environment. 262(3): 201–204. doi:10.1016/s0048-9697(00)00522-2.

**Dale, V.H.; Joyce, L.A.; McNulty, S.; Neilson, R.P.; Ayres, M.P.; Flannigan, M.D.; Hanson, P.J.; Irland, L.C.; Lugo, A.E.; Peterson, C.J.; Simberloff, D.; Swanson, F.J.; Stocks, B.J.; Michael Wotton, B. 2001.** Climate change and forest disturbances. BioScience. 51(9): 723. doi:10.1641/0006-3568(2001)051[0723:ccafd]2.0.co;2.

**Dale, V.H.; Lugo, A.E.; Macmahon, J.A.; Pickett, S.T.A. 1998.** Ecosystem management in the context of large, infrequent disturbances. Ecosystems. 1(6): 546–557. doi:10.1007/s100219900050.

**Dalton, M.; Mote, P.W.; Snover, A.K. 2013.** Climate change in the Northwest: implications for our landscapes, waters, and communities. Washington, DC: Island Press. 224 p.

**Daly, C.; Conklin, D.R.; Unsworth, M.H. 2009.** Local atmospheric decoupling in complex topography alters climate change impacts. International Journal of Climatology. 30: 1857–1864. doi:10.1002/joc.2007.

**Daly, C.; Conklin, D.R.; Unsworth, M.H. 2010.** Local atmospheric decoupling in complex topography alters climate change impacts. International Journal of Climatology. 30: 1857–1864.

**Daly, C.; Halbleib, M.; Smith, J.I.; Gibson, W.P.; Doggett, M.K.; Taylor, G.H.; Curtis, J.; Pasteris, P.P. 2008.** Physiographically sensitive mapping of climatological temperature and precipitation across the conterminous United States. International Journal of Climatology. 28(15): 2031–2064. doi:10.1002/joc.1688.

**Damschen, E.I.; Harrison, S.; Grace, J.B. 2010.** Climate change effects on an endemic-rich edaphic flora: resurveying Robert H. Whittaker's Siskiyou sites (Oregon, USA). Ecology. 91(12): 3609–3619. doi:10.1890/09-1057.1.

**Daniels, M.L.; Anderson, R.S.; Whitlock, C. 2005.** Vegetation and fire history since the late Pleistocene from the Trinity Mountains, northwestern California, USA. The Holocene. 15: 1062–1071.

**Das, T.; Maurer, E.P.; Pierce, D.W.; Dettinger, M.D.; Cayan, D.R. 2013.** Increases in flood magnitudes in California under warming climates. Journal of Hydrology. 501: 101–110. doi:10.1016/j.jhydrol.2013.07.042.

**Davis, M.B.; Shaw, R.G. 2001.** Range shifts and adaptive responses to quaternary climate change. Science. 292(5517): 673–679. doi:10.1126/science.292.5517.673.

**Davis, R.J.; Ohmann, J.L.; Kennedy, R.E.; Cohen, W.B.; Gregory, M.J.; Yang, Z.; Roberts, H.M.; Gray, A.N.; Spies; T.A. 2015.** Northwest Forest Plan—the first 20 years (1994–2013): status and trends of late-successional and old-growth forests. Gen. Tech. Rep. PNW-GTR-911. Portland, OR: U.S. Department of Agriculture, Forest Service, Pacific Northwest Research Station. 112 p.

**Davis, R.; Yang, Z.; Yost, A.; Belongie, C.; Cohen, W. 2017.** The normal fire environment—modeling environmental suitability for large forest wildfires using past, present, and future climate normals. Forest Ecology and Management. 390: 173–186. doi:10.1016/j.foreco.2017.01.027.

**De Frenne, P.; Rodriguez-Sánchez, F.; Coomes, D.A.; Baeten, L.; Verstraeten, G.; Vellend, M.; Bernhardt-Römermann, M.; Brown, C.D.; Brunet, J.; Cornelis, J.; Decocq, G.M.; Dierschke, H.; Eriksson, O.; Gilliam, F.S.; Hédl, R.; Heinken, T.; Hermy, M.; Hommel, P.; Jenkins, M.A.; Kelly, D.L.; Kirby, K.J.; Mitchell, F.J.G.; Naaf, T.; Newman, M.; Peterken, G.; Petřík, P.; Schultz, J.; Sonnier, G.; Van Calster, H.; Waller, D.M.; Walther, G.R.; White, P.S.; Woods, K.D.; Wulf, M.; Graae, B.J.; Verheyen, K. 2013.** Microclimate moderates plant responses to macroclimate warming. Proceedings of the National Academy of Sciences of the United States of America. 110(46): 18561–18565. doi:10.1073/pnas.1311190110.

**DellaSala, D.A.; Brandt, P.; Koopman, M.; Leonard, J.; Meisch, C.; Herzog, P.; Alaback, P.; Goldstein, M.I.; Jovan, S.; MacKinnon, A.; Von Wehrden, H. 2015.** Climate change may trigger broad shifts in North America's Pacific coastal rainforests. In: Reference Module in Earth Systems and Environmental Sciences. Elsevier ScienceDirect. https://doi.org/10.1016/B978-0-12-409548-9.09367-2. (20 November 2017).

**Depro, B.M.; Murray, B.C.; Alig, R.J.; Shanks, A. 2008.** Public land, timber harvests, and climate mitigation: quantifying carbon sequestration potential on U.S. public timberlands. Forest Ecology and Management. 255(3–4): 1122–1134. doi:10.1016/j.foreco.2007.10.036.

**Deser, C.; Knutti, R.; Solomon, S.; Phillips, A.S. 2012.** Communication of the role of natural variability in future North American climate. Nature Climate Change. 2(11): 775–779. doi:10.1038/nclimate1562.

**Deser, C.; Phillips, A.S.; Alexander, M.A.; Smoliak, B.V. 2014.** Projecting North American climate over the next 50 years: uncertainty due to internal variability. Journal of Climate. 27(6): 2271–2296. doi:10.1175/jcli-d-13-00451.1.

**Dettinger, M. 2011.** Climate change, atmospheric rivers, and floods in California—a multimodel analysis of storm frequency and magnitude changes. Journal of the American Water Resources Association. 47(3): 514–523. doi:10.1111/j.1752-1688.2011.00546.x.

**Dickman, A.; Cook, S. 1989.** Fire and fungus in a mountain hemlock forest. Canadian Journal of Botany. 67(7): 2005–2016. doi:10.1139/b89-254.

**Diffenbaugh, N.S.; Swain, D.L.; Touma, D. 2015.** Anthropogenic warming has increased drought risk in California. Proceedings of the National Academy of Sciences of the United States of America. 112(13): 3931–3936. doi:10.1073/pnas.1422385112.

**Dobrowski, S.Z. 2011.** A climatic basis for microrefugia: the influence of terrain on climate. Global Change Biology. 17(2): 1022–1035. doi:10.1111/j.1365-2486.2010.02263.x.

**Dobrowski, S.Z.; Abatzoglou, J.; Swanson, A.K.; Greenberg, J.A.; Mynsberge, A.R.; Holden, Z.A.; Schwartz, M.K. 2013.** The climate velocity of the contiguous United States during the 20th century. Global Change Biology. 19(1): 241–251. doi:10.1111/gcb.12026.

**Dodson, E.K.; Root, H.T. 2015.** Native and exotic plant cover vary inversely along a climate gradient 11 years following stand-replacing wildfire in a dry coniferous forest, Oregon, USA. Global Change Biology. 21(2): 666–675. doi:10.1111/gcb.12775.

**Donato, D.C.; Harvey, B.J.; Romme, W.H.; Simard, M.; Turner, M.G. 2013.** Bark beetle effects on fuel profiles across a range of stand structures in Douglas-fir forests of greater Yellowstone. Ecological Applications. 23(1): 3–20. doi:10.1890/12-0772.1.

**Elliott, M.; Edmonds, R.L.; Mayer, S. 2002.** Role of fungal diseases in decline of Pacific madrone. Northwest Science. 76: 293–303.

**Elsner, M.M.; Cuo, L.; Voisin, N.; Deems, J.S.; Hamlet, A.F.; Vano, J.A.; Mickelson, K.E.B.; Lee, S.-Y.; Lettenmaier, D.P. 2010.** Implications of 21st century climate change for the hydrology of Washington state. Climatic Change. 102(1–2): 225–260. doi:10.1007/s10584-010-9855-0.

**Ettinger, A.K.; Ford, K.R.; HilleRisLambers, J. 2011.** Climate determines upper, but not lower, altitudinal range limits of Pacific Northwest conifers. Ecology. 92(6): 1323–1331. doi:10.1890/10-1639.1.

**Ettinger, A.K.; HilleRisLambers, J. 2013.** Climate isn't everything: competitive interactions and variation by life stage will also affect range shifts in a warming world. American Journal of Botany. 100(7): 1344–1355. doi:10.3732/ajb.1200489.

**Everett, R.L.; Schellhaas, R.; Keenum, D.; Spurbeck, D.; Ohlson, P. 2000.** Fire history in the ponderosa pine/Douglas-fir forests on the east slope of the Washington Cascades. Forest Ecology and Management. 129(1–3): 207–225. doi:10.1016/s0378-1127(99)00168-1.

**Fahnestock, G.R.; Agee, J.K. 1983.** Biomass consumption and smoke production by prehistoric and modern forest fires in western Washington. Journal of Forestry. 81: 653–657.

**Finkral, A.J.; Evans, A.M. 2008.** The effects of a thinning treatment on carbon stocks in a northern Arizona ponderosa pine forest. Forest Ecology and Management. 255(7): 2743–2750. doi:10.1016/j.foreco.2008.01.041.

**Finney, M.A.; Martin, R.E. 1989.** Fire history in a *Sequoia sempervirens* forest at Salt Point State Park, California. Canadian Journal of Forest Research. 19(11): 1451–1457. doi:10.1139/x89-221.

**Ford, K.R.; Breckheimer, I.K.; Franklin, J.F.; Freund, J.A.; Kroiss, S.J.; Larson, A.J.; Theobald, E.J.; Hillerislambers, J. 2017.** Competition alters tree growth responses to climate at individual and stand scales. Canadian Journal of Forest Research. 47(1): 53–62. doi:10.1139/cjfr-2016-0188.

**Ford, K.R.; Ettinger, A.K.; Lundquist, J.D.; Raleigh, M.S.; HilleRisLambers, J. 2013.** Spatial heterogeneity in ecologically important climate variables at coarse and fine scales in a high-snow mountain landscape. PLoS One. 8(6): e65008. doi:10.1371/journal.pone.0065008.

**Ford, K.R.; Harrington, C.A.; Bansal, S.; Gould, P.J.; St. Clair, J.B. 2016.** Will changes in phenology track climate change? A study of growth initiation timing in coast Douglas-fir. Global Change Biology. 22(11): 3712–3723. doi:10.1111/gcb.13328.

**Foster, D.R.; Knight, D.H.; Franklin, J.F. 1998.** Landscape patterns and legacies resulting from large, infrequent forest disturbances. Ecosystems. 1(6): 497–510. doi:10.1007/s100219900046.

**Franklin, J.F.; Dyrness, C.T. 1973.** Natural vegetation of Oregon and Washington. Gen. Tech. Rep. PNW-GTR-8. Portland, OR: U.S. Department of Agriculture, Forest Service, Pacific Northwest Forest and Range Experiment Station. 427 p.

**Franklin, J.F.; Swanson, F.J.; Harmon, M.E.; Perry, D.A.; Spies, T.A.; Dale, V.H.; McKee, A.; Ferrell, W.K.; Means, J.E.; Gregory, S.V.; Lattin, J.D.; Schowalter, T.D.; Larson, D. 1991.** Effects of global climatic change on forests in northwestern North America. Northwest Environmental Journal. 7: 233–254.

**Frey, S.J.K.; Hadley, A.S.; Johnson, S.L.; Schulze, M.; Jones, J.A.; Betts, M.G. 2016.** Spatial models reveal the microclimatic buffering capacity of old-growth forests. Science Advances. 2(4): e1501392-e1501392. doi:10.1126/sciadv.1501392.

**Fried, J.S.; Torn, M.S.; Mills, E. 2004.** The impact of climate change on wildfire severity: a regional forecast for northern California. Climatic Change. 64(1/2): 169–191. doi:10.1023/b:clim.0000024667.89579.ed.

**Garfin, G.; Franco, G.; Blanco, H.; Comrie, A.; Gonzalez, P.; Piechota, T.; Smyth, R.; Waskom, R. 2014.** Southwest. In: Melillo, J.M.; Richmond, T.; Yohe, G.W., eds. Climate change impacts in the United States: the third national climate assessment. Washington, DC: U.S. Global Change Research Program: 462–486. Chapter 20. doi:10.7930/J08G8HMN.

**Garza, E.S. 1995.** Fire history and fire regimes of East Humbug and Scorpion Creeks and their relation to the range of *Pinus lambertiana* Dougl. Corvallis, OR: Oregon State University. 75 p. M.S. thesis.

**Gavin, D.G.; Fitzpatrick, M.C.; Gugger, P.F.; Heath, K.D.; Rodríguez-Sánchez, F.; Dobrowski, S.Z.; Hampe, A.; Hu, F.S.; Ashcroft, M.B.; Bartlein, P.J.; Blois, J.L.; Carstens, B.C.; Davis, E.B.; De Lafontaine, G.; Edwards, M.E.; Fernandez, M.; Henne, P.D.; Herring, E.M.; Holden, Z.A.; Kong, W.-S.; Liu, J.; Magri, D.; Matzke, N.J.; McGlone, M.S.; Saltré, F.; Stigall, A.L.; Tsai, Y.-H.E.; Williams, J.W. 2014.** Climate refugia: joint inference from fossil records, species distribution models and phylogeography. New Phytologist. 204(1): 37–54. doi:10.1111/nph.12929.

**Gavin, D.G.; McLachlan, J.S.; Brubaker, L.B.; Young, K.A. 2001.** Post-glacial history of subalpine forests, Olympic Peninsula, Washington, USA. The Holocene. 11(2): 177–188.

**Gedalof, Z.E.; Berg, A.A. 2010.** Tree ring evidence for limited direct $CO_2$ fertilization of forests over the 20$^{th}$ century. Global Biogeochemical Cycles. 24(3): GB3027. doi:10.1029/2009gb003699.

**Gedalof, Z.E.; Peterson, D.L.; Mantua, N.J. 2005.** Atmospheric, climatic, and ecological controls on extreme wildfire years in the northwestern United States. Ecological Applications. 15(1): 154–174. doi:10.1890/03-5116.

**Gedalof, Z.E.; Smith, D.J. 2001.** Interdecadal climate variability and regime-scale shifts in Pacific North America. Geophysical Research Letters. 28(8): 1515–1518. doi:10.1029/2000gl011779.

**Gershunov, A.; Barnett, T.P.; Cayan, D.R. 1999.** North Pacific interdecadal oscillation seen as factor in ENSO-related North American climate anomalies. Eos, Transactions American Geophysical Union. 80(3): 25. doi:10.1029/99eo00019.

**Gershunov, A.; Guirguis, K. 2012.** California heat waves in the present and future. Geophysical Research Letters. 39(18): L18710. doi:10.1029/2012gl052979.

**Goheen, E.M.; Goheen, D.J. 2014.** Status of sugar and western white pines on federal forest lands in southwest Oregon: inventory query and natural stand survey results. Report No. SWWOFIDSC-14-01. Portland, OR: U.S. Department of Agriculture, Forest Service, Pacific Northwest Region. 71 p.

**Goheen, E.M.; Goheen, D.J.; Marshall, K.; Danchok, R.S.; Petrick, J.A.; White, D.E. 2002.** The status of whitebark pine along the Pacific Crest National Scenic Trail on the Umpqua National Forest. Gen. Tech. Rep. PNW-GTR-530. Portland, OR: U.S. Department of Agriculture, Forest Service, Pacific Northwest Research Station. 21 p. https://www.fs.usda.gov/treesearch/pubs/29520. (4 December 2017).

**Goheen, E.M.; Willhite, E.A. 2006.** Field guide to the common diseases and insect pests of Oregon and Washington conifers. Portland, OR: U.S. Department of Agriculture, Forest Service, Pacific Northwest Region. 325 p.

**Gray, A. 2008.** Monitoring and assessment of regional impacts from nonnative invasive plants in forests of the Pacific coast, United States. In: Jose, S.; Singh, H.P.; Batish, D.R.; Kohli, R.K., eds. Invasive plants and forest ecosystems. Boca Raton, FL: CRC Press: 217–235. doi:10.1201/9781420043389.ch13.

**Gray, A.N. 2005.** Eight nonnative plants in western Oregon forests: associations with environment and management. Environmental Monitoring and Assessment. 100(1–3): 109–127. doi:10.1007/s10661-005-7060-9.

**Gray, A.N.; Barndt, K.; Reichard, S.H. 2011.** Nonnative invasive plants of Pacific coast forests: a field guide for identification. Gen. Tech. Rep. PNW-GTR-817. Portland, OR: U.S. Department of Agriculture, Forest Service, Pacific Northwest Research Station. 91 p. https://www.fs.fed.us/pnw/pubs/pnw_gtr817.pdf. (4 December 2017).

**Gray, A.N.; Whittier, T.R. 2014.** Carbon stocks and changes on Pacific Northwest national forests and the role of disturbance, management, and growth. Forest Ecology and Management. 328: 167–178. doi:10.1016/j.foreco.2014.05.015.

**Gray, A.N.; Whittier, T.R.; Harmon, M.E. 2016.** Carbon stocks and accumulation rates in Pacific Northwest forests: role of stand age, plant community, and productivity. Ecosphere. 7(1): e01224. doi:10.1002/ecs2.1224.

**Griffin, D.; Anchukaitis, K.J. 2014.** How unusual is the 2012–2014 California drought? Geophysical Research Letters. 41(24): 9017–9023. doi:10.1002/2014gl062433.

**Grigg, L.; Whitlock, C. 1998.** Late-glacial vegetation and climate change in western Oregon. Quarternary Research. 49(3): 287–298.

**Haas, S.E.; Cushman, J.H.; Dillon, W.W.; Rank, N.E.; Rizzo, D.M.; Meentemeyer, R.K. 2015.** Effects of individual, community, and landscape drivers on the dynamics of a wildland forest epidemic. Ecology. doi:10.1890/15-0767.1.

**Halofsky, J.; Peterson, D. 2016.** Climate change vulnerabilities and adaptation options for forest vegetation management in the northwestern USA. Atmosphere. 7(3): 46. doi:10.3390/atmos7030046.

**Halofsky, J.; Peterson, D.; Metlen, K.; Myer, M.; Sample, V. 2016.** Developing and implementing climate change adaptation options in forest ecosystems: a case study in southwestern Oregon, USA. Forests. 7(11): 268. doi:10.3390/f7110268.

**Halofsky, J.E.; Hemstrom, M.A.; Conklin, D.R.; Halofsky, J.S.; Kerns, B.K.; Bachelet, D. 2013.** Assessing potential climate change effects on vegetation using a linked model approach. Ecological Modelling. 266: 131–143. doi:10.1016/j.ecolmodel.2013.07.003.

**Halofsky, J.E.; Peterson, D.L.; Prendeville, H.R. 2017.** Assessing vulnerabilities and adapting to climate change in northwestern U.S. forests. Climatic Change. doi:10.1007/s10584-017-1972-6.

**Halofsky, J.S.; Halofsky, J.E.; Burcsu, T.; Hemstrom, M.A. 2014.** Dry forest resilience varies under simulated climate-management scenarios in a central Oregon, USA landscape. Ecological Applications. 24(8): 1908–1925. doi:10.1890/13-1653.1.

**Hamlet, A.F.; Mote, P.W.; Clark, M.P.; Lettenmaier, D.P. 2005.** Effects of temperature and precipitation variability on snowpack trends in the western United States. Journal of Climate. 18(21): 4545–4561. doi:10.1175/jcli3538.1.

**Hampe, A.; Jump, A.S. 2011.** Climate relicts: past, present, future. Annual Review of Ecology, Evolution, and Systematics. 42(1): 313–333. doi:10.1146/annurev-ecolsys-102710-145015.

**Hampe, A.; Petit, R.J. 2005.** Conserving biodiversity under climate change: the rear edge matters. Ecology Letters. 8(5): 461–467. doi:10.1111/j.1461-0248.2005.00739.x.

**Hansen, E.M.; Goheen, D.J.; Jules, E.S.; Ullian, B. 2000a.** Managing Port-Orford-cedar and the introduced pathogen *Phytophthora lateralis*. Plant Disease. 84(1): 4–14. doi:10.1094/pdis.2000.84.1.4.

**Hansen, E.M.; Goheen, E.M. 2000.** *Phellinus weirii* and other native root pathogens as determinants of forest structure and process in western North America. Annual Review of Phytopathology. 38: 515–539.

**Hansen, E.M.; Stone, J.K.; Capitano, B.R.; Rosso, P.; Sutton, W.; Winton, L.; Kanaskie, A.; McWilliams, M.G. 2000b.** Incidence and impact of Swiss needle cast in forest plantations of Douglas-fir in coastal Oregon. Plant Disease. 84(7): 773–778. doi:10.1094/pdis.2000.84.7.773.

**Hargrove, W.W.; Hoffman, F.M. 2004.** Potential of multivariate quantitative methods for delineation and visualization of ecoregions. Environmental Management. 34(S1): S39–S60. doi:10.1007/s00267-003-1084-0.

**Harrington, C.A.; Gould, P.J. 2015.** Tradeoffs between chilling and forcing in satisfying dormancy requirements for Pacific Northwest tree species. Frontiers in Plant Science. 6: art120. doi:10.3389/fpls.2015.00120.

**Harrington, T.B.; Reichard, S.H. 2007.** Meeting the challenge: invasive plants in Pacific Northwest ecosystems. Gen. Tech. Rep. PNW-GTR-694. Portland, OR: U.S. Department of Agriculture, Forest Service, Pacific Northwest Research Station. 166 p. https://www.fs.fed.us/pnw/pubs/pnw_gtr694.pdf. (4 December 2017).

**Harrison, S.; Damschen, E.I.; Grace, J.B. 2010.** Ecological contingency in the effects of climatic warming on forest herb communities. Proceedings of the National Academy of Sciences of the United States of America. 107(45): 19362–19367. doi:10.1073/pnas.1006823107.

**Hart, S.J.; Schoennagel, T.; Veblen, T.T.; Chapman, T.B. 2015.** Area burned in the western United States is unaffected by recent mountain pine beetle outbreaks. Proceedings of the National Academy of Sciences of the United States of America. 112(14): 4375–4380. doi:10.1073/pnas.1424037112.

**Hartmann, H.; Adams, H.D.; Anderegg, W.R.L.; Jansen, S.; Zeppel, M.J.B. 2015.** Research frontiers in drought-induced tree mortality: crossing scales and disciplines. New Phytologist. 205(3): 965–969. doi:10.1111/nph.13246.

**Harvey, B.J.; Donato, D.C.; Romme, W.H.; Turner, M.G. 2013.** Influence of recent bark beetle outbreak on fire severity and postfire tree regeneration in montane Douglas-fir forests. Ecology. 94(11): 2475–2486. doi:10.1890/13-0188.1.

**Hawkins, E.; Sutton, R. 2009.** The potential to narrow uncertainty in regional climate predictions. Bulletin of the American Meteorological Society. 90(8): 1095–1107. doi:10.1175/2009bams2607.1.

**Hemstrom, M.A.; Franklin, J.F. 1982.** Fire and other disturbances of the forests in Mount Rainier National Park. Quaternary Research. 18(01): 32–51. doi:10.1016/0033-5894(82)90020-5.

**Hessburg, P.F.; Churchill, D.J.; Larson, A.J.; Haugo, R.D.; Miller, C.; Spies, T.A.; North, M.P.; Povak, N.A.; Belote, R.T.; Singleton, P.H.; Gaines, W.L.; Keane, R.E.; Aplet, G.H.; Stephens, S.L.; Morgan, P.; Bisson, P.A.; Rieman, B.E.; Salter, R.B.; Reeves, G.H. 2015.** Restoring fire-prone inland Pacific landscapes: seven core principles. Landscape Ecology. 30(10): 1805–1835. doi:10.1007/s10980-015-0218-0.

**Hessburg, P.F.; Salter, R.B.; Jame, K.M. 2007.** Re-examining fire severity relations in pre-management era mixed conifer forests: inferences from landscape patterns of forest structure. Landscape Ecology. 22: 5–24.

**Hessl, A.E. 2011.** Pathways for climate change effects on fire: models, data, and uncertainties. Progress in Physical Geography. 35(3): 393–407. doi:10.1177/0309133311407654.

**Hessl, A.E.; McKenzie, D.; Schellhaas, R. 2004.** Drought and Pacific decadal oscillation linked to fire occurrence in the inland Pacific Northwest. Ecological Applications. 14(2): 425–442. doi:10.1890/03-5019.

**Heyerdahl, E.K.; McKenzie, D.; Daniels, L.D.; Hessl, A.E.; Littell, J.S.; Mantua, N.J. 2008.** Climate drivers of regionally synchronous fires in the inland Northwest (1651–1900). International Journal of Wildland Fire. 17(1): 40. doi:10.1071/wf07024.

**Hicke, J.A.; Johnson, M.C.; Hayes, J.L.; Preisler, H.K. 2012.** Effects of bark beetle-caused tree mortality on wildfire. Forest Ecology and Management. 271: 81–90. doi:10.1016/j.foreco.2012.02.005.

**Hicke, J.A.; Meddens, A.J.H.; Kolden, C.A. 2016.** Recent tree mortality in the western United States from bark beetles and forest fires. Forest Science. 62(2): 141–153. doi:10.5849/forsci.15-086.

**HilleRisLambers, J.; Anderegg, L.D.L.; Breckheimer, I.; Burns, K.M.; Ettinger, A.K.; Franklin, J.F.; Freund, J.A.; Ford, K.R.; Krolss, S.J. 2015.** Implications of climate change for turnover in forest composition. Northwest Science. 89(3): 201–218. doi:10.3955/046.089.0304.

**Holah, J.C.; Wilson, M.V.; Hansen, E.M. 1997.** Impacts of a native root-rotting pathogen on successional development of old-growth Douglas fir forests. Oecologia. 111(3): 429–433. doi:10.1007/s004420050255.

**Hudiburg, T.; Law, B.; Turner, D.P.; Campbell, J.; Donato, D.; Duane, M. 2009.** Carbon dynamics of Oregon and northern California forests and potential land-based carbon storage. Ecological Applications. 19(1): 163–180. doi:10.1890/07-2006.1.

**Hurteau, M.; North, M. 2009.** Fuel treatment effects on tree-based forest carbon storage and emissions under modeled wildfire scenarios. Frontiers in Ecology and the Environment. 7(8): 409–414. doi:10.1890/080049.

**Hurteau, M.D.; Koch, G.W.; Hungate, B.A. 2008.** Carbon protection and fire risk reduction: toward a full accounting of forest carbon offsets. Frontiers in Ecology and the Environment. 6(9): 493–498. doi:10.1890/070187.

**Hurteau, M.D.; Liang, S.; Martin, K.L.; North, M.P.; Koch, G.W.; Hungate, B.A. 2016.** Restoring forest structure and process stabilizes forest carbon in wildfire-prone southwestern ponderosa pine forests. Ecological Applications. 26(2): 382–391. doi:10.1890/15-0337.

**Hurteau, M.D.; Stoddard, M.T.; Fulé, P.Z. 2011.** The carbon costs of mitigating high-severity wildfire in southwestern ponderosa pine. Global Change Biology. 17(4): 1516–1521. doi:10.1111/j.1365-2486.2010.02295.x.

**Ibáñez, I.; Clark, J.S.; Dietze, M.C.; Feeley, K.; Hersh, M.; Ladeau, S.; McBride, A.; Welch, N.E.; Wolosin, M.S. 2006.** Predicting biodiversity change: outside the climate envelope, beyond the species–area curve. Ecology. 87(8): 1896–1906. doi:10.1890/0012-9658(2006)87[1896:pbcotc]2.0.co;2.

**Impara, P.C. 1997.** Spatial and temporal patterns of fire in the forests of the central Oregon Coast Range. Corvallis, OR: Oregon State University. 354 p. Ph.D. dissertation.

**Intergovernmental Panel on Climate Change [IPCC]. 2014.** Climate change 2013—the physical science basis: contribution of working group I to the fifth assessment report of the Intergovernmental Panel on Climate Change. Cambridge, United Kingdom: Cambridge University Press. 1535 p.

**Jackson, S.T.; Betancourt, J.L.; Booth, R.K.; Gray, S.T. 2009.** Ecology and the ratchet of events: climate variability, niche dimensions, and species distributions. Proceedings of the National Academy of Sciences of the United States of America. 106(Supplement 2): 19685–19692. doi:10.1073/pnas.0901644106.

**Jentsch, A.; Kreyling, J.; Beierkuhnlein, C. 2007.** A new generation of climate-change experiments: events, not trends. Frontiers in Ecology and the Environment. 5(7): 365–374. doi:10.1890/1540-9295(2007)5[365:ANGOCE]2.0.CO;2.

**Johnstone, J.A.; Dawson, T.E. 2010.** Climatic context and ecological implications of summer fog decline in the coast redwood region. Proceedings of the National Academy of Sciences of the United States of America. 107(10): 4533–4538. doi:10.1073/pnas.0915062107.

**Johnstone, J.A.; Mantua, N.J. 2014a. Atmospheric controls on northeast Pacific temperature variability and change, 1900–2012.** Proceedings of the National Academy of Sciences of the United States of America. 111(40): 14360–14365. doi:10.1073/pnas.1318371111.

**Johnstone, J.A.; Mantua, N.J. 2014b. Reply to Abatzoglou et al.: atmospheric controls on northwest United States air temperatures, 1948–2012.** Proceedings of the National Academy of Sciences of the United States of America. 111(52): E5607–E5608. doi:10.1073/pnas.1421618112.

**Jolly, W.M.; Cochrane, M.A.; Freeborn, P.H.; Holden, Z.A.; Brown, T.J.; Williamson, G.J.; Bowman, D.M.J.S. 2015.** Climate-induced variations in global wildfire danger from 1979 to 2013. Nature Communications. 6: 7537. doi:10.1038/ncomms8537.

**Jules, E.S.; Kauffman, M.J.; Ritts, W.D.; Carroll, A.L. 2002.** Spread of an invasive pathogen over a variable landscape: a nonnative root rot on Port Orford cedar. Ecology. 83(11): 3167. doi:10.2307/3071851.

**Jules, E.S.; Steenbock, C.M.; Carroll, A.L. 2014.** Update on the 35-year expansion of the invasive root pathogen, *Phytophthora lateralis*, across a landscape of Port Orford cedar (*Chamaecyparis lawsoniana*). Forest Pathology. 45(2): 165–168. doi:10.1111/efp.12158.

**Jung, I.-W.; Chang, H. 2010.** Assessment of future runoff trends under multiple climate change scenarios in the Willamette river basin, Oregon, USA. Hydrologic Processes. 25(2): 258–277. doi:10.1002/hyp.7842.

**Keppel, G.; Van Niel, K.P.; Wardell-Johnson, G.W.; Yates, C.J.; Byrne, M.; Mucina, L.; Schut, A.G.T.; Hopper, S.D.; Franklin, S.E. 2012.** Refugia: identifying and understanding safe havens for biodiversity under climate change. Global Ecology and Biogeography. 21(4): 393–404. doi:10.1111/j.1466-8238.2011.00686.x.

**Kerns, B.K.; Kim, J.B.; Kline, J.D.; Day, M.A. 2016.** US exposure to multiple landscape stressors and climate change. Regional Environmental Change. 16(7): 2129–2140. doi:10.1007/s10113-016-0934-2.

**Keyser, A.; Westerling, A.L. 2017.** Climate drives inter-annual variability in probability of high severity fire occurrence in the western United States. Environmental Research Letters. 12(6): 065003. doi:10.1088/1748-9326/aa6b10.

**Killam, D.; Bui, A.; Ladochy, S.; Ramirez, P.; Willis, J.; Patzert, W. 2014.** California getting wetter to the north, drier to the south: natural variability or climate change? Climate. 2(3): 168–180. doi:10.3390/cli2030168.

**Kim, J. 2005.** A projection of the effects of the climate change induced by increased $CO_2$ on extreme hydrologic events in the western U.S. Climatic Change. 68(1–2): 153–168. doi:10.1007/s10584-005-4787-9.

**Kitzberger, T.; Brown, P.M.; Heyerdahl, E.K.; Swetnam, T.W.; Veblen, T.T. 2007.** Contingent Pacific-Atlantic Ocean influence on multicentury wildfire synchrony over western North America. Proceedings of the National Academy of Sciences of the United States of America. 104(2): 543–548. doi:10.1073/pnas.0606078104.

**Klausmeyer, K.R.; Shaw, M.R.; Mackenzie, J.B.; Cameron, D.R. 2011.** Landscape-scale indicators of biodiversity's vulnerability to climate change. Ecosphere. 2(8): art88. doi:10.1890/es11-00044.1.

**Kline, J.D.; Harmon, M.E.; Spies, T.A.; Morzillo, A.T.; Pabst, R.J.; McComb, B.C.; Schnekenburger, F.; Olsen, K.A.; Csuti, B.; Vogeler, J.C. 2016.** Evaluating carbon storage, timber harvest, and habitat possibilities for a western Cascades (USA) forest landscape. Ecological Applications. 26(7): 2044–2059. doi:10.1002/eap.1358.

**Klos, P.Z.; Link, T.E.; Abatzoglou, J.T. 2014.** Extent of the rain-snow transition zone in the western U.S. under historic and projected climate. Geophysical Research Letters. 41: 4560–4568. doi:10.1002/2014gl060500.

**Knapp, P.A.; Hadley, K.S. 2012.** A 300-year history of Pacific Northwest windstorms inferred from tree rings. Global and Planetary Change. 92–93: 257–266. doi:10.1016/j.gloplacha.2012.06.002.

**Knowles, N. 2015.** Trends in snow cover and related quantities at weather stations in the conterminous United States. Journal of Climate. 28(19): 7518–7528. doi:10.1175/jcli-d-15-0051.1.

**Knutson, K.C.; Pyke, D.A. 2008.** Western juniper and ponderosa pine ecotonal climate-growth relationships across landscape gradients in southern Oregon. Canadian Journal of Forest Research. 38(12): 3021–3032. doi:10.1139/x08-142.

**Kolb, T.E.; Fettig, C.J.; Ayres, M.P.; Bentz, B.J.; Hicke, J.A.; Mathiasen, R.; Stewart, J.E.; Weed, A.S. 2016a.** Observed and anticipated impacts of drought on forest insects and diseases in the United States. Forest Ecology and Management. 380: 321–334. doi:10.1016/j.foreco.2016.04.051.

**Kolb, T.E.; Grady, K.C.; McEttrick, M.P.; Herrero, A. 2016b.** Local-scale drought adaptation of ponderosa pine seedlings at habitat ecotones. Forest Science. 62(6): 641–651. doi:10.5849/forsci.16-049.

**Krapek, J.; Buma, B. 2015.** Yellow-cedar: climate change and natural history at odds. Frontiers in Ecology and the Environment. 13(5): 280–281. doi:10.1890/1540-9295-13.5.280.

**Krawchuk, M.A.; Moritz, M.A.; Parisien, M.-A.; Van Dorn, J.; Hayhoe, K. 2009.** Global pyrogeography: the current and future distribution of wildfire. PLoS One. 4(4): e5102. doi:10.1371/journal.pone.0005102.

**Kroiss, S.J.; HilleRisLambers, J. 2015.** Recruitment limitation of long-lived conifers: implications for climate change responses. Ecology. 96(5): 1286–1297. doi:10.1890/14-0595.1.

**Krosby, M.; Tewksbury, J.; Haddad, N.M.; Hoekstra, J. 2010.** Ecological connectivity for a changing climate. Conservation Biology. 24(6): 1686–1689. doi:10.1111/j.1523-1739.2010.01585.x.

**Kueppers, L.M.; Snyder, M.A.; Sloan, L.C.; Zavaleta, E.S.; Fulfrost, B. 2005.** Modeled regional climate change and California endemic oak ranges. Proceedings of the National Academy of Sciences of the United States of America. 102(45): 16281–16286. doi:10.1073/pnas.0501427102.

**Kusnierczyk, E.; Ettl, G.J. 2002.** Growth response of ponderosa pine (*Pinus ponderosa*) to climate in the eastern Cascade Mountains, Washington, U.S.A.: implications for climatic change. Écoscience. 9(4): 544–551. doi:10.1080/11956860.2002.11682742.

**Ladeau, S.L.; Clark, J.S. 2001.** Rising $CO_2$ levels and the fecundity of forest trees. Science. 292(5514): 95–98. doi:10.1126/science.1057547.

**Ladeau, S.L.; Clark, J.S. 2006.** Elevated $CO_2$ and tree fecundity: the role of tree size, interannual variability, and population heterogeneity. Global Change Biology. 12: 822–833. doi:10.1111/j.1365-2486.2006.01137.x

**Larson, A.J.; Franklin, J.F. 2010.** The tree mortality regime in temperate old-growth coniferous forests: the role of physical damage. Canadian Journal of Forest Research. 40(11): 2091–2103. doi:10.1139/x10-149.

**Larson, A.J.; Lutz, J.A.; Donato, D.C.; Freund, J.A.; Swanson, M.E.; Hillerislambers, J.; Sprugel, D.G.; Franklin, J.F. 2015.** Spatial aspects of tree mortality strongly differ between young and old-growth forests. Ecology. 96(11): 2855–2861. doi:10.1890/15-0628.1.

**Latta, G.; Temesgen, H.; Adams, D.; Barrett, T. 2010.** Analysis of potential impacts of climate change on forests of the United States Pacific Northwest. Forest Ecology and Management. 259(4): 720–729. doi:10.1016/j.foreco.2009.09.003.

**Law, B.E.; Harmon, M.E. 2011.** Forest sector carbon management, measurement and verification, and discussion of policy related to climate change. Carbon Management. 2(1): 73–84. doi:10.4155/cmt.10.40.

**Law, B.E.; Turner, D.; Campbell, J.; Sun, O.J.; Van Tuyl, S.; Ritts, W.D.; Cohen, W.B. 2004.** Disturbance and climate effects on carbon stocks and fluxes across western Oregon USA. Global Change Biology. 10(9): 1429–1444. doi:10.1111/j.1365-2486.2004.00822.x.

**Law, B.E.; Waring, R.H. 2015.** Carbon implications of current and future effects of drought, fire and management on Pacific Northwest forests. Forest Ecology and Management. 355: 4–14. doi:10.1016/j.foreco.2014.11.023.

**Lee, S.-Y.; Ryan, M.E.; Hamlet, A.F.; Palen, W.J.; Lawler, J.J.; Halabisky, M. 2015.** Projecting the hydrologic impacts of climate change on montane wetlands. PLoS One. 10(9): e0136385. doi:10.1371/journal.pone.0136385.

**Leiberg, J.B. 1903.** Southern part of the Cascade Range forest reserve. In: Langille, H.D.; Plummer, F.G.; Dodwell, A.; Rixon, T.F.; Leiberg, J.B.; Ganett, H., eds. Forest conditions in the Cascade Range forest reserve, Oregon. U.S. Geological Survey Professional Paper No. 9, Series H, Forestry, 6. Washington, DC: Government Printing Office: 229–289.

**Lenihan, J.M.; Bachelet, D.; Neilson, R.P.; Drapek, R. 2008.** Response of vegetation distribution, ecosystem productivity, and fire to climate change scenarios for California. Climatic Change. 87(S1): 215–230. doi:10.1007/s10584-007-9362-0.

**Lintz, H.E.; Gray, A.N.; Yost, A.; Sniezko, R.; Woodall, C.; Reilly, M.; Hutten, K.; Elliott, M. 2016.** Quantifying density-independent mortality of temperate tree species. Ecological Indicators. 66: 1–9. doi:10.1016/j.ecolind.2015.11.011.

**Littell, J.S.; McKenzie, D.; Kerns, B.K.; Cushman, S.; Shaw, C.G. 2011.** Managing uncertainty in climate-driven ecological models to inform adaptation to climate change. Ecosphere. 2(9): art102. doi:10.1890/es11-00114.1.

**Littell, J.S.; McKenzie, D.; Peterson, D.L.; Westerling, A.L. 2009.** Climate and wildfire area burned in western U.S. ecoprovinces, 1916–2003. Ecological Applications. 19(4): 1003–1021. doi:10.1890/07-1183.1.

**Littell, J.S.; Oneil, E.E.; McKenzie, D.; Hicke, J.A.; Lutz, J.A.; Norheim, R.A.; Elsner, M.M. 2010.** Forest ecosystems, disturbance, and climatic change in Washington state, USA. Climatic Change. 102(1–2): 129–158. doi:10.1007/s10584-010-9858-x.

**Littell, J.S.; Peterson, D.L.; Riley, K.L.; Liu, Y.; Luce, C.H. 2016.** A review of the relationships between drought and forest fire in the United States. Global Change Biology. 22(7): 2353–2369. doi:10.1111/gcb.13275.

**Littell, J.S.; Peterson, D.L.; Tjoelker, M. 2008.** Douglas-fir growth in mountain ecosystems: water limits tree growth from stand to region. Ecological Monographs. 78(3): 349–368. doi:10.1890/07-0712.1.

**Littlefield, C.E.; McRae, B.H.; Michalak, J.L.; Lawler, J.J.; Carroll, C. 2017.** Connecting today's climates to future climate analogs to facilitate movement of species under climate change. Conservation Biology. 31(6): 1397–1408. doi:10.1111/cobi.12938.

**Liu, Y.; Goodrick, S.L.; Stanturf, J.A. 2013.** Future U.S. wildfire potential trends projected using a dynamically downscaled climate change scenario. Forest Ecology and Management. 294: 120–135. doi:10.1016/j.foreco.2012.06.049.

**Loarie, S.R.; Carter, B.E.; Hayhoe, K.; McMahon, S.; Moe, R.; Knight, C.A.; Ackerly, D.D. 2008.** Climate change and the future of California's endemic flora. PLoS One. 3(6): e2502. doi:10.1371/journal.pone.0002502.

**Long, C.J.; Whitlock, C.; Bartlein, P.J.; Millspaugh, S.H. 1998.** A 9000-year fire history from the Oregon Coast Range, based on a high-resolution charcoal study. Canadian Journal of Forest Research. 28(5): 774–787. doi:10.1139/x98-051.

**Loudermilk, E.L.; Stanton, A.; Scheller, R.M.; Dilts, T.E.; Weisberg, P.J.; Skinner, C.; Yang, J. 2014.** Effectiveness of fuel treatments for mitigating wildfire risk and sequestering forest carbon: a case study in the Lake Tahoe basin. Forest Ecology and Management. 323: 114–125. doi:10.1016/j.foreco.2014.03.011.

**Luo, Y.; Chen, H.Y.H. 2013.** Observations from old forests underestimate climate change effects on tree mortality. Nature Communications. 4: 1655. doi:10.1038/ncomms2681.

**Lutz, J.A.; Halpern, C.B. 2006.** Tree mortality during early forest development: a long-term study of rates, causes, and consequences. Ecological Monographs. 76(2): 257–275. doi:10.1890/0012-9615(2006)076[0257:tmdefd]2.0.co;2.

**Lydersen, J.M.; North, M.P.; Collins, B.M. 2014.** Severity of an uncharacteristically large wildfire, the rim fire, in forests with relatively restored frequent fire regimes. Forest Ecology and Management. 328: 326–334. doi:10.1016/j.foreco.2014.06.005.

**Lynn, B.H.; Healy, R.; Druyan, L.M. 2009.** Quantifying the sensitivity of simulated climate change to model configuration. Climatic Change. 92(3–4): 275–298. doi:10.1007/s10584-008-9494-x.

**Lynott, R.E.; Cramer, O.P. 1966.** Detailed analysis of the 1962 Columbus Day windstorm in Oregon and Washington. Monthly Weather Review. 94(2): 105–117. doi:10.1175/1520-0493(1966)094<0105:daotcd>2.3.co;2.

**Manion, P.D. 1981.** Tree disease concepts. Englewood Cliffs, NJ: Prentice-Hall. 416 p.

**Mann, M.E.; Gleick, P.H. 2015.** Climate change and California drought in the 21st century. Proceedings of the National Academy of Sciences of the United States of America. 112(13): 3858–3859. doi: 10.1073/pnas.1503667112.

**Manning, M.R.; Edmonds, J.; Emori, S.; Grubler, A.; Hibbard, K.; Joos, F.; Kainuma, M.; Keeling, R.F.; Kram, T.; Manning, A.C.; Meinshausen, M.; Moss, R.; Nakicenovic, N.; Riahi, K.; Rose, S.K.; Smith, S.; Swart, R.; Van Vuuren, D.P. 2010.** Misrepresentation of the IPCC $CO_2$ emission scenarios. Nature Geoscience. 3(6): 376–377. doi:10.1038/ngeo880.

**Manter, D.K.; Reeser, P.W.; Stone, J.K. 2005.** A climate-based model for predicting geographic variation in Swiss needle cast severity in the Oregon Coast Range. Phytopathology. 95: 1256–1265.

**Mantua, N.J.; Hare, S.R.; Zhang, Y.; Wallace, J.M.; Francis, R.C. 1997.** A Pacific interdecadal climate oscillation with impacts on salmon production. Bulletin of the American Meteorological Society. 78(6): 1069–1079. doi:10.1175/1520-0477(1997)078<1069:apicow>2.0.co;2.

**Marias, D.E.; Meinzer, F.C.; Woodruff, D.R.; Shaw, D.C.; Voelker, S.L.; Brooks, J.R.; Lachenbruch, B.; Falk, K.; McKay, J. 2014.** Impacts of dwarf mistletoe on the physiology of host *Tsuga heterophylla* trees as recorded in tree-ring C and O stable isotopes. Tree Physiology. 34(6): 595–607. doi:10.1093/treephys/tpu046.

**Marlon, J.R.; Bartlein, P.J.; Gavin, D.G.; Long, C.J.; Anderson, R.S.; Briles, C.E.; Brown, K.J.; Colombaroli, D.; Hallett, D.J.; Power, M.J.; Scharf, E.A.; Walsh, M.K. 2012.** Long-term perspective on wildfires in the western USA. Proceedings of the National Academy of Sciences of the United States of America. 109(9): E535–E543. doi:10.1073/pnas.1112839109.

**Marlon, J.R.; Bartlein, P.J.; Walsh, M.K.; Harrison, S.P.; Brown, K.J.; Edwards, M.E.; Higuera, P.E.; Power, M.J.; Anderson, R.S.; Briles, C.; Brunelle, A.; Carcaillet, C.; Daniels, M.; Hu, F.S.; Lavoie, M.; Long, C.; Minckley, T.; Richard, P.J.H.; Scott, A.C.; Shafer, D.S.; Tinner, W.; Umbanhowar, C.E.; Whitlock, C. 2009.** Wildfire responses to abrupt climate change in North America. Proceedings of the National Academy of Sciences of the United States of America. 106(8): 2519–2524. doi:10.1073/pnas.0808212106.

**Marris, E. 2009.** Forestry: planting the forest of the future. Nature. 459(7249): 906–908. doi:10.1038/459906a.

**Mass, C.; Dotson, B. 2010.** Major extratropical cyclones of the northwest United States: historical review, climatology, and synoptic environment. Monthly Weather Review. 138(7): 2499–2527. doi:10.1175/2010mwr3213.1.

**Mathys, A.S.; Coops, N.C.; Waring, R.H. 2016.** An ecoregion assessment of projected tree species vulnerabilities in western North America through the 21st century. Global Change Biology. 23(2): 920–932. doi:10.1111/gcb.13440.

**McAfee, S.A. 2014.** Consistency and the lack thereof in Pacific decadal oscillation impacts on North American winter climate. Journal of Climate. 27(19): 7410–7431. doi:10.1175/jcli-d-14-00143.1.

**McCabe, G.J.; Dettinger, M.D. 1999.** Decadal variations in the strength of ENSO teleconnections with precipitation in the western United States. International Journal of Climatology. 19(13): 1399–1410. doi:10.1002/(sici)1097-0088(19991115)19:13<1399::aid-joc457>3.0.co;2-a.

**McDowell, N.; Brooks, J.R.; Fitzgerald, S.A.; Bond, B.J. 2003.** Carbon isotope discrimination and growth response of old *Pinus ponderosa* trees to stand density reductions. Plant, Cell and Environment. 26(4): 631–644. doi:10.1046/j.1365-3040.2003.00999.x.

**McDowell, N.G.; Adams, H.D.; Bailey, J.D.; Hess, M.; Kolb, T.E. 2006.** Homeostatic maintenance of ponderosa pine gas exchange in response to stand density changes. Ecological Applications. 16(3): 1164–1182. doi:10.1890/1051-0761(2006)016[1164:hmoppg]2.0.co;2.

**McDowell, N.; Pockman, W.T.; Allen, C.D.; Breshears, D.D.; Cobb, N.; Kolb, T.; Plaut, J.; Sperry, J.; West, A.; Williams, D.G.; Yepez, E.A. 2008.** Mechanisms of plant survival and mortality during drought: Why do some plants survive while others succumb to drought? New Phytologist. 178: 719–739.

**McKenney, D.W.; Pedlar, J.H.; Lawrence, K.; Campbell, K.; Hutchinson, M.F. 2007.** Potential impacts of climate change on the distribution of North American trees. BioScience. 57(11): 939. doi:10.1641/b571106.

**McKenney, D.W.; Pedlar, J.H.; Rood, R.B.; Price, D. 2011.** Revisiting projected shifts in the climate envelopes of North American trees using updated general circulation models. Global Change Biology. 17(8): 2720–2730. doi:10.1111/j.1365-2486.2011.02413.x.

**McKenzie, D.; Gedalof, Z.E.; Peterson, D.L.; Mote, P. 2004.** Climatic change, wildfire, and conservation. Conservation Biology. 18(4): 890–902. doi:10.1111/j.1523-1739.2004.00492.x.

**McKenzie, D.; Hessl, A.E.; Peterson, D.L. 2001.** Recent growth of conifer species of western North America: assessing spatial patterns of radial growth trends. Canadian Journal of Forest Research. 31(3): 526–538. doi:10.1139/x00-191.

**McKenzie, D.; Littell, J.S. 2017.** Climate change and the eco-hydrology of fire: will area burned increase in a warming western USA? Ecological Applications. 27: 26–36. doi:10.1002/eap.1420.

**McKenzie, D.; Shankar, U.; Keane, R.E.; Stavros, E.N.; Heilman, W.E.; Fox, D.G.; Riebau, A.C. 2014.** Smoke consequences of new wildfire regimes driven by climate change. Earth's Future. 2: 35–59. doi:10.1002/2013EF000180.

**McRae, B.H.; Popper, K.; Jones, A.; Schindel, M.; Buttrick, S.; Hall, K.; Unnasch, R.S.; Platt, J. 2016.** Conserving nature's stage: mapping omnidirectional connectivity for resilient terrestrial landscapes in the Pacific Northwest. Portland, OR: The Nature Conservancy. 47 p. http://nature.org/resilienceNW. (30 June 2016).

**Means, J.E. 1982.** Development history of dry coniferous forests in the central western Cascade Range of Oregon. In: Means, J.E., ed. Forest succession and stand development research in the Northwest. Proceedings of the symposium of the Northwest Scientific Association. Corvallis, OR: Oregon State University, School of Forestry, Forest Resource Laboratory: 142–158.

**Meehl, G.A.; Tebaldi, C. 2004.** More intense, more frequent, and longer lasting heat waves in the 21st century. Science. 305(5686): 994–997. doi:10.1126/science.1098704.

**Meentemeyer, R.K.; Cunniffe, N.J.; Cook, A.R.; Filipe, J.A.N.; Hunter, R.D.; Rizzo, D.M.; Gilligan, C.A. 2011.** Epidemiological modeling of invasion in heterogeneous landscapes: spread of sudden oak death in California (1990–2030). Ecosphere. 2(2): art17. doi:10.1890/es10-00192.1.

**Meigs, G.W.; Kennedy, R.E.; Gray, A.N.; Gregory, M.J. 2015.** Spatiotemporal dynamics of recent mountain pine beetle and western spruce budworm outbreaks across the Pacific Northwest region, USA. Forest Ecology and Management. 339: 71–86. doi:10.1016/j.foreco.2014.11.030.

**Meigs, G.W.; Zald, H.S.J.; Campbell, J.L.; Keeton, W.S.; Kennedy, R.E. 2016.** Do insect outbreaks reduce the severity of subsequent forest fires? Environmental Research Letters. 11(4): 045008. doi:10.1088/1748-9326/11/4/045008.

**Metz, M.R.; Frangioso, K.M.; Meentemeyer, R.K.; Rizzo, D.M. 2011.** Interacting disturbances: wildfire severity affected by stage of forest disease invasion. Ecological Applications. 21(2): 313–320. doi:10.1890/10-0419.1.

**Mildrexler, D.; Yang, Z.; Cohen, W.B.; Bell, D.M. 2016.** A forest vulnerability index based on drought and high temperatures. Remote Sensing of Environment. 173: 314–325. doi:10.1016/j.rse.2015.11.024.

**Millar, C.I.; Stephenson, N.L.; Stephens, S.L. 2007.** Climate change and forests of the future: managing in the face of uncertainty. Ecological Applications. 17(8): 2145–2151. doi:10.1890/06-1715.1.

**Miller, J.D.; Skinner, C.N.; Safford, H.D.; Knapp, E.E.; Ramirez, C.M. 2012.** Trends and causes of severity, size, and number of fires in northwestern California, USA. Ecological Applications. 22(1): 184–203. doi:10.1890/10-2108.1.

**Mitchell, R.G.; Buffam, P.E. 2001.** Patterns of long-term balsam woolly adelgid infestations and effects in Oregon and Washington. Western Journal of Applied Forestry. 16: 121–126.

**Mitchell, S.R.; Harmon, M.E.; O'Connell, K.E.B. 2009.** Forest fuel reduction alters fire severity and long-term carbon storage in three Pacific Northwest ecosystems. Ecological Applications. 19(3): 643–655. doi:10.1890/08-0501.1.

**Mohr, J.A.; Whitlock, C.; Skinner, C.N. 2000.** Postglacial vegetation and fire history, eastern Klamath Mountains, California, USA. The Holocene. 10(4): 587–601.

**Monleon, V.J.; Lintz, H.E. 2015.** Evidence of tree species' range shifts in a complex landscape. PLoS One. 10(1): e0118069. doi:10.1371/journal.pone.0118069.

**Morelli, T.L.; Daly, C.; Dobrowski, S.Z.; Dulen, D.M.; Ebersole, J.L.; Jackson, S.T.; Lundquist, J.D.; Millar, C.I.; Maher, S.P.; Monahan, W.B.; Nydick, K.R.; Redmond, K.T.; Sawyer, S.C.; Stock, S.; Beissinger, S.R. 2016.** Managing climate change refugia for climate adaptation. PLoS One. 11(8): e0159909. doi:10.1371/journal.pone.0159909.

**Morrison, P.H.; Swanson, F.J. 1990.** Fire history and pattern in a Cascade Range landscape. Gen. Tech. Rep. PNW-GTR-254. Portland, OR: U.S. Department of Agriculture, Forest Service, Pacific Northwest Research Station. 77 p. https://www.fs.fed.us/pnw/pubs/gtr254/gtr254a.pdf. (4 December 2017).

**Morrow, R.J. 1985.** Age structure and spatial pattern of old-growth ponderosa pine in Pringle Falls Experimental Forest, central Oregon. Corvallis, OR: Oregon State University. 80 p. M.S. thesis.

**Mortenson, L.A.; Gray, A.N.; Shaw, D.C. 2015.** A forest health inventory assessment of red fir (*Abies magnifica*) in upper montane California. Écoscience. 22(1): 47–58. doi:10.1080/11956860.2015.1047142.

**Mote, P.; Snover, A.K.; Capalbo, S.; Eigenbrode, S.D.; Glick, P.; Littell, J.; Raymondi, R.; Reeder, S. 2014.** Northwest. In: Melillo, J.M.; Richmond, T.; Yohe, G.W., eds. Climate change impacts in the United States: the third national climate assessment. Washington, DC: U.S. Global Change Research Program: 487–513. Chapter 21. doi:10.7930/J04Q7RWX.

**Mote, P.W. 2003.** Trends in temperature and precipitation in the Pacific Northwest during the 20th century. Northwest Science. 77: 271–282.

**Mote, P.W. 2006.** Climate-driven variability and trends in mountain snowpack in western North America. Journal of Climate. 19(23): 6209–6220. doi:10.1175/jcli3971.1.

**Mote, P.W.; Hamlet, A.F.; Clark, M.P.; Lettenmaier, D.P. 2005.** Declining mountain snowpack in western North America. Bulletin of the American Meteorological Society. 86(1): 39–49. doi:10.1175/bams-86-1-39.

**Nakawatase, J.M.; Peterson, D.L. 2006.** Spatial variability in forest growth—climate relationships in the Olympic mountains, Washington. Canadian Journal of Forest Research. 36(1): 77–91. doi:10.1139/x05-224.

**Neelin, J.D.; Langenbrunner, B.; Meyerson, J.E.; Hall, A.; Berg, N. 2013.** California winter precipitation change under global warming in the coupled model intercomparison project phase 5 ensemble. Journal of Climate. 26(17): 6238–6256. doi:10.1175/jcli-d-12-00514.1.

**Neill, A.R.; Puettmann, K.J. 2013.** Managing for adaptive capacity: thinning improves food availability for wildlife and insect pollinators under climate change conditions. Canadian Journal of Forest Research. 43(5): 428–440. doi:10.1139/cjfr-2012-0345.

**Nelson, D.B.; Abbott, M.B.; Steinman, B.; Polissar, P.J.; Stansell, N.D.; Ortiz, J.D.; Rosenmeier, M.F.; Finney, B.P.; Riedel, J. 2011.** Drought variability in the Pacific Northwest from a 6,000-yr lake sediment record. Proceedings of the National Academy of Sciences of the United States of America. 108(10): 3870–3875. doi:10.1073/pnas.1009194108.

**Newman, M.; Alexander, M.A.; Ault, T.R.; Cobb, K.M.; Deser, C.; Di Lorenzo, E.; Mantua, N.J.; Miller, A.J.; Minobe, S.; Nakamura, H.; Schneider, N.; Vimont, D.J.; Phillips, A.S.; Scott, J.D.; Smith, C.A. 2016.** The Pacific Decadal Oscillation, revisited. Journal of Climate. 29(12): 4399–4427. doi:10.1175/jcli-d-15-0508.1.

**North, M.; Hurteau, M.; Innes, J. 2009.** Fire suppression and fuels treatment effects on mixed-conifer carbon stocks and emissions. Ecological Applications. 19(6): 1385–1396. doi:10.1890/08-1173.1.

**North, M.P.; Hurteau, M.D. 2011.** High-severity wildfire effects on carbon stocks and emissions in fuels treated and untreated forest. Forest Ecology and Management. 261(6): 1115–1120. doi:10.1016/j.foreco.2010.12.039.

**Noss, R.F. 2001.** Beyond Kyoto: forest management in a time of rapid climate change. Conservation Biology. 15(3): 578–590. doi:10.1046/j.1523-1739.2001.015003578.x.

**Nuñez, T.A.; Lawler, J.J.; McRae, B.H.; Pierce, D.J.; Krosby, M.B.; Kavanagh, D.M.; Singleton, P.H.; Tewksbury, J.J. 2013.** Connectivity planning to address climate change. Conservation Biology. 27(2): 407–416. doi:10.1111/cobi.12014.

**Olson, D.; DellaSala, D.A.; Noss, R.F.; Strittholt, J.R.; Kass, J.; Koopman, M.E.; Allnutt, T.F. 2012.** Climate change refugia for biodiversity in the Klamath-Siskiyou ecoregion. Natural Areas Journal. 32(1): 65–74. doi:10.3375/043.032.0108.

**Olson, D.L.; Agee, J.K. 2005.** Historical fires in Douglas-fir dominated riparian forests of the southern Cascades, Oregon. Fire Ecology. 1(1): 50–74. doi:10.4996/fireecology.0101050.

**Paine, R.T.; Tegner, M.J.; Johnson, E.A. 1998.** Compounded perturbations yield ecological surprises. Ecosystems. 1: 535–545.

**Parendes, L.A.; Jones, J.A. 2000.** Role of light availability and dispersal in exotic plant invasion along roads and streams in the H.J. Andrews Experimental Forest, Oregon. Conservation Biology. 14(1): 64–75. doi:10.1046/j.1523-1739.2000.99089.x.

**Parks, S.A.; Holsinger, L.M.; Miller, C.; Nelson, C.R. 2015.** Wildland fire as a self-regulating mechanism: the role of previous burns and weather in limiting fire progression. Ecological Applications. 25(6): 1478–1492. doi:10.1890/14-1430.1.

**Parks, S.A.; Miller, C.; Abatzoglou, J.T.; Holsinger, L.M.; Parisien, M.-A.; Dobrowski, S.Z. 2016.** How will climate change affect wildland fire severity in the western US? Environmental Research Letters. 11(3): 035002. doi:10.1088/1748-9326/11/3/035002.

**Parks, S.A.; Miller, C.; Nelson, C.R.; Holden, Z.A. 2014.** Previous fires moderate burn severity of subsequent wildland fires in two large western US wilderness areas. Ecosystems. 17(1): 29–42. doi:10.1007/s10021-013-9704-x.

**Parks, S.A.; Miller, C.; Parisien, M.-A.; Holsinger, L.M.; Dobrowski, S.Z.; Abatzoglou, J. 2015.** Wildland fire deficit and surplus in the western United States, 1984–2012. Ecosphere. 6(12): art275. doi:10.1890/es15-00294.1.

**Pasalodos-Tato, M.; Mäkinen, A.; Garcia-Gonzalo, J.; Borges, J.G.; Lämås, T.; Eriksson, L.O. 2013.** Assessing uncertainty and risk in forest planning and decision support systems: review of classical methods and introduction of new approaches. Forest Systems. 22(2): 282. doi:10.5424/fs/2013222-03063.

**Peterson, D.W.; Kerns, B.K.; Dodson, E.K. 2014a.** Climate change effects on vegetation in the Pacific Northwest: a review and synthesis of the scientific literature and simulation model projections. Gen. Tech. Rep. PNW-GTR-900. Portland, OR: U.S. Department of Agriculture, Forest Service, Pacific Northwest Research Station. 183 p. doi:10.2737/pnw-gtr-900.

**Peterson, D.W.; Peterson, D.L. 2001.** Mountain hemlock growth responds to climatic variability at annual and decadal time scales. Ecology. 82(12): 3330–3345. doi:10.1890/0012-9658(2001)082[3330:mhgrtc]2.0.co;2.

**Peterson, D.W.; Peterson, D.L.; Ettl, G.J. 2002.** Growth responses of subalpine fir to climatic variability in the Pacific Northwest. Canadian Journal of Forest Research. 32(9): 1503–1517. doi:10.1139/x02-072.

**Peterson, E.; Hansen, E.; Hulbert, J. 2014b.** Source or sink? The role of soil and water borne inoculum in the dispersal of *Phytophthora ramorum* in Oregon tanoak forests. Forest Ecology and Management. 322: 48–57. doi:10.1016/j.foreco.2014.02.031.

**Pickett, S.T.A.; White, P.S. 1985.** The ecology of natural disturbance and patch dynamics. New York: Academic Press. 472 p.

**Pierce, D.W.; Barnett, T.P.; Santer, B.D.; Gleckler, P.J. 2009.** Selecting global climate models for regional climate change studies. Proceedings of the National Academy of Sciences of the United States of America. 106(21): 8441–8446. doi:10.1073/pnas.0900094106.

**Powers, J.S.; Sollins, P.; Harmon, M.E.; Jones, J.A. 1999.** Plant-pest interactions in time and space: a Douglas-fir bark beetle outbreak as a case study. Landscape Ecology. 14(2): 105–120. doi:10.1023/a:1008017711917.

**Preisler, H.K.; Hicke, J.A.; Ager, A.A.; Hayes, J.L. 2012.** Climate and weather influences on spatial patterns and population growth rates for mountain pine beetle, an outbreak insect. Ecology. 93: 2421–2434.

**Prichard, S.J.; Gedalof, Z.; Oswald, W.W. 2009.** Holocene fire and vegetation dynamics in a montane forest, North Cascade Range, Washington, USA. Quarternary Research. 72(1): 57–67.

**Prichard, S.J.; Peterson, D.L.; Jacobson, K. 2010.** Fuel treatments reduce the severity of wildfire effects in dry mixed conifer forest, Washington, USA. Canadian Journal of Forest Research. 40(8): 1615–1626. doi:10.1139/x10-109.

**Raffa, K.F.; Aukema, B.H.; Bentz, B.J.; Carroll, A.L.; Hicke, J.A.; Turner, M.G.; Romme, W.H. 2008.** Cross-scale drivers of natural disturbances prone to anthropogenic amplification: the dynamics of bark beetle eruptions. BioScience. 58(6): 501–517. doi:10.1641/b580607.

**Rapacciuolo, G.; Maher, S.P.; Schneider, A.C.; Hammond, T.T.; Jabis, M.D.; Walsh, R.E.; Iknayan, K.J.; Walden, G.K.; Oldfather, M.F.; Ackerly, D.D.; Beissinger, S.R. 2014.** Beyond a warming fingerprint: individualistic biogeographic responses to heterogeneous climate change in California. Global Change Biology. 20(9): 2841–2855. doi:10.1111/gcb.12638.

**Raymond, C.L.; McKenzie, D. 2012.** Carbon dynamics of forests in Washington, USA: 21st century projections based on climate-driven changes in fire regimes. Ecological Applications. 22(5): 1589–1611. doi:10.1890/1051-0761-22.5.1589.

**Rehfeldt, G.E.; Crookston, N.L.; Warwell, M.V.; Evans, J.S. 2006.** Empirical analyses of plant-climate relationships for the western United States. International Journal of Plant Sciences. 167(6): 1123–1150. doi:10.1086/507711.

**Reichard, S.H.; White, P. 2001.** Horticulture as a pathway of invasive plant introductions in the United States. BioScience. 51(2): 103. doi:10.1641/0006-3568(2001)051[0103:haapoi]2.0.co;2.

**Reilly, M.J.; Dunn, C.J.; Meigs, G.W.; Spies, T.A.; Kennedy, R.E.; Bailey, J.D.; Briggs, K. 2017.** Contemporary patterns of fire extent and severity in forests of the Pacific Northwest, USA (1985–2010). Ecosphere. 8(3): e01695. doi:10.1002/ecs2.1695.

**Reilly, M.J.; Spies, T.A. 2015.** Regional variation in stand structure and development in forests of Oregon, Washington, and inland northern California. Ecosphere. 6(10): art192. doi:10.1890/es14-00469.1.

**Reilly, M.J.; Spies, T.A. 2016.** Disturbance, tree mortality, and implications for contemporary regional forest change in the Pacific Northwest. Forest Ecology and Management. 374: 102–110. doi:10.1016/j.foreco.2016.05.002.

**Reinhardt, E.D.; Keane, R.E.; Calkin, D.E.; Cohen, J.D. 2008.** Objectives and considerations for wildland fuel treatment in forested ecosystems of the interior western United States. Forest Ecology and Management. 256(12): 1997–2006. doi:10.1016/j.foreco.2008.09.016.

**Restaino, C.M.; Peterson, D.L.; Littell, J. 2016.** Increased water deficit decreases Douglas fir growth throughout western US forests. Proceedings of the National Academy of Sciences of the United States of America. 113(34): 9557–9562. doi:10.1073/pnas.1602384113.

**Restaino, J.C.; Peterson, D.L. 2013.** Wildfire and fuel treatment effects on forest carbon dynamics in the western United States. Forest Ecology and Management. 303: 46–60. doi:10.1016/j.foreco.2013.03.043.

**Ritóková, G.; Shaw, D.; Filip, G.; Kanaskie, A.; Browning, J.; Norlander, D. 2016.** Swiss needle cast in western Oregon Douglas-fir plantations: 20-year monitoring results. Forests. 7(8): 155. doi:10.3390/f7080155.

**Rizzo, D.M.; Garbelotto, M. 2003.** Sudden oak death: endangering California and Oregon forest ecosystems. Frontiers in Ecology and the Environment. 1(4): 197–204. doi:10.1890/1540-9295(2003)001[0197:sodeca]2.0.co;2.

**Robbins, D. 2005.** Temporal and spatial variability of historic fire frequency in the southern Willamette Valley foothills of Oregon. Corvallis, OR: Oregon State University. 112 p. M.S. thesis.

**Rogers, B.M.; Neilson, R.P.; Drapek, R.; Lenihan, J.M.; Wells, J.R.; Bachelet, D.; Law, B.E. 2011.** Impacts of climate change on fire regimes and carbon stocks of the U.S. Pacific Northwest. Journal of Geophysical Research. 116(G3). doi:10.1029/2011jg001695.

**Romme, W.H.; Everham, E.H.; Frelich, L.E.; Moritz, M.A.; Sparks, R.E. 1998.** Are large, infrequent disturbances qualitatively different from small, frequent disturbances? Ecosystems. 1(6): 524–534. doi:10.1007/s100219900048.

**Rosso, P.; Hansen, E.M. 2003.** Predicting Swiss needle cast disease distribution and severity in young Douglas-fir plantations in coastal Oregon. Phytopathology. 93: 790–798.

**Rubenstein, S.M.; Dechaine, J.M. 2015.** Native-nonnative seed dispersal and establishment along an interstate highway. Northwest Science. 89(4): 324–335. doi:10.3955/046.089.0403.

**Rupp, D.E.; Abatzoglou, J.T.; Hegewisch, K.C.; Mote, P.W. 2013.** Evaluation of CMIP5 20th century climate simulations for the Pacific Northwest USA. Journal of Geophysical Research: Atmospheres. 118(19): 10884–10906. doi:10.1002/jgrd.50843.

**Ryan, M.G.; Harmon, M.E.; Birdsey, R.A.; Giardina, C.P.; Heath, L.S.; Houghton, R.A.; Jackson, R.B.; McKinley, D.C.; Morrison, J.F.; Murray, B.C. 2010.** A synthesis of the science on forests and carbon for US forests. Issues in Ecology. 13(1): 16.

**Salathé, E.P.; Hamlet, A.F.; Mass, C.F.; Lee, S.-Y.; Stumbaugh, M.; Steed, R. 2014.** Estimates of twenty-first-century flood risk in the Pacific Northwest based on regional climate model simulations. Journal of Hydrometeorology. 15(5): 1881–1899. doi:10.1175/jhm-d-13-0137.1.

**Sandel, B.; Dangremond, E.M. 2012.** Climate change and the invasion of California by grasses. Global Change Biology. 18(1): 277–289. doi:10.1111/j.1365-2486.2011.02480.x.

**Saxon, E.; Baker, B.; Hargrove, W.; Hoffman, F.; Zganjar, C. 2005.** Mapping environments at risk under different global climate change scenarios. Ecology Letters. 8(1): 53–60. doi:10.1111/j.1461-0248.2004.00694.x.

**Scheller, R.M.; Domingo, J.B.; Sturtevant, B.R.; Williams, J.S.; Rudy, A.; Gustafson, E.J.; Mladenoff, D.J. 2007.** Design, development, and application of LANDIS-II, a spatial landscape simulation model with flexible temporal and spatial resolution. Ecological Modelling. 201(3–4): 409–419. doi:10.1016/j.ecolmodel.2006.10.009.

**Schoennagel, T.; Veblen, T.T.; Romme, W.H.; Sibold, J.S.; Cook, E.R. 2005.** ENSO and PDO variability affect drought-induced fire occurrence in Rocky Mountain subalpine forests. Ecological Applications. 15(6): 2000–2014. doi:10.1890/04-1579.

**Schwartz, R.E.; Gershunov, A.; Iacobellis, S.F.; Cayan, D.R. 2014.** North American west coast summer low cloudiness: broadscale variability associated with sea surface temperature. Geophysical Research Letters. 41(9): 3307–3314. doi:10.1002/2014gl059825.

**Sea, D.S.; Whitlock, C. 1995.** Postglacial vegetation and climate of the Cascade range, central Oregon. Quaternary Research. 43(03): 370–381. doi:10.1006/qres.1995.1043.

**Seidl, R.; Thom, D.; Kautz, M.; Martin-Benito, D.; Peltoniemi, M.; Vacchiano, G.; Wild, J.; Ascoli, D.; Petr, M.; Honkaniemi, J.; Lexer, M.J.; Trotsiuk, V.; Mairota, P.; Svoboda, M.; Fabrika, M.; Nagel, T.A.; Reyer, C.P.O. 2017.** Forest disturbances under climate change. Nature Climate Change. 7(6): 395–402. doi:10.1038/nclimate3303.

**Shafer, S.L.; Bartlein, P.J.; Gray, E.M.; Pelltier, R.T. 2015.** Projected future vegetation changes for the Northwest United States and southwest Canada at a fine spatial resolution using a dynamic global vegetation model. PLoS One. 10(10): e0138759. doi:10.1371/journal.pone.0138759.

**Shaw, D.C.; Oeaster, P.T.; Filip, G.M. 2009.** Managing insects and disease of Oregon conifers. Publication # EM 8980. Corvallis, OR: Oregon State University Extension Services.

**Sheehan, T.; Bachelet, D.; Ferschweiler, K. 2015.** Projected major fire and vegetation changes in the Pacific Northwest of the conterminous United States under selected CMIP5 climate futures. Ecological Modelling. 317: 16–29. doi:10.1016/j.ecolmodel.2015.08.023.

**Simard, M.; Romme, W.H.; Griffin, J.M.; Turner, M.G. 2011.** Do mountain pine beetle outbreaks change the probability of active crown fire in lodgepole pine forests? Ecological Monographs. 81(1): 3–24. doi:10.1890/10-1176.1.

**Simpson, M. 2013.** Modeled potential vegetation zones of Washington and Oregon. Bend, OR: U.S. Department of Agriculture, Forest Service, Pacific Northwest Region, Central Oregon Area Ecology and Forest Health Program. https://ecoshare.info/category/gis-data-vegzones/. (8 December 2017).

**Sinton, D.S.; Jones, J.A. 2002.** Extreme winds and windthrow in the western Columbia River gorge. Northwest Science. 76: 173–182.

**Slesak, R.A.; Harrington, T.B.; D'Amato, A.W. 2016.** Invasive Scotch broom alters soil chemical properties in Douglas-fir forests of the Pacific Northwest, USA. Plant and Soil. 398(1–2): 281–289. doi:10.1007/s11104-015-2662-7.

**Smithwick, E.A.H.; Harmon, M.E.; Remillard, S.M.; Acker, S.A.; Franklin, J.F. 2002.** Potential upper bounds of carbon stores in forests of the Pacific Northwest. Ecological Applications. 12(5): 1303–1317. doi:10.1890/1051-0761(2002)012[1303:pubocs]2.0.co;2.

**Soeriaatmadja, R.E. 1966.** Fire history of the ponderosa pine forests of the Warm Springs Indian Reservation. Corvallis, OR: Oregon State University. 123 p. Ph.D. dissertation. http://ir.library.oregonstate.edu/concern/graduate_thesis_or_dissertations/cc08hh857 (11 November 2017).

**Soule, P.T.; Knapp, P.A. 2006.** Radial growth rate increases in naturally occurring ponderosa pine trees: a late-20$^{th}$ century $CO_2$ fertilization effect? New Phytologist. 171(2): 379–390. doi:10.1111/j.1469-8137.2006.01746.x.

**Spies, T.A.; Franklin, J.F. 1989.** Gap characteristics and vegetation response in coniferous forests of the Pacific Northwest. Ecology. 70(3): 543–545. doi:10.2307/1940198.

**Spies, T.A.; Giesen, T.W.; Swanson, F.J.; Franklin, J.F.; Lach, D.; Johnson, K.N. 2010.** Climate change adaptation strategies for federal forests of the Pacific Northwest, USA: ecological, policy, and socio-economic perspectives. Landscape Ecology. 25(8): 1185–1199. doi:10.1007/s10980-010-9483-0.

**Spies, T.A.; White, E.; Ager, A.; Kline, J.D.; Bolte, J.P.; Platt, E.K.; Olsen, K.A.; Pabst, R.J.; Barros, A.M.G.; Bailey, J.D.; Charnley, S.; Morzillo, A.T.; Koch, J.; Steen-Adams, M.M.; Singleton, P.H.; Sulzman, J.; Schwartz, C.; Csuti, B. 2017.** Using an agent-based model to examine forest management outcomes in a fire-prone landscape in Oregon, USA. Ecology and Society. 22(1): 25. doi:10.5751/es-08841-220125.

**Spracklen, D.V.; Mickley, L.J.; Logan, J.A.; Hudman, R.C.; Yevich, R.; Flannigan, M.D.; Westerling, A.L. 2009.** Impacts of climate change from 2000 to 2050 on wildfire activity and carbonaceous aerosol concentrations in the western United States. Journal of Geophysical Research. 114(D20) doi:10.1029/2008jd010966.

**St. Clair, J.B.; Howe, G.T. 2007.** Genetic maladaptation of coastal Douglas-fir seedlings to future climates. Global Change Biology. 13(7): 1441–1454. doi:10.1111/j.1365-2486.2007.01385.x.

**Stavros, E.N.; Abatzoglou, J.T.; McKenzie, D.; Larkin, N.K. 2014.** Regional projections of the likelihood of very large wildland fires under a changing climate in the contiguous western United States. Climatic Change. 126(3–4): 455–468. doi:10.1007/s10584-014-1229-6.

**Steinman, B.A.; Abbott, M.B.; Mann, M.E.; Ortiz, J.D.; Feng, S.; Pompeani, D.P.; Stansell, N.D.; Anderson, L.; Finney, B.P.; Bird, B.W. 2014.** Ocean-atmosphere forcing of centennial hydroclimate variability in the Pacific Northwest. Geophysical Research Letters. 41(7): 2553–2560. doi:10.1002/2014gl059499.

**Steinman, B.A.; Abbott, M.B.; Mann, M.E.; Stansell, N.D.; Finney, B.P. 2012.** 1,500 year quantitative reconstruction of winter precipitation in the Pacific Northwest. Proceedings of the National Academy of Sciences of the United States of America. 109(29): 11619–11623. doi:10.1073/pnas.1201083109.

**Stephens, S.L.; Fry, D.L. 2005.** Fire history in coast redwood stands in the northeastern Santa Cruz Mountains, California. Fire Ecology. 1(1): 2–19. doi:10.4996/fireecology.0101002.

**Stephens, S.L.; Moghaddas, J.J.; Edminster, C.; Fiedler, C.E.; Haase, S.; Harrington, M.; Keeley, J.E.; Knapp, E.E.; McIver, J.D.; Metlen, K.; Skinner, C.N.; Youngblood, A. 2009.** Fire treatment effects on vegetation structure, fuels, and potential fire severity in western U.S. forests. Ecological Applications. 19(2): 305–320. doi:10.1890/07-1755.1.

**Stephenson, N. 1998.** Actual evapotranspiration and deficit: biologically meaningful correlates of vegetation distribution across spatial scales. Journal of Biogeography. 25(5): 855–870. doi:10.1046/j.1365-2699.1998.00233.x.

**Stewart, G.H. 1986.** Forest development in canopy openings in old-growth *Pseudotsuga* forests of the western Cascade Range, Oregon. Canadian Journal of Forest Research. 16(3): 558–568. doi:10.1139/x86-097.

**Stewart, I.T.; Cayan, D.R.; Dettinger, M.D. 2004.** Changes in snowmelt runoff timing in western North America under 'business as usual' climate change scenario. Climatic Change. 62(1–3): 217–232. doi:10.1023/b:clim.0000013702.22656.e8.

**Stewart, I.T.; Cayan, D.R.; Dettinger, M.D. 2005.** Changes toward earlier streamflow timing across western North America. Journal of Climate. 18(8): 1136–1155. doi:10.1175/jcli3321.1.

**Stewart, J.R.; Lister, A.M.; Barnes, I.; Dalen, L. 2010.** Refugia revisited: individualistic responses of species in space and time. Proceedings of the Royal Society B: Biological Sciences. 277(1682): 661–671. doi:10.1098/rspb.2009.1272.

**Stone, J.K.; Coop, L.B.; Manter, D.K. 2008.** Predicting effects of climate change on Swiss needle cast disease severity in Pacific Northwest forests. Canadian Journal of Plant Pathology. 30(2): 169–176. doi:10.1080/07060661.2008.10540533.

**Stuart, J.D. 1987.** Fire history of an old-growth forest of *Sequoia sempervirens* (Taxodiaceae) in Humboldt Redwoods State Park, California. Madrono. 34: 128–141.

**Stuart, J.D.; Salazar, L.A. 2000.** Fire history of white fir forests in the coastal mountains of northwestern California. Northwest Science. 74: 280–285.

**Sturrock, R.N.; Frankel, S.J.; Brown, A.V.; Hennon, P.E.; Kliejunas, J.T.; Lewis, K.J.; Worrall, J.J.; Woods, A.J. 2011.** Climate change and forest diseases. Plant Pathology. 60(1): 133–149. doi:10.1111/j.1365-3059.2010.02406.x.

**Suggitt, A.J.; Gillingham, P.K.; Hill, J.K.; Huntley, B.; Kunin, W.E.; Roy, D.B.; Thomas, C.D. 2011.** Habitat microclimates drive fine-scale variation in extreme temperatures. Oikos. 120(1): 1–8. doi:10.1111/j.1600-0706.2010.18270.x.

**Tague, C.; Grant, G.; Farrell, M.; Choate, J.; Jefferson, A. 2008.** Deep groundwater mediates streamflow response to climate warming in the Oregon Cascades. Climatic Change. 86(1–2): 189–210. doi:10.1007/s10584-007-9294-8.

**Taylor, A.H.; Skinner, C.N. 1998.** Fire history and landscape dynamics in a late-successional reserve, Klamath mountains, California, USA. Forest Ecology and Management. 111(2–3): 285–301. doi:10.1016/s0378-1127(98)00342-9.

**Taylor, A.H.; Skinner, C.N. 2003.** Spatial patterns and controls on historical fire regimes and forest structure in the Klamath Mountains. Ecological Applications. 13(3): 704–719. doi:10.1890/1051-0761(2003)013[0704:spacoh]2.0.co;2.

**Tebaldi, C.; Knutti, R. 2007.** The use of the multi-model ensemble in probabilistic climate projections. Philosophical Transactions of the Royal Society A: Mathematical, Physical and Engineering Sciences. 365(1857): 2053–2075. doi:10.1098/rsta.2007.2076.

**Teensma, P.D. 1987.** Fire history and fire regimes of the central western Cascades of Oregon. Eugene, OR: University of Oregon. 188 p. Ph.D. dissertation. https://andrewsforest.oregonstate.edu/publications/716. (17 November 2017).

**Teensma, P.D.A.; Rienstra, J.T.; Yeiter, M.A. 1991.** Preliminary reconstruction and analysis of change in forest stand age classes of the Oregon Coast Range from 1850 to 1940. Tech. Note. OR-9. Portland, OR: U.S. Department of the Interior, Bureau of Land Management, Oregon State Office. 9 p.

**Thuiller, W.; Albert, C.; Araújo, M.B.; Berry, P.M.; Cabeza, M.; Guisan, A.; Hickler, T.; Midgley, G.F.; Paterson, J.; Schurr, F.M.; Sykes, M.T.; Zimmermann, N.E. 2008.** Predicting global change impacts on plant species' distributions: future challenges. Perspectives in Plant Ecology, Evolution and Systematics. 9(3–4): 137–152. doi:10.1016/j.ppees.2007.09.004.

**Tohver, I.M.; Hamlet, A.F.; Lee, S.-Y. 2014.** Impacts of 21st-century climate change on hydrologic extremes in the Pacific Northwest region of North America. Journal of the American Water Resources Association. 50(6): 1461–1476. doi:10.1111/jawr.12199.

**Trouet, V.; Taylor, A.H.; Carleton, A.M.; Skinner, C.N. 2006.** Fire-climate interactions in forests of the American Pacific coast. Geophysical Research Letters. 33(18): L18704. doi:10.1029/2006gl027502.

**Turner, D.P.; Conklin, D.R.; Bolte, J.P. 2015.** Projected climate change impacts on forest land cover and land use over the Willamette River basin, Oregon, USA. Climatic Change. doi:10.1007/s10584-015-1465-4.

**van Mantgem, P.J.; Caprio, A.C.; Stevenson, N.L.; Das, A.J. 2016.** Does prescribed fire promote resistance to drought in low elevation forests of the Sierra Nevada, California, USA? Fire Ecology. 12(1): 13–25. doi:10.4996/fireecology.1201013.

**van Mantgem, P.J.; Sarr, D.A. 2015.** Structure, diversity, and biophysical properties of old-growth forests in the Klamath region, USA. Northwest Science. 89(2): 170–181. doi:10.3955/046.089.0208.

**van Mantgem, P.J.; Stephenson, N.L.; Byrne, J.C.; Daniels, L.D.; Franklin, J.F.; Fule, P.Z.; Harmon, M.E.; Larson, A.J.; Smith, J.M.; Taylor, A.H.; Veblen, T.T. 2009.** Widespread increase of tree mortality rates in the western United States. Science. 323(5913): 521–524. doi:10.1126/science.1165000.

**Van Norman, K.J. 1998.** Historical fire regime in the Little River watershed, southwestern Oregon. Corvallis, OR: Oregon State University. 103 p. M.S. thesis

**van Vuuren, D.P.; Edmonds, J.; Kainuma, M.; Riahi, K.; Thomson, A.; Hibbard, K.; Hurtt, G.C.; Kram, T.; Krey, V.; Lamarque, J.-F.; Masui, T.; Meinshausen, M.; Nakicenovic, N.; Smith, S.J.; Rose, S.K. 2011.** The representative concentration pathways: an overview. Climatic Change. 109(1–2): 5–31. doi:10.1007/s10584-011-0148-z.

**van Vuuren, D.P.; Edmonds, J.; Smith, S.J.; Calvin, K.V.; Karas, J.; Kainuma, M.; Nakicenovic, N.; Riahi, K.; Van Ruijven, B.J.; Swart, R.; Thomson, A. 2010.** What do near-term observations tell us about long-term developments in greenhouse gas emissions? Climatic Change. 103(3–4): 635–642. doi:10.1007/s10584-010-9940-4.

**Vos, C.C.; Berry, P.; Opdam, P.; Baveco, H.; Nijhof, B.; O'Hanley, J.; Bell, C.; Kuipers, H. 2008.** Adapting landscapes to climate change: examples of climate-proof ecosystem networks and priority adaptation zones. Journal of Applied Ecology. 45(6): 1722–1731. doi:10.1111/j.1365-2664.2008.01569.x.

**Vose, J.M.; Peterson, D.L.; Patel-Weynand, T. 2012.** Effects of climatic variability and change on forest ecosystems: a comprehensive science synthesis for the U.S. forest sector. Gen. Tech. Rep. PNW-GTR-870. Portland, OR: U.S. Department of Agriculture, Forest Service, Pacific Northwest Research Station. 265 p. https://www.fs.fed.us/psw/publications/pnw/pnw_gtr870_210.pdf. (6 December 2017).

**Walsh, M.K.; Marlon, J.R.; Goring, S.J.; Brown, K.J.; Gavin, D.G. 2015.** A regional perspective on Holocene fire-climate-human interactions in the Pacific Northwest of North America. Annals of the Association of American Geographers. 105(6): 1135–1157.

**Walsh, M.K.; Whitlock, C.; Bartlein, P.J. 2008.** A 14,300-year-long record of fire-vegetation-climate linkages at Battle Ground Lake, southwestern Washington. Quaternary Research. 70: 251–264.

**Wang, S.Y.S.; Yoon, J.-H.; Becker, E.; Gillies, R. 2017.** California from drought to deluge. Nature Climate Change. 7(7): 465–468. doi:10.1038/nclimate3330.

**Ward, K.; Shoal, R.; Aubry, C. 2006.** Whitebark pine in Washington and Oregon: a synthesis of current studies and historical data. Portland, OR: U.S. Department of Agriculture, Forest Service, Pacific Northwest Region. 29 p.

**Weaver, H. 1959.** Ecological changes in the ponderosa pine forest of the Warm Springs Indian Reservation in Oregon. Journal of Forestry. 57: 15–20.

**Weisberg, P.J. 2009.** Historical fire frequency on contrasting slope facets along the McKenzie River, western Oregon Cascades. Western North American Naturalist. 69(2): 206–214. doi:10.3398/064.069.0209.

**Weisberg, P.J.; Swanson, F.J. 2003.** Regional synchroneity in fire regimes of western Oregon and Washington, USA. Forest Ecology and Management. 172(1): 17–28. doi:10.1016/s0378-1127(01)00805-2.

**Wendel, R.; Zabowskl, D. 2010.** Fire history within the lower Elwha River watershed, Olympic National Park, Washington. Northwest Science. 84(1): 88–97. doi:10.3955/046.084.0109.

**West, N.E. 1969.** Tree patterns in central Oregon ponderosa pine forests. American Midland Naturalist. 81(2): 584. doi:10.2307/2423994.

**Westerling, A.L.; Bryant, B.P.; Preisler, H.K.; Holmes, T.P.; Hidalgo, H.G.; Das, T.; Shrestha, S.R. 2011.** Climate change and growth scenarios for California wildfire. Climatic Change. 109(S1): 445–463. doi:10.1007/s10584-011-0329-9.

**Westerling, A.L.; Hidalgo, H.G.; Cayan, D.R.; Swetnam, T.W. 2006.** Warming and earlier spring increase western U.S. forest wildfire activity. Science. 313(5789): 940–943. doi:10.1126/science.1128834.

**Wetzel, S.A.; Fonda, R.W. 2000.** Fire history of Douglas-fir forests in the Morse Creek drainage of Olympic National Park, Washington. Northwest Science. 74: 263–279.

**Whitlock, C. 1992.** Vegetational and climatic history of the Pacific Northwest during the last 20,000 years: implications for understanding present-day biodiversity. The Northwest Environmental Journal. 8: 5–28.

**Whitlock, C.; Bartlein, P.J.; Markgraf, V.; Ashworth, A.C. 2001.** The midlatitudes of North and South America during the last glacial maximum and early Holocene. In: Markgraf, V., ed. Interhemispheric climate linkages. New York: Academic Press: 391–416. doi:10.1016/b978-012472670-3/50024-0.

**Whitlock, C.; Marlon, J.; Briles, C.; Brunelle, A.; Long, C.; Bartlein, P. 2008.** Long-term relations among fire, fuel, and climate in the north-western US based on lake-sediment studies. International Journal of Wildland Fire. 17(1): 72. doi:10.1071/wf07025.

**Whitlock, C.; Shafer, S.L.; Marlon, J. 2003.** The role of climate and vegetation change in shaping past and future fire regimes in the northwestern US and the implications for ecosystem management. Forest Ecology and Management. 178(1–2): 5–21. doi:10.1016/s0378-1127(03)00051-3.

**Whittier, T.R.; Gray, A.N. 2016.** Tree mortality based fire severity classification for forest inventory surveys: a Pacific Northwest national forests example. Forest Ecology and Management. 359: 199–209.

**Williams, D.W.; Liebhold, A.M. 1995.** Forest defoliators and climatic change: potential changes in spatial distribution of outbreaks of western spruce budworm (Lepidoptera: Tortricidae) and gypsy moth (Lepidoptera: Lymantriidae). Environmental Entomology. 24(1): 1–9. doi:10.1093/ee/24.1.1.

**Wills, R.D.; Stuart, J.D. 1994.** Fire history and stand development of a Douglas-fir/hardwood forest in Northern California. Northwest Science. 68(3): 205–212.

**Wilson, B.T.; Woodall, C.W.; Griffith, D.M. 2013.** Forest carbon stocks of the contiguous United States (2000–2009). [Dataset] Newtown Square, PA: U.S. Department of Agriculture, Forest Service, Northern Research Station. http://dx.doi.org/10.2737/rds-2013-0004.

**Wimberly, M.C.; Cochrane, M.A.; Baer, A.D.; Pabst, K. 2009.** Assessing fuel treatment effectiveness using satellite imagery and spatial statistics. Ecological Applications. 19(6): 1377–1384. doi:10.1890/08-1685.1.

**Winthers, E.; Fallon, D.; Haglund, J.; Demeo, T.; Nowacki, G.; Tart, D.; Ferwerda, M.; Robertson, G.; Gallegos, A.; Rorick, A.; Cleland, D.T.; Robbie, W. 2005.** Terrestrial ecological unit inventory technical guide: landscape and land unit scales. Gen. Tech. Rep. WO-68. Washington, DC: U.S. Department of Agriculture, Forest Service, Ecosystem Management Coordination Staff. 254 p.

**Wise, E.K. 2010.** Spatiotemporal variability of the precipitation dipole transition zone in the western United States. Geophysical Research Letters. 37(7) doi:10.1029/2009gl042193.

**Wright, C.S.; Agee, J.K. 2004.** Fire and vegetation history in the eastern Cascade mountains, Washington. Ecological Applications. 14(2): 443–459. doi:10.1890/02-5349.

**Yamaguchi, D.K. 1986.** The development of old-growth Douglas-fir forests northeast of Mount St. Helens, Washington, following an A.D. 1480 eruption. Seattle, WA: University of Washington. 100 p. Ph.D. dissertation.

## 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 dead-wood 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

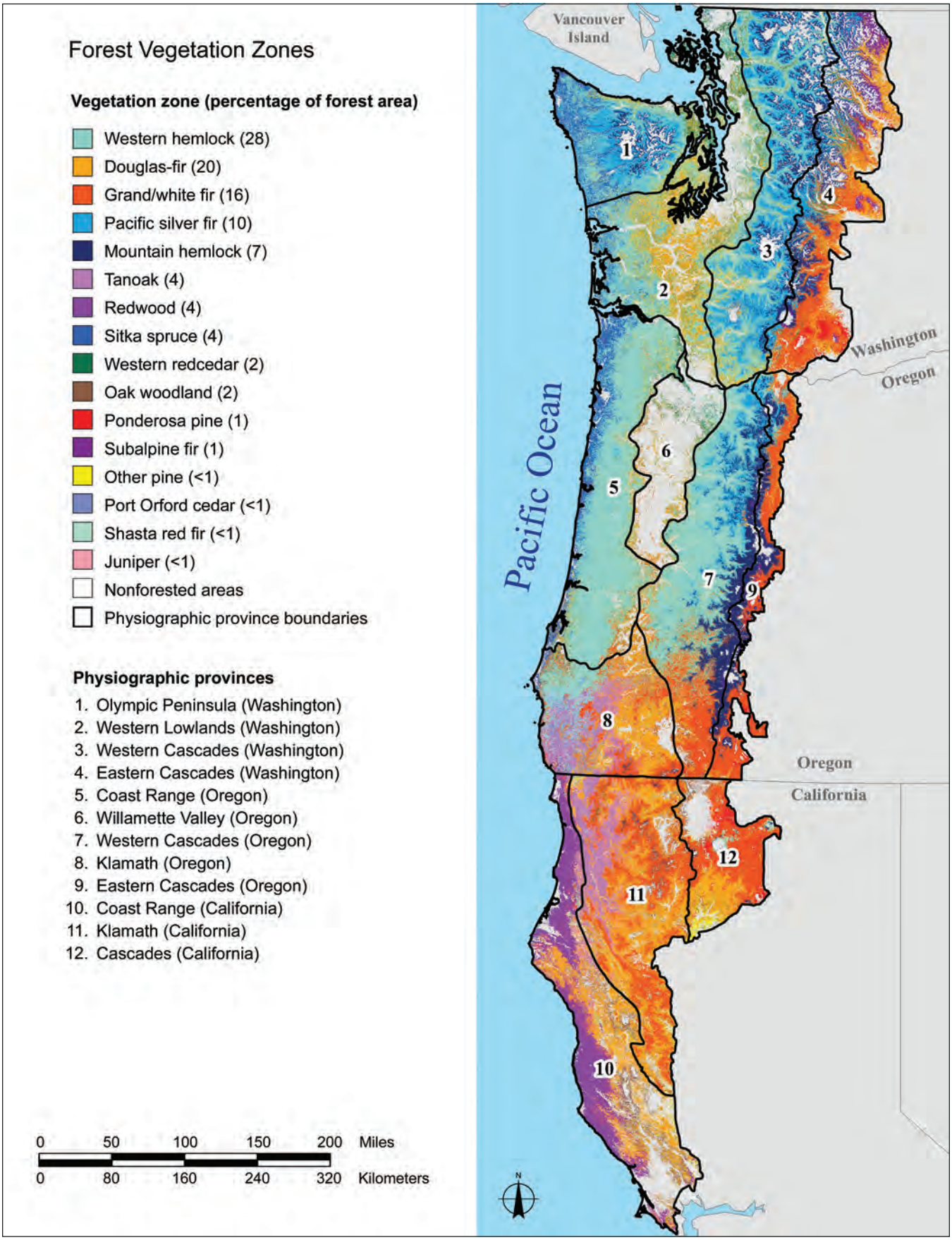



Figure 3-4—Geographic distribution of potential vegetation zones (aka vegetation types) (Simpson 2013) and physiographic provinces across the Northwest Forest Plan area.

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).

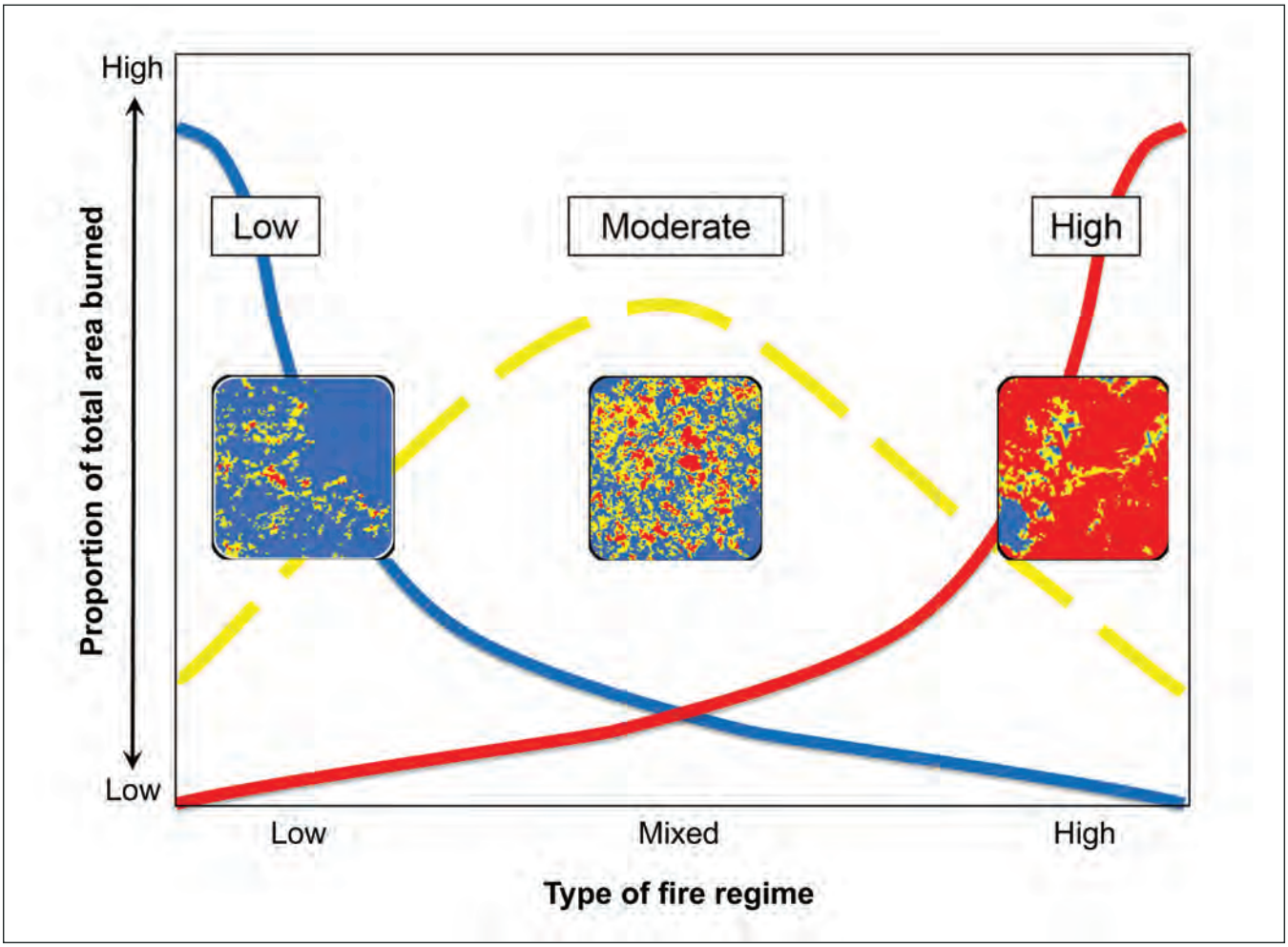



Figure 3-5—Conceptual diagram characterizing the proportions of low-, moderate-, and high-severity fires in three major fire regime classes. Inset panels represent idealized landscape dynamics associated with each regime based on proportions and size class distributions of patches at each of the three 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<br>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<br>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<br>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<br>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.

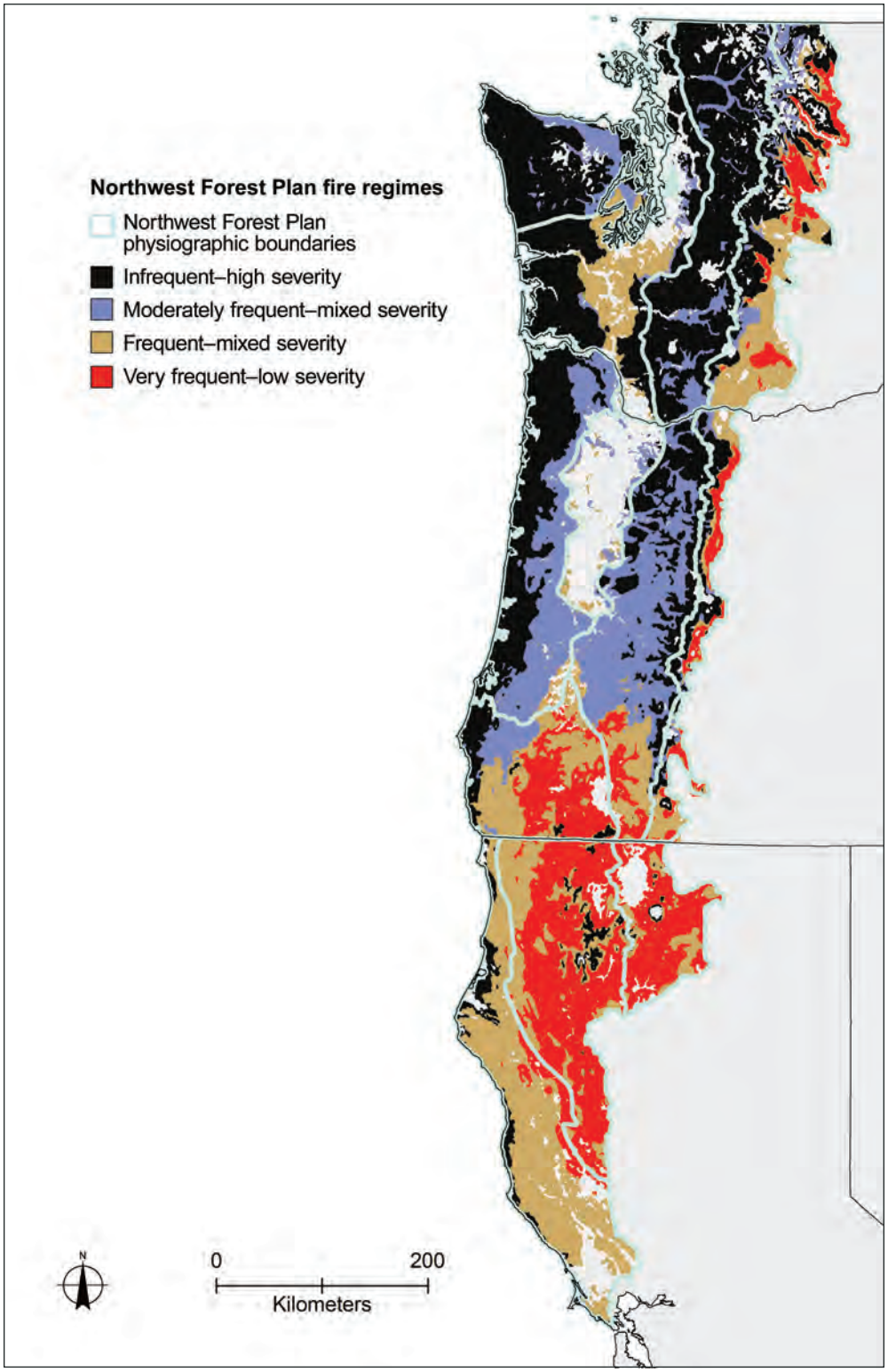



Figure 3-6—Generalized fire regimes for the Northwest Forest Plan (NWFP) area based on climate and lighting density. Fire frequency, particularly in coastal areas of California, may be underestimated because historical ignitions by American Indians are not included in the model. See table 3-1 for more 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.

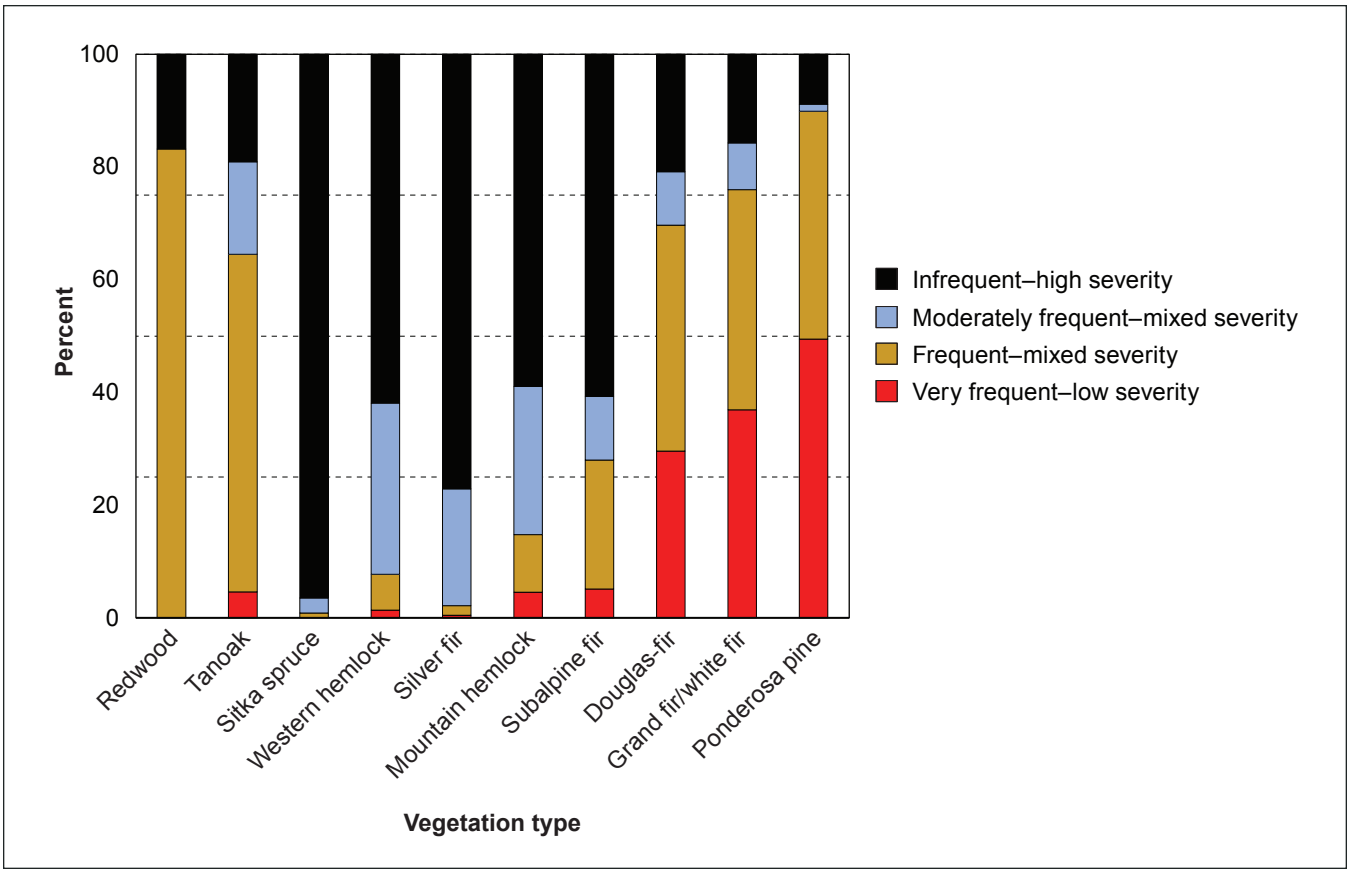



Figure 3-7—Percentage of major potential vegetation types (PVTs) in the four different fire regimes. Small percentages of a fire regime within a PVT may be a result of errors in the PVT maps, fire regime maps, or both.

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.

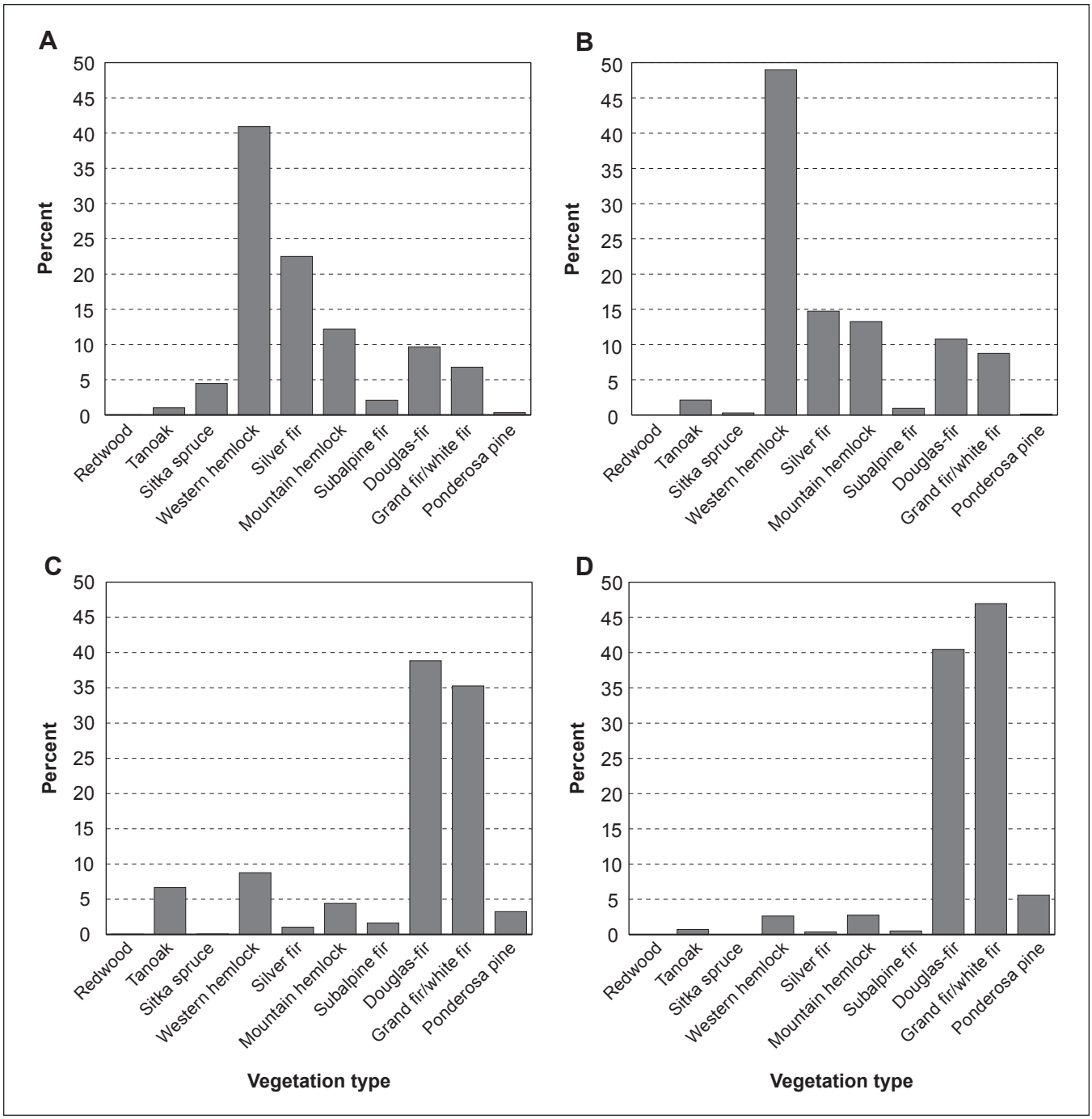
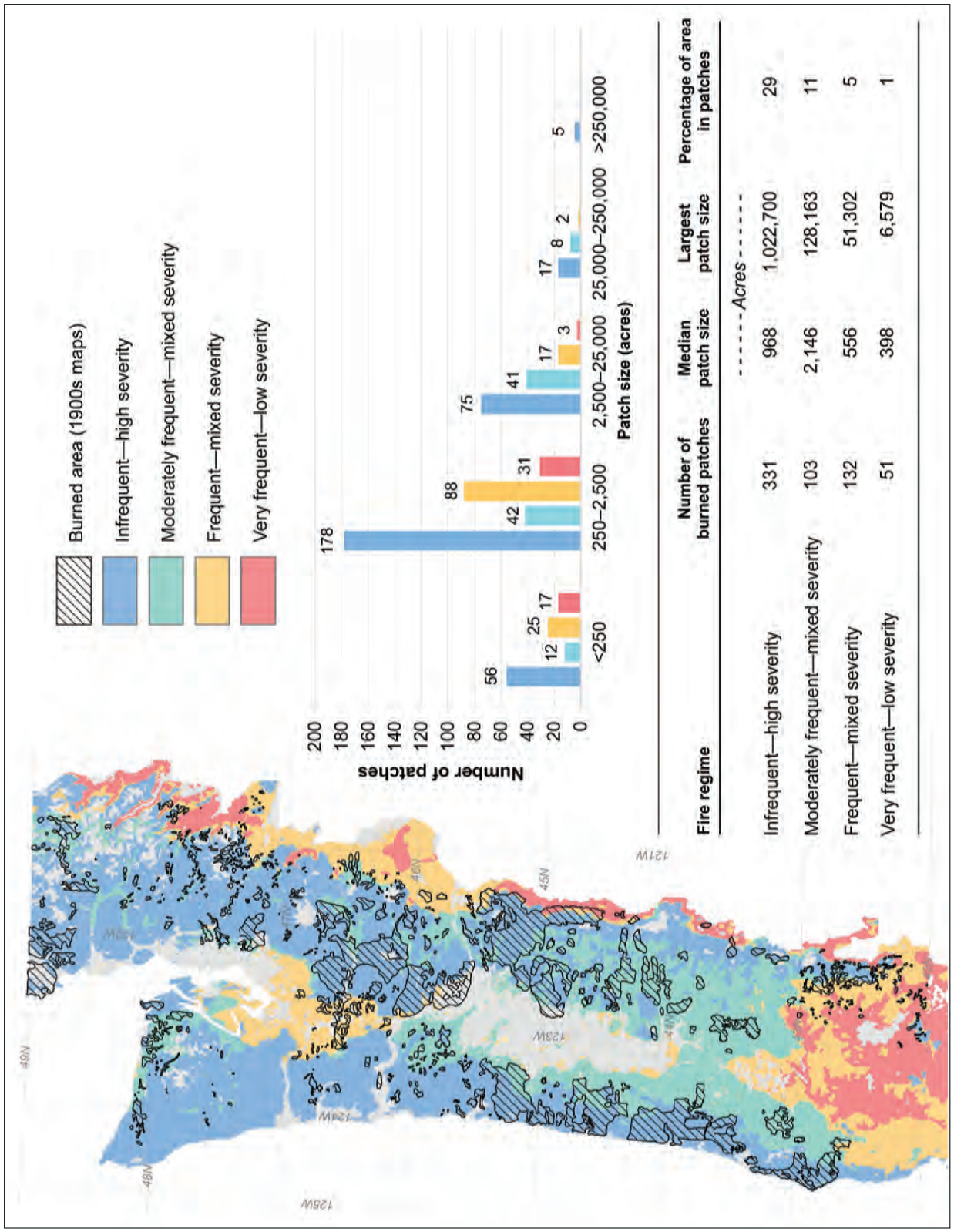



Figure 3-8—Distribution of major potential vegetation types (PVTs) within the (A) infrequent, high-severity regime; (B) moderately frequent, mixed-severity regimes of the moist forests; (C) frequent, mixed-severity regime; and (D) very frequent, low-severity regimes of the dry forests. Only major PVTs 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.

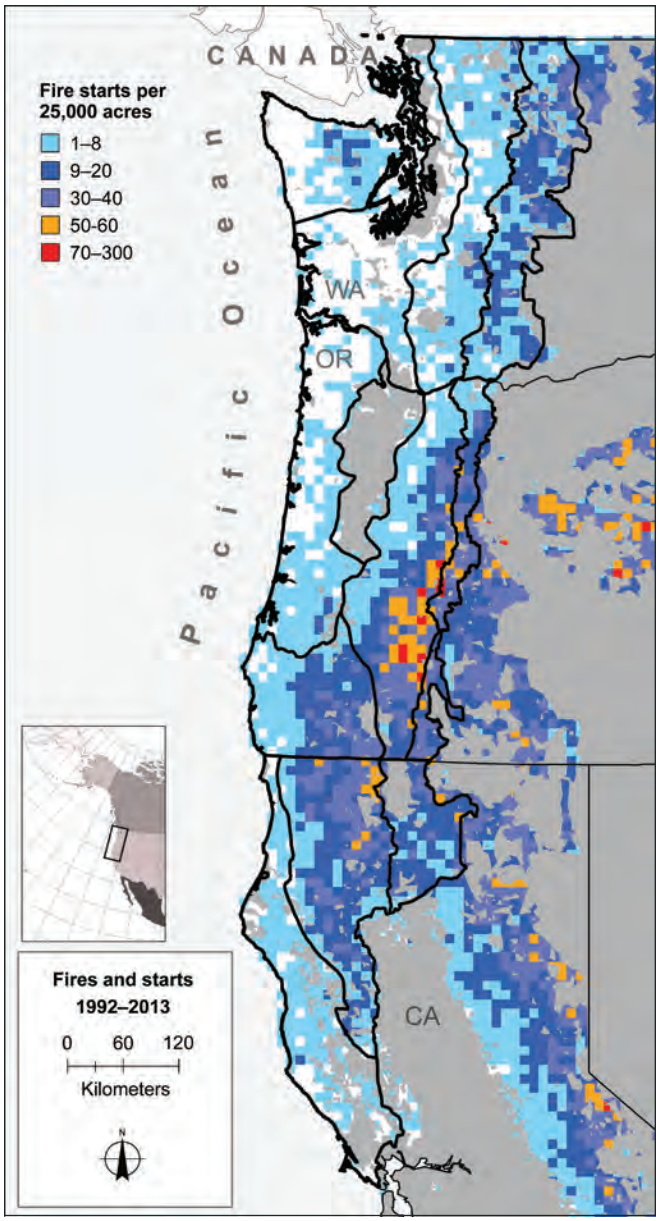



Figure 3-10—Density of lightning-ignited fires per 25,000 ac on forest lands in the Northwest Forest Plan area for the period 1992–2013. Black lines are physiographic provinces as delineated in figure 3-4.

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

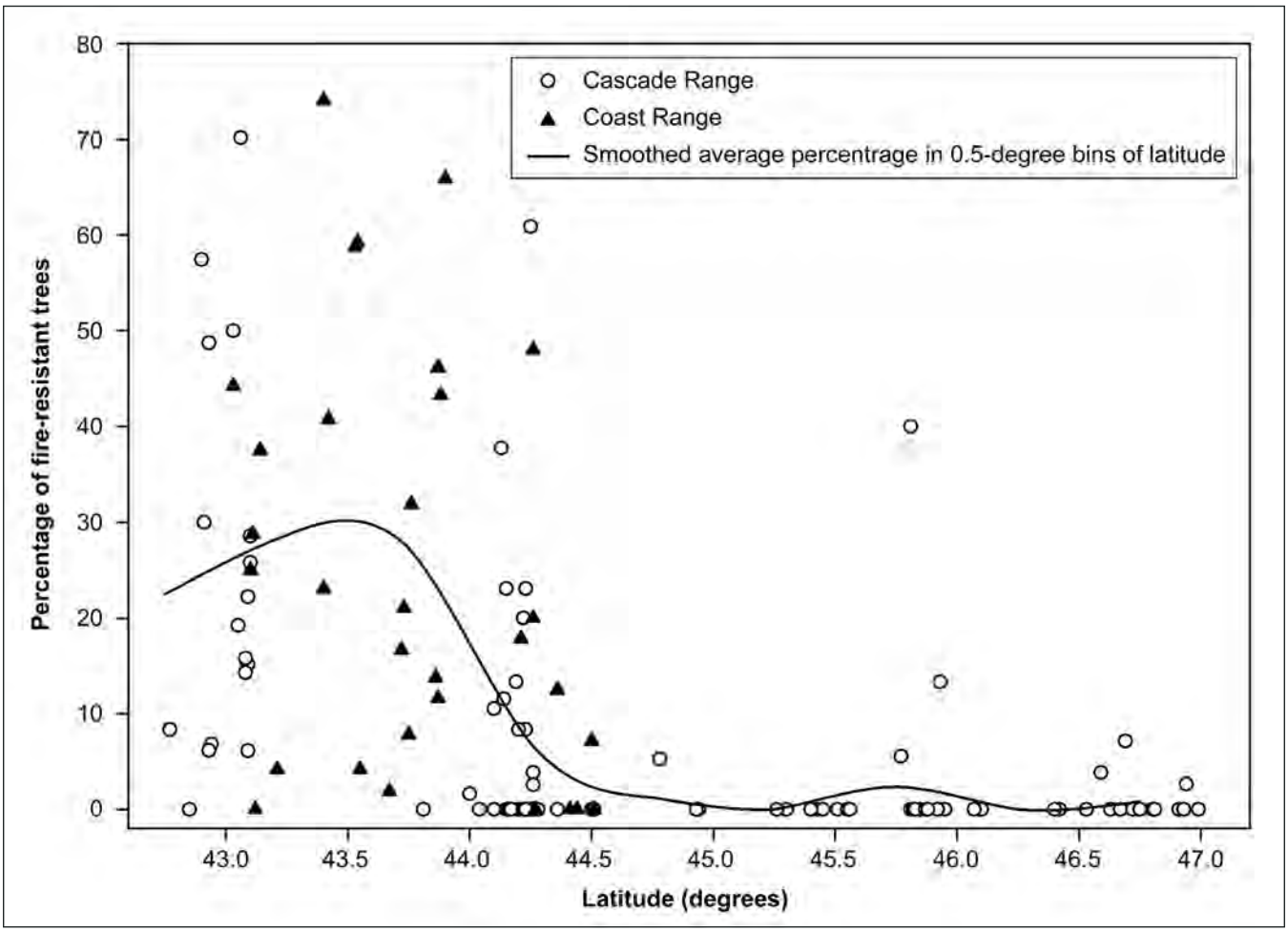



Figure 3-11—Percentage of fire-resistant mature and old trees with evidence of fire (scars or charred bark) in the western Cascades and Oregon Coast Range in relation to latitude. Line is smoothed running average in 0.5° bins. The increase in evidence of fire on tree boles around latitude 44.5° N in 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 14$^{th}$ 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<br>Creates ant/termite habitat<br>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

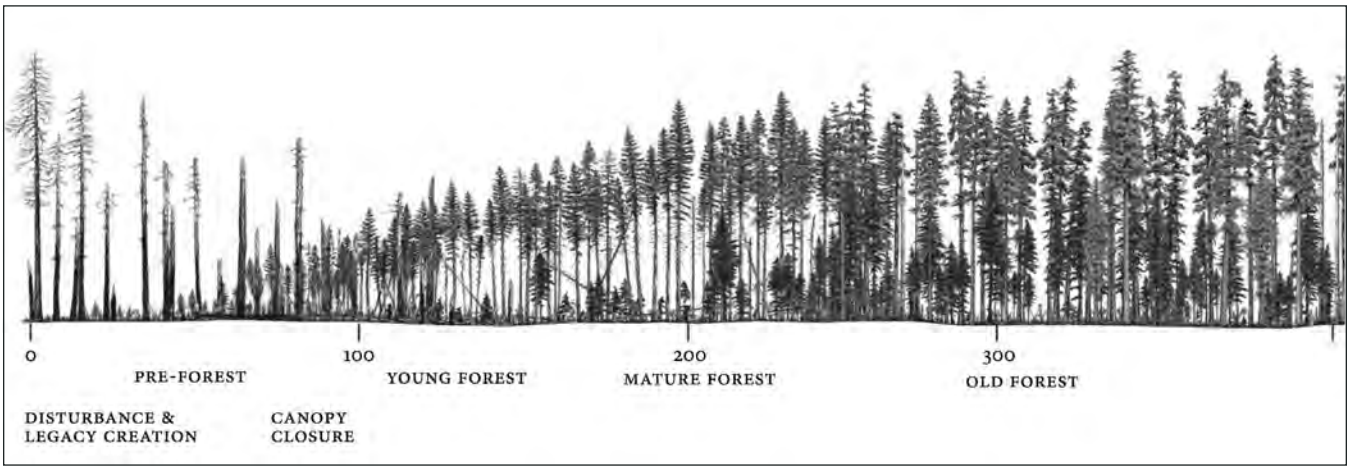



Figure 3-13—A common stand developmental pathway for a Douglas-fir and western hemlock forest following stand-replacement wildfire (from Franklin et al. 2018).

(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).

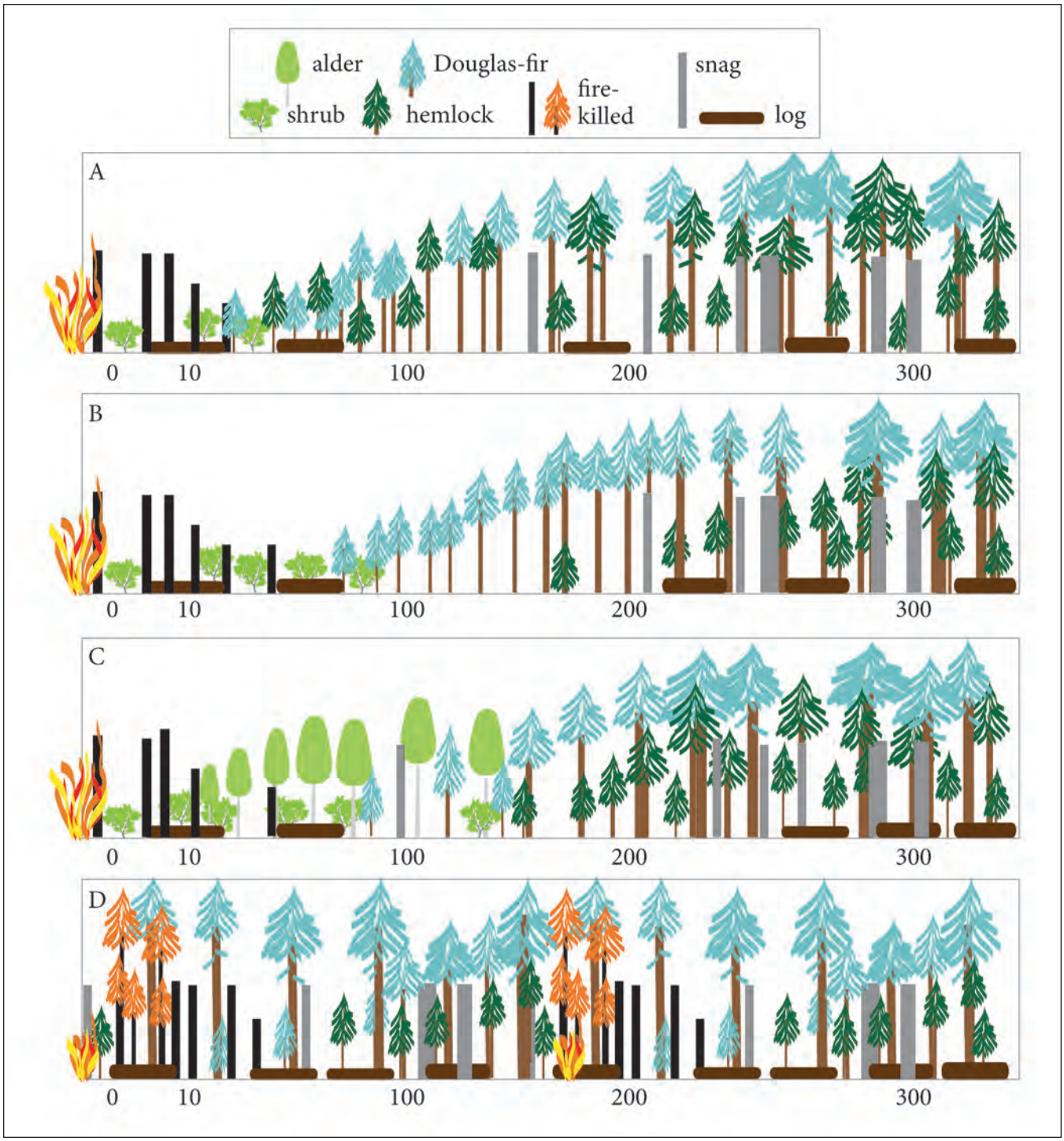



Figure 3-14—Multiple pathways of succession that could occur in the moist forests. Pathway A occurs when Douglas-fir canopy closure occurs within 50 years after a fire and western hemlock establishes early in succession. Pathway B occurs when the pre-forest shrub-dominated stage persists for many decades 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.

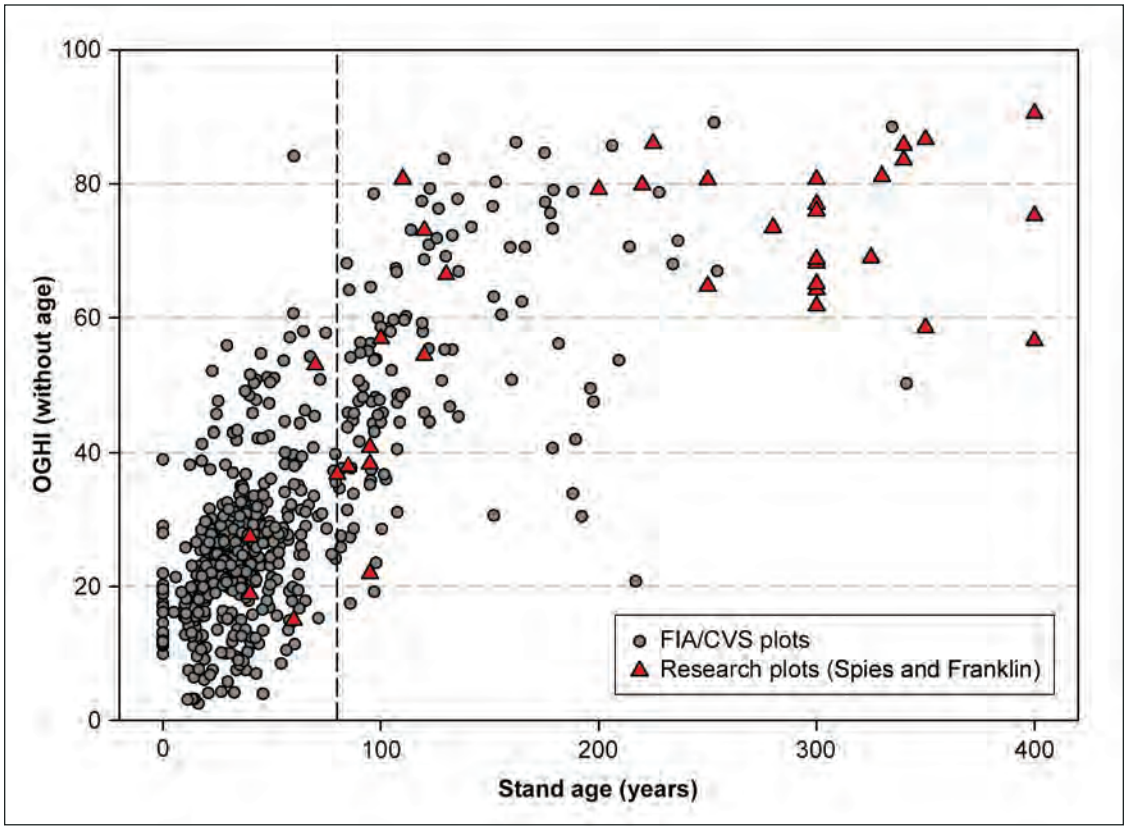



Figure 3-15—An old-growth forest habitat index (OGHI) (Franklin et al. 2005) in relation to stand age for forest inventory and research plots in the Oregon Coast Range. The index is based on number of large trees, large snags, volume of down woody debris, and tree size diversity, which is a surrogate 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

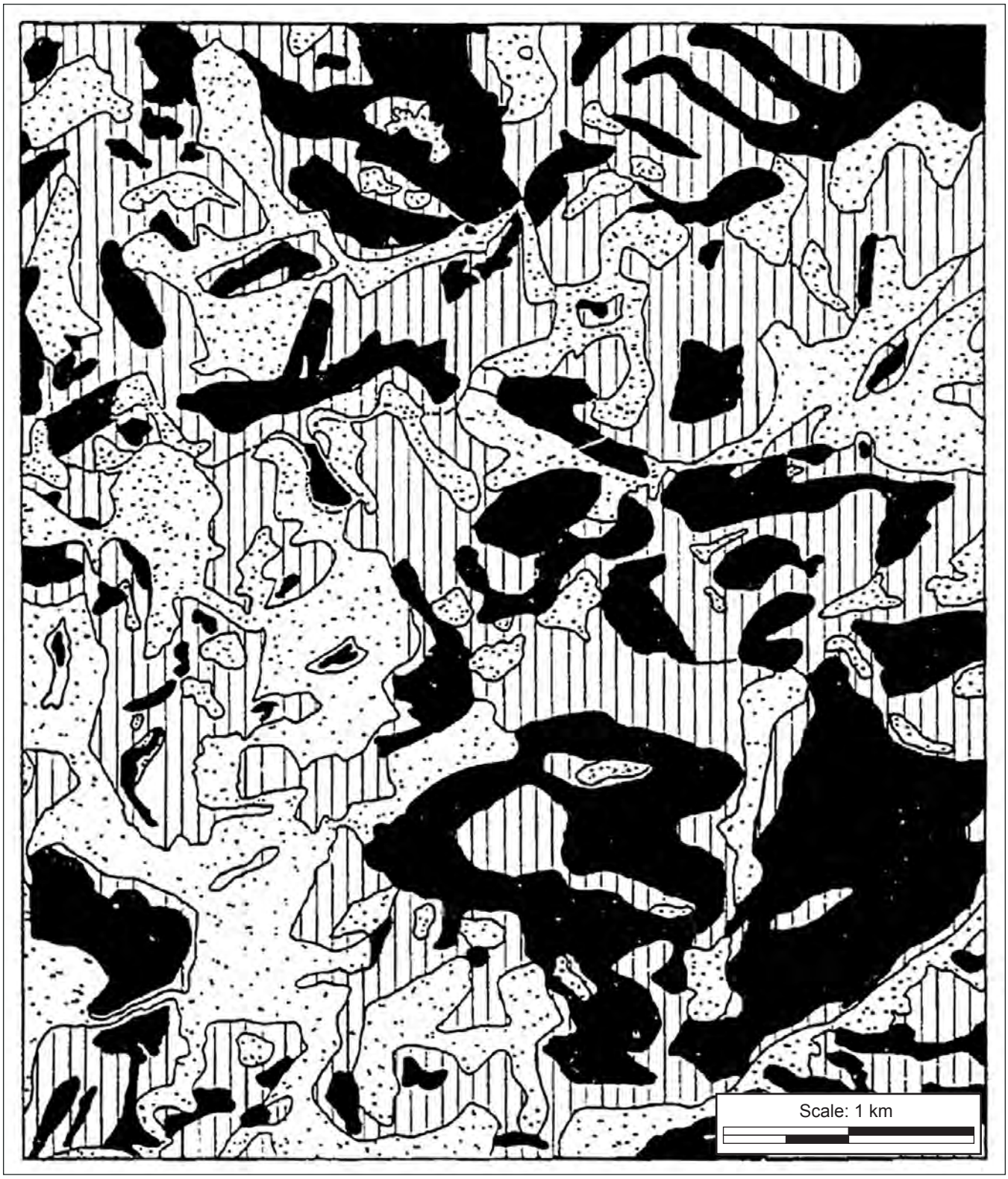



Figure 3-16—Mosaic of fire severity patches in a Douglas-fir and western hemlock landscape in the western Cascade Range of Oregon. Black = a high mortality area (>70 percent), vertical lines = moderate mortality (30 to 70 percent), and stippled = low mortality areas (<30 percent). From Morrison and Swanson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

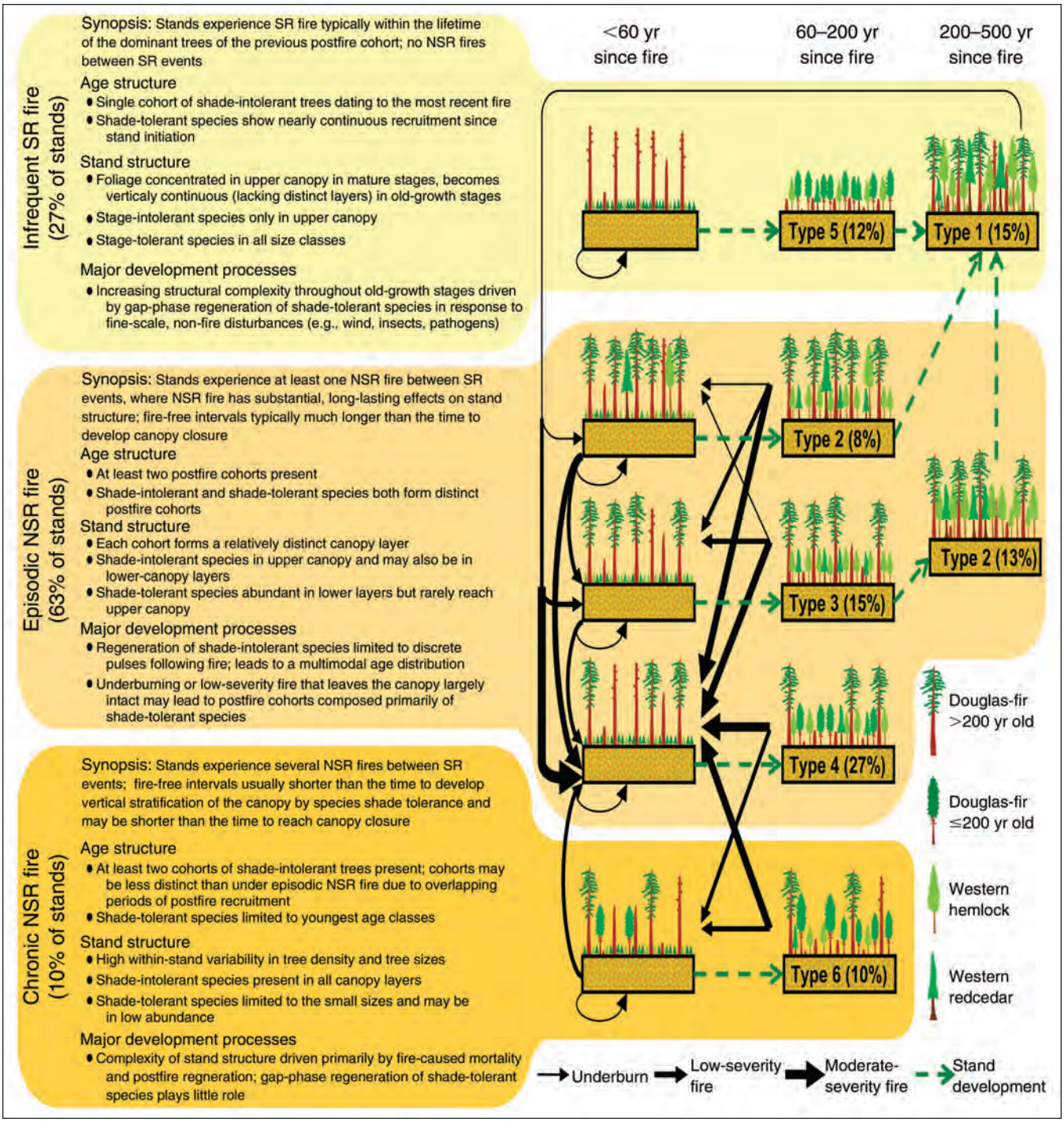



Figure 3-17—Conceptual model of stand-development pathways in Douglas-fir/western hemlock (current vegetation) forests in the moderately frequent, mixed-severity fire regime of the central western Cascade Range of Oregon. Dashed arrows represent stand development in the absence of fire, and solid arrows 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-20$^{th}$ 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 20$^{th}$ 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 20$^{th}$ 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 20$^{th}$ 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).

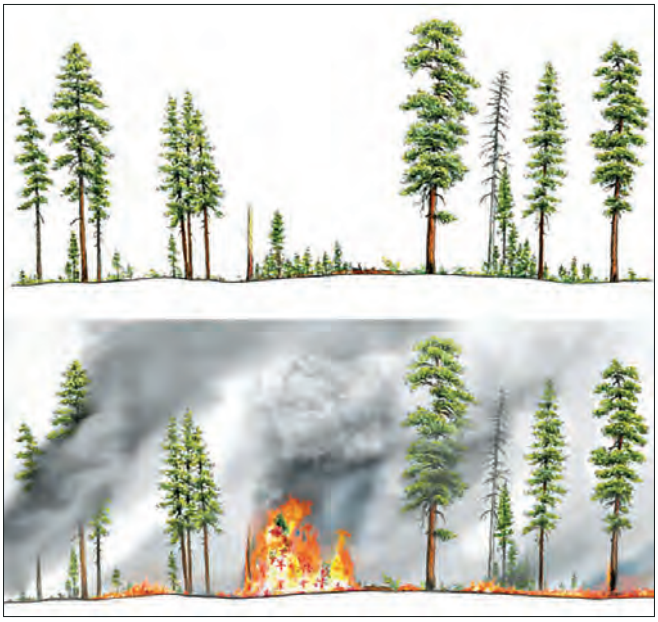


Figure 3-19—Hypothetical structural profile and typical historical fire behavior in a ponderosa pine forest of the eastern Cascade Range of Washington. From Van Pelt (2008).

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.

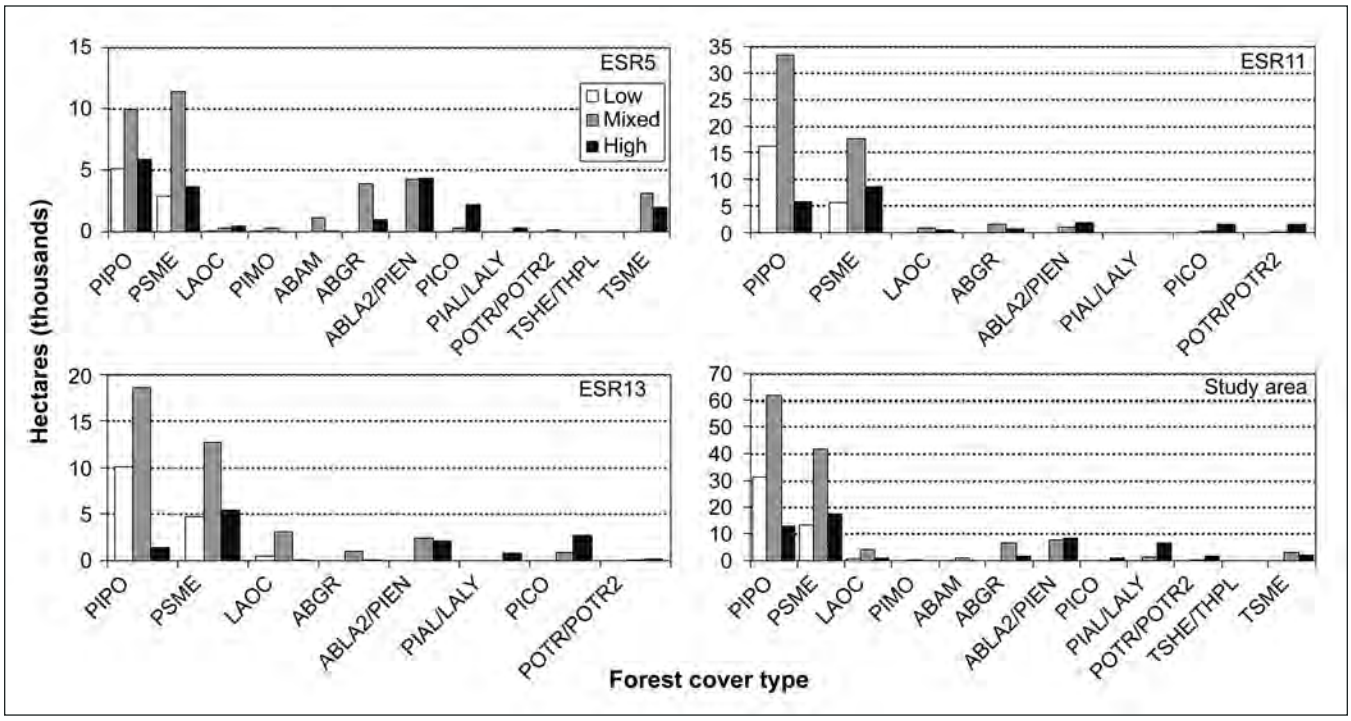



Figure 3-23—The proportions of premanagement-era total forest area (hectares) by forest cover type in low-, mixed-, and high-severity fire (corresponding with percentage of canopy mortality values of ≤20 percent, 20.1 to 69.9 percent, and ≥70 percent, respectively) of Ecological Subregions (ESRs) 5, 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).

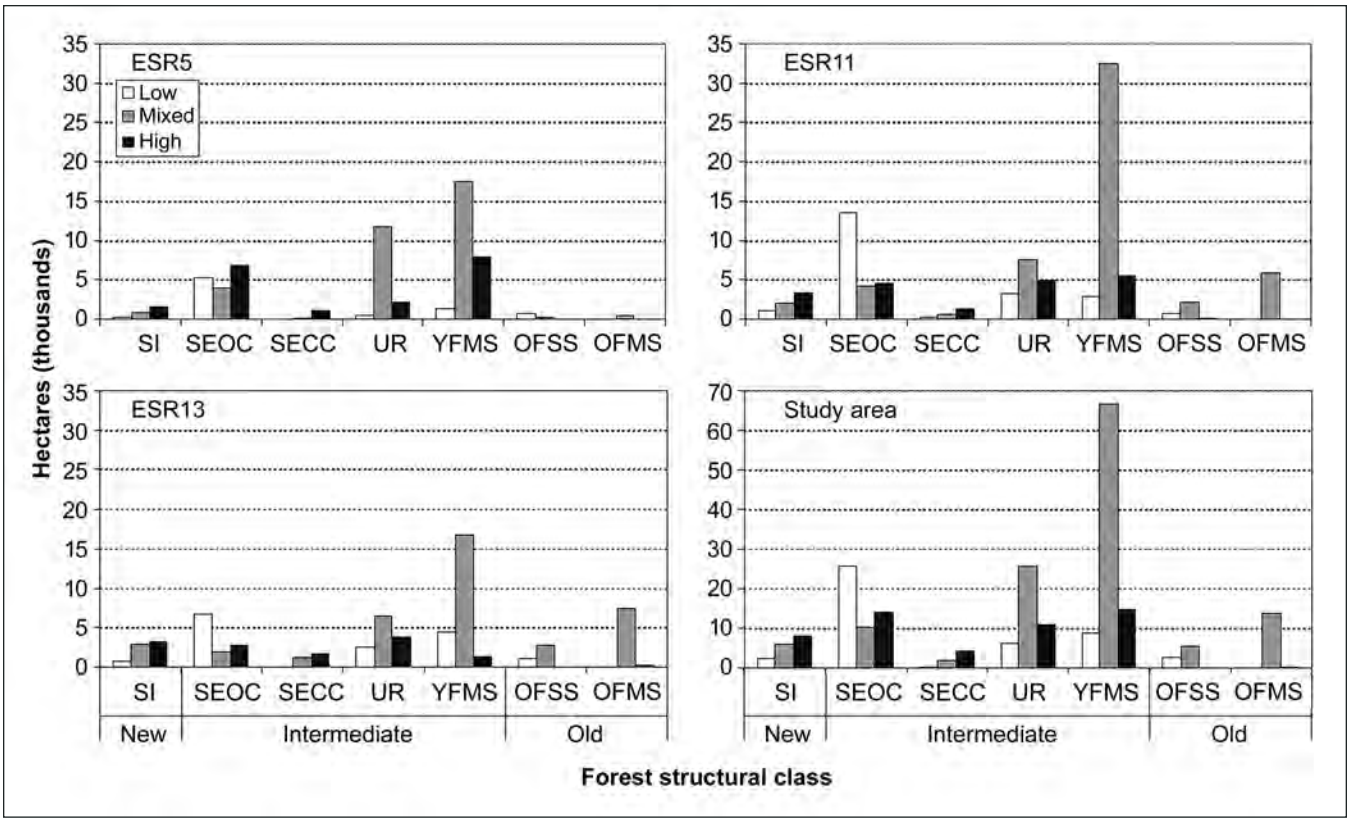



Figure 3-24—The proportions of the premanagement-era dry forest area (hectares) by forest structural class in low-, mixed-, and high-severity fire (corresponding with percentage of canopy mortality values of ≤20 percent, 20.1 to 69.9 percent, and ≥70 percent, respectively) of Ecological Subregions (ESRs) 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.

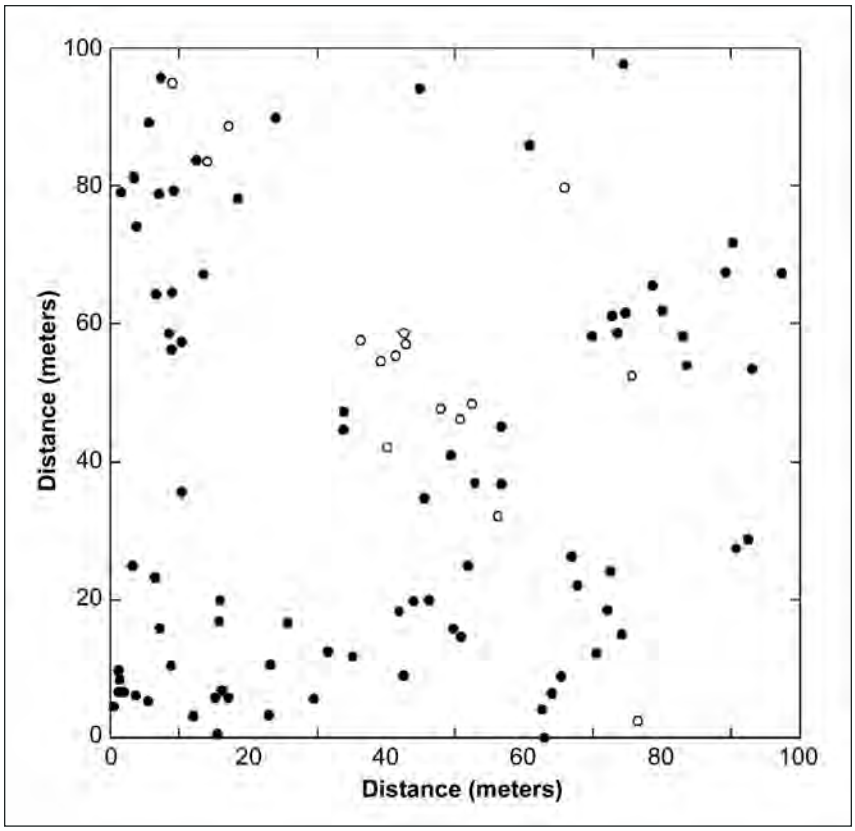



Figure 3-25—Spatial patterns of live (filled circle) and dead trees (open circle) in the upper canopy of an old-growth ponderosa pine forest in central Oregon. From Youngblood et al. (2004).

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 ($25^{th}$ to $75^{th}$) 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

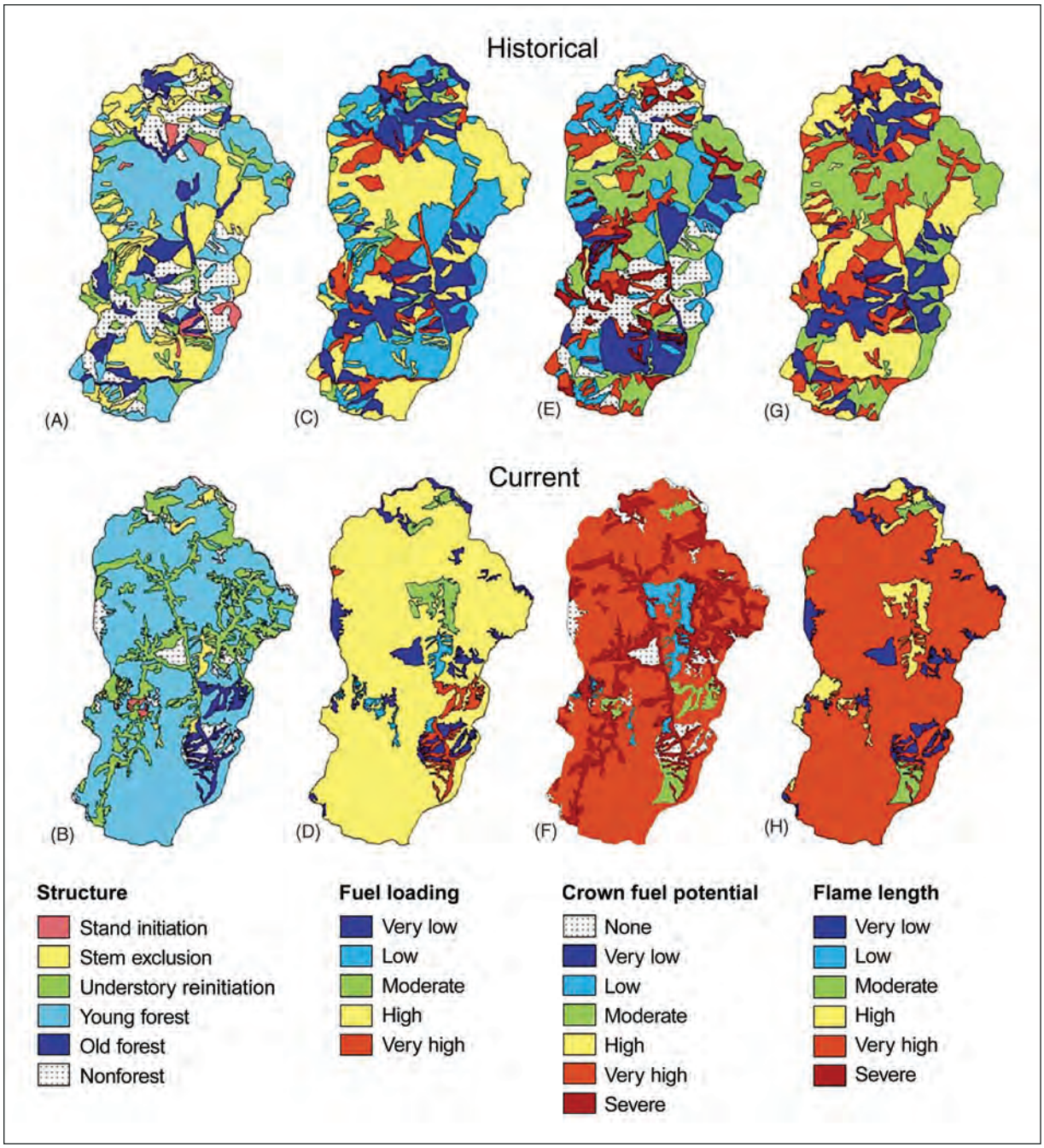



Figure 3-26—Reconstructed historical (1900s) and current (1990s) maps of dry forest subwatershed of the Lower Grand Ronde subbasin in the Blue Mountains province displaying historical and current structural classes (A and B), fuel loading (C and D), crown fire potential under average wildfire conditions (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

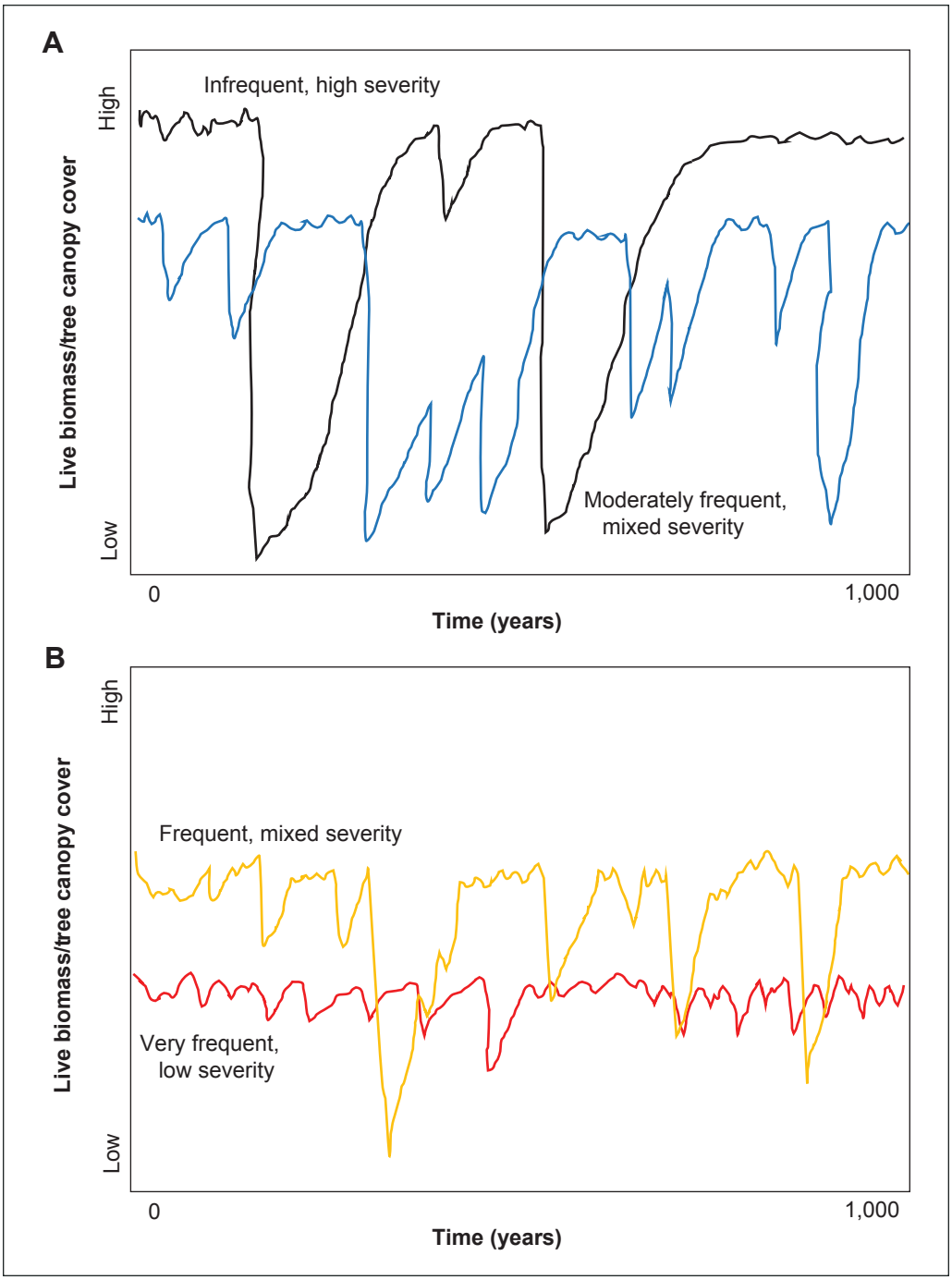



Figure 3-27—Hypothesized dynamics (historical range of variation) in live forest structure (biomass or cover) over a hypothetical 1,000-year period during the pre-Euro-American settlement period for an area of several thousand acres for (A) moist forest fire regimes and (B) dry forest fire regimes. Large 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, 20$^{th}$ 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.<br>Reduce fragmentation and increase connectivity of older forest patches.<br>Create or promote early-seral vegetation where needed to provide seral stage and landscape diversity.<br>Restore disturbance processes (e.g., fire) where feasible. |
| Dry forests | Protect existing large fire-tolerant trees in areas of dense and open forest.<br>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.<br>Increase areas of open old forests to promote resilience to fire and climate change and meet needs of species.<br>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.<br>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 20$^{th}$ 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).

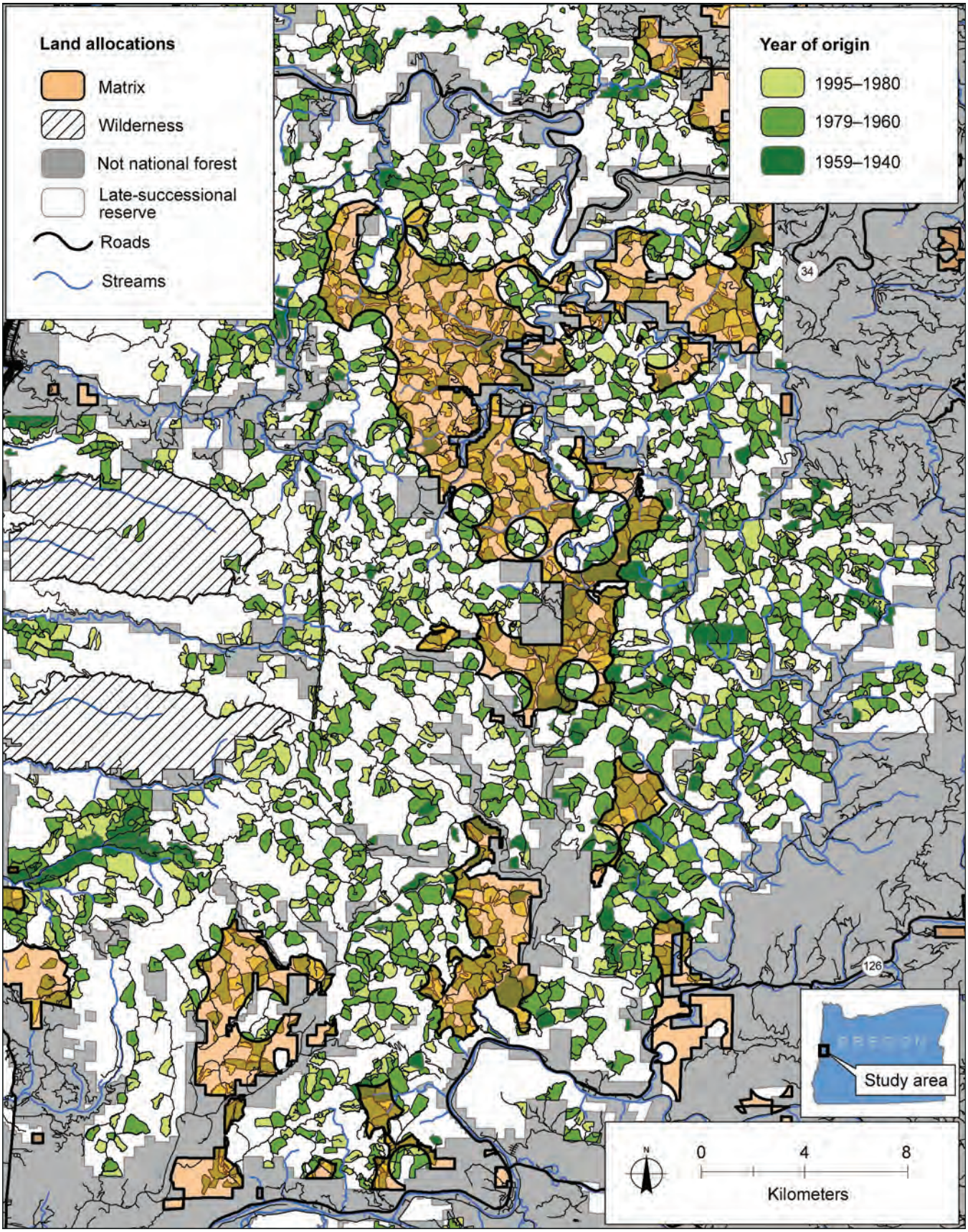



Figure 3-28—Plantations and the dates of their origin in a landscape containing late-successional reserve (in white), wilderness (striped), and matrix (orange) lands on the Siuslaw National Forest in coastal Oregon. From Stewart Johnston (retired), Siuslaw National Forest.

**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-20$^{th}$ 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.

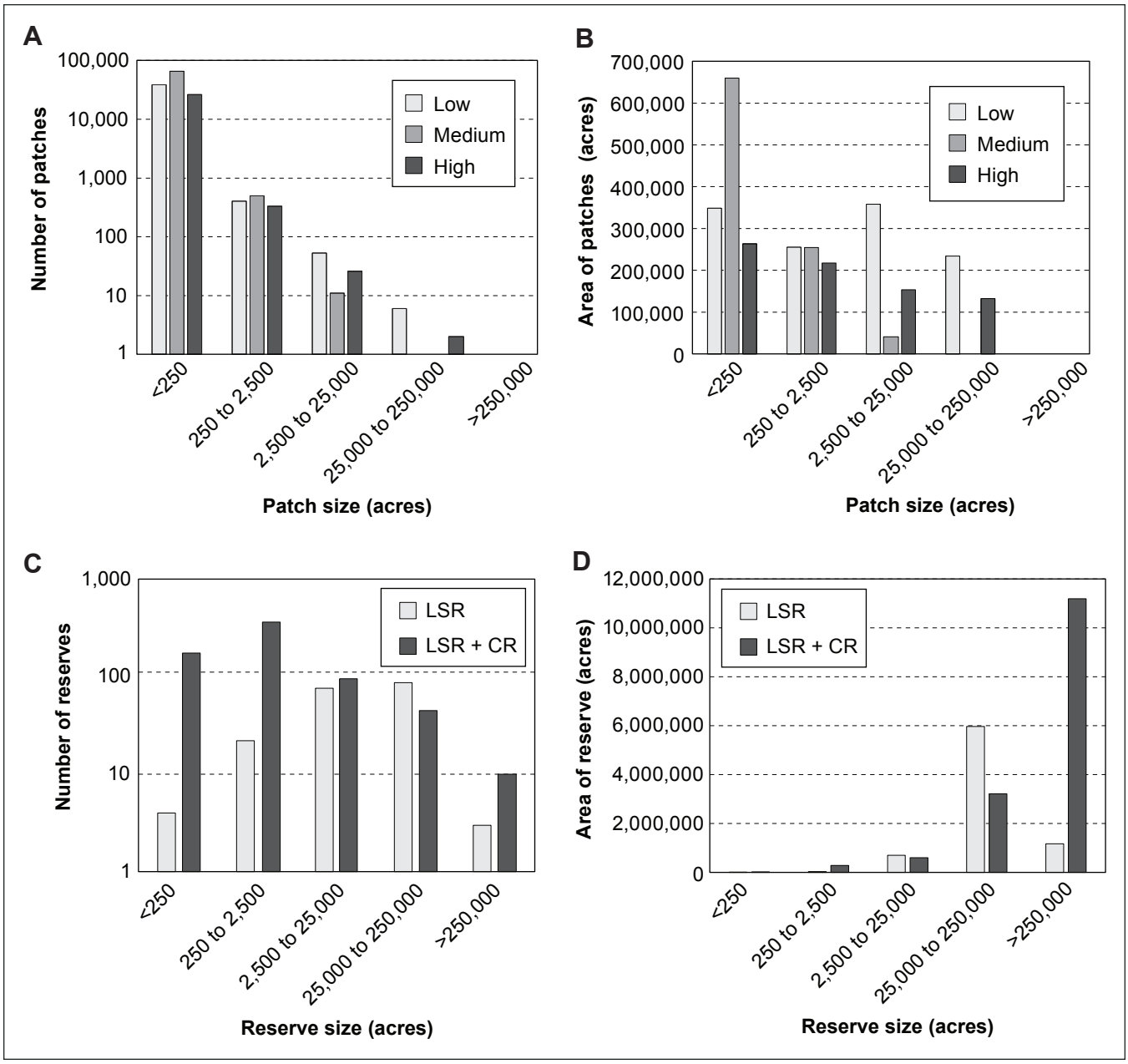



Figure 3-30—Patch distributions of recent fire (2000–2012) and sizes of Northwest Forest Plan (NWFP) reserves: (A) frequency distribution of number patches by fire severity and area class, (B) fire-severity patch sizes by area class, (C) frequency distribution of number of NWFP reserves (late-successonal 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 20$^{th}$ 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

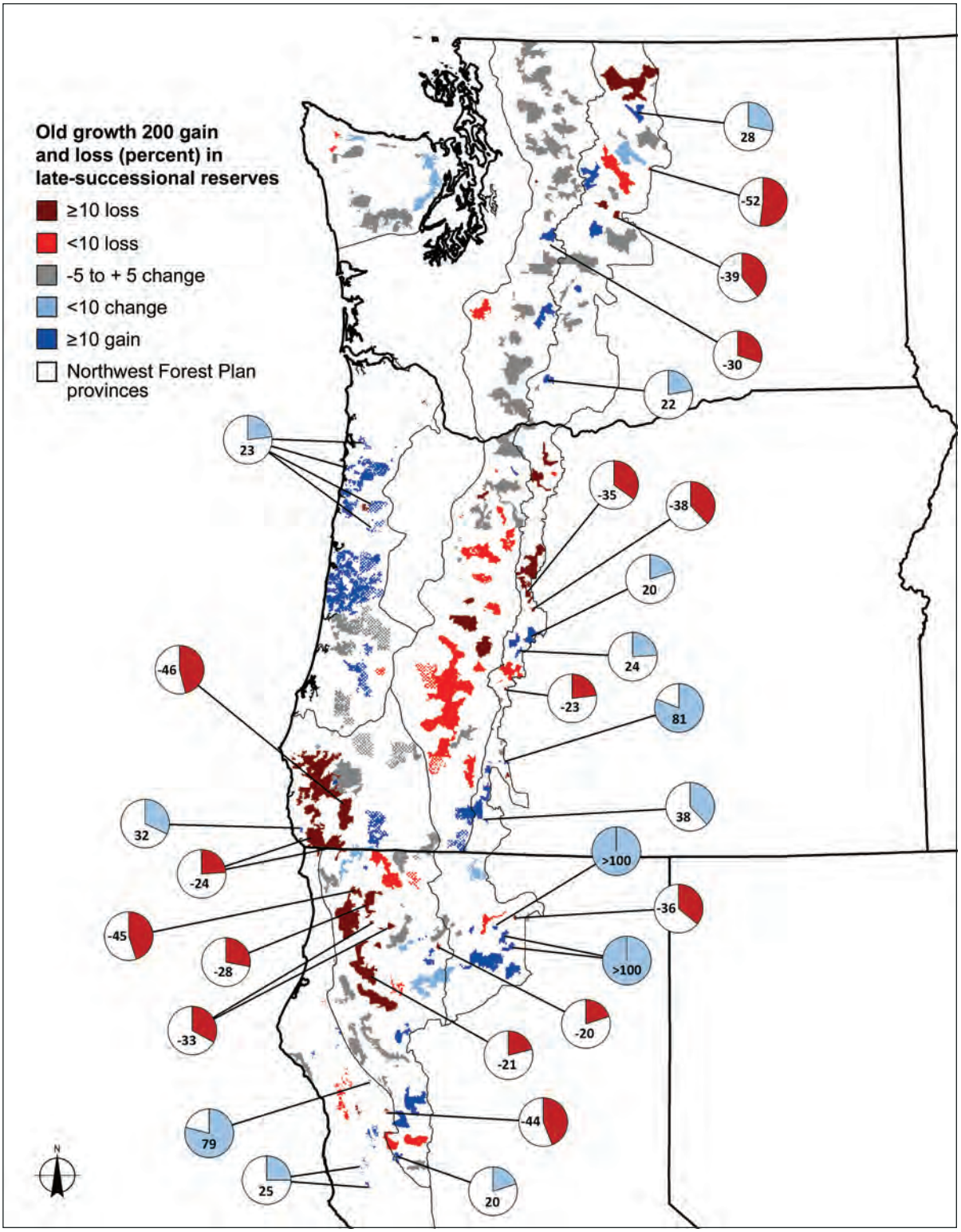



Figure 3-31—Map of 192 late-successional reserves (LSRs) in the Northwest Forest Plan area showing percentage of net change (gain or loss) in old-growth structure index (OGSI) 200 from 1993 to 2012. The LSRs are color coded by degree of gain (blue) or loss (red). The LSRs with little net change are shown 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.

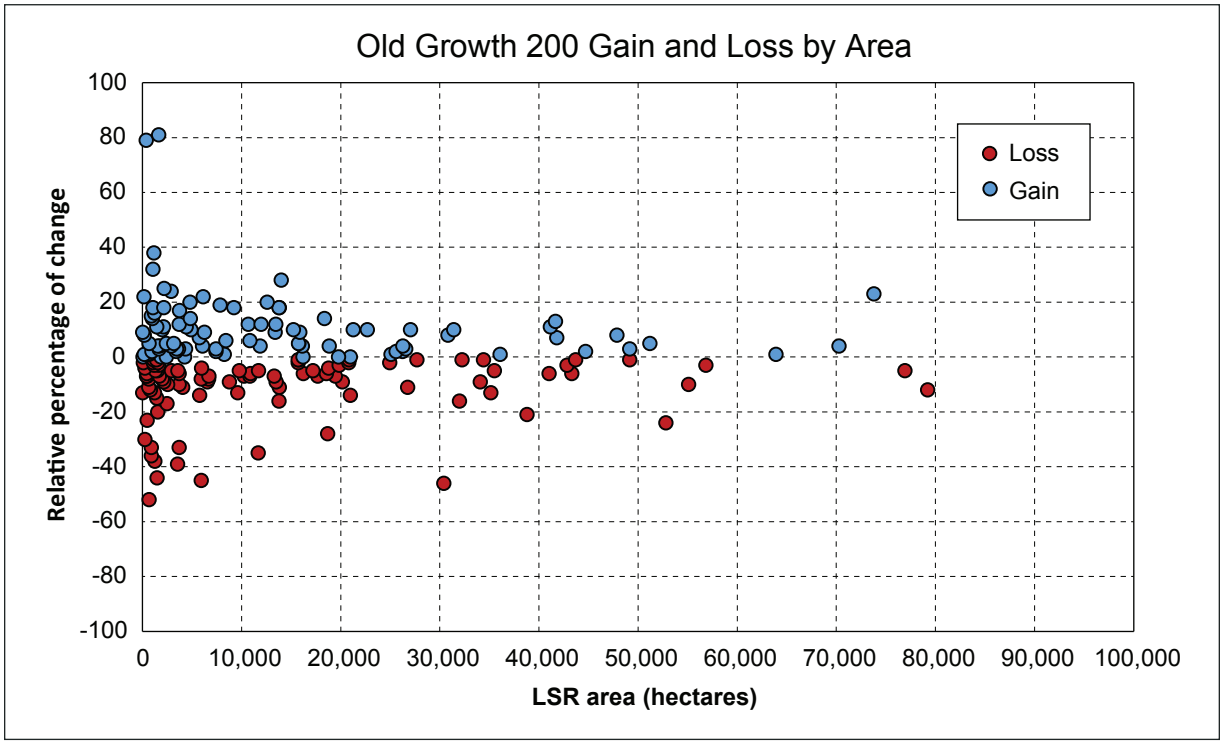



Figure 3-32—Relative change in old-growth strucuture index (OGSI) 200 in reserves between 1993 and 2013 in relation to late-successional reserve (LSR) size. Reserves smaller than 50,000 ac (20 224 ha) tended to show more change than larger reserves.

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

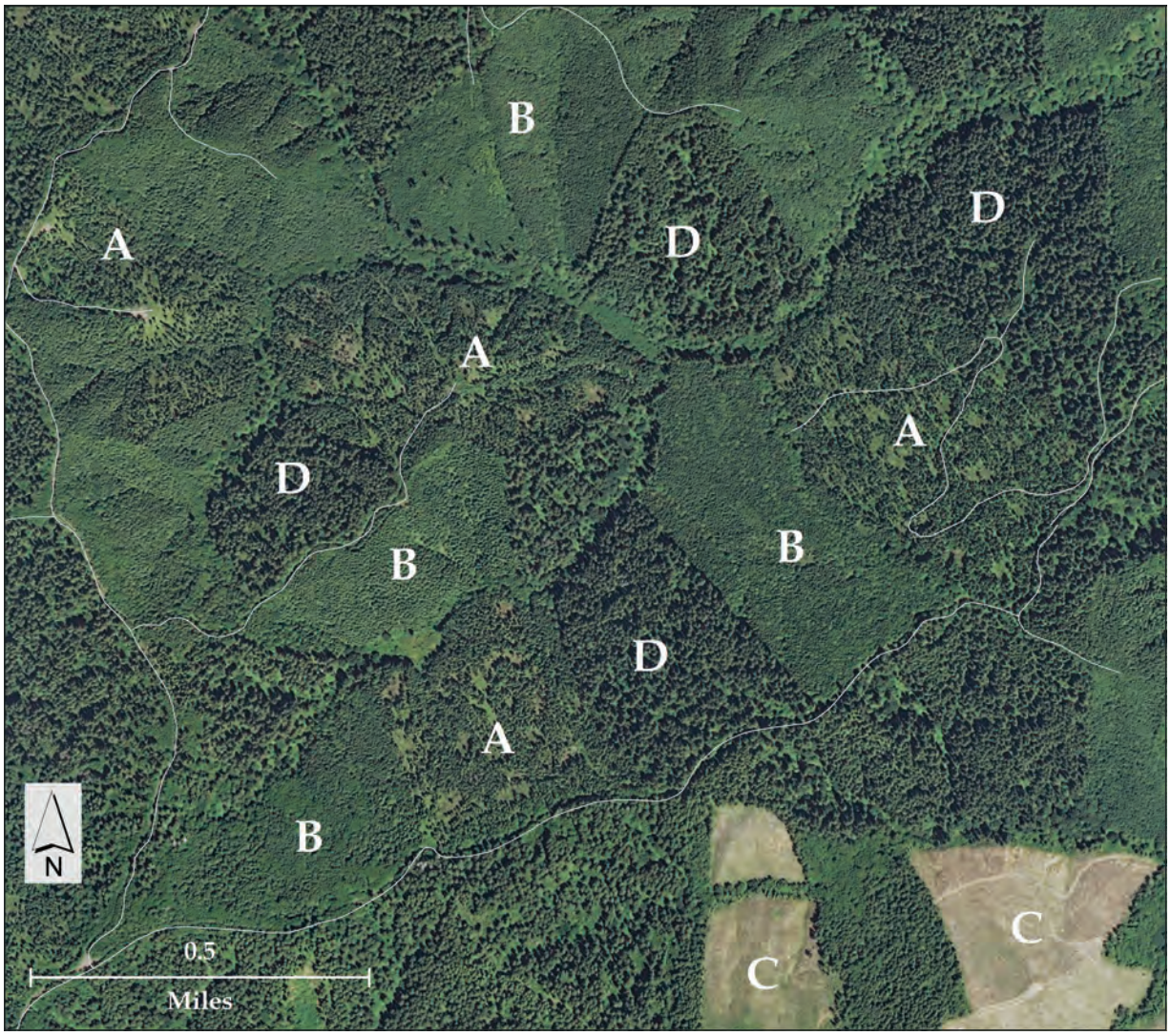



Figure 3-33—Aerial image from 2011 of management units and unmanaged stands in an area of late-successional and riparian reserves and matrix allocation on the Siuslaw National Forest, and private lands in the Oregon Coast Range: (A) plantations treated with variable density thinning, (B) uniform plantations 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

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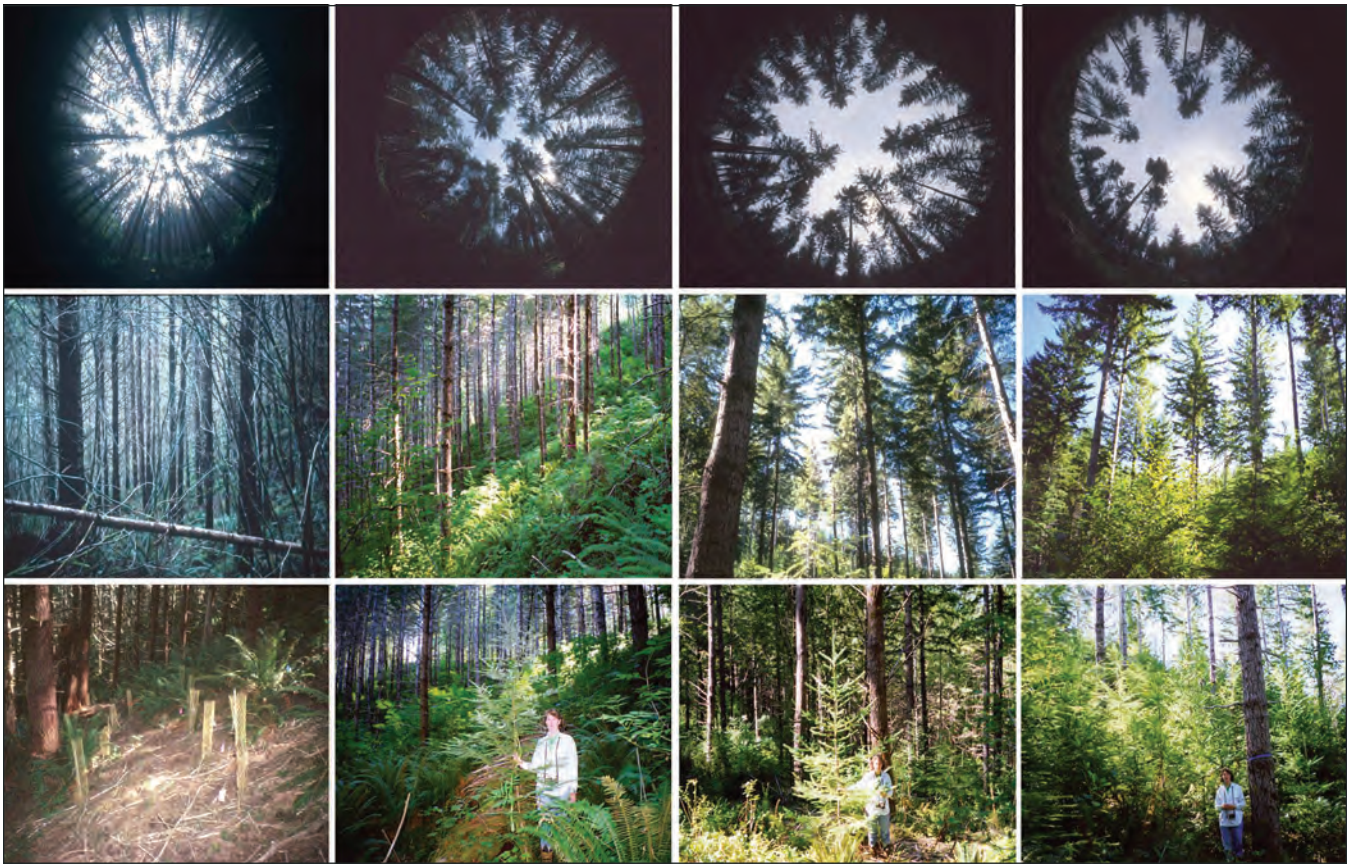
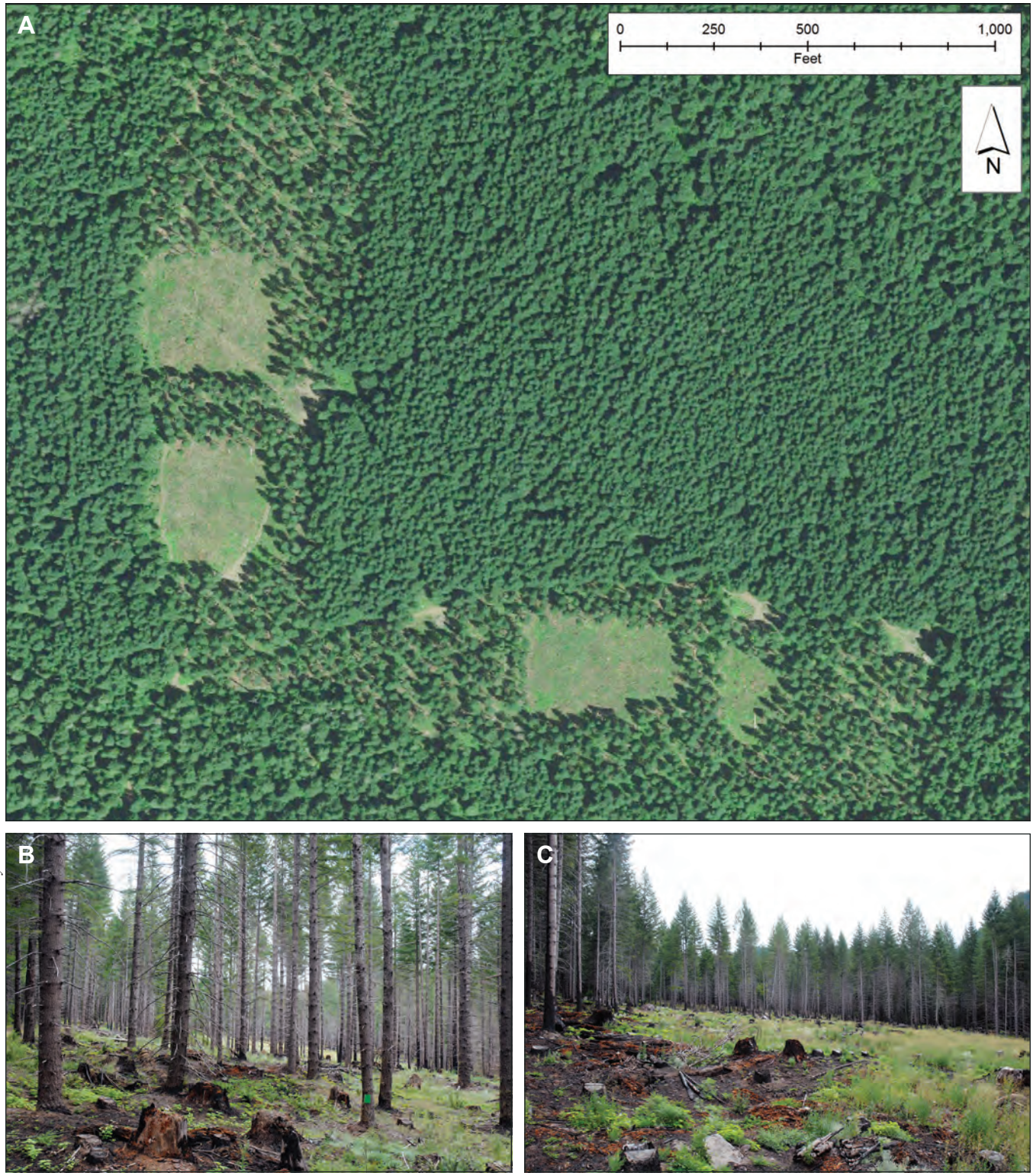


Figure 3-34—Canopy (fisheye) and understory photographs of unthinned and thinned 30 to 40 year old plantations of Douglas-fir on the Siuslaw National Forest. Densities of four stands from left to right: unthinned; 100 trees/acre; 60 trees/acre; and 30 trees/acre.

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

Gary Rost

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.

***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).

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

***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).

*Bureau of Land Management*

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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## Literature Cited

**Abatzoglou, J.T.; Rupp, D.E.; Mote, P.W. 2014.** Seasonal climate variability and change in the Pacific Northwest of the United States. Journal of Climate. 27(5): 2125–2142. doi: 10.1175/jcli-d-13-00218.1.

**Abella, S.R.; Springer, J.D. 2015.** Effects of tree cutting and fire on understory vegetation in mixed conifer forests. Forest Ecology and Management. 335: 281–299. doi: 10.1016/j.foreco.2014.09.009.

**Acker, S.A.; Kertis, J.; Bruner, H.; O'Connell, K.; Sexton, J. 2013.** Dynamics of coarse woody debris following wildfire in a mountain hemlock (*Tsuga mertensiana*) forest. Forest Ecology and Management. 302: 231–239. doi:10.1016/j.foreco.2013.03.013.

**Agee, J.K. 1991.** Fire history along an elevational gradient in the Siskiyou Mountains, Oregon. Northwest Science. 65(4): 188–199.

**Agee, J.K. 1993.** Fire ecology of Pacific Northwest forests. Washington, DC: Island Press. 511 p.

**Agee, J.K. 1994.** Fire and weather disturbances in terrestrial ecosystems of the eastern Cascades. Gen. Tech. Rep. PNW-GTR-320. Portland, OR: U.S. Department of Agriculture, Forest Service, Pacific Northwest Research Station. 52 p. doi:10.2737/pnw-gtr-320.

**Agee, J.K. 1998.** The landscape ecology of western forest fire regimes. Northwest Science. 72(17): 24–34.

**Agee, J.K. 2003.** Historical range of variability in eastern Cascades forests, Washington, USA. Landscape Ecology. 18(8): 725–740. doi:10.1023/b:land.0000014474.49803.f9.

**Agee, J.K.; Edmonds, R.L. 1992.** Forest protection guidelines for the northern spotted owl. In: U.S. Department of the Interior, Fish and Wildlife Service, ed. Recovery plan for the northern spotted owl. Washington, DC. Appendix F.

**Agee, J.K.; Finney, M.; Gouvenain, R.D. 1990a.** Forest fire history of Desolation Peak, Washington. Canadian Journal of Forest Research. 20(3): 350–356. doi:10.1139/x90-051.

**Agee, J.K.; Krusemark, F. 2001.** Forest fire regime of the Bull Run Watershed, Oregon. Northwest Science. 75: 292–306.

**Agee, J.K.; Potash, L.L.; Gracz, M. 1992b.** Oregon Caves forest and fire history. Report CPSU/UW 90-1. Seattle, WA: University of Washington, College of Forest Resources. 77 p.

**Agee, J.K.; Skinner, C.N. 2005.** Basic principles of forest fuel reduction treatments. Forest Ecology and Management. 211(1–2): 83–96. doi:10.1016/j.foreco.2005.01.034.

**Agee, J.K.; Wright, C.S.; Williamson, N.; Huff, M.H. 2002.** Foliar moisture content of Pacific Northwest vegetation and its relation to wildland fire behavior. Forest Ecology and Management. 167(1-3): 57–66. doi:10.1016/s0378-1127(01)00690-9.

**Ager, A.A.; Barros, A.M.G.; Preisler, H.K.; Day, M.A.; Spies, T.A.; Bailey, J.D.; Bolte, J. 2017.** Effects of accelerated wildfire on future fire regimes and implications for US Federal fire policy. Ecology and Society. 22(4): 12. doi.org/10.5751/ES-09680-220412

**Ager, A.A.; Finney, M.A.; McMahan, A.; Cathcart, J. 2010.** Measuring the effect of fuel treatments on forest carbon using landscape risk analysis. Natural Hazards and Earth System Science. 10(12): 2515–2526. doi:10.5194/nhess-10-2515-2010.

**Ahlgren, I.F.; Ahlgren, C.E. 1960.** Ecological effects of forest fires. The Botanical Review. 26(4): 483–533. doi:10.1007/bf02940573.

**Allen, C.D.; Macalady, A.K.; Chenchouni, H.; Bachelet, D.; McDowell, N.; Vennetier, M.; Kitzberger, T.; Rigling, A.; Breshears, D.D.; Hogg, E.H.; Gonzalez, P.; Fensham, R.; Zhang, Z.; Castro, J.; Demidova, N.; Lim, J.-H.; Allard, G.; Running, S.W.; Semerci, A.; Cobb, N. 2010.** A global overview of drought and heat-induced tree mortality reveals emerging climate change risks for forests. Forest Ecology and Management. 259(4): 660–684. doi:10.1016/j.foreco.2009.09.001.

**Allen, C.D.; Savage, M.; Falk, D.A.; Suckling, K.F.; Swetnam, T.W.; Schulke, T.; Stacey, P.B.; Morgan, P.; Hoffman, M.; Klingel, J.T. 2002.** Ecological restoration of southwestern ponderosa pine ecosystems: a broad perspective. Ecological Applications. 12(5): 1418–1433. doi:10.1890/1051-0761(2002)012[1418:erospp]2.0.co;2.

**Anderson, K. 2005.** Tending the wild: Native American knowledge and the management of California's natural resources. Berkeley, CA: University of California Press. 558 p.

**Anderson, P.D.; Ronnenberg, K.L. 2013.** Density management in the 21st century: west side story. Gen. Tech. Rep. PNW-GTR-880. Portland, OR: U.S. Department of Agriculture, Forest Service, Pacific Northwest Research Station. 249 p. doi:10.2737/pnw-gtr-880.

**Andrews, H.J.; Cowlin, R.W. 1940.** Forest resources of the Douglas-fir region. Misc. Publ. 389. Washington, DC: U.S. Department of Agriculture, Forest Service. 169 p. plus maps.

**Antoine, M.E. 2004.** An ecophysiological approach to quantifying nitrogen fixation by *Lobaria oregana*. The Bryologist. 107(1): 82–87. doi:10.1639/0007-2745(2004)107[82:aeatqn]2.0.co;2.

**Apigian, K.O.; Dahlsten, D.L.; Stephens, S.L. 2006.** Fire and fire surrogate treatment effects on leaf litter arthropods in a western Sierra Nevada mixed-conifer forest. Forest Ecology and Management. 221(1–3): 110–122. doi:10.1016/j.foreco.2005.09.009.

**Ares, A.; Berryman, S.D.; Puettmann, K.J. 2009.** Understory vegetation response to thinning disturbance of varying complexity in coniferous stands. Applied Vegetation Science. 12(4): 472–487. doi:10.1111/j.1654-109x.2009.01042.x.

**Ares, A.; Bright, C.; Puettmann, K. 2012.** Mesoscale variation in snag and hardwood densities and sizes in old-growth forests in western Oregon. Western Journal of Applied Forestry. 27: 12–17.

**Ares, A.; Neill, A.R.; Puettmann, K.J. 2010.** Understory abundance, species diversity and functional attribute response to thinning in coniferous stands. Forest Ecology and Management. 260(7): 1104–1113. doi:10.1016/j.foreco.2010.06.023.

**Azuma, D.L.; Donnegan, J.; Gedney, D. 2004.** Southwest Oregon Biscuit Fire: an analysis of forest resources and fire severity. Res. Pap. PNW-RP-560. Portland, OR: U.S. Department of Agriculture, Forest Service, Pacific Northwest Research Station. 32 p. doi:10.2737/pnw-rp-560.

**Bailey, J.D.; Tappeiner, J.C. 1998.** Effects of thinning on structural development in 40- to 100-year-old Douglas-fir stands in western Oregon. Forest Ecology and Management. 108(1–2): 99–113. doi:10.1016/s0378-1127(98)00216-3.

**Baird, M.; Zabowski, D.; Everett, R.L. 1999.** Wildfire effects on carbon and nitrogen in inland coniferous forests. Plant and Soil. 209(2): 233–243. doi:10.1023/a:1004602408717.

**Baker, S.C.; Halpern, C.B.; Wardlaw, T.J.; Kern, C.; Edgar, G.J.; Thomson, R.J.; Bigley, R.E.; Franklin, J.F.; Gandhi, K.J.K.; Gustafsson, L.; Johnson, S.; Palik, B.J.; Spies, T.A.; Steel, E.A.; Weslien, J.; Strengbom, J. 2016.** A cross-continental comparison of plant and beetle responses to retention of forest patches during timber harvest. Ecological Applications. 26(8): 2495–2506. doi:10.1002/eap.1406.

**Baker, W.L. 2012.** Implications of spatially extensive historical data from surveys for restoring dry forests of Oregon's eastern Cascades. Ecosphere. 3(3): art23. doi:10.1890/es11-00320.1.

**Baker, W.L. 2014.** Historical forest structure and fire in Sierran mixed-conifer forests reconstructed from General Land Office survey data. Ecosphere. 5(7): art79. doi:10.1890/es14-00046.1.

**Baker, W.L. 2015a.** Are high-severity fires burning at much higher rates recently than historically in dry-forest landscapes of the western USA? PLoS ONE. 10(9): e0136147. doi:10.1371/journal.pone.0136147.

**Baker, W.L. 2015b.** Historical northern spotted owl habitat and old-growth dry forests maintained by mixed-severity wildfires. Landscape Ecology. 30(4): 655–666.

**Baker, W.L.; Ehle, D. 2001.** Uncertainty in surface-fire history: the case of ponderosa pine forests in the western United States. Canadian Journal of Forest Research. 31(7): 1205–1226. doi:10.1139/x01-046.

**Baker, W.L.; Williams, M.A. 2015.** Bet-hedging dry-forest resilience to climate-change threats in the western USA based on historical forest structure. Frontiers in Ecology and Evolution. 2: 88. doi:10.3389/fevo.2014.00088.

**Barnes, B.V.; Zak, D.R.; Denton, S.R.; Spurr, S.H. 1998.** Forest ecology. 4th ed. New York: John Wiley & Sons, Inc. 774 p.

**Batavia, C.; Nelson, M.P. 2016.** Conceptual ambiguities and practical challenges of ecological forestry: a critical review. Journal of Forestry. 114(5): 572–581. doi:10.5849/jof.15-103.

**Beaty, R.M.; Taylor, A.H. 2001.** Spatial and temporal variation of fire regimes in a mixed conifer forest landscape, Southern Cascades, California, USA. Journal of Biogeography. 28(8): 955–966. doi:10.1046/j.1365-2699.2001.00591.x.

**Bednářová, M.; Dvorák, M.; Janoušek, J.; Jankovský, L. 2013.** Other foliar diseases of coniferous trees. In: Gonthier, P.; Nicolotti, G., eds. Infectious forest diseases. Wallingford, United Kingdom; Boston, MA: CABI: 458–487. Chapter 23.

**Beh, M.M.; Metz, M.R.; Frangioso, K.M.; Rizzo, D.M. 2012.** The key host for an invasive forest pathogen also facilitates the pathogen''s survival of wildfire in California forests. New Phytologist. 196(4): 1145–1154. doi:10.1111/j.1469-8137.2012.04352.x.

**Bekker, M.F.; Taylor, A.H. 2001.** Gradient analysis of fire regimes in montane forests of the southern Cascade Range, Thousand Lakes Wilderness, California, USA. Plant Ecology. 155(1): 15–26.

**Bekker, M.F.; Taylor, A.H. 2010.** Fire disturbance, forest structure, and stand dynamics in montane forests of the southern Cascades, Thousand Lakes Wilderness, California, USA. Écoscience. 17(1): 59–72. doi:10.2980/17-1-3247.

**Benda, L.; Miller, D.; Bigelow, P.; Andras, K. 2003.** Effects of post-wildfire erosion on channel environments, Boise River, Idaho. Forest Ecology and Management. 178(1–2): 105–119. doi:10.1016/s0378-1127(03)00056-2.

**Bengtsson, J.; Angelstam, P.; Elmqvist, T.; Emanuelsson, U.; Folke, C.; Ihse, M.; Moberg, F.; Nyström, M. 2003.** Reserves, resilience and dynamic landscapes. AMBIO: A Journal of the Human Environment. 32(6): 389–396. doi:10.1579/0044-7447-32.6.389.

**Berger, A.I.; Fischer, D.G. 2016.** Soil impacts due to the invasion of *ilex aquifolium* (English holly) into second growth forests of the Pacific Northwest. Metamorphosis. Fall. http://metamorphosis.coplac.org/index.php/metamorphosis/article/view/36. (20 September 2017).

**Berger, C.A.; Puettmann, K.J.; McKenna, J. 2012.** Understory response to repeated thinning in Douglas-fir forests of western Oregon. Journal of Sustainable Forestry. 31(6): 589–605. doi:10.1080/10549811.2011.628523.

**Beschta, R.L.; Rhodes, J.J.; Kauffman, J.B.; Gresswell, R.E.; Minshall, G.W.; Karr, J.R.; Perry, D.A.; Hauer, F.R.; Frissel, C.A. 2004.** Postfire management on forested public lands of the western United States. Conservation Biology. 18: 957–967.

**Bettinger, P.; Kellogg, L.D. 1993.** Residual stand damage from cut-to-length thinning of second-growth timber in the Cascade Range of western Oregon. Forest Products Journal. 43(11, 12): 59.

**Betts, M.G.; Fahrig, L.; Hadley, A.S.; Halstead, K.E.; Bowman, J.; Robinson, W.D.; Wiens, J.A.; Lindenmayer, D.B. 2014.** A species-centered approach for uncovering generalities in organism responses to habitat loss and fragmentation. Ecography. 37(6): 517–527. doi:10.1111/ecog.00740.

**Bigelow, S.W.; North, M.P. 2012.** Microclimate effects of fuels-reduction and group-selection silviculture: Implications for fire behavior in Sierran mixed-conifer forests. Forest Ecology and Management. 264: 51–59. doi:10.1016/j.foreco.2011.09.031.

**Bisson, P.A.; Rieman, B.E.; Luce, C.; Hessburg, P.F.; Lee, D.C.; Kershner, J.L.; Reeves, G.H.; Gresswell, R.E. 2003.** Fire and aquatic ecosystems of the western USA: current knowledge and key questions. Forest Ecology and Management. 178(1–2): 213–229. doi:10.1016/s0378-1127(03)00063-x.

**Bohlman, G.N.; North, M.; Safford, H.D. 2016.** Shrub removal in reforested post-fire areas increases native plant species richness. Forest Ecology and Management. 374: 195–210. doi:10.1016/j.foreco.2016.05.008.

**Bordelon, M.A.; McAllister, D.C.; Holloway, R. 2000.** Sustainable forestry Oregon style. Journal of Forestry. 98(1): 26–34.

**Bork, J.L. 1984.** Fire history in three vegetation types on the eastern side of the Oregon Cascades. Corvallis, OR: Oregon State University. 94 p. Ph.D. dissertation.

**Bormann, B.T.; Darbyshire, R.L.; Homann, P.S.; Morrissette, B.A.; Little, S.N. 2015.** Managing early succession for biodiversity and long-term productivity of conifer forests in southwestern Oregon. Forest Ecology and Management. 340: 114–125. doi:10.1016/j.foreco.2014.12.016.

**Bradford, J.B.; Bell, D.M. 2017.** A window of opportunity for climate-change adaptation: easing tree mortality by reducing forest basal area. Frontiers in Ecology and the Environment. 15(1): 11–17.

**Bradford, J.B.; Fraver, S.; Milo, A.M.; D'Amato, A.W.; Palik, B.; Shinneman, D.J. 2012.** Effects of multiple interacting disturbances and salvage logging on forest carbon stocks. Forest Ecology and Management. 267: 209–214. doi:10.1016/j.foreco.2011.12.010.

**Bradshaw, G.A.; Spies, T.A. 1992.** Characterizing canopy gap structure in forests using wavelet analysis. The Journal of Ecology. 80(2): 205. doi:10.2307/2261007.

**Briles, C.E.; Whitlock, C.; Bartlein, P.J. 2005.** Postglacial vegetation, fire, and climate history of the Siskiyou Mountains, Oregon, USA. Quaternary Research. 64(01): 44–56. doi:10.1016/j.yqres.2005.03.001.

**Briles, C.E.; Whitlock, C.; Skinner, C.N.; Mohr, J. 2011.** Holocene forest development and maintenance on different substrates in the Klamath Mountains, northern California, USA. Ecology. 92(3): 590–601. doi:10.1890/09-1772.1.

**Brookes, M.H.; Campbell, R.W.; Colbert, J.J.; Mitchell, R.G.; Stark, R.W., tech. eds. 1987.** Western spruce budworm. Tech. Bull. 1694. Washington, DC: U.S. Department of Agriculture, Forest Service, Cooperative State Research Service. 198 p.

**Brookes, M.H.; Stark, R.W.; Campbell, R.W., eds. 1978.** The Douglas-fir tussock moth: a synthesis. Tech. Bull. 1585. Washington, DC: U.S. Department of Agriculture, Forest Service. 331 p.

**Brooks, M.L.; Brown, C.S.; Chambers, J.C.; D'Antonio, C.M.; Keeley, J.E.; Belnap, J. 2016.** Exotic annual *Bromus* invasions: comparisons among species and ecoregions in the western United States. In: Germino, M.J.; Chambers, J.C.; Brown, C.S., eds. Exotic brome-grasses in arid and semiarid ecosystems of the western US. Cham, Switzerland: Springer International Publishing: 11–60. http://dx.doi.org/10.1007/978-3-319-24930-8_2.

**Brown, M.J.; Kertis, J.; Huff, M.H. 2013.** Natural tree regeneration and coarse woody debris dynamics after a forest fire in the western Cascade Range. Res. Pap. PNW-RP-592. Portland, OR: U.S. Department of Agriculture, Forest Service, Pacific Northwest Research Station. 50 p. doi:10.2737/pnw-rp-592.

**Brown, P.M.; Baxter, W.T. 2003.** Fire history in coast redwood forests of the Mendocino Coast California. Northwest Science. 77(2): 147–158.

**Brown, P.M.; Cook, B. 2006.** Early settlement forest structure in Black Hills ponderosa pine forests. Forest Ecology and Management. 223(1–3): 284–290. doi:10.1016/j.foreco.2005.11.008.

**Brown, P.M.; Kaye, M.W.; Buckley, D. 1999.** Fire history in Douglas-fir and coast redwood forests at Point Reyes National Seashore, California. Northwest Science. 73(3): 205–216.

**Brown, P.M.; Swetnam, T.W. 1994.** A cross-dated fire history from coast redwood near Redwood National Park, California. Canadian Journal of Forest Research. 24(1): 21–31. doi:10.1139/x94-004.

**Brown, T.J.; Hall, B.L.; Mohrle, C.R.; Reinbold, H.J. 2002.** Coarse assessment of federal wildland fire occurrence data: report for the National Wildfire Coordinating Group. CEFA Report 02-04. Reno, NV: Desert Research Institute, Division of Atmospheric Sciences, Program for Climate, Ecosystem and Fire Applications. 31 p.

**Buchanan, J.B.; Irwin, L.L.; McCutchen, E.L. 1995.** Within-stand nest site selection by spotted owls in the eastern Washington Cascades. Journal of Wildlife Management. 59(2): 301–310. doi:10.2307/3808943.

**Bull, E.L. 2002.** The value of coarse woody debris to vertebrates in the Pacific Northwest. In: Laudenslayer, W.F., Jr.; Shea, P.J.; Valentine, B.E.; Weatherspoon, C.P.; Lisle, T.E., eds. Proceedings of the symposium on the ecology and management of dead wood in western forests. Gen. Tech. Rep. PSW-GTR-181. Albany, CA: U.S. Department of Agriculture, Forest Service, Pacific Southwest Research Station: 171–178.

**Bull, E.L.; Aubry, K.B.; Wales, B.C. 2001.** Effects of disturbance on forest carnivores of conservation concern in eastern Oregon and Washington. Northwest Science. 75: 180–184.

**Bunnell, F.L. 2013.** Sustaining cavity-using species: patterns of cavity use and implications to forest management. ISRN Forestry. 2013: 1–33. doi:10.1155/2013/457698.

**Burton, J.I.; Ganio, L.M.; Puettmann, K.J. 2014.** Multi-scale spatial controls of understory vegetation in Douglas-fir–western hemlock forests of western Oregon, USA. Ecosphere. 5(12): art151.doi:10.1890/es14-00049.1.

**Busse, M.D.; Cochran, P.H.; Barrett, J.W. 1996.** Changes in ponderosa pine site productivity following removal of understory vegetation. Soil Science Society of America Journal. 60(6): 1614. doi:10.2136/sssaj1996.03615995006000060004x.

**Cale, W.G.; Henebry, G.M.; Yeakley, J.A. 1989.** Inferring process from pattern in natural communities. BioScience. 39(9): 600–605. doi:10.2307/1311089.

**Calkin, D.E.; Thompson, M.P.; Finney, M.A. 2015.** Negative consequences of positive feedbacks in US wildfire management. Forest Ecosystems. 2:9 doi:10.1186/s40663-015-0033-8.

**Camp, A.; Oliver, C.; Hessburg, P.; Everett, R. 1997.** Predicting late-successional fire refugia pre-dating European settlement in the Wenatchee mountains. Forest Ecology and Management. 95(1): 63–77. doi:10.1016/s0378-1127(97)00006-6.

**Camp, A.E. 1999.** Age structure and species composition changes resulting from altered disturbance regimes on the eastern slopes of the Cascades Range, Washington. Journal of Sustainable Forestry. 9(3–4): 39–67. doi:10.1300/j091v09n03_03.

**Campbell, J.; Donato, D.; Azuma, D.; Law, B. 2007.** Pyrogenic carbon emission from a large wildfire in Oregon, United States. Journal of Geophysical Research. 112(G4) doi:10.1029/2007jg000451.

**Campbell, J.L.; Donato, D.C.; Fontaine, J.B. 2016a.** Effects of post-fire logging on fuel dynamics in a mixed-conifer forest, Oregon, USA: a 10-year assessment. International Journal of Wildland Fire. 25(6): 646. doi:10.1071/wf15119.

**Campbell, J.L.; Fontaine, J.B.; Donato, D.C. 2016b.** Carbon emissions from decomposition of fire-killed trees following a large wildfire in Oregon, United States. Journal of Geophysical Research: Biogeosciences. 121(3): 718–730. doi:10.1002/2015jg003165.

**Cansler, C.A.; McKenzie, D. 2014.** Climate, fire size, and biophysical setting control fire severity and spatial pattern in the northern Cascade Range, USA. Ecological Applications. 24(5): 1037–1056. doi:10.1890/13-1077.1.

**Carey, A.B. 2003.** Biocomplexity and restoration of biodiversity in temperate coniferous forest: inducing spatial heterogeneity with variable-density thinning. Forestry. 76(2): 127–136. doi:10.1093/forestry/76.2.127.

**Carlson, C.H.; Dobrowski, S.Z.; Safford, H.D. 2012.** Variation in tree mortality and regeneration affect forest carbon recovery following fuel treatments and wildfire in the Lake Tahoe Basin, California, USA. Carbon Balance and Management. 7: 7. doi:10.1186/1750-0680-7-7.

**Carroll, C. 2010.** Role of climatic niche models in focal-species-based conservation planning: assessing potential effects of climate change on northern spotted owl in the Pacific Northwest, USA. Biological Conservation. 143(6): 1432–1437. doi:10.1016/j.biocon.2010.03.018.

**Carroll, C.; Dunk, J.R.; Moilanen, A. 2010.** Optimizing resiliency of reserve networks to climate change: multispecies conservation planning in the Pacific Northwest, USA. Global Change Biology. 16(3): 891–904.

**Certini, G. 2005.** Effects of fire on properties of forest soils: a review. Oecologia. 143(1): 1–10. doi:10.1007/s00442-004-1788-8.

**Chen, J.; Franklin, J.F.; Spies, T.A. 1992.** Vegetation responses to edge environments in old-growth Douglas-fir forests. Ecological Applications. 2(4): 387–396. doi:10.2307/1941873.

**Christensen, N.L.; Bartuska, A.M.; Brown, J.H.; Carpenter, S.; D'Antonio, C.; Francis, R.; Franklin, J.F.; MacMahon, J.A.; Noss, R.F.; Parsons, D.J.; Peterson, C.H.; Turner, M.G.; Woodmansee, R.G. 1996.** The report of the Ecological Society of America Committee on the scientific basis for ecosystem management. Ecological Applications. 6(3): 665–691. doi:10.2307/2269460.

**Churchill, D.J.; Larson, A.J.; Dahlgreen, M.C.; Franklin, J.F.; Hessburg, P.F.; Lutz, J.A. 2013.** Restoring forest resilience: from reference spatial patterns to silvicultural prescriptions and monitoring. Forest Ecology and Management. 291: 442–457. doi:10.1016/j.foreco.2012.11.007.

**Cissel, J.H.; Anderson, P.D.; Olson, D.; Puettmann, K.J.; Berryman, S.; Chan, S.; Thompson, C. 2006.** BLM density management and riparian buffer study: establishment report and study plan. Scientific Investigations Report 2006-5087. Reston, VA: U.S. Department of the Interior, Geological Survey. 151 p.

**Cissel, J.H.; Mayo, J.H.; Garman, S.L.; Swanson, F.J. 2002.** Application of landscape objectives to stand-level silviculture: Blue River, Oregon. In: Parker, S.; Hummel, S.S., eds. Beyond 2001: a silvicultural odyssey to sustaining terrestrial and aquatic ecosystems. Proceedings of the 2001 national silviculture workshop. Gen. Tech. Rep. PNW-GTR-546. Portland, OR: U.S. Department of Agriculture, Forest Service, Pacific Northwest Research Station: 21–37.

**Cissel, J.H.; Swanson, F.J.; Weisberg, P.J. 1999.** Landscape management using historical fire regimes: Blue River, Oregon. Ecological Applications. 9(4): 1217–1231. doi:10.1890/1051-0761(1999)009[1217:lmuhfr]2.0.co;2.

**Cobb, R.C.; Rizzo, D.M.; Frangioso, K.; Hartsough, P.; Klein, J.; Swezy, M.; Williams, A.; Sanders, C.; Frankel, S.J. 2017.** Restoration of Mt. Tamalpais forests destroyed by the sudden oak death pathogen. In: Abstracts book, sixth sudden oak death science symposium. Gen. Tech. Rep. GTR-PSW-255. Albany, CA: U.S. Department of Agriculture, Forest Service, Pacific Southwest Research Station: 23.

**Cocking, M.I.; Varner, J.M.; Knapp, E.E. 2014.** Long-term effects of fire severity on oak–conifer dynamics in the southern Cascades. Ecological Applications. 24(1): 94–107. doi:10.1890/13-0473.1.

**Cocking, M.I.; Varner, J.M.; Sherriff, R.L. 2012.** California black oak responses to fire severity and native conifer encroachment in the Klamath Mountains. Forest Ecology and Management. 270: 25–34. doi:10.1016/j.foreco.2011.12.039.

**Cohen, W.B.; Spies, T.A.; Alig, R.J.; Oetter, D.R.; Maiersperger, T.K.; Fiorella, M. 2002.** Characterizing 23 years (1972–95) of stand replacement disturbance in western Oregon forests with Landsat imagery. Ecosystems. 5(2): 122–137. doi:10.1007/s10021-001-0060-x.

**Cole, E.C.; Newton, M. 1987.** Fifth-year responses of Douglas-fir to crowding and nonconiferous competition. Canadian Journal of Forest Research. 17(2): 181–186. doi:10.1139/x87-031.

**Collins, B.M.; Kelly, M.; van Wagtendonk, J.W.; Stephens, S.L. 2006.** Spatial patterns of large natural fires in Sierra Nevada wilderness areas. Landscape Ecology. 22(4): 545–557. doi:10.1007/s10980-006-9047-5.

**Collins, B.M.; Miller, J.D.; Thode, A.E.; Kelly, M.; van Wagtendonk, J.W.; Stephens, S.L. 2008.** Interactions among wildland fires in a long-established Sierra Nevada natural fire area. Ecosystems. 12(1): 114–128. doi:10.1007/s10021-008-9211-7.

**Collins, B.M.; Roller, G.B. 2013.** Early forest dynamics in stand-replacing fire patches in the northern Sierra Nevada, California, USA. Landscape Ecology. 28(9): 1801–1813. doi:10.1007/s10980-013-9923-8.

**Collins, B.M.; Stephens, S.L. 2010.** Stand-replacing patches within a 'mixed severity' fire regime: quantitative characterization using recent fires in a long-established natural fire area. Landscape Ecology. 25(6): 927–939. doi:10.1007/s10980-010-9470-5.

**Collins, B.M.; Stephens, S.L.; Moghaddas, J.J.; Battles, J. 2010.** Challenges and approaches in planning fuel treatments across fire-excluded forested landscapes. Journal of Forestry. 108(1): 24–31.

**Colombaroli, D.; Gavin, D.G. 2010.** Highly episodic fire and erosion regime over the past 2,000 y in the Siskiyou Mountains, Oregon. Proceedings of the National Academy of Sciences of the United States of America. 107(44): 18909–18914. doi:10.1073/pnas.1007692107.

**Compton, J.E.; Church, M.R.; Larned, S.T.; Hogsett, W.E. 2003.** Nitrogen export from forested watersheds in the Oregon Coast Range: the role of $N_2$-fixing red alder. Ecosystems. 6(8): 773–785. doi:10.1007/s10021-002-0207-4.

**Coppoletta, M.; Merriam, K.E.; Collins, B.M. 2015.** Post-fire vegetation and fuel development influences fire severity patterns in reburns. Ecological Applications. 26(3): 686–699. doi:10.1890/15-0225.1.

**Countryman, C.C. 1955.** Old-growth conversion also converts fire climate. In: Proceedings of the Society of American Foresters annual meeting. Portland, OR: Society of American Foresters: 158–160.

**Creutzburg, M.K.; Scheller, R.M.; Lucash, M.S.; LeDuc, S.D.; Johnson, M.G. 2017.** Forest management scenarios in a changing climate: trade-offs between carbon, timber, and old forest. Ecological Applications. 27(2): 503–518. doi:10.1002/eap.1460.

**Crookston, N.L.; Dixon, G.E. 2005.** The forest vegetation simulator: a review of its structure, content, and applications. Computers and Electronics in Agriculture. 49(1): 60–80. doi:10.1016/j.compag.2005.02.003.

**Crotteau, J.S.; Morgan Varner, J.; Ritchie, M.W. 2013.** Post-fire regeneration across a fire severity gradient in the southern Cascades. Forest Ecology and Management. 287: 103–112. doi:10.1016/j.foreco.2012.09.022.

**Curtis, R.O.; Marshall, D.D. 1986.** Levels-of-growing-stock cooperative study in Douglas-fir: report no. 08—The LOGS Study, 20-year results. Res. Pap. PNW-RP-356. Portland, OR: U.S. Department of Agriculture, Forest Service, Pacific Northwest Research Station. 124 p. doi:10.2737/pnw-rp-356.

**Cushman, S.A.; McKelvey, K.S.; Flather, C.H.; McGarigal, K. 2008.** Do forest community types provide a sufficient basis to evaluate biological diversity? Frontiers in Ecology and the Environment. 6(1): 13–17. doi:10.1890/070039.

**Dale, V.H.; Swanson, F.J.; Crisafulli, C.M. 2005.** Disturbance, survival, and succession: understanding ecological responses to the 1980 eruption of Mount St. Helens. In: Ecological responses to the 1980 eruption of Mount St. Helens. New York: Springer: 3–11.

**Dalton, M.M.; Mote, P.W.; Snover, A.K., eds. 2013.** Climate change in the Northwest: implications for our landscapes, waters, and communities. Washington, DC: Island Press. 270 p.

**D'Amato, A.W.; Palik, B.J.; Franklin, J.F.; Foster, D.R. 2017.** Exploring the origins of ecological forestry in North America. Journal of Forestry. 115(2): 126–127. doi:10.5849/jof.16-013.

**D'Antonio, C.; Meyerson, L.A. 2002.** Exotic plant species as problems and solutions in ecological restoration: a synthesis. Restoration Ecology. 10(4): 703–713. doi:10.1046/j.1526-100x.2002.01051.x.

**Davis, L.R.; Puettmann, K.J. 2009.** Initial response of understory vegetation to three alternative thinning treatments. Journal of Sustainable Forestry. 28(8): 904–934. doi:10.1080/10549810903344611.

**Davis, L.R.; Puettmann, K.J.; Tucker, G.F. 2007.** Overstory response to alternative thinning treatments in young Douglas-fir forests of western Oregon. Northwest Science. 81(1): 1–14. doi:10.3955/0029-344x-81.1.1.

**Davis, R.J.; Ohmann, J.L.; Kennedy, R.E.; Cohen, W.B.; Gregory, M.J.; Yang, Z.; Roberts, H.M.; Gray, A.N.; Spies, T.A. 2015.** Northwest Forest Plan—the first 20 years (1994–2013): status and trends of late-successional and old-growth forests. Gen. Tech. Rep. PNW-GTR-911. Portland, OR: U.S. Department of Agriculture, Forest Service, Pacific Northwest Research Station. 112 p. doi:10.2737/pnw-gtr-911.

**DellaSala, D.A.; Anthony, R.G.; Bond, M.L.; Fernandez, E.S.; Frissell, C.A.; Hanson, C.T.; Spivak, R. 2013.** Alternative views of a restoration framework for federal forests in the Pacific Northwest. Journal of Forestry. 111(6): 420–429. doi:10.5849/jof.13-040.

**DellaSala, D.A.; Hanson, C.T. 2015.** Preface. In: DellaSala, D.A.; Hanson, C.T., eds. The ecological importance of mixed-severity fires: nature's phoenix. Amsterdam: Elsevier BV: xxiii–xxxviii. http://dx.doi.org/10.1016/b978-0-12-802749-3.10000-3.

**DellaSala, D.A.; Martin, A.; Spivak, R.; Schulke, T.; Bird, B.; Criley, M.; van Daalen, C.; Kreilick, J.; Brown, R.; Aplet, G. 2003.** A citizen's call for ecological forest restoration: forest restoration principles and criteria. Ecological Restoration. 21(1): 14–23. doi:10.3368/er.21.1.14.

**DeLuca, T.H.; Aplet, G.H. 2008.** Charcoal and carbon storage in forest soils of the Rocky Mountain West. Frontiers in Ecology and the Environment. 6(1): 18–24. doi:10.1890/070070.

**DeMeo, T.E.; Swanson, F.J.; Smith, E.B.; Buttrick, S.C.; Kertis, J.; Rice, J.; Ringo, C.D.; Waltz, A.; Zanger, C.; Friesen, C.A.; Cissel, J.H. 2012.** Applying historical fire-regime concepts to forest management in the western United States: three case studies. In: Wiens, J.A.; Hayward, G.D.; Hugh, D.; Giffen, C., eds. Historical environmental variation in conservation and natural resource management. Chichester, West Sussex, United Kingdom: John Wiley & Sons, Ltd: 194–204. http://dx.doi.org/10.1002/9781118329726.ch13.

**DeSiervo, M.H.; Jules, E.S.; Safford, H.D. 2015.** Disturbance response across a productivity gradient: postfire vegetation in serpentine and nonserpentine forests. Ecosphere. 6(4): art60. doi:10.1890/es14-00431.1.

**Devine, W.D.; Harrington, C.A. 2006.** Changes in Oregon white oak (*Quercus garryana* Dougl. ex Hook.) following release from overtopping conifers. Trees. 20(6): 747–756. doi:10.1007/s00468-006-0089-8.

**Dickman, A.; Cook, S. 1989.** Fire and fungus in a mountain hemlock forest. Canadian Journal of Botany. 67(7): 2005–2016. doi:10.1139/b89-254.

**Dodson, E.K.; Ares, A.; Puettmann, K.J. 2012.** Early responses to thinning treatments designed to accelerate late successional forest structure in young coniferous stands of western Oregon, USA. Canadian Journal of Forest Research. 42(2): 345–355. doi:10.1139/x11-188.

**Dodson, E.K.; Burton, J.I.; Puettmann, K.J. 2014.** Multiscale controls on natural regeneration dynamics after partial overstory removal in Douglas-fir forests in western Oregon, USA. Forest Science. 60(5): 953–961. doi:10.5849/forsci.13-011.

**Dodson, E.K.; Peterson, D.W. 2010.** Dry coniferous forest restoration and understory plant diversity: the importance of community heterogeneity and the scale of observation. Forest Ecology and Management. 260(10): 1702–1707. doi:10.1016/j.foreco.2010.08.012.

**Dodson, E.K.; Peterson, D.W.; Harrod, R.J. 2008.** Understory vegetation response to thinning and burning restoration treatments in dry conifer forests of the eastern Cascades, USA. Forest Ecology and Management. 255(8–9): 3130–3140. doi:10.1016/j.foreco.2008.01.026.

**Dodson, E.K.; Root, H.T. 2013.** Conifer regeneration following stand-replacing wildfire varies along an elevation gradient in a ponderosa pine forest, Oregon, USA. Forest Ecology and Management. 302: 163–170. doi:10.1016/j.foreco.2013.03.050.

**Dolph, K.L.; Mori, S.R.; Oliver, W.W. 1995.** Long-term response of old-growth stands to varying levels of partial cutting in the eastside pine type. Western Journal of Applied Forestry. 10(3): 101–108.

**Donato, D.C. 2006.** Post-wildfire logging hinders regeneration and increases fire risk. Science. 311(5759): 352–352. doi:10.1126/science.1122855.

**Donato, D.C.; Campbell, J.L.; Franklin, J.F. 2011.** Multiple successional pathways and precocity in forest development: Can some forests be born complex? Journal of Vegetation Science. 23(3): 576–584. doi:10.1111/j.1654-1103.2011.01362.x.

**Donato, D.C.; Fontaine, J.B.; Campbell, J.L. 2016.** Burning the legacy? Influence of wildfire reburn on dead wood dynamics in a temperate conifer forest. Ecosphere. 7(5): e01341. doi:10.1002/ecs2.1341.

**Donato, D.C.; Fontaine, J.B.; Campbell, J.L.; Robinson, W.D.; Kauffman, J.B.; Law, B.E. 2009.** Conifer regeneration in stand-replacement portions of a large mixed-severity wildfire in the Klamath–Siskiyou Mountains. Canadian Journal of Forest Research. 39(4): 823–838. doi:10.1139/x09-016.

**Donato, D.C.; Fontaine, J.B.; Kauffman, J.B.; Robinson, W.D.; Law, B.E. 2013.** Fuel mass and forest structure following stand-replacement fire and post-fire logging in a mixed-evergreen forest. International Journal of Wildland Fire. 22(5): 652. doi:10.1071/wf12109.

**Driscoll, D.A.; Lindenmayer, D.B.; Bennett, A.F.; Bode, M.; Bradstock, R.A.; Cary, G.J.; Clarke, M.F.; Dexter, N.; Fensham, R.; Friend, G.; Gill, M.; James, S.; Kay, G.; Keith, D.A.; MacGregor, C.; Russell-Smith, J.; Salt, D.; Watson, J.E.M.; Williams, R.J.; York, A. 2010.** Fire management for biodiversity conservation: key research questions and our capacity to answer them. Biological Conservation. 143(9): 1928–1939. doi:10.1016/j.biocon.2010.05.026.

**Dunbar-Irwin, M.; Safford, H. 2016.** Climatic and structural comparison of yellow pine and mixed-conifer forests in northern Baja California (México) and the eastern Sierra Nevada (California, USA). Forest Ecology and Management. 363: 252–266. doi:10.1016/j.foreco.2015.12.039.

**Dunham, J.B.; Young, M.K.; Gresswell, R.E.; Rieman, B.E. 2003.** Effects of fire on fish populations: landscape perspectives on persistence of native fishes and nonnative fish invasions. Forest Ecology and Management. 178(1–2): 183–196. doi:10.1016/s0378-1127(03)00061-6.

**Dunham, P.A. 2008.** Incidence of insects, diseases, and other damaging agents in Oregon forests. Resour. Bull. PNW-RB-257. Portland, OR: U.S. Department of Agriculture, Forest Service, Pacific Northwest Research Station. 89 p. doi:10.2737/pnw-rb-257.

**Dunn, C.J. 2015.** Mixed-severity fire effects on biological legacies and vegetation response in *Pseudotsuga* forests of western Oregon's Central Cascades, USA. Corvallis, OR: Oregon State University. 126 p. Ph.D. dissertation.

**Dunn, C.J.; Bailey, J.D. 2015.** Modeling the direct effects of salvage logging on long-term temporal fuel dynamics in dry-mixed conifer forests. Forest Ecology and Management. 341: 93–109. doi:10.1016/j.foreco.2015.01.002.

**Dunn, C.J.; Bailey, J.D. 2016.** Tree mortality and structural change following mixed-severity fire in *Pseudotsuga* forests of Oregon's western Cascades, USA. Forest Ecology and Management. 365: 107–118. doi:10.1016/j.foreco.2016.01.031.

**Dunwiddie, P.W. 1986.** A 6000-year record of forest history on Mount Rainier, Washington. Ecology. 67(1): 58–68. doi:10.2307/1938503.

**Ellison, A.M.; Bank, M.S.; Clinton, B.D.; Colburn, E.A.; Elliott, K.; Ford, C.R.; Foster, D.R.; Kloeppel, B.D.; Knoepp, J.D.; Lovett, G.M.; Mohan, J.; Orwig, D.A.; Rodenhouse, N.L.; Sobczak, W.V.; Stinson, K.A.; Stone, J.K.; Swan, C.M.; Thompson, J.; Von Holle, B.; Webster, J.R. 2005.** Loss of foundation species: consequences for the structure and dynamics of forested ecosystems. Frontiers in Ecology and the Environment. 3(9): 479–486. doi:10.1890/1540-9295(2005)003[0479:lofscf]2.0.co;2.

**Engber, E.A.; Varner Iii, J.M.; Arguello, L.A.; Sugihara, N.G. 2011.** The effects of conifer encroachment and overstory structure on fuels and fire in an oak woodland landscape. Fire Ecology. 7(2): 32–50. doi:10.4996/fireecology.0702032.

**Estes, B.L.; Knapp, E.E.; Skinner, C.N.; Miller, J.D.; Preisler, H.K. 2017.** Factors influencing fire severity under moderate burning conditions in the Klamath mountains, northern California, USA. Ecosphere. 8(5): e01794. doi:10.1002/ecs2.1794.

**Estes, B.L.; Knapp, E.E.; Skinner, C.N.; Uzoh, F.C.C. 2012.** Seasonal variation in surface fuel moisture between unthinned and thinned mixed conifer forest, northern California, USA. International Journal of Wildland Fire. 21(4): 428. doi:10.1071/wf11056.

**Everett, R.L.; Schellhaas, R.; Keenum, D.; Spurbeck, D.; Ohlson, P. 2000.** Fire history in the ponderosa pine/Douglas-fir forests on the east slope of the Washington Cascades. Forest Ecology and Management. 129(1–3): 207–225. doi:10.1016/s0378-1127(99)00168-1.

**Fahnestock, G.R.; Agee, J.K. 1983.** Biomass consumption and smoke production by prehistoric and modern forest fires in western Washington. Journal of Forestry. 81(10): 653–657.

**Fahrig, L. 2013.** Rethinking patch size and isolation effects: the habitat amount hypothesis. Journal of Biogeography. 40(9): 1649–1663. doi:10.1111/jbi.12130.

**Fairweather, S.E. 2004.** Recommendation of growth model and tree volume estimation system to the H&H core team for the Tillamook, Forest Grove, Astoria, West Oregon, and Western Lane districts. In: Oregon Department of Forestry harvest scheduling model yield table creation project. Portland, OR: Mason, Bruce and Girard Inc: 23.

**Falk, D.A.; Heyerdahl, E.K.; Brown, P.M.; Farris, C.; Fulé, P.Z.; McKenzie, D.; Swetnam, T.W.; Taylor, A.H.; Van Horne, M.L. 2011.** Multi-scale controls of historical forest-fire regimes: new insights from fire-scar networks. Frontiers in Ecology and the Environment. 9(8): 446–454. doi:10.1890/100052.

**Fettig, C.J.; Klepzig, K.D.; Billings, R.F.; Munson, A.S.; Nebeker, T.E.; Negrón, J.F.; Nowak, J.T. 2007.** The effectiveness of vegetation management practices for prevention and control of bark beetle infestations in coniferous forests of the western and southern United States. Forest Ecology and Management. 238(1–3): 24–53. doi:10.1016/j.foreco.2006.10.011.

**Filip, G.M.; Goheen, D.J. 1984.** Root diseases cause severe mortality in white and grand fir stands of the Pacific Northwest. Forest Science. 30: 138–142.

**Finney, M.A.; Martin, R.E. 1989.** Fire history in a *Sequoia sempervirens* forest at Salt Point State Park, California. Canadian Journal of Forest Research. 19(11): 1451–1457. doi:10.1139/x89-221.

**Finney, M.A.; Seli, R.C.; McHugh, C.W.; Ager, A.A.; Bahro, B.; Agee, J.K. 2007.** Simulation of long-term landscape-level fuel treatment effects on large wildfires. International Journal of Wildland Fire. 16(6): 712. doi:10.1071/wf06064.

**Fischer, J.; Lindenmayer, D.B.; Manning, A.D. 2006.** Biodiversity, ecosystem function, and resilience: ten guiding principles for commodity production landscapes. Frontiers in Ecology and the Environment. 4(2): 80–86. doi:10.1890/1540-9295(2006)004[0080:befart]2.0.co;2.

**Flitcroft, R.L.; Falke, J.A.; Reeves, G.H.; Hessburg, P.F.; McNyset, K.M.; Benda, L.E. 2016.** Wildfire may increase habitat quality for spring Chinook salmon in the Wenatchee River subbasin, WA, USA. Forest Ecology and Management. 359: 126–140. doi:10.1016/j.foreco.2015.09.049.

**Fontaine, J.B.; Donato, D.C.; Campbell, J.L.; Martin, J.G.; Law, B.E. 2010.** Effects of post-fire logging on forest surface air temperatures in the Siskiyou Mountains, Oregon, USA. Forestry. 83(5): 477–482. doi:10.1093/forestry/cpq030.

**Fontaine, J.B.; Donato, D.C.; Robinson, W.D.; Law, B.E.; Kauffman, J.B. 2009.** Bird communities following high-severity fire: response to single and repeat fires in a mixed-evergreen forest, Oregon, USA. Forest Ecology and Management. 257(6): 1496–1504. doi:10.1016/j.foreco.2008.12.030.

**Forest Ecosystem Management Assessment Team [FEMAT]. 1993.** Forest ecosystem management: an ecological, economic, and social assessment. Portland, OR: U.S. Department of Agriculture; U.S. Department of the Interior [and others]. [Irregular pagination].

**Forman, R.T.T.; Alexander, L.E. 1998.** Roads and their major ecological effects. Annual Review of Ecology, Evolution, and Systematics. 29(1): 207–231. doi:10.1146/annurev.ecolsys.29.1.207.

**Forrestel, A.B.; Ramage, B.S.; Moody, T.; Moritz, M.A.; Stephens, S.L. 2015.** Disease, fuels and potential fire behavior: impacts of sudden oak death in two coastal California forest types. Forest Ecology and Management. 348: 23–30. doi:10.1016/j.foreco.2015.03.024.

**Forsman, E.D.; Meslow, E.C.; Wight, H.M. 1984.** Distribution and biology of the spotted owl in Oregon. Wildlife Monographs. 87: 3–64.

**Foster, D.R.; Knight, D.H.; Franklin, J.F. 1998.** Landscape patterns and legacies resulting from large, infrequent forest disturbances. Ecosystems. 1(6): 497–510. doi:10.1007/s100219900046.

**Franklin, A.B.; Anderson, D.R.; Gutiérrez, R.J.; Burnham, K.P. 2000.** Climate, habitat quality, and fitness in northern spotted owl populations in northwestern California. Ecological Monographs. 70(4): 539–590. doi:10.1890/0012-9615(2000)070[0539:chqafi]2.0.co;2.

**Franklin, J.F. 1989.** The "new forestry." Journal of Soil and Water Conservation. 44(6): 549.

**Franklin, J.F. 1997.** Ecosystem management: an overview. In: Boyce, M.S.; Haney, A., eds. Ecosystem management: applications for sustainable forest and wildlife resources. New Haven, CT: Yale University Press: 21–53.

**Franklin, J.F.; Dyrness, C.T. 1973.** Natural vegetation of Oregon and Washington. Gen. Tech. Rep. PNW-GTR-8. Portland, OR: U.S. Department of Agriculture, Forest Service, Pacific Northwest Forest and Range Experiment Station. 417 p.

**Franklin, J.F.; Forman, R.T.T. 1987.** Creating landscape patterns by forest cutting: Ecological consequences and principles. Landscape Ecology. 1(1): 5–18. doi:10.1007/bf02275261.

**Franklin, J.F.; Hemstrom, M.A. 1981.** Aspects of succession in the coniferous forests of the Pacific Northwest. In: West, D.C.; Shugart, H.H.; Botkin, D.B., eds. Forest succession: concepts and application. New York: Springer: 212–229. doi:10.1007/978-1-4612-5950-3_14.

**Franklin, J.F.; Hemstrom, M.A.; Van Pelt, R.; Buchanan, J.B.; Hull, S. 2008.** The case for active management of dry forest types in eastern Washington: perpetuating and creating old forest structures and functions. Olympia, WA: Washington Department of Natural Resources. 97 p.

**Franklin, J.F.; Hemstrom, M.; Van Pelt, R.; Buchanan, J.; Hull, S.; Crawford, R.; Curry, S.; Obermeyer, W. 2007a.** Extent and distribution of old forest conditions on DNR-managed state trust lands in eastern Washington. Olympia, WA: Washington State Department of Natural Resources. 32 p.

**Franklin, J.F.; Johnson, K.N. 2012.** A restoration framework for federal forests in the Pacific Northwest. Journal of Forestry. 110(8): 429–439. doi:10.5849/jof.10-006.

**Franklin, J.F.; Johnson, K.N.; Churchill, D.J.; Hagmann, K.; Johnson, D.; Johnston, J. 2013.** Restoration of dry forests in eastern Oregon: a field guide. Portland, OR: The Nature Conservancy. 202 p.

**Franklin, J.F.; Johnson, K.N.; Johnson, D.L. 2018.** Ecological forest management. Long Grove, IL: Waveland Press.

**Franklin, J.F.; Mitchell, R.J.; Palik, B.J. 2007b.** Natural disturbance and stand development principles for ecological forestry. Gen. Tech. Rep. NRS-GTR-19. Newtown Square, PA: U.S. Department of Agriculture, Forest Service, Northern Research Station. 44 p. doi:10.2737/nrs-gtr-19.

**Franklin, J.F.; Spies, T.A. 1991.** Composition, function, and structure of old-growth Douglas-fir forests. In: Ruggiero, L.F.; Aubry, K.B.; Carey, A.B.; Huff, M.H., eds. Wildlife and vegetation of unmanaged Douglas-fir forests. Gen. Tech. Rep. PNW-GTR-285. Portland, OR: U.S. Department of Agriculture, Forest Service, Pacific Northwest Research Station: 71–80.

**Franklin, J.F.; Spies, T.A.; Van Pelt, R. 2005.** Definition and inventory of old growth forests on DNR-managed state lands. Olympia, WA: Report to the Washington Department of Natural Resources. 22 p.

**Franklin, J.F.; Spies, T.A.; Van Pelt, R.; Carey, A.B.; Thornburgh, D.A.; Berg, D.R.; Lindenmayer, D.B.; Harmon, M.E.; Keeton, W.S.; Shaw, D.C.; Bible, K.; Chen, J. 2002.** Disturbances and structural development of natural forest ecosystems with silvicultural implications, using Douglas-fir forests as an example. Forest Ecology and Management. 155(1–3): 399–423. doi:10.1016/s0378-1127(01)00575-8.

**Freund, J.A.; Franklin, J.F.; Larson, A.J.; Lutz, J.A. 2014.** Multi-decadal establishment for single-cohort Douglas-fir forests. Canadian Journal of Forest Research. 44(9): 1068–1078. doi:10.1139/cjfr-2013-0533.

**Froehlich, H.A.; Aulerich, D.E.; Curtis, R. 1981.** Designing skid trail systems to reduce soil impacts from tractive logging machines. Corvallis, OR: Oregon State University, School of Forestry, Forest Research Laboratory. 15 p.

**Fry, D.L.; Stephens, S.L. 2006.** Influence of humans and climate on the fire history of a ponderosa pine-mixed conifer forest in the southeastern Klamath Mountains, California. Forest Ecology and Management. 223(1–3): 428–438. doi:10.1016/j.foreco.2005.12.021.

**Fry, D.L.; Stephens, S.L.; Collins, B.M.; North, M.P.; Franco-Vizcaíno, E.; Gill, S.J. 2014.** Contrasting spatial patterns in active-fire and fire-suppressed Mediterranean climate old-growth mixed conifer forests. PLoS ONE. 9(2): e88985. doi:10.1371/journal.pone.0088985.

**Fulé, P.Z.; Swetnam, T.W.; Brown, P.M.; Falk, D.A.; Peterson, D.L.; Allen, C.D.; Aplet, G.H.; Battaglia, M.A.; Binkley, D.; Farris, C.; Keane, R.E.; Margolis, E.Q.; Grissino-Mayer, H.; Miller, C.; Sieg, C.H.; Skinner, C.; Stephens, S.L.; Taylor, A. 2013.** Unsupported inferences of high-severity fire in historical dry forests of the western United States: response to Williams and Baker. Global Ecology and Biogeography. 23(7): 825–830. doi:10.1111/geb.12136.

**Furniss, M.M. 2014a.** The Douglas-fir beetle in western forests: a historical perspective—part 1. American Entomologist. 60(2): 84–96. doi:10.1093/ae/60.2.84.

**Furniss, M.M. 2014b.** The Douglas-fir beetle in western forests a historical perspective—part 2. American Entomologist. 60(3): 166–181. doi:10.1093/ae/60.3.166.

**Furniss, M.M.; Kegley, S. 2014.** Douglas-fir beetle. Portland, OR: U.S. Department of Agriculture, Forest Service, Pacific Northwest Region.

**Furniss, R.L.; Carolin, V.M. 1977.** Western forest insects. Misc. Pub. 1339. Washington DC: U.S. Department of Agriculture, Forest Service. 655 p.

**Ganzlin, P.W.; Gundale, M.J.; Becknell, R.E.; Cleveland, C.C. 2016.** Forest restoration treatments have subtle long-term effects on soil C and N cycling in mixed conifer forests. Ecological Applications. 26(5): 1503–1516. doi:10.1002/15-1100.

**Garman, S.L.; Cissel, J.H.; Mayo, J.H. 2003.** Accelerating development of late-successional conditions in young managed Douglas-fir stands: a simulation study. Gen. Tech. Rep. PNW-GTR-557. Portland, OR: U.S. Department of Agriculture, Forest Service, Pacific Northwest Research Station. 57 p. doi:10.2737/pnw-gtr-557.

**Gashwiler, J.S. 1970.** Plant and animal changes on a clearcut in west-central Oregon. Ecology. 51(6): 1018–1026. doi:10.2307/1933628.

**Gavin, D.G.; Hallett, D.J.; Hu, F.S.; Lertzman, K.P.; Prichard, S.J.; Brown, K.J.; Lynch, J.A.; Bartlein, P.; Peterson, D.L. 2007.** Forest fire and climate change in western North America: insights from sediment charcoal records. Frontiers in Ecology and the Environment. 5(9): 499–506. doi:10.1890/060161.

**Gholz, H.L. 1982.** Environmental limits on aboveground net primary production, leaf area, and biomass in vegetation zones of the Pacific Northwest. Ecology. 63(2): 469–481. doi:10.2307/1938964.

**Gibson, K.; Kegley, S.; Bentz, B. 2009.** Mountain pine beetle. Forest insect and disease leaflet 2. Washington, DC: U.S. Department of Agriculture, Forest Service. 12 p.

**Gibson, K.; Negrón, J.F. 2009.** Fire and bark beetle interactions. Gen. Tech. Rep. PNW-GTR-784. Portland, OR: U.S. Department of Agriculture, Forest Service, Pacific Northwest Research Station. 69 p.

**Goheen, E.M.; Willhite, E.A. 2006.** Field guide to the common diseases and insect pests of Oregon and Washington conifers. Portland, OR: U.S. Department of Agriculture, Forest Service, Pacific Northwest Region. 337 p.

**Gould, P.J.; Harrington, C.A.; Devine, W.D. 2011.** Growth of Oregon white oak (*Quercus garryana*). Northwest Science. 85(2): 159–171. doi:10.3955/046.085.0207.

**Gray, A.N. 2005.** Eight nonnative plants in western Oregon forests: associations with environment and management. Environmental Monitoring and Assessment. 100(1–3): 109–127. doi:10.1007/s10661-005-7060-9.

**Gray, A.N.; Franklin, J.F. 1997.** Effects of multiple fires on the structure of southwestern Washington forests. Northwest Science. 71: 174–185.

**Grumbine, R.E. 1994.** What is ecosystem management? Conservation Biology. 8: 27–38. doi:10.1046/j.1523-1739.1994.08010027.x.

**Guarín, A.; Taylor, A.H. 2005.** Drought triggered tree mortality in mixed conifer forests in Yosemite National Park, California, USA. Forest Ecology and Management. 218(1–3): 229–244. doi:10.1016/j.foreco.2005.07.014.

**Gustafsson, L.; Baker, S.C.; Bauhus, J.; Beese, W.J.; Brodie, A.; Kouki, J.; Lindenmayer, D.B.; Lõhmus, A.; Pastur, G.M.; Messier, C.; Neyland, M.; Palik, B.; Sverdrup-Thygeson, A.; Volney, W.J.A.; Wayne, A.; Franklin, J.F. 2012.** Retention forestry to maintain multifunctional forests: a world perspective. BioScience. 62(7): 633–645. doi:10.1525/bio.2012.62.7.6.

**Hadfield, J.S.; Goheen, D.J.; Filip, G.M.; Schmitt, C.L.; Harvey, R.D. 1986.** Root diseases in Oregon and Washington conifers. R6-FPM-250-86: Portland, OR: U.S. Department of Agriculture, Forest Service, Pacific Northwest Region. https://www.fs.usda.gov/detail/r6/forest-grasslandhealth/insects-diseases/?cid=fsbdev2_027376. (12 January 2018).

**Hagar, J.; Howlin, S.; Ganio, L. 2004.** Short-term response of songbirds to experimental thinning of young Douglas-fir forests in the Oregon Cascades. Forest Ecology and Management. 199(2–3): 333–347. doi:10.1016/j.foreco.2004.05.054.

**Hagmann, R.K.; Franklin, J.F.; Johnson, K.N. 2013.** Historical structure and composition of ponderosa pine and mixed-conifer forests in south-central Oregon. Forest Ecology and Management. 304: 492–504. doi:10.1016/j.foreco.2013.04.005.

**Hagmann, R.K.; Franklin, J.F.; Johnson, K.N. 2014.** Historical conditions in mixed-conifer forests on the eastern slopes of the northern Oregon Cascade Range, USA. Forest Ecology and Management. 330: 158–170. doi:10.1016/j.foreco.2014.06.044.

**Hagmann, R.K.; Johnson, D.L.; Johnson, K.N. 2017.** Historical and current forest conditions in the range of the northern spotted owl in south central Oregon, USA. Forest Ecology and Management. 389: 374–385. doi:10.1016/j.foreco.2016.12.029.

**Hall, T.E.; Slothower, M. 2009.** Cognitive factors affecting homeowners' reactions to defensible space in the Oregon Coast Range. Society & Natural Resources. 22(2): 95–110. doi:10.1080/08941920802392187.

**Hallett, D.J.; Lepofsky, D.S.; Mathewes, R.W.; Lertzman, K.P. 2003.** 11 000 years of fire history and climate in the mountain hemlock rain forests of southwestern British Columbia based on sedimentary charcoal. Canadian Journal of Forest Research. 33(2): 292–312. doi:10.1139/x02-177.

**Halofsky, J.E.; Donato, D.C.; Hibbs, D.E.; Campbell, J.L.; Cannon, M.D.; Fontaine, J.B.; Thompson, J.R.; Anthony, R.G.; Bormann, B.T.; Kayes, L.J.; Law, B.E.; Peterson, D.L.; Spies, T.A. 2011.** Mixed-severity fire regimes: lessons and hypotheses from the Klamath-Siskiyou ecoregion. Ecosphere. 2(4): art40. doi:10.1890/es10-00184.1.

**Halpern, C.B. 1988.** Early successional pathways and the resistance and resilience of forest communities. Ecology. 69(6): 1703–1715. doi:10.2307/1941148.

**Halpern, C.B. 1989.** Early successional patterns of forest species: interactions of life history traits and disturbance. Ecology. 70(3): 704–720. doi:10.2307/1940221.

**Halpern, C.B.; Halaj, J.; Evans, S.A.; Dovciak, M. 2012.** Level and pattern of overstory retention interact to shape long-term responses of understories to timber harvest. Ecological Applications. 22(8): 2049–2064. doi:10.1890/12-0299.1.

**Han, H.S.; Kellogg, L.D. 2000.** Damage characteristics in young Douglas-fir stands from commercial thinning with four timber harvesting systems. Western Journal of Applied Forestry. 15(1): 27–33.

**Hann, W.; Shlisky, A.; Havlina, D.; Schon, K.; Barrett, S.; DeMeo, T.; Pohl, K.; Menakis, J.; Hamilton, D.; Jones, J.; Levesque, M. 2004.** Interagency fire regime condition class guidebook. Boise, ID: National Interagency Fire Center. http://www.frcc.gov/. (1 April 2008).

**Hann, W.J.; Bunnell, D.L. 2001.** Fire and land management planning and implementation across multiple scales. International Journal of Wildland Fire. 10(4): 389. doi:10.1071/wf01037.

**Hann, W.J.; Jones, J.L.; Karl, M.G.; Hessburg, P.F.; Keane, R.E.; Long, D.G.; Menakis, J.P.; McNicoll, C.H.; Leonard, S.G.; Gravenmier, R.A.; Smith, B.G. 1997.** An assessment of landscape dynamics of the Basin. In: Quigley, T.M.; Arbelbide, S.J., eds. An assessment of ecosystem components in the interior Columbia Basin and portions of the Klamath and Great Basins. Gen. Tech. Rep. PNW-GTR-405. Portland, OR: U.S. Department of Agriculture, Forest Service, Pacific Northwest Research Station. 1055 p. Vol. II.

**Hansen, A.J.; Garman, S.L.; Marks, B.; Urban, D.L. 1993.** An approach for managing vertebrate diversity across multiple-use landscapes. Ecological Applications. 3(3): 481. doi:10.2307/1941917.

**Hansen, A.J.; Garman, S.L.; Weigand, J.F.; Urban, D.L.; McComb, W.C.; Raphael, M.G. 1995a.** Alternative silvicultural regimes in the Pacific Northwest: simulations of ecological and economic effects. Ecological Applications. 5(3): 535. doi:10.2307/1941965.

**Hansen, A.J.; McComb, W.C.; Vega, R.; Raphael, M.G.; Hunter, M. 1995b.** Bird habitat relationships in natural and managed forests in the west Cascades of Oregon. Ecological Applications. 5(3): 555–569. doi:10.2307/1941966.

**Hansen, E.M.; Goheen, E.M. 2000.** *Phellinus weirii* and other native root pathogens as determinants of forest structure and process in western North America. Annual Review of Phytopathology. 38(1): 515–539. doi:10.1146/annurev.phyto.38.1.515.

**Harcombe, P.A.; Greene, S.E.; Kramer, M.G.; Acker, S.A.; Spies, T.A.; Valentine, T. 2004.** The influence of fire and windthrow dynamics on a coastal spruce–hemlock forest in Oregon, USA, based on aerial photographs spanning 40 years. Forest Ecology and Management. 194(1–3): 71–82. doi:10.1016/j.foreco.2004.02.016.

**Harmon, M.; Bible, K.; Ryan, M.; Shaw, D.; Chen, H.; Klopatek, J.; Li, X. 2004.** Production, respiration, and overall carbon balance in an old-growth *Pseudotsuga-Tsuga* forest ecosystem. Ecosystems. 7(5): 498–512. doi:10.1007/s10021-004-0140-9.

**Harmon, M.E.; Ferrell, W.K.; Franklin, J.F. 1990.** Effects on carbon storage of conversion of old-growth forests to young forests. Science. 247(4943): 699–702. doi:10.1126/science.247.4943.699.

**Harr, R.D. 1986.** Effects of clearcutting on rain-on-snow runoff in western Oregon: a new look at old studies. Water Resources Research. 22(7): 1095–1100. doi:10.1029/wr022i007p01095.

**Harrod, R.J.; Peterson, D.W.; Povak, N.A.; Dodson, E.K. 2009.** Thinning and prescribed fire effects on overstory tree and snag structure in dry coniferous forests of the interior Pacific Northwest. Forest Ecology and Management. 258(5): 712–721. doi:10.1016/j.foreco.2009.05.011.

**Haugo, R.; Zanger, C.; DeMeo, T.; Ringo, C.; Shlisky, A.; Blankenship, K.; Simpson, M.; Mellen-McLean, K.; Kertis, J.; Stern, M. 2015.** A new approach to evaluate forest structure restoration needs across Oregon and Washington, USA. Forest Ecology and Management. 335: 37–50. doi:10.1016/j.foreco.2014.09.014.

**Hawkins, A.E.; Henkel, T.W. 2011.** Native forest pathogens facilitate persistence of Douglas-fir in old-growth forests of northwestern California. Canadian Journal of Forest Research. 41(6): 1256–1266. doi:10.1139/x11-053.

**Hawksworth, F.G.; Wiens, D. 1996.** Dwarf mistletoes: biology, pathology, and systematics. Agric. Handb. 709. Washington, DC: U.S. Department of Agriculture, Forest Service. 410 p.

**Hebel, C.L.; Smith, J.E.; Cromack, K. 2009.** Invasive plant species and soil microbial response to wildfire burn severity in the Cascade Range of Oregon. Applied Soil Ecology. 42(2): 150–159. doi:10.1016/j.apsoil.2009.03.004.

**Heinselman, M.L. 1981.** Fire and succession in the conifer forests of northern North America. In: West, D.C.; Shugart, H.H.; Botkin, D.B., eds. Forest succession: concepts and applications. New York: Springer: 374–405. doi:10.1007/978-1-4612-5950-3_23.

**Hemstrom, M.A.; Franklin, J.F. 1982.** Fire and other disturbances of the forests in Mount Rainier National Park. Quaternary Research. 18(01): 32–51. doi:10.1016/0033-5894(82)90020-5.

**Henderson, J.A.; Peter, D.H.; Lesher, R.D.; Shaw, D.C. 1989.** Forested plant associations of the Olympic National Forest. Portland, OR: U.S. Department of Agriculture, Forest Service, Pacific Northwest Region. 502 p.

**Hessburg, P.F.; Agee, J.K. 2003.** An environmental narrative of inland Northwest United States forests, 1800–2000. Forest Ecology and Management. 178(1–2): 23–59. doi:10.1016/s0378-1127(03)00052-5.

**Hessburg, P.F.; Agee, J.K.; Franklin, J.F. 2005.** Dry forests and wildland fires of the inland Northwest USA: Contrasting the landscape ecology of the pre-settlement and modern eras. Forest Ecology and Management. 211(1–2): 117–139. doi:10.1016/j.foreco.2005.02.016.

**Hessburg, P.F.; Churchill, D.J.; Larson, A.J.; Haugo, R.D.; Miller, C.; Spies, T.A.; North, M.P.; Povak, N.A.; Belote, R.T.; Singleton, P.H.; Gaines, W.L.; Keane, R.E.; Aplet, G.H.; Stephens, S.L.; Morgan, P.; Bisson, P.A.; Rieman, B.E.; Salter, R.B.; Reeves, G.H. 2015.** Restoring fire-prone Inland Pacific landscapes: seven core principles. Landscape Ecology. 30(10): 1805–1835. doi:10.1007/s10980-015-0218-0.

**Hessburg, P.F.; Reynolds, K.M.; Salter, R.B.; Richmond, M.B. 2004.** Using a decision support system to estimate departures of present forest landscape patterns from historical reference condition—an example from the inland Northwest region of the United States. In: Perera, A.H.; Buse, L.J.; Weber, M.G., eds. Emulating natural forest landscape disturbances: concepts and applications. New York, NY: Columbia University Press: 158–174. Chapter 13.

**Hessburg, P.F.; Salter, R.B.; James, K.M. 2007.** Re-examining fire severity relations in pre-management era mixed conifer forests: inferences from landscape patterns of forest structure. Landscape Ecology. 22(S1): 5–24. doi:10.1007/s10980-007-9098-2.

**Hessburg, P.F.; Salter, R.B.; Richmond, M.B.; Smith, B.G. 2000a.** Ecological subregions of the interior Columbia Basin, USA. Applied Vegetation Science. 3: 163–180. doi:10.2307/1478995.

**Hessburg, P.F.; Smith, B.G.; Kreiter, S.D.; Miller, C.A.; Salter, R.B.; McNicoll, C.H.; Hann, W.J. 1999a.** Historical and current forest and range landscapes in the interior Columbia River basin and portions of the Klamath and Great Basins. Part 1: Linking vegetation patterns and landscape vulnerability to potential insect and pathogen disturbances. Gen. Tech. Rep. PNW-GTR-458. Portland, OR: U.S. Department of Agriculture, Forest Service, Pacific Northwest Research Station. 357 p. doi:10.2737/pnw-gtr-458.

**Hessburg, P.F.; Smith, B.G.; Salter, R.B. 1999b.** Detecting change in forest spatial patterns from reference conditions. Ecological Applications. 9(4): 1232–1252. doi:10.1890/1051-0761(1999)009[1232:dcifsp]2.0.co;2.

**Hessburg, P.F.; Smith, B.G.; Salter, R.B. 1999c.** Using estimates of natural variation to detect ecologically important change in forest spatial patterns: a case study, Cascade Range, eastern Washington. Res. Pap. PNW-RP-514. Portland, OR: U.S. Department of Agriculture, Forest Service, Pacific Northwest Research Station. 21 p. doi:10.2737/pnw-rp-514.

**Hessburg, P.F.; Smith, B.G.; Salter, R.B.; Ottmar, R.D.; Alvarado, E. 2002b.** Recent changes (1930s–1990s) in spatial patterns of interior northwest forests, USA. Forest Ecology and Management. 136(1–3): 53–83. doi:10.1016/s0378-1127(99)00263-7.

**Hessburg, P.F.; Spies, T.A.; Perry, D.A.; Skinner, C.N.; Taylor, A.H.; Brown, P.M.; Stephens, S.L.; Larson, A.J.; Churchill, D.J.; Povak, N.A.; Singleton, P.H.; McComb, B.; Zielinski, W.J.; Collins, B.M.; Salter, R.B.; Keane, J.J.; Franklin, J.F.; Riegel, G. 2016.** Tamm Review: Management of mixed-severity fire regime forests in Oregon, Washington, and Northern California. Forest Ecology and Management. 366: 221–250. doi:10.1016/j.foreco.2016.01.034.

**Hessl, A.E.; McKenzie, D.; Schellhaas, R. 2004.** Drought and Pacific Decadal Oscillation linked to fire occurrence in the inland Pacific Northwest. Ecological Applications. 14(2): 425–442. doi:10.1890/03-5019.

**Heyerdahl, E.K.; Brubaker, L.B.; Agee, J.K. 2001.** Spatial controls of historical fire regimes: a multiscale example from the interior west, USA. Ecology. 82(3): 660–678. doi:10.1890/0012-9658(2001)082[0660:scohfr]2.0.co;2.

**Heyerdahl, E.K.; McKenzie, D.; Daniels, L.D.; Hessl, A.E.; Littell, J.S.; Mantua, N.J. 2008.** Climate drivers of regionally synchronous fires in the inland Northwest (1651–1900). International Journal of Wildland Fire. 17(1): 40. doi:10.1071/wf07024.

**Hicke, J.A.; Johnson, M.C.; Hayes, J.L.; Preisler, H.K. 2012.** Effects of bark beetle-caused tree mortality on wildfire. Forest Ecology and Management. 271: 81–90. doi:10.1016/j.foreco.2012.02.005.

**Hiers, J.K.; Jackson, S.T.; Hobbs, R.J.; Bernhardt, E.S.; Valentine, L.E. 2016.** The precision problem in conservation and restoration. Trends in Ecology & Evolution. 31(11): 820–830. doi:10.1016/j.tree.2016.08.001.

**Hobbs, R.J.; Higgs, E.; Hall, C.M.; Bridgewater, P.; Chapin, F.S.; Ellis, E.C.; Ewel, J.J.; Hallett, L.M.; Harris, J.; Hulvey, K.B.; Jackson, S.T.; Kennedy, P.L.; Kueffer, C.; Lach, L.; Lantz, T.C.; Lugo, A.E.; Mascaro, J.; Murphy, S.D.; Nelson, C.R.; Perring, M.P.; Richardson, D.M.; Seastedt, T.R.; Standish, R.J.; Starzomski, B.M.; Suding, K.N.; Tognetti, P.M.; Yakob, L.; Yung, L. 2014.** Managing the whole landscape: historical, hybrid, and novel ecosystems. Frontiers in Ecology and the Environment. 12(10): 557–564. doi:10.1890/130300.

**Holden, Z.A.; Morgan, P.; Hudak, A.T. 2010.** Burn severity of areas reburned by wildfires in the Gila National Forest, New Mexico, USA. Fire Ecology. 6(3): 77–85. doi:10.4996/fireecology.0603085.

**Horton, T.R.; Bruns, T.D.; Parker, V.T. 1999.** Ectomycorrhizal fungi associated with *Arctostaphylos* contribute to *Pseudotsuga menziesii* establishment. Canadian Journal of Botany. 77(1): 93–102. doi:10.1139/b98-208.

**Hubbard, S. 2015.** BLM timber harvest plans displease Lane County commissioners. Salem, OR: Statesman Journal. http://www.statesmanjournal.com/story/news/politics/2015/08/18/blm-timber-harvest-plans-displease-lane-county-commissioners/31918119/. (9 October 2017).

**Huff, M.H.; Ottmar, R.D.; Alvarado, E.; Vihnanek, R.E.; Lehmkuhl, J.F.; Hessburg, P.F.; Everett, R.L. 1995.** Historical and current forest landscapes in eastern Oregon and Washington. Part II: Linking vegetation characteristics to potential fire behavior and related smoke production. Gen. Tech. Rep. PNW-GTR-355. Portland, OR: U.S. Department of Agriculture, Forest Service, Pacific Northwest Research Station. 43 p. doi:10.2737/pnw-gtr-355.

**Huff, T.D.; Bailey, J.D. 2009.** Longevity and dynamics of fatally and nonfatally topped Douglas-fir in the Coast Range of Oregon. Canadian Journal of Forest Research. 39(11): 2224–2233. doi:10.1139/x09-141.

**Hunt, J.; Krueger, K.W. 1962.** Decay associated with thinning wounds in young-growth western hemlock and Douglas-fir. Journal of Forestry. 60(5): 336–340.

**Hunter, J.C.; Parker, V.T. 1993.** The disturbance regime of an old-growth forest in coastal California. Journal of Vegetation Science. 4(1): 19–24. doi:10.2307/3235729.

**Hunter, M.L., Jr. 1990.** Wildlife, forests, and forestry: principles of managing forests for biological diversity. Englewood Cliffs, NJ: Prentice Hall. 370 p.

**Hunter, M.L., Jr.; White, A.S. 1997.** Ecological thresholds and the definition of old-growth forest stands. Natural Areas Journal. 17(4): 292–296.

**Hurteau, M.; North, M. 2009.** Fuel treatment effects on tree-based forest carbon storage and emissions under modelled wildfire scenarios. Frontiers in Ecology and the Environment. 7(8): 409–414. https://doi.org/10.1890/080049. (3 April 2018).

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

**Hutto, R.L.; Gallo, S.M. 2006.** The effects of postfire salvage logging on cavity-nesting birds. The Condor. 108(4): 817. doi:10.1650/0010-5422(2006)108[817:teopsl]2.0.co;2.

**Hutto, R.L.; Keane, R.E.; Sherriff, R.L.; Rota, C.T.; Eby, L.A.; Saab, V.A. 2016.** Toward a more ecologically informed view of severe forest fires. Ecosphere. 7(2): e01255. doi:10.1002/ecs2.1255.

**Ice, G.G.; Neary, D.G.; Adams, P.W. 2004.** Effects of wildfire on soils and watershed processes. Journal of Forestry. 102(6): 16–20.

**Impara, P.C. 1997.** Spatial and temporal patterns of fire in the forests of the central Oregon Coast Range. Corvallis, OR: Oregon State University. 354 p. Ph.D. dissertation.

**Jennings, T.N.; Smith, J.E.; Cromack, K.; Sulzman, E.W.; McKay, D.; Caldwell, B.A.; Beldin, S.I. 2011.** Impact of postfire logging on soil bacterial and fungal communities and soil biogeochemistry in a mixed-conifer forest in central Oregon. Plant and Soil. 350(1–2): 393–411. doi:10.1007/s11104-011-0925-5.

**Jimerson, T.M.; Jones, D.W. 2003.** Megram: blowdown, wildfire, and the effects of fuel treatment. In: Galley, K.E.M.; Klinger, R.C.; Sugihara, N.G., eds. Proceedings of fire conference 2000: the first national congress on fire ecology, prevention, and management. Misc. Pub. 13. Tallahassee, FL: Tall Timbers Research Station: 55–59.

**Johnson, D.W.; Murphy, J.F.; Susfalk, R.B.; Caldwell, T.G.; Miller, W.W.; Walker, R.F.; Powers, R.F. 2005.** The effects of wildfire, salvage logging, and post-fire N-fixation on the nutrient budgets of a Sierran forest. Forest Ecology and Management. 220(1–3): 155–165. doi:10.1016/j.foreco.2005.08.011.

**Johnson, K.N.; Bettinger, P.; Kline, J.D.; Spies, T.A.; Lennette, M.; Lettman, G.; Garber-Yonts, B.; Larsen, T. 2007.** Simulating forest structure, timber production, and socioeconomic effects in a multi-owner province. Ecological Applications. 17(1): 34–47. doi:10.1890/1051-0761(2007)017[0034:sfstpa]2.0.co;2.

**Kane, V.R.; Lutz, J.A.; Roberts, S.L.; Smith, D.F.; McGaughey, R.J.; Povak, N.A.; Brooks, M.L. 2013.** Landscape-scale effects of fire severity on mixed-conifer and red fir forest structure in Yosemite National Park. Forest Ecology and Management. 287: 17–31. doi:10.1016/j.foreco.2012.08.044.

**Karr, J.R.; Rhodes, J.J.; Minshall, G.W.; Hauer, F.R.; Beschta, R.L.; Frissell, C.A.; Perry, D.A. 2004.** The effects of postfire salvage logging on aquatic ecosystems in the American West. BioScience. 54(11): 1029. doi:10.1641/0006-3568(2004)054[1029:teopsl]2.0.co;2.

**Keane, R.E.; Hessburg, P.F.; Landres, P.B.; Swanson, F.J. 2009.** The use of historical range and variability (HRV) in landscape management. Forest Ecology and Management. 258(7): 1025–1037. doi:10.1016/j.foreco.2009.05.035.

**Keane, R.E.; Parsons, R.A.; Hessburg, P.F. 2002.** Estimating historical range and variation of landscape patch dynamics: limitations of the simulation approach. Ecological Modelling. 151(1): 29–49. doi:10.1016/s0304-3800(01)00470-7.

**Keeley, J.E. 2000.** Fire and invasive species in Mediterranean-climate ecosystems of California. In: Galley, K.E.; Wilson, T.P., eds. Proceedings of the invasive species workshop: the role of fire in the control and spread of invasive species; Fire conference 2000: first national congress on fire ecology, prevention, and management. Tallahassee, FL: Tall Timbers Research Station: 81–94.

**Kellogg, L.-K.B.; McKenzie, D.; Peterson, D.L.; Hessl, A.E. 2007.** Spatial models for inferring topographic controls on historical low-severity fire in the eastern Cascade Range of Washington, USA. Landscape Ecology. 23(2): 227–240. doi:10.1007/s10980-007-9188-1.

**Kelly, L.T.; Brotons, L. 2017.** Using fire to promote biodiversity. Science. 355(6331): 1264–1265. doi:10.1126/science.aam7672.

**Kennedy, R.S.H.; Wimberly, M.C. 2009.** Historical fire and vegetation dynamics in dry forests of the interior Pacific Northwest, USA, and relationships to northern spotted owl (*Strix occidentalis caurina*) habitat conservation. Forest Ecology and Management. 258(5): 554–566. doi:10.1016/j.foreco.2009.04.019.

**Kimmins, J.P. 2003.** Old-growth forest: an ancient and stable sylvan equilibrium, or a relatively transitory ecosystem condition that offers people a visual and emotional feast? Answer: it depends. The Forestry Chronicle. 79(3): 429–440. doi:10.5558/tfc79429-3.

**Kiser, J.; Morrell, J.J.; Filip, G. 2011.** Fungal colonization of coastal Douglas-fir following mechanical commercial thinning damage. Forest Pathology. 41(4): 317–324. https://doi.org/10.1111/j.1439-0329.2010.00675.x. (3 April 2018).

**Kishchuk, B.E.; Thiffault, E.; Lorente, M.; Quideau, S.; Keddy, T.; Sidders, D. 2015.** Decadal soil and stand response to fire, harvest, and salvage-logging disturbances in the western boreal mixedwood forest of Alberta, Canada. Canadian Journal of Forest Research. 45(2): 141–152. doi:10.1139/cjfr-2014-0148.

**Kitzberger, T.; Aráoz, E.; Gowda, J.H.; Mermoz, M.; Morales, J.M. 2011.** Decreases in fire spread probability with forest age promotes alternative community states, reduced resilience to climate variability and large fire regime shifts. Ecosystems. 15(1): 97–112. doi:10.1007/s10021-011-9494-y.

**Kitzberger, T.; Brown, P.M.; Heyerdahl, E.K.; Swetnam, T.W.; Veblen, T.T. 2006.** Contingent Pacific-Atlantic Ocean influence on multicentury wildfire synchrony over western North America. Proceedings of the National Academy of Sciences. 104(2): 543–548. doi:10.1073/pnas.0606078104.

**Klein, J.; Williams, A.; Menke, J. 2013.** Determining landscape-scale changes in forest structure and possible management responses to *Phytophthora ramorum* in the Mt. Tamalpais watershed, Marin County, California. In: Frankel, S.J.; Kliejunas, J.T.; Palmieri, K.M.; Alexander, J.M., eds. Proceedings of the sudden oak death fifth science symposium. Gen. Tech. Rep. PSW-GTR-243. Albany, CA: U.S. Department of Agriculture, Forest Service, Pacific Southwest Research Station: 93.

**Kline, J.D.; Harmon, M.E.; Spies, T.A.; Morzillo, A.T.; Pabst, R.J.; McComb, B.C.; Schnekenburger, F.; Olsen, K.A.; Csuti, B.; Vogeler, J.C. 2016.** Evaluating carbon storage, timber harvest, and habitat possibilities for a Western Cascades (USA) forest landscape. Ecological Applications. 26(7): 2044–2059. doi:10.1002/eap.1358.

**Klinka, K.; Chen, H.Y.; Wang, Q.; De Montigny, L. 1996.** Forest canopies and their influence on understory vegetation in early-seral stands on west Vancouver Island. Northwest Science. 70(3): 193–200.

**Knapp, E.E.; Keeley, J.E.; Ballenger, E.A.; Brennan, T.J. 2005.** Fuel reduction and coarse woody debris dynamics with early season and late season prescribed fire in a Sierra Nevada mixed conifer forest. Forest Ecology and Management. 208(1–3): 383–397. doi:10.1016/j.foreco.2005.01.016.

**Knapp, E.E.; North, M.; Benech, M.; Estes, B. 2012a.** The variable-density thinning study at Stanislaus-Tuolumne Experimental Forest. In: North, M., ed. Managing Sierra Nevada forests. Gen. Tech. Rep. PSW-GTR-237. Albany, CA: U.S. Department of Agriculture, Forest Service, Pacific Southwest Research Station: 127–139.

**Knapp, E.E.; Weatherspoon, J.; Skinner, C.N. 2012b.** Shrub seed banks in mixed conifer forests of northern California and the role of fire in regulatinabundance. Fire Ecology. 7(1): 32–48. doi:10.4996/fireecology.0801032.

**Knapp, P.A.; Hadley, K.S. 2011.** Lewis and Clarks' Tempest: The 'perfect storm' of November 1805, Oregon, USA. The Holocene. 21(4): 693–697. doi:10.1177/0959683610391319.

**Knapp, P.A.; Hadley, K.S. 2012.** A 300-year history of Pacific Northwest windstorms inferred from tree rings. Global and Planetary Change. 92–93: 257–266. doi:10.1016/j.gloplacha.2012.06.002.

**Küchler, A.W. 1964.** Potential natural vegetation of the conterminous United States: Map. Vol. 36. New York: American Geographical Society.

**Küchler, A.W. 1974.** A new vegetation map of Kansas. Ecology. 55(3): 586–604. doi:10.2307/1935149.

**Kuehne, C.; Puettmann, K.J. 2008.** Natural regeneration in thinned Douglas-fir stands in western Oregon. Journal of Sustainable Forestry. 27(3): 246–274. doi:10.1080/10549810802256221.

**Kuehne, C.; Weiskittel, A.R.; Fraver, S.; Puettmann, K.J. 2015.** Effects of thinning-induced changes in structural heterogeneity on growth, ingrowth, and mortality in secondary coastal Douglas-fir forests. Canadian Journal of Forest Research. 45(11): 1448–1461. doi:10.1139/cjfr-2015-0113.

**Kuljian, H.; Varner, J.M. 2010.** The effects of sudden oak death on foliar moisture content and crown fire potential in tanoak. Forest Ecology and Management. 259(10): 2103–2110. doi:10.1016/j.foreco.2010.02.022.

**Landres, P.B.; Morgan, P.; Swanson, F.J. 1999.** Overview of the use of natural variability concepts in managing ecological systems. Ecological Applications. 9(4): 1179–1188. doi:10.1890/1051-0761(1999)009[1179:ootuon]2.0.co;2.

**Larson, A.J.; Belote, R.T.; Cansler, C.A.; Parks, S.A.; Dietz, M.S. 2013.** Latent resilience in ponderosa pine forest: effects of resumed frequent fire. Ecological Applications. 23(6): 1243–1249. doi:10.1890/13-0066.1.

**Larson, A.J.; Churchill, D. 2012.** Tree spatial patterns in fire-frequent forests of western North America, including mechanisms of pattern formation and implications for designing fuel reduction and restoration treatments. Forest Ecology and Management. 267: 74–92. doi:10.1016/j.foreco.2011.11.038.

**Larson, A.J.; Franklin, J.F. 2005.** Patterns of conifer tree regeneration following an autumn wildfire event in the western Oregon Cascade Range, USA. Forest Ecology and Management. 218(1–3): 25–36. doi:10.1016/j.foreco.2005.07.015.

**Latham, P.; Tappeiner, J. 2002.** Response of old-growth conifers to reduction in stand density in western Oregon forests. Tree Physiology. 22(2–3): 137–146. doi:10.1093/treephys/22.2-3.137.

**Lauvaux, C.A.; Skinner, C.N.; Taylor, A.H. 2016.** High severity fire and mixed conifer forest-chaparral dynamics in the southern Cascade Range, USA. Forest Ecology and Management. 363: 74–85. doi:10.1016/j.foreco.2015.12.016.

**Law, B.E.; Waring, R.H. 2015.** Carbon implications of current and future effects of drought, fire and management on Pacific Northwest forests. Forest Ecology and Management. 355: 4–14. doi:10.1016/j.foreco.2014.11.023.

**Lehmkuhl, J.F.; Hessburg, P.F.; Everett, R.L.; Huff, M.H.; Ottmar, R.D. 1994.** Historical and current forest landscapes of eastern Oregon and Washington Part I: vegetation pattern and insect and disease hazards. Gen. Tech. Rep. PNW-GTR-328. Portland, OR: U.S. Department of Agriculture, Forest Service, Pacific Northwest Research Station. 88 p. doi:10.2737/pnw-gtr-328.

**Lehmkuhl, J.F.; Kennedy, M.; Ford, E.D.; Singleton, P.H.; Gaines, W.L.; Lind, R.L. 2007.** Seeing the forest for the fuel: integrating ecological values and fuels management. Forest Ecology and Management. 246(1): 73–80. doi:10.1016/j.foreco.2007.03.071.

**Leiberg, J.B. 1903.** Southern part of Cascade Range Forest Reserve. In: Forest conditions in the Cascade Range Forest Reserve, Oregon. Professional Paper 9. Reston, VA: U.S. Department of the Interior, Geological Survey: 229–289.

**Levine, C.R.; Cogbill, C.V.; Collins, B.M.; Larson, A.J.; Lutz, J.A.; North, M.P.; Restaino, C.M.; Safford, H.D.; Stephens, S.L.; Battles, J.J. 2017.** Evaluating a new method for reconstructing forest conditions from General Land Office survey records. Ecological Applications. 27(5): 1498–1513. doi:10.1002/eap.1543.

**Lewis, J.C. 1998.** Creating snags and wildlife trees in commercial forest landscapes. Western Journal of Applied Forestry. 13: 97–101.

**Lillybridge, T.R.; Kovalchik, B.L.; Williams, C.K.; Smith, B.G. 1995.** Field guide for forested plant associations of the Wenatchee National Forest. Gen. Tech. Rep. PNW-GTR-359. Portland, OR: U.S. Department of Agriculture, Forest Service, Pacific Northwest Research Station. 335 p. doi:10.2737/pnw-gtr-359.

**Lindenmayer, D.B. 2004.** Ecology: enhanced: salvage harvesting policies after natural disturbance. Science. 303(5662): 1303–1303. doi:10.1126/science.1093438.

**Lindenmayer, D.B.; Franklin, J.F. 1997.** Managing stand structure as part of ecologically sustainable forest management in Australian mountain ash forests. Conservation Biology. 11(5): 1053–1068. doi:10.1046/j.1523-1739.1997.96150.x.

**Lindenmayer, D.B.; Franklin, J.F. 2002.** Conserving forest biodiversity: a comprehensive multiscaled approach. Washington, DC: Island Press. 352 p.

**Lindenmayer, D.B.; Noss, R.F. 2006.** Salvage logging, ecosystem processes, and biodiversity conservation. Conservation Biology. 20(4): 949–958. doi:10.1111/j.1523-1739.2006.00497.x.

**Lindh, B.C.; Muir, P.S. 2004.** Understory vegetation in young Douglas-fir forests: Does thinning help restore old-growth composition? Forest Ecology and Management. 192(2–3): 285–296. doi:10.1016/j.foreco.2004.01.018.

**Lint, J. 2005.** Northwest Forest Plan—the first 10 years (1994–2003): status and trends of northern spotted owl populations and habitat. Gen. Tech. Rep. PNW-GTR-648. Portland, OR: U.S. Department of Agriculture, Forest Service, Pacific Northwest Research Station. 176 p. doi:10.2737/pnw-gtr-648.

**Littell, J.S.; McKenzie, D.; Peterson, D.L.; Westerling, A.L. 2009.** Climate and wildfire area burned in western U.S. ecoprovinces, 1916–2003. Ecological Applications. 19(4): 1003–1021. doi:10.1890/07-1183.1.

**Littell, J.S.; Oneil, E.E.; McKenzie, D.; Hicke, J.A.; Lutz, J.A.; Norheim, R.A.; Elsner, M.M. 2010.** Forest ecosystems, disturbance, and climatic change in Washington State, USA. Climatic Change. 102(1–2): 129–158. doi:10.1007/s10584-010-9858-x.

**Lockman, I.B.; Kearns, H.J.S. 2016.** Forest root diseases across the United States. Gen. Tech. Rep. RMRS-GTR-342. Ogden, UT: U.S. Department of Agriculture, Forest Service, Rocky Mountain Research Station. 55 p.

**Long, C.J.; Whitlock, C.; Bartlein, P.J.; Millspaugh, S.H. 1998.** A 9000-year fire history from the Oregon Coast Range, based on a high-resolution charcoal study. Canadian Journal of Forest Research. 28(5): 774–787. doi:10.1139/x98-051.

**Long, J.W.; Anderson, M.K.; Quinn-Davidson, L.N.; Goode, R.W.; Lake, F.K.; Skinner, C.N. 2016.** Restoring California black oak ecosystems to promote tribal values and wildlife. Gen. Tech. Rep. PSW-GTR-252. Albany, CA: U.S. Department of Agriculture, Forest Service, Pacific Southwest Research Station. 111 p.

**Long, J.W.; Quinn-Davidson, L.; Skinner, C.N. 2014.** Science synthesis to support socioecological resilience in the Sierra Nevada and southern Cascade Range. Gen. Tech. Rep. PSW-GTR-247. Albany, CA: U.S. Department of Agriculture, Forest Service, Pacific Southwest Research Station. 723 p. doi:10.2737/psw-gtr-247.

**Lord, R. 2005.** Leave tree effect growth analysis. Portland, OR: Mason, Bruce, and Girard.

**Lorimer, C.G.; Porter, D.J.; Madej, M.A.; Stuart, J.D.; Veirs, S.D.; Norman, S.P.; O'Hara, K.L.; Libby, W.J. 2009.** Presettlement and modern disturbance regimes in coast redwood forests: implications for the conservation of old-growth stands. Forest Ecology and Management. 258(7): 1038–1054. doi:10.1016/j.foreco.2009.07.008.

**Loudermilk, E.L.; Scheller, R.M.; Weisberg, P.J.; Kretchun, A. 2016.** Bending the carbon curve: fire management for carbon resilience under climate change. Landscape Ecology. 32(7): 1461–1472. doi:10.1007/s10980-016-0447-x.

**Loudermilk, E.L.; Scheller, R.M.; Weisberg, P.J.; Yang, J.; Dilts, T.E.; Karam, S.L.; Skinner, C. 2013.** Carbon dynamics in the future forest: the importance of long-term successional legacy and climate-fire interactions. Global Change Biology. 19: 3502–3515. doi:10.1111/gcb.12310.

**Loudermilk, E.L.; Stanton, A.; Scheller, R.M.; Dilts, T.E.; Weisberg, P.J.; Skinner, C.; Yang, J. 2014.** Effectiveness of fuel treatments for mitigating wildfire risk and sequestering forest carbon: a case study in the Lake Tahoe Basin. Forest Ecology and Management. 323: 114–125. doi:10.1016/j.foreco.2014.03.011.

**Lovett, G.M.; Weiss, M.; Liebhold, A.M.; Holmes, T.P.; Leung, B.; Lambert, K.F.; Orwig, D.A.; Campbell, F.T.; Rosenthal, J.; McCullough, D.G.; Wildova, R.; Ayres, M.P.; Canham, C.D.; Foster, D.R.; LaDeau, S.L.; Weldy, T. 2016.** Nonnative forest insects and pathogens in the United States: impacts and policy options. Ecological Applications. 26(5): 1437–1455. doi:10.1890/15-1176.

**Lutz, J.A.; Halpern, C.B. 2006.** Tree mortality during early forest development: a long-term study of rates, causes, and consequences. Ecological Monographs. 76(2): 257–275. doi:10.1890/0012-9615(2006)076[0257:tmdefd]2.0.co;2.

**Lutz, J.A.; van Wagtendonk, J.W.; Thode, A.E.; Miller, J.D.; Franklin, J.F. 2009.** Climate, lightning ignitions, and fire severity in Yosemite National Park, California, USA. International Journal of Wildland Fire. 18(7): 765. doi:10.1071/wf08117.

**Lydersen, J.; North, M. 2012.** Topographic variation in structure of mixed-conifer forests under an active-fire regime. Ecosystems. 15(7): 1134–1146. doi:10.1007/s10021-012-9573-8.

**Lydersen, J.M.; North, M.P.; Knapp, E.E.; Collins, B.M. 2013.** Quantifying spatial patterns of tree groups and gaps in mixed-conifer forests: reference conditions and long-term changes following fire suppression and logging. Forest Ecology and Management. 304: 370–382. doi:10.1016/j.foreco.2013.05.023.

**Lynott, R.E.; Cramer, O.P. 1966.** Detailed analysis of the 1962 Columbus Day windstorm in Oregon and Washington. Monthly Weather Review. 94(2): 105–117.

**Ma, S.; Concilio, A.; Oakley, B.; North, M.; Chen, J. 2010.** Spatial variability in microclimate in a mixed-conifer forest before and after thinning and burning treatments. Forest Ecology and Management. 259(5): 904–915. doi:10.1016/j.foreco.2009.11.030.

**Maguire, D.A.; Kanaskie, A.; Voelker, W.; Johnson, R.; Johnson, G. 2002.** Growth of young Douglas-fir plantations across a gradient in Swiss needle cast severity. Western Journal of Applied Forestry. 17: 86–95.

**Maguire, D.A.; Mainwaring, D.B.; Kanaskie, A. 2011.** Ten-year growth and mortality in young Douglas-fir stands experiencing a range in Swiss needle cast severity. Canadian Journal of Forest Research. 41(10): 2064–2076. doi:10.1139/x11-114.

**Mallek, C.; Safford, H.; Viers, J.; Miller, J. 2013.** Modern departures in fire severity and area vary by forest type, Sierra Nevada and southern Cascades, California, USA. Ecosphere. 4(12): art153. doi:10.1890/es13-00217.1.

**Marañón-Jiménez, S.; Castro, J.; Querejeta, J.I.; Fernández-Ondoño, E.; Allen, C.D. 2013.** Post-fire wood management alters water stress, growth, and performance of pine regeneration in a Mediterranean ecosystem. Forest Ecology and Management. 308: 231–239. doi:10.1016/j.foreco.2013.07.009.

**Margules, C.R.; Pressey, R.L. 2000.** Systematic conservation planning. Nature. 405(6783): 243–253. doi:10.1038/35012251.

**Marshall, D.; Curtis, R.O. 2002.** Levels-of-growing-stock cooperative study in Douglas-fir: Report No. 15—Hoskins: 1963–1998. Res. Pap. PNW-RP-537. Portland, OR: U.S. Department of Agriculture, Forest Service, Pacific Northwest Research Station. 80 p.

**Martin, P.H.; Canham, C.D.; Marks, P.L. 2009.** Why forests appear resistant to exotic plant invasions: intentional introductions, stand dynamics, and the role of shade tolerance. Frontiers in Ecology and the Environment. 7(3): 142–149. doi:10.1890/070096.

**Mathiasen, R.; Marshall, K. 1999.** Dwarf mistletoe diversity in the Siskiyou-Klamath Mountain Region. Natural Areas Journal. 19(4): 379–385.

**McIver, J.D.; Ottmar, R. 2007.** Fuel mass and stand structure after post-fire logging of a severely burned ponderosa pine forest in northeastern Oregon. Forest Ecology and Management. 238(1–3): 268–279. doi:10.1016/j.foreco.2006.10.021.

**McIver, J.D.; Starr, L. 2001.** A literature review on the environmental effects of post-fire logging. Western Journal of Applied Forestry. 16: 159–168.

**McIver, J.D.; Stephens, S.L.; Agee, J.K.; Barbour, J.; Boerner, R.E.J.; Edminster, C.B.; Erickson, K.L.; Farris, K.L.; Fettig, C.J.; Fiedler, C.E.; Haase, S.; Hart, S.C.; Keeley, J.E.; Knapp, E.E.; Lehmkuhl, J.F.; Moghaddas, J.J.; Otrosina, W.; Outcalt, K.W.; Schwilk, D.W.; Skinner, C.N.; Waldrop, T.A.; Weatherspoon, C.P.; Yaussy, D.A.; Youngblood, A.; Zack, S. 2013.** Ecological effects of alternative fuel-reduction treatments: highlights of the National Fire and Fire Surrogate study (FFS). International Journal of Wildland Fire. 22(1): 63. doi:10.1071/wf11130.

**McKenzie, D.; Gedalof, Z.e.; Peterson, D.L.; Mote, P. 2004.** Climatic change, wildfire, and conservation. Conservation Biology. 18(4): 890–902. doi:10.1111/j.1523-1739.2004.00492.x.

**McNeil, R.C.; Zobel, D.B. 1980.** Vegetation and fire history of a ponderosa pine-white fir forest in Crater Lake National Park. Northwest Science. 54(1): 30–46.

**Meentemeyer, R.K.; Rank, N.E.; Anacker, B.L.; Rizzo, D.M.; Cushman, J.H. 2008.** Influence of land-cover change on the spread of an invasive forest pathogen. Ecological Applications. 18(1): 159–171. doi:10.1890/07-0232.1.

**Meigs, G.W.; Campbell, J.L.; Zald, H.S.J.; Bailey, J.D.; Shaw, D.C.; Kennedy, R.E. 2015.** Does wildfire likelihood increase following insect outbreaks in conifer forests? Ecosphere. 6(7): art118. doi:10.1890/es15-00037.1.

**Meigs, G.W.; Donato, D.C.; Campbell, J.L.; Martin, J.G.; Law, B.E. 2009.** Forest fire impacts on carbon uptake, storage, and emission: the role of burn severity in the eastern Cascades, Oregon. Ecosystems. 12(8): 1246–1267. doi:10.1007/s10021-009-9285-x.

**Merschel, A.G.; Spies, T.A.; Heyerdahl, E.K. 2014.** Mixed-conifer forests of central Oregon: effects of logging and fire exclusion vary with environment. Ecological Applications. 24(7): 1670–1688. doi:10.1890/13-1585.1.

**Messier, M.S.; Shatford, J.P.A.; Hibbs, D.E. 2012.** Fire exclusion effects on riparian forest dynamics in southwestern Oregon. Forest Ecology and Management. 264: 60–71. doi:10.1016/j.foreco.2011.10.003.

**Metz, M.R.; Frangioso, K.M.; Meentemeyer, R.K.; Rizzo, D.M. 2011.** Interacting disturbances: wildfire severity affected by stage of forest disease invasion. Ecological Applications. 21(2): 313–320. doi:10.1890/10-0419.1.

**Millar, C.I. 2014.** Historic variability: informing restoration strategies, not prescribing targets. Journal of Sustainable Forestry. 33(Suppl.): S28–S42. doi:10.1080/10549811.2014.887474.

**Millar, C.I.; Stephenson, N.L.; Stephens, S.L. 2007.** Climate change and forests of the future: managing in the face of uncertainty. Ecological Applications. 17(8): 2145–2151. doi:10.1890/06-1715.1.

**Miller, C.; Urban, D.L. 1999a.** Forest pattern, fire, and climatic change in the Sierra Nevada. Ecosystems. 2(1): 76–87. doi:10.1007/s100219900060.

**Miller, C.; Urban, D.L. 1999b.** Interactions between forest heterogeneity and surface fire regimes in the southern Sierra Nevada. Canadian Journal of Forest Research. 29(2): 202–212. doi:10.1139/x98-188.

**Miller, C.; Urban, D.L. 2000a.** Connectivity of forest fuels and surface fire regimes. Landscape Ecology. 15(2): 145–154. doi:10.1023/a:1008181313360.

**Miller, C.; Urban, D.L. 2002b.** Modeling the effects of fire management alternatives on Sierra Nevada mixed-conifer forests. Ecological Applications. 10(1): 85–94. doi:10.1890/1051-0761(2000)010[0085:mteofm]2.0.co;2.

**Miller, J.D.; Safford, H. 2012.** Trends in wildfire severity: 1984 to 2010 in the Sierra Nevada, Modoc Plateau, and southern Cascades, California, USA. Fire Ecology. 8(3): 41–57. doi:10.4996/fireecology.0803041.

**Miller, J.D.; Safford, H.D.; Crimmins, M.; Thode, A.E. 2008.** Quantitative evidence for increasing forest fire severity in the Sierra Nevada and Southern Cascade Mountains, California and Nevada, USA. Ecosystems. 12(1): 16–32. doi:10.1007/s10021-008-9201-9.

**Miller, J.D.; Skinner, C.N.; Safford, H.D.; Knapp, E.E.; Ramirez, C.M. 2012.** Trends and causes of severity, size, and number of fires in northwestern California, USA. Ecological Applications. 22(1): 184–203. doi:10.1890/10-2108.1.

**Mohr, J.A.; Whitlock, C.; Skinner, C.N. 2000.** Postglacial vegetation and fire history, eastern Klamath Mountains, California, USA. The Holocene. 10(5): 587–601. doi:10.1191/095968300675837671.

**Monsanto, P.G.; Agee, J.K. 2008.** Long-term post-wildfire dynamics of coarse woody debris after salvage logging and implications for soil heating in dry forests of the eastern Cascades, Washington. Forest Ecology and Management. 255(12): 3952–3961. doi:10.1016/j.foreco.2008.03.048.

**Moore, J.M.; Woods, A.E.; Moran, B. 2017.** 2016 aerial survey results: California. Report R5-PR-034. Davis, CA: U.S. Department of Agriculture, Forest Service, Pacific Southwest Region, Forest Health Monitoring Program. 13 p.

**Moore, K.D. 2007.** In the shadow of the cedars: the spiritual values of old-growth forests. Conservation Biology. 21(4): 1120–1123. doi:10.1111/j.1523-1739.2007.00669.x.

**Morgan, P.; Aplet, G.H.; Haufler, J.B.; Humphries, H.C.; Moore, M.M.; Wilson, W.D. 1994.** Historical range of variability. Journal of Sustainable Forestry. 2(1–2): 87–111. doi:10.1300/j091v02n01_04.

**Morgan, P.; Heyerdahl, E.K.; Gibson, C.E. 2008.** Multi-season climate synchronized forest fires throughout the 20th century, northern Rockies, USA. Ecology. 89(3): 717–728. doi:10.1890/06-2049.1.

**Moritz, M.A. 2003.** Spatiotemporal analysis of controls on shrubland fire regimes: age dependency and fire hazard. Ecology. 84(2): 351–361. doi:10.1890/0012-9658(2003)084[0351:saocos]2.0.co;2.

**Moritz, M.A.; Hessburg, P.F.; Povak, N.A. 2011.** Native fire regimes and landscape resilience. In: McKenzie, D.; Miller, C.; Falk, D.A., eds. The landscape ecology of fire. New York: Springer: 51–86. Chapter 3.

**Moritz, M.A.; Hurteau, M.D.; Suding, K.N.; D'Antonio, C.M. 2013.** Bounded ranges of variation as a framework for future conservation and fire management. Annals of the New York Academy of Sciences. 1286(1): 92–107. doi:10.1111/nyas.12104.

**Moritz, M.A.; Morais, M.E.; Summerell, L.A.; Carlson, J.M.; Doyle, J. 2005.** Wildfires, complexity, and highly optimized tolerance. Proceedings of the National Academy of Sciences of the United States of America. 102(50): 17912–17917. doi:10.1073/pnas.0508985102.

**Moritz, M.A.; Odion, D.C. 2005.** Examining the strength and possible causes of the relationship between fire history and sudden oak death. Oecologia. 144(1): 106–114. doi:10.1007/s00442-005-0028-1.

**Morrison, P.H.; Swanson, F.J. 1990.** Fire history and pattern in a Cascade Range landscape. Gen. Tech. Rep. PNW-GTR-254. Portland, OR: U.S. Department of Agriculture, Forest Service, Pacific Northwest Research Station. 73 p. doi:10.2737/pnw-gtr-254.

**Morrow, R.J. 1985.** Age structure and spatial pattern of old-growth ponderosa pine in Pringle Falls Experimental Forest, central Oregon. Corvallis, OR: Oregon State University. 80 p. M.S. thesis.

**Moser, W.K.; Barnard, E.L.; Billings, R.F.; Crocker, S.J.; Dix, M.E.; Gray, A.N.; Ice, G.G.; Kim, M.S.; Reid, R.; Rodman, S.U.; McWilliams, W.H. 2009.** Impacts of nonnative invasive species on US forests and recommendations for policy and management. Journal of Forestry. 107(6): 320–327.

**Muir, J.A.; Hennon, P.E. 2007.** A synthesis of the literature on the biology, ecology, and management of western hemlock dwarf mistletoe. Gen. Tech. Rep. PNW-GTR-718. Portland, OR: U.S. Department of Agriculture, Forest Service, Pacific Northwest Research Station. 142 p. doi:10.2737/pnw-gtr-718.

**Muir, P.S.; Mattingly, R.L.; Tappeiner, J.C., II; Bailey, J.C.; Elliot, W.E.; Hagar, J.C.; Miller, J.C.; Peterson, E.B.; Starkey, E.E. 2002.** Managing for biodiversity in young Douglas-fir forests of western Oregon. Biological Science Report SGS/BRD/BSR-2002-0006. Corvallis, OR: U.S. Department of the Interior, Geological Survey, Biological Resources Division. 76 p.

**Munger, T.T. 1917.** Western yellow pine in Oregon. Bulletin 418. Washington, DC: U.S. Department of Agriculture. doi:10.5962/bhl.title.108047.

**Nagel, T.A.; Taylor, A.H. 2005.** Fire and persistence of montane chaparral in mixed conifer forest landscapes in the northern Sierra Nevada, Lake Tahoe Basin, California, USA. The Journal of the Torrey Botanical Society. 132(3): 442–457. doi:10.3159/1095-5674(2005)132[442:fapomc]2.0.co;2.

**Navarro, S.; Norlander, D. 2016.** 2016 Swiss needle cast aerial survey. In: Ritokova, G.; Shaw, C., eds. Swiss Needle Cast Cooperative 2016 annual report. Corvallis, OR: Oregon State University: 7–11. http://sncc.forestry.oregonstate.edu/annual-reports. (20 September 2017).

**Neill, A.R.; Puettmann, K.J. 2013.** Managing for adaptive capacity: thinning improves food availability for wildlife and insect pollinators under climate change conditions. Canadian Journal of Forest Research. 43(5): 428–440. doi:10.1139/cjfr-2012-0345.

**Niwa, C.G.; Peck, R.W. 2002.** Influence of prescribed fire on carabid beetle (Carabidae) and spider (Araneae) assemblages in forest litter in southwestern Oregon. Environmental Entomology. 31(5): 785–796. doi:10.1603/0046-225x-31.5.785.

**Nonaka, E. 2003.** Disturbance and landscape history as a reference for evaluating forest management effects at a regional scale: examples from the Coast Range of Oregon, USA. Corvallis, OR: Oregon State University. 149 p. Ph.D. dissertation.

**Nonaka, E.; Spies, T.A. 2005.** Historical range of variability in landscape structure: a simulation study in Oregon, USA. Ecological Applications. 15(5): 1727–1746. doi:10.1890/04-0902.

**Nonaka, E.; Spies, T.A.; Wimberly, M.C.; Ohmann, J.L. 2007.** Historical range of variability in live and dead wood biomass: a regional-scale simulation study. Canadian Journal of Forest Research. 37(11): 2349–2364. doi:10.1139/x07-064.

**Norman, S.P.; Taylor, A.H. 2005.** Pine forest expansion along a forest-meadow ecotone in northeastern California, USA. Forest Ecology and Management. 215(1–3): 51–68. doi:10.1016/j.foreco.2005.05.003.

**North, M.; Collins, B.M.; Stephens, S. 2012.** Using fire to increase the scale, benefits, and future maintenance of fuels treatments. Journal of Forestry. 110(7): 392–401. doi:10.5849/jof.12-021.

**North, M.; Sherlock, J. 2012.** Marking and assessing forest heterogeneity. In: North, M., ed. Managing Sierra Nevada forests. Gen. Tech. Rep. PSW-GTR-237. Albany, CA: U.S. Department of Agriculture, Forest Service, Pacific Southwest Research Station: 95–105.

**North, M.; Stine, P.; O'Hara, K.; Zielinski, W.; Stephens, S. 2009.** An ecosystem management strategy for Sierran mixed-conifer forests. Gen. Tech. Rep. PSW-GTR-220. Albany, CA: U.S. Department of Agriculture, Forest Service, Pacific Southwest Research Station. 49 p. doi:10.2737/psw-gtr-220.

**North, M.P.; Stephens, S.L.; Collins, B.M.; Agee, J.K.; Aplet, G.; Franklin, J.F.; Fule, P.Z. 2015.** Reform forest fire management. Science. 349(6254): 1280–1281. doi:10.1126/science.aab2356.

**Noss, R.F. 1990.** Indicators for monitoring biodiversity: a hierarchical approach. Conservation Biology. 4(4): 355–364.

**O'Hara, K.L.; Latham, P.A.; Hessburg, P.; Smith, B.G. 1996.** A structural classification for inland Northwest forest vegetation. Western Journal of Applied Forestry. 11: 97–102.

**Odion, D.C.; Frost, E.J.; Strittholt, J.R.; Jiang, H.; Dellasala, D.A.; Moritz, M.A. 2004.** Patterns of fire severity and forest conditions in the western Klamath Mountains, California. Conservation Biology. 18(4): 927–936. doi:10.1111/j.1523-1739.2004.00493.x.

**Odion, D.C.; Hanson, C.T.; Arsenault, A.; Baker, W.L.; DellaSala, D.A.; Hutto, R.L.; Klenner, W.; Moritz, M.A.; Sherriff, R.L.; Veblen, T.T.; Williams, M.A. 2014.** Examining historical and current mixed-severity fire regimes in ponderosa pine and mixed-conifer forests of western North America. PLoS ONE. 9(2): e87852. doi:10.1371/journal.pone.0087852.

**Odion, D.C.; Hanson, C.T.; Baker, W.L.; DellaSala, D.A.; Williams, M.A. 2016.** Areas of agreement and disagreement regarding ponderosa pine and mixed conifer forest fire regimes: a dialogue with Stevens et al. PLoS ONE. 11(5): e0154579. doi:10.1371/journal.pone.0154579.

**Odion, D.C.; Moritz, M.A.; DellaSala, D.A. 2010.** Alternative community states maintained by fire in the Klamath Mountains, USA. Journal of Ecology. 98(1): 96–105. doi:10.1111/j.1365-2745.2009.01597.x.

**Oliver, C.D. 2009.** Maintaining forest landscapes and sustaining old growth. In: Spies, T.A.; Duncan, S.L., eds. Old growth in a new world: a Pacific Northwest icon reexamined. Washington, DC: Island Press: 274–285.

**Oliver, C.D.; Larson, B.C. 1990.** Forest stand dynamics. New York: McGraw-Hill, Inc. 624 p.

**Pabst, R.J.; Goslin, M.N.; Garman, S.L.; Spies, T.A. 2008.** Calibrating and testing a gap model for simulating forest management in the Oregon Coast Range. Forest Ecology and Management. 256(5): 958–972. doi:10.1016/j.foreco.2008.05.046.

**Palmer, M.A., Bernhardt, E.S.; Allan, J.D.; Lake, P.S.; Alexander, G.; Brooks, S.; Carr, J.; Clayton, S.; Dahm, C.N.; Follstad Shah, J.; Galat, D.L.; Loss, S.G.; Goodwin, P.; Hart, D.D.; Hassett, B.; Jenkinson, R.; Kondof, G.M.; Lave, R.; Meyer, J.T.; O'Donnell, T.K.; Pagano, L.; Sudduth, E. 2005.** Standards for ecologically successful river restoration. Journal of Applied Ecology. 42(2): 208–217. http://www.jstor.org/stable/3505713.

**Parendes, L.A.; Jones, J.A. 2000.** Role of light availability and dispersal in exotic plant invasion along roads and streams in the H.J. Andrews Experimental Forest, Oregon. Conservation Biology. 14(1): 64–75. doi:10.1046/j.1523-1739.2000.99089.x.

**Parks, S.A.; Miller, C.; Holsinger, L.M.; Baggett, L.S.; Bird, B.J. 2016.** Wildland fire limits subsequent fire occurrence. International Journal of Wildland Fire. 25(2): 182. doi:10.1071/wf15107.

**Parks, S.A.; Miller, C.; Nelson, C.R.; Holden, Z.A. 2013.** Previous fires moderate burn severity of subsequent wildland fires in two large western US wilderness areas. Ecosystems. 17(1): 29–42. doi:10.1007/s10021-013-9704-x.

**Parks, S.A.; Miller, C.; Parisien, M.-A.; Holsinger, L.M.; Dobrowski, S.Z.; Abatzoglou, J. 2015.** Wildland fire deficit and surplus in the western United States, 1984–2012. Ecosphere. 6(12): art275. doi:10.1890/es15-00294.1.

**Parsons, D.J. 1978.** Fire and fuel accumulation in a giant sequoia forest. Journal of Forestry. 76: 104–105.

**Parsons, D.J.; DeBenedetti, S.H. 1979.** Impact of fire suppression on a mixed-conifer forest. Forest Ecology and Management. 2: 21–33. doi:10.1016/0378-1127(79)90034-3.

**Perchemlides, K.A.; Muir, P.S.; Hosten, P.E. 2008.** Responses of chaparral and oak woodland plant communities to fuel-reduction thinning in southwestern Oregon. Rangeland Ecology and Management. 61(1): 98–109. doi:10.2111/07-026r1.1.

**Perry, D.A.; Hessburg, P.F.; Skinner, C.N.; Spies, T.A.; Stephens, S.L.; Taylor, A.H.; Franklin, J.F.; McComb, B.; Riegel, G. 2011.** The ecology of mixed severity fire regimes in Washington, Oregon, and Northern California. Forest Ecology and Management. 262(5): 703–717. doi:10.1016/j.foreco.2011.05.004.

**Perry, D.A.; Jing, H.; Youngblood, A.; Oetter, D.R. 2004.** Forest structure and fire susceptibility in volcanic landscapes of the Eastern High Cascades, Oregon. Conservation Biology. 18(4): 913–926. doi:10.1111/j.1523-1739.2004.00530.x.

**Peterson, D.L.; Agee, J.K.; Aplet, G.H.; Dykstra, D.P.; Graham, R.T.; Lehmkuhl, J.F.; Pilliod, D.S.; Potts, D.F.; Powers, R.F.; Stuart, J.D. 2009.** Effects of timber harvest following wildfire in western North America. Gen. Tech. Rep. PNW-GTR-776. Portland, OR: U.S. Department of Agriculture, Forest Service, Pacific Northwest Research Station. 51 p. doi:10.2737/pnw-gtr-776.

**Peterson, D.W.; Dodson, E.K. 2016.** Post-fire logging produces minimal persistent impacts on understory vegetation in northeastern Oregon, USA. Forest Ecology and Management. 370: 56–64. doi:10.1016/j.foreco.2016.04.004.

**Peterson, D.W.; Dodson, E.K.; Harrod, R.J. 2015.** Post-fire logging reduces surface woody fuels up to four decades following wildfire. Forest Ecology and Management. 338: 84–91. doi:10.1016/j.foreco.2014.11.016.

**Pfister, R.D.; Arno, S.F. 1980.** Classifying forest habitat types based on potential climax vegetation. Forest Science. 26(1): 52–70.

**Picchio, R.; Magagnotti, N.; Sirna, A.; Spinelli, R. 2012.** Improved winching technique to reduce logging damage. Ecological Engineering. 47: 83–86. doi:10.1016/j.ecoleng.2012.06.037.

**Pickett, S.T.A.; Thompson, J.N. 1978.** Patch dynamics and the design of nature reserves. Biological Conservation. 13(1): 27–37. doi:10.1016/0006-3207(78)90016-2.

**Pilz, D.; Molina, R.; Mayo, J. 2006.** Effects of thinning young forests on chanterelle mushroom production. Journal of Forestry. 104(1): 9–14.

**Plummer, G.H.; Plummer, F.G.; Rankine, J.H. 1902.** Map of Washington showing classification of lands. Washington, DC: U.S. Department of the Interior, Geological Survey.

**Poage, N.J.; Anderson, P.D. 2007.** Large-scale silviculture experiments of western Oregon and Washington. Gen. Tech. Rep. PNW-GTR-713. Portland, OR: U.S. Department of Agriculture, Forest Service, Pacific Northwest Research Station. 44 p. doi:10.2737/pnw-gtr-713.

**Poff, R.J. 1989.** Compatibility of timber salvage operations with watershed values. In: Berg, N.H., ed. Proceedings of the symposium on fire and watershed management. Gen. Tech. Rep. PSW-GTR-109. Albany, CA: U.S. Department of Agriculture, Forest Service, Pacific Southwest Research Station: 137–140.

**Poirier, V.; Paré, D.; Boiffin, J.; Munson, A.D. 2014.** Combined influence of fire and salvage logging on carbon and nitrogen storage in boreal forest soil profiles. Forest Ecology and Management. 326: 133–141. doi:10.1016/j.foreco.2014.04.021.

**Pollock, M.M.; Beechie, T.J. 2014.** Does riparian forest restoration thinning enhance biodiversity? The ecological importance of large wood. Journal of the American Water Resources Association. 50(3): 543–559. doi:10.1111/jawr.12206.

**Powell, D.C. 2012.** A stage is a stage is a stage…or is it? Successional stages, structural stages, seral stages. White paper F14-SO-WP-Silv-10. Pendleton, OR: U.S. Department of Agriculture, Forest Service, Pacific Northwest Region, Umatilla National Forest. 14 p.

**Powers, E.M.; Marshall, J.D.; Zhang, J.; Wei, L. 2013.** Post-fire management regimes affect carbon sequestration and storage in a Sierra Nevada mixed conifer forest. Forest Ecology and Management. 291: 268–277. doi:10.1016/j.foreco.2012.07.038.

**Pressey, R.L.; Cabeza, M.; Watts, M.E.; Cowling, R.M.; Wilson, K.A. 2007.** Conservation planning in a changing world. Trends in Ecology & Evolution. 22(11): 583–592. doi:10.1016/j.tree.2007.10.001.

**PRISM. 2015.** PRISM Climate Group. http://prism.oregonstate.edu. (12 February 2015).

**Puettmann, K.; Ares, A.; Burton, J.; Dodson, E. 2016.** Forest restoration using variable density thinning: lessons from Douglas-fir stands in western Oregon. Forests. 7(12): 310. doi:10.3390/f7120310.

**Puettmann, K.J.; Dodson, E.K.; Ares, A.; Berger, C. 2013.** Over-and understory responses to thinning treatments: can we accelerate late successional stand structures? In: Anderson, P.D.; Ronneberg, K.L., eds. Density management in the 21st century: west side story. Gen. Tech. Rep. PNW-GTR-880. Portland, OR: U.S. Department of Agriculture, Forest Service, Pacific Northwest Research Station: 44–58.

**Ramage, B.S.; O'Hara, K.L.; Caldwell, B.T. 2010.** The role of fire in the competitive dynamics of coast redwood forests. Ecosphere. 1(6): art20. doi:10.1890/es10-00134.1.

**Ramsey, A.; Omdal, D.; Dozic, A.; Kohler, G.; Hoff, J.; Lynch, D. 2016.** Coastal Washington Swiss needle cast aerial and ground survey, 2016. In: Ritokova, G.; Shaw, D.C., eds. Swiss Needle Cast Cooperative 2016 annual report. Corvallis, OR: Oregon State University: 12–16. http://sncc.forestry.oregonstate.edu/annual-reports. (20 September 2017).

**Raymond, C.L.; Peterson, D.L. 2005.** Fuel treatments alter the effects of wildfire in a mixed-evergreen forest, Oregon, USA. Canadian Journal of Forest Research. 35(12): 2981–2995. doi:10.1139/x05-206.

**Reeves, G.H.; Benda, L.E.; Burnett, K.M.; Bisson, P.A.; Sedell, J.R. 1995.** A disturbance-based ecosystem approach to maintaining and restoring freshwater habitats of evolutionarily significant units of anadromous salmonids in the Pacific Northwest. American Fisheries Society Symposium. 17: 334–349.

**Reeves, G.H.; Bisson, P.A.; Rieman, B.E.; Benda, L.E. 2006.** Postfire logging in riparian areas. Conservation Biology. 20(4): 994–1004. doi:10.1111/j.1523-1739.2006.00502.x.

**Reilly, M.J.; Dunn, C.J.; Meigs, G.W.; Spies, T.A.; Kennedy, R.E.; Bailey, J.D.; Briggs, K. 2017.** Contemporary patterns of fire extent and severity in forests of the Pacific Northwest, USA (1985-2010). Ecosphere. 8(3): e01695. doi:10.1002/ecs2.1695.

**Reilly, M.J.; Spies, T.A. 2015.** Regional variation in stand structure and development in forests of Oregon, Washington, and inland northern California. Ecosphere. 6(10): art192. doi:10.1890/es14-00469.1.

**Reilly, M.J.; Spies, T.A. 2016.** Disturbance, tree mortality, and implications for contemporary regional forest change in the Pacific Northwest. Forest Ecology and Management. 374: 102–110. doi:10.1016/j.foreco.2016.05.002.

**Rhodes, J.J.; Baker, W.L. 2008.** Fire probability, fuel treatment effectiveness and ecological tradeoffs in western US public forests. Open Forest Science Journal. 1(1): 1–7.

**Rieman, B.; Clayton, J. 1997.** Wildfire and native fish: issues of forest health and conservation of sensitive species. Fisheries. 22(11): 6–15. doi:10.1577/1548-8446(1997)022<0006:wanfio>2.0.co;2.

**Rieman, B.E.; Lee, D.; Chandler, G.; Myers, D. 1997.** Does wildfire threaten extinction for salmonids: responses of redband trout and bull trout following recent large fires on the Boise National Forest. In: Proceedings of the symposium on fire effects on threatened and endangered species and their habitats. Fairfield, VI: International Association of Wildland Fire: 47–57.

**Ritchie, M.; Zhang, J.; Hamilton, T. 2013a.** Aboveground tree biomass for *Pinus ponderosa* in northeastern California. Forests. 4(1): 179–196. doi:10.3390/f4010179.

**Ritchie, M.W. 2005.** Ecological research at the Goosenest Adaptive Management Area in northeastern California. Gen. Tech. Rep. PSW-GTR-192. Albany, CA: U.S. Department of Agriculture, Forest Service, Pacific Southwest Research Station. 128 p. doi:10.2737/psw-gtr-192.

**Ritchie, M.W.; Knapp, E.E. 2014.** Establishment of a long-term fire salvage study in an interior ponderosa pine forest. Journal of Forestry. 112(5): 395–400. doi:10.5849/jof.13-093.

**Ritchie, M.W.; Knapp, E.E.; Skinner, C.N. 2013b.** Snag longevity and surface fuel accumulation following post-fire logging in a ponderosa pine dominated forest. Forest Ecology and Management. 287: 113–122. doi:10.1016/j.foreco.2012.09.001.

**Ritchie, M.W.; Skinner, C.N.; Hamilton, T.A. 2007.** Probability of tree survival after wildfire in an interior pine forest of northern California: effects of thinning and prescribed fire. Forest Ecology and Management. 247(1–3): 200-208. doi:10.1016/j.foreco.2007.04.044.

**Ritchie, M.W.; Wing, B.M.; Hamilton, T.A. 2008.** Stability of the large tree component in treated and untreated late-seral interior ponderosa pine stands. Canadian Journal of Forest Research. 38(5): 919–923. doi:10.1139/x07-242.

**Ritóková, G.; Shaw, D.; Filip, G.; Kanaskie, A.; Browning, J.; Norlander, D. 2016.** Swiss needle cast in western Oregon Douglas-fir plantations: 20-year monitoring results. Forests. 7(8): 155. doi:10.3390/f7080155.

**Rizzo, D.M.; Garbelotto, M.; Hansen, E.M. 2005.** *Phytophthora ramorum*: integrative research and management of an emerging pathogen in California and Oregon forests. Annual Review of Phytopathology. 43(1): 309–335. doi:10.1146/annurev.phyto.42.040803.140418.

**Robbins, D. 2005.** Temporal and spatial variability of historic fire frequency in the southern Willamette valley foothills of Oregon. Corvallis, OR: Oregon State University. 111 p. M.S. thesis.

**Robbins, W.G. 1999.** Landscapes of promise: the Oregon story, 1800–1940. Seattle, WA: University of Washington Press. 416 p.

**Rollins, M.G. 2009.** LANDFIRE: a nationally consistent vegetation, wildland fire, and fuel assessment. International Journal of Wildland Fire. 18(3): 235. doi:10.1071/wf08088.

**Roloff, G.J.; Mealey, S.P.; Clay, C.; Barry, J.; Yanish, C.; Neuenschwander, L. 2005.** A process for modeling short- and long-term risk in the southern Oregon Cascades. Forest Ecology and Management. 211(1–2): 166–190. doi:10.1016/j.foreco.2005.02.006.

**Root, H.T.; McCune, B.; Neitlich, P. 2010.** Lichen habitat may be enhanced by thinning treatments in young *Tsuga heterophylla–Pseudotsuga menziesii* forests. The Bryologist. 113(2): 292–307. doi:10.1639/0007-2745-113.2.292.

**Rosenvald, R.; Lõhmus, A. 2008.** For what, when, and where is green-tree retention better than clear-cutting? A review of the biodiversity aspects. Forest Ecology and Management. 255(1): 1–15. doi:10.1016/j.foreco.2007.09.016.

**Rothermel, R.C. 1983.** How to predict the spread and intensity of forest and range fires. Gen. Tech. Rep. INT-GTR-143. Ogden, UT: U.S. Department of Agriculture, Forest Service, Intermountain Forest and Range Experiment Station. 161 p. doi:10.2737/int-gtr-143.

**Ruggiero, L.F.; Aubry, K.B.; Carey, A.B.; Huff, M.H. 1991.** Wildlife and vegetation of unmanaged Douglas-fir forests. Gen. Tech. Rep. PNW-GTR-285. Portland, OR: U.S. Department of Agriculture, Forest Service, Pacific Northwest Research Station. 533 p. doi:10.2737/pnw-gtr-285.

**Russell, R.E.; Saab, V.A.; Dudley, J.G.; Rotella, J.J. 2006.** Snag longevity in relation to wildfire and postfire salvage logging. Forest Ecology and Management. 232(1–3): 179–187. doi:10.1016/j.foreco.2006.05.068.

**Ruzicka, K.J.; Puettmann, K.J.; Olson, D.H. 2014.** Management of riparian buffers: upslope thinning with downslope impacts. Forest Science. 60(5): 881–892. doi:10.5849/forsci.13-107.

**Ryan, M.G.; Harmon, M.E.; Birdsey, R.A.; Giardina, C.P.; Heath, L.S.; Houghton, R.A.; Jackson, R.B.; McKinley, D.C.; Morrison, J.F.; Murray, B.C.; Pataki, D.E. 2010.** A synthesis of the science on forests and carbon for US forests. Issues in Ecology. 13(1): 16.

**Saab, V.A.; Russell, R.E.; Dudley, J.G. 2007.** Nest densities of cavity-nesting birds in relation to postfire salvage logging and time since wildfire. Condor. 109: 97–108.

**Safford, H.D.; Miller, J.; Schmidt, D.; Roath, B.; Parsons, A. 2007.** BAER soil burn severity maps do not measure fire effects to vegetation: a comment on Odion and Hanson (2006). Ecosystems. 11(1): 1–11. doi:10.1007/s10021-007-9094-z.

**Safford, H.D.; Stevens, J.T. 2016.** Natural range of variation (NRV) for yellow pine and mixed conifer forests in the Sierra Nevada, southern Cascades, and Modoc and Inyo National Forests, California, USA. Gen. Tech. Rep. PSW-GTR-256. Albany, CA: U.S. Department of Agriculture, Forest Service, Pacific Southwest Research Station. 86 p.

**Safford, H.D.; Stevens, J.T.; Merriam, K.; Meyer, M.D.; Latimer, A.M. 2012.** Fuel treatment effectiveness in California yellow pine and mixed conifer forests. Forest Ecology and Management. 274: 17–28. doi:10.1016/j.foreco.2012.02.013.

**Safford, H.D.; Wiens, J.A.; Hayward; Gregory, D. 2012.** The growing importance of the past in managing ecosystems of the future. In: Wiens, J.A.; Hayward, G.D.; Hugh, D.; Giffen, C., eds. Historical environmental variation in conservation and natural resource management. Chichester, West Sussex, United Kingdom: Wiley-Blackwell: 319–327. http://dx.doi.org/10.1002/9781118329726.ch24.

**Scharpf, R.F. 1993.** Cankers, diebacks, and galls. In: Bega, R.V., tech. coord. Diseases of Pacific Coast conifers. Agric. Handb. 521. Washington, DC: U.S. Department of Agriculture, Forest Service: 61–82.

**Scheller, R.M.; Spencer, W.D.; Rustigian-Romsos, H.; Syphard, A.D.; Ward, B.C.; Strittholt, J.R. 2011.** Using stochastic simulation to evaluate competing risks of wildfires and fuels management on an isolated forest carnivore. Landscape Ecology. 26(10): 1491–1504. doi:10.1007/s10980-011-9663-6.

**Schmidt, D.A.; Taylor, A.H.; Skinner, C.N. 2008.** The influence of fuels treatment and landscape arrangement on simulated fire behavior, Southern Cascade Range, California. Forest Ecology and Management. 255(8–9): 3170-3184. doi:10.1016/j.foreco.2008.01.023.

**Schmidt, K.M.; Menakis, J.P.; Hardy, C.C.; Hann, W.J.; Bunnell, D.L. 2002.** Development of coarse-scale spatial data for wildland fire and fuel management. Gen. Tech. Rep. RMRS-GTR-87. Ogden, UT: U.S. Department of Agriculture, Forest Service, Rocky Mountain Research Station. 41 p. doi:10.2737/rmrs-gtr-87.

**Scholl, A.E.; Taylor, A.H. 2010.** Fire regimes, forest change, and self-organization in an old-growth mixed-conifer forest, Yosemite National Park, USA. Ecological Applications. 20(2): 362–380. doi:10.1890/08-2324.1.

**Schultz, C.A.; Sisk, T.D.; Noon, B.R.; Nie, M.A. 2013.** Wildlife conservation planning under the United States Forest Service's 2012 planning rule. Journal of Wildlife Management. 77: 428–444. doi:10.1002/jwmg.513.

**Seidel, D.; Ruzicka, K.J.; Puettmann, K. 2016.** Canopy gaps affect the shape of Douglas-fir crowns in the western Cascades, Oregon. Forest Ecology and Management. 363: 31–38. doi:10.1016/j.foreco.2015.12.024.

**Sensenig, T.; Bailey, J.D.; Tappeiner, J.C. 2013.** Stand development, fire and growth of old-growth and young forests in southwestern Oregon, USA. Forest Ecology and Management. 291: 96–109. doi:10.1016/j.foreco.2012.11.006.

**Seymour, R.S.; Hunter, M.L. 1999.** Principles of ecological forestry. In: Hunter, M.L., ed. Maintaining biodiversity in forest ecosystems. Cambridge, United Kingdom: Cambridge University Press: 22–61.

**Shaben, J.; Myers, J.H. 2009.** Relationships between Scotch broom (*Cytisus scoparius*), soil nutrients, and plant diversity in the Garry oak savannah ecosystem. Plant Ecology. 207(1): 81–91. doi:10.1007/s11258-009-9655-7.

**Shatford, J.; Bailey, J.D.; Tappeiner, J.C. 2009.** Understory tree development with repeated stand density treatments in coastal Douglas-fir forests of Oregon. Western Journal of Applied Forestry. 24(1): 11–16.

**Shatford, J.P.A.; Hibbs, D.E.; Puettmann, K.J. 2007.** Conifer regeneration after forest fire in the Klamath-Siskiyous: how much, how soon? Journal of Forestry. 105(3): 139–146.

**Shaw, D.C.; Agne, M.C. 2017.** Fire and dwarf mistletoe (Viscaceae: *Arceuthobium* species) in western North America: contrasting *Arceuthobium tsugense* and *Arceuthobium americanum*. Botany. 95(3): 231–246. doi:10.1139/cjb-2016-0245.

**Shaw, D.C.; Filip, G.M.; Kanaskie, A.; Maguire, D.A.; Littke, W. 2011.** Managing an epidemic of Swiss needle cast in the Douglas-fir region of Oregon: the Swiss needle cast cooperative. Journal of Forestry. 109: 109–119.

**Shaw, D.C.; Oester, P.T.; Filip, G.M. 2009.** Managing insects and diseases of Oregon conifers. Extension Publications EM 8980. Corvallis, OR: Oregon State University. 104 p.

**Shaw, D.C.; Watson, D.M.; Mathiasen, R.L. 2004.** Comparison of dwarf mistletoes (*Arceuthobium* spp., Viscaceae) in the western United States with mistletoes (*Amyema* spp., Loranthaceae) in Australia—ecological analogs and reciprocal models for ecosystem management. Australian Journal of Botany. 52(4): 481. doi:10.1071/bt03074.

**Sheridan, C.D.; Puettmann, K.J.; Huso, M.M.P.; Hagar, J.C.; Falk, K.R. 2013.** Management, morphological, and environmental factors influencing Douglas-fir bark furrows in the Oregon Coast Range. Western Journal of Applied Forestry. 28(3): 97–106. doi:10.5849/wjaf.12-011.

**Simpson, M. 2007.** Forested plant associations of the Oregon east Cascades. Portland, OR: U.S. Department of Agriculture, Forest Service, Pacific Northwest Region. https://ir.library.oregonstate.edu/downloads/7s75dj14z. (12 January 2018).

**Simpson, M. 2013.** Forest vegetation zone map. Bend, OR: U.S. Department of Agriculture, Forest Service, Central Oregon Area Ecology and Forest Health Program.

**Sinton, D.S.; Jones, J.A.; Ohmann, J.L.; Swanson, F.J. 2000.** Windthrow disturbance, forest composition, and structure in the Bull Run basin, Oregon. Ecology. 81(9): 2539–2556. doi:10.1890/0012-9658(2000)081[2539:wdfcas]2.0.co;2.

**Skinner, C.N. 1995.** Change in spatial characteristics of forest openings in the Klamath Mountains of northwestern California, USA. Landscape Ecology. 10(4): 219–228. doi:10.1007/bf00129256.

**Skinner, C.N. 2002.** Influence of fire on dead woody material in forests of California and southwestern Oregon. In: Laudenslayer, W.F.; Shea, P.J.; Valentine, B.E.; Weatherspoon, C.P.; Lisle, T.E., eds. Proceedings of the symposium on the ecology and management of dead wood in western forests. Gen. Tech. Rep. PSW-GTR-181. Albany, CA: U.S. Department of Agriculture, Forest Service, Pacific Southwest Research Station: 445–454.

**Skinner, C.N. 2003.** Fire regimes of upper montane and subalpine glacial basins in the Klamath Mountains of northern California. In: Galley, K.E.M.; Klinger, R.C.; Sugihara, N.G., eds. Proceedings of fire conference 2000: the first national congress on fire ecology, prevention, and management. Misc. Publ. 13. Tallahassee, FL: Tall Timbers Research Station: 145–151.

**Skinner, C.N.; Abbott, C.S.; Fry, D.L.; Stephens, S.L.; Taylor, A.H.; Trouet, V. 2009.** Human and climatic influences on fire occurrence in California's north Coast Range, USA. Fire Ecology. 5(3): 76–99. doi:10.4996/fireecology.0503076.

**Skinner, C.N.; Chang, C. 1996.** Fire regimes, past and present. Status of the Sierra Nevada. In: Erman, D.C., ed. Status of the Sierra Nevada: Sierra Nevada Ecosystem Project, final report to Congress. Report No. 37. Davis, CA: University of California, Centers for Water and Wildland Resources: 1041–1070. Vol. II: Assessments and scientific basis for management options.

**Skinner, C.N.; Taylor, A.H. [In press].** Southern Cascades bioregion. In: van Wagtendonk, J.W.; Sugihara, N.G.; Stephens, S.L.; Thode, A.E.; Shaffer, K.E.; Fites-Kaufmann, J., eds. Fire in California's ecosystems. 2nd ed. Berkeley, CA: University of California Press.

**Skinner, C.N.; Taylor, A.H.; Agee, J.K.; Briles, C.E.; Whitlock, C. [In press].** Klamath Mountains bioregion. In: van Wagtendonk, J.W.; Sugihara, N.G.; Stephens, S.L.; Thode, A.E.; Shaffer, K.E.; Fites-Kaufmann, J., eds. Fire in California's ecosystems. 2nd ed. Berkeley, CA: University of California Press.

**Skinner, C.N.; Weatherspoon, C.P. 1996.** Plantation characteristics affecting damage from wildfire. In: Proceedings of the 17th forest vegetation management conference. Redding, CA: [Publisher unknown]: 137–142. https://www.fs.fed.us/psw/publications/skinner/psw_1996_skinner003.pdf. (2017 October 2017).

**Smucker, K.M.; Hutto, R.L.; Steele, B.M. 2005.** Changes in bird abundance after wildfire: importance of fire severity and time since fire. Ecological Applications. 15(5): 1535–1549. doi:10.1890/04-1353.

**Soeriaatmadja, R.E. 1965.** Fire history of the ponderosa pine forests of the Warm Springs Indian Reservation, Oregon. Corvallis, OR: Oregon State University, Department of Botany and Plant Pathology. 123 p. Ph.D. dissertation. http://ir.library.oregonstate.edu/xmlui/handle/1957/47833. (1 August 2017).

**Sollmann, R.; White, A.M.; Tarbill, G.L.; Manley, P.N.; Knapp, E.E. 2016.** Landscape heterogeneity compensates for fuel reduction treatment effects on northern flying squirrel populations. Forest Ecology and Management. 373: 100–107. doi:10.1016/j.foreco.2016.04.041.

**Soulé, M.E. 1985.** What is conservation biology? A new synthetic discipline addresses the dynamics and problems of perturbed species, communities, and ecosystems. BioScience. 35(11): 727–734.

**Spies, T.A. 1991.** Plant species diversity and occurrence in young, mature, and old-growth Douglas-fir stands in western Oregon and Washington. In: Ruggiero, L.F.; Aubry, K.B.; Carey, A.B.; Huff, M.H., eds. Wildlife and vegetation of unmanaged Douglas-fir forests. Gen. Tech. Rep. PNW-GTR-285. Portland, OR: U.S. Department of Agriculture, Forest Service, Pacific Northwest Research Station: 111–121.

**Spies, T.A. 1997.** Forest stand structure, composition, and function. In: Kohm, A.; Franklin, J.F., eds. Creating a forestry for the 21st century—the science of ecosystem management. Washington, DC: Island Press: 11–30.

**Spies, T.A. 2004.** Ecological concepts and diversity of old-growth forests. Journal of Forestry. 102(3): 14–20.

**Spies, T.A. 2006.** Maintaining old-growth forests. In: Haynes, R.W.; Bormann, B.T.; Lee, D.C.; Martin, J.R., eds. Northwest Forest Plan—the first 10 years (1994–2003): synthesis of monitoring and research results. Gen. Tech. Rep. PNW-GTR-651. Portland, OR: U.S. Department of Agriculture, Forest Service, Pacific Northwest Research Station: 83–116.

**Spies, T.A.; Cline, S.P. 1988.** Coarse woody debris in forests and plantations of coastal Oregon. In: Maser, C.; Tarrant, R.F.; Trappe, J.M.; Franklin, J.F., eds. From the forest to the sea: a story of fallen trees. Gen. Tech. Rep. PNW-GTR-229. Portland, OR: U.S. Department of Agriculture, Forest Service, Pacific Northwest Research Station: 5–23. Chapter 1.

**Spies, T.A.; Duncan, S.L. 2009.** Old growth in a new world: a Pacific Northwest icon reexamined. Washington, DC: Island Press. 360 p.

**Spies, T.A.; Franklin, J.F. 1989.** Gap characteristics and vegetation response in coniferous forests of the Pacific Northwest. Ecology. 70(3): 543–545. doi:10.2307/1940198.

**Spies, T.A.; Franklin, J.F. 1991.** The structure of natural young, mature, and old-growth Douglas-fir forests in Oregon and Washington. In: Ruggiero, L.F.; Aubry, K.B.; Carey, A.B.; Huff, M.H., eds. Wildlife and vegetation of unmanaged Douglas-fir forests. Gen. Tech. Rep. PNW-GTR-285. Portland, OR: U.S. Department of Agriculture, Forest Service, Pacific Northwest Research Station: 111–121.

**Spies, T.A.; Franklin, J.F. 1996.** The diversity and maintenance of old-growth forests. In: Szaro, R.C.; Johnston, D.W., eds. Biodiversity in managed landscapes: theory and practice. New York, NY: Oxford University Press: 296–314.

**Spies, T.A.; Franklin, J.F.; Klopsch, M. 1990.** Canopy gaps in Douglas-fir forests of the Cascade Mountains. Canadian Journal of Forest Research. 20(5): 649–658. doi:10.1139/x90-087.

**Spies, T.A.; Franklin, J.F.; Thomas, T.B. 1988.** Coarse woody debris in Douglas-fir forests of western Oregon and Washington. Ecology. 69(6): 1689–1702. doi:10.2307/1941147.

**Spies, T.A.; Giesen, T.W.; Swanson, F.J.; Franklin, J.F.; Lach, D.; Johnson, K.N. 2010.** Climate change adaptation strategies for federal forests of the Pacific Northwest, USA: ecological, policy, and socio-economic perspectives. Landscape Ecology. 25(8): 1185–1199. doi:10.1007/s10980-010-9483-0.

**Spies, T.A.; Hemstrom, M.A.; Youngblood, A.; Hummel, S. 2006.** Conserving old-growth forest diversity in disturbance-prone landscapes. Conservation Biology. 20(2): 351–362. doi:10.1111/j.1523-1739.2006.00389.x.

**Spies, T.A.; Hibbs, D.E.; Ohmann, J.L.; Reeves, G.H.; Pabst, R.J.; Swanson, F.J.; Whitlock, C.; Jones, J.A.; Wemple, B.C.; Parendes, L.A.; Schrader, B.A. 2002.** The ecological basis of forest ecosystem management in the Oregon Coast Range. In: Hobbs, S.D.; Hayes, J.P., eds. Forest and stream management in the Oregon Coast Range. Corvallis, OR: Oregon State University Press: 31–67.

**Spies, T.A.; Johnson, K.N.; Burnett, K.M.; Ohmann, J.L.; McComb, B.C.; Reeves, G.H.; Bettinger, P.; Kline, J.D.; Garber-Yonts, B. 2007.** Cumulative ecological and socioeconomic effects of forest policies in coastal Oregon. Ecological Applications. 17(1): 5–17. doi:10.1890/1051-0761(2007)017[0005:ceaseo]2.0.co;2.

**Spies, T.A.; Lindenmayer, D.B.; Gill, A.M.; Stephens, S.L.; Agee, J.K. 2012.** Challenges and a checklist for biodiversity conservation in fire-prone forests: perspectives from the Pacific Northwest of USA and southeastern Australia. Biological Conservation. 145(1): 5–14. doi:10.1016/j.biocon.2011.09.008.

**Spies, T.A.; McComb, B.C.; Kennedy, R.S.H.; McGrath, M.T.; Olsen, K.; Pabst, R.J. 2007.** Potential effects of forest policies on terrestrial biodiversity in a multi-ownership province. Ecological Applications. 17(1): 48–65. doi:10.1890/1051-0761(2007)017[0048:peofpo]2.0.co;2.

**Spies, T.A.; Ripple, W.J.; Bradshaw, G.A. 1994.** Dynamics and pattern of a managed coniferous forest landscape in Oregon. Ecological Applications. 4(3): 555–568. doi:10.2307/1941957.

**Spies, T.A.; Turner, M.G. 1999.** Dynamic forest mosaics. In: Hunter, M.L., ed. Maintaining biodiversity in forest ecosystems. Cambridge, United Kingdom: Cambridge University Press: 95–160.

**Spies, T.A.; White, E.; Ager, A.; Kline, J.D.; Bolte, J.P.; Platt, E.K.; Olsen, K.A.; Pabst, R.J.; Barros, A.M.G.; Bailey, J.D.; Charnley, S.; Morzillo, A.T.; Koch, J.; Steen-Adams, M.M.; Singleton, P.H.; Sulzman, J.; Schwartz, C.; Csuti, B. 2017.** Using an agent-based model to examine forest management outcomes in a fire-prone landscape in Oregon, USA. Ecology and Society. 22(1) doi:10.5751/es-08841-220125.

**Stavros, E.N.; Abatzoglou, J.T.; McKenzie, D.; Larkin, N.K. 2014.** Regional projections of the likelihood of very large wildland fires under a changing climate in the contiguous western United States. Climatic Change. 126(3–4): 455–468. http://dx.doi.org/10.1007/s10584-014-1229-6.

**Stephens, S.L. 1998.** Evaluation of the effects of silvicultural and fuels treatments on potential fire behaviour in Sierra Nevada mixed-conifer forests. Forest Ecology and Management. 105(1–3): 21–35. doi:10.1016/s0378-1127(97)00293-4.

**Stephens, S.L.; Finney, M.A. 2002.** Prescribed fire mortality of Sierra Nevada mixed conifer tree species: effects of crown damage and forest floor combustion. Forest Ecology and Management. 162(2–3): 261–271. doi:10.1016/s0378-1127(01)00521-7.

**Stephens, S.L.; Fry, D.L. 2005.** Fire history in coast redwood stands in the northeastern Santa Cruz mountains, California. Fire Ecology. 1(1): 2–19. doi:10.4996/fireecology.0101002.

**Stephens, S.L.; Fule, P.Z. 2005.** Western pine forests with continuing frequent fire regimes: possible reference sites for management. Journal of Forestry. 103(7): 357–362.

**Stephens, S.L.; Lydersen, J.M.; Collins, B.M.; Fry, D.L.; Meyer, M.D. 2015.** Historical and current landscape-scale ponderosa pine and mixed conifer forest structure in the southern Sierra Nevada. Ecosphere. 6(5): art79. doi:10.1890/es14-00379.1.

**Stephens, S.L.; Millar, C.I.; Collins, B.M. 2010.** Operational approaches to managing forests of the future in Mediterranean regions within a context of changing climates. Environmental Research Letters. 5(2): 024003. doi:10.1088/1748-9326/5/2/024003.

**Stephens, S.L.; Moghaddas, J.J. 2005.** Experimental fuel treatment impacts on forest structure, potential fire behavior, and predicted tree mortality in a California mixed conifer forest. Forest Ecology and Management. 215(1–3): 21–36. doi:10.1016/j.foreco.2005.03.070.

**Stephens, S.L.; Moghaddas, J.J.; Edminster, C.; Fiedler, C.E.; Haase, S.; Harrington, M.; Keeley, J.E.; Knapp, E.E.; McIver, J.D.; Metlen, K.; Skinner, C.N.; Youngblood, A. 2009.** Fire treatment effects on vegetation structure, fuels, and potential fire severity in western U.S. forests. Ecological Applications. 19(2): 305–320. doi:10.1890/07-1755.1.

**Stephens, S.L.; Ruth, L.W. 2005.** Federal forest-fire policy in the United States. Ecological Applications. 15(2): 532–542. doi:10.1890/04-0545.

**Stephenson, N.L.; Das, A.J.; Condit, R.; Russo, S.E.; Baker, P.J.; Beckman, N.G.; Coomes, D.A.; Lines, E.R.; Morris, W.K.; Rüger, N.; Álvarez, E.; Blundo, C.; Bunyavejchewin, S.; Chuyong, G.; Davies, S.J.; Duque, Á.; Ewango, C.N.; Flores, O.; Franklin, J.F.; Grau, H.R.; Hao, Z.; Harmon, M.E.; Hubbell, S.P.; Kenfack, D.; Lin, Y.; Makana, J.R.; Malizia, A.; Malizia, L.R.; Pabst, R.J.; Pongpattananurak, N.; Su, S.H.; Sun, I.F.; Tan, S.; Thomas, D.; van Mantgem, P.J.; Wang, X.; Wiser, S.K.; Zavala, M.A. 2014.** Rate of tree carbon accumulation increases continuously with tree size. Nature. 507(7490): 90–93. doi:10.1038/nature12914.

**Stevens, J.T.; Safford, H.D.; Harrison, S.; Latimer, A.M. 2015.** Forest disturbance accelerates thermophilization of understory plant communities. Journal of Ecology. 103(5): 1253–1263. doi:10.1111/1365-2745.12426.

**Stevens, J.T.; Safford, H.D.; North, M.P.; Fried, J.S.; Gray, A.N.; Brown, P.M.; Dolanc, C.R.; Dobrowski, S.Z.; Falk, D.A.; Farris, C.A.; Franklin, J.F.; Fulé, P.Z.; Hagmann, R.K.; Knapp, E.E.; Miller, J.D.; Smith, D.F.; Swetnam, T.W.; Taylor, A.H. 2016.** Average stand age from forest inventory plots does not describe historical fire regimes in ponderosa pine and mixed-conifer forests of western North America. PLoS ONE. 11(5): e0147688. doi:10.1371/journal.pone.0147688.

**Stewart, G.H. 1986.** Forest development in canopy openings in old-growth *Pseudotsuga* forests of the western Cascade Range, Oregon. Canadian Journal of Forest Research. 16(3): 558–568. doi:10.1139/x86-097.

**Stewart, O.C. 2002.** Forgotten fires: Native Americans and the transient wilderness. Norman, OK: University of Oklahoma Press. 384 p.

**Stine, P.; Hessburg, P.; Spies, T.; Kramer, M.; Fettig, C.J.; Hansen, A.; Lehmkuhl, J.; O'Hara, K.; Polivka, K.; Singleton, P.; Charnley, S.; Merschel, A.; White, R. 2014.** 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. Gen. Tech. Rep. PNW-GTR-897. Portland, OR: U.S. Department of Agriculture, Forest Service, Pacific Northwest Research Station. 254 p. doi:10.2737/pnw-gtr-897.

**Stokes, D.L.; Church, E.D.; Cronkright, D.M.; Lopez, S. 2014.** Pictures of an invasion: English holly (*Ilex aquifolium*) in a semi-natural Pacific Northwest forest. Northwest Science. 88(2): 75–93. doi:10.3955/046.088.0204.

**Stuart, J.D. 1987.** Fire history of an old-growth forest of *Sequoia sempervirens (*Taxodiaceae*)* in Humboldt Redwoods State Park, California. Madrono. 34: 128–141.

**Stuart, J.D.; Grifantini, M.C.; Fox, L. 1993.** Early successional pathways following wildfire and subsequent silvicultural treatment in Douglas-fir /hardwood forests, NW California. Forest Science. 39: 561–572.

**Stuart, J.D.; Salazar, L.A. 2000.** Fire history of white fir forests in the coastal mountains of northwestern California. Northwest Science. 74(4): 280–284.

**Swanson, F.J.; Fredriksen, R.L.; McCorison, F.M. 1982.** Material transfer in a western Oregon forested watershed. Stroudsburg, PA: Hutchinson Ross Publishing Company: 233–266.

**Swanson, M.E.; Franklin, J.F.; Beschta, R.L.; Crisafulli, C.M.; DellaSala, D.A.; Hutto, R.L.; Lindenmayer, D.B.; Swanson, F.J. 2011.** The forgotten stage of forest succession: early-successional ecosystems on forest sites. Frontiers in Ecology and the Environment. 9(2): 117–125. doi:10.1890/090157.

**Swanson, M.E.; Studevant, N.M.; Campbell, J.L.; Donato, D.C. 2014.** Biological associates of early-seral pre-forest in the Pacific Northwest. Forest Ecology and Management. 324: 160–171. doi:10.1016/j.foreco.2014.03.046.

**Swetnam, T.W.; Allen, C.D.; Betancourt, J.L. 1999.** Applied historical ecology: using the past to manage for the future. Ecological Applications. 9(4): 1189–1206. doi:10.1890/1051-0761(1999)009[1189:aheutp]2.0.co;2.

**Swiecki, T.J.; Bernhardt, E.A. 2013.** A reference manual for managing sudden oak death in California. Gen. Tech. Rep. PSW-GTR-242. Albany, CA: U.S. Department of Agriculture, Forest Service, Pacific Southwest Research Station. 129 p. doi:10.2737/psw-gtr-242.

**Tappeiner, J.C.; Huffman, D.W.; Marshall, D.; Spies, T.A.; Bailey, J.D. 1997.** Density, ages, and growth rates in old-growth and young-growth forests in coastal Oregon. Canadian Journal of Forest Research. 27(5): 638–648. doi:10.1139/x97-015.

**Tappeiner, J.C.; Maguire, D.A.; Harrington, T.B. 2007.** Silviculture and ecology of western US forests. Corvallis, OR: Oregon State University Press. 416 p.

**Taylor, A.H. 1993.** Fire history and structure of red fir (*Abies magnifica*) forests, Swain Mountain Experimental Forest, Cascade Range, northeastern California. Canadian Journal of Forest Research. 23(8): 1672–1678. doi:10.1139/x93-208.

**Taylor, A.H. 2000.** Fire regimes and forest changes in mid and upper montane forests of the southern Cascades, Lassen Volcanic National Park, California, U.S.A. Journal of Biogeography. 27(1): 87–104. doi:10.1046/j.1365-2699.2000.00353.x.

**Taylor, A.H.; Halpern, C.B. 1991.** The structure and dynamics of *Abies magnifica* forests in the southern Cascade Range, USA. Journal of Vegetation Science. 2(2): 189–200. doi:10.2307/3235951.

**Taylor, A.H.; Skinner, C.N. 1998.** Fire history and landscape dynamics in a late-successional reserve, Klamath Mountains, California, USA. Forest Ecology and Management. 111(2–3): 285–301. doi:10.1016/s0378-1127(98)00342-9.

**Taylor, A.H.; Skinner, C.N. 2003.** Spatial patterns and controls on historical fire regimes and forest structure in the Klamath mountains. Ecological Applications. 13(3): 704–719. doi:10.1890/1051-0761(2003)013[0704:spacoh]2.0.co;2.

**Taylor, A.H.; Solem, M.N. 2001.** Fire regimes and stand dynamics in an upper montane forest landscape in the southern Cascades, Caribou Wilderness, California. Journal of the Torrey Botanical Society. 128(4): 350. doi:10.2307/3088667.

**Taylor, A.H.; Trouet, V.; Skinner, C.N. 2008.** Climatic influences on fire regimes in montane forests of the southern Cascades, California, USA. International Journal of Wildland Fire. 17(1): 60. doi:10.1071/wf07033.

**Taylor, L.A.V.; Cruzan, M.B. 2015.** Propagule pressure and disturbance drive the invasion of perennial false-brome (*Brachypodium sylvaticum*). Invasive Plant Science and Management. 8(02): 169–180. doi:10.1614/ipsm-d-14-00042.1.

**Teensma, P.D. 1987.** Fire history and fire regimes of the central western Cascades of Oregon. Eugene, OR: University of Oregon. 188 p. Ph.D. dissertation.

**Teensma, P.D.A.; Rienstra, J.T.; Yeiter, M.A. 1991.** Preliminary reconstruction and analysis of change in forest stand age classes of the Oregon Coast Range from 1850 to 1940. Portland, OR: U.S. Department of the Interior, Bureau of Land Management, Oregon State Office. 22 p.

**Tepley, A.J. 2010.** Age structure, developmental pathways, and fire regime characterization of Douglas-fir/western hemlock forests in the central western Cascades of Oregon. Corvallis, OR: Oregon State University. 278 p. Ph.D. dissertation. https://ir.library.oregonstate.edu/xmlui/handle/1957/18113. (9 October 2017).

**Tepley, A.J.; Swanson, F.J.; Spies, T.A. 2013.** Fire-mediated pathways of stand development in Douglas-fir/western hemlock forests of the Pacific Northwest, USA. Ecology. 94(8): 1729–1743. doi:10.1890/12-1506.1.

**Tepley, A.J.; Swanson, F.J.; Spies, T.A. 2014.** Post-fire tree establishment and early cohort development in conifer forests of the western Cascades of Oregon, USA. Ecosphere. 5(7): art80. doi:10.1890/es14-00112.1.

**Tepley, A.J.; Thompson, J.R.; Epstein, H.E.; Anderson-Teixeira, K.J. 2017.** Vulnerability to forest loss through altered postfire recovery dynamics in a warming climate in the Klamath Mountains. Global Change Biology. 23(10): 4117–4132. doi:10.1111/gcb.13704.

**Thomas, S.C.; Halpern, C.B.; Falk, D.A.; Liguori, D.A.; Austin, K.A. 1999.** Plant diversity in managed forests: understory responses to thinning and fertilization. Ecological Applications. 9(3): 864–879. doi:10.1890/1051-0761(1999)009[0864:pdimfu]2.0.co;2.

**Thompson, G.; Johnson, A.J. 1900.** Map of the state of Oregon showing the classification of lands and forests. Washington, DC: U.S. Department of the Interior, Geological Survey.

**Thompson, J.R.; Spies, T.A. 2009.** Vegetation and weather explain variation in crown damage within a large mixed-severity wildfire. Forest Ecology and Management. 258(7): 1684–1694. doi:10.1016/j.foreco.2009.07.031.

**Thompson, J.R.; Spies, T.A. 2010.** Factors associated with crown damage following recurring mixed-severity wildfires and post-fire management in southwestern Oregon. Landscape Ecology. 25(5): 775–789. doi:10.1007/s10980-010-9456-3.

**Thompson, J.R.; Spies, T.A.; Ganio, L.M. 2007.** Reburn severity in managed and unmanaged vegetation in a large wildfire. Proceedings of the National Academy of Sciences of the United States of America. 104(25): 10743–10748. doi:10.1073/pnas.0700229104.

**Thompson, J.R.; Spies, T.A.; Olsen, K.A. 2011.** Canopy damage to conifer plantations within a large mixed-severity wildfire varies with stand age. Forest Ecology and Management. 262(3): 355–360. doi:10.1016/j.foreco.2011.04.001.

**Tingley, M.W.; Ruiz-Gutiérrez, V.; Wilkerson, R.L.; Howell, C.A.; Siegel, R.B. 2016.** Pyrodiversity promotes avian diversity over the decade following forest fire. Proceedings of the Royal Society B: Biological Sciences. 283(1840): 20161703. doi:10.1098/rspb.2016.1703.

**Trombulak, S.C.; Frissell, C.A. 2000.** Review of ecological effects of roads on terrestrial and aquatic communities. Conservation Biology. 14(1): 18–30. doi:10.1046/j.1523-1739.2000.99084.x.

**Trouet, V.; Taylor, A.H.; Wahl, E.R.; Skinner, C.N.; Stephens, S.L. 2010.** Fire-climate interactions in the American West since 1400 CE. Geophysical Research Letters. 37(4): doi:10.1029/2009gl041695.

**Turner, M.G.; Hargrove, W.W.; Gardner, R.H.; Romme, W.H. 1994.** Effects of fire on landscape heterogeneity in Yellowstone National Park, Wyoming. Journal of Vegetation Science. 5(5): 731–742. doi:10.2307/3235886.

**U.S. Department of Agriculture, Forest Service [USDA FS]. 1989.** Generic definition of old-growth forests and position statement on national forest old-growth values. Memorandum to regional foresters, station directors and Washington Office staff, October 11, 1989. Washington, DC.

**U.S. Department of Agriculture, Forest Service [USDA FS]. 2012a.** National Forest System land management planning. 36 C.F.R. Part 219. Federal Register. 77: 21260–21276.

**U.S. Department of Agriculture, Forest Service [USDA FS]. 2012b.** The Okanogan-Wenatchee National Forest restoration strategy: adaptive ecosystem management to restore landscape resiliency. Portland, OR: Pacific Northwest Region. 118 p. http://www.fs.usda.gov/Internet/FSE_DOCUMENTS/stelprdb5340103.pdf. (17 February 2016).

**U.S. Department of Agriculture, Forest Service; U.S. Department of the Interior, Bureau of Land Management [USDA and USDI]. 1994.** Standards and guidelines for management of habitat for late-successional and old-growth forest related species within the range of the northern spotted owl; attachment A to the record of decision. [Place of publication unknown].

**U.S. Department of the Interior [USDI]. 2012.** Endangered and threatened wildlife and plants; designation of revised critical habitat for the northern spotted owl. 50 C.F.R. Part 17. Federal Register. 77(233): 71876–72066.

**U.S. Department of the Interior, Fish and Wildlife Service [USFWS]. 2008.** Final recovery plan for the northern spotted owl (*Strix occidentalis caurina*). Portland, OR. xii + 142 p.

**Urgenson, L.S.; Halpern, C.B.; Anderson, P.D. 2013.** Twelve-year responses of planted and naturally regenerating conifers to variable-retention harvest in the Pacific Northwest, USA. Canadian Journal of Forest Research. 43(1): 46–55. doi:10.1139/cjfr-2012-0323.

**Uzoh, F.C.C.; Skinner, C.N. 2009.** Effects of creating two forest structures and using prescribed fire on coarse woody debris in northeastern California, USA. Fire Ecology. 5(2): 1–13. doi:10.4996/fireecology.0502001.

**van Norman, K.J. 1998.** Historical fire regime in the Little River watershed, southwestern Oregon. Corvallis, OR: Oregon State University. 103 p. Ph.D. dissertation.

**van Pelt, R. 2008.** Identifying old trees and forests in eastern Washington. Olympia, WA: Washington State Department of Natural Resources. 166 p.

**van Wagner, C.E. 1978.** Age-class distribution and the forest fire cycle. Canadian Journal of Forest Research. 8(2): 220–227. doi:10.1139/x78-034.

**van Wagtendonk, J.W.; van Wagtendonk, K.A.; Thode, A.E. 2012.** Factors associated with the severity of intersecting fires in Yosemite National Park, California, USA. Fire Ecology. 7(1): 11–31. doi:10.4996/fireecology.0801011.

**Veldman, J.W.; Buisson, E.; Durigan, G.; Fernandes, G.W.; Le Stradic, S.; Mahy, G.; Negreiros, D.; Overbeck, G.E.; Veldman, R.G.; Zaloumis, N.P.; Putz, F.E.; Bond, W.J. 2015.** Toward an old-growth concept for grasslands, savannas, and woodlands. Frontiers in Ecology and the Environment. 13(3): 154–162. doi:10.1890/140270.

**Virkkala, R.; Heikkinen, R.K.; Fronzek, S.; Leikola, N. 2013.** Climate change, northern birds of conservation concern and matching the hotspots of habitat suitability with the reserve network. PLoS ONE. 8(5): e63376. doi:10.1371/journal.pone.0063376.

**Wagenbrenner, J.W.; MacDonald, L.H.; Coats, R.N.; Robichaud, P.R.; Brown, R.E. 2015.** Effects of post-fire salvage logging and a skid trail treatment on ground cover, soils, and sediment production in the interior western United States. Forest Ecology and Management. 335: 176–193. doi:10.1016/j.foreco.2014.09.016.

**Walsh, M.K.; Marlon, J.R.; Goring, S.J.; Brown, K.J.; Gavin, D.G. 2015.** A regional perspective on holocene fire–climate–human interactions in the Pacific Northwest of North America. Annals of the Association of American Geographers. 105(6): 1135–1157. doi:10.1080/00045608.2015.1064457.

**Walter, S.T.; Maguire, C.C. 2005.** Snags, cavity-nesting birds, and silvicultural treatments in western Oregon. Journal of Wildlife Management. 69(4): 1578–1591. doi:10.2193/0022-541X(2005)69[1578:SCBAST]2.0.CO;2.

**Waring, K.M.; O'Hara, K.L. 2008.** Redwood/tanoak stand development and response to tanoak mortality caused by *Phytophthora ramorum*. Forest Ecology and Management. 255(7): 2650–2658. doi:10.1016/j.foreco.2008.01.025.

**Washington State Academy of Sciences. 2013.** Opportunities for addressing laminated root rot caused by *Phellinus sulphuracens* in Washington's forests. Olympia, WA: Washington State Academy of Sciences in cooperation with the Washington State Department of Natural Resources. 110 p.

**Weatherspoon, C.P.; Husari, S.J.; van Wagtendonk, J.W. 1992.** Fire and fuels management in relation to owl habitat in forests of the Sierra Nevada and southern California. In: Verner, J.; McKelvey, K.S.; Noon, B.R.; Gutierrez, R.J.; G.I. Gould, J.; Beck, T.W., eds. The California spotted owl: a technical assessment of its current status. Gen. Tech. Rep. PSW-GTR-133. Albany, CA: U.S. Department of Agriculture, Forest Service, Pacific Southwest Research Station: 247–260. Chapter 12.

**Weatherspoon, C.P.; Skinner, C.N. 1995.** An assessment of factors associated with damage to tree crowns from the 1987 wildfires in northern California. Forest Science. 41: 430–451.

**Weaver, H. 1943.** Fire as an ecological and silvicultural factor in the ponderosa-pine region of the Pacific slope. Journal of Forestry. 41(1): 7–15.

**Weaver, H. 1959.** Ecological changes in the ponderosa pine forest of the Warm Springs Indian Reservation in Oregon. Journal of Forestry. 57: 15–20.

**Webster, K.M.; Halpern, C.B. 2010.** Long-term vegetation responses to reintroduction and repeated use of fire in mixed-conifer forests of the Sierra Nevada. Ecosphere. 1(5): 9. doi:10.1890/es10-00018.1.

**Weisberg, P.J. 1998.** Fire history, fire regimes, and development of forest structure in the central western Oregon Cascades. Corvallis, OR: Oregon State University. 256 p. Ph. D. dissertation.

**Weisberg, P.J. 2004.** Importance of non-stand-replacing fire for development of forest structure in the Pacific Northwest, USA. Forest Science. 50(2): 245–258.

**Weisberg, P.J. 2009.** Historical fire frequency on contrasting slope facets along the Mckenzie River, western Oregon Cascades. Western North American Naturalist. 69(2): 206–214. doi:10.3398/064.069.0209.

**Weisberg, P.J.; Swanson, F.J. 2001.** Fire dating from tree rings in western Cascades Douglas-fir forests: an error analysis. Northwest Science. 75: 145–156.

**Weisberg, P.J.; Swanson, F.J. 2003.** Regional synchroneity in fire regimes of western Oregon and Washington, USA. Forest Ecology and Management. 172(1): 17–28. doi:10.1016/s0378-1127(01)00805-2.

**Weiskittel, A.R.; Garber, S.M.; Johnson, G.P.; Maguire, D.A.; Monserud, R.A. 2007.** Annualized diameter and height growth equations for Pacific Northwest plantation-grown Douglas-fir, western hemlock, and red alder. Forest Ecology and Management. 250(3): 266–278. doi:10.1016/j.foreco.2007.05.026.

**Wender, B.W.; Harrington, C.A.; Tappeiner, J.C. 2004.** Flower and fruit production of understory shrubs in western Washington and Oregon. Northwest Science. 78(2): 124–140.

**West, N.E. 1969.** Tree patterns in central Oregon ponderosa pine forests. American Midland Naturalist. 81(2): 584. doi:10.2307/2423994.

**Westerling, A.L. 2006.** Warming and earlier spring increase western U.S. Forest wildfire activity. Science. 313(5789): 940–943. doi:10.1126/science.1128834.

**Wetzel, S.A.; Fonda, R.W. 2000.** Fire history of Douglas-fir forests in the Morse Creek drainage of Olympic National Park, Washington. Northwest Science. 74: 263–278.

**White, A.M.; Manley, P.N.; Tarbill, G.L.; Richardson, T.W.; Russell, R.E.; Safford, H.D.; Dobrowski, S.Z. 2015.** Avian community responses to post-fire forest structure: implications for fire management in mixed conifer forests. Animal Conservation. 19(3): 256–264. doi:10.1111/acv.12237.

**White, D.E.; Newton, M. 1989.** Competitive interactions of whiteleaf manzanita, herbs, Douglas-fir, and ponderosa pine in southwest Oregon. Canadian Journal of Forest Research. 19(2): 232–238. doi:10.1139/x89-032.

**Whitlock, C. 1992.** Vegetational and climatic history of the Pacific Northwest during the last 20,000 years: implications for understanding present-day biodiversity. The Northwest Environmental Journal. 8: 5–28.

**Whitlock, C.; Marlon, J.; Briles, C.; Brunelle, A.; Long, C.; Bartlein, P. 2008.** Long-term relations among fire, fuel, and climate in the north-western US based on lake-sediment studies. International Journal of Wildland Fire. 17(1): 72. doi:10.1071/wf07025.

**Whitlock, C.; Skinner, C.N.; Bartlein, P.J.; Minckley, T.; Mohr, J.A. 2004.** Comparison of charcoal and tree-ring records of recent fires in the eastern Klamath Mountains, California, USA. Canadian Journal of Forest Research. 34(10): 2110–2121. doi:10.1139/x04-084.

**Wiens, J.A.; Hayward, G.D.; Hugh, D.; Giffen, C., eds. 2012.** Historical environmental variation in conservation and natural resource management. Chichester, West Sussex, United Kingdom: John Wiley & Sons, Inc. 352 p.

**Williams, M.A.; Baker, W.L. 2011**. Testing the accuracy of new methods for reconstructing historical structure of forest landscapes using GLO survey data. Ecological Monographs. 81*:* 63–88. doi:10.1890/10-0256.1.

**Williams, M.A.; Baker, W.L. 2012.** Spatially extensive reconstructions show variable-severity fire and heterogeneous structure in historical western United States dry forests. Global Ecology and Biogeography. 21(10): 1042–1052. doi:10.1111/j.1466-8238.2011.00750.x.

**Williams, M.A.; Baker, W.L. 2014.** High-severity fire corroborated in historical dry forests of the western United States: response to Fulé et al. Global Ecology and Biogeography. 23(7): 831–835. doi:10.1111/geb.12152.

**Wilson, J.S.; Oliver, C.D. 2000.** Stability and density management in Douglas-fir plantations. Canadian Journal of Forest Research. 30(6): 910–920. doi:10.1139/x00-027.

**Wilson, T.M.; Forsman, E.D. 2013.** Thinning effects on spotted owl prey and other forest-dwelling small mammals. In: Anderson, P.D.; Ronnenberg, K.L., eds. Density management for the 21st century: west side story. Gen. Tech. Rep. PNW-GTR-880. Portland, OR: U.S. Department of Agriculture, Forest Service, Pacific Northwest Research Station: 79–90.

**Wimberly, M.C. 2002.** Spatial simulation of historical landscape patterns in coastal forests of the Pacific Northwest. Canadian Journal of Forest Research. 32(8): 1316–1328. doi:10.1139/x02-054.

**Wimberly, M.C.; Spies, T.A. 2001.** Influences of environment and disturbance on forest patterns in coastal Oregon watersheds. Ecology. 82(5): 1443–1459. doi:10.1890/0012-9658(2001)082[1443:ioeado]2.0.co;2.

**Wimberly, M.C.; Spies, T.A.; Long, C.J.; Whitlock, C. 2000.** Simulating historical variability in the amount of old forests in the Oregon Coast Range. Conservation Biology. 14(1): 167-180. doi:10.1046/j.1523-1739.2000.98284.x.

**Winter, L.E.; Brubaker, L.B.; Franklin, J.F.; Miller, E.A.; DeWitt, D.Q. 2002a.** Canopy disturbances over the five-century lifetime of an old-growth Douglas-fir stand in the Pacific Northwest. Canadian Journal of Forest Research. 32(6): 1057–1070. doi:10.1139/x02-030.

**Winter, L.E.; Brubaker, L.B.; Franklin, J.F.; Miller, E.A.; DeWitt, D.Q. 2002b.** Initiation of an old-growth Douglas-fir stand in the Pacific Northwest: a reconstruction from tree-ring records. Canadian Journal of Forest Research. 32(6): 1039–1056. doi:10.1139/x02-031.

**Wondzell, S.M. 2001.** The influence of forest health and protection treatments on erosion and stream sedimentation in forested watersheds of eastern Oregon and Washington. Northwest Science. 75: 128–140.

**Wood, D.; Koerber, T.; Scharpf, R.; Storer, A. 2003.** Pests of the native California conifers. Berkeley, CA: University of California Press. 245 p.

**Wright, C.S.; Agee, J.K. 2004.** Fire and vegetation history in the eastern Cascade Mountains, Washington. Ecological Applications. 14(2): 443–459. doi:10.1890/02-5349.

**Yang, Z.; Cohen, W.B.; Harmon, M.E. 2005.** Modeling early forest succession following clear-cutting in western Oregon. Canadian Journal of Forest Research. 35(8): 1889–1900. doi:10.1139/x05-132.

**Yegorova, S.; Betts, M.G.; Hagar, J.; Puettmann, K.J. 2013.** Bird-vegetation associations in thinned and unthinned young Douglas-fir forests 10 years after thinning. Forest Ecology and Management. 310: 1057–1070. doi:10.1016/j.foreco.2013.06.052.

**Youngblood, A.; Max, T.; Coe, K. 2004.** Stand structure in eastside old-growth ponderosa pine forests of Oregon and northern California. Forest Ecology and Management. 199(2–3): 191–217. doi:10.1016/j.foreco.2004.05.056.

## 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.

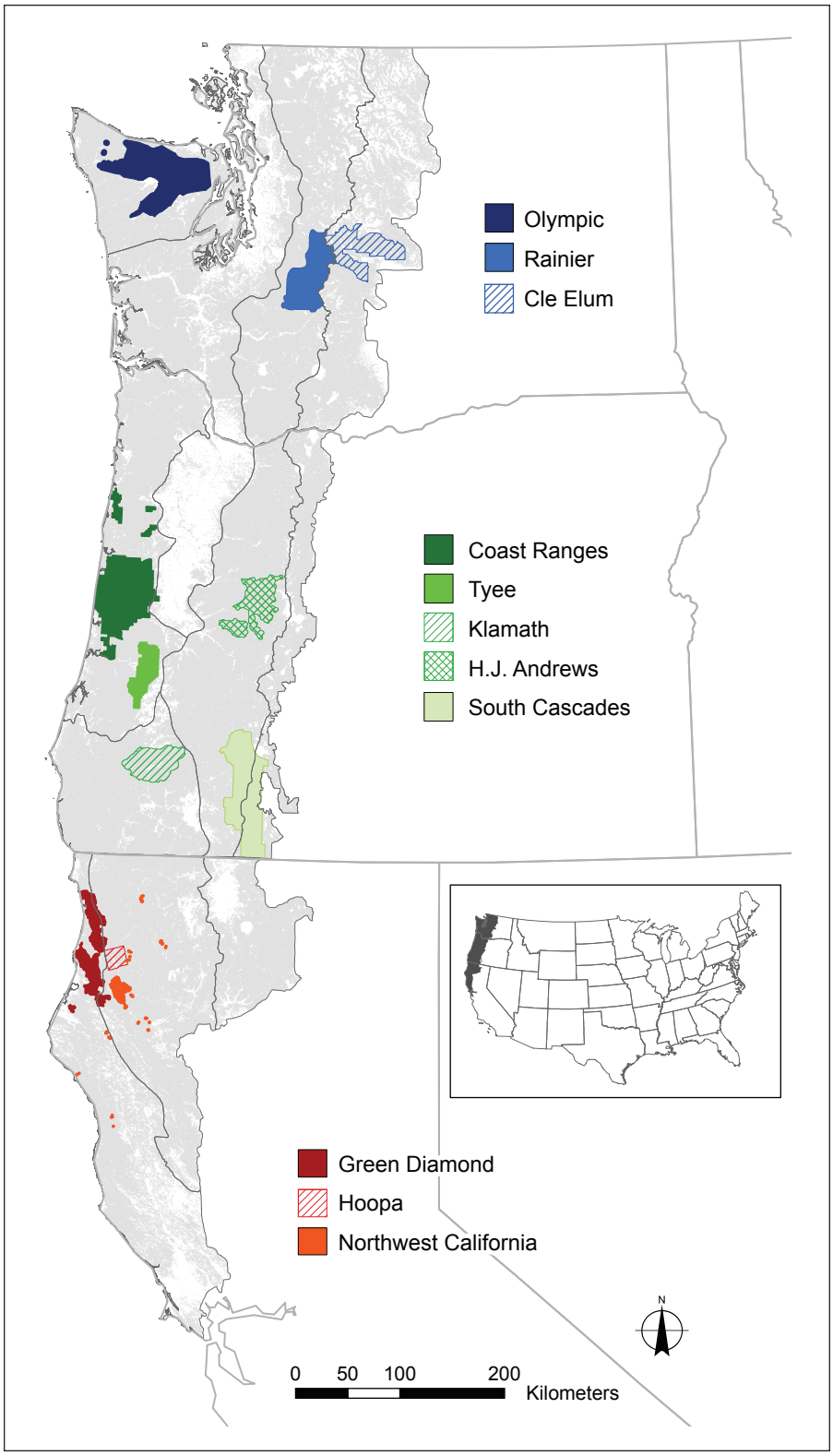



Figure 4-1—Locations of 11 study areas used in the analysis of vital rates and population trends of northern spotted owls, 1985 to 2013 (Dugger et al. 2016).

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).

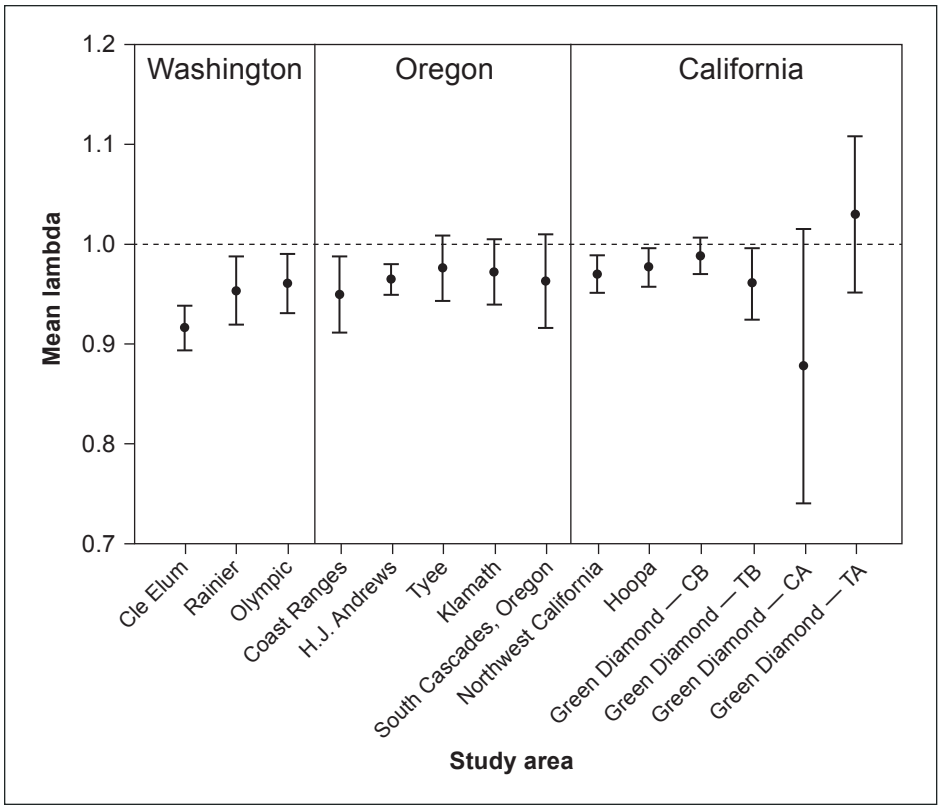



Figure 4-2—Estimated mean rates of population change (mean lambda) and 95 percent confidence limits for spotted owls from 1985 to 2013 at 11 sites: Cle Elum, Rainier, and Olympic, Washington; Coast Range, H.J. Andrews, and Tyee, Klamath, and South Cascades, Oregon; and northwest California, Hoopa, and 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

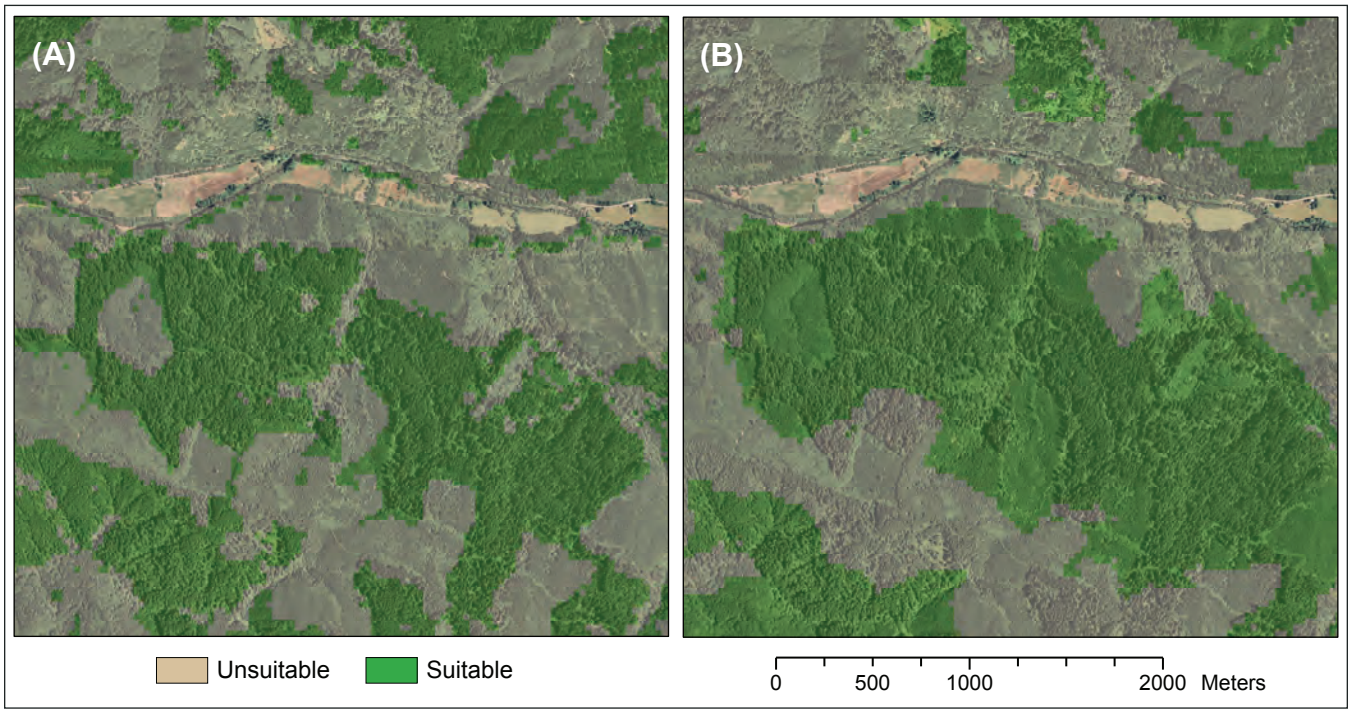



Figure 4-3—Examples of the suitable forest cover at (A) the stand scale developed by the Northwest Forest Plan monitoring program (Davis et al. 2011), and (B) the 200-ha (~250-foot radius) core-area scale used for modeling and delineating critical habitat (USFWS 2012a).

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

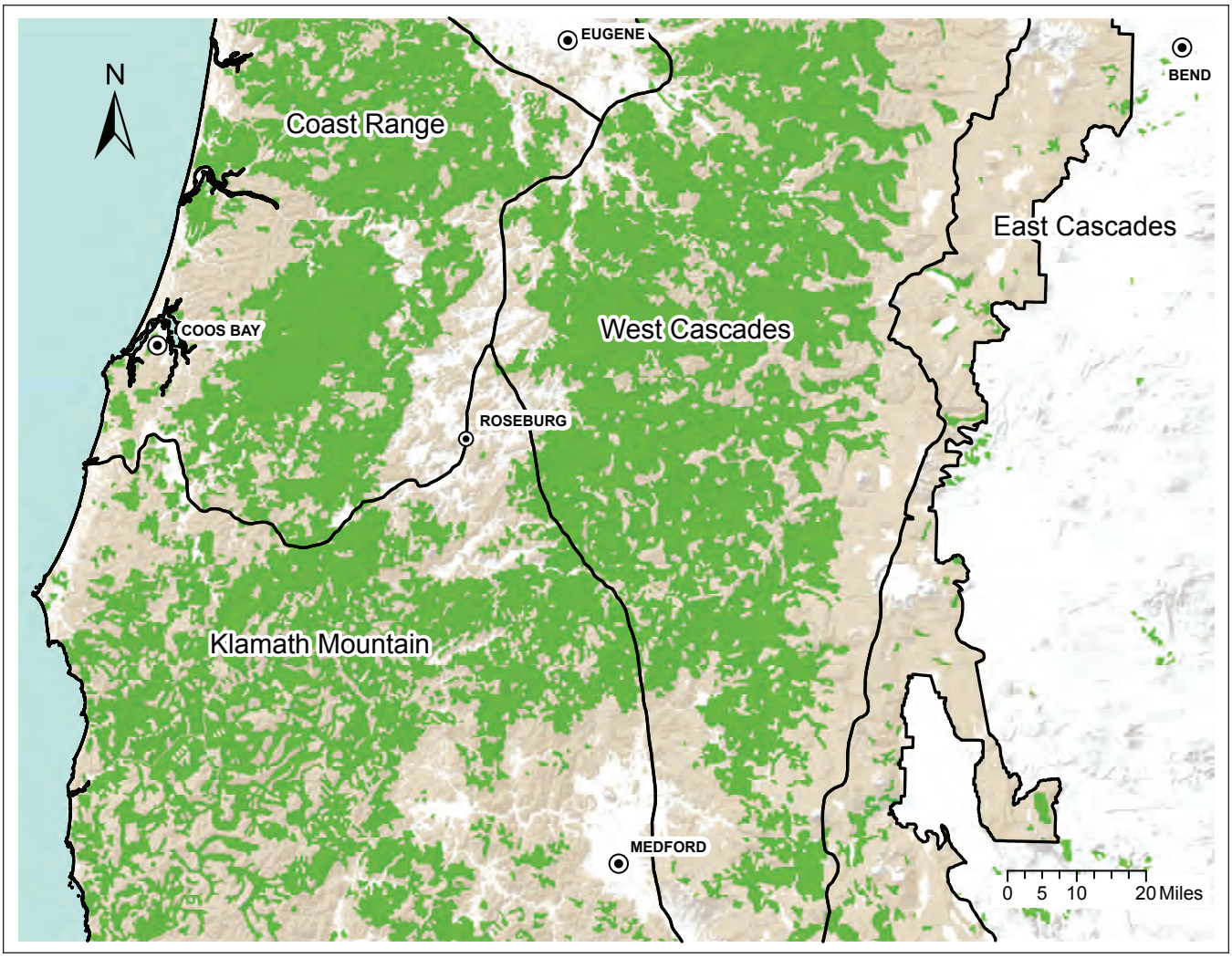



Figure 4-4—Differing historical patterns of old-growth Douglas-fir and mixed-conifer forest (green shaded areas) in west-central Oregon (Andrews and Cowlin 1940) within four areas with different fire regimes (Coast Range, infrequent—high severity; West Cascades, moderately frequent—mixed severity; Klamath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

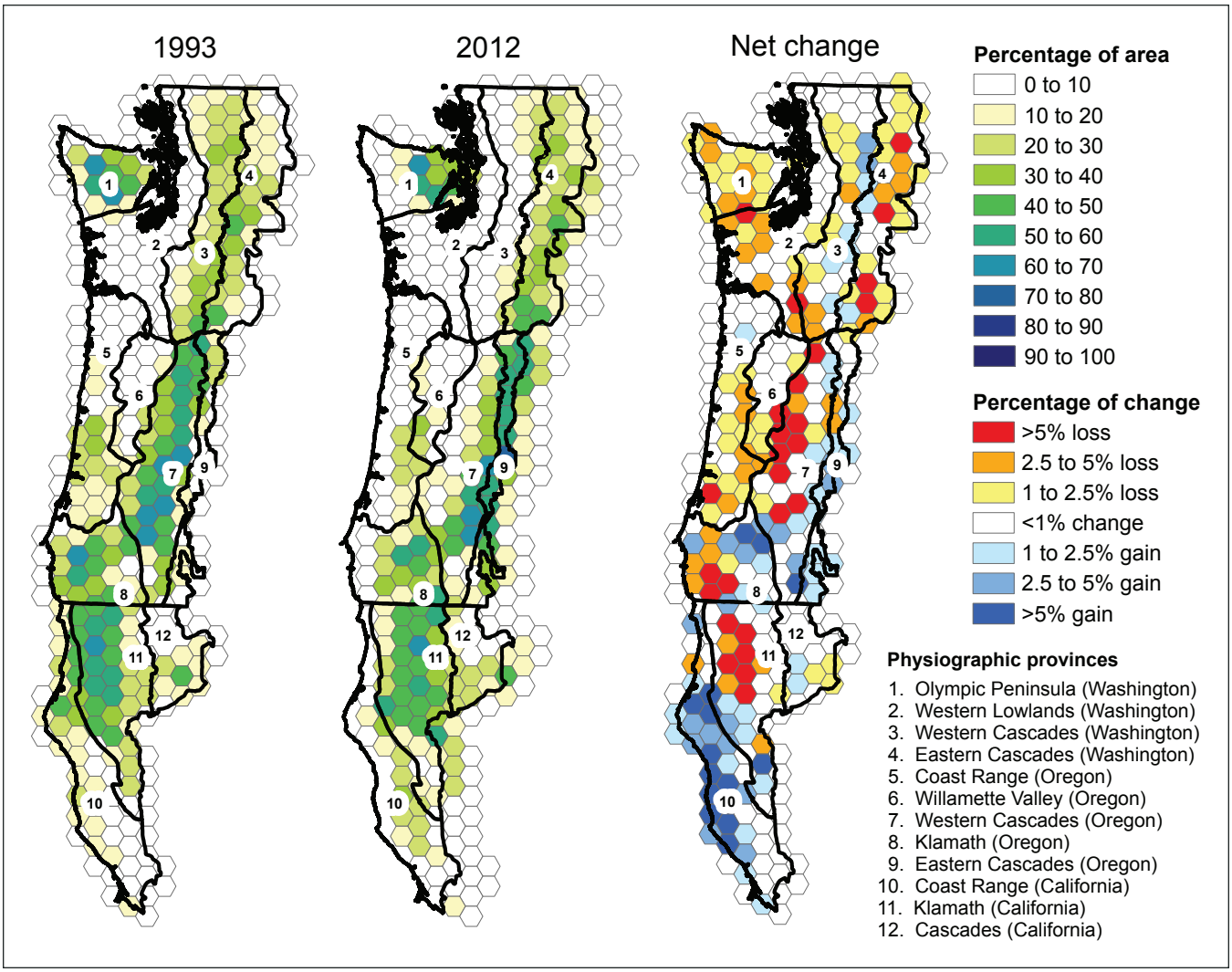



Figure 4-5—Loss and recruitment of forest types suitable for nesting and roosting by spotted owls in 1993 versus 2012, based on change in percentage of cover estimated within 18.6 mi (30 km) (center to center) hexagons across the Northwest Forest Plan area.

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

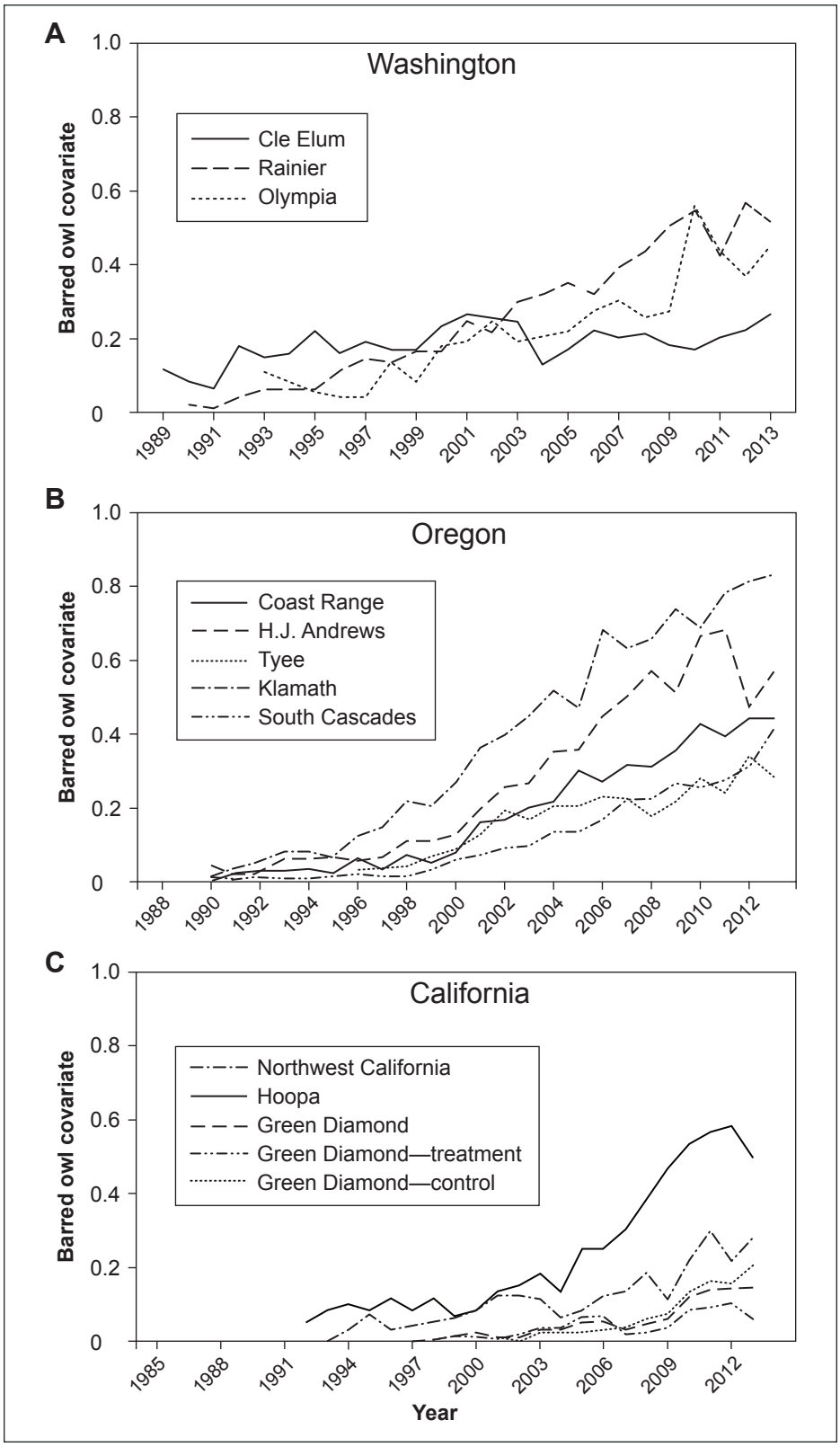



Figure 4-6—Annual increase in the proportion of spotted owl territories with detections of barred owls at (A) three sites in Washington, (B) five sites in Oregon, and (C) three sites in California from 1985 to 2013 (Dugger et al. 2016). Estimates for Green Diamond are presented separately for control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

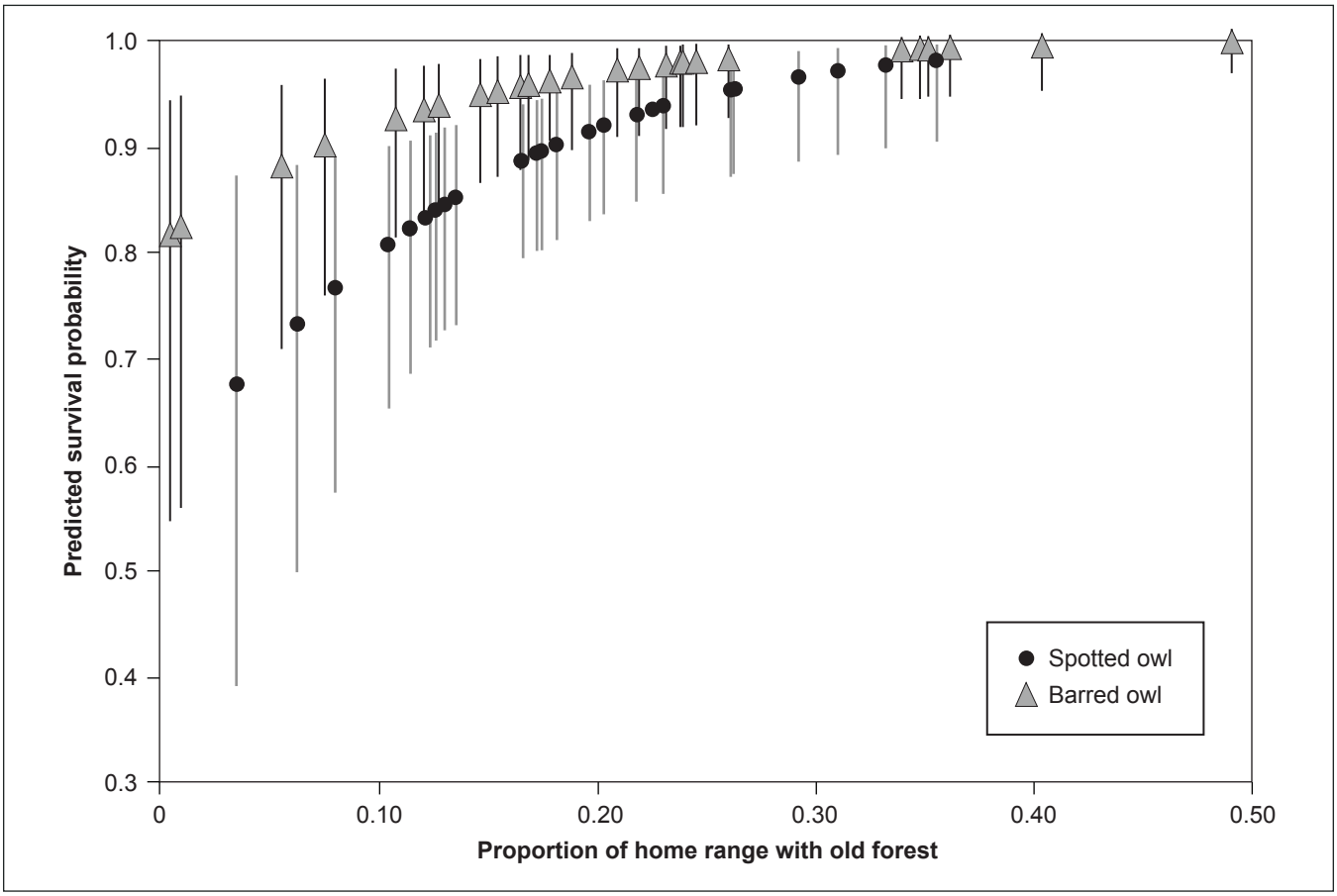



Figure 4-7—Survival (with 95-percent confidence limits) of individual adult barred owls and spotted owls increased with increasing amount of old forest (≥120 years) conifer forest within their home ranges in the Oregon Coast Range (Wiens et al. 2014).

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

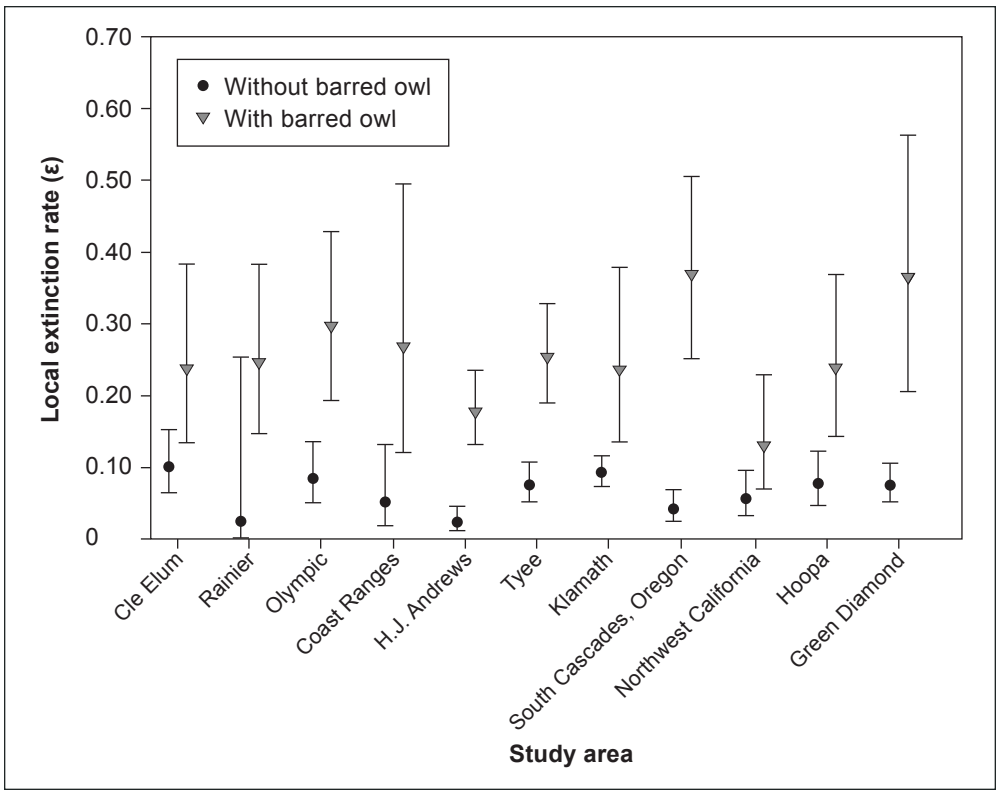



Figure 4-8—Mean annual local extinction rates (with 95 percent confidence limits) for northern spotted owls on 11 study areas in Washington, Oregon, and California relative to with (gray triangles) and without (black circles) the presence of barred owls (Dugger et al. 2016).

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.

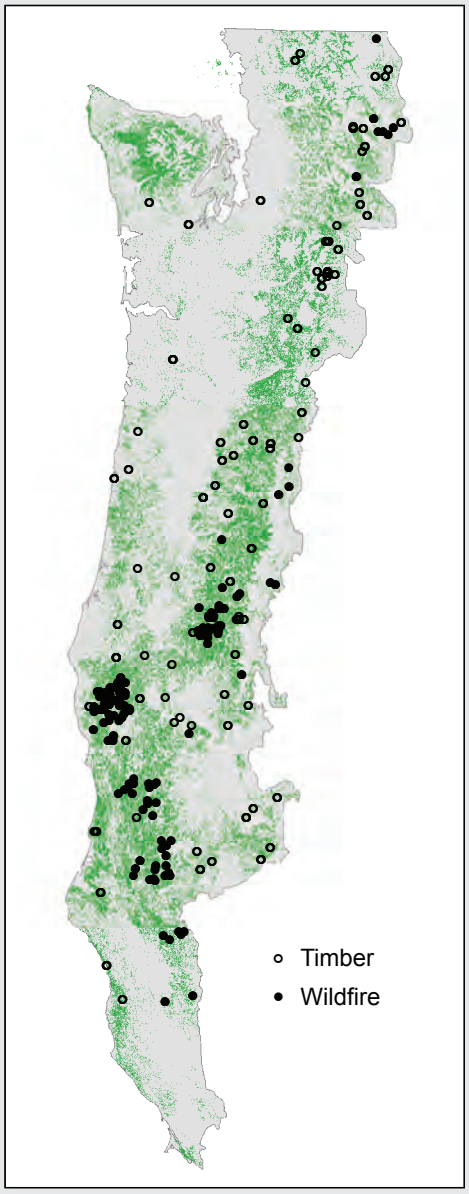
Abundance and distribution of primary prey species can influence space use by spotted owls. For example, spotted owls more frequently use riparian areas within their home ranges (Wiens et al. 2014), perhaps because the cool microclimates associated with stream drainages may be favorable for thermoregulatory 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.

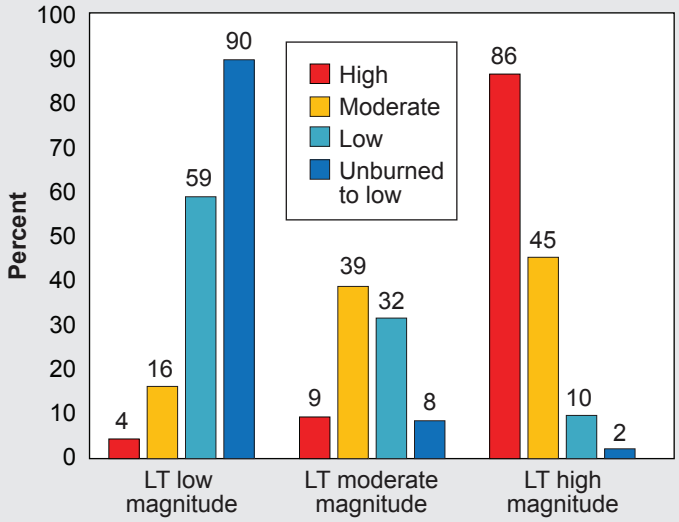
Plots used in this analysis occurred in mapped suitable nesting/roosting cover type in 1993 that experienced a disturbance between 1994 and 2012 from either timber harvesting or wildfire, which occurred between the initial plot measurement and remeasurement dates.

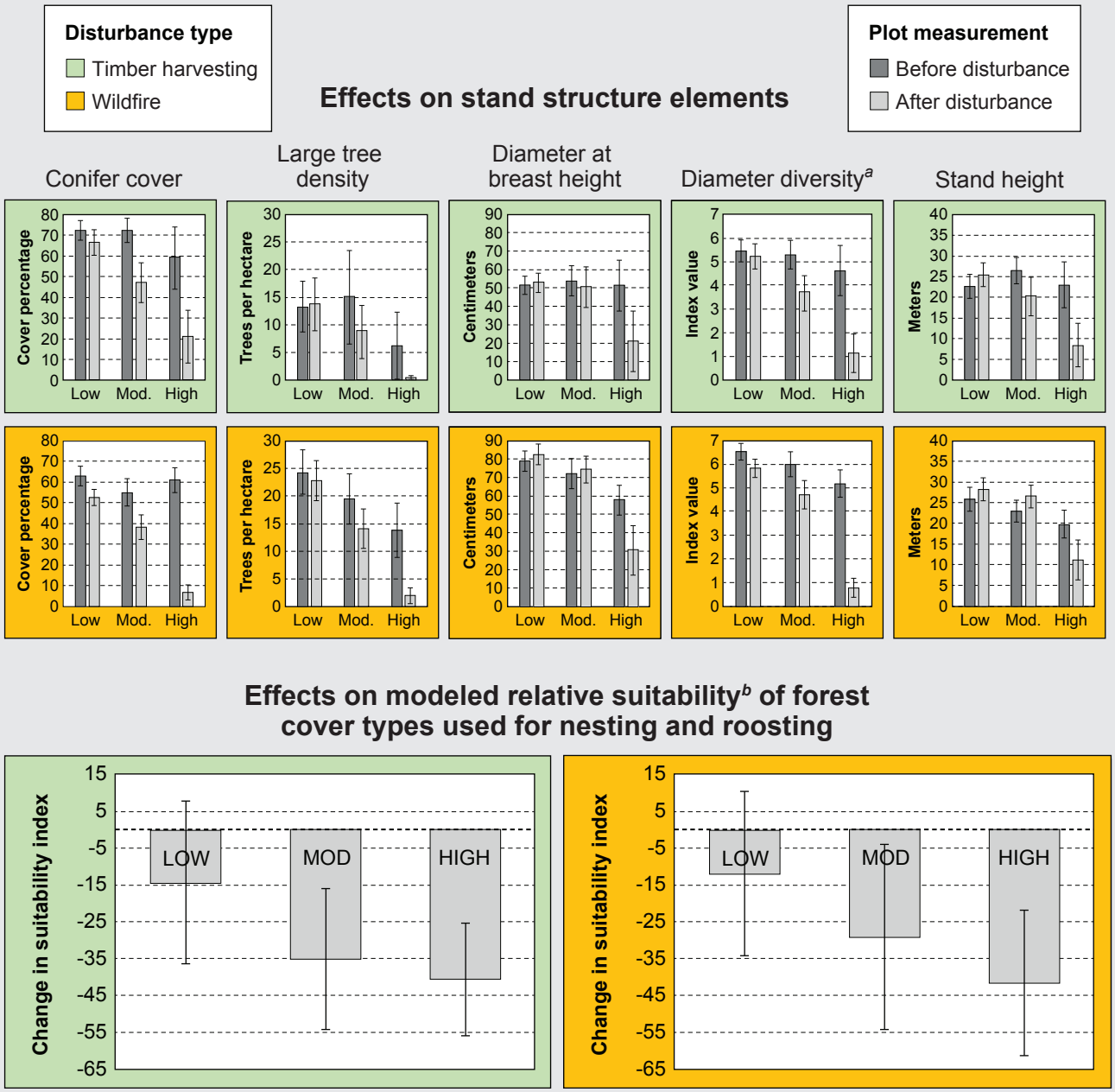
Changes in the mapped nesting/roosting relative suitability index were also analyzed by differencing the 2012 and 1993 relative suitability maps.

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

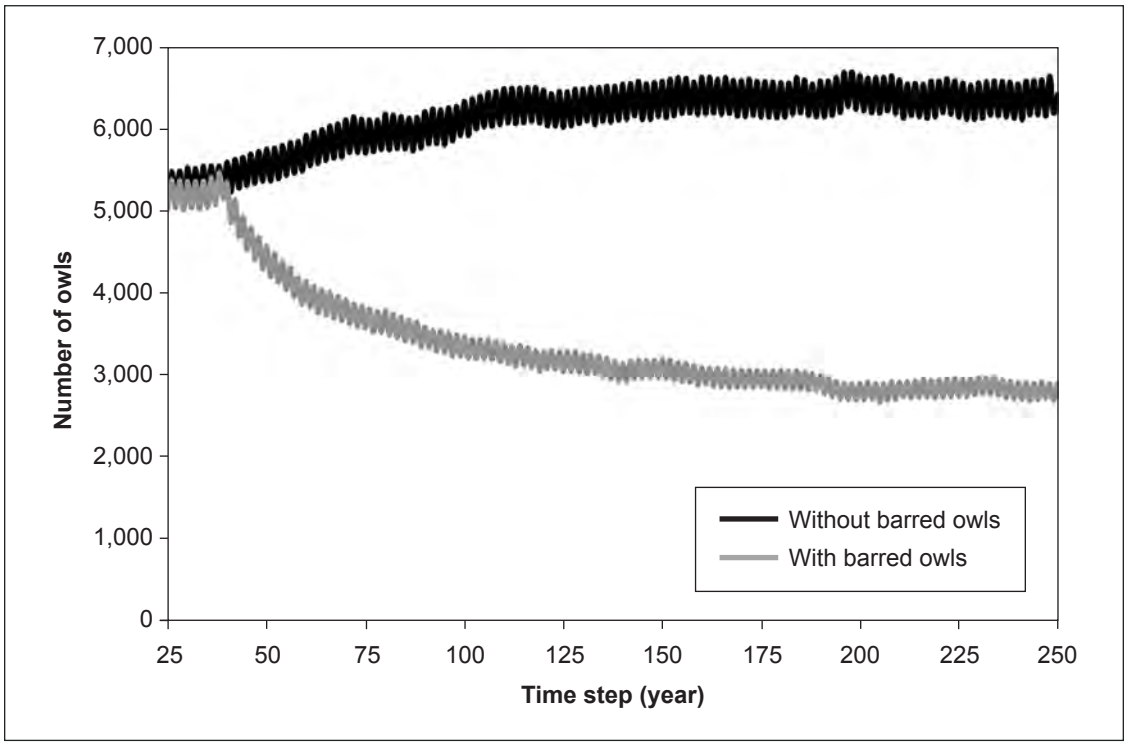



Figure 4-9—HexSim model runs with five replicates each for without barred owl impacts and with barred owl impacts for the spotted owl's geographic range in the United States. The apparent within-year variation that appears in the figure is a function of an "even-odd" year effect on reproduction (USFWS 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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## Literature Cited

**Acker, J. 2012.** Recent trends in western screech-owl and barred owl abundances on Bainbridge Island, Washington. Northwestern Naturalist. 93(2): 133–137. doi:10.1898/nwn11-21.1.

**Ackers, S.H.; Davis, R.J.; Olsen, K.A.; Dugger, K.M. 2015.** The evolution of mapping habitat for northern spotted owls (*Strix occidentalis caurina*): a comparison of photo-interpreted, Landsat-based, and LiDAR-based habitat maps. Remote Sensing of Environment. 156: 361–373. doi:10.1016/j.rse.2014.09.025.

**Agee, J.K. 1993.** Fire ecology of Pacific Northwest forests. Washington, DC: Island Press. 505 p.

**Agee, J.K.; Skinner, C.N. 2005.** Basic principles of forest fuel reduction treatments. Forest Ecology and Management. 211(1–2): 83–96. doi:10.1016/j.foreco.2005.01.034.

**Ager, A.A.; Finney, M.A.; Kerns, B.K.; Maffei, H. 2007.** Modeling wildfire risk to northern spotted owl (*Strix occidentalis caurina*) habitat in central Oregon, USA. Forest Ecology and Management. 246(1): 45–56.

**Albert, C.A.; Wilson, L.K.; Mineau, P.; Trudeau, S.; Elliott, J.E. 2010.** Anticoagulant rodenticides in three owl species from western Canada, 1988–2003. Archives of Environmental Contamination and Toxicology. 58(2): 451–459. doi:10.1007/s00244-009-9402-z.

**Anderson, P.D.; Ronnenberg, K.L., eds. 2013.** Density management in the 21st century: west side story. Gen. Tech. Rep. PNW-GTR-880. Portland, OR: U.S. Department of Agriculture, Forest Service, Pacific Northwest Research Station. 249 p.

**Andrews, H.J.; Cowlin, R.W. 1940.** Forest resources of the Douglas-fir region. Misc. Publ. 389. Portland, OR: U.S. Department of Agriculture, Forest Service. 200 p.

**Andrews, L.S.; Perkins, J.P.; Thrailkill, J.A.; Poage, N.J.; Tappeiner, J.C. 2005.** Silvicultural approaches to develop northern spotted owl nesting sites, central coast ranges, Oregon. Western Journal of Applied Forestry. 20(1): 13–27.

**Angert, A.L.; LaDeau, S.L.; Ostfeld, R.S. 2013.** Climate change and species interactions: ways forward. Annals of the New York Academy of Sciences. 1297: 1–7. doi: 10.1111/nyas.12286.

**Anthony, R.G.; Forsman, E.D.; Franklin, A.B.; Anderson, D.R.; Burnham, K.P.; White, G.C.; Schwarz, C.J.; Nichols, J.D.; Hines, J.E.; Olson, G.S.; Ackers, S.H.; Andrews, L.S.; Biswell, B.L.; Carlson, P.C.; Diller, L.V.; Dugger, K.M.; Fehring, K.E.; Fleming, T.L.; Gerhardt, R.P.; Gremel, S.A.; Gutierrez, R.J.; Happe, P.J.; Herter, D.R.; Higley, J.M.; Horn, R.B.; Irwin, L.L.; Loschl, P.J.; Reid, J.A.; Sovern, S.G. 2006.** Status and trends in demography of northern spotted owls, 1985–2003. Wildlife Monographs. 163(1): 1–48.

**Anthony, R.G.; O'Connell, M.A.; Pollock, M.M.; Hallett, J.G. 2003.** Associations of mammals with riparian ecosystems in Pacific Northwest forests. In: Zabel, C.J.; Anthony, R.G., eds. Mammal community dynamics: management and conservation in the coniferous forests of western North America. Cambridge, United Kingdom: Cambridge University Press: 510–566. Chapter 15.

**Aubry, K.B.; Halpern, C.B.; Peterson, C.E. 2009.** Variable-retention harvests in the Pacific Northwest: a review of short-term findings from the demo study. Forest Ecology and Management. 258(4): 398–408. doi:10.1016/j.foreco.2009.03.013.

**Avery, D.M.; Avery, G.; Palmer, N.G. 2005.** Micromammalian distribution and abundance in the Western Cape Province, South Africa, as evidenced by barn owls *Tyto alba* (scopoli). Journal of Natural History. 39(22): 2047–2071. doi:10.1080/00222930500044631.

**Bailey, L.L.; Reid, J.A.; Forsman, E.D.; Nichols, J.D. 2009.** Modeling co-occurrence of northern spotted and barred owls: accounting for detection probability differences. Biological Conservation. 142(12): 2983–2989. doi:10.1016/j.biocon.2009.07.028.

**Baker, W.L. 2015.** Historical northern spotted owl habitat and old-growth dry forests maintained by mixed-severity wildfires. Landscape Ecology. 30(4): 655–666. doi:10.1007/s10980-014-0144-6.

**Bamford, M.J.; Calver, M.C. 2014.** A precise definition of habitat is needed for effective conservation and communication. Australian Zoologist. 37(2): 245–247.

**Barrowclough, G.F.; Groth, J.G.; Mertz, L.A.; Gutiérrez, R.J. 2005.** Genetic structure, introgression, and a narrow hybrid zone between northern and California spotted owls (*Strix occidentalis*). Molecular Ecology. 14(4): 1109–1120.

**Barrowclough, G.F.; Gutierrez, R.J.; Groth, J.G.; Lai, J.E.; Rock, D.F. 2011.** The hybrid zone between northern and California spotted owls in the Cascade-Sierran suture zone. Condor. 113(3): 581–589.

**Barrows, C. 1980.** Feeding ecology of the spotted owl in California. Journal of Raptor Research. 12(3): 73–78.

**Barrows, C.W. 1981.** Roost selection by spotted owls: an adaptation to heat stress. Condor. 83: 302–309.

**Bell, D.M.; Gregory, M.J.; Roberts, H.M.; Davis, R.J.; Ohmann, J.L. 2015.** How sampling and scale limit accuracy assessment of vegetation maps: a comment on Loehle et al. (2015). Forest Ecology and Management. 358: 361–364. doi:10.1016/j.foreco.2015.07.017.

**Bevis, K.R.; Richards, J.E.; King, G.M.; Hanson, E.E. 1997.** Food habits of the northern spotted owl (*Strix occidentalis caurina*) at six nest sites in Washington's east Cascades. In: Duncan, J.R.; Johnson, D.H.; Nicholls, T.H., eds. Biology and conservation of owls of the northern hemisphere: second international symposium. Gen. Tech. Rep. GTR-NC-190. St. Paul, MN: U.S. Department of Agriculture, Forest Service, North Central Forest Experiment Station: 68–73.

**Bodine, E.N.; Capaldi, A. 2017.** Can culling barred owls save a declining northern spotted owl population? Natural Resource Modeling. e12131. doi:10.1111/nrm.12131.

**Bond, M.L. 2016.** The heat is on: spotted owls and wildfire. Reference Module in Earth Systems and Environmental Sciences: 1–12. doi:10.1016/B978-0-12-409548-9.10014-4.

**Bond, M.L.; Bradley, C.; Lee, D.E. 2016.** Foraging habitat selection by California spotted owls after fire. Journal of Wildlife Management. doi:10.1002/jwmg.21112.

**Bond, M.L.; Lee, D.E.; Siegel, R.B.; Ward, J.P. 2009.** Habitat use and selection by California spotted owls in a postfire landscape. Journal of Wildlife Management. 73(7): 1116–1124. doi:10.2193/2008-248.

**Bortolotti, G.R.; Marchant, T.; Blas, J.; Cabezas, S. 2009.** Tracking stress: localisation, deposition and stability of corticosterone in feathers. Journal of Experimental Biology. 212(Pt 10): 1477–1482. doi:10.1242/jeb.022152.

**Buchanan, J.B.; Fleming, T.L.; Irwin, L.L. 2004.** A comparison of barred and spotted owl nest-site characteristics in the eastern Cascade Mountains, Washington. Journal of Raptor Research. 38(3): 231–237.

**Buchanan, J.B.; Gutiérrez, R.J.; Anthony, R.G.; Cullinan, T.; Diller, L.V.; Forsman, E.D.; Franklin, A.B. 2007.** A synopsis of suggested approaches to address potential competitive interactions between barred owls (*Strix varia*) and spotted owls (*S. occidentalis*). Biological Invasions. 9(6): 679–691. doi:10.1007/s10530-006-9068-7.

**Burnham, K.P.; Anderson, D.R.; White, G.C. 1996.** Meta-analysis of vital rates of the northern spotted owl. In: Forsman, E.D.; DeStefano, S.; Raphael, M.G.; Gutierrez, R.J., eds. Demography of the northern spotted owl. Studies in Avian Biology No. 17. Lawrence, KS: Cooper Ornithological Society: 92–101. Vol. 17.

**Carey, A.B. 2000.** Effects of new forest management strategies on squirrel populations. Ecological Applications. 10(1): 248–257.

**Carey, A.B.; Horton, S.P.; Biswell, B.L. 1992.** Northern spotted owls: influence of prey base and landscape character. Ecological Monographs. 62(2): 223–250.

**Carey, A.B.; Johnson, M.L. 1995.** Small mammals in managed, naturally young, and old-growth forests. Ecological Applications. 5(2): 336–352.

**Carey, A.B.; Maguire, C.C.; Biswell, B.L.; Wilson, T.M. 1999.** Distribution and abundance of neotoma in western Oregon and Washington. Northwest Science. 73(2): 65–80.

**Carey, A.B.; Reid, J.A.; Horton, S.P. 1990.** Spotted owl home range and habitat use in southern Oregon Coast Ranges. Journal of Wildlife Management. 54(1): 11–17.

**Carey, M.P.; Sanderson, B.L.; Barnas, K.A.; Olden, J.D. 2012.** Native invaders—challenges for science, management, policy, and society. Frontiers in Ecology and the Environment. 10(7): 373–381. doi:10.1890/110060.

**Carrete, M.; Bortolotti, G.R.; Sánchez-Zapata, J.A.; Delgado, A.; Cortés-Avizanda, A.; Grande, J.M.; Donázar, J.A. 2013.** Stressful conditions experienced by endangered Egyptian vultures on African wintering areas. Animal Conservation. 16(3): 353–358. doi:10.1111/acv.12001.

**Carroll, C. 2010.** Role of climatic niche models in focal-species-based conservation planning: assessing potential effects of climate change on northern spotted owl in the Pacific Northwest, USA. Biological Conservation. 143(6): 1432–1437.

**Carroll, C.; Dunk, J.R.; Moilanen, A. 2010.** Optimizing resiliency of reserve networks to climate change: multispecies conservation planning in the Pacific Northwest, USA. Global Change Biology. 16(3): 891–904.

**Carroll, C.; Johnson, D.S. 2008.** The importance of being spatial (and reserved): assessing northern spotted owl habitat relationships with hierarchical bayesian models. Conservation Biology. 22(4): 1026–1036.

**Churchill, D.J.; Larson, A.J.; Dahlgreen, M.C.; Franklin, J.F.; Hessburg, P.F.; Lutz, J.A. 2013.** Restoring forest resilience: from reference spatial patterns to silvicultural prescriptions and monitoring. Forest Ecology and Management. 291: 442–457. doi:10.1016/j.foreco.2012.11.007.

**Clark, D.A.; Anthony, R.G.; Andrews, L.S. 2011.** Survival rates of northern spotted owls in post-fire landscapes of southwest Oregon. Journal of Raptor Research. 45(1): 38–47.

**Clark, D.A.; Anthony, R.G.; Andrews, L.S. 2013.** Relationship between wildfire, salvage logging, and occupancy of nesting territories by northern spotted owls. Journal of Wildlife Management. 77(4): 672–688. doi:10.1002/jwmg.523.

**Cochrane, M.A.; Moran, C.J.; Wimberly, M.C.; Baer, A.D.; Finney, M.A.; Beckendorf, K.L.; Eidenshink, J.; Zhu, Z. 2012.** Estimation of wildfire size and risk changes due to fuels treatments. International Journal of Wildland Fire. 21(4): 357. doi:10.1071/wf11079.

**Comfort, E.J.; Clark, D.A.; Anthony, R.G.; Bailey, J.; Betts, M.G. 2016.** Quantifying edges as gradients at multiple scales improves habitat selection models for northern spotted owl. Landscape Ecology. 31: 1227–1240. doi:10.1007/s10980-015-0330-1.

**Courtney, S.P.; Blakesley, J.A.; Bigley, R.E.; Cody, M.L.; Dumbacher, J.P.; Fleishcher, R.C.; Franklin, A.B.; Franklin, J.F.; Gutiérrez, R.J.; Marzluff, J.M.; Sztukowski, L. 2004.** Scientific evaluation of the status of the northern spotted owl. Portland, OR: Sustainable Ecosystems Institute. 498 p.

**Crespin, L.; Verhagen, R.; Stenseth, N.C.; Yoccoz, N.G.; Prevot-Julliard, A.; Lebreton, J. 2002.** Survival in fluctuating bank vole populations: seasonal and yearly variations. Oikos. 98: 467–479.

**Crozier, M.L.; Seamans, M.E.; Gutierrez, R.J.; Loschl, P.J.; Horn, R.B.; Sovern, S.G.; Forsman, E.D. 2006.** Does the presence of barred owls suppress the calling behavior of spotted owls? Condor. 108(4): 760–769.

**Cutler, T.L.; Hays, D.W. 1991.** Food habits of northern spotted owls in high elevation forests of pelican butte, southwest Oregon. Northwestern Naturalist. 72: 66–69.

**Dale, V.H.; Joyce, L.A.; McNulty, S.; Neilson, R.P.; Ayres, M.P.; Flannigan, M.D.; Hanson, P.J.; Irland, L.C.; Lugo, A.E.; Peterson, C.J.; Simberloff, D.; Swanson, F.J.; Stocks, B.J.; Michael Wotton, B. 2001.** Climate change and forest disturbances. BioScience. 51(9): 723. doi:10.1641/0006-3568(2001)051[0723:ccafd]2.0.co;2.

**Dark, S.J.; Gutierrez, R.J.; Gould, G.I., Jr. 1998.** The barred owl (*Strix varia*) invasion in California. Auk. 115(1): 50–56.

**Davis, R.J.; Dugger, K.M.; Mohoric, S.; Evers, L.; Aney, W.C. 2011.** Northwest forest plan—the first 15 years (1994–2008): status and trends of northern spotted owl populations and habitat. Gen. Tech. Rep. PNW-GTR-850. Portland, OR: U.S. Department of Agriculture, Forest Service, Pacific Northwest Research Station. 147 p.

**Davis, R.J.; Hollen, B.; Hobson, J.; Gower, J.E.; Keenum, D. 2016.** Northwest Forest Plan—the first 20 years (1994–2013): status and trends of northern spotted owl habitats. Gen. Tech. Rep. PNW-GTR-929. Portland, OR: U.S. Department of Agriculture, Forest Service, Pacific Northwest Research Station. 111 p.

**Davis, R.J.; Lint, J. 2005.** Habitat status and trends. In: Lint, J., ed. Northwest Forest Plan—the first 10 years (1994–2003): status and trends of northern spotted owl populations and habitat. Gen. Tech. Rep. PNW-GTR-648. Portland, OR: U.S. Department of Agriculture, Forest Service, Pacific Northwest Research Station: 21–82. Chapter 3.

**DellaSala, D.A.; Anthony, R.G.; Bond, M.L.; Fernandes, E.S.; Frissell, C.A.; Hanson, C.T.; Spivak, R. 2013.** Alternative views of a restoration framework for federal forests in the Pacific Northwest. Journal of Forestry. 111(6): 420–429.

**Dennison, P.E.; Brewer, S.C.; Arnold, J.D.; Moritz, M.A. 2014.** Large wildfire trends in the Western United States, 1984–2011. Geophysical Research Letters. 41(8): 2928–2933. doi:10.1002/2014gl059576.

**Devaney, E.E. 2008.** Report of investigation—the endangered species act and the conflict between science and policy. Washington, DC: U.S. Department of the Interior, Office of Inspector General. 141 p.

**Diller, L.; Hamm, K.; Lamphear, D.; McDonald, T. 2012.** Two decades of research and monitoring of the northern spotted owl on private timberlands in the redwood region: what do we know and what challenges remain? In: Standiford, R.B.; Weller, T.J.; Piirto, D.D.; Stuart, J.D., eds. Proceedings of the coast redwood forests in a changing California: a symposium for scientists and managers. Gen. Tech. Rep. PSW-GTR-238. Albany, CA: U.S. Department of Agriculture, Forest Service, Pacific Southwest Research Station: 399–408.

**Diller, L.V.; Hamm, K.A.; Early, D.A.; Lamphear, D.W.; Dugger, K.M.; Yackulic, C.B.; Schwartz, C.J.; Carlson, P.C.; McDonald, T.L. 2016.** Demographic responses of northern spotted owls to barred owl removal. Journal of Wildlife Management. 80(4): 691–707. doi:10.1002/jwmg.1046.

**Dugger, K.M.; Anthony, R.G.; Andrews, L.S. 2011.** Transient dynamics of invasive competition: barred owls, spotted owls, habitat, and the demons of competition present. Ecological Applications. 21(7): 2459–2468.

**Dugger, K.M.; Forsman, E.D.; Franklin, A.B.; Davis, R.J.; White, G.C.; Schwarz, C.J.; Burnham, K.P.; Nichols, J.D.; Hines, J.E.; Yackulic, C.B.; Doherty Jr., P.F.; Bailey, L.L.; Clark, D.A.; Ackers, S.H.; Andrews, L.S.; Augustine, B.; Biswell, B.L.; Blakesley, J.A.; Carlson, P.C.; Clement, M.J.; Diller, L.V.; Glenn, E.M.; Green, A.; Gremel, S.A.; Herter, D.R.; Higley, J.M.; Hobson, J.; Horn, R.B.; Huyvaert, K.P.; McCafferty, C.; McDonald, T.L.; McDonnell, K.; Olson, G.S.; Reid, J.A.; Rockweit, J.; Ruiz, V.; Saenz, J.; Sovern, S.G. 2016.** The effects of habitat, climate and barred owls on the long-term population demographics of northern spotted owls. Condor. 118: 57–116. doi:10.1650/CONDOR-15-24.1.

**Dugger, K.M.; Wagner, F.; Anthony, R.G.; Olson, G.S. 2005.** The relationship between habitat characteristics and demographic performance of northern spotted owls in southern Oregon. Condor. 107(4): 863–878.

**Dunbar, D.L.; Booth, B.P.; Forsman, E.D.; Hetherington, A.E.; Wilson, D.J. 1991.** Status of the spotted owl, *Strix occidentalis*, and barred owl, *Strix varia*, in southwestern British Columbia. Canadian Field-Naturalist. 105(4): 464–468.

**Dunk, J.R.; Hawley, J.J.V.G. 2009.** Red-tree vole habitat suitability modeling: implications for conservation and management. Forest Ecology and Management. 258(5): 626–634. doi:10.1016/j.foreco.2009.04.041.

**Dunk, J.R.; Woodbridge, B.; Glenn, E.M.; Davis, R.J.; Fitzgerald, K.; Henson, P.; LaPlante, D.W.; Marcot, B.G.; Noon, B.R.; Raphael, M.G.; Schumaker, N.H.; White, B. 2015.** The scientific basis for modeling northern spotted owl habitat: a response to Loehle, Irwin, Manly, and Merrill. Forest Ecology and Management. 358: 355–360. doi:10.1016/j.foreco.2015.06.035.

**Elliot, K. 2006.** Declining numbers of western screech-owl in the lower mainland of British Columbia. British Columbia Birds. 14: 2–11.

**Farber, S.L.; Kroll, A.J. 2012.** Site occupancy dynamics of northern spotted owls in managed interior Douglas-fir forests, California, USA, 1995–2009. Journal of Wildlife Management. 76(6): 1145–1152. doi:10.1002/jwmg.368.

**Forest Ecosystem Management Assessment Team [FEMAT]. 1993.** Forest ecosystem management: an ecological, economic, and social assessment. Portland, OR: U.S. Department of Agriculture; U.S. Department of the Interior [and others]. [Irregular pagination].

**Forsman, E.D.; Anthony, R.G.; Dugger, K.M.; Glenn, E.M.; Franklin, A.B.; White, G.C.; Schwartz, C.J.; Burnham, K.P.; Anderson, D.R.; Nichols, J.D.; Hines, J.E.; Lint, J.B.; Davis, R.J.; Ackers, S.H.; Andrews, L.S.; Biswell, B.L.; Carlson, P.C.; Diller, L.V.; Gremel, S.A.; Herter, D.R.; Higley, J.M.; Horn, R.B.; Reid, J.A.; Rockweit, J.; Schaberl, J.P.; Snetsinger, T.J.; Sovern, S.G. 2011.** Population demography of northern spotted owls. Studies in Avian Biology. 40: 1–106.

**Forsman, E.D.; Anthony, R.G.; Meslow, E.C.; Zabel, C.J. 2004.** Diets and foraging behavior of northern spotted owls in Oregon. Journal of Raptor Research. 38(3): 214–230.

**Forsman, E.D.; Anthony, R.G.; Reid, J.A.; Loschl, P.J.; Sovern, S.G.; Taylor, M.; Biswell, B.L.; Ellingson, A.; Meslow, E.C.; Miller, G.S.; Swindle, K.A.; Thrailkill, J.A.; Wagner, F.F.; Seaman, D.E. 2002.** Natal and breeding dispersal of northern spotted owls. Wildlife Monographs. 149: 1–35.

**Forsman, E.D.; Kaminski, T.J.; Lewis, J.C.; Maurice, K.J.; Sovern, S.G.; Ferland, C.; Glenn, E.M. 2005.** Home range and habitat use of northern spotted owls on the Olympic Peninsula, Washington. Journal of Raptor Research. 39(4): 365–377.

**Forsman, E.D.; Meslow, E.C.; Wight, H.M. 1984.** Distribution and biology of the spotted owl in Oregon. Wildlife Monographs. 48(2): 1–64.

**Forsman, E.D.; Otto, I.A.; Sovern, S.C.; Taylor, M.; Hays, D.W.; Allen, H.; Roberts, S.L.; Seaman, D.E. 2001.** Spatial and temporal variation in diets of spotted owls in Washington. Journal of Raptor Research. 35(2): 141–150.

**Forsman, E.D.; Sovern, S.G.; Taylor, M.; Biswell, B.L. 2015.** Home range and habitat selection by northern spotted owls on the eastern slope of the Cascade mountains, Washington. Journal of Raptor Research. 49(2): 109–128.

**Franklin, A.B.; Anderson, D.R.; Forsman, E.D.; Burnham, K.P.; Wagner, F.W. 1996.** Methods for collecting and analyzing demographic data on the northern spotted owl. Studies in Avian Biology. 17: 12–20.

**Franklin, A.B.; Anderson, D.R.; Gutierrez, R.J.; Burnham, K.P. 2000.** Climate, habitat quality, and fitness in northern spotted owl populations in northwestern California. Ecological Monographs. 70(4): 539–590.

**Franklin, A.B.; Gutiérrez, R.J. 2002.** Spotted owls, forest fragmentation, and forest heterogeneity. Studies in Avian Biology. 25: 203–220.

**Franklin, A.B.; Nichols, J.D.; Anthony, R.G.; Burnham, K.P.; White, G.C.; Forsman, E.D.; Anderson, D.R. 2006.** Comment on "are survival rates for northern spotted owls biased?" Canadian Journal of Zoology. 84(9): 1375–1379. doi:10.1139/z06-129.

**Franklin, J.F.; Johnson, K.N. 2012.** A restoration framework for federal forests in the Pacific Northwest. Journal of Forestry. 110(8): 429–439. doi:10.5849/jof.10-006.

**Frey, S.J.K.; Hadley, A.S.; Johnson, S.L.; Schulze, M.; Jones, J.A.; Betts, M.G. 2016.** Spatial models reveal the microclimatic buffering capacity of old-growth forests. Science Advances. 2: e1501392.

**Funk, W.C.; Forsman, E.D.; Johnson, M.; Mullins, T.D.; Haig, S.M. 2010.** Evidence for recent population bottlenecks in northern spotted owls (*Strix occidentalis caurina*). Conservation Genetics. 11(3): 1013–1021.

**Funk, W.C.; Forsman, E.D.; Mullins, T.D.; Haig, S.M. 2008.** Introgression and dispersal among spotted owl (*Strix occidentalis*) subspecies. Evolutionary Applications. 1(1): 161–171.

**Funk, W.C.; Mullins, T.D.; Forsman, E.D.; Haig, S.M. 2007.** Microsatellite loci for distinguishing spotted owls (*Strix occidentalis*), barred owls (*Strix varia*), and their hybrids. Molecular Ecology Notes. 7(2): 284–286. doi:10.1111/j.1471-8286.2006.01581.x.

**Gabriel, M.W.; Woods, L.W.; Poppenga, R.; Sweitzer, R.A.; Thompson, C.; Matthews, S.M.; Higley, J.M.; Keller, S.M.; Purcell, K.; Barrett, R.H.; Wengert, G.M.; Sacks, B.N.; Clifford, D.L. 2012.** Anticoagulant rodenticides on our public and community lands: spatial distribution of exposure and poisoning of a rare forest carnivore. PLoS ONE. 7: e40163. doi:10.1371/journal.pone.0040163.

**Gaines, W.L.; Harrod, R.J.; Dickinson, J.; Lyons, A.L.; Halupka, K. 2010.** Integration of northern spotted owl habitat and fuels treatments in the eastern Cascades, Washington, USA. Forest Ecology and Management. 260(11): 2045–2052.

**Ganey, J.L.; Ward, J.P.J.; Willey, D. 2011.** Status and ecology of Mexican spotted owls in the Upper Gila Mountains recovery unit, Arizona and New Mexico. Gen. Tech. Rep. RMRS-GTR-256. Fort Collins, CO: U.S. Department of Agriculture, Forest Service, Rocky Mountain Research Station. 94 p.

**Glenn, E.M.; Anthony, R.G.; Forsman, E.D. 2010.** Population trends in northern spotted owls: associations with climate in the Pacific Northwest. Biological Conservation. 143(11): 2543–2552.

**Glenn, E.M.; Anthony, R.G.; Forsman, E.D.; Olson, G.S. 2011a.** Local weather, regional climate, and annual survival of the northern spotted owl. Condor. 113(1): 150–158. doi:10.1525/cond.2011.100118.

**Glenn, E.M.; Anthony, R.G.; Forsman, E.D.; Olson, G.S. 2011b.** Reproduction of northern spotted owls: the role of local weather and regional climate. Journal of Wildlife Management. 75(6): 1279–1294. doi:10.1002/jwmg.177.

**Glenn, E.M.; Hansen, M.C.; Anthony, R.G. 2004.** Spotted owl home-range and habitat use in young forests of western Oregon. Journal of Wildlife Management. 68(1): 33–50.

**Glenn, E.M.; Lesmeister, D.B.; Davis, R.J.; Hollen, B.; Poopatanapong, A. 2017.** Estimating density of a territorial species in a dynamic landscape. Landscape Ecology. 32: 563–579. doi:10.1007/s10980-016-0467-6.

**Gomez, D.; Anthony, R.G. 1998.** Small mammal abundance in riparian and upland areas of five seral stages in western Oregon. Northwest Science. 72(4): 293–302.

**Graham, S.A. 2012.** Diet composition, niche and geographic characteristics, and prey size preference of barred owls (*Strix varia*) in the Pacific Northwest. Boise, ID: Boise State University. 149 p. M.S. thesis.

**Gutiérrez, R.J.; Barrowclough, G.F. 2005.** Redefining the distributional boundaries of the northern and California spotted owls: implications for conservation. Condor. 107(1): 182–187.

**Gutiérrez, R.J.; Cody, M.; Courtney, S.; Franklin, A.B. 2007.** The invasion of barred owls and its potential effect on the spotted owl: a conservation conundrum. Biological Invasions. 9(2): 181–196. doi:10.1007/s10530-006-9025-5.

**Hagar, J.C. 2007.** Wildlife species associated with non-coniferous vegetation in Pacific Northwest conifer forests: A review. Forest Ecology and Management. 246(1): 108–122. doi:10.1016/j.foreco.2007.03.054.

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

**Hagmann, R.K.; Franklin, J.F.; Johnson, K.N. 2014.** Historical conditions in mixed-conifer forests on the eastern slopes of the northern Oregon Cascade Range, USA. Forest Ecology and Management. 330: 158–170. doi:10.1016/j.foreco.2014.06.044.

**Haig, S.M.; Mullins, T.D.; Forsman, E.D.; Trail, P.W.; Wennerberg, L. 2004.** Genetic identification of spotted owls, barred owls, and their hybrids: legal implications of hybrid identity. Conservation Biology. 18(5): 1347–1357.

**Hall, L.S.; Krausman, P.R.; Morrison, M.L. 1997.** The habitat concept and a plea for standard terminology. Wildlife Society Bulletin. 25(1): 173–182.

**Hamer, T.E.; Forsman, E.D.; Fuchs, A.D.; Walters, M.L. 1994.** Hybridization between barred and spotted owls. Auk. 111(2): 487–492.

**Hamer, T.E.; Forsman, E.D.; Glenn, E.M. 2007.** Home range attributes and habitat selection of barred owls and spotted owls in an area of sympatry. Condor. 109(4): 750–768.

**Hamer, T.E.; Hays, D.L.; Senger, C.M.; Forsman, E.D. 2001.** Diets of northern barred owls and northern spotted owls in an area of sympatry. Journal of Raptor Research. 35(3): 221–227.

**Hanson, C.T.; Odion, D.C.; Dellasala, D.A.; Baker, W.L. 2009.** Overestimation of fire risk in the northern spotted owl recovery plan. Conservation Biology. 23(5): 1314–1319.

**Hanson, C.T.; Odion, D.C.; Dellasala, D.A.; Baker, W.L. 2010.** More-comprehensive recovery actions for northern spotted owls in dry forests: reply to Spies et al. Conservation Biology. 24(1): 334–337. doi:10.1111/j.1523-1739.2009.01415.x.

**Hayward, L.S.; Bowles, A.E.; Ha, J.C.; Wasser, S.K. 2011.** Impacts of acute and long-term vehicle exposure on physiology and reproductive success of the northern spotted owl. Ecosphere. 2(6): 1–20.

**Heithecker, T.D.; Halpern, C.B. 2006.** Variation microclimate associated with dispersed-retention harvests in coniferous forests of western Washington. Forest Ecology and Management. 226(1–3): 60–71.

**Henson, P.; Thrailkill, J.; Glenn, B.; Woodbridge, B.; White, B. 2013.** Using ecological forestry to reconcile spotted owl conservation and forest management. Journal of Forestry. 111(6): 433–437. doi:10.5849/jof.13-072.

**Herter, D.R.; Hicks, L.L. 2000.** Barred owl and spotted owl populations and habitat in the central Cascade range of Washington. Journal of Raptor Research. 34(4): 279–286.

**Hessburg, P.F.; Churchill, D.J.; Larson, A.J.; Haugo, R.D.; Miller, C.; Spies, T.A.; North, M.P.; Povak, N.A.; Belote, R.T.; Singleton, P.H.; Gaines, W.L.; Keane, R.E.; Aplet, G.H.; Stephens, S.L.; Morgan, P.; Bisson, P.A.; Rieman, B.E.; Salter, R.B.; Reeves, G.H. 2015.** Restoring fire-prone inland pacific landscapes: seven core principles. Landscape Ecology. 30(10): 1805–1835. doi:10.1007/s10980-015-0218-0.

**Hessburg, P.F.; Salter, R.B.; James, K.M. 2007.** Re-examining fire severity relations in pre-management era mixed conifer forests: Inferences from landscape patterns of forest structure. Landscape Ecology. 22: 5–24.

**Hessburg, P.F.; Spies, T.A.; Perry, D.A.; Skinner, C.N.; Taylor, A.H.; Brown, P.M.; Stephens, S.L.; Larson, A.J.; Churchill, D.J.; Povak, N.A.; Singleton, P.H.; McComb, B.; Zielinski, W.J.; Collins, B.M.; Salter, R.B.; Keane, J.J.; Franklin, J.F.; Riegel, G. 2016.** Tamm review: management of mixed-severity fire regime forests in Oregon, Washington, and northern California. Forest Ecology and Management. 366: 221–250. doi:10.1016/j.foreco.2016.01.034.

**Holm, S.R.; Noon, B.R.; Wiens, J.D.; Ripple, W.J. 2016.** Potential trophic cascades triggered by the barred owl range expansion. Wildlife Society Bulletin. 40(4): 615–624. doi:10.1002/wsb.714.

**Irwin, L.L.; Rock, D.F.; Miller, G.P. 2000.** Stand structures used by northern spotted owls in managed forests. Journal of Raptor Research. 34(3): 175–186.

**Irwin, L.L.; Rock, D.F.; Rock, S.C. 2012.** Habitat selection by northern spotted owls in mixed-coniferous forests. Journal of Wildlife Management. 76(1): 200–213. doi:10.1002/jwmg.218.

**Irwin, L.L.; Rock, D.F.; Rock, S.C. 2013.** Do northern spotted owls use harvested areas? Forest Ecology and Management. 310: 1029–1035. doi:10.1016/j.foreco.2013.04.001.

**Irwin, L.L.; Rock, D.F.; Rock, S.C.; Loehle, C.; Van Deusen, P. 2015.** Forest ecosystem restoration: initial response of spotted owls to partial harvesting. Forest Ecology and Management. 354: 232–242. doi:10.1016/j.foreco.2015.06.009.

**Ishak, H.D.; Dumbacher, J.P.; Anderson, N.L.; Keane, J.J.; Valkiunas, G.; Haig, S.M.; Tell, L.A.; Sehgal, R.N. 2008.** Blood parasites in owls with conservation implications for the spotted owl (*Strix occidentalis*). PLoS One. 3(5): e2304. doi:10.1371/journal.pone.0002304.

**Jennings, S.B.; Brown, N.B.; Sheil, D. 1999.** Assessing forest canopies and understory illumination: canopy closure, canopy cover and other measures. Forestry. 72(1): 59–73.

**Johnson, K.N.; Franklin, J.F. 2013.** Recommendations for future implementation of ecological forestry projects on BLM western Oregon forests. Portland, OR: U.S. Department of the Interior, Bureau of Land Management, Oregon/Washington State Office. 45 p.

**Jones, G.M.; Gutiérrez, R.J.; Tempel, D.J.; Whitmore, S.A.; Berigan, W.J.; Peery, M.Z. 2016.** Megafires: an emerging threat to old-forest species. Frontiers in Ecology and the Environment. 14(6): 300–306. doi:10.1002/fee.1298.

**Kelly, E.G.; Forsman, E.D. 2004.** Recent records of hybridization between barred owls (*Strix varia*) and northern spotted owls (*S. Occidentalis caurina*). Auk. 121(3): 806–810.

**Kelly, E.G.; Forsman, E.D.; Anthony, R.G. 2003.** Are barred owls displacing spotted owls? Condor. 105(1): 45–53.

**Kennedy, R.E.; Yang, Z.; Cohen, W.B.; Pfaff, E.; Braaten, J.; Nelson, P. 2012.** Spatial and temporal patterns of forest disturbance and regrowth within the area of the northwest forest plan. Remote Sensing of Environment. 122: 117–133. doi:10.1016/j.rse.2011.09.024.

**Kennedy, R.S.H.; Wimberly, M.C. 2009.** Historical fire and vegetation dynamics in dry forests of the interior Pacific Northwest, USA, and relationships to northern spotted owl (*Strix occidentalis caurina*) habitat conservation. Forest Ecology and Management. 258(5): 554–566.

**Koren, L.; Nakagawa, S.; Burke, T.; Soma, K.K.; Wynne-Edwards, K.E.; Geffen, E. 2012.** Non-breeding feather concentrations of testosterone, corticosterone and cortisol are associated with subsequent survival in wild house sparrows. Proceedings of the Royal Society—Biological Sciences (Series B). 279(1733): 1560–1566. doi:10.1098/rspb.2011.2062.

**Kroll, A.J.; Fleming, T.L.; Irwin, L.L. 2010.** Site occupancy dynamics of northern spotted owls in the eastern Cascades, Washington, USA, 1990–2003. Journal of Wildlife Management. 74(6): 1264–1274. doi:10.2193/2009-429.

**Lee, D.E.; Bond, M.L. 2015.** Occupancy of California spotted owl sites following a large fire in the Sierra Nevada, California. Condor. 117(2): 228–236. doi:10.1650/condor-14-155.1.

**Lee, D.E.; Bond, M.L.; Borchert, M.I.; Tanner, R. 2013.** Influence of fire and salvage logging on site occupancy of spotted owls in the San Bernardino and San Jacinto Mountains of southern California. Journal of Wildlife Management. 77(7): 1327–1341. doi:10.1002/jwmg.581.

**Lehmkuhl, J.F.; Kistler, K.D.; Begley, J.S. 2006a.** Bushy-tailed woodrat abundance in dry forests of eastern Washington. Journal of Mammology. 87(2): 371–379.

**Lehmkuhl, J.F.; Kistler, K.D.; Begley, J.S.; Boulanger, J. 2006b.** Demography of northern flying squirrels informs ecosystem management of western interior forests. Ecological Applications. 16(2): 584–600.

**Lehmkuhl, J.F.; Peterson, D.W.; Gaines, W.L.; Bailey, J.; Youngblood, A., eds. 2015.** Silviculture and monitoring guidelines for integrating restoration of dry mixed-conifer forest and spotted owl habitat management in the eastern Cascade Range. Gen. Tech. Rep. PNW-GTR-915. Portland, OR: U.S. Department of Agriculture, Forest Service, Pacific Northwest Research Station. 168 p.

**Leskiw, T.; Gutiérrez, R.J. 1998.** Possible predation of a spotted owl by a barred owl. Western Birds. 29: 225–226.

**Lewicki, K.E.; Huyvaert, K.P.; Piaggio, A.J.; Diller, L.V.; Franklin, A.B. 2015.** Effects of barred owl (*Strix varia*) range expansion on haemoproteus parasite assemblage dynamics and transmission in barred and northern spotted owls (*Strix occidentalis caurina*). Biological Invasions. 17(6): 1713–1727.

**Lindenmayer, D.B.; Noss, R.F. 2006.** Salvage logging, ecosystem processes, and biodiversity conservation. Conservation Biology. 20(4): 949–958. doi:10.1111/j.1523-1739.2006.00497.x.

**Lint, J., ed. 2005.** Northwest Forest Plan—the first 10 years (1994–2003): status and trends of northern spotted owl populations and habitat. Gen. Tech. Rep. PNW-GTR-648. Portland, OR: U.S. Department of Agriculture, Forest Service, Pacific Northwest Research Station. 176 p.

**Lint, J.; Noon, B.; Anthony, R.; Forsman, E.; Raphael, M.; Collopy, M.; Starkey, E. 1999.** Northern spotted owl effectiveness monitoring plan for the Northwest Forest Plan. Gen. Tech. Rep. PNW-GTR-440. Portland, OR: U.S. Department of Agriculture, Forest Service, Pacific Northwest Research Station. 43 p.

**Livezey, K.B. 2007.** Barred owl habitat and prey: a review and synthesis of the literature. Journal of Raptor Research. 41(3): 177–201.

**Livezey, K.B. 2009.** Range expansion of barred owls, part I: chronology and distribution. American Midland Naturalist. 161: 49–56.

**Livezey, K.B. 2010.** Killing barred owls to help spotted owls I: a global perspective. Northwestern Naturalist. 91: 107–133.

**Loehle, C.; Irwin, L. 2006.** Reply to the comment by Franklin et al. on "are survival rates for northern spotted owls biased?" Canadian Journal of Zoology. 84(9): 1380–1382.

**Loehle, C.; Irwin, L. 2015.** Range wide analysis of northern spotted owl habitat relations: reply to comments. Forest Ecology and Management. 358: 365–368. doi:10.1016/j.foreco.2015.09.033.

**Loehle, C.; Irwin, L.; Manly, B.F.J.; Merrill, A. 2015.** Range-wide analysis of northern spotted owl nesting habitat relations. Forest Ecology and Management. 342: 8–20. doi:10.1016/j.foreco.2015.01.010.

**Loehle, C.; Irwin, L.; Rock, D.; Rock, S. 2005.** Are survival rates for northern spotted owls biased? Canadian Journal of Zoology. 83: 1386–1390. doi:10.1139/Z05-137.

**MacKenzie, D.I.; Nichols, J.D.; Royle, J.A.; Pollock, K.H.; Bailey, L.L.; Hines, J.E. 2006.** Occupancy estimation and modeling: Inferring patterns and dynamics of species occurrence. New York: Academic Press. 324 p.

**Manning, T.; Hagar, J.C.; McComb, B.C. 2012.** Thinning of young Douglas-fir forests decreases density of northern flying squirrels in the Oregon Cascades. Forest Ecology and Management. 264: 115–124. doi:10.1016/j.foreco.2011.09.043.

**Marcot, B.G.; Raphael, M.G.; Schumaker, N.H.; Galleher, B. 2013.** How big and how close? Habitat patch size and spacing to conserve a threatened species. Natural Resource Modeling. 26(2): 194–214.

**Marcot, B.G.; Singleton, P.H.; Schumaker, N.H. 2015.** Analysis of sensitivity and uncertainty in an individual-based model of a threatened wildlife species. Natural Resource Modeling. 28(1): 37–58.

**Mathewson, H.A.; Morrison, M.L. 2015.** The misunderstanding of habitat. In: Morrison, M.L.; Mathewson, H.A., eds. Wildlife habitat conservation: concepts, challenges, and solutions. Baltimore, MD: Johns Hopkins University Press: 3–8. Chapter 1.

**McDonald, T.L.; Manly, B.F.J.; Nielson, R.M.; Diller, L.V. 2006.** Discrete-choice modeling in wildlife studies exemplified by northern spotted owl nighttime habitat selection. Journal of Wildlife Management. 70(2): 375–383.

**McKelvey, K.S. 2015.** The effects of disturbance and succession on wildlife habitat and animal communities. In: Morrison, M.L.; Mathewson, H.A., eds. Wildlife habitat conservation: concepts, challenges, and solutions. Baltimore, MD: Johns Hopkins University Press: 143–156. Chapter 11.

**Merschel, A.G.; Spies, T.A.; Heyerdahl, E.K. 2014.** Mixed-conifer forests of central Oregon: effects of logging and fire exclusion vary with environment. Ecological Applications. 24(7): 1670–1688.

**Miller, G.S.; Small, R.J.; Meslow, E.C. 1997.** Habitat selection by spotted owls during natal dispersal in western Oregon. Journal of Wildlife Management. 61(1): 140–150.

**Miller, M.P.; Mullins, T.D.; Forsman, E.D.; Haig, S.M. 2017.** Genetic differentiation and inferred dynamics of a hybrid zone between northern spotted owls (*Strix occidentalis caurina*) and California spotted owls (*S. o. occidentalis*) in northern California. Ecology and Evolution. 00: 1–13. doi:10.1002/ece3.3260.

**Monahan, W.B.; Hijmans, R.J. 2007.** Distribution dynamics of invasion and hybridization by *Strix* spp. in western North America. Ornithological Monographs. 63: 55–66.

**Morrison, M.L.; Marcot, B.G.; Mannan, R.W. 2006.** Wildlife-habitat relationships: concepts and applications. 3rd ed. Washington, DC: Island Press. 520 p.

**Mote, P.W.; Parson, E.A.; Hamlet, A.F.; Keeton, W.S.; Lettenmaier, D.; Mantua, N.; Miles, E.L.; Peterson, D.W.; Peterson, D.L.; Slaughter, R.; Snover, A.K. 2003.** Preparing for climate change: the water, salmon, and forests of the Pacific Northwest. Climate Change. 61: 45–88.

**Noon, B.R.; Blakesley, J.A. 2006.** Conservation of the northern spotted owl under the Northwest Forest Plan. Conservation Biology. 20(2): 288–296.

**Olson, G.; Anthony, R.G.; Forsman, E.D.; Ackers, S.H.; Loschl, P.J.; Reid, J.A.; Dugger, K.M.; Glenn, E.M.; Ripple, W.J. 2005.** Modeling of site occupancy dynamics for northern spotted owls, with emphasis on the effects of barred owls. Journal of Wildlife Management. 69(3): 918–932.

**Olson, G.S.; Glenn, E.M.; Anthony, R.G.; Forsman, E.D.; Reid, J.A.; Loschl, P.J.; Ripple, W.J. 2004.** Modeling demographic performance of northern spotted owls relative to forest habitat in Oregon. Journal of Wildlife Management. 68(4): 1039–1053.

**Pearson, R.R.; Livezey, K.B. 2003.** Distribution, numbers, and site characteristics of spotted owls and barred owls in the Cascade Mountains of Washington. Journal of Raptor Research. 37(4): 265–276.

**Pearson, R.R.; Livezey, K.B. 2007.** Spotted owls, barred owls, and late-successional reserves. Journal of Raptor Research. 41(2): 156–161.

**Perry, D.A.; Hessburg, P.F.; Skinner, C.N.; Spies, T.A.; Stephens, S.L.; Taylor, A.H.; Franklin, J.F.; McComb, B.; Riegel, G. 2011.** The ecology of mixed severity fire regimes in Washington, Oregon, and northern California. Forest Ecology and Management. 262(5): 703–717.

**Pierce, D.J.; Buchanan, J.B.; Consentino, B.L.; Snyder, S. 2005.** An assessment of spotted owl habitat on non-federal lands in Washington between 1996 and 2004. Final report. Olympia, WA: Washington Department of Fish and Wildlife. 187 p.

**Price, A.L.; Mowdy, J.S.; Swingle, J.K.; Forsman, E.D. 2015.** Distribution and abundance of tree voles in the northern Coast Ranges of Oregon. Northwestern Naturalist. 96(1): 37–49. doi:10.1898/NWN14-04.1.

**Raphael, M.G. 2006.** Conservation of listed species: the northern spotted owl and marbled murrelet. In: Haynes, R.W.; Borman, B.T.; Lee, D.C.; Martin, J.R., eds. Northwest Forest Plan—the first 10 years (1994–2003): synthesis of monitoring and research results. Gen. Tech. Rep. PNW-GTR-651. Portland, OR: U.S. Department of Agriculture, Forest Service, Pacific Northwest Research Station: 117–143. Chapter 7.

**Reilly, M.J.; Dunn, C.J.; Meigs, G.W.; Spies, T.S.; Kennedy, R.E.; Bailey, J.D.; Briggs, K. 2017.** Contemporary patterns of fire extent and severity in forests of the Pacific Northwest, USA (1985–2010). Ecosphere. 8(3): e01695.

**Riley, S.; Bromley, C.; Poppenga, R.; Uzal, F.; Whited, L.; Sauvajot, R. 2007.** Anticoagulant exposure and notoedric mange in bobcats and mountain lions in urban southern California. Journal of Wildlife Management. 71(6): 1874–1884.

**Roberts, S.L.; van Wagtendonk, J.W.; Miles, A.K.; Kelt, D.A. 2011.** Effects of fire on spotted owl site occupancy in a late-successional forest. Biological Conservation. 144(1): 610–619. doi:10.1016/j.biocon.2010.11.002.

**Rockweit, J.T.; Franklin, A.B.; Carlson, P.C. 2017.** Differential impacts of wildfire on the population dynamics of an old-forest species. Ecology. 98(6): 1574–1582.

**Roloff, G.J.; Mealey, S.P.; Bailey, J.D. 2012.** Comparative hazard assessment for protected species in a fire-prone landscape. Forest Ecology and Management. 277: 1–10. doi:10.1016/j.foreco.2012.04.015.

**Roloff, G.J.; Mealey, S.P.; Clay, C.; Barry, J.; Yanish, C.; Neuenschwander, L. 2005.** A process for modeling short- and long-term risk in the southern Oregon Cascades. Forest Ecology and Management. 211(1–2): 166–190.

**Romero, L.M.; Reed, J.M.; Wingfield, J.C. 2000.** Effects of weather on corticosterone responses in wild free-living passerine birds. General and Comparative Endocrinology. 118: 113–122.

**Romero, L.M.; Wikelski, M. 2001.** Corticosterone levels predict survival probabilities of Galapagos marine iguanas during El Niño events. Proceedings of the National Academy of Sciences. 98(13): 7366–7370. doi:10.1073/pnas.131091498.

**Rosenberg, D.K.; Swindle, K.A.; Anthony, R.G. 2003.** Influence of prey abundance on northern spotted owl reproductive success in western Oregon. Canadian Journal of Zoology. 81(10): 1715–1725.

**Rosenberg, D.K.; Vesely, D.G.; Gervais, J.A. 2012.** Maximizing endangered species research. Science. 337: 799.

**Rullman, S.; Marzluff, J.M. 2014.** Raptor presence along an urban-wildland gradient: influences of prey abundance and land cover. Journal of Raptor Research. 48(3): 257–272. doi:10.3356/jrr-13-32.1.

**Safford, H.D.; Stevens, J.T.; Merriam, K.; Meyer, M.D.; Latimer, A.M. 2012.** Fuel treatment effectiveness in California yellow pine and mixed conifer forests. Forest Ecology and Management. 274: 17–28. doi:10.1016/j.foreco.2012.02.013.

**Sakai, H.F.; Noon, B.R. 1993.** Dusky-footed woodrat abundance in different-aged forests in northwestern California. Journal of Wildlife Management. 57(2): 373–382.

**Schilling, J.W.; Dugger, K.M.; Anthony, R.G. 2013.** Survival and home-range size of northern spotted owls in southwestern Oregon. Journal of Raptor Research. 47(1): 1–14.

**Schumaker, N.H. 2008.** Hexsim (version 1.3.6.9). http://www.hexsim.net. (13 September 2017).

**Schumaker, N.H. 2015.** Hexsim (version 4.0.9). http://www.hexsim.net. (13 September 2017).

**Schumaker, N.H.; Brookes, A.; Dunk, J.R.; Woodbridge, B.; Heinrichs, J.A.; Lawler, J.J.; Carroll, C.; LaPlante, D. 2014.** Mapping sources, sinks, and connectivity using a simulation model of northern spotted owls. Landscape Ecology. 29(4): 579–592.

**Seamans, M.E.; Corcoran, J.; Rex, A. 2004.** Southernmost record of a spotted owl x barred owl hybrid in the Sierra Nevada. Western Birds. 35: 173–174.

**Seamans, M.E.; Gutiérrez, R.J. 2007.** Habitat selection in a changing environment: the relationship between habitat alteration and spotted owl territory occupancy and breeding dispersal. The Condor. 109: 566–576.

**Sheriff, M.J.; Dantzer, B.; Delehanty, B.; Palme, R.; Boonstra, R. 2011.** Measuring stress in wildlife: techniques for quantifying glucocorticoids. Oecologia. 166(4): 869–887.

**Singleton, P.H. 2013.** Barred owls and northern spotted owls in the eastern Cascade Range, Washington. Seattle, WA: University of Washington. 214 p. Ph.D. dissertation.

**Singleton, P.H. 2015.** Forest structure within barred owl (*Strix varia*) home ranges in the eastern Cascade Range, Washington. Journal of Raptor Research. 49(2): 129–140.

**Singleton, P.H.; Lehmkuhl, J.F.; Gaines, W.L.; Graham, S.A. 2010.** Barred owl space use and habitat selection in the eastern Cascades, Washington. Journal of Wildlife Management. 74(2): 285–294. doi:10.2193/2008-548.

**Sollmann, R.; White, A.M.; Tarbill, G.L.; Manley, P.N.; Knapp, E.E. 2016.** Landscape heterogeneity compensates for fuel reduction treatment effects on northern flying squirrel populations. Forest Ecology and Management. 373: 100–107. doi:10.1016/j.foreco.2016.04.041.

**Sovern, S.G.; Forsman, E.D.; Dugger, K.M.; Taylor, M. 2015.** Roosting habitat use and selection by northern spotted owls during natal dispersal. Journal of Wildlife Management. 79(2): 254–262. doi:10.1002/jwmg.834.

**Sovern, S.G.; Forsman, E.D.; Olson, G.S.; Biswell, B.L.; Taylor, M.; Anthony, R.G. 2014.** Barred owls and landscape attributes influence territory occupancy of northern spotted owls. Journal of Wildlife Management. 78(8): 1436–1443.

**Sovern, S.G.; Taylor, M.; Forsman, E.D. 2011.** Nest reuse by northern spotted owls on the east slope of the Cascade Range, Washington. Northwestern Naturalist. 92(2): 101–106. doi:10.1898/10-01.1.

**Spies, T.A.; Miller, J.D.; Buchanan, J.B.; Lehmkuhl, J.F.; Franklin, J.F.; Healey, S.P.; Hessburg, P.F.; Safford, H.D.; Cohen, W.B.; Kennedy, R.S.; Knapp, E.E.; Agee, J.K.; Moeur, M. 2010.** Underestimating risks to the northern spotted owl in fire-prone forests: response to Hanson et al. Conservation Biology. 24(1): 330–333; discussion 334–337. doi:10.1111/j.1523-1739.2009.01414.x.

**Spies, T.A.; White, E.; Ager, A.; Kline, J.D.; Bolte, J.P.; Platt, E.K.; Olsen, K.A.; Pabst, R.J.; Barros, A.M.G.; Bailey, J.D.; Charnley, S.; Koch, J.; Steen-Adams, M.M.; Singleton, P.H.; Sulzman, J.; Schwartz, C.; Csuti, B. 2017.** Using an agent-based model to examine forest management outcomes in a fire-prone landscape in Oregon, USA. Ecology and Society. 22(1): 25. doi:https://doi.org/10.5751/ES-08841-220125.

**Stephens, S.L.; Moghaddas, J.J.; Edminster, C.; Fieldler, C.E.; Haase, S.; Harrington, M.; Keeley, J.E.; Knapp, E.E.; McIver, J.D.; Metlen, K.; Skinner, C.N.; Youngblood, A. 2009.** Fire treatment effects on vegetation structure, fuels, and potential fire severity in western U.S. Forests. Ecological Applications. 19(2): 305–320.

**Stine, P.; Hessburg, P.; Spies, T.; Kramer, M.; Fettig, C.J.; Hansen, A.; Lehmkuhl, J.; O'Hara, K.; Polivka, K.; Singleton, P.; Charnley, S.; Merschel, A.; White, R. 2014.** 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. Gen. Tech. Rep. PNW-GTR-897. Portland, OR: U.S. Department of Agriculture, Forest Service, Pacific Northwest Research Station. 254 p.

**Stone, W.B.; Okoniewski, J.C.; Stedelin, J.R. 1999.** Poisoning of wildlife with anticoagulant rodenticides in New York. Journal of Wildlife Diseases. 35(2): 187–193.

**Stralberg, D.; Fehring, K.E.; Pomara, L.Y.; Nur, N.; Adams, D.B.; Hatch, D.; Geupel, G.R.; Allen, S. 2009.** Modeling nest-site occurrence for the northern spotted owl at its southern range limit in central California. Landscape and Urban Planning. 90(1–2): 76–85.

**Strong, R.J.; Pereira, M.G.; Shore, R.F.; Henrys, P.A.; Pottinger, T.G. 2015.** Feather corticosterone content in predatory birds in relation to body condition and hepatic metal concentration. General and Comparative Endocrinology. 214: 47–55. doi:10.1016/j.ygcen.2015.03.002.

**Swanson, M.E.; Franklin, J.F.; Beschta, R.L.; Crisafulli, C.M.; DellaSala, D.A.; Hutto, R.L.; Lindenmayer, D.B.; Swanson, F.J. 2011.** The forgotten stage of forest succession: early-successional ecosystems on forest sites. Frontiers in Ecology and the Environment. 9(2): 117–125. doi:10.1890/090157.

**Swingle, J.K.; Forsman, E.D. 2009.** Home range areas and activity patterns of red tree voles (*Arborimus longicaudus*) in western Oregon. Northwest Science. 83(3): 273–286.

**Taylor, A.L.; Forsman, E.D. 1976.** Recent range extension of the barred owl in western North America, including the first records for Oregon. Condor. 78(4): 560–561.

**Thomas, J.W.; Forsman, E.D.; Lint, J.B.; Meslow, E.C.; Noon, B.B.; Verner, J. 1990.** A conservation strategy for the northern spotted owl. Portland, OR: Interagency Scientific Committee to Address the Conservation of the Northern Spotted Owl. 427 p.

**Thomas, J.W.; Franklin, J.F.; Gordon, J.; Johnson, K.N. 2006.** The Northwest Forest Plan: origins, components, implementation experience, and suggestions for change. Conservation Biology. 20(2): 277–287.

**Thompson, J.R.; Spies, T.A. 2009.** Vegetation and weather explain variation in crown damage within a large mixed-severity wildfire. Forest Ecology and Management. 258(7): 1684–1694. doi:10.1016/j.foreco.2009.07.031.

**Thorn, S.; Bässler, C.; Brandl, R.; Burton, P.J.; Cahall, R.; Campbell, J.L.; Castro, J.; Choi, C.-Y.; Cobb, T.; Donato, D.C.; Durska, E.; Fontaine, J.B.; Gauthier, S.; Hebert, C.; Hothorn, T.; Hutto, R.L.; Lee, E.-J.; Leverkus, A.B.; Lindenmayer, D.B.; Obrist, M.K.; Rost, J.; Seibold, S.; Seidl, R.; Thom, D.; Waldron, K.; Wermelinger, B.; Winter, M.-B.; Zmihorski, M.; Müller, J. 2017.** Impacts of salvage logging on biodiversity—a meta-analysis. Journal of Applied Ecology. doi:10.1111/1365-2664.12945.

**U.S. Department of Agriculture [USDA]. 2012.** National Forest System land management planning. Federal Register. 77(68): 21162–21276.

**U.S. Department of Agriculture, Forest Service; U.S. Department of the Interior, Bureau of Land Management [USDA and USDI]. 1994.** Final supplemental environmental impact statement on management of habitat for late-successional and old-growth forest related species within the range of the northern spotted owl. [Place of publication unknown]. Vol. 1–2 + Record of Decision.

**U.S. Department of the Interior [USDI]. 2013.** Experimental removal of barred owls to benefit threatened northern spotted owls; record of decision for final environmental impact statement. Federal Register. 78(180): 57171–57173.

**U.S. Department of the Interior, Fish and Wildlife Service [USFWS]. 1990.** Endangered and threatened wildlife and plants: determination of threatened status for the northern spotted owl. Federal Register. 55(123): 26114–26194.

**U.S. Department of the Interior, Fish and Wildlife Service [USFWS]. 1992.** Recovery plan for the northern spotted owl. Portland, OR: Pacific Region. 322 p.

**U.S. Department of the Interior, Fish and Wildlife Service [USFWS]. 2008.** Recovery plan for the northern spotted owl (*Strix occidentalis caurina*). Portland, OR: Pacific Region. 142 p.

**U.S. Department of the Interior, Fish and Wildlife Service [USFWS]. 2011a.** Endangered and threatened wildlife and plants: revised recovery plan for the northern spotted owl (*Strix occidentalis caurina*). Federal Register. 76(127): 38575–38576.

**U.S. Department of the Interior, Fish and Wildlife Service [USFWS]. 2011b.** Revised recovery plan for the northern spotted owl (*Strix occidentalis caurina*). Portland, OR: Pacific Region. 258 p.

**U.S. Department of the Interior, Fish and Wildlife Service [USFWS]. 2012a.** Endangered and threatened wildlife and plants: revised critical habitat for the northern spotted owl. Federal Register. 77(233): 71876–72068.

**U.S. Department of the Interior, Fish and Wildlife Service [USFWS]. 2012b.** Final environmental assessment: revision of critical habitat for the northern spotted owl. Portland, OR: Pacific Region. 329 p.

**U.S. Department of the Interior, Fish and Wildlife Service [USFWS]. 2013.** Experimental removal of barred owls to benefit threatened northern spotted owls: final environmental impact statement. Portland, OR: Oregon Fish and Wildlife Office. 505 p.

**Valéry, L.; Fritz, H.; Lefeuvre, J.C.; Simberloff, D. 2009.** Invasive species can also be native. Trends in Ecology & Evololution. 24(11): 585. doi:10.1016/j.tree.2009.07.003.

**Van Lanen, N.J.; Franklin, A.B.; Huyvaert, K.P.; Reiser, R.F.; Carlson, P.C. 2011.** Who hits and hoots at whom? Potential for interference competition between barred and northern spotted owls. Biological Conservation. 144(9): 2194–2201. doi:10.1016/j.biocon.2011.05.011.

**Van Mantgem, P.J.; Stephenson, N.L.; Byrne, J.C.; Daniels, L.D.; Franklin, J.F.; Fulé, P.Z.; Harmon, M.E.; Larson, A.J.; Smith, J.M.; Taylor, A.H.; Veblen, T.T. 2009.** Widespread increase of tree mortality rates in the western United States. Science. 323(5913): 521–524.

**Walters, J.R.; Zabel, C.J. 1995.** Northern flying squirrel densities in fir forests of northeastern California. Journal of Wildlife Management. 59(4): 858–866.

**Waters, J.R.; Zabel, C.J. 1995.** Northern flying squirrel densities in fir forests of northeastern California. Journal of Wildlife Management. 59(4): 858–866.

**Weisel, L.E. 2015.** Northern spotted owl and barred owl home range size and habitat selection in coastal northwestern California. Arcata, CA: Humboldt State University. 54 p. M.S. thesis.

**Westerling, A.L.; Hidalgo, H.G.; Cayan, D.R.; Swetnam, T.W. 2006.** Warming and earlier spring increase western U.S. forest wildfire activity. Science. 313(5789): 940–943. doi: 10.1126/science.1128834.

**Wiens, J.D.; Anthony, R.G.; Forsman, E.D. 2011.** Barred owl occupancy surveys within the range of the northern spotted owl. Journal of Wildlife Management. 75(3): 531–538. doi:10.1002/jwmg.82.

**Wiens, J.D.; Anthony, R.G.; Forsman, E.D. 2014.** Competitive interactions and resource partitioning between northern spotted owls and barred owls in western Oregon. Wildlife Monographs. 185(1): 1–50. doi:10.1002/wmon.1009.

**Wiens, J.D.; Dugger, K.M.; Lewicki, K.E.; Simon, D.C. 2017.** Effects of experimental removal of barred owls on population demography of northern spotted owls in Washington and Oregon—2016 progress report. Open-File Report 2017-1040. Reston, VA: U.S. Department of the Interior, Geological Survey. 23 p. https://doi.org/10.3133/ofr20171040. (13 September 2017).

**Wilson, T.M. 2010.** Limiting factors for northern flying squirrels (*Glaucomys sabrinus*) in the Pacific Northwest: a spatio-temporal analysis. Cincinnati, OH: Union Institute & University. 207 p. Ph.D. dissertation.

**Wilson, T.M.; Forsman, E.D. 2013.** Thinning effects on spotted owl prey and other forest-dwelling small mammals. In: Anderson, P.D.; Ronnenberg, K.L., eds. Density management in the 21st century: west side story. Gen. Tech. Rep. PNW-GTR-880. Portland, OR: U.S. Department of Agriculture, Forest Service, Pacific Northwest Research Station: 79–90.

**Yackulic, C.B.; Reid, J.; Davis, R.; Hines, J.E.; Nichols, J.D.; Forsman, E. 2012.** Neighborhood and habitat effects on vital rates: expansion of the barred owl in the Oregon Coast Ranges. Ecology. 93(8): 1953–1966.

**Yackulic, C.B.; Reid, J.; Nichols, J.D.; Hines, J.E.; Davis, R.; Forsman, E. 2014.** The roles of competition and habitat in the dynamics of populations and species distributions. Ecology. 95(2): 265–279.

**Zabel, C.J.; Mckelvey, K.; Ward, J.P. 1995.** Influence of primary prey on home-range size and habitat-use patterns of northern spotted owls (*Strix occidentalis caurina*). Canadian Journal of Zoology. 73(3): 433–439.

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.

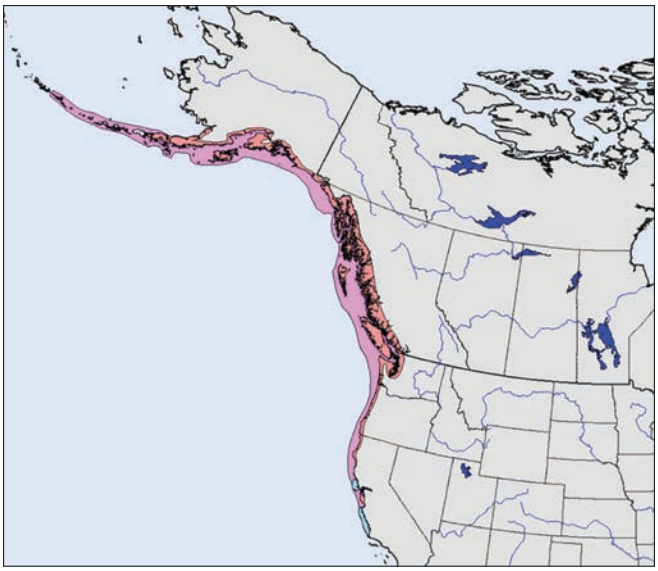

Figure 5-2—Range of the marbled murrelet in North America. Map by Terry Sohl from NatureServe data.

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 occupied sites**

| State/province | Inland distance: Nest[a] | Inland distance: 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, Washington Department of Fish and Wildlife, 111 Washington Street SE, Olympia, WA 98501.
[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

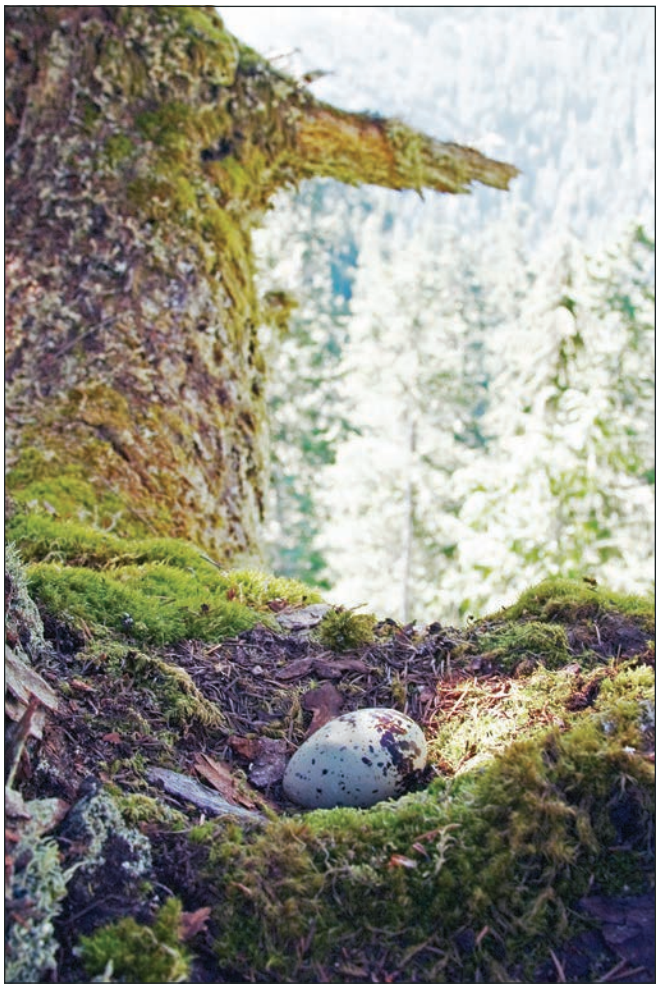

Figure 5-3—Nesting platforms usually include large branches and other structures large enough to provide a platform for a nesting adult murrelet.

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.

[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.

## 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).

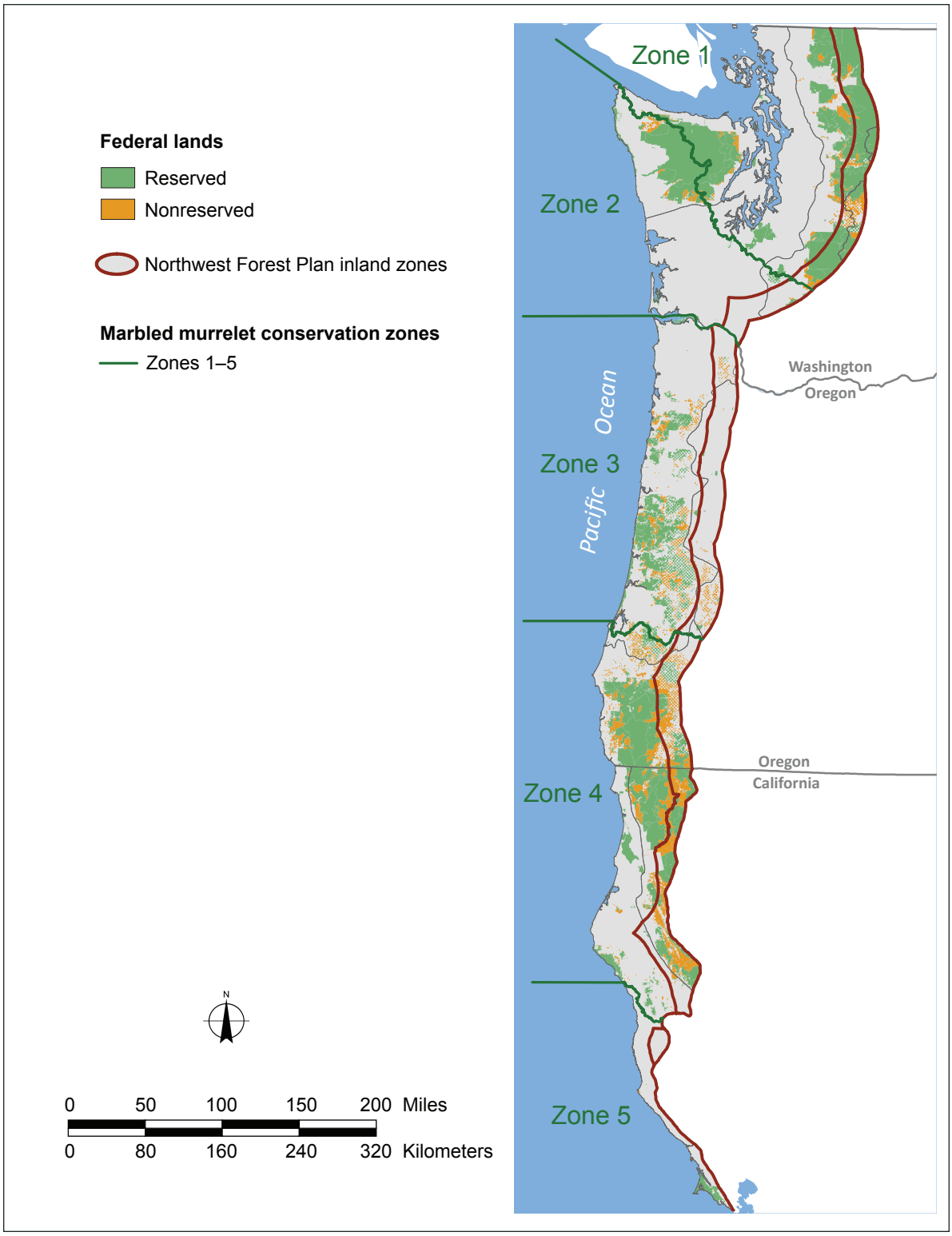


Figure 5-4—Range of the marbled murrelet with boundaries of conservation zones and locations of federal reserves, other federal lands, and nonfederal lands. Also shown are boundaries of the "inland zones" designated by the Northwest Forest Plan; see "Research Needs, Uncertainties, Information Gaps, and 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).

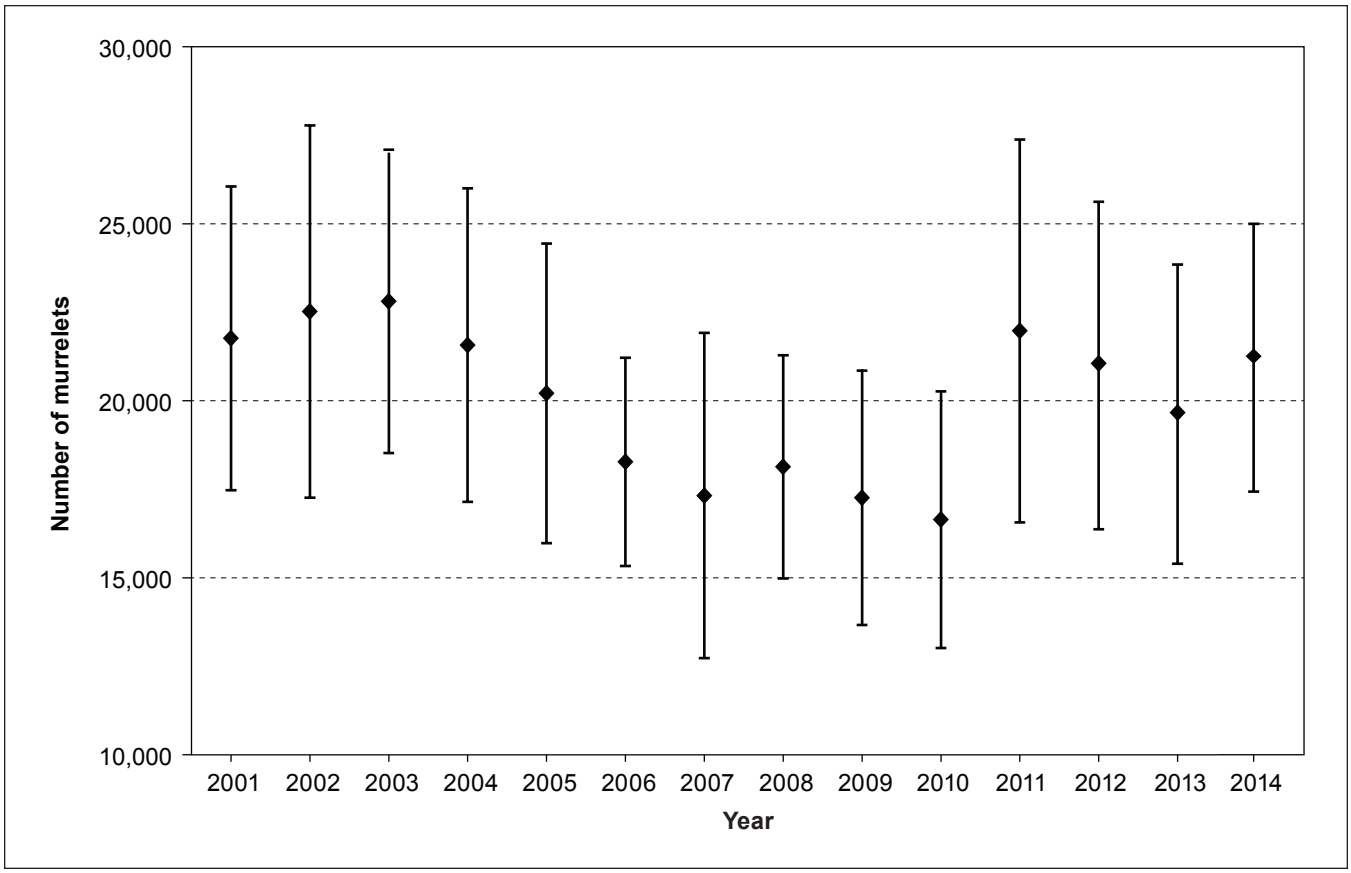


Figure 5-5—Annual marbled murrelet population estimates and 95 percent confidence intervals for the Northwest Forest Plan area (conservation zones 1 through 5 combined) based on 2000–2014 data (Falxa et al. 2016, Lynch et al. 2016).

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.

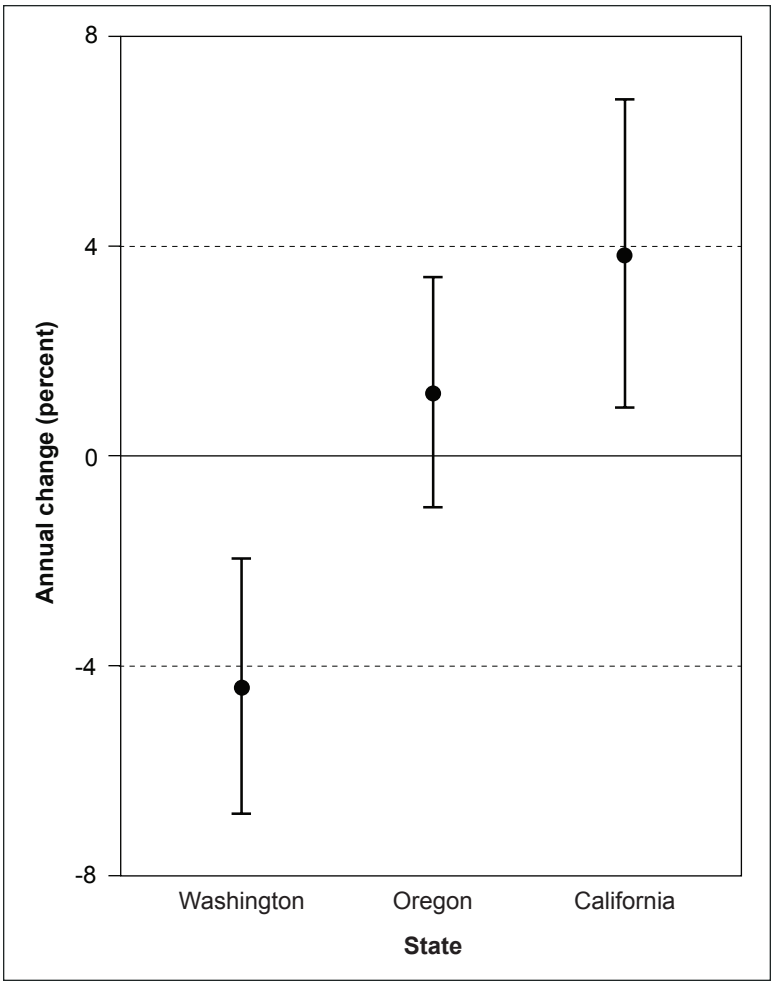


Figure 5-6—Trend results: average rate of annual change by state, 2000 to 2013, with 95 percent confidence intervals. Washington trend is based on 2001–2015 data, Oregon on 2000–2014 data, and California on 2000–2015 data (Falxa et al. 2016, Lynch et al. 2016).

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

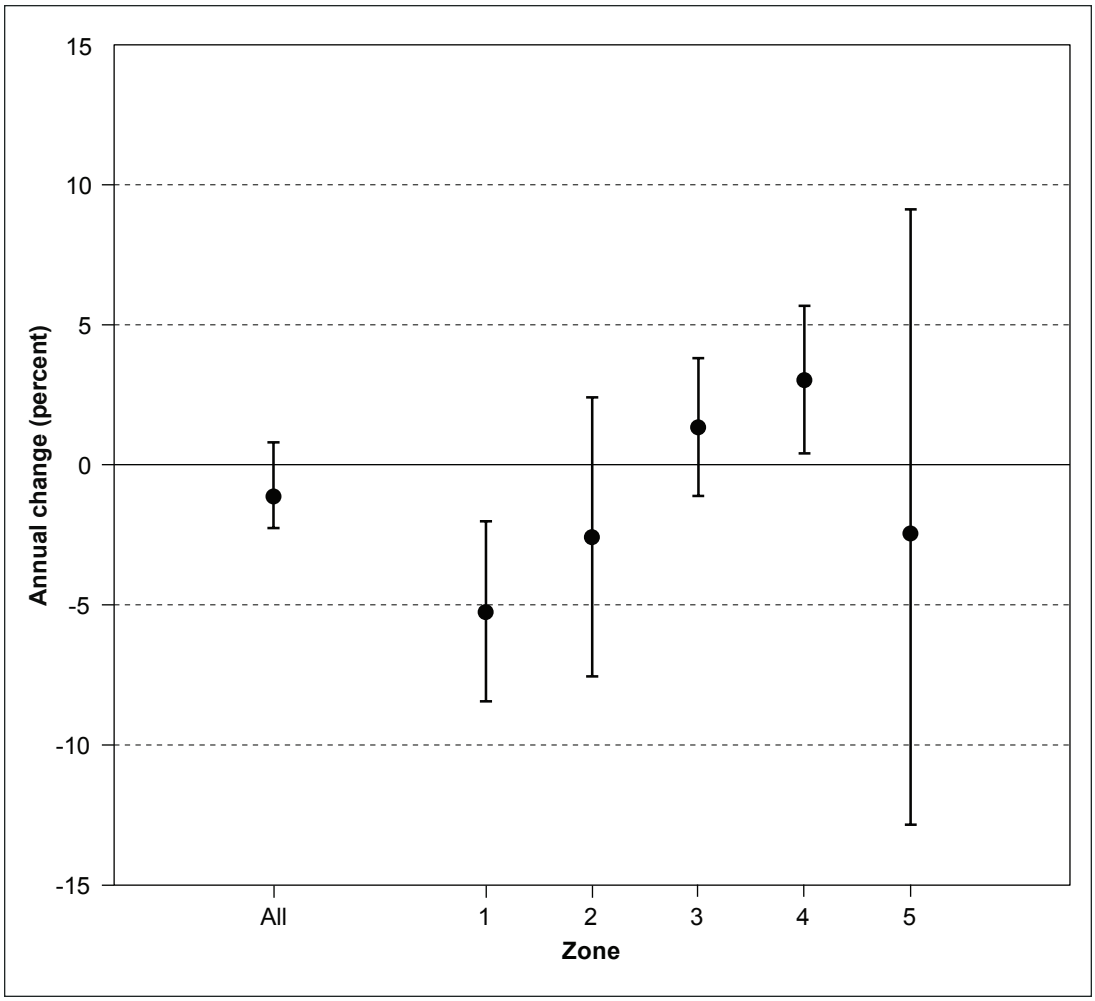


Figure 5-7—Trend results: average rate of annual change by conservation zone (see fig. 5-2 for zone locations) and for all conservation zones combined, with 95 percent confidence intervals. All zones based on 2001–2014 data, zones 1 and 2 on 2001–2015 data, zone 3 on 2000–2014 data, zone 4 on 2000–2015 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.

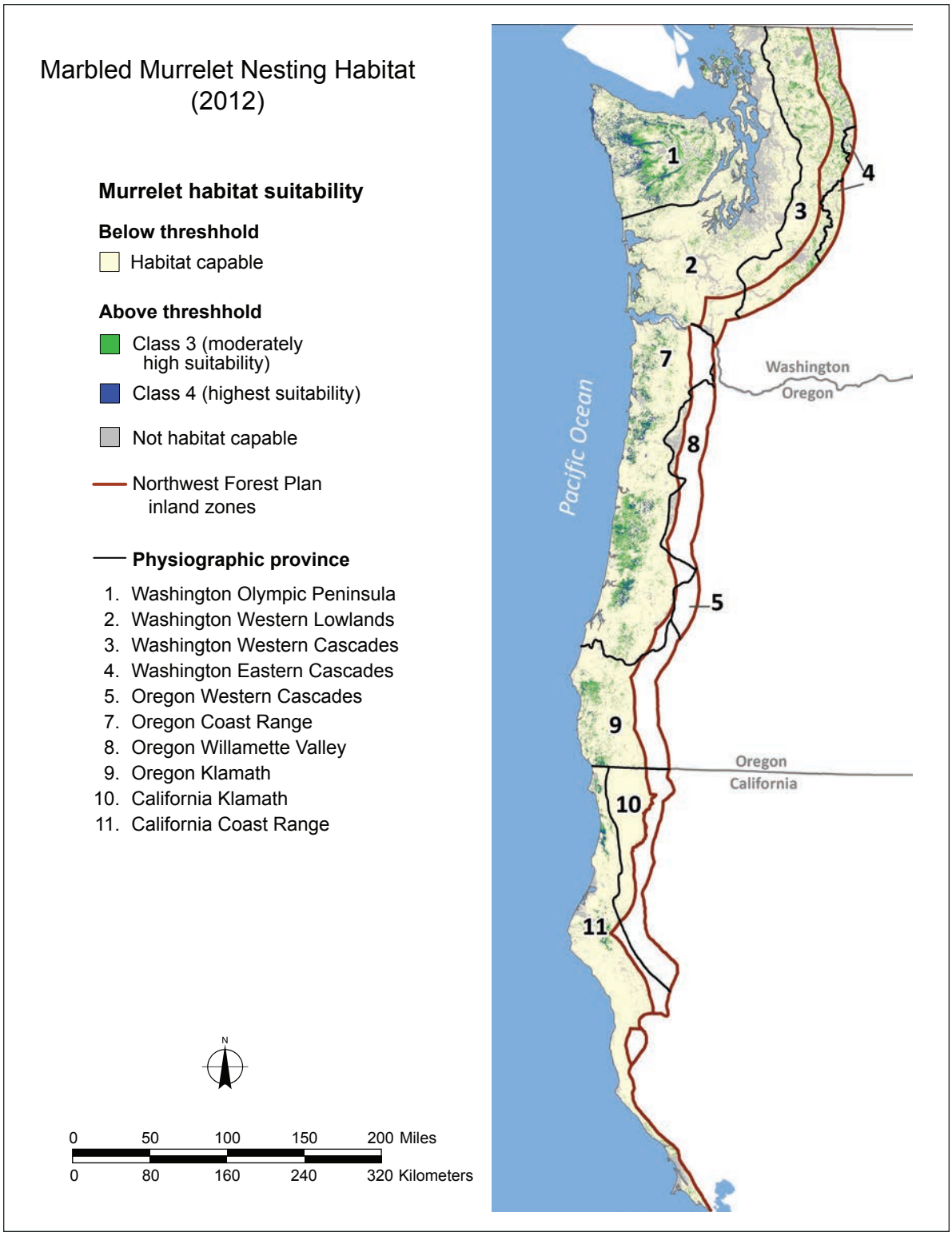


Figure 5-8—Map of suitability for marbled murrelet nesting habitat, 2012 (Raphael et al. 2016a).

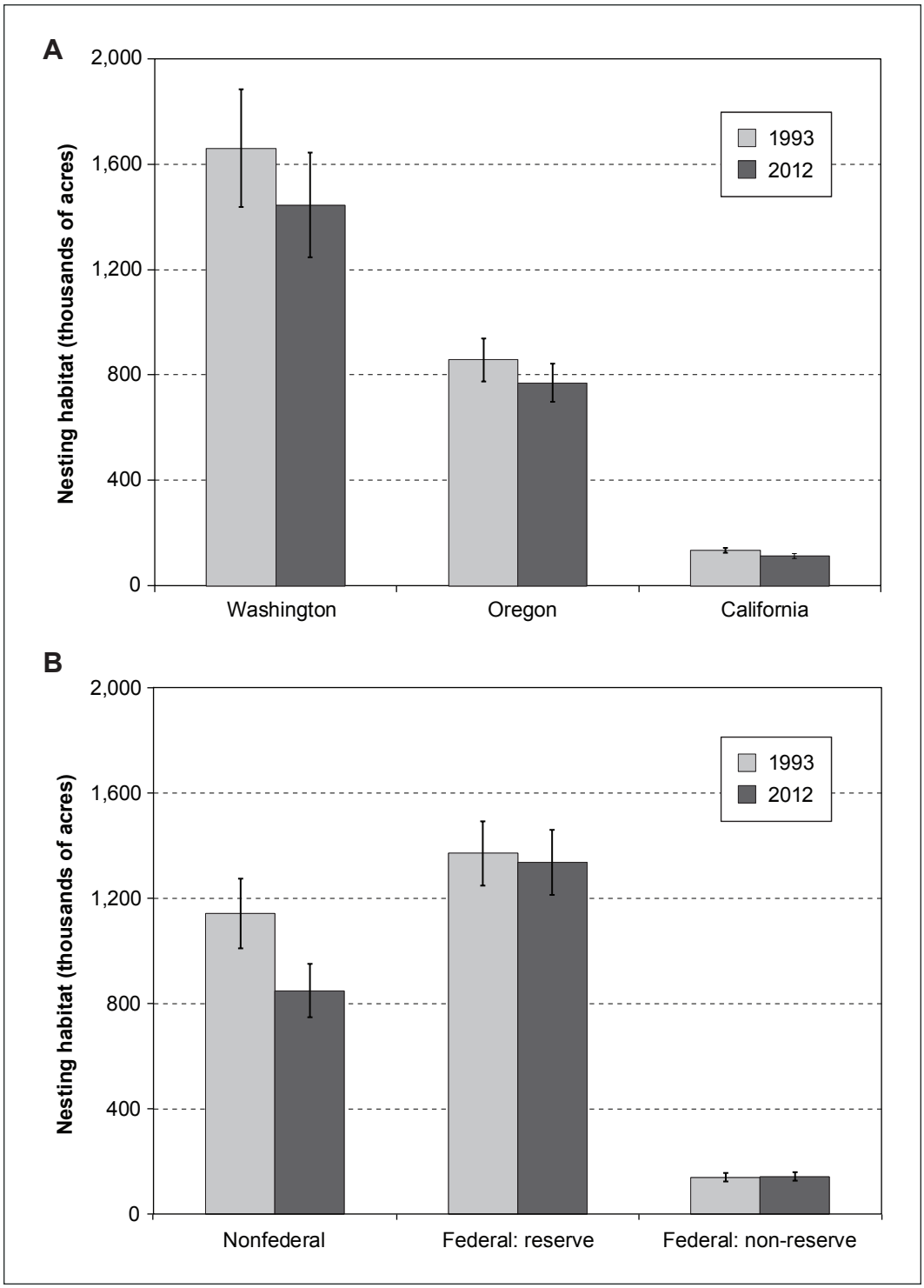


Figure 5-9—Estimated amounts of higher suitability nesting habitat of the marbled murrelet in 1993 and 2012, by (A) state, and (B) land allocation (Raphael et al. 2016a). Error bars are 95-percent confidence intervals from 25 replicated model runs. See table on page 338 for metric equivelents.

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**

| Land allocation | Losses[a] | | | |
|---|---|---|---|---|
| | 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. 2014). 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 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 the 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 20$^{th}$ 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 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 |

## Literature Cited

**Ackerman, J.T.; Adams, J.; Takekawa, J.Y.; Carter, H.R.; Whitworth, D.L.; Newman, S.H. 2004.** Effects of radio transmitters on the reproductive performance of Cassin's auklets. Wildlife Society Bulletin. 32: 1229–1241.

**Agee, J.K. 1993.** Fire ecology of Pacific Northwest forests. Washington, DC: Island Press. 493 p.

**Ainley, D.G.; Hyrenbach, D.K. 2010.** Top-down and bottom-up factors affecting seabird population trends in the California Current system (1985–2006). Progress in Oceanography. 84: 242–254.

**Ainley, D.G.; Sydeman, W.J.; Norton, J. 1995.** Upper-trophic level predators indicate interannual negative and positive anomalies in the California Current food web. Marine Ecology in Progress Series. 118: 69–79.

**Alegria, J.; Folliard, L.; Lint, J.; Madsen, S.; Max, T.; Webb, L. 2002.** Southwest Oregon inland survey assessment for marbled murrelets. Final report. Portland, OR: U.S. Fish and Wildlife Service. 17 p. On file with: U.S. Department of the Interior, Fish and Wildlife Service, Oregon State Office, 2600 SE 98th Ave, Suite 100, Portland, OR 97266.

**Baker, L.M.; Peery, M.Z.; Burkett, E.E.; Singer, S.W.; Suddjian, D.L.; Beissinger, S.R. 2006.** Nesting habitat characteristics of the marbled murrelet in central California redwood forests. Journal of Wildlife Management. 70(4): 939–946.

**Barbaree, B.A.; Nelson, S.K.; Dugger, B.D.; Roby, D.D.; Carter, H.R.; Whitworth, D.L.; Newman, S.H. 2014.** Nesting ecology of marbled murrelets at a remote mainland fjord in southeast Alaska. Condor. 116: 173–184.

**Batáry, P.; Báldi, A. 2004.** Evidence of an edge effect on avian nest success. Conservation Biology. 18: 389–400.

**Batchelder, H.P.; Barth, J.A.; Kosro, P.M.; Strub, P.T.; Brodeur, R.D.; Peterson, W.T.; Tynan, C.T.; Ohman, M.D.; Botsford, L.W.; Powell, T.M.; Schwing, F.B.; Ainley, D.G.; Mackas, D.L.; Hickey, B.M.; Ramp, S.R. 2002.** The GLOBEC Northeast Pacific California Current System Program. Oceanography. 15: 36–47.

**Battin, J. 2004.** When good animals love bad habitats: ecological traps and the conservation of animal populations. Conservation Biology. 18: 1482–1491.

**Becker, B.H.; Beissinger, S.R. 2006.** Centennial decline in the trophic level of an endangered seabird after fisheries decline. Conservation Biology. 20: 470–479.

**Becker, B.H.; Peery, M.Z.; Beissinger, S.R. 2007.** Ocean climate and prey availability affect the trophic level and reproductive success of the marbled murrelet, an endangered seabird. Marine Ecology Progress Series. 329: 267–279.

**Beissinger, S. 1995.** Population trends of the marbled murrelet projected from demographic analyses. In: Ralph, C.J.; Hunt, G.L.; Raphael, M.G.; Piatt, J.F., eds. Ecology and conservation of the marbled murrelet. Gen. Tech. Rep. PSW-152. Albany, CA: U.S. Department of Agriculture, Forest Service, Pacific Southwest Research Station: 385–393.

**Beissinger, S.R.; Peery, M.Z. 2007.** Reconstructing the historic demography of an endangered seabird. Ecology. 88: 296–305. doi:10.1890/06-0869.

**Bertram, D.F.; Drever, M.C.; McAllister, M.K.; Schroeder, B.K.; Lindsay, D.J.; Faust, D.A. 2015a.** Estimation of coast-wide population trends of marbled murrelets in Canada using a Bayesian hierarchical model. PLoS ONE. 10(8): e0134891. doi:10.1371/journal.pone.0134891.

**Bertram, D.F.; Harfenist, A.; Smith, B.D. 2005.** Ocean climate and El Niño impacts on survival of Cassin's auklets from upwelling and downwelling domains of British Columbia. Canadian Journal of Fisheries and Aquatic Sciences. 62: 2841–2853.

**Bertram, D.F.; MacDonald, C.A.; O'Hara, P.D.; Cragg, J.L.; Janssen, M.H.; McAdie, M.; Boyd, W.S. 2015b.** Marbled murrelet *Brachyramphus marmoratus* movements and marine habitat use near proposed tanker routes to Kitimat, BC, Canada. Marine Ornithology. 44: 3–9.

**Binford, L.C.; Elliot, B.G.; Singer, S.W. 1975.** Discovery of a nest and the downy young of the marbled murrelet. Wilson Bulletin. 87: 303–319.

**Bishop, M.A.; Watson, J.T.; Kuletz, K.; Morgan, T. 2014.** Pacific herring (*Clupea pallasii*) consumption by marine birds during winter in Prince William Sound, Alaska. Fisheries Oceanography. 1–13. doi:10.1111/fog.12073.

**Black, B.A.; Schroeder, I.D.; Sydeman, W.J.; Bograd, S.J.; Wells, B.K.; Schwing, F.B. 2011.** Winter and summer upwelling modes and their biological importance in the California Current Ecosystem. Global Change Biology. 17: 2536–2545.

**Bloxton, T.D.; Raphael, M.G. 2009.** Breeding ecology of the marbled murrelet in Washington state: 5-year project summary. Unpublished report. On file with: U.S. Department of Agriculture, Forest Service, Pacific Northwest Research Station, Portland, OR.

**Bradley, J.E.; Marzluff, J.M.; Thompson, F.R. 2003.** Rodents as nest predators: influences on predatory behavior and consequences to nesting birds. Auk. 120: 1180–1187.

**Bradley, R.W. 2002.** Breeding ecology of radio-marked marbled murrelets (*Brachyramphus marmoratus*) in Desolation Sound, British Columbia. Burnaby, BC: Simon Fraser University. 86 p. M.Sc. thesis.

**Burger, A.E. 2001.** Using radar to estimate populations and assess habitat associations of marbled murrelets. Journal of Wildlife Management. 65: 696–715.

**Burger, A.E. 2002.** Conservation assessment of marbled murrelets in British Columbia, a review of the biology, populations, habitat associations, and conservation. Tech. Report 387. Delta, BC, Canada: Canadian Wildlife Service. 194 p.

**Burger, A.E.; Chatwin, T.A.; Cullen, S.A.; Holmes, N.P.; Manley, I.A.; Mather, M.H.; Schroeder, B.K.; Steventon, J.D.; Duncan, J.E.; Arcese, P.; Selak, E. 2004.** Application of radar surveys in the management of nesting habitat of marbled murrelets *Brachyramphus marmoratus*. Marine Ornithology. 32: 1–11.

**Burger, A.E.; Manley, I.A.; Silvergieter, M.P.; Lank, D.B.; Jordan, K.M.; Bloxton, T.D.; Raphael, M.R. 2009.** Re-use of nest sites by marbled murrelets (*Brachyramphus marmoratus*) in British Columbia. Northwestern Naturalist. 90: 217–226.

**Burger, A.E.; Page, R.E. 2007.** The need for biological realism in habitat modeling: a reinterpretation of Zharikov et al. (2006). Landscape Ecology. 22: 1273-1281.

**Burger, A.E.; Ronconi, R.A.; Silvergieter, M.P.; Conroy, C.; Bahn, V.; Manley, I.A.; Cober, A.; Lank, D.B. 2010.** Factors affecting the availability of thick epiphyte mats and other potential nest platforms for marbled murrelets in British Columbia. Canadian Journal of Forest Research. 40: 727–746.

**Burkett, E.E. 1995.** Marbled murrelet food habits and prey ecology. In: Ralph, C.J.; Hunt, G.L.; Raphael, M.G.; Piatt, J.F., eds. Ecology and conservation of the marbled murrelet. Gen. Tech. Rep. PSW-152. Albany, CA: U.S. Department of Agriculture, Forest Service, Pacific Southwest Research Station: 223–246.

**Chen, J.; Saunders, S.C.; Crow, T.R.; Naiman, R.J.; Brosofske, K.D.; Mroz, G.D.; Brookshire, B.L.; Franklin, J.F. 1999.** Microclimate in forest ecosystem and landscape ecology. Bioscience. 49: 288–297.

**Cortese, L. 2011.** Picking patches: What is the utility of habitat fragmentation in determining habitat use by local populations of the marbled murrelet, *Brachyramphus marmoratus*? Burnaby, BC: School of Resource and Environmental Management, Simon Fraser University. 61 p. M.R.M. thesis.

**Committee on the Status of Endangered Wildlife in Canada [COSEWIC]. 2012.** COSEWIC assessment and status report on the marbled murrelet *Brachyramphus marmoratus* in Canada. Ottawa, ON: Government of Canada. 82 p. http://www.registrelep-sararegistry.gc.ca/document/default_e.cfm?documentID=2463. (8 August 2017).

**Cury, P.M.; Boyd, I.L.; Bonhommeau, S.; Anker-Nilssen, T.; Crawford, R.J.; Furness, R.W.; Mills, J.A.; Murphy, E.J.; Österblom, H.; Paleczny, M.; Piatt, J.F. 2011.** Global seabird response to forage fish depletion—one-third for the birds. Science. 334: 1703–1706.

**Dale, V.H.; Joyce, L.A.; McNulty, S.; Neilson, R.P.; Ayres, M.P.; Flannigan, M.D.; Hanson, P.J.; Irland, L.C.; Lugo, A.E.; Peterson, C.J.; Simberloff, D.; Swanson, F.J.; Stocks, B.J.; Wotton, M. 2001.** Climate change and forest disturbances: climate change can affect forests by altering the frequency, intensity, duration, and timing of fire, drought, introduced species, insect and pathogen outbreaks, hurricanes, windstorms, ice storms, or landslides. BioScience. 51: 723–734.

**Dalton, M.; Mote, P.W.; Snover, A.K. eds. 2013.** Climate change in the Northwest: implications for our landscapes, waters, and communities. Washington, DC: Island Press. 224 p.

**Davis, R.J.; Ohmann, J.L.; Kennedy, R.E.; Cohen, W.B.; Gregory, M.J.; Yang, Z.; Roberts, H.M.; Gray, A.N.; Spies, T.A. 2015.** Northwest Forest Plan—the first 20 years (1994–2013): status and trends of late-successional and old-growth forests. Gen. Tech. Rep. PNW-GTR-911. Portland, OR: U.S. Department of Agriculture, Forest Service, Pacific Northwest Research Station. 112 p.

**Day, R.H.; Nigro, D.A. 2000.** Feeding ecology of Kittlitz's and marbled murrelets in Prince William Sound, Alaska. Waterbirds. 23: 1–14.

**Di Lorenzo, E.; Miller, A. J.; Schneider, N.; McWilliams, J. C. 2005.** The warming of the California Current system: dynamics and ecosystem implications. Journal of Physical Oceanography. 35: 336–362.

**Doney, S.C.; Ruckelshaus, M.; Duffy, J.E.; Barry, J.P.; Chan, F.; English, C.A.; Talley, L.D. 2012.** Climate change impacts on marine ecosystems. Annual Review of Marine Science. 4: 11–37.

**Durant, J.M.; Hjermann, D.Ø.; Frederiksen, M.; Charrassin, J.B.; Le Maho, Y.; Sabarros, P.S.; Crawford, R.J.M.; Stenseth, N.C. 2009.** Pros and cons of using seabirds as ecological indicators. Climate Research. 39: 115–129.

**Endangered Species Act of 1973 [ESA]; 16 U.S.C. 1531–1536, 1538–1540.**

**Environment Canada. 2014.** Recovery strategy for the marbled murrelet (*Brachyramphus marmoratus*) in Canada. Species at risk act recovery strategy series. Ottawa, ON: Environment Canada. 49 p. http://www.sararegistry.gc.ca/virtual_sara/files/plans/rs_guillemot_marbre_marbled_murrelet_0614_e.pdf. (8 August 2017).

**Evans Mack, D.; Ritchie, W.P.; Nelson, S.; Kuo-Harrison, E.; Harrison, P.; Hamer, T.E., eds. 2003.** Methods for surveying marbled murrelets in forests: a revised protocol for land management and research. Pacific Seabird Group. https://www.pacificseabirdgroup.org/publications/PSG_TechPub2_MAMU_ISP.pdf. (8 August 2017).

**Faaborg, J. 2004.** Truly artificial nest studies. Conservation Biology. 18: 369–370.

**Fahrig, L. 1997.** Relative effects of habitat loss and fragmentation on population extinction. Journal of Wildlife Management. 61: 603–610.

**Falxa, G.; Baldwin, J.; Lance, M.; Lynch, D.; Nelson, S.K.; Pearson, S.F.; Raphael, M.G.; Strong, C.; Young, R. 2015.** Marbled murrelet effectiveness monitoring, Northwest Forest Plan: 2014 summary report. Portland, OR: Regional Ecosystem Office. 18 p. https://reo.gov/monitoring/reports/murrelet/MAMU_2014_NWFP%20EM_Annual%20Report_Aug2015.pdf. (8 August 2017).

**Falxa, G.A.; Raphael, M.G., tech. coords. 2016.** Northwest Forest Plan—the first 20 years (1994–2013): status and trend of marbled murrelet populations and nesting habitat. Gen. Tech. Rep. PNW-GTR-933. Portland, OR: U.S. Department of Agriculture, Forest Service, Pacific Northwest Research Station. 132 p.

**Falxa, G.A.; Raphael, M.G.; Strong, C.; Baldwin, J.; Lance, M.; Lynch, D.; Pearson, S.F.; Young, R.D. 2016.** Status and trend of marbled murrelet populations in the Northwest Forest Plan area. In: Falxa, G.A.; Raphael, M.G., tech. coords. Northwest Forest Plan—the first 20 years (1994–2013): status and trend of marbled murrelet populations and nesting habitat. Gen. Tech. Rep. PNW-GTR-933. Portland, OR: U.S. Department of Agriculture, Forest Service, Pacific Northwest Research Station: 1–36. Chapter 1.

**Feely, R.A.; Sabine, C.L.; Hernandez-Ayon, J.M.; Ianson, D.; Hales, B. 2008.** Evidence for upwelling of corrosive "acidified" water onto the continental shelf. Science. 320: 1490–1492.

**Forest Ecosystem Management Assessment Team [FEMAT]. 1993.** Forest ecosystem management: an ecological, economic, and social assessment. Portland, OR: U.S. Department of Agriculture; U.S. Department of the Interior [and others]. [Irregular pagination].

**Franklin, J.F.; Forman, R.T.T. 1987.** Creating landscape patterns by forest cutting: ecological consequences and principles. Landscape Ecology. 1: 5–18.

**Frederiksen, M.; Edwards, M.; Richardson, A.J.; Halliday, N.C.; Wanless, S. 2006.** From plankton to top predators: bottom-up control of a marine food web across four trophic levels. Journal of Animal Ecology. 75: 1259–1268.

**Frey, S.J.; Hadley, A.S.; Johnson, S.L.; Schulze, M.; Jones, J.A.; Betts, M.G. 2016.** Spatial models reveal the microclimatic buffering capacity of old-growth forests. Science Advances. 2(4): e1501392.

**Furness, R.W.; Tasker, M.L. 2000.** Seabird-fishery interactions: quantifying the sensitivity of seabirds to reductions in sandeel abundance, and identification of key areas for sensitive seabirds in the North Sea. Marine Ecology Progress Series. 202: 253–264.

**Gabriel, P.O.; Golightly, R. 2014.** Aversive conditioning of Steller's jays to improve marbled murrelet nest survival. Journal of Wildlife Management. 78: 894–903.

**Goldenberg, W.P. 2013.** Steller's jay space use and behavior in campground and non-campground sites within Redwood National and State Parks. Arcata, CA: Humboldt State University, Department of Wildlife. 36 p. M.S. thesis.

**Gratowski, H.J. 1956.** Windthrow around staggered settings in old-growth Douglas-fir. Forest Science. 2: 60–74.

**Grémillet, D.; Boulinier, T. 2009.** Spatial ecology and conservation of seabirds facing global climate change: a review. Marine Ecology in Progress Series. 391: 121–137.

**Gutowsky, S.; Janssen, M.H.; Arcese, P.; Kyser, T.K.; Ethier, D.; Wunder, M.B.; Bertram, D.F.; Tranquilla, L.M.; Lougheed, C.; Norris, D.R. 2009.** Concurrent declines in nestling diet quality and reproductive success of a threatened seabird over 150 years. Endangered Species Research. 9: 247–254.

**Hallowed, A.B.; Hare, S.R.; Wooster, W.S. 2001.** Pacific Basin climate variability and patterns of northeast Pacific marine fish production. Progress in Oceanography. 49: 257–282.

**Hamer, T.E.; Cummins, E.B. 1991.** Relationships between forest characteristics and use of inland sites by marbled murrelets in northwestern Washington. Olympia, WA: Washington Department of Fish and Wildlife. 47 p.

**Hamer, T.E.; Meekins, D.J. 1999.** Marbled murrelet nest site selection in relation to habitat characteristics in western Washington: 1998 report. Portland, OR: U.S. Department of the Interior, Fish and Wildlife Service. 26 p.

**Hamer T.E.; Nelson, S.K. 1995.** Characteristics of marbled murrelet nest trees and nesting stands. In: Ralph, C.J; Hunt, G.L.; Raphael, M.G.; Piatt, J.F., eds. Ecology and conservation of the marbled murrelet, Gen. Tech. Rep. PSW-GTR-152. Albany, CA: U.S. Department of Agriculture, Forest Service, Pacific Southwest Experiment Station. 420 p.

**Hamer, T.E.; Ritchie, W.P.; Cummins, E.B.; Turley, C.W. 1994.** Forest habitat relationships of marbled murrelets in western Washington. 51 p. Unpublished report. On file with: Washington Department of Fish and Wildlife, Nongame Division, 1111 Washington St. SE, Olympia, WA 98501.

**Harper, K.A.; Macdonald, S.E.; Burton, P.J.; Chen, J.; Brosofske, K.D.; Saunders, S.C.; Euskirchen, E.S.; Roberts, D.A.R.; Jaiteh, M.S.; Esseen, P.A. 2005.** Edge influence on forest structure and composition in fragmented landscapes. Conservation Biology. 19: 768–782.

**Hazen, E.L.; Jorgensen, S.; Rykaczewski, R.R.; Bograd, S.J.; Foley, D.G.; Jonsen, I.D.; Shaffer, S.A.; Dunne, J.P.; Costa, D.P.; Crowder, L.B.; Block, B.A. 2012.** Predicted habitat shifts of Pacific top predators in a changing climate. Nature Climate Change. 3(3): 234. doi:10.1038/NCLIMATE1686.

**Hébert P.N.; Golightly, R.T. 2006.** Movements, nesting, and response to anthropogenic disturbance of marbled murrelets (*Brachyramphus marmoratus*) in Redwood National and State Parks, California. Report FHWA/CA/IR-2006/04. Sacramento, CA: California Department of Transportation. 321 p. http://www.dot.ca.gov/newtech/researchreports/2002-2006/2006/fhwa-ca-ir-2006-04.pdf. (8 August 2017).

**Hébert, P.N.; Golightly, R.T. 2007.** Observations of predation by corvids at a marbled murrelet nest. Journal of Field Ornithology. 78: 221–224.

**Henry, R.W.; Tyler, W.B. 2014.** 2014 at-sea monitoring of marbled murrelets in central California. Summary presented to the Cosco Busan Trustee Council, Nov. 2014. 5 p.

**Henry, R.W.; Tyler, W.B.; Peery, M.Z. 2012.** Abundance and productivity of marbled murrelets off central California during the 2010 and 2011 breeding seasons. Final report submitted to the Luckenbach and Command Oil Spill trustees. 17 p. https://nrm.dfg.ca.gov/FileHandler.ashx?DocumentID=57216. (8 August 2017).

**Hodder, J.; Graybill, M.R. 1985.** Reproduction and survival of seabirds in Oregon during the 1982–1983 El Niño. Condor. 87: 535–541.

**Hunter, J.E.; Schmidt, K.N.; Stauffer, H.B.; Miller, S.L.; Ralph, C.J.; Roberts, L. 1998.** Status of the marbled murrelet in the inner north coast ranges of California. Northwestern Naturalist. 79(3): 92–103.

**Huot, Y.; Babin, M.; Bruyant, F.; Grob, C.; Twardowski, M.S.; Claustre, H. 2007.** Does chlorophyll a provide the best index of phytoplankton biomass for primary productivity studies? Biogeosciences Discussions. 4: 707–745.

**Intergovernmental Panel on Climate Change [IPCC]. 2014.** Climate change 2014: synthesis report. Contribution of working groups I, II and III to the Fifth Assessment Report of the Intergovernmental Panel on Climate Change. Geneva, Switzerland. 151 p.

**Irvine, J.R.; Crawford, W.R. 2012.** State of physical, biological, and selected fishery resources of Pacific Canadian marine ecosystems in 2011. Canadian Science Advisory Secretariat Research Document 2012/072. Ottawa, ON: Government of Canada, Department of Fisheries and Oceans. http://waves-vagues.dfo-mpo.gc.ca/Library/347247.pdf. (8 August 2017).

**Johnstone, J.A.; Dawson, T.E. 2010.** Climatic context and ecological implications of summer fog decline in the coast redwood region. Proceedings of the National Academy of Sciences of the United States of America. 107: 4533–4538.

**Jones, J. 2001.** Habitat selection in avian ecology: a critical review. Auk. 118: 557–562.

**Jones, T.L.; Miller, A.; Senichenko, G. 2006.** Marbled murrelet inventory in Elk Creek, Chilliwack Forest District, British Columbia, Canada. Unpublished report. On file with: Western Canada Wilderness Committee, 46 East 6th Avenue, Vancouver, BC V5T 1J4.

**Kissling, M.L.; Gende, S.M.; Lewis, S.B.; Lukacs, P.M. 2015.** Reproductive performance of Kittlitz's murrelet in a glaciated landscape, Icy Bay, Alaska, USA. Condor. 117: 237–248.

**Kitaysky, A.S.; Piatt, J.F.; Hatch, S.A.; Kitaiskaia, A.V.; Benowitz-Fredericks, Z.M. 2010.** Food availability and population processes: severity of nutritional stress during reproduction predicts survival of long-lived seabirds. Functional Ecology. 24: 625–637.

**Kliejunas, J.T.; Geils, B.W.; Glaeser, J.M.; Goheen, E.M.; Hennon, P.; Kim, M.; Kope, H.; Stone, J.; Sturrock, R.; Frankel, S.J. 2009.** Review of literature on climate change and forest diseases of western North America. Gen. Tech. Rep. PSW-GTR-225. Albany, CA: U.S. Department of Agriculture, Forest Service, Pacific Southwest Research Station. 54 p.

**Kremsater, L.; Bunnell, F.L. 1999.** Edge effects: theory, evidence and implications to management of western North American forests. In: Rochelle, J.L.; Lehmann, L.A.; Wisniewski, J., eds. Forest fragmentation: wildlife and management implications. Leiden, Netherlands: Brill: 117–153.

**Kuletz, K.J. 2005.** Foraging behavior and productivity of a non-colonial seabird, the marbled murrelet (*Brachyramphus marmoratus*), relative to prey and habitat. Victoria, BC: University of Victoria. 195 p. Ph.D. dissertation.

**Kuletz, K.J.; Nations, C.S.; Manly, B.; Allyn, A.; Irons, D.B.; McKnight, A. 2011.** Distribution, abundance, and population trends of the Kittlitz's murrelet *Brachyramphus brevirostris* in Prince William Sound, Alaska. Marine Ornithology. 39: 97–109.

**Lindenmayer, D.B.; Fischer, J. 2007.** Tackling the habitat fragmentation panchreston. Trends in Ecology and Evolution. 22: 127–132.

**Littell, J.S.; McKenzie, D.; Peterson, D.L.; Westerling, A.L. 2009.** Climate and wildfire area burned in western U.S. ecoprovinces, 1916–2003. Ecological Applications. 19: 1003–1021.

**Littell, J.S.; Oneil, E.E.; McKenzie, D.; Hicke, J.A.; Lutz, J.A.; Norheim, R.A.; Elsner, M.M. 2010.** Forest ecosystems, disturbance, and climatic change in Washington State, USA. Climatic Change. 102: 129–158.

**Loehman, R.; Anderson, G. 2009.** Understanding the science of climate change: talking points—impacts to western mountains and forests. Natural Resource Report NPS/NRPC/NRR—2009/090. Fort Collins, CO: U.S. Department of the Interior, National Park Service. 32 p.

**Lorenz, T.J.; Raphael, M.R.; Bloxton, T.D.; Cunningham, P.G. 2017.** Low breeding propensity and wide-ranging movements by marbled murrelets in Washington. Journal of Wildlife Management. 81: 306–321.

**Lougheed, C. 1999.** Breeding chronology, breeding success, distribution and movements of marbled murrelets (*Brachyramphus marmoratus*) in Desolation Sound, British Columbia. CWS Tech. Report 352. Ottawa, ON: Environment Canada, Canadian Wildlife Service. 89 p.

**Luginbuhl, J.M.; Marzluff, J.M.; Bradley, J.E.; Raphael, M.G.; Varland, D.E. 2001.** Corvid survey techniques and the relationship between corvid relative abundance and nest predation. Journal of Field Ornithology. 72: 556–572.

**Lynch, D.; Falxa, G.; Baldwin, J.; Lance, M.M.; Nelson, S.K.; Pearson, S.F.; Raphael, M.G.; Strong, C.; Young, R. 2016.** Marbled murrelet effectiveness monitoring, Northwest Forest Plan: 2015 summary report. Portland, OR: Regional Ecosystem Office. 19 p. https://reo.gov/monitoring/reports/murrelet/Lynch%20etal%202016_NWFPEM%20Marbled%20Murrelet%202015%20Summary%20Report.pdf. (8 August 2017).

**Madsen, S.; Evans, D.; Hamer, T.; Henson, P.; Miller, S.; Nelson, S.K.; Roby, D.; Stapanian, M. 1999.** Marbled murrelet effectiveness monitoring plan for the Northwest Forest Plan. Gen. Tech. Rep. PNW-GTR-439. Portland, OR: U.S. Department of Agriculture, Forest Service, Pacific Northwest Research Station. 51 p.

**Malt, J.M.; Lank, D.B. 2007.** Temporal dynamics of edge effects on nest predation risk for the marbled murrelet. Biological Conservation. 140: 160–173.

**Malt, J.M.; Lank, D.B. 2009.** Marbled murrelet nest predation risk in managed forest landscapes: dynamic fragmentation effects at multiple scales. Ecological Applications. 19: 1274–1287.

**Manley, I.A. 1999.** Behavior and habitat selection of marbled murrelets nesting on the Sunshine Coast. Burnaby, BC: Simon Fraser University. 326 p. M.Sc. thesis.

**Manley, I.A.; Nelson, S.K. 1999.** Habitat characteristics associated with nest success and predation at marbled murrelet tree nests (abstract). Pacific Seabirds. 26: 40.

**Mantua, N.J.; Hare, S.R.; Zhang, Y.; Wallace, J.M.; Francis, R.C. 1997.** A Pacific interdecadal climate oscillation with impacts on salmon production. Bulletin of the American Meteorological Society. 78: 1069–1079.

**Marzluff, J.M.; Neatherlin, E. 2006.** Corvid response to human settlements and campgrounds: causes, consequences, and challenges for conservation. Biological Conservation. 130: 301–314.

**Marzluff, J.M.; Raphael, M.G.; Sallabanks, R. 2000.** Understanding the effects of forest management on avian species. Wildlife Society Bulletin. 28: 1132–1143.

**Marzluff, J.M.; Restani, M. 1999.** The effects of forest fragmentation on avian nest predation. In: Rochelle, J.L.; Lehmann, L.A.; Wisniewski, J., eds. Forest fragmentation: wildlife and management implications. Leiden, Netherlands: Brill: 155–169.

**Masselink, M.N.M. 2001.** Responses of Steller's jays to forest fragmentation on southwest Vancouver Island and potential impacts on marbled murrelets. Victoria, BC: University of Victoria, Department of Biology. 138 p. M.S. thesis.

**McShane, C.; Hamer, T.; Carter, H.; Swartzman, G.; Friesen, V.; Ainley, D.; Tressler, R.; Nelson, K.; Burger, A.; Spear, L.; Mohagen, T.; Martin, R.; Henkel, L.; Prindle, K.; Strong, C.; Keany, J. 2004.** Evaluation report for the 5-year status review of the marbled murrelet in Washington, Oregon, and California. Report prepared for the U.S. Fish and Wildlife Service, Region 1. Seattle, WA: EDAW, Inc. 272 p. https://www.fws.gov/arcata/es/birds/MM/documents/mm5yr_rpt_final_web.pdf. (8 August 2017).

**Melillo, J.M.; Richmond, T.C.; Yohe, G.W., eds. 2014.** Climate change impacts in the United States: the third national climate assessment. Washington, DC: U.S. Global Change Research Program. 841 p. doi:10.7930/J0Z31WJ2.

**Mestas-Nunez, A.M.; Miller, A.J. 2006.** Interdecadal variability and climate change in the eastern tropical Pacific: a review. Progress in Oceanography. 69: 267–284.

**Meyer, C.B.; Miller, S.L. 2002.** Use of fragmented landscapes by marbled murrelets for nesting in southern Oregon. Conservation Biology. 16: 755–766.

**Meyer, C.B.; Miller, S.L.; Ralph, C.J. 2004.** Stand-scale habitat associations across a large geographic region of an old-growth specialist, the marbled murrelet. Wilson Bulletin. 116: 197–210.

**Millar, C.; Neilson, R.; Bachelet, D.; Drapek, R.; Lenihan, J. 2006.** Climate change at multiple scales. In: Salwasser, H.; Cloughesy, M., eds. Forests, carbon and climate change: a synthesis of science findings. Portland, OR: Oregon Forest Resources Institute: 31–60. Chapter 3. https://www.treesearch.fs.fed.us/pubs/31819. (8 August 2017).

**Miller, S.L.; Meyer, C.B.; Ralph, C.J. 2002.** Land and seascape patterns associated with marbled murrelet abundance offshore. Waterbirds. 25: 100–108.

**Miller, S.L.; Raphael, M.G.; Falxa, G.A.; Strong, C.; Baldwin, J.; Bloxton, T.; Galleher, B.M.; Lance, M.; Lynch, D.; Pearson, S.F.; Ralph, C.J.; Young, R.D. 2012.** Recent population decline of the marbled murrelet in the Pacific Northwest. Condor. 114: 771–781.

**Mitchell, S.J.; Hailemariam, T.; Kulis, Y. 2001.** Empirical modeling of cutblock edge windthrow risk on Vancouver Island, Canada, using stand level information. Forest Ecology and Management. 154: 117–130.

**Moeur, M.; Ohmann, J.L.; Kennedy, R.E.; Cohen, W.B.; Gregory, M.J.; Yang, Z.; Roberts, H.M.; Spies, T.A.; Fiorella, M. 2011.** Northwest Forest Plan—the first 15 years (1994–2008): status and trends of late-successional and old-growth forests. Gen. Tech. Rep. PNW-GTR-853. Portland, OR: U.S. Department of Agriculture, Forest Service, Pacific Northwest Research Station. 48 p.

**Moeur, M.; Spies, T.A.; Hemstrom, M.A.; Alegria, J.; Browning, J.; Cissel, J.H.; Cohen, W.B.; Demeo, T.E.; Healy, S.; Warbington, R. 2005.** Northwest Forest Plan—the first 10 years (1994–2003): status and trends of late-successional and old-growth forests. Gen Tech. Rep. PNW-GTR-646. Portland, OR: U.S. Department of Agriculture, Forest Service, Pacific Northwest Research Station. 142 p.

**Montevecchi, W.A.; Myers, R.A. 1997.** Centurial and decadal oceanographic influences on changes in northern gannet populations and diets in the northwest Atlantic: implications for climate change. ICES Journal of Marine Science. 54: 608–614.

**Mote, P.W.; Parson, E.A.; Hamlet, A.F.; Keeton, W.S.; Lettenmaier, D.; Mantua, N.; Miles, E.L.; Peterson, D.W.; Peterson, D.L.; Slaughter, R.; Snover, A.K. 2003.** Preparing for climate change: the water, salmon, and forests of the Pacific Northwest. Climate Change. 61: 45–88.

**Mote, P.W.; Salathé, E.P., Jr. 2010.** Future climate in the Pacific Northwest. Climatic Change. doi:10.1007/s10584-010-9848-z.

**Muth, C.C.; Bazzaz, F.A. 2002.** Tree canopy displacement at forest gap edges. Canadian Journal of Forest Research. 32: 247–254.

**Naslund, N.L.; Kuletz, K.J.; Cody, M.B.; Marks, D.K. 1995.** Tree and habitat characteristics and reproductive success at marbled murrelet tree nests in Alaska. Northwestern Naturalist. 76: 12–25.

**Nelson, S.K. 1997.** Marbled murrelet *Brachyramphus marmoratus*. In: Poole, A.; Gill, F., eds. The Birds of North America, No. 276. Philadelphia, PA: The Academy of Natural Sciences; Washington, DC: The American Ornithologists' Union.

**Nelson, S.K.; Barbaree, B.A.; Dugger, B.D.; Newman, S.H. 2010.** Marbled murrelet breeding ecology at Port Snettisham, southeast Alaska, in 2005–2008. Unpublished report prepared for Alaska Department of Fish and Game. Corvallis, OR: Oregon State University, and New York: Wildlife Trust.

**Nelson, S.K.; Hamer, T.E. 1995.** Nest success and the effects of predation on marbled murrelets. In: Ralph, C.J.; Hunt, G.L., Jr.; Raphael, M.G.; Piatt, J.F., eds. Ecology and conservation of the marbled murrelet. Gen. Tech. Rep. PSW-GTR-152. Albany, CA: U.S. Department of Agriculture, Forest Service, Pacific Southwest Research Station: 89–97.

**Nelson, S.K.; Wilson, A.K. 2002.** Marbled murrelet habitat characteristics on state lands in western Oregon. Corvallis, OR: Oregon Cooperative Fish and Wildlife Research Unit. 151 p.

**Norris, D.R.; Arcese, P.; Preikshot, D.; Bertram, D.F.; Kyser, T.K. 2007.** Diet reconstruction and historic population dynamics in a threatened seabird. Journal of Applied Ecology. 44: 875–884.

**Ohmann, J.L.; Gregory, M.J. 2002.** Predictive mapping of forest composition and structure with direct gradient analysis and nearest-neighbor imputation in coastal Oregon, U.S.A. Canadian Journal of Forest Research. 32: 725–741.

**Peery, M.Z. 2004.** Ecology of marbled murrelets in central California. Berkeley, CA: University of California. 210 p. Ph.D. dissertation.

**Peery, M.Z.; Becker, B.H.; Beissinger, S.R. 2006.** Combining demographic and count-based approaches to identify source-sink dynamics of a threatened seabird. Ecological Applications. 16: 1516–1528.

**Peery, M.Z.; Beissinger, S.R.; Burkett, E.; Newman, S.H. 2006b.** Local survival of marbled murrelets in central California: roles of oceanographic processes, sex, and radiotagging. Journal of Wildlife Management. 70: 78–88.

**Peery, M.Z.; Beissinger, S.R.; Newman, S.H.; Burkett, E.B.; Williams, T.D. 2004.** Applying the declining population paradigm: diagnosing causes of poor reproduction in the marbled murrelet. Conservation Biology. 18: 1088–1098.

**Peery, M.Z.; Hall, L.A.; Sellas, A.; Beissinger, S.R.; Moritz, C.; Berube, M.; Raphael, M.G; Nelson, S.K.; Golightly, R.T.; McFarlane-Tranquilla, L.; Newman, S.; Palsbøll, P.J. 2007.** Genetic analyses of historic and modern marbled murrelets suggest decoupling of migration and gene flow after habitat fragmentation. Proceedings of the Royal Society B. 277: 697–706.

**Peery, M.Z.; Henry, R.W. 2010.** Recovering marbled murrelets via corvid management: a population viability analysis approach. Biological Conservation. 143: 2414–2424.

**Perry, D.A. 1995.** Status of forest habitat of the marbled murrelet. In: Ralph, C.J.; Hunt, G.L.; Raphael, M.G.; Piatt, J.F., eds. Ecology and conservation of the marbled murrelet. Gen. Tech. Rep. PSW-152. Albany, CA: U.S. Department of Agriculture, Forest Service, Pacific Southwest Research Station: 381–384.

**Piatt, J.F.; Kuletz, K.J.; Burger, A.E.; Hatch, S.A.; Friesen, V.L; Birt, T.P.; Arimitsu, M.L.; Drew, G.S.; Harding, A.M.A.; Bixler, K.S. 2007.** Status review of the marbled murrelet (*Brachyramphus marmoratus*) in Alaska and British Columbia. Open-File Report 2006-1387. Reston, VA: U.S. Department of the Interior, Geological Survey. 258 p. https://pubs.er.usgs.gov/publication/ofr20061387. (8 August 2017).

**Plissner, J.H.; Cooper, B.A.; Day, R.H.; Sanzenbacher, P.M.; Burger, A.E.; Raphael, M.G. 2015.** A review of marbled murrelet research related to nesting habitat use and nest success. Salem, OR: Oregon Department of Forestry. http://www.oregon.gov/ODF/Documents/WorkingForests/ReviewofMAMUResearchRelatedToNestingHabitatUseandNestSuccess.pdf. (8 August 2017).

**Quinlan, S.E.; Hughes, J.H. 1990.** Location and description of a marbled murrelet tree nest site in Alaska. Condor. 92: 1068–1073.

**Ralph, C.J.; Hunt, G.L.; Raphael, M.G.; Piatt, J.F., eds. 1995a.** Ecology and conservation of the marbled murrelet. Gen. Tech. Rep. PSW-GTR-152. Albany, CA: U.S. Department of Agriculture, Forest Service, Pacific Southwest Experiment Station. 420 p.

**Ralph, C.J.; Hunt, G.L., Jr.; Raphael, M.G.; Piatt, J.F. 1995b.** Ecology and conservation of the marbled murrelet in North America: an overview. In: Ralph, C.J.; Hunt, G.L., Jr.; Raphael, M.G.; Piatt, J.F., eds. Ecology and conservation of the marbled murrelet. Gen. Tech. Rep. PSW-GTR-152. Albany, CA: U.S. Department of Agriculture, Forest Service, Pacific Southwest Research Station: 3–22.

**Raphael, M.G. 2006.** Conservation of listed species: the northern spotted owl and marbled murrelet. In: Haynes, R.W.; Bormann, B.T.; Lee, D.C.; Martin, J.R., tech. eds. Northwest Forest Plan—the first 10 years (1994–2003): synthesis of monitoring and research results. Gen. Tech. Rep. PNW-GTR-651. Portland, OR: U.S. Department of Agriculture, Forest Service, Pacific Northwest Research Station: 117–144.

**Raphael, M.G.; Baldwin, J.; Falxa, G.A.; Huff, M.H.; Lance, M.; Miller, S.L.; Pearson, S.F.; Ralph, C.J.; Strong, C.; Thompson, C. 2007.** Regional population monitoring of the marbled murrelet: field and analytical methods. Gen. Tech. Rep. PNW-GTR-716. Portland, OR: U.S. Department of Agriculture, Forest Service, Pacific Northwest Research Station. 70 p.

**Raphael, M.G.; Evans Mack, D.; Cooper, B.A. 2002a.** Landscape-scale relationships between abundance of marbled murrelets and distribution of nesting habitat. Condor. 104: 331–342.

**Raphael, M.G.; Evans Mack, D.; Marzluff, J.M.; Luginbuhl, J. 2002b.** Effects of forest fragmentation on populations of the marbled murrelet. Studies in Avian Biology. 25: 221–235.

**Raphael, M.G.; Galleher, B.M.; Huff, M.H.; Miller, S.L.; Nelson, S.K.; Young, R.D. 2006.** Spatially explicit estimates of potential nesting habitat for the marbled murrelet. In: Huff, M., tech. coord. Northwest Forest Plan—the first 10 years (1994–2003): status and trend of populations and nesting habitat for the marbled murrelet. Gen. Tech. Rep. PNW-GTR-650. Portland, OR: U.S. Department of Agriculture, Forest Service, Pacific Northwest Research Station: 97–146. Chapter 5.

**Raphael, M.G.; Falxa, G.A.; Lynch, D.; Nelson, S.K.; Pearson, S.F.; Shirk, A.J.; Young, R.D. 2016a.** Status and trend of nesting habitat for the marbled murrelet under the Northwest Forest Plan. In: Falxa, G.A.; Raphael, M.G., tech. coords. Northwest Forest Plan—the first 20 years (1994–2013): status and trend of marbled murrelet populations and nesting habitat. Gen. Tech. Rep. PNW-GTR-933. Portland, OR: U.S. Department of Agriculture, Forest Service, Pacific Northwest Research Station: 37–94. Chapter 2.

**Raphael, M.G.; Shirk, A.J.; Falxa, G.A.; Pearson, S.F. 2015.** Habitat associations of marbled murrelets during the nesting season in nearshore waters along the Washington to California coast. Journal of Marine Systems. 146: 17–25.

**Raphael, M.G.; Shirk, A.J.; Falxa, G.A.; Lynch, D.; Nelson, S.K.; Pearson, S.F.; Strong, C.; Young, R.D. 2016b.** Factors influencing status and trend of marbled murrelet populations: an integrated perspective. In: Falxa, G.A.; Raphael, M.G., tech. coords. Northwest Forest Plan—the first 20 years (1994–2013): status and trend of marbled murrelet populations and nesting habitat. Gen. Tech. Rep. PNW-GTR-933. Portland, OR: U.S. Department of Agriculture, Forest Service, Pacific Northwest Research Station: 95–120. Chapter 3.

**Reese, D.C.; Brodeur, R.D. 2006.** Identifying and characterizing biological hotspots in the northern California Current. Deep Sea Research Part II: Topical Studies in Oceanography. 53: 291–314.

**Ritchie, W.P.; Rodrick, E. 2002.** Marbled murrelet *Brachyramphus marmoratus*. In: Priority habitats and species recommendations, Vol. IV: Birds. Olympia, WA: Washington Department of Fish and Wildlife: 4.1–4.23.

**Rodway, M.S. 1990.** Status report on the marbled murrelet in Canada. Ottawa, ON: Government of Canada, Committee on the Status of Endangered Wildlife in Canada. 64 p.

**Rodway, M.S.; Regehr, H.M. 2002.** Inland activity and forest structural characteristics as indicators of marbled murrelet nesting habitat in Clayoquot Sound. In: Burger, A.E.; Chatwin, T.A., eds. Multi-scale studies of populations, distribution and habitat associations of marbled murrelets in Clayoquot Sound, British Columbia. Victoria, BC: Government of British Columbia, Ministry of Water, Land, and Air Protection: 57–87.

**Ronconi, R.A. 2008.** Patterns and processes of marine habitat selection: foraging ecology, competition and coexistence among coastal seabirds. Victoria, BC: University of Victoria. 214 p. Ph.D. dissertation.

**Ronconi, R.A.; Burger, A.E. 2008.** Limited foraging flexibility: increased foraging effort by a marine predator does not buffer against scarce prey. Marine Ecology Progress Series. 366: 245–258.

**Rucklehaus, M.H.; McClure, M.M., tech. coords. 2007.** Sound science: synthesizing ecological and socioeconomic information about the Puget Sound ecosystem. Seattle, WA: U.S. Department of Commerce, National Oceanic & Atmospheric Administration, Northwest Fisheries Science Center. 93 p.

**Ryder, G.R.; Campbell, R.W.; Carter, H.R.; Sealy, S.G. 2012.** Earliest well-described tree nest of the marbled murrelet: Elk Creek, British Columbia, 1955. Wildlife Afield. 9(1): 49–58.

**Schmidt, K.N.; Hunter, J.E.; Stauffer, H.B.; Miller, S.L.; Ralph, C.J.; Roberts, L. 2000.** Status and distribution of the marbled murrelet in interior northwestern California: final report. Eureka, CA: U.S. Department of Agriculture, Forest Service, Six Rivers National Forest. 28 p.

**Schrimpf, M.B.; Parrish, J.K.; Pearson, S.F. 2012.** Trade-offs in prey quality and quantity revealed through the behavioral compensation of breeding seabirds. Marine Ecology Progress Series. 460: 247–259.

**Schwing, F.F.; Murphreeb, T.; deWitta, L.; Green, P.M. 2002.** The evolution of oceanic and atmospheric anomalies in the northeast Pacific during the El Niño and La Niña events of 1995–2001. Progress in Oceanography. 54: 459–491.

**Sergeant, C.J.; Hoekman, S.T.; Johnson, W.F. 2015.** Monitoring Kittlitz's and marbled murrelets in Glacier Bay National Park and Preserve: 2015 annual report. Natural Resource Report NPS/SEAN/NRR—2015/1076. Fort Collins, CO: U.S. Department of the Interior, National Park Service. 15 p.

**Sherich, K.; Pocewicz, A.; Morgan, P. 2013.** Canopy characteristics and growth rates of ponderosa pine and Douglas-fir at long-established forest edges. Canadian Journal of Forest Research. 37: 2096–2105.

**Sinton, D.S.; Jones, J.A.; Ohmann, J.L.; Swanson, F.J. 2000.** Windthrow disturbance, forest composition, and structure in the Bull Run basin, Oregon. Ecology. 81: 2539–2556.

**Strong, C.S. 2003.** Decline of the marbled murrelet population on the central Oregon coast during the 1990s. Northwestern Naturalist. 84: 31–37.

**Strong, C.S.; Keitt, B.S.; McIver, W.R.; Palmer, C.J.; Gaffney, I. 1995.** Distribution and population estimates of marbled murrelets at sea in Oregon during the summers of 1992 and 1993. In: Ralph, C.J.; Hunt, G.L.; Raphael, M.G.; Piatt, J.F., eds. Ecology and conservation of the marbled murrelet. Gen. Tech. Rep. PSW-152. Albany, CA: U.S. Department of Agriculture, Forest Service, Pacific Southwest Research Station: 339–352.

**Suryan, R.S.; Santora, J.A.; Sydeman, W.J. 2012.** New approach for using remotely sensed chlorophyll a to identify seabird hotspots. Marine Ecology Progress Series. 451: 213–225.

**Sydeman, W.J.; Bradley, R.W.; Warzybok, P.; Abraham, C.L.; Jahncke, J.; Hyrenbach, K.D.; Kousky, V.; Hipfner, J.M.; Ohman, M.D. 2006.** Planktivorous auklet *Ptychoramphus aleuticus* responses to ocean climate, 2005. Unusual atmospheric blocking? Geophysical Research Letters. 33: L22S09. doi:10.1029/2006GL026736.

**Sydeman, W.J.; García-Reyes, M.; Schoeman, D.S.; Rykaczewski, R.R.; Thompson, S.A.; Black, B.A.; Bograd, S.J. 2014.** Climate change and wind intensification in coastal upwelling ecosystems. Science. 345: 77–80.

**Sydeman, W.J.; Mills, K.L.; Santora, J.A.; Thompson, S.A.; Bertram, D.F.; Morgan, K.H.; Wells, B.K.; Hipfner, J.M.; Wolf, S.G. 2009.** Seabirds and climate in the California Current—a synthesis of change. CalCOFI Reports. 50: 82–104.

**Thayer, J.A.; Sydeman, W.J. 2007.** Spatio-temporal variability in prey harvest and reproductive ecology of a piscivorous seabird, *Cerorhinka monocerata*, in an upwelling system. Marine Ecology Progress Series. 329: 253–265.

**Thomas, J.W.; Forsman, E.D.; Lint, J.B.; Meslow, E.C.; Noon, B.R.; Verner, J. 1990.** A conservation strategy for the northern spotted owl: report to the Interagency Scientific Committee to address the conservation of the northern spotted owl. Portland, OR: U.S. Department of Agriculture, Forest Service; U.S. Department of the Interior, Bureau of Land Management, Fish and Wildlife Service, National Park Service. 427 p.

**Trenberth, K.E. 1997.** The definition of El Nino. Bulletin of the American Meteorological Society. 78: 2771–2777.

**U.S. Department of Agriculture, Forest Service; U.S. Department of the Interior, Bureau of Land Management [USDA and USDI]. 1994a.** Final supplemental environmental impact statement on management of habitat for late-successional and old-growth forest related species within the range of the northern spotted owl. Portland, OR. 526 p.

**U.S. Department of Agriculture, Forest Service; U.S. Department of the Interior, Bureau of Land Management [USDA and USDI]. 1994b.** Record of decision for amendments to Forest Service and Bureau of Land Management planning documents within the range of the northern spotted owl. Standards and guidelines for management of habitat for late-successional and old-growth forest related species within the range of the northern spotted owl. [Place of publication unknown]. 74 p.

**U.S. Department of the Interior, Bureau of Land Management [USDI BLM]. 2016.** Northwestern and coastal Oregon record of decision and approved resource management plan. Portland, OR: Oregon State Office. 308 p. https://www.blm.gov/or/plans/rmpswesternoregon/files/rod/Northwestern_and_Coastal_Oregon_ROD_RMP.pdf. (17 August 2017).

**U.S. Department of the Interior, Fish and Wildlife Service [USFWS]. 1997.** Recovery plan for the threatened marbled murrelet (*Brachyramphus marmoratus*) in Washington, Oregon, and California. Portland, OR: Pacific Region. 202 p.

**U.S. Department of the Interior, Fish and Wildlife Service [USFWS]. 2009.** Marbled murrelet (*Brachyramphus marmoratus*) 5-year review. Lacey, WA: Washington State Office. 105 p. http://www.fws.gov/wafwo/species/Fact%20sheets/5%20Year%20Review%202009.pdf. (8 August 2017).

**Van Mantgen, P.J.; Stephenson, N.L.; Byrne, J.C.; Daniels, L.D.; Franklin, J.F.; Fule, P.Z.; Harmon, M.E.; Larson, A.J.; Smith, J.M.; Taylor, A.H.; Veblen, T.T. 2009.** Widespread increase of tree mortality rates in the Western United States. Science. 323: 521–524.

**Van Rooyen, J.C.; Malt, J.M.; Lank, D.B. 2011.** Relating microclimate to epiphyte availability: edge effects on nesting habitat availability for the marbled murrelet. Northwest Science. 85: 549–561.

**Vigallon, S.M.; Marzluff, J.M. 2005.** Is nest predation by Steller's jays (*Cyanocitta stelleri*) incidental or the result of a specialized search strategy? Auk. 122: 36–49.

**Ware, D.M.; Thomson, R.E. 2005.** Bottom-up ecosystem trophic dynamics determine fish production in the northeast Pacific. Science. 308. 1280–1284.

**Whitworth, D.L.; Nelson, S.K.; Newman, S.H.; Van Vliet, G.B.; Smith, W.P. 2000.** Foraging distances of radio-marked marbled murrelets from inland areas in southeast Alaska. Condor. 102: 452–456.

**Wilk, R.J.; Raphael, M.G.; Bloxton, T. 2016.** Nesting habitat characteristics of marbled murrelets occurring in near-shore waters of the Olympic Peninsula, Washington. Journal of Field Ornithology. 87: 162–175.

**Witt, J.W. 1998a.** Distribution of the marbled murrelet in southwestern Oregon. Northwest Science. 72: 96–101.

**Witt, J.W. 1998b.** Notes on activity and characteristics of an inland marbled murrelet nest site in Douglas County, Oregon. Northwestern Naturalist. 79: 27–32.

**Yen, P.P.W.; Huettmann, F.; Cooke, F. 2004.** A large-scale model for the at-sea distribution and abundance of marbled murrelets (*Brachyramphus marmoratus*) during the breeding season in coastal British Columbia, Canada. Ecological Modeling. 171: 395–413.

**Zharikov, Y.; Lank, D.; Cooke, F. 2007.** Influence of landscape pattern on breeding distribution and success in a threatened alcid, the marbled murrelet: model transferability and management implications. Journal of Applied Ecology. 44: 748–759.

**Zharikov, Y.; Lank, D.B.; Huettmann, F.; Bradley, R.W.; Parker, N.; Yen, P.P-W.; McFarlane Tranquilla, L.A.; Cooke, F.C. 2006.** Habitat selection and breeding success in a forest-nesting alcid, the marbled murrelet, in two landscapes with differing degrees of forest fragmentation. Landscape Ecology. 21: 107–120.

**Zharikov, Y., Lank, D.; Huettmann, F.; Cooke, F. 2007b.** Interpreting habitat distribution models of an elusive species, the marbled murrelets: a response to Burger and Page. Landscape Ecology Landscape Ecology. 22: 1283–1289.

## 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.

## Glossary Literature Cited

**Berkes, F.; Colding, J.; Folke, C. 2000.** Rediscovery of traditional ecological knowledge as adaptive management. Ecological Applications. 10(5): 1251–1262.

**Brown, G.G.; Reed, P. 2009.** Public participation GIS: a new method for use in national forest planning. Forest Science. 55(2): 166–182.

**Carey, A.B. 2003.** Restoration of landscape function: reserves or active management? Forestry. 76(2): 221–230.

**Churchill, D.J.; Larson, A.J.; Dahlgreen, M.C.; Franklin, J.F.; Hessburg, P.F.; Lutz, J.A. 2013.** Restoring forest resilience: from reference spatial patterns to silvicultural prescriptions and monitoring. Forest Ecology and Management. 291: 442–457.

**Council on Environmental Quality [CEQ]. 1997.** Environmental justice guidance under the National Environmental Policy Act. [Updated]. https://www.epa.gov/sites/production/files/2015-02/documents/ej_guidance_nepa_ceq1297.pdf. (21 December 2017).

**Doremus, H. 2007.** Precaution, science, and learning while doing in natural resource management. Washington Law Review. 82: 547–580.

**Espy, M.; Babbit, B. 1994.** Record of decision: amendments to Forest Service and Bureau of Land Management planning documents within the range of the northern spotted owl. Washington, DC: U.S. Department of Agriculture, Forest Service, and U.S. Department of the Interior, Bureau of Land Management. 74 p.

**Executive Order No. 13112.** 64 FR 6183, February 8, 1999. Invasive species. https://www.gpo.gov/fdsys/pkg/FR-1999-02-08/pdf/99-3184.pdf. (June 2017).

**Forest Ecosystem Management Assessment Team [FEMAT]. 1993.** Forest ecosystem management: an ecological, economic, and social assessment. Washington, DC: U.S. Government Printing Office.

**Franklin, J.F. 1990.** Biological legacies: a critical management concept from Mount St. Helens. In: Sabol, K.J., ed. Transactions of the 55th North American wildlands natural resource conference. Wildlife Management Institute: 216–219.

**Franklin, J.F.; Mitchell, R.J.; Palik, B.J. 2007.** Natural disturbance and stand development principles for ecological forestry. Gen. Tech. Rep. GTR-NRS-19. Newtown Square, PA: U.S. Department of Agriculture, Forest Service, Northern Research Station. 44 p.

**Franklin, J.F.; Spies, T.A.; Pelt, R.V.; Carey, A.B.; Thornburgh, D.A.; Berg, D.R.; Lindenmayer, D.B.; Harmon, M.E.; Keeton, W.S.; Shaw, D.C.; Bible, K.; Chen, J. 2002.** Disturbances and structural development of natural forest ecosystems with silvicultural implications, using Douglas-fir forests as an example. Forest Ecology and Management. 155(1–3): 399–423.

**Garibaldi, A.; Turner, N. 2004.** Cultural keystone species: implications for ecological conservation and restoration. Ecology and Society. 9(3): art1.

**Gosnell, H.; Abrams, J. 2011.** Amenity migration: diverse conceptualizations of drivers, socioeconomic dimensions, and emerging challenges. GeoJournal. 76(4): 303–322.

**Hobbs, R.J.; Higgs, E.; Harris, J.A. 2009.** Novel ecosystems: implications for conservation and restoration. Trends in Ecology & Evolution. 24(11): 599–605.

**Hunter, M.L. 2005.** A mesofilter conservation strategy to complement fine and coarse filters. Conservation Biology. 19(4): 1025–1029.

**Jenerette, G.D.; Wu, J. 2000.** On the definitions of scale. Bulletin of the Ecological Society of America. 81(1): 104–105.

**Jorgensen, B.S.; Stedman, R.C. 2001.** Sense of place as an attitude: lakeshore owners attitudes toward their properties. Journal of Environmental Psychology. 21(3): 233–-248.

**Keane, R.E.; Ryan, K.C.; Veblen, T.T.; Allen, C.D.; Logan, J.A.; Hawkes, B. 2002.** The cascading effects of fire exclusion in Rocky Mountain ecosystems: a literature review. Gen. Tech. Rep. RMRS-GTR-91. Fort Collins, CO: U.S. Department of Agriculture, Forest Service, Rocky Mountain Research Station. 24 p.

**Liu, J.G.; Dietz, T.; Carpenter, S.R.; Alberti, M.; Folke, C.; Moran, E.; Pell, A.N.; Deadman, P.; Kratz, T.; Lubchenco, J.; Ostrom, E.; Ouyang, Z.; Provencher, W.; Redman, C.L.; Schneider, S.H.; Taylor, W.W. 2007.** Complexity of coupled human and natural systems. Science. 317(5844): 1513–1516.

**Long, J.W.; Tarnay, L.W.; North, M.P. 2017.** Aligning smoke management with ecological and public health goals. Journal of Forestry. https://doi.org/10.5849/jof.16-042. (22 December 2017).

**Lowery, D.R.; Morse, W.C. 2013.** A qualitative method for collecting spatial data on important places for recreation, livelihoods, and ecological meanings: integrating focus groups with public participation geographic information systems. Society & Natural Resources. 26(12): 1422–1437.

**Nelson, P.B. 2001.** Rural restructuring in the American West: land use, family and class discourses. Journal of Rural Studies. 17(4): 395–407.

**North, M.; Stine, P.A.; O'Hara, K.L.; Zielinski, W.J.; Stephens, S.L. 2009.** An ecosystems management strategy for Sierra mixed-conifer forests, with addendum. Gen. Tech. Rep. PSW-GTR-220. Albany, CA: U.S. Department of Agriculture, Forest Service, Pacific Southwest Research Station. 49 p.

**Perry, D.A.; Hessburg, P.F.; Skinner, C.N.; Spies, T.A.; Stephens, S.L.; Taylor, A.H.; Franklin, J.F.; McComb, B.; Riegel, G. 2011.** The ecology of mixed severity fire regimes in Washington, Oregon, and northern California. Forest Ecology and Management. 262(5): 703–717.

**Pfister, R.D.; Arno, S.F. 1980.** Classifying forest habitat types based on potential climax vegetation. Forest Science. 26(1): 52–70.

**Power, M.E.; Tilman, D.; Estes, J.A.; Menge, B.A.; Bond, W.J.; Mills, L.S.; Daily, G.; Castilla, J.C.; Lubchenco, J.; Paine, R.T. 1996.** Challenges in the quest for keystones. BioScience. 46(8): 609–620.

**Proshansky, H.M. 1978.** The city and self-identity. Environment and Behavior. 10(2): 147–169.

**Redman, C.L.; Grove, J.M.; Kuby, L.H. 2004.** Integrating social science into the long-term ecological research (LTER) network: social dimensions of ecological change and ecological dimensions of social change. Ecosystems. 7(2): 161–171.

**Riitters, H.K.; Wickham, D.J.; O'Neill, V.R.; Jones, B.K.; Smith, R.E.; Coulston, W.J.; Wade, G.T.; Smith, H.J. 2002.** Fragmentation of continental United States forests. Ecosystems. 5(8): 815–822.

**Sarukhán, J.; Whyte, A. 2005.** Ecosystems and human well-being: synthesis. Millennium Ecosystem Assessment. Washington, DC: Island Press, World Resources Institute. 137 p.

**Stankey, G.H.; Bormann, B.T.; Ryan, C.; Shindler, B.; Sturtevant, V.; Clark, R.N.; Philpot, C. 2003.** Adaptive management and the Northwest Forest Plan: rhetoric and reality. Journal of Forestry. 101(1): 40–46.

**Steffen, W.; Crutzen, P.J.; McNeill, J.R. 2007.** The Anthropocene: Are humans now overwhelming the great forces of nature? AMBIO: A Journal of the Human Environment. 36(8): 614–621.

**Stokols, D.; Shumaker, S.A. 1982.** The psychological context of residential mobility and well-being. Journal of Social Issues. 38(3): 149–171.

**Swanson, M.E.; Studevant, N.M.; Campbell, J.L.; Donato, D.C. 2014.** Biological associates of early-seral pre-forest in the Pacific Northwest. Forest Ecology and Management. 324: 160–171.

**Teitelbaum, S.; Beckley, T.; Nadeau, S. 2006.** A national portrait of community forestry on public land in Canada. The Forestry Chronicle. 82(3): 416–428.

**Turner, M.G.; Romme, W.H. 1994.** Landscape dynamics in crown fire ecosystems. Landscape Ecology. 9(1): 59–77.

**U.S. Department of Agriculture, Forest Service [USDA FS]. 2011.** Watershed condition framework. FS-977. Washington, DC: U.S. Department of Agriculture, Forest Service. 34 p.

**U.S. Global Change Research Program. 2017.** Glossary. [Updated]. http://www.globalchange.gov/climate-change/glossary. (11 October 2017).

**Williams, D.R.; Stewart, S.I. 1998.** Sense of place: an elusive concept that is finding a home in ecosystem management. Journal of Forestry. 96(5): 18–23.

**Yamamoto, S.-I. 2000.** Forest gap dynamics and tree regeneration. Journal of Forest Research. 5(4): 223–229.

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