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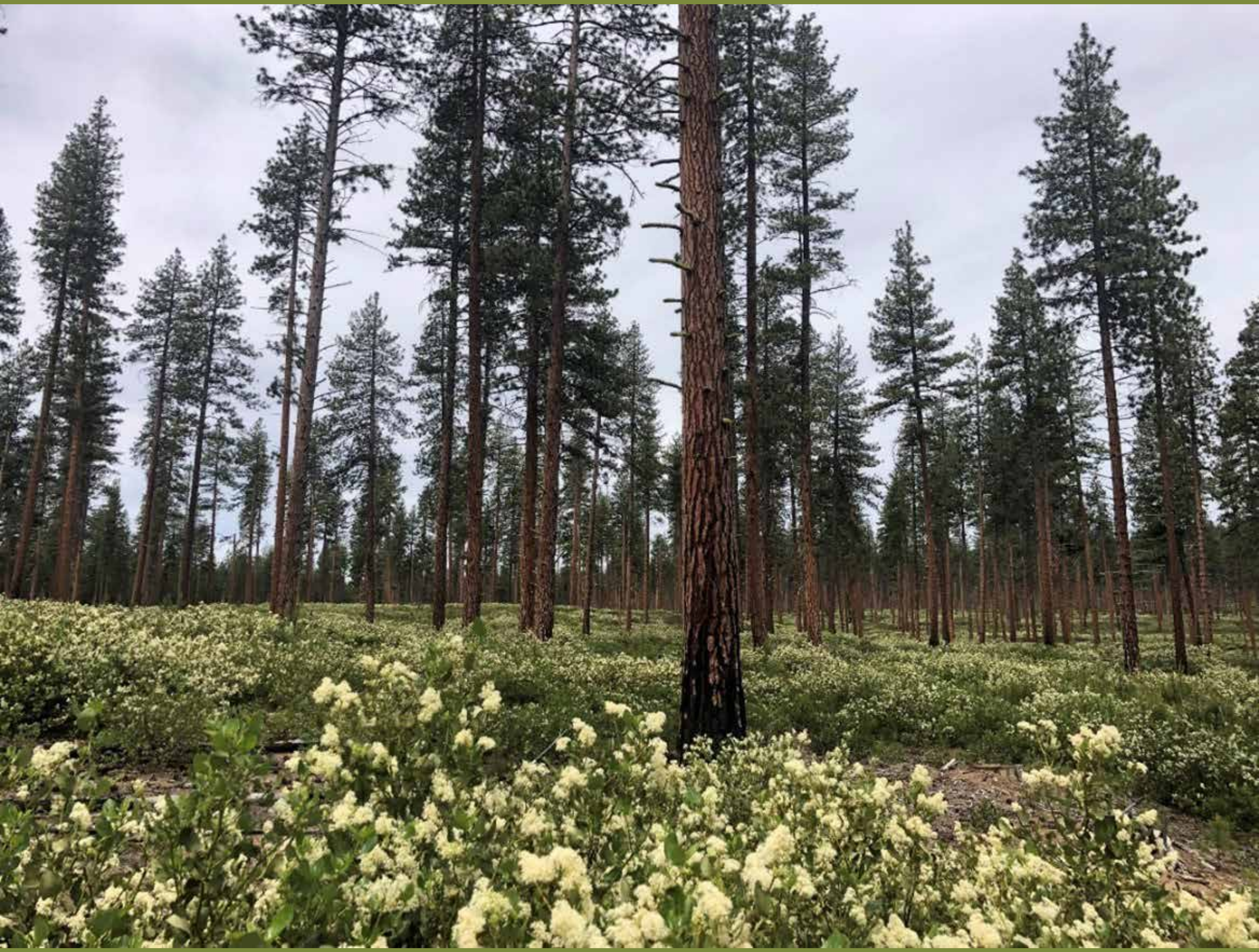
Forest Dynamics After Thinning and Fuel Reduction in the Pringle Falls Experimental Forest

Establishment and Early Observations of the Lookout Mountain Thinning and Fuels Reduction Study



Technical Editors

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Ceanothus velutinus in bloom in the understory of a forest stand that was thinned and treated to reduce fuels as part of the Lookout Mountain Study in the Pringle Falls Experimental Forest, Deschutes National Forest, Oregon, USA.
Photo by Lorelle Sherman.

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Abstract

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The Lookout Mountain Thinning and Fuels Reduction Study is an operational-scale, long-term management experiment to evaluate the effectiveness of silvicultural treatments to mitigate fire risk and the associated implications for ecological processes and functions. The study is being conducted within the Pringle Falls Experimental Forest, which encompasses about 1019 ha of ponderosa pine (*Pinus ponderosa*) and mixed-conifer forest originating predominantly from an 1845 wildfire. Study treatments include four levels of overstory thinning, each followed by mastication of understory vegetation and broadcast burning, and an untreated Reference. Thinning treatments target residual tree densities of 50, 75 or 100 percent of the upper management zone (UMZ) and a fourth treatment targeting 75 percent UMZ with gap openings (75 UMZ + Gaps). Treatments were assigned to experimental units in a randomized block design.

The study treatments were implemented between 2011 and 2015 in a series of three sales using a combination of conventional timber sale and stewardship contracts. The detection of a pair of nesting northern spotted owls (*Strix occidentalis caurina*) and litigation complicated implementation. Court rulings determined that the proposed experimental activities addressing research questions stated in the project purpose and need could fall outside of standard practices employed on other National Forest System lands. The nesting spotted owls were accommodated by establishing a no-treatment buffer around the nest site, reconfiguring some treatment unit boundaries, and eliminating of one replication of the 75 UMZ + Gaps treatment.

Initial posttreatment observations indicate that the thinning treatments resulted in stands that compositionally shifted to ponderosa pine dominance with increased average stem diameter. Target basal area densities were met within 88.5 to 180.9 percent. Target stand density indices were met within 51 to 193 percent. Thinning and fuel reduction treatments resulted in decreased shrub and subshrub cover and increased diversity of understory plant species. Posttreatment tree mortality from bark beetles occurred at low frequency and is not expected to confound the reduction of landscape-scale risk, the primary objective of the thinning and fuels reduction treatments. Thinning treatments combined with understory burning resulted in a fivefold increase in woodpecker nest occurrence, suggesting a positive response by cavity excavators. Soil heating experiments demonstrated that the occurrence of extreme soil heating is harmful to soil microorganisms and organic matter-derived nutrients. Understory microclimate varied geospatially in relation to physiography and residual overstory density. Presumed future overstory canopy densification arising from crown expansion will likely further mediate understory temperatures through decreasing incoming solar radiation and insolation of nighttime heat loss.

Future monitoring will determine the timing of subsequent, repeated understory fuels reduction and longer term dynamics of vegetation composition and structure, wildfire risk, and ecological and biophysical performance.

Keywords: Oregon, Pacific Northwest, ponderosa pine, silviculture, wildland fire.

Contents

Part I:
Lookout Mountain Thinning and Fuels Reduction Study—Context and Setting

Chapter 1: Introduction

Lorelle M. Sherman and Paul D. Anderson[1]

Establishment Report Overview

The Lookout Mountain Thinning and Fuels Reduction Study (hereafter LMS), explores silvicultural treatments intended to restore resiliency in dry, fire-prone forests east of the Cascade crest in Oregon. The LMS experimental framework facilitates the evaluation of thinning and fuel reduction treatment effects on the susceptibility of these forests to fire, insects, wind, environmental change, and other forest disturbances over time. It also addresses the impacts of these silvicultural treatments on a broader array of forest functions and objectives, including ecological and biophysical attributes.

The study is located on the Lookout Mountain unit of the Pringle Falls Experimental Forest (PFEF) near La Pine, Oregon. The U.S. Department of Agriculture (USDA) Forest Service Pacific Northwest Research Station manages the 4472-ha experimental forest, an administrative withdrawal from the Deschutes National Forest. Established in 1931, PFEF was the first experimental forest in the Pacific Northwest and functioned as a center for silviculture, forest management, and insect and disease research in ponderosa pine (*Pinus ponderosa*) forests east of the Oregon Cascade Range. Experimental forests and ranges constitute a national network of outdoor laboratories that the Forest Service designated for the express purpose of providing sites for research. The Forest and Rangeland Renewable Resources Research Act of 1978 reaffirmed the use of experimental forests and ranges for research. Experimental forests and ranges have a rich legacy of providing information to guide forest management activities. They offer a unique platform from which researchers can address forest management questions at various scales, and experimental forests and ranges offer important opportunities for collaborative research.

The LMS was conceived jointly by researchers from the Pacific Northwest Research Station and resource managers from the Deschutes National Forest. The initial emphasis was to assess the responses of overstory and understory vegetation as well as downed wood surface fuels to alternative thinning and fuels reduction treatments. The core vegetation study (described in chapter 7 of this report) represents this emphasis. A cadre of Forest Service and university researchers posed additional research questions addressing a broader array of ecological responses. These queries prompted several ancillary studies embedded within

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the main LMS experimental design. The core vegetation study represents many agencies, whereas the ancillary studies are a direct product of the participants.

This establishment report first reviews the key issues and drivers that framed the need for alternative silvicultural treatments in dry conifer forests. It then describes the LMS experimental design and implementation. This report also outlines important operational details and legal challenges that had to be addressed for the LMS implementation. It concludes by summarizing initial postimplementation observations.

As a long-term study spanning several decades, the LMS will likely engage multiple generations of researchers and managers. This documentation of study conceptualization, implementation, and early observations facilitates continuity of effective research-management collaboration with the LMS over the long term. Furthermore, the lessons learned in the implementation of the LMS can serve as a guide for forest managers interested in implementing similar silvicultural treatments in their operational programs of work.

An objective that this report addresses is the documentation of initial, near-term observations of treatment implementation effects on vegetation composition and structure, as well as some other ecological and biophysical responses. In a sense, this report documents how the treatment implementation reset the forest system. A near-term objective is determining the degree to which the specified stand density targets were met through thinning treatments and subsequent influences of prescribed fire. Beyond this summarization, the realization of many LMS objectives and the development of rigorous inferences will require additional monitoring and statistical analyses over longer timeframes—from a few years to decades.

Regional Context, Background, and Research Needs

Before Euro-American settlement, low-elevation, dry conifer forests of the Western United States were fundamentally shaped by frequent low- or mixed-severity disturbances, such as wildfires (Agee 1993, Arabas et al. 2006, Everett et al. 2000, Hagmann et al. 2021, Hessburg et al. 2005) and insect attacks (Hayes and Daterman 2001, McCullough et al. 1998), mediated by diverse environmental gradients. Today, resource managers face multiple challenges in reestablishing and maintaining patterns of disturbance that mitigate fire risk, improve forest health, and ensure forest resiliency. One of these challenges is determining how to manage forest composition and structure to reduce the flammability of forest stands. Resource managers also must understand the effectiveness of contemporary thinning and understory fuel treatments in creating and maintaining forest stand resistance and resilience to disturbance. An additional challenge is identifying the effects these treatments have on other ecosystem attributes (such as wildlife habitat, watershed function, or soil health).

Historically, surface fires ignited predominantly by lightning during the dry season (Agee 1993, Rorig and Ferguson 1999) limited regeneration of fire-intolerant species, reduced densities of small-diameter stems, consumed litter and downed wood, created variable-sized openings, and eliminated fuel ladders between the forest floor and the overstory canopy. Consequently, these dry forests were composed of widely spaced, medium to large old live trees and scattered dead trees with a continuous herbaceous understory and low abundance of ground and surface fuels (Agee 1994, Arabas et al. 2006, Wickman 1992, Youngblood et al. 2004). These conditions supported a fire regime dominated by low-severity fire, with the occasional moderate- to high-severity fire occurring during extreme weather conditions (Hagmann et al. 2013, 2019; Hessburg et al. 2020; Merschel et al. 2014, 2018).

Today, many of these same dry forests face a greater risk of succumbing to common fire and bark beetle disturbances, increased environmental stress, and their interactions. Contributing risk factors include the following:

- An accumulated mass of downed wood and continuity of the fuels mosaic at landscape scales (Covington and Moore 1994, Harrod et al. 1999, Youngblood et al. 2004)
- More small trees and fewer large trees (Arno et al. 1997, Stephens and Gill 2005, Taylor and Skinner 1998)
- Greater extent of young, multistoried forests with fire-intolerant conifers in both understory and overstory strata (Hessburg et al. 2005, Merschel et al. 2014)
- Increased fuel ladders that contribute to increased torching and crown fire initiation (Agee 1993)
- Altered regional climate impacts on plant community structure and organization (Johnston et al. 2017, McKenney et al. 2007)

In many dry forests of the Pacific Northwest, altered fuelbeds and shifts in forest structure and composition resulted from fire exclusion and fire suppression, livestock grazing, land use changes, timber management activities, and changes in climate (Arno et al. 1997, Bergoffen 1976, Dolph et al. 1995, Richardson et al. 2007, Steele et al. 1986). Moreover, there is an increased risk of loss of human lives and property. A steady increase in use of public forest lands, the number of people living within 80 to 161 km of these public lands, and the frequency and intensity of hazardous conditions in the wildland-urban interface amplify the risk of catastrophic loss (English et al. 2015, Olsen et al. 2017).

Recent wildfires across the Western United States have led to an increased awareness of and urgency in restoring resiliency to wildfire in western forests (Hagmann et al. 2021, Prichard et al. 2021). There is broad agreement that forest restoration treatments can be effective tools to restore forest resiliency (Agee and Skinner 2005, Graham et al. 2004, Kane et al. 2019, Sohn et al. 2016). The U.S. Department of the Interior (USDI), the USDA, their land management agencies, and their partners initiated the National Cohesive Wildland Fire Management Strategy as a multi-entity collaborative effort to promote forest restoration efforts

(USDI and USDA 2014). Presidential initiatives, such as the National Fire Plan, the 10-year Comprehensive Strategy, Executive Order 13855 of 2019, and the Healthy Forests Initiative, as well as legislation, such as the Healthy Forests Restoration Act of 2003 (Public Law 108–148), have promoted the consideration of large-scale and strategically located fuel reduction and forest restoration projects to manage landscapes within the context of ecological processes. Fuel reduction and forest restoration treatments are often applied in dry forests as thinning or underburning projects with the goal of reducing the risk of severe surface and crown fires (Agee and Skinner 2005, Brown et al. 2004). They are designed to reduce surface fuels and canopy density while retaining large, live trees of fire-resistant species. They also aim to increase the live canopy height. The National Fire and Fire Surrogate (FFS) Study provided partial validation of these precepts and demonstrated that thinning and underburning result in at least short-term reductions in woody fuels mass and thus reduce the potential for and severity of wildland fires (McIver et al. 2013; Stephens et al. 2009, 2012; Youngblood et al. 2008).

Ecological consequences of silvicultural treatments implemented with the intention of mitigating compounding forest disturbances (e.g., wildfire, drought stress, changing environmental conditions) are still uncertain. First, most response assessments have been limited to the short term. Few studies are carried out on large-scale, long-term experimental platforms. Second, while treatments often have been designed to restore fuel load and forest composition and structure to historical conditions, they have ignored the risks, uncertainties, and benefits of multiple environmental stressors in the context of modern environmental conditions (Parker et al. 2006). For example, in forests where water was not limiting, changes in temperature, precipitation, and atmospheric carbon dioxide levels over the past six decades may have had a generally positive effect on forest productivity (Boisvenue and Running 2006). However, over the past two decades, many western forests have experienced increased tree mortality due to severe drought (van Mantgem and Stephenson 2007). Management options are needed that accommodate uncertainty in future environmental conditions and dynamically shifting forest ecosystems (Hessburg et al. 2019, Kerns et al. 2020). Landscape-scale, long-term monitoring of natural disturbances and forest resilience is crucial for revealing novel management concerns.

Forest restoration projects promoting the release of old trees and a compositional shift toward dominance of fire- and drought-tolerant species have become a priority in dry ponderosa pine forests (Franklin and Johnson 2012). Maintaining healthy, old ponderosa pine forests in Oregon is important for upholding social and cultural values, providing ecosystem services (e.g., carbon sequestration, water regulation), and conserving critical wildlife habitat.

Studies have found substantial variation among small mammal and bird taxa responses to thinning and fuels reduction treatments. The white-headed woodpecker (WHWO; *Dryobates albolarvatus*), a species of concern in the

Pacific Northwest, frequently uses ponderosa pine snags for nest cavities; thus, as fire suppression efforts have converted forests from ponderosa pine-dominated to true fir (*Abies* spp.)-dominated, the population of the WHWO has declined (Garrett et al. 2020). Thinning treatments and thinning in combination with prescribed burning as part of the FFS Study have positively influenced WHWO foraging rates when compared with untreated stands (Lyons et al. 2008). Bark beetle populations were attracted to burned trees in FFS Study stands (Schwilk et al. 2006). WHWO sometimes consume bark beetles as part of their diet (Kozma et al. 2020), so treatments involving burning may indirectly positively influence WHWO populations (Farris et al. 2010). Other species that rely on dry conifer forest in resilient and natural conditions include the flammulated owl (*Psiloscops flammeolus*), Lewis's woodpecker (*Melanerpes lewis*), long-legged myotis (*Myotis volans*), and pallid bat (*Antrozous pallidus),* all of which have been identified as species of greatest conservation need under the Oregon Department of Fish and Wildlife Oregon Conservation Strategy (ODFW 2016).

The importance of ecosystem services, including conditions, processes, and components of the natural environment that benefit human life (Daily 1997, Smith et al. 2011), and maintenance of forest health and resilience have been highlighted by the growing uncertainties of future conditions (Grulke and Heath 2020). Natural disturbances can substantially reduce the ability of forests to store carbon and maintain water resources. Interactions of natural disturbances with altered environmental conditions further amplify the risk of major changes in forest functionality, potentially transitioning a forest from a net carbon accumulator (sink) to a net carbon emitter (source) (Lal 2005, Zhang et al. 2015). Natural disturbance can also alter water availability and resources through changes in canopy cover, albedo (solar reflectance) and potential evapotranspiration, and soil moisture, subsequently altering soil heat load and erosion (Elliot 2003, Lal 2005).

Mycorrhizal fungal networks buffer nutrient and water availability for vegetation during times of environmental stress. Van der Heijden et al. (2015) estimated that mycorrhizal fungal networks obtain up to 80 percent of the nitrogen and phosphorous plants require. The inoculation of tree seedling roots by mycorrhizal fungi facilitates the regeneration of forests after a natural or silvicultural disturbance. Saprobic fungi break down decaying wood and plant materials, cycling nutrients in a form usable to mycorrhizal fungi and plants back into the soil. High-intensity fire can result in severe soil heating, soil nutrient volatilization, and damage to the function of soil microorganisms, ectomycorrhizal fungi (Hebel et al. 2009, Neary et al. 1999, Smith et al. 2017), and overall function of the forest.

Insects are major components of forest ecosystems, representing most of the biological diversity and affecting virtually all processes and uses. Bark beetles (Coleoptera: Curculionidae, Scolytinae) heavily influence the structure and function of low-elevation, dry conifer forests by regulating aspects of primary production;

nutrient cycling; ecological succession; and the size, distribution, and abundance of forest trees (Fettig et al. 2007). Bark beetle attacks lead to the mortality of individual and small groups of trees, alter the accumulation of fuels and vegetation, and create canopy gaps that provide opportunities for new seedling cohorts (Hayes and Daterman 2001, Hessburg et al. 1994).

Increases in overall stand density over the past century have led to increased competition among trees, reduced tree vigor, and increased susceptibility to attack from bark beetles and other forest insects and diseases (Fettig et al. 2007, Hessburg et al. 1994, Mitchell 1990, Oliver 1995). Thinning decreases mortality of ponderosa pine by mountain pine beetle (*Dendroctonus ponderosae*) (Fettig et al. 2007, Hood et al. 2016). The mountain pine beetle often kills large numbers of trees when contiguous stands or landscapes become vulnerable (e.g., during severe single-year or multiyear droughts) (Kolb et al. 2016). These changes have occurred more recently in the context of shifting resource availability, including decreased soil moisture, increased air temperature and heat load, and greater minimum temperatures (Halofsky et al. 2019). Collectively, these altered conditions contribute to increased probability of multiple, interacting stresses and may lead to novel disturbance regimes and changes in forest structure and function (Fettig et al. 2021, Vose et al. 2018).

Objectives of the Lookout Mountain Thinning and Fuels Reduction Study

A strategic priority of the USDA Forest Service is to sustain our nation's forests and grasslands by addressing goals to (a) foster resilient and adaptive ecosystems to mitigate climate change and (b) mitigate wildfire risk.[2] The applicability of these goals is strongly reflected in the historical, contemporary, and anticipated future of the fire-prone ponderosa pine ecosystems of the Pacific Northwest. The agency's ability to address these goals depends on an evolving body of science—knowledge and tools development—generated through research and operational application.

The LMS was undertaken to mitigate wildfire risk to long-term and ongoing research being conducted at the experimental forest. In doing so, the study examines the effectiveness of different mitigation treatments for decreasing wildfire risk and assesses the collateral effects of those treatments on other ecological processes and values. The objectives of the study are to (1) refine management options for restoring resiliency in ponderosa pine ecosystems; (2) evaluate the effects of thinning and fuel reduction on long-term susceptibility to fire, insects, wind, climate change, and other forest disturbances; and (3) protect long-term experiments and enhance future research opportunities.

[2] The USDA Forest Service Strategic Plan (https://www.fs.usda.gov/strategicplan) contains four outcome-oriented goals: (1) sustain our nation's forests and grasslands, (2) deliver benefits to the public, (3) apply knowledge globally, and (4) excel as a high-performing agency.

The above objectives were translated into six research questions (Youngblood 2009) that focused on the dynamic responses of forest composition and structure and associated disturbance risk:

1. What set of fuel reduction treatments best accelerates the development of large trees, while over the long-term reintroduces natural disturbance processes that provide greater ecosystem resiliency?
2. What is the long-term influence of climate change interacting with a set of fuel reduction treatments on vegetation dynamics and forest structure?
3. Can single-cohort stands be readily converted to multicohort stands?
4. Do multicohort stands share the same risks of multiple, interacting stresses as single-cohort stands?
5. How does the dominant shrub, giant chinquapin (*Chrysolepis chrysophylla*), respond in the near term to a set of fuel reduction treatments?
6. How does the residual stand structure resulting from a set of fuel reduction treatments interact locally and in the near term with wind to cause additional structural changes?

The PFEF offers the unique opportunity for robust, long-term study of forest health and resilience of fire-prone ponderosa pine forests experiencing an array of climatic and biotic stressors. The array of studies established since its inception (Youngblood 2011) provide a robust foundation for researching the contemporary and future forest resilience at an operational landscape scale. Specifically, historical studies have provided detailed examinations of stand density—growth relationships for ponderosa pine and other mixed-conifer species (e.g., Barrett 1982, Kolbe and McKay 1939), and there has been detailed investigation of forest insect population dynamics and impacts (e.g., Keen 1936, Larsson et al. 1983), two important drivers of the forest resilience issue for ponderosa pine forests in the bioregion.

As an experimental forest, the PFEF provides opportunities for experimentation, demonstration, and learning about novel resource management approaches. The LMS is a unique investigation of silvicultural treatments at operational scales (Monserud 2002), providing information directly applicable to management plans and their implementation in dry conifer forests. With more than 1000 ha available for consideration, the Lookout Mountain unit of the PFEF provided a landscape large enough to accommodate operational-scale treatment units with replication to test thinning intensity options. At this scale, treatments could be implemented using timber sale and contracting mechanisms typically available to the National Forest System, with the observed responses reflecting the ecological and biophysical processes and impacts occurring in typical Forest Service management projects.

Intended Use of the Establishment Report

This establishment report is designed for use by those interested in the implementation of the LMS, the early observations of the core vegetation study nested within the LMS, and the progress in addressing ancillary research questions posed by other researchers. It is important to understand regional silvicultural issues, research needs, and the overarching experimental design before considering finer scale components of the study. For this reason, part I frames the research needs and process of conceptualizing the study in addition to providing an indepth description of the setting and context of the LMS. Part II outlines the implementation of the landscape-scale study and the overarching study design followed by information about litigation and study reorganization in response to the detection of a northern spotted owl (*Strix occidentalis caurina*) (NSO) nesting pair. Details about timber harvest and vegetation treatment contracts and litigation are included for forest managers and researchers interested in implementing similar forest restoration treatments. Part III dives into the detailed methods and early observations of the core vegetation study, the foundational research component of the LMS.

Ancillary studies embedded within the broader context of the LMS address ecological and biophysical research attributes beyond the core vegetation responses. Part III also describes the methods and early observations associated with these ancillary studies. We encourage investigators with research questions that fit within the scope of the overarching treatments to explore using this long-term, landscape-scale experiment. Current and previous work at the PFEF serve as an excellent foundation for additional studies as new management concerns arise with shifting values and objectives in applied forestry, social perceptions, and regulatory requirements. The establishment report concludes with a discussion of intended and recommended future work in part IV.

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Chapter 2: Pringle Falls Experimental Forest Overview

Lorelle M. Sherman and Paul D. Anderson[1]

Geographic, Physiographic, and Ecological Setting

Geography

The Lookout Mountain Thinning and Fuels Reduction Study (hereafter LMS) is located entirely within the Lookout Mountain unit of the Pringle Falls Experimental Forest (PFEF; latitude 43°42′ N, longitude 121°37′ W) in the Deschutes National Forest, about 48 km southwest of Bend, Oregon.

Physiography

The Lookout Mountain unit lies on an ancient shield volcano with an upper elevation of 1900 m. As part of the High Cascades physiographic province, cinder cones and volcanic peaks surround the LMS area (Youngblood 2011). The LMS area within the Lookout Mountain unit (1430 ha) ranges from south to east aspects and elevations from 1370 to 1645 m. There is minimal change in elevation at the perimeters and steep slopes toward the center (Youngblood 2011).

Soils

The soils in the Lookout Mountain unit are relatively well-drained, loamy, coarse sands and ash at 80 to 130 cm in depth (Youngblood 2011). Soils are remnant of the Mount Mazama (Crater Lake) eruption and were deposited around 6,600 years ago. Subsequent, nearby eruptions deposited more recent layers of ash and pumice. Soils are low in organic matter, nitrogen, sulfur, and phosphorus because of their lack of development and type of parent material (Youngblood 2004a).

Climate

The position of the PFEF among the Cascade Mountain Range and the Great Basin Desert strongly influences its climate. Current mean annual precipitation at Wickiup Dam, Oregon, is 519 mm and falls mostly as snow or early summer rains. Precipitation at the summit of Lookout Mountain approaches 1000 mm annually. The study area lies within a region that has frequent lightning storms, which historically have been a predominant source of fire ignition. Mean annual temperature is 6.2 °C, though it has increased by 0.05 °C per decade since 1895 (Halofsky et al. 2019). Vose et al. (2017) predicted a 1.67 to 2.59 °C increase in mean annual temperature for the Pacific Northwest in 2036 to 2065 when compared with the beginning of the 20th century.

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Kim et al. (2019) developed future climate projection models for south-central Oregon, and Halofsky et al. (2019) incorporated these models into the south-central Oregon climate change vulnerability assessment. Using 21 of the 42 global circulation models that best fit the Pacific Northwest, the average annual warming of air temperature is expected to increase from an average of 1.3 to 4.0 °C by 2050 and from an average of 2.7 to 4.8 °C by 2080 (Halofsky et al. 2019). Models predict that warming will occur in all seasons, but the most extreme increases are expected in the summer months (Mote et al. 2014). Future trends for annual precipitation are unclear, but large changes in the timing and magnitude of snowpack and snowmelt are expected. Residence time of snowpack is predicted to decrease by 7 to 8 weeks (Halofsky et al. 2019). A reduction in snowpack and earlier snowmelt dates will result in more variable but overall lower average streamflow and increased stress on aquatic and riparian systems. Downstream effects of lower flow rates are expected to have negative impacts on fisheries, wildlife, livestock, firefighting capabilities, recreation, and municipal water sources.

The dry conifer forests of PFEF will likely be more resilient to higher ambient air temperatures than other forest types in the region (Halofsky et al. 2019). Additional interacting stressors, such as decreased water availability and increased temperatures, will compromise the inherent resiliency of ponderosa pine (*Pinus ponderosa*) forests to wildfire.

Vegetation

Three plant associations (Volland 1988) were mapped within the bounds of the LMS area. Table 2.1 provides descriptions of their dominant vegetation components. The ponderosa pine/bitterbrush (*Purshia tridentata*)-snowbrush (*Ceanothus velutinus*)/sedge plant association (CP-S3-12) occurs in drier and warmer locations on toe slopes and undulating slopes with east aspects at elevations up to about 1370 m. From about 1370 m upslope to about 1500 m on east to southeast aspects is the mixed-conifer/greenleaf manzanita (*Arctostaphylos patula*)-snowbrush/sedge-penstemon plant association (CW-S1-13). Ponderosa pine is the dominant tree species on south aspects, while grand fir (*Abies grandis*) or white fir (*A. concolor*) increases in prevalence on east to northeast aspects. The mixed-conifer/snowbrush/sedge plant association (CW-S1-15) occurs on south and east aspects up to about 1615 m in elevation, supporting ponderosa pine as a major seral tree species, as well as small amounts of grand fir, white fir, sugar pine (*Pinus lambertiana*), and Shasta red fir (*Abies magnifica* var *shastensis*) as advance regeneration. The tall shrub giant chinquapin (*Chrysolepis chrysophylla*) occurs in the CW-S1-15 plant association. A fourth plant association, mixed conifer/greenleaf manzanita (CR-S1-11), occurs at higher elevations but was not a part of the LMS.

Table 2.1—Dominant vegetation components for plant associations of the Lookout Mountain Thinning and Fuels Reduction Study area

Plant association and block(s)	Species	Cover
		Percent
CP-S3-12 (ponderosa pine/ bitterbrush-snowbrush/sedge)—block 4	Ponderosa pine	8–50
	Lodgepole pine (*Pinus contorta*)	0–10
	Bitterbrush	3–35
	Snowbrush	3–20
	Western needlegrass (*Achnatherum occidentale*)	T–10
	Long-stolon sedge (*Carex inops*)	2–25
CW-S1-13 (mixed conifer/manzanita-snowbrush/sedge-penstemon)—block 3	Ponderosa pine	10–60
	Lodgepole pine	0–5
	White fir	T–15
	Greenleaf manzanita	T–11
	Snowbrush	0–40
	Long-stolon sedge	T–20
	Glaucous beardtongue (penstemon)	5–60
CW-S1-15 (mixed conifer/snowbrush/ sedge)—blocks 1 and 2	Ponderosa pine	8–50
	Lodgepole pine	0–10
	White fir	3–35
	Snowbrush	3–20
	Western needlegrass	T–10
	Long-stolon sedge	2–25

T = trace; represent <1 percent.

Source: Volland 1988.

Note: Blocks are experimental replicates and were delineated by relatively similar elevation, aspect, and plant association to minimize experimental unit differences within a block.

Euro-American pioneers described the historical biological structure of east-side dry conifer forests as containing continuous understories punctuated by widely spaced, medium- to large-diameter ponderosa pines (Agee 1994). The PFEF still maintains stands that represent these conditions. However, a suite of modern human management practices (e.g., fire suppression, selective logging, etc.) have largely altered the landscape within and surrounding the PFEF.

Youngblood (2004b) described the attributes of old-growth ponderosa pine forests in the Pacific Northwest, including the Pringle Butte Research Natural Area located within the PFEF. These structural characteristics likely emulate the historical forest conditions. According to Youngblood (2004b), the conditions for old-growth ponderosa pine forest are as follows (mean ± standard error [SE]):

- Stand age: multiaged with >3 age classes
- Live upper canopy tree density: 50 ± 3.5 trees/ha
- Live upper canopy tree diameter: 60.0 ± 1.55 cm diameter at breast height (d.b.h.)
- Snag density: 9.0 ± 0.97 trees/ha
- Snag diameter: 61.7 ± 4.33 cm
- Coarse downed wood: 47.0 ± 5.28 logs/ha
- Spatial patterns: aggregated, clumped, or randomly distributed

Historical Disturbance and Current Conditions

Before its designation as an experimental forest in 1931, Euro-Americans explored the area surrounding the PFEF during the first half of the 19th century. Timber surveys by the U.S. Geological Survey recorded expansive, old-growth ponderosa pine forests in the area. Harvest of the region's ponderosa pine forests rapidly followed, inspiring Thornton T. Munger, the first director of the Pacific Northwest Forest Experiment Station, to set aside what is now the PFEF as protected old-growth forest in 1914 (Youngblood 2011).

The relatively large block of closed-canopy forest in the Lookout Mountain unit originated after a stand-replacement fire that burned around 1845, leading to the establishment of dense ponderosa pine at lower elevations and Douglas-fir (*Pseudotsuga menziesii*), grand fir, white fir, sugar pine, western white pine (*Pinus monticola*), and mountain hemlock (*Tsuga mertensiana*) at higher elevations. These closed-canopy stands have grown exceptionally well and may represent some of the most productive ponderosa pine sites in central Oregon (Youngblood 2011). Near the eastern portion of the study, about 129.5 ha show signs of a second notable fire ignited by lightning in the early 1900s that burned in the vicinity of Lookout Mountain (Bork 1984). Smaller fires occurred between stand-replacing fires. Using dendrochronology, Bork (1984) estimated that the mean fire return interval prior to 1900 was 8 years for the Lookout Mountain unit. In general, insect and pathogen outbreaks associated with large-diameter ponderosa pine forests in this region include the pandora moth (*Coloradia pandora*), western pine beetle (*Dendroctonus brevicomis*), red turpentine beetle (*Dendroctonus valens*), Comandra blister rust (*Cronartium comandrae*), dwarf mistletoe (*Arceuthobium* spp.), and pine butterfly (*Neophasia menapia*).

Prior to the LMS, research use of the Lookout Mountain unit affected a limited amount of the area through several small-scale silvicultural treatments (fig. 2.1, table 2.2). Additionally, limited low thinning occurred in the 1970s when the road network was established.

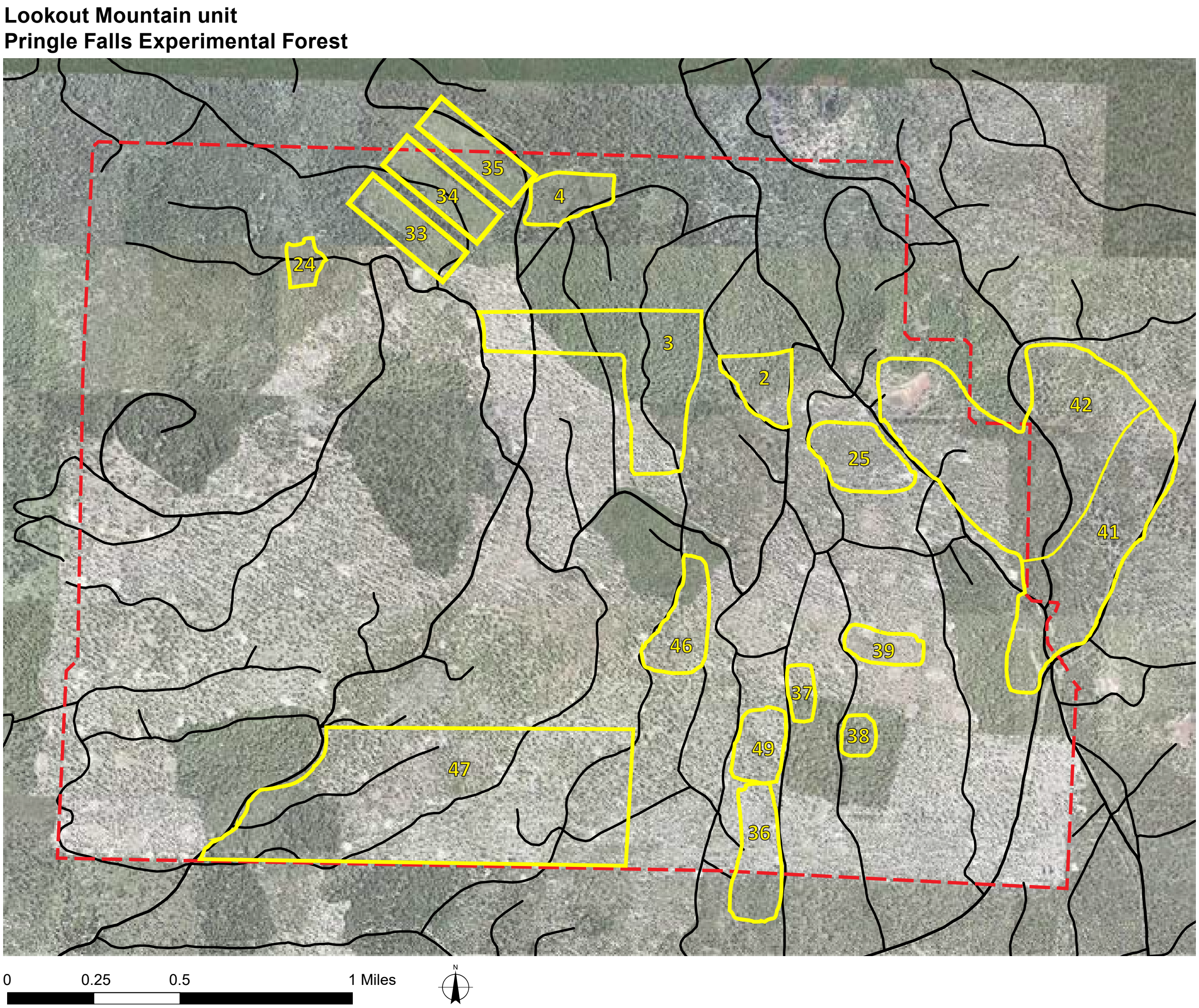

Figure 2.1—Past research activities with associated Pringle Falls Experimental Forest (PFEF) unit designations labeled as numbers, activity perimeters denoted as solid lines, and the PFEF boundary denoted as a dashed line. Table 2.2 describes the type of research activities associated with unit areas.

Table 2.2—Past research activities and associated scientists and publications by unit in the Pringle Falls Experimental Forest

Unit(s)	Research activity	Scientist(s)	Publication(s)
2	Underburning for shrub control (1976, 1979, 1980)	R. Martin	Martin (1982)
3	Pruning in a 55-year-old stand (1941)	E. Mowat, W. Dahms, J. Barrett, J. Cahill	Barrett (1968) Cahill (1991) Fight, Bolon, and Cahill (1993)
4	Ponderosa pine-grand fir spacing study	K. Seidel	
24	Release and thinning in a 42-year-old suppressed grand fir and Shasta red (noble) fir stand	K. Seidel	Seidel (1977)
25	Thinning from above in a 94-year-old stand (1938)	E. Mowat, J. Barrett	Barrett and Newman (1974)
33–35	Ponderosa pine-grand fir plantation in a spacing study	K. Seidel	Seidel (1985)
36–42	Ponderosa pine-lodgepole pine sale: commercial thinning (1949)	E. Mowat, J. Sowder	Sowder (1953)
46	Underburned	R. Martin	Evans (1983)
47	Levels-of-growing-stock established in 65-year-old ponderosa pine with six growing-stock levels (1965)	J. Barrett, P. Cochran, W. Oliver	Barrett (1983) Cochran and Barrett (1999) Oliver (2005)

Note: Unit numbers are not associated with the Lookout Mountain Thinning and Fuels Reduction Study unit designations. Figure 2.1 shows the area of research activities associated with units.

Prior and Ongoing Research Activity

The PFEF is the oldest experimental forest in the Pacific Northwest and was established in May 1931 as a center for silviculture, forest management, and insect and disease research in ponderosa pine forests east of the Oregon Cascade Range. It originally comprised 3043 ha centered on Pringle Butte. In 1937, the Forest Service incorporated the southern and eastern flanks of Lookout Mountain, augmenting the experimental forest with a 1430 ha unit.[2] The Deschutes National Forest management plan (USDA 1990) addresses the experimental forest as a single management area, with forest-level standards and guides applying when they do not conflict with research.

Early PFEF research resulted in a rating system for ponderosa pine susceptibility to western pine beetle attack (Keen 1936, 1943). A methods-of-cutting study used the Pringle Butte unit to look at the effects of different intensities of cutting large, old ponderosa pine on forest stand growth (Dahms 1951, Kolbe 1939). In 1965, James W. Barrett implemented the long-term, Levels-of-Growing-Stock Study to demonstrate the effects of tree spacing on individual tree and whole stand growth rates (Barrett 1965, 1970, 1973). Barrett also attempted to determine whether suppressed ponderosa pine could develop into large, old trees through silvicultural manipulations and how the understory vegetation affected growth rates

[2] U.S. Department of Agriculture, Forest Service. 1937. Report on the Lookout Mountain unit of the Pringle Falls Experimental Forest. On file with: U.S. Department of Agriculture, Forest Service, Pacific Northwest Research Station, 1220 SW 3rd Avenue, Suite 1400, Portland, OR 97204.

in thinned stands. The sample plots established by Barrett in 1957 are protected, and the Levels-of-Growing-Stock Study continues as long-term work with regional and national significance.

The Lookout Mountain unit was chosen as the location for the LMS for many reasons. Past research efforts in the neighboring Pringle Butte unit (Levels-of-Growing-Stock Study and ponderosa pine-grand fir spacing study) are regionally and nationally significant. The ongoing research can be used to leverage the LMS for a more robust understanding of the short- and long-term impacts of natural disturbance and management on ponderosa pine forests and to help identify some of the uncertainties and unknowns of forest community response. Additional justifications for using the Lookout Mountain unit for this study are as follows:

- Designated a field laboratory for ponderosa pine since 1931.
- A methods-of-cutting study in the 1930s shifted the intensity of harvest to more sustainable levels and left large blocks of old-growth ponderosa pine onsite.
- A lack of broadscale severe natural disturbances since 1845, except localized incursion of a 1914 fire.
- A lack of major Euro-American settlement within the unit.
- Experimental reduction of fire risk provides pragmatic benefits of (1) protection of the investment in long-term silviculture trials and (2) mitigation of wildfire transmission into residential communities developed at the boundaries of PFEF.

Chapter Summary

The LMS site encompasses potential variation in physiography and initial vegetation community composition. Furthermore, historical, long-term research activities based on relatively small plot designs fall within the study bounds. The LMS addresses both physiographic and compositional variations through delineation of experimental units and the use of an imbalanced randomized block experimental design. The preservation of small plots of important historical studies within the study footprint is important for future sustained monitoring.

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Part II:
Experimental Design and Operational Implementation

Chapter 3: Treatments and Layout

Lorelle M. Sherman and Paul D. Anderson[1]

Introduction

Land use changes and forest management practices have significantly altered dry conifer forests in Oregon since Euro-American settlement, as discussed in chapters 1 and 2. Because of its early designation as an experimental forest in the 1930s, the Pringle Falls Experimental Forest (PFEF) was largely spared from the harvest activity typical of the region in the 20th century. Limited past management activity may allow forest development to track shifting environmental conditions (however lagged), and therefore, the PFEF provides an ideal location for large-scale research on forest health and resilience. However, prior and current studies located at the PFEF are at risk of damage resulting from natural disturbances.

The Lookout Mountain Thinning and Fuels Reduction Study (hereafter LMS) was undertaken to accomplish two major objectives: (1) to mitigate wildfire risk to long-term and ongoing research being conducted at the experimental forest; and in doing so, (2) to research the effectiveness of different mitigation treatments for decreasing wildfire risk as well as to assess the collateral effects of those treatments on other ecological processes and values.

Specifically, the treatment prescriptions in the LMS were designed to decrease risk of stand-replacing fire by reducing stand density and understory fuels loading. All treatments included thinning of the overstory coupled with understory mastication and broadcast burning. The treatments differed in the degree and spatial patterning of overstory density reduction through thinning.

Thinning for Density Management

Thinning is a common silvicultural practice used to manage tree species composition and growing space, and therefore the access that individual trees in a forest stand have to site resources (e.g., water, nutrients, light). Adequate access to site resources is important to maintaining individual tree vigor and growth and avoiding biotic and abiotic stressors that can result in loss of vigor or mortality. Overly dense stands increase the risk for stress-induced loss of vigor and mortality, increased fuels and combustibility, and therefore increased risk for severe wildfire. In mixed-species stands, thinning can be used to alter the composition by retaining tree species with desirable attributes, such as fire tolerance or those that provide other ecological functions or cultural and economic values. The LMS objectives were to identify the degree to which thinning stands (characteristic of the central

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Oregon pumice zone) could enhance stand vigor and to mitigate risks related to overcrowding, preferably retaining tree species and individuals best adapted to anticipated stressors and a more frequent, lower intensity fire regime.

The LMS thinning treatments, as defined by Youngblood (2009), were specified in terms of interrelated metrics—basal area (BA), stand density index (SDI), and upper (limit of the) management zone (UMZ).

Basal area is a common stand metric that describes the cross-sectional area of trees at 1.37 m from the ground. As an absolute density metric, the degree of site occupancy, or crowding, reflected by a given BA value can vary markedly by site quality, species, and tree age. In contrast, SDI and UMZ quantify relative density across a wide variety of stand conditions and are relatively independent of site quality and stand age (Long and Daniel 1990). This makes SDI and UMZ ideal metrics to use in the LMS because varying elevation and aspect within the Lookout Mountain unit result in variable site quality, which is evident as spatially varying plant communities across the study landscape.

SDI is a measure of relative stand density based on the relationship between mean tree size and the number of trees per unit of land area in a fully stocked stand. Reineke (1933) originally developed SDI for use on single-species, single-cohort stands. However, Shaw (2000) demonstrated application of SDI to mixed-species stands while considering individual species characteristics. Maximum SDI can be interpreted as the maximum tree density, or full stocking, possible for a given mean tree size in a self-thinning stand. Maximum SDI for a species varies by plant association, and relative density is expressed as a percentage found by comparing the measured SDI of a particular stand to the maximum SDI characteristic of the plant association.

Cochran et al. (1994) extended the concept of relative density to set a UMZ, or upper density limit, at which tree-to-tree competition leads to the suppression of trees. Specifically, the UMZ of a stand represents the level of stocking, or relative density, beyond which there is imminent risk of substantial losses of overstory trees due to resource limitation and drought- and other stress-induced bark beetle infestations. As with SDI on which it is based, the UMZ for a species varies with plant association. Cochran et al. (1994) recommended setting the UMZ at around 75 percent of full stocking (or maximum SDI) for trees in the Wallowa-Snake and Blue Mountain provinces to minimize density-related mortality.

The UMZ accounts for site differences as reflected by stand density relationships for specific plant communities. Plant associations are groups of plant communities that respond similarly to management activities and remain uniquely identifiable throughout different states of development (Volland 1988). Simpson (2007) refined Volland's forest plant association classifications for the central Oregon pumice zone. However, for consistency with previously collected data, we refer to Volland (1988) classifications in this study. For individual tree species, SDI values at full stocking, and thus the UMZ values, can differ among plant associations. Cochran et al. (1994) demonstrates adjustment of SDI and UMZ estimates based on plant association, with exceptions for lodgepole pine (*Pinus contorta*) and ponderosa pine (*P. pine*) (see Cochran et al. 1994: 7). Booser and White established UMZ values for individual species specific to the Deschutes National Forest in central Oregon as reference for those who do not have local SDI information.[2] Sampling data were used to develop SDI targets specific to the study area for this study.

The thinning treatments evaluated in the LMS include:

1. **100 percent UMZ (hereafter 100 UMZ):** Thinned from below to 100 percent of the UMZ specified for the dominant plant association, based on metric SDI values for ponderosa pine. Removing trees from the subdominant crown classes or smaller diameters with the objective to improve growth and overall vigor of trees in the upper crown classes or larger diameters reduced stand density. Any clumped or random distribution of large trees was intended to be preserved. Ground and surface fuel reduction using a combination of mastication and underburning followed thinning.
2. **75 percent UMZ (hereafter 75 UMZ):** Thinned from below to 75 percent of the UMZ as above. Mastication and underburning followed thinning.
3. **50 percent UMZ (hereafter 50 UMZ):** Thinned from below to 50 percent of the UMZ as above. Mastication and underburning followed thinning.
4. **75 percent UMZ + Gaps (hereafter 75 UMZ + Gaps):** A free thin across all diameters designed to initiate a transition to an all-aged stand structure (e.g., uneven-age management). Because this treatment involved limited removal of codominant and dominant trees, small canopy gaps (0.1 ha) were created to provide space for regeneration. This was achieved by either augmenting existing gaps or creating new gaps. Mastication and underburning followed overstory harvest.
5. **Reference (untreated):** Current structure retained (i.e., no manipulation via thinning or fuel reduction). There was no followup ground or surface fuel reduction with mastication or underburning.

Understory Fuels Reduction

Slash Pile Burning

Understory vegetation and dead and down wood contribute to subcanopy fuels that can carry fire through a stand and potentially into the forest canopy. Stand thinning typically generates slash that can considerably increase the abundance of surface fuels beyond the preexisting live and dead fuels. Slash pile burning mitigates wildfire risk by reducing the concentrated fuel loads of coarse downed wood. A loader piled slash in preparation for burning piles in the fall.

[2] Booser, J.; White, J. 1984. Calculating maximum stand density indexes (SDI) for Deschutes National Forest plant associations. Unpublished paper. On file with: U.S. Department of Agriculture, Forest Service, Deschutes National Forest, 63095 Deschutes Market Road, Bend, OR 97701.

Mastication

Mechanical treatment of understory vegetation is commonly used to alter fuelbed structure before prescribed burning treatments. Thinning operations were followed by mastication to further reduce the height of subcanopy fuels, primarily small-diameter, undesirable conifers and shrubs. Undesirable biomass was defined as trees that were >0.31 m tall, shrubs, and slash <22.9 cm in diameter and >0.91 m long. Mastication of undesirable biomass occurred across the entire study area except where it was not safe to do so (e.g., rocky outcroppings).

Broadcast Underburning

Prescribed broadcast underburning emulated natural fire regimes for Lookout Mountain and reduced buildup of understory biomass. Individual broadcast burn plans were developed for each experimental unit to account for differences in physiographic features and fuel accumulations. In general, the goal of burn plans was to reduce 1- and 10-hour surface fuels by 50 to 100 percent and shrubs by 25 to 100 percent, with less than 50 percent crown scorch. In some cases, broadcast burning included both backing and strip head fire when weather and fuels moisture conditions were conducive to low flame heights.

Weather conditions that were conducive to achieving broadcast underburning goals were as follows:

- Temperatures between 10 to 27 °C
- Relative humidity between 20 to 44 percent
- Windspeeds of 3.2 to 14.5 km/hour

All broadcast underburning, except for about half of the area within unit 12, occurred in the spring season to limit fire intensity and the risk of unintended tree mortality. The northeastern portion of unit 12 had deep duff and surface fuel accumulations; broadcast burning occurred during autumn to allow time for fuels to cure and to ensure fall rains would extinguish any smoldering hotspots.

Experimental Design

The LMS comprises a randomized complete block design with four blocks within the Lookout Mountain unit at PFEF. Block delineation was based on relatively similar elevation, aspect, and plant associations to minimize experimental unit differences within a block (fig. 3.1). Each block was further delineated into five experimental units, except for one block, which was delineated into four experimental units to adjust for the northern spotted owl (*Strix occidentalis caurina*) (NSO) detections as outlined in chapter 4. Each unit was randomly assigned one of four thinning treatments or left untreated as a Reference (untreated control). Experimental unit boundaries were delineated based on topography and existing roads where possible.

The 75 UMZ + Gaps treatment and the Reference have greater risk of catastrophic wildfire because they retain much of the structural characteristics associated with overstocked stands (e.g., high fuel loads, presence of ladder fuels, non-fire-adapted species). Therefore, units assigned to 75 UMZ + Gaps treatment and Reference were intentionally smaller in areal extent. Specifically, for each unit randomly assigned to the 75 UMZ + Gaps treatment, the unit was split in half to accommodate both a 75 UMZ + Gaps treatment unit and a Reference unit. The SDI values resulting from the 75 UMZ + Gaps treatment were greater and were expected to have less risk reduction than 100 UMZ, 75 UMZ, and 50 UMZ treatments. The intentional assignment of 75 UMZ + Gaps treatment and Reference to units half the size of 100 UMZ, 75 UMZ, and 50 UMZ was based on the assumption that variation within experimental units was similar regardless of experimental unit size.

Before treatment implementation, a sampling area of 6.25-ha was established within each experimental unit based on a gridded array of 25 permanent sampling plots with 50-m spacing between plot centers (fig. 3.2). These sampling plots were used for the core vegetation study. Sampling area grid locations were determined by an initial randomly selected point within a buffered area of each unit. Buffers were placed at a minimum of 50 m from unit and stand boundaries, roads, and skid trails. The specific grid configuration (numbers of rows and columns) differed among experimental units based on the polygon shape. Gridded sampling plot centers were established pretreatment using a compass and measuring tape based on GPS coordinates extracted from aerial imagery. All sampling plot centers were georeferenced and permanently monumented with fiberglass rods in 2014 through 2016 after treatment implementation (fig. 3.2).

Data on overstory tree structure and composition and understory vegetation were collected based on 400-m^2, 11.28-m-radius, circular sampling plots before treatment and in the first year after underburning. The consequences of the sampling plot size included a lack of posttreatment retained live overstory trees in some plots and high variability in attributes among sampling plots within an experimental unit. Other landscape-scale silvicultural studies in the Western United States used plot sizes ranging from <0.1 (Curtis et al. 2005, Youngblood et al. 2006) to 0.2 ha (Barrett 1983, Cochran and Barrett 1999, Oliver 1979). Curtis et al. (2005) noted that the size of the sampling plot is not as important as the number of sampling plots in landscape-scale studies where the unit of analysis is the experimental unit, so long as the number of sampling plots adequately captures stand condition.

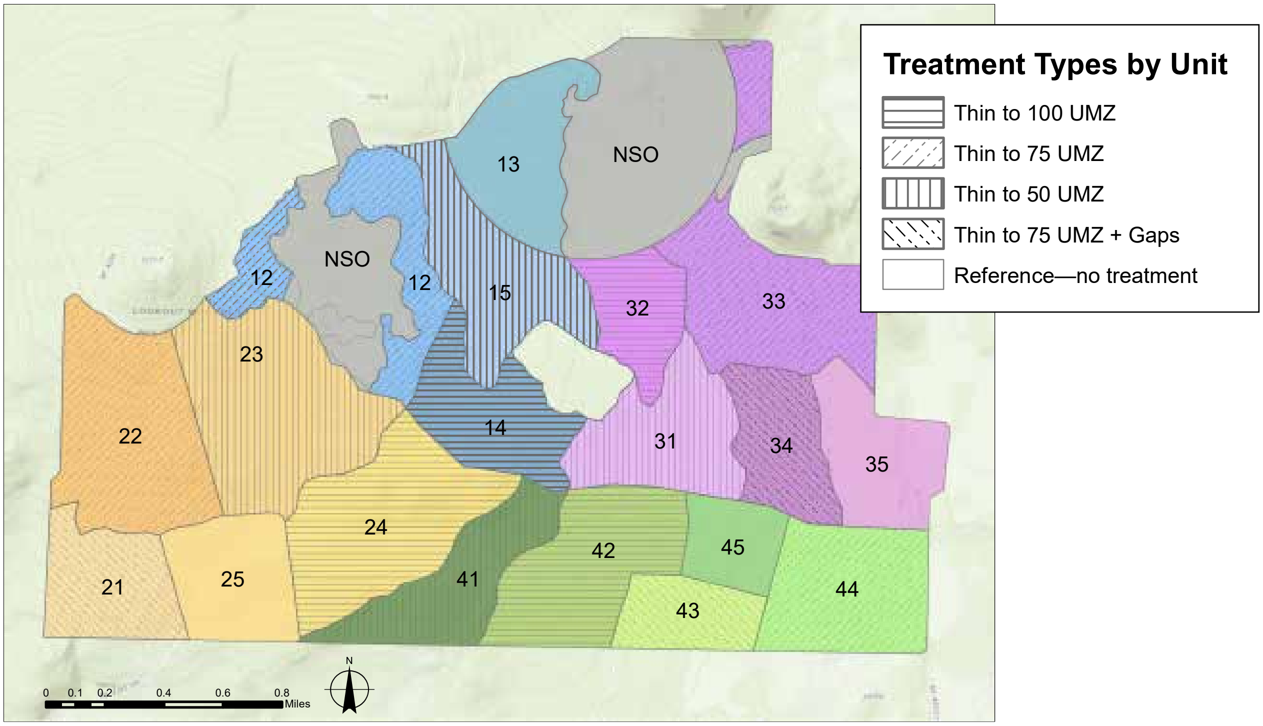

Figure 3.1—The layout for the Lookout Mountain core vegetation study. Polygons delineate experimental units, which are labeled with designated 2-digit unit codes. The first digit as well as polygon color (block 1 = blue, block 2 = orange, block 3 = purple, block 4 = green) designate blocks. Polygon patterns designate treatments. Gray polygons (labeled NSO) were removed from the study, and unit 13 was reassigned to the Reference because of nesting, roosting, and foraging habitat of northern spotted owl. UMZ = upper management zone.

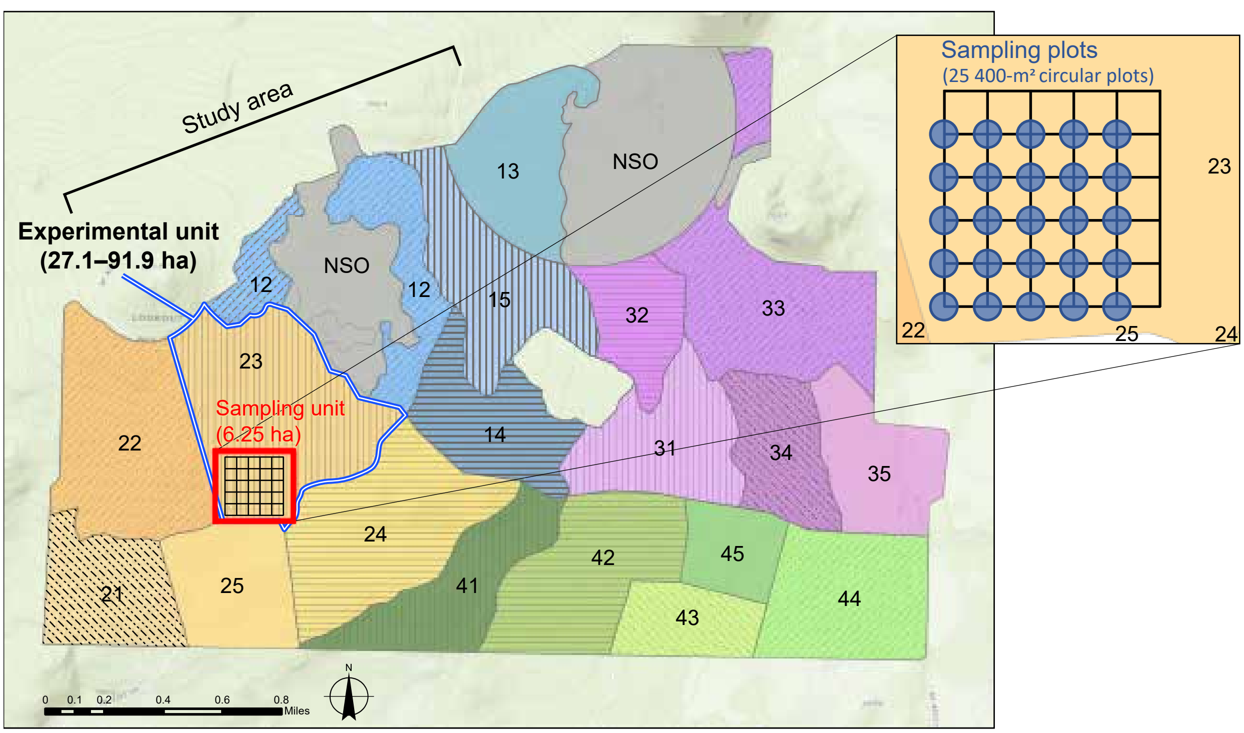

Figure 3.2—Lookout Mountain Thinning and Fuels Reduction Study experimental design, showing the four blocks and the five treatments (see fig. 3.1 for descriptions). For each treatment, a 6.25-ha sampling area (square grid) was delineated; within the sampling area, a grid was established of 25 permanent 400-m^2 circular plots (circles, in inset), centered on grid points.

Chapter Summary

The LMS is one of few operational-scale experiments in dry, fire-prone forests east of the Cascade crest. Combinations of overstory thinning coupled with understory fuels reduction were chosen to test the effectiveness of active management in reducing the risk of stand-replacing disturbance in dynamic environmental conditions. Treatments were underpinned by the following management objectives:

1. Restore stand structure to more fire-adapted conditions by reducing stand density, favoring fire-tolerant tree species, and reducing understory and ladder fuels.
2. Sustain fire-adapted conditions into the future with additional fuels reduction treatments as needed.
3. Adaptively select silvicultural treatments as trends in pyrogenicity of posttreatment stands and the duration of treatment effects are revealed through the characterization of vegetation dynamics.

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Chapter 4: Environmental Assessment and Northern Spotted Owl Adjustments

Lorelle M. Sherman and Paul D. Anderson[1]

Introduction

The purpose of the experimental forest designation is to facilitate ecological research and long-term research (Forest and Rangeland Renewable Resource Research Act of 1978). The Forest Service manages the Pringle Falls Experimental Forest to promote long-term research on forest management strategies, especially those that increase the resistance and resilience of the forest to wildfire and insect attack. Experimental forests are subject to National Environmental Policy Act (NEPA) of 1969 but with consideration of the research objectives of the forest. The NEPA process requires federal agencies to assess the environmental impact of a proposed action, draft an environmental impact statement, and accept and consider public comment. The Forest Service had to adhere to NEPA procedures before implementation of thinning and fuels reduction treatments as part of the Lookout Mountain Thinning and Fuels Reduction Study (hereafter LMS). Supported by the final environmental impact statement, the Forest Service rendered a decision to proceed with the management actions to implement the LMS. However, various interest groups subjected that decision to a series of litigations.

As of 2021, the northern spotted owl (*Strix occidentalis caurina*) (NSO), an old-growth obligate forest species, retains its threatened species status. Before implementation of LMS harvest treatments, Forest Service personnel conducted surveys for the presence of NSO. Detection of a nesting pair in 2011 near the proposed LMS treatment units warranted reconsideration of the original study layout.

To differing extents, litigation and NSO detection led to rethinking of the LMS layout between 2011 and 2012. This chapter reviews the implications of two factors: (1) reaffirming the role of experimental forests as designated units suitable for exploring novel management approaches and (2) modifying the experimental design and layout to establish a buffer zone for the nesting owl pair.

NEPA Process

In 2007, the Pacific Northwest Research Station, in collaboration with the Deschutes National Forest Bend-Fort Rock Ranger District wrote a study plan, *Forest Dynamics After Thinning and Fuel Reduction in Dry Forests*, for the LMS (Youngblood 2009). The study plan identified six research questions aimed to reduce the risk of catastrophic wildfire and insect attack:

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1. What set of fuel reduction treatments best accelerates the development of large trees while over the long-term reintroduces natural disturbance processes that provide greater ecosystem resiliency?
2. What is the long-term influence of climate change interacting with a set of fuel reduction treatments on vegetation dynamics and forest structure?
3. Can single cohort stands be readily converted to multicohort stands?
4. Do multicohort stands share the same risks of multiple, interacting stresses as single-cohort stands?
5. How does the dominant shrub, giant chinquapin (*Chrysolepis chrysophylla*), respond in the near-term to a set of fuel reduction treatments?
6. How does the residual stand structure resulting from a set of fuel reduction treatments interact locally and in the near term with wind to cause additional structural changes?

In April 2008, the Forest Service released a scoping letter for public comment and published a notice of intent to prepare an environmental impact statement (EIS) for the proposed LMS plan. The Confederated Tribes of Warm Springs, Burns Paiute Tribe, and Klamath Tribes were included in the public scoping and comment period and expressed no concerns about the project. From 2008 through 2009, the LMS plan underwent internal technical review and blind, external technical and statistical reviews before the director of the Pacific Northwest Research Station approved the plan in 2009. The Forest Service released the draft environmental impact statement (DEIS) in September 2009 and allowed for a 45-day comment period under the provisions of the NEPA In November 2009, the U.S. Environmental Protection Agency wrote a letter of support for the proposed project, citing that it was "a means to address the risk of severe insect epidemic or catastrophic fire, and as an opportunity to study forest dynamics after thinning and fuels reduction in dry forests." The DEIS outlined three alternative plans of action, including the proposed action and two alternatives:

1. Alternative 1, No Action.
2. Alternative 2 (preferred), the Proposed Action, "would result in logging approximately 27–29 million board feet of timber on [1034 ha]—or about 70 percent of the trees larger than [15 cm] in diameter at breast height. It would 'thin from below' by cutting the smallest trees in an area until the target SDI for a particular area is reached, thereby retaining the largest trees in that area…"[2]
3. Alternative 3, developed to respond to the issue of maintaining NSO habitat, "uses the same research design and has the same target numbers as Alternative 2 for SDIs… However, Alternative 3 would leave undisturbed

[2] Retrieved from https://www.elr.info/litigation/42/22/league-wilderness-defenders-v-united-states-forest-service.

[150 ha] of spotted owl habitat that would be thinned under Alternative 2. This would reduce the total logged area in the Project by about 15 percent."[3]

The final environmental impact statement (FEIS)[4] and record of decision (ROD) selecting alternative 2 were signed in May 2010. The U.S. Department of the Interior Fish and Wildlife Service (USFWS) provided a 2010 biological opinion stating that the LMS was not likely to jeopardize the continued existence of the NSO. It was determined that implementation of proposed activities would have no impact on threatened or endangered fish or plant species and would either have no impact on sensitive wildlife species and associated habitat or might impact individuals or habitat but not cause a trend toward federal listing (FEIS: 128–144, 226–227, 239).

Administrative Appeals

After the signing of the ROD and FEIS, two environmental groups filed appeals against the decision of the Forest Service to move forward with alternative 2 of the LMS. The appellants stated the following:

Appellant 1. Oregon Chapter Sierra Club, League of Wilderness Defenders-Blue Mountain Biodiversity Project, and Cascadia Wildlands (appeal number 10-06-00-18-215, 7 May 2010)

a. Inappropriate use of experimental forests
b. Arbitrarily contrived purpose and need
c. Contrary to the intent of the Eastside Screens
d. Inadequate range of alternatives
e. Adverse effect to wildlife
f. Scientific controversy over density reduction and fire return interval
g. Scientific controversy over treatments in mixed-conifer forests
h. Climate change

Appellant 2. Oregon Wild (appeal number 10-06-00-19-215, 10 May 2010)

a. Purpose and need are not clear and compelling
b. Project is contrary to current ecological priorities
c. Analysis is inadequate
d. Response to the public is inadequate

Following the filing of appeals in May 2010, the Forest Service hosted two field trips with appellant groups. After review of the appellant statements in June 2010, a government review team recommended denial of appeals requests and the responsible official's decision be affirmed on all points. The next day, the appeals were resolved with the decision that the administrative procedures carried out to date regarding the LMS complied with law, regulation, and policy.

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[3] https://www.elr.info/litigation/42/22/league-wilderness-defenders-v-united-states-forest-service.

[4] Information about the final environmental impact statement can be found at https://cdxapps.epa.gov/cdx-enepa-II/public/action/eis/details?eisId=86796#links.

Federal District Court Appeals

On 22 September 2010, the League of Wilderness Defenders filed a lawsuit against the Forest Service in Federal District Court citing the Administrative Procedure Act of 1946. This act states that a court may reverse a final agency action if it finds that the action is arbitrary, capricious, an abuse of discretion, or otherwise contrary to law; the court is not a finder of disputed facts but instead the role of the court is to determine whether the evidence in the administrative record permitted the agency to make the decision it did. In May of 2011, the Federal District Court heard the following challenges by the plaintiff:

1. Failed to consider a reasonable range of alternatives.
2. Failed to ensure scientific accuracy and integrity of the FEIS; failure to disclose that thinning would remove more trees than would be expected to die from fire or insects.
3. Failed to consider impacts of effects on snag levels and snag-dependent Management Indicator Species and NSO.

District Court response to plaintiff challenge 1:

a. The research aspect of the proposed action warranted the narrowing of the consideration of alternatives, because the purpose of an experimental forest is to conduct research.
b. The proposed action is intended to protect large, old ponderosa pine throughout the study area by thinning from below and retaining the largest trees. Altering the resultant stand densities by retaining all trees 21-inch (53.3-cm) d.b.h. and larger would compromise the validity of the experimental design and deny the reduction of risk of loss to bark beetles.
c. The Forest Service reasonably considered the range of alternatives at hand but found that they were inconsistent with research objectives of the proposed action. There is no expectation that the Forest Service must consider all possible alternatives.
d. Considering alternatives that eliminate research objectives in favor of further limiting tree thinning over alternatives that prioritize research objectives and may result in short-term tree mortality would be short-sighted when the proposed action utilizes research activities on thinning and management techniques to reduce risk to widespread tree mortality.

District Court response to plaintiff challenge 2:

a. Thinning is "a practical means of lowering the probability of serious mortality from mountain pine beetle… and perhaps western pine beetle." "A review of the average mortality rate tables reasonably supports the conclusion that larger mortality rates due to beetle infestation occur in greater rather than lower densities even after treatments."

b. Records of average mortality rate tables support the conclusion that higher stand densities lead to higher tree mortality rates due to beetle infestations, and while mortality rates immediately after treatment implementation are greatest, high mortality rates at longer time periods were restricted to stands with high stocking levels.
c. Stands can reach UMZ (upper management zone) and continue to grow, but it is not sustainable and remains at imminent risk of catastrophic loss.

District Court response to plaintiff challenge 3:

a. The FEIS uses the modeling technology and resources at hand to analyze the impact of the proposed action on snags and snag habitat. This constitutes a "hard look."
b. NEPA requires the Service to take a hard look at and to publicly disclose the environmental consequences before implementing the proposed action. It does not necessitate the elevation of environmental concerns.
c. The inability to exactly quantify tree mortality as a result of thinning operations or if stand densities grow unchecked does not equate to a failure of the agency to take a hard look at ecological consequences.

The District Court ultimately ruled that the plaintiff (League of Wilderness Defenders-Blue Mountains Diversity Project) failed to raise questions substantial enough to warrant a deliberative investigation and denied the appeal on 14 June 2011.

Ninth Circuit Court of Appeals

The League of Wilderness Defenders-Blue Mountains Biodiversity Project filed an appeals case with the Ninth Circuit Court of Appeals on 1 September 2011, alleging that the Forest Service:

1. Failed to consider reasonable range of alternatives.
2. Failed to ensure scientific accuracy and integrity of the FEIS.
3. Failed to take a hard look at environmental impacts.

The Circuit Court completed a de novo review of the District Court's decision on summary judgement, followed by oral arguments in Portland, Oregon, on 7 March 2012. The Ninth Circuit Court of Appeals affirmed the District Court ruling in favor of the Forest Service. FindLaw[5] offers the following review on the court's decision:

1. "The agency need only evaluate alternatives that are 'reasonably related to the purpose of the project.'"
2. "The League contends that, as a result of the narrowness of the stated purpose and need, only a single alternative—the Study Plan—could satisfy them. However, the statement does not incorporate the specifics of the

[5] https://caselaw.findlaw.com/us-9th-circuit/1607559.html.

Plan's proposed experiment. Rather, the statement refers to the Plan because it contains an extensive discussion of the research objectives and working hypotheses behind the six study questions."

3. "The League bases its challenge in part on an argument that the Service created the Study Plan 'prior to initiating the NEPA process'… Here, the Service began the NEPA process nearly a year before the Plan was finally approved. Based on the record before us, we conclude that the purpose and need in the challenged EIS adequately informed decisions by the Service and participation by the public."
4. "An agency need not consider alternatives that extend beyond those reasonably related to the purposes of the project… The EIS only needs to consider in detail alternatives that would address both Project's stated purpose and needs by meaningfully reducing the risk of beetle infestation and wildfire while attempting to answer the six research questions."
5. "In another context, an EIS analyzing in detail two action alternatives that differed only in proposed acreage would likely be inadequate. But we agree with the district court that the special circumstances of a research project in an experimental forest 'necessarily narrowed consideration of alternatives.'"
6. "The Study Plan, which underwent both internal and external peer review, also concluded that trees within the Unit currently have structural characteristics which place them at imminent risk of catastrophic loss to bark beetles. We therefore cannot say that the EIS lacks scientific integrity or misrepresents scientific literature..."

The Circuit Court found that the Eastside Screens (Powell 2013) represent a Forest Service policy that this review did not need to meet because the retention of trees according to screen limits would have precluded attainment of the targeted experimental densities. According to Pioneer Law Group,[6] "the EIS explained that the Service briefly considered the League's proffered alternative but eliminated it from detailed study because modeling indicated that retaining all trees greater than 21 inches (53.3 cm) in diameter would not achieve the targeted stand densities and thus not fulfill the project objectives. Under prior case law, the court explained, agencies need not consider in detail proposed alternatives that fail to meet specifically identified targets or densities. Further, the 21-inch (53.3-cm) alternative would not fulfill the research purpose of the project. The diameter-limit alternative would not yield statistically valid comparative data among the areas specified for different densities of logging."

[6] https://www.pioneerlawgroup.net/forest-services-eis-experimental-forest-thinning-fuels-reduction-research-project-satisfied-nepa/.

Northern Spotted Owl Detection

The original scope of the LMS included consideration of NSO nesting, roosting, and foraging habitat by providing the opportunity to examine the treatment response with respect to short-term changes in NSO habitat attributes. Ongoing monitoring for NSO presence within the boundaries of the Lookout Mountain unit yielded no detections before 2011. In May 2011, ongoing NSO surveys led to the detection of a single mated pair within the Lookout Mountain unit. In spring 2012, a nest tree was detected in block 3 of the LMS area where the pair produced offspring that fledged in June 2012.

The Forest Service and USFWS used the NSO detection as an opportunity to collaborate in creating a revised NSO critical habitat rule. In 2011, the USFWS recovery plan described management actions to restore NSO habitat based on a landscape-scale active restoration strategy that departed substantially from current, mostly passive, reserve-network strategies of the Northwest Forest Plan (NWFP). The NWFP recognized that land managers have a variety of management obligations in dry, disturbance-prone forests, and NSO may not be the sole driver of all landscape management decisions (USFWS 2011). Also, projects designed to provide long-term forest resiliency and restore natural forest dynamic processes are needed even if there are short-term effects to NSO habitat.

NSO core habitat within the LMS area was delineated as a circle centered over the nest tree located in 2012 (fig. 4.1). All NSO core habitat was withdrawn from treatment areas, and all entry and activity within this zone was prohibited. A seasonal restriction on a broader circle outside of the core habitat zone representing the NSO home range was enacted from 1 March through 30 September (fig. 4.1). In addition to NSO surveys, the LMS supported research activities focused on NSO prey abundance and habitat, stand microclimate and microsite variation resulting from treatments, and an inventory of stand structure and composition.

The inclusion of owl nesting core habitat led to reconsideration of the original unit delineations within block 1 (fig. 4.2). Before treatment implementation, the original layout was modified; experimental unit 13 was reassigned to Reference (originally assigned to a 75 upper management zone [UMZ] + Gaps treatment, see chapter 3 for treatment descriptions). Furthermore, experimental unit 11 was combined with unit 12, and the entirety was assigned 75 UMZ. The net effect of changes due to the occurrence of a nesting pair of owls was a decrease in the total number of experimental units to 19—100 UMZ, 75 UMZ, 50 UMZ, and Reference had four replicates; and 75 UMZ + Gaps had three replicates. The decision to remove one 75 UMZ + Gaps treatment replicate highlights the importance of maintaining four replicates for each of three thinning prescriptions (100 UMZ, 75 UMZ, and 50 UMZ). This preserved the robustness of the test of effects of harvest intensities on forest attributes. The resultant experimental design as implemented in 2012 was an incomplete randomized block design with either three or four replicates of five treatments (four thinning treatments and one Reference) distributed among four blocks. The station statistician performed a power analysis to assess and confirm the statistical validity of the overall design.

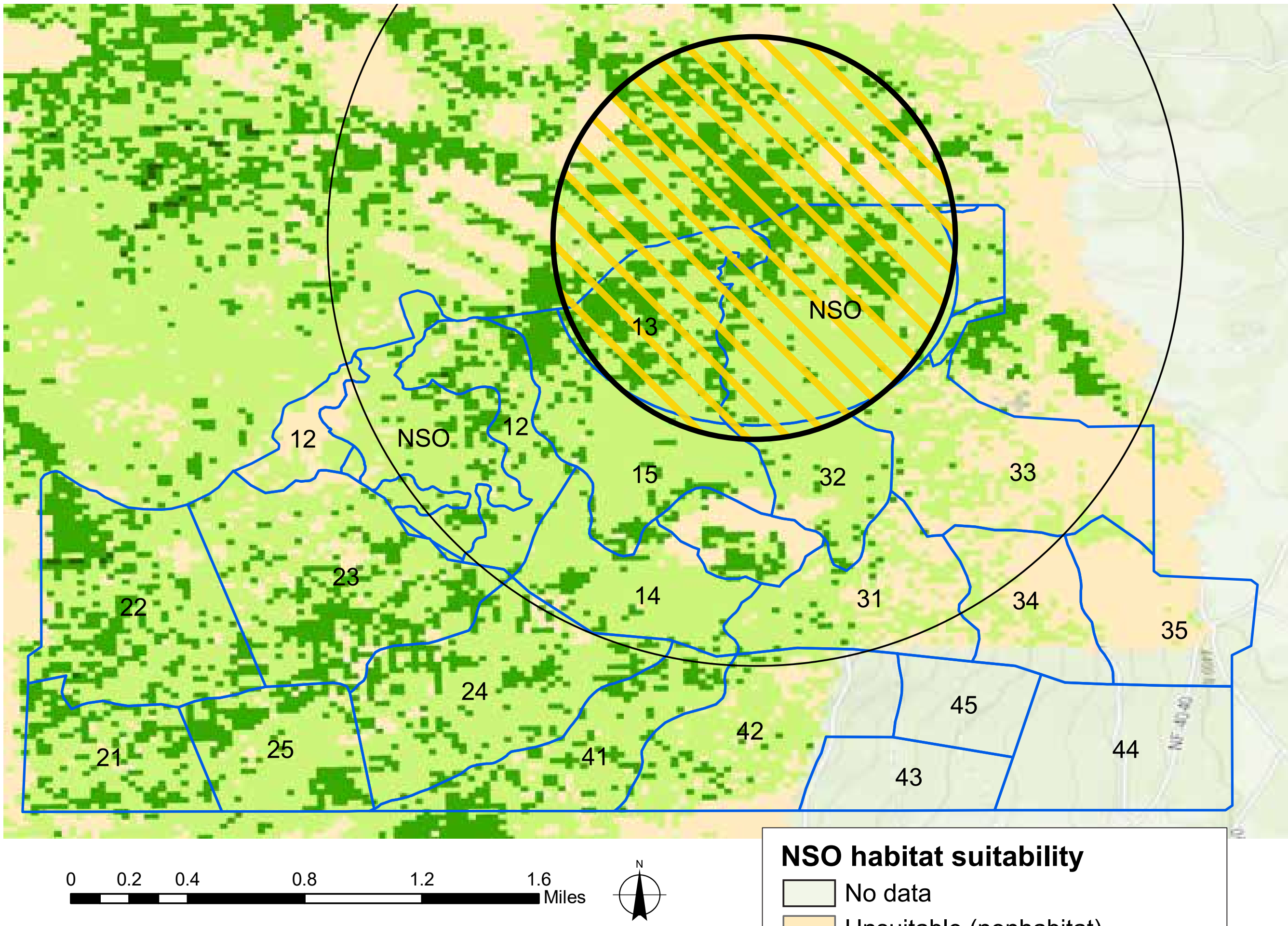

Figure 4.1—The area of northern spotted owl (NSO) nesting, roosting, and foraging habitat within and surrounding the Lookout Mountain Thinning and Fuels Reduction Study area. The circle with hatch marks represents the core habitat area centered on the nest tree detected in 2012. The broader circle represents the NSO home range area. Adapted from Davis et al. 2022.

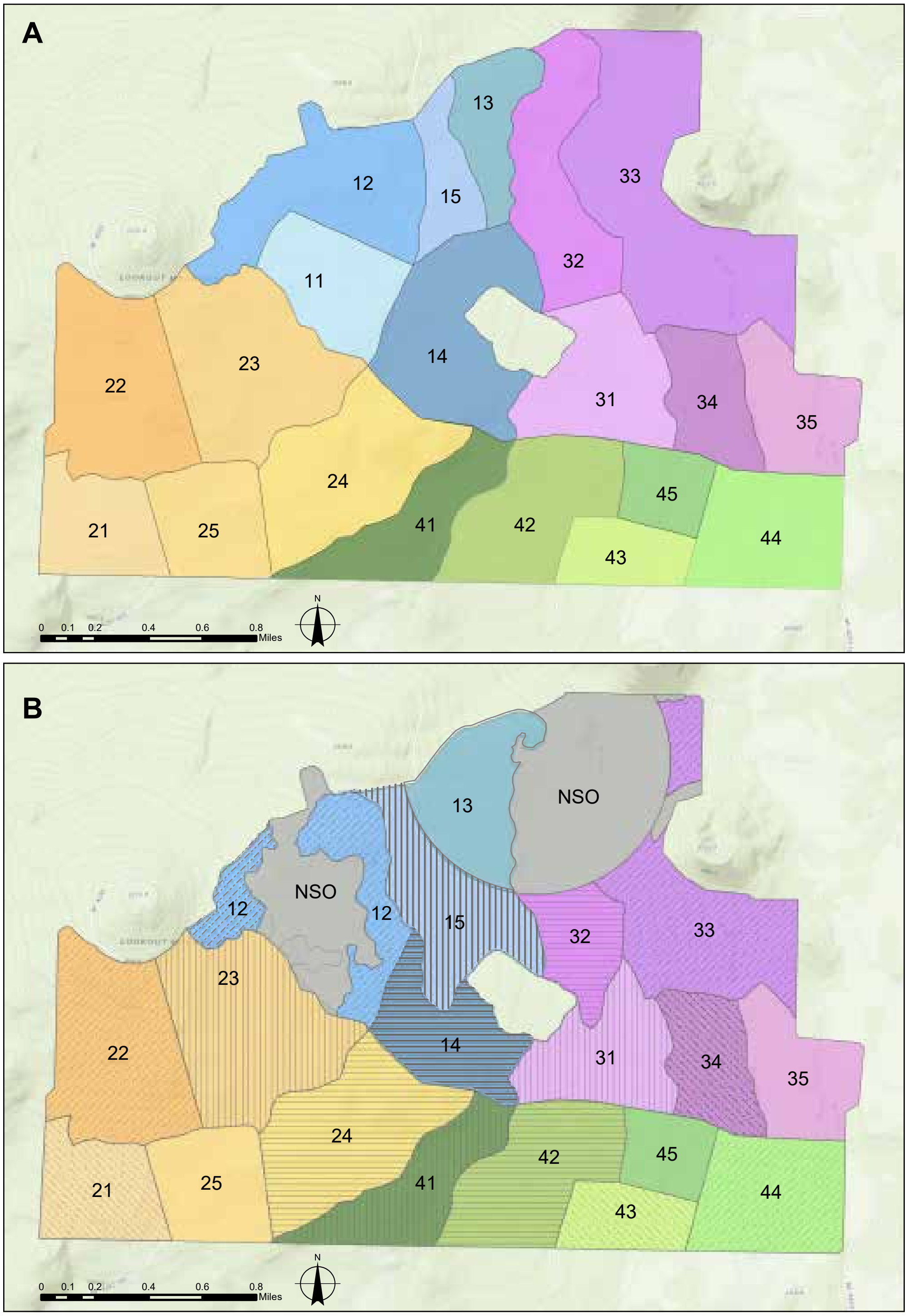

Figure 4.2—The Lookout Mountain Thinning and Fuels Reduction Study experimental design: (A) before and (B) after the removal of northern spotted owl (NSO) nesting, roosting, and foraging habitat and subsequent reorganization of unit layouts.

Chapter Summary

The Forest Service followed NEPA procedures involving public scoping and comment, internal technical review, and external technical and statistical reviews before approving implementation of the LMS. Environmental groups appealed the Forest Service NEPA decision citing inappropriate use of experimental forests and the failure of the agency to adequately consider alternatives. The courts ultimately concluded that the Forest Service was in compliance with both appropriate use of experimental forests and the NEPA process. The LMS originally supported research that examined the ecological response of NSO nesting, roosting, and foraging habitat to a suite of thinning and fuel reduction treatments. The detection of a nesting pair of NSO within the LMS led to the withdrawal of NSO nesting, roosting, and foraging habitat and the removal of this research opportunity.

Changes in the treatment unit configuration also resulted in a revised experimental design; the modifications decreased the number of experimental treatment units to 19 and resulted in the loss of one 75 UMZ + Gaps treatment replication. The station statistician assessed and confirmed the statistical validity of the modified design with respect to primary research questions.

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Chapter 5: Harvest and Fuel Contracts

Lorelle M. Sherman and Paul D. Anderson[1]

The large-scale nature of the Lookout Mountain Thinning and Fuels Reduction Study (hereafter LMS) required three separate harvest and fuel treatment contracts staggered over 5 years to accomplish implementation of silvicultural treatments.

Ponderosa pine (*Pinus ponderosa*) is an economically important species in the interior Pacific Northwest, providing conventional timber products as well as potentially emerging innovations, including laminated structural products or marketable biomass. Manipulating growing stock to produce a desired future stand structure and prevent losses from a stand-replacing disturbance is one of the greatest challenges facing silviculturists. Smaller dimension woody biomass, which are often costly to remove, are products of thinning treatments intended to reduce stand density. However, small-dimension wood material may have potential as a marketable resource where money acquired from sales could be reinvested into forest restoration project costs. Hence, the removal and effective use of small-dimension materials has been a topic of growing research interest as thinning and fuel reduction treatments are increasingly used to remove biomass from overstocked forests (Lowell and Green 2001).

Thinning and fuel reduction treatments generated conventional timber and smaller dimension material outputs under three harvest and fuel treatment contracts: Bind (FS-2400-T), Extra (FS-2400-6T), and Fine (FS-2400-13T). Bind and Fine were stewardship contracts that allowed the U.S. Department of Agriculture Forest Service to trade forest products in return for the completion of forest restoration projects where the value of the forest products offsets the costs. The forest products removed as part of these contracts were sawtimber and green biomass convertible. Sawtimber is defined here as stemwood from conifer trees >22.9 cm diameter at breast height (d.b.h.) and hardwood trees >27.9 cm d.b.h., where a live tree of commercial species contains at least a 3.7-m sawlog or two noncontiguous 2.4-m sawlogs. Green biomass convertible is defined as wood fiber used as fuel.

The operational contracts were aligned with the four experimental blocks: Extra covered block 4, Bind covered block 2, and Fine covered blocks 1 and 3 (fig. 3.1). Chapter 3 provides an indepth description of the experimental design.

Surveys and Marking

All experimental units within a block were surveyed and marked before harvesting began, guided by the marking protocol. These objectives were based on historical documentation of presettlement forests in central Oregon, which suggests

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that before fire exclusion in the 1900s, fire-tolerant tree species dominated the composition, while species with low fire tolerance were sparse (Brown et al. 2000, Languille et al. 1903). The species marking guide (table 5.1) was based on a combination of presumed fire tolerance of tree species and historic tree species compositions. Lodgepole pine (*Pinus contorta*), grand fir (*Abies grandis*), and white fir (*A. concolor*), were preferentially removed because of a combination of morphological and physiological attributes that make these species susceptible to mortality from fire events typical of the study area (Brown et al. 2000, Spalt and Reifsnyder 1962).

Table 5.1—Marking guide with tree species preferences for thinning operations as part of the Lookout Mountain Thinning and Fuels Reduction Study

Cutting preference	Species
1	Lodgepole pine
2	Grand/white fir
3	Small-diameter ponderosa pine
4	Small-diameter mountain hemlock
5	Small-diameter Douglas-fir

Mountain hemlock (*Tsuga mertensiana*) is also readily killed by fire, but large-diameter trees were uncommon in the study area, so marking and removal was not imperative to achieve harvest objectives (Spalt and Reifsnyder 1962). Although ponderosa pine and Douglas-fir (*Pseudotsuga menziesii*) develop fire tolerance characteristics at maturity, small-diameter individuals are unlikely to have fully developed these characteristics, thus making them more vulnerable to fire mortality (Fitzgerald 2005, Volland and Dell 1981). Small-diameter ponderosa pine were more likely to be cut when compared with small-diameter mountain hemlock and Douglas-fir because of their greater abundance in the study area.

The intent of marking was to preserve the existing clumped or random distribution of large trees in all thinning prescriptions and the existing spatial distribution of dominant and codominant trees in all thinning prescriptions without regeneration gaps. Leave trees were designated with orange marking paint.

As a research study on a designated experimental forest, marking guides for the LMS did not reflect the 21-inch (53.3-cm) harvest limitation common to the Eastside Screens (Powell 2013) applied to National Forest System lands. As such, harvest treatments were designed to achieve target residual densities by thinning from below. However, larger trees of fire-tolerant species were also harvested as needed to reach the target stocking. Chapter 8 provides an assessment of how the treatments implemented as part of the LMS affected larger trees that the 21-inch (53.3-cm) rule may have otherwise protected.

Plots associated with the west-wide Levels-of-Growing-Stock (LOGS) Study for even-aged ponderosa pine (Barrett 1983, Cochran and Barrett 1999) at Lookout Mountain were thinned again under Bind and Extra contracts for operational convenience; these plots had separately defined marking rules that sustained the integrity of the LOGS Study. For LOGS Study plots located within the LMS timber sale area, trees designated for cutting were marked with blue paint above and below stump height. No dead trees were intended to be removed. Experimental units 24, 25, 41, 42, and 45 contained LOGS Study plots.

Harvest and Fuel Contracts

The Extra Contract: Block 4

The Extra contract was awarded to Interfor Pacific, Inc. on 13 April 2011. It included all experimental units in block 4, and the contractor completed the work in November 2011. Treatment implementation removed a total of about 47 300 m^3 of wood biomass (about 39 700 m^3 of ponderosa pine) from 199.5 ha, of which about 22 400 m^3 was merchantable timber. Table 5.2 provides a breakdown of volume removed by species and product, as well as slash disposal costs. Under the Knutson-Vandenberg Act of 1930, a portion of timber receipts funding was reinvested into

Table 5.2—The estimated quantity removed and payment by tree species, unit, and product for the block 4 Extra contract

Species	Product	Estimated quantity removed	Estimated payment	Total slash disposal deposits
		Cubic meters	- - - - - *U.S. dollars* - - - - -	
Unit 41 (48.6 ha):				5,039.36
Lodgepole pine and other coniferous species	Sawtimber	408	6,176.16	
Ponderosa pine	Sawtimber	9662	199,431.40	
White fir	Sawtimber	603	9,218.64	
All species	Green biomass convertible	835	73.75	
Unit 42 (64.3 ha):				6,677.40
Lodgepole pine and other coniferous species	Sawtimber	541	8,191.99	
Ponderosa pine	Sawtimber	12 802	264,252.45	
White fir	Sawtimber	799	12,204.96	
All species	Green biomass convertible	1107	97.75	
Unit 43 (27.1 ha):				2,814.80
Lodgepole pine and other coniferous species	Sawtimber	229	3,474.09	
Ponderosa pine	Sawtimber	5394	111,347.25	
White fir	Sawtimber	337	5,150.32	
All species	Green biomass convertible	467	41.25	
Unit 44 (59.5 ha):				6,175.20
Lodgepole pine and other coniferous species	Sawtimber	501	7,591.53	
Ponderosa pine	Sawtimber	11 836	244,321.00	
White fir	Sawtimber	739	11,296.08	
All species	Green biomass convertible	1025	90.50	

Note: Sawtimber is defined as a commercial species of tree with at least one 3.7-m sawlog or two noncontiguous, 2.4-m or longer sawlogs. Green biomass convertible is defined as wood fiber used as fuel. The purchaser made slash disposal deposits.

postharvest site development activities in block 4, including thinning and disposal of small-diameter trees and brush, hand piling, slash pile removal, and subsoiling of skid trails and landings. Mastication of undesirable (excess) trees and brush occurred on all of block 4 as part of a separate contract.

The Bind Stewardship Contract: Block 2

The Bind stewardship contract was awarded to Interfor Pacific, Inc. on 15 May 2012. It included all experimental units in block 2, and the contractor completed the work in November 2012. This sale was prepared as a stewardship contract. Mandatory stewardship projects included as part of this contract were as follows:

1. Mastication of undesirable conifers >0.31 m tall, shrubs, and slash <22.9 cm in diameter and >0.91 m long (on 297 ha)
2. Subsoiling of landings, skid trails, and temporary roads and miscellaneous areas with detrimental soil compaction to a depth of 60.96 cm where feasible (on 15 ha)
3. Delayed felling and delayed removal of 30 trees in unit 21

Treatment implementation removed a total of about 66 550 m^3 (about 47 600 m^3 of ponderosa pine) on 296.6 ha, of which about 30 200 m^3 was merchantable timber. Retained receipts totaled $1.8 million for this contract. Table 5.3 presents a breakdown of volume removed by tree species and slash disposal costs. Knutson-Vandenberg Act of 1930 funding was targeted for boundary tag removal.

This stewardship contract was an integrated resource timber contract (IRTC) and allowed the Forest Service to trade forest products in return for the completion of forest restoration projects where the value of the forest products offset the costs. These cooperative contracts are used in conjunction with traditional timber sale contracts to achieve a set of stewardship-based goals, especially in overstocked forests at high risk of insect and disease infestations and catastrophic wildfire. Typically, much of the biomass removed in these contracts does not meet utilization standards (see table 6.1 for LMS utilization standards) and has little commercial value. In return for providing services on stewardship projects, contractors received an in-kind reduced price on timber harvested as part of the silvicultural prescription.

Unlike traditional timber sales in which the highest bidder wins the contract, IRTCs involve a contractor selection process with proposals. IRTC contractor selection is based on evaluating technical/price tradeoffs and best value for achieving the end result, where the contractor obtains credits as receipts, which they can use to reduce stumpage costs. In this IRTC, the contractor obtained receipts to apply to the list of stewardship projects. The contractor earned stewardship credits at an approved offer rate when they performed stewardship project work and could use the credits to pay for included timber value in excess of base rates and required deposits.

Table 5.3—The estimated quantity removed and payment by tree species, unit, and product for the block 2 Bind stewardship contract

Species	Product	Estimated quantity removed	Estimated payment	Total slash disposal deposits
		Cubic meters	*U.S. dollars*	
Unit 21 (43.3 ha):				2,976.60
Ponderosa pine	Sawtimber	7770	268,857.12	
White fir and other coniferous species	Sawtimber	3780	103,115.40	
All species	Green biomass convertible	1220	124.99	
Unit 22 (83.4 ha):				4,270.86
Ponderosa pine	Sawtimber	11 083	383,493.72	
White fir and other coniferous species	Sawtimber	6125	167,070.12	
All species	Green biomass convertible	1116	114.26	
Unit 23 (91.9 ha):				5,460.18
Ponderosa pine	Sawtimber	19 825	686,055.96	
White fir and other coniferous species	Sawtimber	2939	80,175.12	
All species	Green biomass convertible	660	67.57	
Unit 24 (78.1 ha):				2,803.68
Ponderosa pine	Sawtimber	8914	308,441.04	
White fir and other coniferous species	Sawtimber	2390	65,190.56	
All species	Green biomass convertible	725	74.24	

Note: Sawtimber is defined as a commercial species of tree with at least one 3.7-m sawlog or two noncontiguous, 2.4-m or longer sawlogs. Green biomass convertible is defined as wood fiber used as fuel. The purchaser made slash disposal deposits.

The Fine Stewardship Contract: Blocks 1 and 3

The Fine stewardship contract, another IRTC, was awarded to Interfor Pacific, Inc. It included treatments for all experimental units in blocks 1 and 3, and the contractor completed the work on 30 May 2013. Mandatory stewardship projects included under this contract were:

1. Mastication of undesirable conifers >0.31 m tall, shrubs, and slash <22.9 cm in diameter and >0.91 m.
2. Subsoiling of landings, skid trails, temporary roads, and miscellaneous areas with detrimental soil compaction to a depth of 60.96 cm where feasible (on 16 ha)
3. Machine fireline construction (11 km)
4. Hand fireline construction (0.8 km)

Treatment implementation removed a total of about 69 300 m^3 of wood biomass (54 400 m^3 of ponderosa pine) on 159 ha, of which about 30 700 m^3 was merchantable timber. Retained receipts totaled $1.5 million for this contract. Table 5.4 provides a breakdown of volume removed by species and slash disposal costs.

As with the Bind stewardship contract, the Fine stewardship contract was used in conjunction with a traditional timber sale contract to achieve stewardship-based goals by trading forest products in return for forest restoration projects. The contractor earned credits for work completed on stewardship projects, which they could use to pay for included timber value in excess of base rates and required deposits. Interfor Pacific, Inc. reported that the Bind and Fine stewardship contracts supported 450 jobs (direct and indirect combined).

Table 5.4—The estimated quantity removed and payment by tree species, unit, and product for the blocks 1 and 3 Fine stewardship contract

Species	Product	Estimated quantity removed	Estimated payment	Total slash disposal deposits
		Cubic meters	- - - - - - *U.S. dollars* - - - - - -	
Unit 11 (28.7 ha):				1,319.17
Lodgepole pine and other coniferous species	Sawtimber	337	9,474.78	
Ponderosa pine	Sawtimber	617	17,357.16	
White fir	Sawtimber	1608	49,455.76	
All species	Green biomass convertible	289	25.50	
Unit 12 (52.2 ha):				806.96
Lodgepole pine and other coniferous species	Sawtimber	623	17,516.40	
Ponderosa pine	Sawtimber	48	1,353.54	
White fir	Sawtimber	745	22,899.41	
All species	Green biomass convertible	329	29.00	
Unit 14 (45.7 ha):				2,212.59
Lodgepole pine and other coniferous species	Sawtimber	552	15,525.90	
Ponderosa pine	Sawtimber	3610	101,515.50	
White fir	Sawtimber	133	4,092.29	
All species	Green biomass convertible	487	43.00	
Unit 15 (61.5 ha):				10,405.33
Lodgepole pine and other coniferous species	Sawtimber	105	2,954.94	
Ponderosa pine	Sawtimber	17 709	497,943.48	
White fir	Sawtimber	3786	116,412.59	
All species	Green biomass convertible	892	78.75	

Table 5.4—The estimated quantity removed and payment by tree species, unit, and product for the blocks 1 and 3 Fine stewardship contract (continued)

Species	Product	Estimated quantity removed	Estimated payment	Total slash disposal deposits
		Cubic meters	*U.S. dollars*	
Unit 31 (51.0 ha):				6,306.34
Lodgepole pine and other coniferous species	Sawtimber	263	7,404.66	
Ponderosa pine	Sawtimber	12 100	340,216.26	
White fir	Sawtimber	0	0.00	
All species	Green biomass convertible	1269	112.00	
Unit 32 (31.2 ha):				962.85
Lodgepole pine and other coniferous species	Sawtimber	113	3,184.80	
Ponderosa pine	Sawtimber	1501	42,198.60	
White fir	Sawtimber	159	4,875.92	
All species	Green biomass convertible	309	27.25	
Unit 33 (69.6 ha):				5,644.79
Lodgepole pine and other coniferous species	Sawtimber	555	15,605.52	
Ponderosa pine	Sawtimber	10 166	285,835.80	
White fir	Sawtimber	51	1,567.26	
All species	Green biomass convertible	1430	126.25	
Unit 34 (39.3 ha):				4,405.53
Lodgepole pine and other coniferous species	Sawtimber	416	11,704.14	
Ponderosa pine	Sawtimber	8651	243,239.10	
White fir	Sawtimber	0	0.00	
All species	Green biomass convertible	456	40.25	

Note: Sawtimber is defined as a commercial species of tree with at least one 3.7-m sawlog or two noncontiguous, 2.4-m or longer sawlogs. Green biomass convertible is defined as wood fiber used as fuel. The purchaser made slash disposal deposits.

Chapter Summary

The LMS employed a combination of conventional timber contracts and stewardship contracts over 5 years to complete harvest and fuel reduction objectives. Two stewardship contracts accomplished stewardship projects in addition to marking and harvest objectives as part of the LMS. Stewardship projects included the mastication of biomass that did not meet utilization standards, subsoiling of temporary roads and landings, machine and hand fireline construction to protect ongoing research, and delayed felling and removal of trees for research purposes.

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Chapter 6: Harvest and Prescribed Fire Operations

Lorelle M. Sherman and Paul D. Anderson[1]

Introduction

Implementation of the Lookout Mountain Thinning and Fuels Reduction Study (hereafter LMS) treatments occurred from 2011 through 2015 with a consistent sequence among blocks. Harvest was completed in 1 year (block 4 in 2011, block 2 in 2012, blocks 1 and 3 in 2013). Mastication occurred the year after harvest. Slash piles were burned the second winter following harvest, and broadcast underburns were conducted the following spring (block 4 in 2013, block 2 in 2014, blocks 1 and 3 in 2015). The one anomaly occurred in the northernmost portion of unit 12, which had a larger proportion of true fir. That portion was burned in the fall of 2015 because of concerns that a deep layer of duff ignited in spring might not fully extinguish before summer and the onset of seasonal conditions that are favorable for fire.

Forest managers interested in implementing similar silvicultural treatments can use the detailed documentation of the operations and implementation of the LMS as a guide. Chapter 5 details sale preparation and awarding of contracts, while this chapter discusses the methods used to achieve harvest and fuel reduction objectives.

Felling and Skidding

Mechanized felling equipment was operated to minimize detrimental soil disturbance and damage to residual trees and tree seedlings. Mechanized felling equipment was prohibited on slopes exceeding 30 percent. Hand felling or bucking was used in circumstances agreed upon by the Forest Service and the purchaser.

Utilization standards for all contracts outlined the minimum specifications for merchantable trees by species (table 6.1). All trees that met the utilization standards and were <10.1 m (blocks 1, 2, and 3) or <12.2 m (block 4) in length were required to be whole tree yarded to the landing.

After processing, logs were decked, sorted, and hauled on log trucks to the Interfor Pacific mill in Gilchrist, Oregon.

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Table 6.1—Utilization standards by species and timber sale contract for the Lookout Mountain Thinning and Fuels Reduction Study

Species	Diameter at breast height (d.b.h.)	Minimum number of pieces per tree	Length required to remove	Diameter inside bark	Net scale of gross scale
	Centimeters		*Meters*	*Centimeters*	*Percent*
Lodgepole pine and other coniferous species	17.8	1	2.4	15.2	40
Ponderosa pine	22.9	1	4.9	15.2	40
White fir	20.3	1	2.4	15.2	40
All species	12.7	1	4.9 (3.7 in blocks 1 and 3 only)	7.6 (5.1 in block 2 only)	40

Note: Utilization standards are the same for all timber sale contracts except where noted otherwise in parentheses.

Mastication

Mechanical mastication preceded underburning to reduce shrub heights and ladder fuels. Mastication happened within 1 year of harvesting operations. Only unmarked stems <12.7 cm in diameter were masticated. All unmarked live conifers >0.3 m in height were severed from the stump, except in units 12 and 32. The proximity of those units to northern spotted owl (*Strix occidentalis caurina*) (NSO) habitat led to the amendment limiting mastication to conifers ≤12.7 cm diameter at breast height (d.b.h.); conifers >12.7 cm d.b.h. were retained. In all experimental units, all standing dead trees >17.8 cm d.b.h. were left except when they posed a safety hazard. Standing dead trees that posed a safety hazard were felled and left on site.

Slash created during thinning and mastication that was <22.9 cm in diameter on the large end and >0.9 m long was removed, mulched, chopped, lopped, or piled to meet specified fuel loadings. Slash >22.9 cm in diameter on the small end was left in place unless it was within a 3.1-m radius of a designated leave tree, in which case it was moved outside the 3.1-m radius.

Prescribed Fire

Broadcast burn objectives were uniform across all blocks as follows:

- Less than 2 percent mortality of ponderosa pine measured 1 year postburn
- Less than 5 percent loss of snags owing to fire measured 5 years postburn
- Bole scorch <6.1 m
- Crown scorch <50 percent
- Ninety to ninety-five percent ignition rate of surface fuels
- Consistent ignition conditions across the landscape
- One hundred percent mortality of unmarked white fir (*Abies concolor*) and lodgepole pine (*Pinus contorta*) in ponderosa pine (*P. ponderosa*) stands
- Reduction of brush by at least 25 percent and up to 100 percent measured 1 year postburn

The desired reduction in fuels was a 50 to 100 percent consumption of 1- and 10-hour fuels measured 1 month postburn. According to the Forest Service Wildland Fire Assessment System, 1-hour fuels are fine, flashy fuels that are <0.64 cm in diameter and take about 1 hour to lose 63 percent of the difference between their current moisture content and their equilibrium moisture content under the current environmental conditions. Ten-hour fuels are 0.64 cm to 2.54 cm in diameter and take about 10 hours to lose 63 percent of the difference between their current moisture content and their equilibrium moisture content under the current environmental conditions.

To implement an ancillary fire effects study, log decks were intentionally constructed at random locations within block 4 and burned to generate high-intensity soil heating presumed characteristic of high-severity wildfire. These soil heating hotspots were subsequently monitored specifically to evaluate the effects of soil heating on soil microorganisms and ectomycorrhizal fungi, which is outlined in chapter 10.

The Extra Contract: Block 4

Mastication reduced 1-hour fuels within the burn area, so the primary carriers of the fire were 10-hour and 100-hour surface fuels and slash that were rearranged during mastication. Ignitions occurred from 14 through 16 May and on 4 June 2013 with little to no wind, and increased needle reddening of large-diameter trees resulted from a prolonged residence time of heat and smoke in the canopy.

No spotting occurred into plots of ongoing long-term studies within the Lookout Mountain unit. Spotting led to minor surface fire in two unburned control plots as part of the ancillary study on combustion of coarse downed wood (chapter 10) located within block 4.

The fire effects monitor (FEMO) estimated that about 90 percent of surface fuels were consumed. Furthermore, crown scorch averaged <40 percent, and bole scorch averaged <6.1 m. Areas burned during a low wind event on 14 May 2013 experienced stagnation of heat and smoke leading to crown scorch levels which exceeded objectives.

The Bind Stewardship Contract: Block 2

Presence of NSO nesting, roosting, and foraging habitat features in the eastern half of unit 23 and the northeast corner of unit 24 triggered occupancy surveys prior to underburning. Fine, flashy fuels were notably sparse, but concentrations (e.g., jackpots) of light slash resulting from mastication served as the primary carrier of fire. Ignitions occurred from 2 through 4 June 2014.

According to visual evidence, the FEMO suggested that the burn successfully achieved the objective of 90 to 95 percent consumption of 1- and 10-hour fuels. Bole scorch and crown scorch objectives were met in 95 percent of the burn area. About 90 to 95 percent of surface fuels were burned. The objective of 100 percent mortality of white fir and lodgepole pine was met. Fire intensity was mostly low to moderate with small areas of higher intensity.

The Fine Stewardship Contract: Blocks 1 and 3

Because of sparse fuels, a strip-lighting pattern was used to generate the heat needed to carry the fire across the burn area. Strips were separated by 15 to 31 m. The primary carrier of the fire was jackpotted light slash resulting from mastication. Ignitions occurred on 28 and 29 April 2015.

Based on visual evidence, the FEMO stated that the burn appeared to have achieved its goal. The fuels consumption objective was met with <50 percent consumption of 1- and 10-hour fuels. Bole scorch height exceeded the 6.1-m threshold in some places, with maximum bole scorch heights up to 28 m observed. Scorch height >6.1 m is likely explained by high winds leading to short duration and local fire whirls and torching.

Regeneration

Across the landscape, posttreatment conifer regeneration was intended to arise naturally. However, small, discrete areas of experimental conifer plantings were established as an ancillary study evaluating seedling response to soil heating from fire.

The abundance and diversity of shrub layer regeneration was variable depending on environmental gradients across the study area. Postharvest and postburn resprouting and self-seeding led to natural shrub regeneration in all treatment types. No planting or seeding of shrubs, subshrubs, forbs, or graminoids occurred as part of this study.

Future Maintenance

Overstory

Periodic future treatments will be essential to realign the desired structure along target stand density index (SDI) values as established in this plan for 100 upper management zone (UMZ), 75 UMZ, and 50 UMZ treatments (see chapter 3 for treatment descriptions). These treatments will likely need to occur about 40 years after initial treatment implementation. Additional treatments will also be needed to sustain the trajectory of stand development into a multicohort structure as initiated by the 75 UMZ + Gaps treatment. These repeated treatments will likely need to occur at about 50-year intervals.

Timing of future thinning treatments will depend on the rate of canopy closure and understory development and connectivity between the understory and overstory. Rapid canopy closure or accumulation of fire-intolerant tree species or the establishment of ladder fuels may warrant future treatments occurring in less than 40 to 50 years.

There is debate over the potential impacts of fuel reduction activities on NSO habitat in Western United States forests (Ager et al. 2007, Lee and Irwin 2005). Future overstory treatments may conflict with the maintenance of preferred NSO habitat structure. The NSO favors multilayered, mixed-conifer forests with large-diameter trees (Courtney et al. 2004), which on the east side are inherently less fire resilient than open-canopy forests dominated by ponderosa pine and other fire-tolerant tree species. Foraging habitat encompasses a broad range of conditions, from high-quality, closed-canopy nesting habitat to more open conifer forest. Therefore, foraging habitat structure and prey abundance may be maintained under a broad range of future overstory and understory treatment prescriptions.

Understory

Additional mastication or underburning to reduce surface and ladder fuels are scheduled at 20-year intervals. However, because of the rapid reestablishment of surface and ladder fuels in some experimental units, a shorter return interval may need to be considered.

The length of time among future understory maintenance treatments will depend on the rate of understory reestablishment and fuels loading as well as logistical constraints. From a fuels management perspective, underburning would occur at a frequency that would eliminate the need for mechanical pretreatment. However, evidence of abundant understory shrub establishment within a 5- to 10-year period may trigger a need for mastication prior to a prescribed reburn. The seasonality of future underburning will be decided based on the combination of environmental factors and fuel loads that moderates potential burn intensity. Broadcast burns can also be planned and scheduled according to desired retention rates for coarse downed wood and creating or improving wildlife habitat.

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Part III: Components of the Lookout Mountain Thinning and Fuels Reduction Study

Chapter 7: Core Vegetation Study—Methods

Lorelle M. Sherman and Paul D. Anderson[1]

Introduction

Since Euro-American settlement, land use changes and forest management practices have significantly altered dry conifer forests in Oregon, as discussed in chapters 1 and 2. Because of its early designation as an experimental forest in the 1930s, the Pringle Falls Experimental Forest (PFEF) was largely spared from the harvest activity typical of the region in the 20th century. Limited past management activity may allow forest development to track shifting environmental conditions (however lagged), and therefore, the PFEF provides an ideal location for large-scale research on forest health and resilience. However, prior and current studies located at the PFEF are at risk of damage owing to natural disturbances. For example, the treatment prescriptions in the Lookout Mountain Thinning and Fuels Reduction Study (hereafter LMS) were designed to decrease risk of stand-replacing fire by reducing stand density and fuels loading, and in application, to mitigate risk to the Levels-of-Growing-Stock Study and other studies within.

Field Protocol

Physiographic information was collected on all 400-m^2 circular plots (475 total) during the first round of posttreatment data collection from 2014 through 2016 (see fig. 3.2). Elevation was estimated to the nearest 15 m using a U.S. Geological Survey (USGS) quadrangle map. Slope, aspect, and horizontal topography of the prevailing slope (either a ridge top, convex slope, even slope, concave slope, a swale, the bottom of a slope, or a flat) were estimated in the field using a compass corrected for declination. The percentage of exposed mineral soil and percentage of area with surface disturbance (defined as the area with surface depressions or ruts at least 2.5 cm deep) were both ocularly estimated to the nearest 5 percent.

Overstory and understory trees >1.37 m in height were also sampled within the 400-m^2 circular plot from 2011 through 2012, pretreatment, and in the first year following treatment implementation (from 2014 through 2016, congruent with the first year following underburning treatments for each block). For each tree, the species and vitality (live or dead) were identified. The diameter at breast height (d.b.h.) was measured at 1.37 m in height with a metric logger's tape. The total stem height (to the base of terminal bud on the main stem), stem height to the lowest live limb (lowest attachment point on the bole supporting a live limb), and stem height to the base of the live crown (lowest point on the bole with branches at least ⅔ circumference) were measured using a laser hypsometer. Tree and crown damages or defects were noted.

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Before and after treatment implementation, ocular estimates of canopy cover by shrubs, forbs, ferns, and graminoids were made for each 400-m^2 circular plot. Canopy cover estimations were averages of estimates made for each of four quadrants within a plot. Plots were divided into quadrants based on cardinal directions and marked. First, an estimate of total canopy cover was made and then individual species canopy covers were estimated as trace (<1 percent), to the nearest 1 percent when cover was below 10 percent cover, and to the nearest 10 percent when cover was above 10 percent. In the case of graminoids, less precise identification led to the binning of sedge cover estimates at the genus level (*Carex* spp.) and grass cover estimates at the family level (Poaceae). Conceptually, the maximum total canopy cover cannot exceed 100 percent. However, because individual species can overlap, cover estimates when summed could exceed the estimated total cover and in theory could exceed 100 percent.

Coarse downed wood (CDW) and tree seedling and sapling attributes were assessed using strip sampling. At each 400-m^2 circular plot center, three 22-m transect lines were established at 0-, 120-, and 240-degrees azimuth. An 80-m^2 rectangular strip sampling plot was established extending 20 m radially and 2 m either side of each transect line (20- by 4-m strip) (fig. 7.1). To calculate CDW volume within each of the three 4- by 20-m strips, length and diameter data were collected for each piece of downed wood at least 1 m in length and with a large-end diameter of ≥15.2 cm. If possible, the decomposition class and species were recorded for each piece. Tree seedlings and saplings <1.37 m in height were tallied by species in a rectangular 20- by 2-m subplot established along the clockwise side of each 40-m^2 strip sampling plot to determine local tree seedling density.

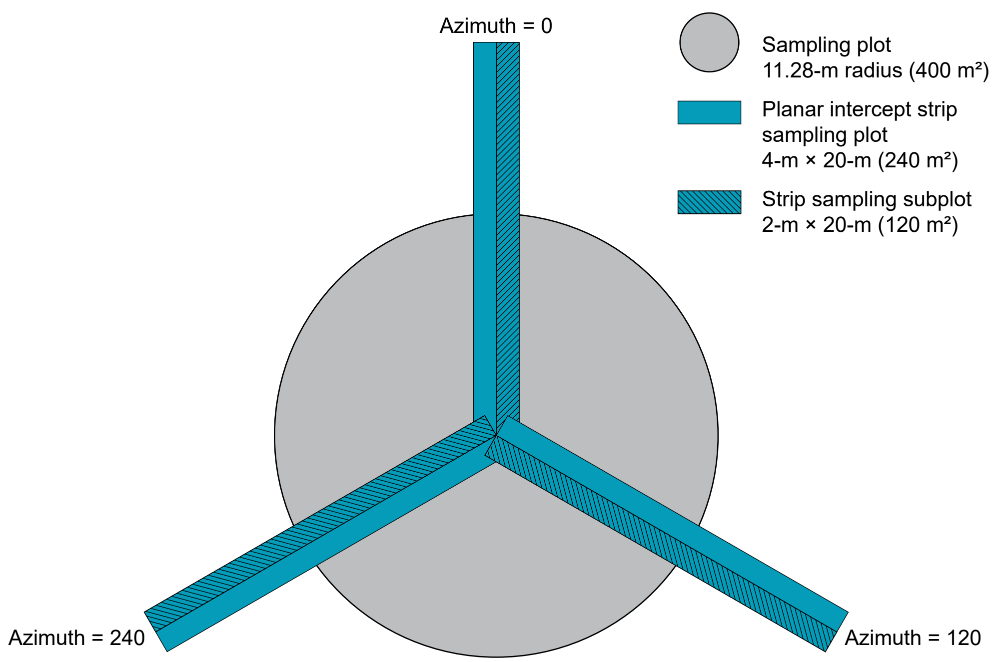

Figure 7.1—Diagram depicting a permanent 400-m^2 sampling plot (circle), with planar intercept strip sampling plots delineated as rectangles and strip sampling subplots as half of each rectangle delineated by hatch marks.

Overstory Structure

Data to characterize pretreatment and posttreatment overstory stand structure were collected at the scale of the 400-m^2 permanent circular sampling plots. All trees ≥1.37 m in height were measured and recorded as overstory trees and trees <1.37 m in height as seedlings and saplings.

For each plot, tree surveys began at 0 degrees north azimuth and rotated clockwise around the plot center. Each tree location was recorded as azimuth and distance (meter) to the center of the stem. For each tree, the species was recorded, and the vitality was coded as either live (1) or dead (9).

Tree d.b.h. at 1.37 m was measured to the nearest 0.1 cm on the uphill side of the tree. Tree height was measured to the nearest 0.1 m with a telescoping height pole or laser hypsometer. The total stem height (to the base of terminal bud on the main stem), stem height to the lowest live limb (lowest attachment point on the bole supporting a live limb), and stem height to the base of the live crown (lowest point on the bole with branches at least ⅔ circumference) were measured using a laser hypsometer. Live crown ratio was determined by ocular estimate to the nearest 10 percent from the base of the live crown to the uppermost limbs.

Notes on tree condition included any crown damage or visible stress (e.g., reduced needle elongation, needle chlorosis, early needle senescence); stem damage and displacement; and presence of insect damage, brooms, mechanical damage, and bole scars. If bole char or crown scorch was observed, they were measured to the nearest 0.01 m in height.

Overstory Composition

Similar to overstory structure, data were collected at the scale of the 400-m^2 permanent circular sampling plots on trees >1.37 m in height. Presence of advanced regeneration (established tree saplings) is important in determining pathways of plant succession because it replaces the stressed, dead, and dying overstory trees following disturbance. Therefore, advanced generation was included with overstory composition data in this report. All trees were identified to species.

At the time of operational implementation, mastication treatments removed unmarked trees >0.3 m in height and <12.7 cm in diameter. This was another source of overstory tree removal in addition to the removal of marked trees as part of thinning prescriptions. However, there is uncertainty in the number of trees removed during mastication.

Understory Structure and Composition

Data to describe pretreatment and posttreatment understory characteristics were collected in 400-m^2 permanent circular plots and 2- by 20-m strip sampling subplots. Plant growth forms (i.e., shrubs, subshrubs, forbs/herbs, and graminoids) were recorded as defined by the U.S. Department of Agriculture Natural Resources Conservation Service (NRCS) growth habit codes (USDA NRCS 2020). Ocular vegetation cover estimates for shrubs, subshrubs, forbs, and graminoids by species or genus were made in each quadrant (100 m^2) of the circular plot delineated by cardinal directions and subsequently averaged cover estimates across all quadrants of the circular plot.

Ocular estimates of shrub canopy cover were made to the nearest 1 percent when cover was below 10 percent and to the nearest 10 percent when cover was above 10 percent. Total shrub cover ignores overlap among individuals of a species, so conceptually the maximum total cover for an individual species was 100 percent. However, total shrub canopy cover estimates could exceed 100 percent when summed across all species in a plot because species estimates were independent of potential vertical overlap among individuals of different species.

Ocular estimates of subshrub, forb/herb, and graminoid canopy cover were made by species and lifeform to the nearest 1 percent with trace amounts were recorded as 0.05 percent.

All trees having a height ≥15 cm and <1.37 m were recorded as understory tree seedlings. Tree seedlings were recorded by species, including both deciduous and conifer species. Estimation of tree seedling canopy cover followed the same protocol used for estimation of shrub canopy cover.

Coarse Downed Wood

Data to describe CDW were collected in three 80-m^2 strip sampling plots that were scaled to the 400-m^2 plots. Within each strip sampling plot, only logs or parts of logs that were ≥1 m in length and had a large-end diameter of ≥15.2 cm were measured and counted. The small-end (≥7.6 cm) and large-end (≥15.2 cm) diameters were measured on all qualifying logs or parts of logs that fell within or crossed the boundaries of the strip plot. If a piece extended outside the strip sampling plot, diameters were measured at the line of intercept of the strip sampling plot boundary and CDW piece.

In addition, the species (if possible) and decomposition class of each log were recorded according to Thomas et al. (1979):

- Class 1: bark is intact, twigs are present, wood texture is intact, log is round, original wood color, log may be elevated on support points.
- Class 2: bark is intact, twigs are absent, wood texture is sound or becoming soft, log is still round, original wood color, log elevated on support points but sagging slightly.
- Class 3: bark has fallen off, twigs are absent, wood texture is hard, log is still round, original color of wood is faded, log is sagging near ground.
- Class 4: bark is absent, twigs are absent, texture of wood is soft and blocky pieces, shape of log is round to oval, wood has faded to light yellow or gray, entire log on the ground.
- Class 5: bark is absent, twigs are absent, wood texture is soft and powdery, shape of log is oval, wood has faded to light yellow or gray, entire log on the ground.

Analysis

To analyze overstory structure, the target stand density index (SDI) reflecting a proportion of upper management zone (UMZ) depending on treatment type was calculated for each plant association according to the methods outlined in Cochran et al. (1994) and Booser and White.[2] To assess treatment implementation, observed posttreatment SDI values were compared with target SDI values calculated from the percentage of UMZ prescribed for each experimental unit (e.g. 50 percent UMZ, 75 percent UMZ, 100 percent UMZ). The percentage change in basal area (BA) and density of snags and live trees was calculated for each experimental unit. Standard deviation was calculated as the variance among sampling plots.

An assessment of how treatments resulted in the target overstory composition involved a comparison of the density (trees/hectare) of overstory trees by species and calculation of the change in Shannon diversity index of overstory trees. Analyses were conducted using the vegan package (Oksanen et al. 2020) with R Statistical Software[3] (version 3.6.3) via RStudio (version 1.2.5033) (RStudio Team 2020).

In line with the 21-inch (53.3-cm) rule, or Eastside Screens, the change in density of large live trees and large snags (≥50.27 cm d.b.h.) for each experimental unit is summarized in tables 8.7 and 8.8, respectively. Change in density was assessed in terms of stems per unit area (trees/hectare) and BA per unit area (square meters/hectare) for all species combined and for species classes reflecting the targeted retention of fire-tolerant pines and the removal prioritization of more fire-sensitive tree species.

The initial changes in understory vegetation were described by the mean percentage of canopy cover of plant growth forms in figure 8.2 and species within plant growth forms in figures 8.3 through 8.7. The change in Simpson's diversity index of understory plant growth forms was calculated using the vegan package (Oksanen et al. 2020) in RStudio (version 1.2.5033) (RStudio Team 2020).

The accumulation of CDW is most often measured as volume and calculated with algebraic formulas that account for the natural tapering of tree diameter growth with tree height. The Smalian formula is one of the formulas most commonly used in the Western United States for estimating volume. When measuring ponderosa pine (*Pinus ponderosa*) and lodgepole pine (*P. contorta*) logs, Plank and Cahill (1984) found that the Smalian formula tended to overestimate volume. CDW volume is reported as calculated by the Smalian formula:

$$Volume = \frac{1}{2}\,(0.00007854 \times L \times (Dsm^2 + Dlrg^2)$$

[2] Booser, J.; White, J. 1984. Calculating maximum stand density indexes (SDI) for Deschutes National Forest plant associations. Unpublished paper. On file with: U.S. Department of Agriculture, Forest Service, Deschutes National Forest, 63095 Deschutes Market Road, Bend, OR 97701.

[3] The use of trade or firm names in this publication is for reader information and does not imply endorsement by the U.S. Department of Agriculture of any product or service.

where *L* represents the log length in meters, *Dsm* represents the diameter of the small log end, and *Dlrg* represents the diameter of the large log end, measured in meters.

CDW percentage cover was calculated as the total area (square meters) divided by the area of the strip sampling plot (240 m^2). Total area was calculated using the formula for the area of a trapezoid:

$$Total\ area = \left(\frac{(Dlrg \div Dsm)}{2}\right) \times L$$

where *L* represents the piece length in meters, *Dsm* represents the diameter of the small end of the piece in meters, and *Dlrg* represents the diameter of the large end of the piece in meters.

Assumptions

The assumption of similar variation among experimental units regardless of unit size was based on legacies of past management and natural disturbances within the Lookout Mountain unit. The Deschutes National Forest Management Plan (USDA FS 1990) addresses the entire PFEF as a single management area. Most of the Lookout Mountain unit consists of stands of a single-age cohort (about 174 years). A small portion along the southern edge is younger. Earlier silvicultural experiments have added limited, fine-scale, patchy variability (see table 2.2 for a summary of past research activities on the PFEF). Except for those isolated experiments (which were excluded from the study sampling units), all of Lookout Mountain has continuous canopy cover, high levels of accumulated ground and surface fuels, and high tree densities. The 1910 wildfire created some variation in the structure and composition of forests in the Lookout Mountain unit.

Block- or unit-scale stand density metrics that take plant association into account may introduce unknown levels of error because some blocks and units may include more than one plant association (fig. 7.2). The delineation of blocks based on similarities in elevation and aspect was intended to reduce the variation among treatment units arising from different physiographic factors and therefore reinforce the ability to detect treatment effects.

Ecological and Logistical Adaptations

The sampling area of experimental unit 12, originally established and surveyed before treatment implementation, was the only sampling area that was burned in the autumn. All other sampling areas were burned in the spring. Prescribed fire specialists assessed the sampling area in unit 12 and found a deep surface duff layer, which they deemed too risky for a spring season burn. For continuity in long-term experimental monitoring, the sampling area was relocated to a portion of unit 12 that was spring burned before the initial posttreatment remeasurement. Therefore, the pretreatment and posttreatment measurements for unit 12 were

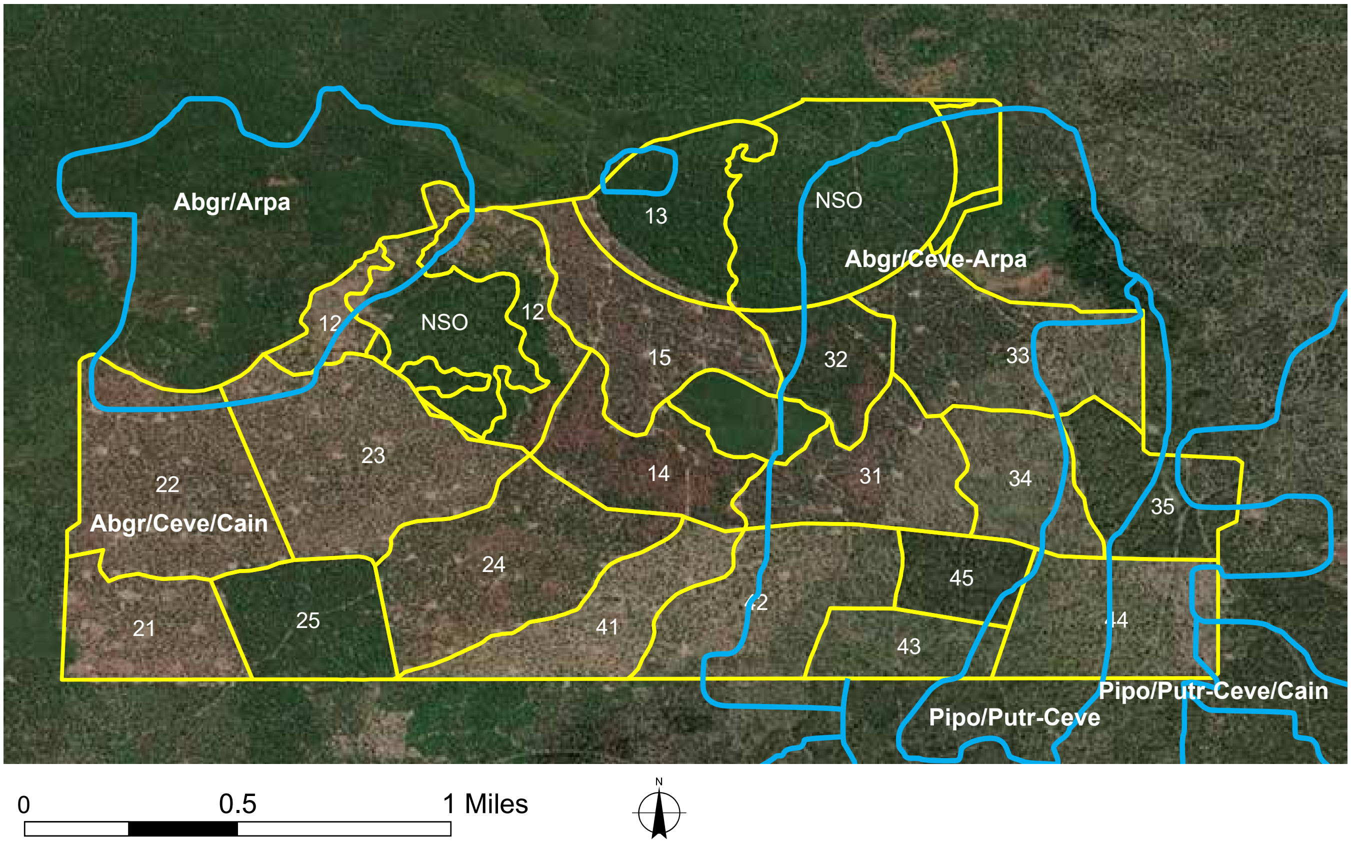

Figure 7.2—Experimental units shown as yellow polygons and labeled with numbers overlaying plant association type boundaries shown as blue lines (Volland 1988) within the Lookout Mountain Thinning and Fuels Reduction Study. Abgr = *Abies grandis*; Arpa = *Arctostaphylos patula*; Cain = *Carex inops*; Ceve = *Ceanothus velutinus*; Pipo = *Pinus ponderosa*; Putr = *Purshia tridentata*.

collected in different areas and plots; that discrepancy is accounted for in the data summaries to follow.

The 2011 detection of a nesting pair of northern spotted owl (*Strix occidentalis caurina*) (NSO) and the resultant designation of a nest site buffer reduced the area available for treatment in experimental units 13, 32, and 33 (see fig. 4.2). This led to the reorganization of experimental units within blocks 1 and 3; several experimental units were decreased in size, and only four of five total treatments were represented in block 1 (see fig. 4.2). The station statistician conducted a power analysis of the incomplete block design that determined the changes in design were tolerable. The resulting imbalance of treatment representation across the four experimental blocks required appropriate statistical treatment in data analyses.

The accuracy and precision of preharvest marking in meeting overstory structure and composition targets was unknown, and on-the-ground changes to marking during active harvest operations were not documented.

Quality Assurance/Quality Control

Deschutes National Forest and Pacific Northwest Research Station staff conducted this research as a collaborative effort. It required a high degree of cooperation and coordination with the Bend-Fort Rock Ranger District. Data collectors, trained and supervised by station scientists, were responsible for operational maintenance and calibration of any instrumentation. Accuracy of each instrument was maintained using manufacture's recommended procedures for daily calibration and annual maintenance. About 5 percent of the plots were remeasured (tree diameters, tree height, canopy cover of herbaceous species, etc.; blind resampling) to determine precision levels. For computer entry and data reduction, the identified personnel were responsible for verification of data entry, data completeness, accuracy, precision, and documenting corrective action.

Data are stored in logbooks, computers, and backup disks. Numeric data were entered and stored in spreadsheet form compatible with Microsoft (MS) Excel®. File headers identify fields, experimental units, experimental conditions, and other relevant information. All files were converted to a MS Access® database. Hardcopy backups and the complete study file, including documentation of methods and equipment used, training and certification of personnel, notes from field personnel, study plan amendments, copies of manuscripts, technology transfer materials, and safety documentation are located at the Corvallis Forestry Sciences Laboratory.

Chapter Summary

The core vegetation study was designed to measure the long-term ecological responses of implemented silvicultural treatments using a suite of measurements on overstory tree, understory vegetation, and CDW characteristics. The core vegetation study comprised 25 permanent sampling plots within a subsample area of each experimental unit. There are two data measurement periods included in this establishment report that follow the field protocols outlined above: pretreatment implementation from 2011 through 2012 and posttreatment implementation from 2014 through 2016. Analysis of the structural response of the overstory was defined by targets set for BA and SDI based off the UMZ.

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Chapter 8: Core Vegetation Study—Early Observations

Lorelle M. Sherman and Paul D. Anderson[1]

Introduction

The long-term response of the Lookout Mountain unit forest to thinning and fuel reduction treatments will emerge over time. However, near-term metrics, such as the initial changes in structure and composition and an assessment of study implementation, are summarized below. This chapter covers an evaluation of study implementation, variation among blocks, variation among treatments, and unforeseen variation. Early observations, variability, and implications of early observations are outlined for each of the focal response categories (overstory structure, overstory composition, understory structure and composition, coarse downed wood [CDW]). The following questions defined the efficacy of and variability in treatment implementation:

- Did thinning and fuel reduction treatments result in the desired forest overstory structure and overstory composition?
- Were there any notable deviations from expected changes in overstory structure or overstory composition?
- How variable were the effects of thinning and fuel reduction treatments on overstory structure, overstory composition, understory vegetation, and CDW within and among blocks and treatments? What was the source of that variability?
- What structural and compositional changes occurred in the understory vegetation and CDW?

Overstory Structure

The desired forest overstory structural response was defined by targets set for basal area (BA) and stand density index (SDI) based on the upper management zone (UMZ) (Cochran et al. 1994) specific to each plant association in the Lookout Mountain Thinning and Fuels Reduction Study (hereafter LMS) area (table 8.1). The desired overstory structure was created by preferentially removing fire-intolerant tree species and retaining fire-tolerant tree species according to a species marking preference system (see table 5.1) written in concurrence with overstory species composition goals.

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Table 8.1—Mean pretreatment overstory tree conditions (all trees >1.37 m in height) by block with the predominant plant associations according to Volland (1988) and his target density metrics

Block	Plant association	Treatment area	Stand density index	Basal area	Density	Quadratic mean diameter
		Hectare		*Square meters per hectare*	*Trees per hectare*	*Centimeters*
1	CW-S1-15 mixed conifer/snowbrush/sedge	209	313	42	1,082	32
2	CW-S1-15 mixed conifer/snowbrush/sedge	340	316	47	451	41
3	CW-S1-13 mixed conifer/greenleaf manzanita-snowbrush/sedge-penstemon	250	331	44	859	31
4	CP-S3-12 ponderosa pine/bitterbrush-snowbrush/sedge	224	310	41	777	29

Early Observations

Live and dead trees per hectare—

The mean percentage decrease of live tree density was similar in all thinning treatment types, with the smallest reduction measured in the least-intensive thinning treatment and the largest reduction measured in the most intensive thinning treatment (table 8.2). The mean change in live tree density was −89, −91, −92, and −93 percent for 100 UMZ, 75 UMZ + Gaps, 75 UMZ, and 50 UMZ, respectively (see chapter 3 for treatment descriptions). The 22 percent mean reduction in live tree density in Reference units was greater than expected. Some decrease in live tree density was expected in Reference units owing to greater density-driven competition for resources and thus greater mortality rates than those in treated units. However, additional explanations for the decrease are bark beetle-induced mortality of trees where prescribed underburning unintentionally crept into Reference unit boundaries (as documented within the sampling area of Reference unit 35) and a blowdown event that took place during treatment implementations. A survey of dead trees in 2016 noted at least one wind-killed tree in Reference unit 35. No direct fire mortality was recorded, but bole scorch was observed outside of plot boundaries in Reference unit 35, which possibly contributed to a delayed within-plot tree mortality owing to spillover of bark beetle attacks on stressed trees. Low levels of bark beetle-induced mortality were observed in Reference units 25 and 45, suggesting potential spillover of bark beetle attacks in other Reference areas.

Notable percentage decreases in snag density posttreatment were measured in all units and treatments (table 8.2). The mean change in snag density was −81, −89, −95, and −96 percent for 100 UMZ, 75 UMZ, 50 UMZ, and 75 UMZ + Gaps, respectively. The large loss of snags in the most intense thinning treatments (50 UMZ and 75 UMZ + Gaps) may be attributed to increased exposure to wind. Exposure to wind may have been amplified in 75 UMZ + Gaps treatment units where at least two sampling plots

Table 8.2—Pretreatment and posttreatment live and dead standing tree (snag) densities per hectare and pretreatment to posttreatment density changes by unit and treatment

Unit	Pretreatment snag density mean ± SD	Posttreatment snag density mean ± SD	Snag density change	Pretreatment live tree density mean ± SD	Posttreatment live tree density mean ± SD	Live tree density change
	- - - - - Stems per hectare - - - - -		*Percent*	*- - - - - Stems per hectare - - - - -*		*Percent*
Treatment 1—100 UMZ						
14	69 ± 75	23 ± 1	−67	670 ± 1172	63 ± 1	−91
24	76 ± 53	3 ± 1	−96	575 ± 544	74 ± 1	−88
32	138 ± 99	27 ± 1	−80	675 ± 355	113 ± 2	−83
42	128 ± 133	23 ± 4	−82	1,042 ± 980	53 ± 1	−95
Average	103 ± 95	19 ± 2	−81	741 ± 830	76 ± 1	−89
Treatment 2—75 UMZ						
12a	141 ± 83	NA	NA	2,451 ± 2,218	NA	NA
12b	NA	23 ± 1	NA	NA	56 ± 1	NA
22	77 ± 59	13 ± 0	−83	490 ± 179	53 ± 1	−89
33	202 ± 223	23 ± 2	−89	916 ± 650	66 ± 2	−93
44	137 ± 101	7 ± 0	−95	1,092 ± 559	46 ± 1	−96
Average	139 ± 133	17 ± 1	−89	1,237 ± 1,192	55 ± 1	−93
Treatment 3—50 UMZ						
15	90 ± 62	4 ± 1	−96	772 ± 987	31 ± 1	−96
23	65 ± 74	2 ± 0	−97	299 ± 137	35 ± 1	−88
31	191 ± 169	16 ± 1	−92	999 ± 655	36 ± 1	−96
41	112 ± 83	8 ± 1	−93	453 ± 214	36 ± 2	−92
Average	115 ± 106	8 ± 1	−95	631 ± 606	35 ± 1	−93
Treatment 4—75 UMZ + Gaps						
21	82 ± 73	0 ± 0	−100	495 ± 215	52 ± 1	−90
34	39 ± 50	3 ± 0	−92	499 ± 430	58 ± 1	−88
43	179 ± 172	7 ± 0	−96	575 ± 313	32 ± 1	−94
Average	100 ± 97	3 ± 0	−96	523 ± 287	47 ± 1	−91
Reference—no treatment						
13	53 ± 50	40 ± 2	−25	434 ± 211	374 ± 7	−14
25	58 ± 55	6 ± 1	−90	393 ± 409	262 ± 4	−33
35	143 ± 126	88 ± 4	−39	1,191 ± 1,071	780 ± 18	−35
45	116 ± 100	63 ± 2	−46	722 ± 516	664 ± 18	−8
Average	93 ± 89	49 ± 3	−50	685 ± 637	520 ± 13	−23

NA = not applicable; SD = standard deviation.

Note: Unit 12 sampling plots (12a) were relocated posttreatment (12b) owing to inconsistencies in surface fuels type and season of understory burning for a portion of unit 12 containing the original gridded sampling points.

overlapped with regeneration gaps in each of the three units that received this treatment. Reference units had a mean decrease in snag density of 50 percent, partly driven by the 90 percent decrease in Reference unit 25. Potential explanations for the percentage decrease in live trees on Reference units also apply to the percentage decrease in snag density. However, loss to fire was an expectation within units treated with underburning, whereas it was not expected to occur in Reference units.

Basal area—

The mean percentage decrease in BA was less than the mean percentage decrease in live tree density (table 8.3). The magnitude of mean BA decrease was greater with increasing harvest intensity as expected. The mean change in BA was −38, −56, −58, and −65 percent for 100 UMZ, 75 UMZ, 75 UMZ + Gaps, and 50 UMZ, respectively. Except for a decrease in unit 45, an increase in BA was measured in all Reference units for a mean increase of 4.7 percent. Posttreatment BA values were greater than respective target BA values in all experimental units except units 14, 33, and 34 (table 8.3). Units 14 and 33 suffered pockets of greater-than-intended underburn intensity. However, other units that also showed signs of greater-than-intended burn intensity resulted in BA values exceeding the target values. Unit 34 was prescribed the 75 UMZ + Gaps treatment. The less-than-intended posttreatment BA may be attributed to the predetermined locations of gaps coinciding with locations containing trees with larger-than-average diameters (e.g., high overstory tree BA).

Posttreatment basal areas for each of the 3 or 4 replicate units receiving the same treatment deviated from the target by less than 5 m^2/ha. The variability in the effect of treatments on Reference unit BA may be due to a combination of yearly BA growth and stochastic disturbance events. Interestingly, the most intensive treatment (50 UMZ) contained the experimental unit with the largest deviation between posttreatment BA and the target BA, with the former being greater than the latter.

Stand density index and upper management zone—

For all treatments, mean posttreatment SDI values fell within 1 standard deviation (SD) of the target SDI (n = 3 or 4), where the target SDI is a given percentage of the SDI for the UMZ. All Reference units fell within 1 SD of SDI at 100 UMZ, except unit 13, in which the posttreatment SDI was over 100 units greater than SDI at 100 UMZ (table 8.4).

All experimental units in blocks 1, 2, and 4 had lower posttreatment mean SDI values than the target SDI (table 8.4). The posttreatment SDI in all experimental units in block 3 was higher than the target SDI (table 8.4). This may suggest that for plant associations CW-S1-15 and CP-W3-12 (Volland 1988), the application of a combined thinning and underburning resulted in lower densities than prescribed. The same application of thinning and underburning resulted in a generally different direction of change in overstory SDI in the CW-S1-13 plant association found in block 3. However, with a sample size of n = 1 for the CW-S1-13 plant association, the variability in the effect of treatments on SDI may be explained by several alternatives (marking crew accuracy, environmental gradients, geologic features, etc.).

Table 8.3—Target, pretreatment, and posttreatment overstory tree basal area (BA), posttreatment BA relative to target BA, and pretreatment to posttreatment BA changes by unit and treatment

Unit	Target BA	Pretreatment BA mean ± SD	Posttreatment BA mean ± SD	Posttreatment BA/ target BA × 100	BA change
	- - - - - - - - - - - Square meters per hectare	*- - - - - - - - - - -*		*- - - - - Percent - - - - -*	
Treatment 1—100 UMZ					
14	23.2	32.0 ± 15.6	20.5 ± 10.2	89	−36
24	25.5	42.9 ± 14.8	26.5 ± 11.6	104	−38
32	32.1	45.1 ± 20.3	34.8 ± 15.6	108	−23
42	16.1	45.2 ± 18.0	19.8 ± 11.1	123	−56
Average	NA	41.3 ± 17.3	25.4 ± 12.3	106	−38
Treatment 2—75 UMZ					
12a	NA	29.5 ± 14.9	NA	NA	NA
12b	17.4	NA	21.5 ± 13.6	123	NA
22	19.1	52.4 ± 17.1	21.1 ± 10.7	110	−60
33	24.1	43.6 ± 27.3	23.1 ± 11.3	96	−47
44	12.2	43.5 ± 18.6	17.1 ± 10.1	140	−61
Average	NA	42.3 ± 20.0	20.7 ± 11.5	117	−56
Treatment 3—50 UMZ					
15	11.7	48.9 ± 17.7	12.2 ± 8.2	104	−75
23	12.9	47.2 ± 15.2	15.0 ± 9.2	117	−68
31	16.1	48.2 ± 19.9	20.9 ± 8.2	130	−57
41	8.0	38.0 ± 19.8	14.5 ± 8.5	181	−62
Average	NA	45.6 ± 18.1	15.7 ± 8.5	133	−65
Treatment 4—75 UMZ + Gaps					
21	19.1	51.0 ± 15.8	19.5 ± 11.0	102	−62
34	24.1	41.2 ± 14.6	21.3 ± 9.2	89	−48
43	12.2	38.2 ± 15.7	13.7 ± 8.0	112	−64
Average	NA	43.5 ± 13.3	18.2 ± 8.2	101	−58
Reference—no treatment					
13	NA	58.8 ± 16.2	61.7 ± 16.5	195	+5
25	NA	43.1 ± 13.4	48.6 ± 15.5	122	+11
35	NA	43.4 ± 13.9	45.6 ± 14.4	108	+5
45	NA	40.2 ± 15.5	39.4 ± 17.2	111	−2
Average	NA	46.4 ± 14.8	48.8 ± 15.9	134	+5

NA = not applicable; SD = standard deviation; UMZ = upper management zone.

Note: Unit 12 sampling plots (12a) were relocated posttreatment (12b) owing to inconsistencies in surface fuels type and season of understory burning for a portion of unit 12 containing the original gridded sampling points.

Table 8.4—Target, pretreatment, and posttreatment overstory stand density index (SDI) and pretreatment and posttreatment SDI relative to target SDI by unit and treatment

Unit	Target SDI	Pretreatment SDI mean ± SD	Posttreatment SDI mean ± SD	Pretreatment SDI mean /target SDI × 100	Posttreatment SDI mean/target SDI × 100
				- - - - - - - - Percent	*- - - - - - -*
Treatment 1—100 UMZ					
14	156	238 ± 105	104 ± 61	153	67
24	156	300 ± 97	132 ± 70	192	85
32	145	326 ± 114	192 ± 83	225	132
42	156	347 ± 105	102 ± 61	222	66
Average	NA	303 ± 105	133 ± 69	198	88
Treatment 2—75 UMZ					
12a	NA	287 ± 111	NA	NA	NA
12b	117	NA	106 ± 72	NA	90
22	117	354 ± 102	108 ± 60	303	92
33	109	327 ± 162	112 ± 68	300	103
44	117	352 ± 133	88 ± 54	301	75
Average	NA	330 ± 129	104 ± 64	301	90
Treatment 3—50 UMZ					
15	78	346 ± 110	57 ± 45	444	73
23	78	295 ± 81	70 ± 50	378	90
31	73	371 ± 135	74 ± 62	508	102
41	78	271 ± 130	60 ± 50	347	76
Average	NA	321 ± 116	65 ± 52	419	88
Treatment 4—75 UMZ + Gaps					
21	117	351 ± 105	98 ± 62	300	83
34	109	281 ± 83	112 ± 54	258	103
43	117	281 ± 104	60 ± 47	240	51
Average	NA	304 ± 85	90 ± 47	266	93
Reference—no treatment					
13	NA	381 ± 97	384 ± 98	135	136
25	NA	282 ± 74	298 ± 89	100	105
35	NA	351 ± 106	344 ± 97	126	123
45	NA	297 ± 91	287 ± 103	105	102
Average	NA	328 ± 93	328 ± 97	117	117

NA = not applicable; SD = standard deviation; UMZ = upper management zone.

Note: Unit 12 sampling plots (12a) were relocated posttreatment (12b) owing to inconsistencies in surface fuels type and season of understory burning for a portion of unit 12 containing the original gridded sampling points.

Variability

Experimental units that had high sampling variability (e.g., large SD, table 8.4) in posttreatment density metrics included 50 UMZ units 15, 23, 31, and 41; and 75 UMZ + Gaps unit 43. Greater variation among experimental units for 50 UMZ units was expected because the wide spacing among leave trees leading to some plots with few or no trees. Variability in experimental units assigned to 75 UMZ + Gaps was partially due to the creation of gaps overlapping some sample plots. This overlap led to few or no retained trees within some plot boundaries. The coefficient of variation (CV, SD divided by mean) of Reference units was consistently less than that of thinned units for BA and SDI. Units assigned to the most intensive thinning treatment (50 UMZ) had the largest CV for BA possibly because each additional tree marked for cutting had a greater impact on the total plot BA. The CV for SDI was relatively consistent among thinning treatments.

There were multiple sources of variability in treatment implementations in meeting target density goals. Pretreatment, all plots had at least one live tree. Overall, 40 plots out of 475 total plots had no live, retained trees posttreatment. The random location of the sampling area within each treatment unit allowed for individual sample plots within the sampling grid to be devoid of residual trees following implementation of the prescribed thinning treatments.

There were 18 instances of plots in Reference units with more live trees posttreatment than pretreatment. This did not occur for any of the plots associated with thinning treatment units. Because Reference units were overstocked with small-diameter trees, there was a high chance of missing or double counting individual trees. There could also have been inconsistent application of small-diameter tree sampling protocols among years by different crews, particularly in accounting for small-diameter trees having severe lean. Also, some of the small-diameter trees could have achieved 1.37 m in height and were subsequently added to the count as ingrowth.

Some areas may have had greater mortality of overstory trees than anticipated owing to prescribed underburning or subsequent insect attack on fire-injured trees. Anecdotal observations of a stalled fire front as part of the prescribed underburn in block 4 noted heat accumulation and canopy scorching. Additionally, an area of dead trees in unit 14 suggested the prescribed fire became more intense than prescribed. Other pockets of fire-induced tree mortality are outlined in figure 8.1.

The LMS was implemented over the course of several years (harvest operations in 2011, 2012, 2013; prescribed burning in 2013, 2014, 2015) and under three timber sale contracts. There was some level of inherent variability associated with different environmental conditions by year, timber sale preparation, marking crews, and harvest operations that was not explicitly observed or documented.

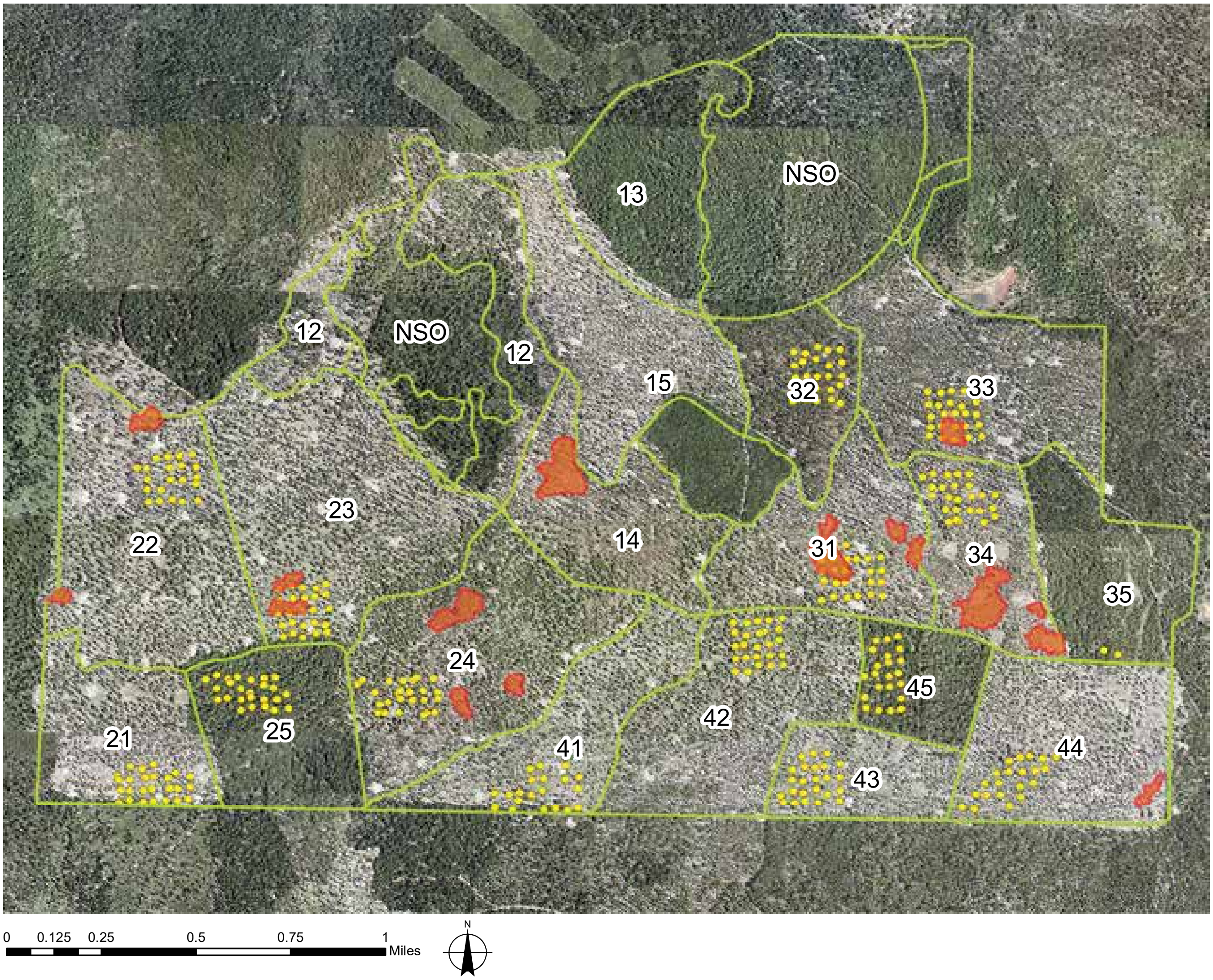

Figure 8.1—Areas where the prescribed fire as part of the Lookout Mountain Thinning and Fuels Reduction Study appeared to get hotter than intended are shown in orange, with experimental units outlined by lime green and permanent 400-m^2 sampling plots marked as yellow points.

Implications

The overstory structural characteristics described above provide a snapshot of the near-term change in overstory structure resulting from treatment application. Additional years of data collection are necessary to inform the response of overstory structure. Although unintended overstory tree mortality cannot be separated from the intended removal of overstory trees during thinning, the LMS demonstrates the potential for variability at the unit, treatment, and block scales. Information from the study will help increase the efficiency, accuracy, and precision of overstory tree removal in meeting target overstory structure goals.

Overstory Composition

Historical documentation and accounts of presettlement forests in central Oregon suggest that fire-tolerant tree species dominated the forest composition, and those with low fire tolerance were sparse (Brown et al. 2000, Languille et al. 1903). Persistent, mixed-conifer forest was present on the landscape where topographical features led to cool, moist microclimates (Merschel et al. 2014). In the absence of fire, historically open and fire-resilient ponderosa pine (*Pinus ponderosa*) forests in hot, dry microclimates have increased in density, while those in cooler, moister microclimates have transitioned into shade-tolerant, mixed-species forest (Merschel et al. 2014). An operational objective of the LMS was to reduce stand density in all stands to a more fire-adapted condition. To achieve the target overstory composition dominated by fire-tolerant tree species, fire-intolerant tree species were preferentially removed and fire-tolerant species were retained, according to a species marking preference system (see table 5.1) written in concurrence with goals for overstory stocking.

Early Observations

All treatments eliminated tree species considered to have low fire tolerance (e.g., Engelmann spruce [*Picea engelmannii*] and subalpine fir [*Abies lasiocarpa*]) (Burns 1990) and reduced the number of moderately fire-tolerant tree species in the overstory (table 8.5). An average of 1 trees/ha of grand fir (*A. grandis*) and 0.25 trees/ha of mountain hemlock (*Tsuga mertensiana*) were retained to maintain the density required for the least-intensive prescription of thinning from below to 100 UMZ (table 8.5). No more than 4 trees/ha each of lodgepole pine (*Pinus contorta*), western white pine (*P. monticola*), Douglas-fir (*Pseudotsuga menziesii*), and grand fir were retained to achieve the prescribed density for the prescription of thinning from below to 75 UMZ (table 8.5). Except for small numbers of lodgepole pine, only ponderosa pine was retained in the overstory of 50 UMZ treatment units (table 8.5). An average of 1.3 trees/ha of grand fir was retained to attain the prescribed density for the 75 UMZ + Gaps treatment (table 8.5).

Although before treatment ponderosa pine occurred throughout the LMS, grand fir, Shasta red fir (*Abies magnifica*), sugar pine (*Pinus lambertiana*), and mountain hemlock had greater presence in blocks 1, 2, and 3, than in block 4, which was dominated by ponderosa pine. Block 1 had the most live conifer trees in the overstory of all sampling plots combined (4,326 trees in 100 plots sampled) and had the highest diversity of overstory conifers (Shannon diversity index [H] = 1.03; table 8.6). Block 1 is the highest in elevation and has east and south aspects, and these conditions favor a mix of conifer species. Experimental unit 12 in block 1 had the highest diversity of all units (H = 1.31). The pretreatment mix of overstory conifer species in unit 12 led to a diverse set of posttreatment conditions as reflected among the sampling plots. Blocks 2 and 3 had combined totals (summed across 125 plots per block) of 2,252

Table 8.5—Density of live overstory trees by species pretreatment (pre) and posttreatment (post) for thinning treatments in the Lookout Mountain Thinning and Fuels Reduction Study

			Treatment type			
		Reference	100 UMZ	75 UMZ	75 UMZ + Gaps	50 UMZ
		Trees per hectare				
Ponderosa pine	Pre	1,738	2,078	1,431	1,067	1,945
	Post	1,408	292	207	138	134
Grand fir	Pre	363	633	1,620	326	407
	Post	271	4	4	4	0
Lodgepole pine	Pre	633	236	1,034	165	149
	Post	397	0	3	0	2
Western white pine	Pre	1	8	274	4	20
	Post	3	0	6	0	0
Sugar pine	Pre	4	3	2	7	0
	Post	1	0	0	0	0
Mountain hemlock	Pre	0	2	32	0	0
	Post	0	1	0	0	0
Douglas-fir	Pre	0	2	4	0	1
	Post	0	0	1	0	0
Shasta red fir	Pre	0	0	530	0	0
	Post	0	0	0	0	0
Subalpine fir	Pre	0	0	20	0	0
	Post	0	0	0	0	0
White fir	Pre	0	0	1	0	0
	Post	0	0	0	0	0
Engelmann spruce	Pre	0	0	1	0	0
	Post	0	0	0	0	0

UMZ = upper management zone.

Table 8.6—Density of live overstory trees pretreatment (pre) and posttreatment (post) by block in the Lookout Mountain Thinning and Fuels Reduction Study

	Block 1 (CW-S1-15)		Block 2 (CW-S1-15)		Block 3 (CW-S1-13)		Block 4 (CP-S3-12)	
	Pre	Post	Pre	Post	Pre	Post	Pre	Post
	Trees per hectare							
Ponderosa pine	749	271	1,184	422	3,391	699	2,935	787
Grand fir	2,186	242	996	36	91	4	76	1
Lodgepole pine	533	3	51	10	771	348	862	41
Shasta red fir	531	0	0	0	0	0	0	0
Western white pine	273	3	16	4	18	0	0	2
Mountain hemlock	32	0	0	0	2	1	0	0
Subalpine fir	20	0	0	0	0	0	0	0
Sugar pine	0	0	1	0	5	1	10	0
Douglas-fir	1	0	3	1	2	0	1	0
White fir	1	0	0	0	0	0	0	0
Engelmann spruce	0	0	1	0	0	0	0	0

Note: Blocks were approximately delineated by Volland's (1988) plant associations where possible: CW-S1-15—mixed conifer/snowbrush-sedge, CW-S1-13—mixed conifer/snowbrush-greenleaf manzanita, CP-S3-12—ponderosa pine/antelope bitterbrush-snowbrush. Total stem counts based on sums across 25 plots per either 5 (blocks 2 to 4) or 4 (block 1) experimental units.

and 4,280 live trees, respectively. Corresponding H indices for blocks 2 and 3 were 0.68 and 0.47, respectively. Block 4 is the lowest in elevation, experiences the highest temperatures, and was occupied by plant communities indicative of dry sites. Block 4 was delineated as a ponderosa pine-dominated plant association with a total of 3,884 live trees (summed across 125 sampled plots) and had the lowest diversity ($H = 0.45$).

Variability

Overstory composition was simplified through implementation of the thinning and understory fuels reduction treatments. Among the 19 experimental units, the number of overstory tree species observed ranged from a minimum of 2 to a maximum of 10, with a total of 11 tree species observed across the study. For treated units, there was an average of 4.8 species per unit pretreatment and 1.8 species per unit posttreatment. The portion of stem density attributable to ponderosa pine ranged among treatments from 0.55 to 0.79. Posttreatment the proportion of stem density attributable to ponderosa pine ranged from 0.94 to 0.99 for treated units and averaged 0.67 for the untreated reference units. Standard deviations for the mean proportion of ponderosa pine ranged from ±0.01 to ±0.06 for treated units and was ±0.23 for reference units.

Implications

The long-term goal of the LMS is to develop a forest compositionally and structurally resilient to wildfire through manipulation of stand structure and composition. However, the study landscape encompasses environmental variation (due to elevation, aspect, and soils), which affects forest composition (both overstory tree and understory shrub and forb vegetation). The interaction of topography and microclimate with historical fire was likely a major driver in determining the pretreatment forest composition. Looking forward, variations in overstory species composition resulting from thinning and fuel reduction treatments may continue to promote distinct community succession by altering the potential composition of natural tree regeneration and continued overstory interactions with varying edaphic (related to the structure and composition of the soil) and climatic drivers of understory plant community succession. To accurately predict these successional pathways, there must be an understanding of how changing environmental factors will shift habitat suitability for individual species, and therefore, emergent plant associations on the landscape.

Eastside Screens

In the mid-1990s, concerns surrounding the accumulated loss of large-diameter, fire-resistant trees in fire-prone forests east of the Cascades crest came to the forefront of forest management in the Pacific Northwest. Of particular concern for both mesic and dry forests was the role large, older trees played in providing habitat for wildlife species dependent on the structures of older-aged forests. In response, the regional forester instituted amendments—known as the Eastside Screens—to forest plans for several eastside forests in Oregon. One element of the screens was the limitation that precluded the harvest of any tree greater than 21-inch (53.3-cm) d.b.h. This restriction became known as the 21-inch rule (Powell 2013) and remained in place until an amendment shifting the rule to a guideline was signed into place on 15 January 2021.

The scientific merit of the 21-inch rule has been questioned from multiple perspectives (Hessburg et al. 2020, Stine et al. 2014). Likewise, east-side national forests are reviewing forest restoration actions with regard to the 21-inch rule in forest plans and amendments. One perspective of the 21-inch rule is that limiting the harvest of larger trees of all species has resulted in a loss of fire resilience in forest stands where fire-intolerant tree species (e.g., true firs and spruce) have become more prevalent. Also, stand densities in these forests have increased to levels that represent a departure from ecological conditions that occurred under historical disturbance regimes. Another perspective suggests that restoration of fire-resilient forested landscapes may require the harvest of some larger, shade-tolerant conifers. From an economic perspective, harvest of larger trees may be needed to support the commercial forest products infrastructure required for large-scale forest restoration. A counterpoint is that there is too little representation of larger, old trees in the forest landscape, and harvest of any large trees, regardless of stand density or composition, may risk the array of values derived from such features at site and bioregional scales. In 2021, the U.S. Department of Agriculture issued a forest plan amendment for eastern Oregon and southeastern Washington to allow for harvest of some large, old individuals of shade-tolerant, fire-intolerant tree species, in replacement of the 21-inch rule.

As research on a designated experimental forest, the LMS thinning treatments were conducted without adherence to the 21-inch harvest limitation. Harvest treatments were designed to achieve target residual densities through thinning from below. However, larger trees of both fire-intolerant and fire-tolerant species were harvested as needed to reach target stocking levels. From the perspective of the 21-inch rule, it is of interest to know how the treatments as implemented in the LMS affected the abundance, growth, and health of remaining large trees. Thus, the LMS offers unique insights into potential modifications of the 21-inch rule.

Early Observations

Live trees—

The numbers of large trees (≥50.27-cm d.b.h.) harvested to meet target densities increased with increasing harvest intensity as expected (table 8.7). Mean change in large tree density was −22.5, −42.7, −41.7 and −60.5 trees/ha for 100 UMZ, 75 UMZ, 75 UMZ + Gaps, and 50 UMZ, respectively. Mean change in BA per acre had a similar pattern of density decrease with increased thinning intensity. Tree density of Reference stands increased an average of 5.5 trees/ha suggesting slight ingrowth in the 3 years between the initial pretreatment and subsequent posttreatment surveys. It should be noted that there was substantial variation among plots within experimental units for both stem and BA densities. Pockets of tree mortality due to higher-than-intended fire intensity were observed in units 14, 22, 23, 31, and 33. In these pockets, fire-associated mortality compounded the harvest removal of large trees.

Although thinning treatments were designed to first remove less fire-tolerant fir and spruce before removing pines, in effect, ponderosa pine accounted for most of the large trees that were harvested to meet prescribed stocking levels. Harvested large ponderosa pine represented 25.9, 41.5, 48.7, and 65.3 percent of the total pretreatment large tree density for 100 UMZ, 75 UMZ, 75 UMZ + Gaps, and 50 UMZ, respectively. In Reference units, ponderosa pine accounted for an increased mean large tree density of 4.4 trees/ha.

Table 8.7—Change in density and basal area (BA) of large live trees (≥50.27-cm diameter at breast height) from pretreatment to posttreatment by experimental unit and treatment

			All species				Ponderosa pine		Other pine		True fir		Other conifer	
Block	Unit	Treatment	Change in density mean ± SD	Change in BA mean ± SD	Density change	BA change	Density change	BA change	Density change	BA change	Density change	BA change	Density change	BA change
			Trees per hectare	*Square meters per hectare*	*Percent*									
1	14	100 UMZ	−11.0 ± 29.8	−2.6 ± 9.6	−19.6	−13.5	−19.6	−13.5	0.0	0.0	0.0	0.0	0.0	0.0
2	24	100 UMZ	−27.0 ± 29.7	−7.9 ± 9.6	−32.9	−25.7	−26.8	−21.0	0.0	0.0	−6.1	−4.7	0.0	0.0
3	32	100 UMZ	−6.0 ± 22.0	−1.2 ± 7.6	−7.1	−3.7	−7.1	−3.2	−1.2	−1.3	1.2	0.8	0.0	0.0
4	42	100 UMZ	−46.0 ± 36.6	−14.3 ± 2.0	−50.0	−43.8	−50.0	−43.8	0.0	0.0	0.0	0.0	0.0	0.0
2	22	75 UMZ	−63.0 ± 43.4	−19.4 ± 13.9	−56.3	−49.9	−39.3	−33.6	0.0	0.2	−15.2	−15.2	−1.8	−1.4
3	33	75 UMZ	−24.0 ± 45.9	−6.4 ± 15.9	−35.3	−26.8	−35.3	−26.8	0.0	−0.1	0.0	0.0	0.0	0.0
4	44	75 UMZ	−41.0 ± 43.8	−11.6 ± 12.3	−50.0	−42.6	−50.0	−42.6	0.0	0.0	0.0	0.0	0.0	0.0
2	21	75 UMZ + Gaps	−45.0 ± 35.4	−13.9 ± 11.9	−52.3	−45.6	−44.2	−39.0	0.0	0.0	−8.1	−6.6	0.0	0.0
3	34	75 UMZ + Gaps	−45.0 ± 33.1	−12.1 ± 11.0	−46.4	−39.3	−46.4	−39.3	0.0	0.0	0.0	0.0	0.0	0.0
4	43	75 UMZ + Gaps	−35.0 ± 45.1	−10.8 ± 14.7	−55.6	−51.1	−55.6	−51.1	0.0	0.0	0.0	0.0	0.0	0.0
1	15	50 UMZ	−65.0 ± 65.7	−23.8 ± 24.5	−69.9	−69.3	−67.7	−67.0	−1.1	−1.4	−1.1	−0.9	0.0	0.0
2	23	50 UMZ	−71.0 ± 44.9	−23.2 ± 16.2	−71.7	−65.3	−70.7	−63.7	0.0	0.0	−1.0	−1.5	0.0	0.0
3	31	50 UMZ	−64.0 ± 69.3	−22.2 ± 24.9	−69.6	−62.6	−69.6	−62.6	0.0	0.0	0.0	0.0	0.0	0.0
4	41	50 UMZ	−42.0 ± 44.9	−10.4 ± 13.4	−63.6	−51.0	−53.0	−42.0	0.0	0.0	−10.6	−9.0	0.0	0.0
1	13	Reference	8.0 ± 15.7	3.6 ± 3.7	7.3	9.4	1.8	4.4	−0.9	−1.0	6.4	5.9	0.0	0.0
2	25	Reference	15.0 ± 16.1	5.4 ± 5.5	17.2	19.6	17.2	19.6	0.0	0.0	0.0	0.1	0.0	0.0
3	35	Reference	7.0 ± 13.5	2.8 ± 3.5	9.5	11.4	9.5	11.4	0.0	0.0	0.0	0.0	0.0	0.0
4	45	Reference	−8.0 ± 27.7	−2.2 ± 7.6	−11.1	−10.0	−11.1	−10.0	0.0	0.0	0.0	0.0	0.0	0.0

UMZ = upper management zone.

Note: Values are absolute mean ± standard deviation (SD) of stem density and basal area for all species and percentages of change from pretreatment stem density and basal area for all species combined and by species type (ponderosa pine, other pine, true fir, or other conifer). Means and standard deviations are based on n = 25 plot estimates per experimental unit.

Snags—

There was relatively little change in large snag density from pre- to posttreatment in any of the treatments as mean change was −1.0, 1.3, −2.3, and 1.3 snags/ha for 100 UMZ, 75 UMZ, 75 UMZ + Gaps, and 50 UMZ, respectively (table 8.8). A lack of a consistent snag density response to harvest intensity was also reflected in the variation in mean snag densities among experimental units receiving the same treatment. For all harvest treatments, there were experimental units exhibiting both net increases and net decreases. Furthermore, variation among sampling plots within experimental units was substantial relative to mean change, suggesting spatial variation at a scale finer than the experimental unit. This within-unit variation may be related to number of factors such as variability in pretreatment distribution, localized effect of harvest or prescribed fire, or logging damage.

Across all Reference stands, there was a mean decrease in snag density of 2.5 snags/ha. In contrast to the harvest and burn treatments, all four Reference units demonstrated a net decrease in snag density ranging from 1 to 4 snags/ha. A decrease in snags over a 3-year period may point to some specific wind or snow event or may be indictive of the chronic loss through structural degradation. Over time, the expectation may be that without an event that substantially reduces stand density, Reference stands will show an increase in snag density due to competition or biotic stress-induced mortality. Intermittent, stochastic events will reduce snag density in short-term pulses.

Implications

Thinning and fuel reduction treatments resulted in substantial removal of large-diameter trees, predominantly of fire-tolerant pines. Pine removal reflected pretreatment stand characteristics—a ponderosa pine-dominated landscape originating from a stand-replacing fire in the 1860s with small areas originating from a subsequent fire in the early 1900s and a thinning in the 1970s that likely removed less fire-tolerant species. While attentive to species composition, the thinning treatments imposed in this study were focused on well-defined stocking guides intended to reduce risks to bark beetle infestations by maintaining adequate crown spacing to increase available light, water, and nutrients to support the vigor of retained trees. With respect to meeting stocking guidelines, the opportunity to harvest less fire-tolerant species occurred in block 1 where there was a greater abundance of true firs and spruce and in block 4 where there was lodgepole pine.

A key question to be addressed at a larger scale—both spatial and temporal—is the extent to which tradeoffs existed among the benefits of harvesting some larger, fire-tolerant trees to reduce physiological stresses and fire risk and the benefits for various resource values directly attributable to retaining large trees. Treatments imposed in the LMS provide an important opportunity to assess these tradeoffs over time.

Table 8.8—Change in density and basal area (BA) of large snags (≥50.27-cm diameter at breast height) from pretreatment to posttreatment by experimental unit and treatment

			All species				Ponderosa pine		Other pine		True fir		Other conifer	
Block	Unit	Treatment	Change in density mean ± SD	Change in BA mean ± SD	Density change	BA change	Density change	BA change	Density change	BA change	Density change	BA change	Density change	BA change
			Snags per hectare	*Square meters per hectare*	*-----Percent-----*									
1	14	100 UMZ	−4.0 ± 18.7	1.0 ± 7.3	200.0	102.8	200.0	102.8	0.0	0.0	0.0	0.0	0.0	0.0
2	24	100 UMZ	−8.0 ± 13.9	−2.4 ± 4.3	−88.9	−82.7	−77.8	−67.4	0.0	0.0	−11.1	−15.3	0.0	0.0
3	32	100 UMZ	1.0 ± 5.0	0.4 ± 2.0	NA	NA	NA	NA	NA	NA	NA	NA	NA	NA
4	42	100 UMZ	−1.0 ± 13.5	0.1 ± 5.2	−33.3	9.2	−33.3	9.2	0.0	0.0	0.0	0.0	0.0	0.0
2	22	75 UMZ	−3.0 ± 13.1	−0.8 ± 3.7	−37.5	−36.7	−62.5	−68.6	0.0	0.4	25.0	31.4	0.0	0.0
3	33	75 UMZ	7.0 ± 18.4	2.7 ± 7.1	NA	NA	NA	NA	NA	NA	NA	NA	NA	NA
4	44	75 UMZ	0.0 ± 0.0	0.0 ± 0.0	NA	NA	NA	NA	NA	NA	NA	NA	NA	NA
2	21	75 UMZ + Gaps	−5.0 ± 10.2	−1.6 ± 3.4	−50.0	−29.1	−75.0	−66.9	25.0	37.8	0.0	0.0	0.0	0.0
3	34	75 UMZ + Gaps	−3.0 ± 11.0	−0.7 ± 2.6	−100.0	−100.0	−100.0	−100.0	0.0	0.0	0.0	0.0	0.0	0.0
4	43	75 UMZ + Gaps	1.0 ± 13.5	−0.3 ± 5.1	50.0	−27.9	50.0	−27.9	0.0	0.0	0.0	0.0	0.0	0.0
1	15	50 UMZ	−2.0 ± 12.3	−0.4 ± 4.5	−100.0	−100.0	−20.0	−17.2	0.0	0.0	−80.0	−82.8	0.0	0.0
2	23	50 UMZ	0.0 ± 10.2	0.2 ± 3.9	0.0	33.2	0.0	33.2	0.0	0.0	0.0	0.0	0.0	0.0
3	31	50 UMZ	10.0 ± 19.1	4.6 ± 8.1	1000	1904	1000	1904	0.0	0.0	0.0	0.0	0.0	0.0
4	41	50 UMZ	−3.0 ± 8.3	−1.1 ± 3.1	−100.0	−100.0	−100.0	−100.0	0.0	0.0	0.0	0.0	0.0	0.0
1	13	Reference	−3.0 ± 8.3	−1.4 ± 4.2	−60.0	−65.8	−40.0	−36.7	0.0	0.0	−20.0	−29.0	0.0	0.0
2	25	Reference	−4.0 ± 9.4	−1.2 ± 2.8	−100.0	−100.0	−100.0	−100.0	0.0	0.0	0.0	0.0	0.0	0.0
3	35	Reference	−2.0 ± 6.9	−0.4 ± 1.5	−66.7	−58.2	−66.7	−58.2	0.0	0.0	0.0	0.0	0.0	0.0
4	45	Reference	−1.0 ± 5.0	−0.2 ± 1.1	−100.0	−100.0	−100.0	−100.0	0.0	0.0	0.0	0.0	0.0	0.0

NA = not applicable; and UMZ = upper management zone.

Note: Values are absolute mean ± standard deviation (SD) of stem density and basal area for all species and percentages of change from pretreatment stem density and basal area for all species combined and by species type (ponderosa pine, other pine, true fir, or other conifer). Means and standard deviations are based on n = 25 plot estimates per experimental unit.

Understory Structure and Composition

Forest understories contribute to biodiversity and regulate processes, such as conifer regeneration, soil and surficial water retention, nutrient cycling, and to some extent watershed function. As habitat, the understory also lays the foundation for faunal diversity. The dynamic response of understory vegetation to silvicultural prescriptions defines the developmental pathway of ecosystems over time. The LMS did not set specific targets for understory structure and composition following treatments. However, monitoring understory vegetation over time can inform decisions on future prescriptions aimed at promoting the development of desired forest attributes and values. It is important to note that all experimental units, including Reference units, are continuously undergoing vegetation dynamics. Therefore, the comparisons made on changes in composition and structure of vegetation before and after treatment implementation are among treatment comparisons and relative comparisons to dynamic Reference units.

Early Observations

Initial changes in understory vegetation as a result of thinning and fuel reduction treatments varied by plant growth form (fig. 8.2). Mean graminoid canopy cover increased in all treatments. Mean shrub and subshrub canopy covers decreased in all treatments. The response of tree and forb/herb canopy covers varied by treatment. Interestingly, all plant growth forms in Reference units increased in canopy cover from pretreatment to posttreatment surveys.

Tree seedlings—

The change in mean tree seedling canopy cover varied by treatment type (fig. 8.3). Ponderosa pine, lodgepole pine, and grand fir seedlings increased in Reference units likely because of their abundance in the overstory (fig. 8.3). Bitter cherry (*Prunus emarginata*) decreased in Reference units likely as a result of a general intolerance of shade (Esser 1995). The decrease of most species of tree seedlings in treated units may be attributed to rapid increases in understory shrub cover reducing the amount of light that hits the forest floor.

A goal of the LMS treatments was to reduce the abundance and density of regenerating fire-intolerant tree species. Most tree seedlings were expected to be killed by fire, but tree seedlings and saplings <1.37 m in height were also occasionally removed as a function of harvesting logistics. Where overstory trees

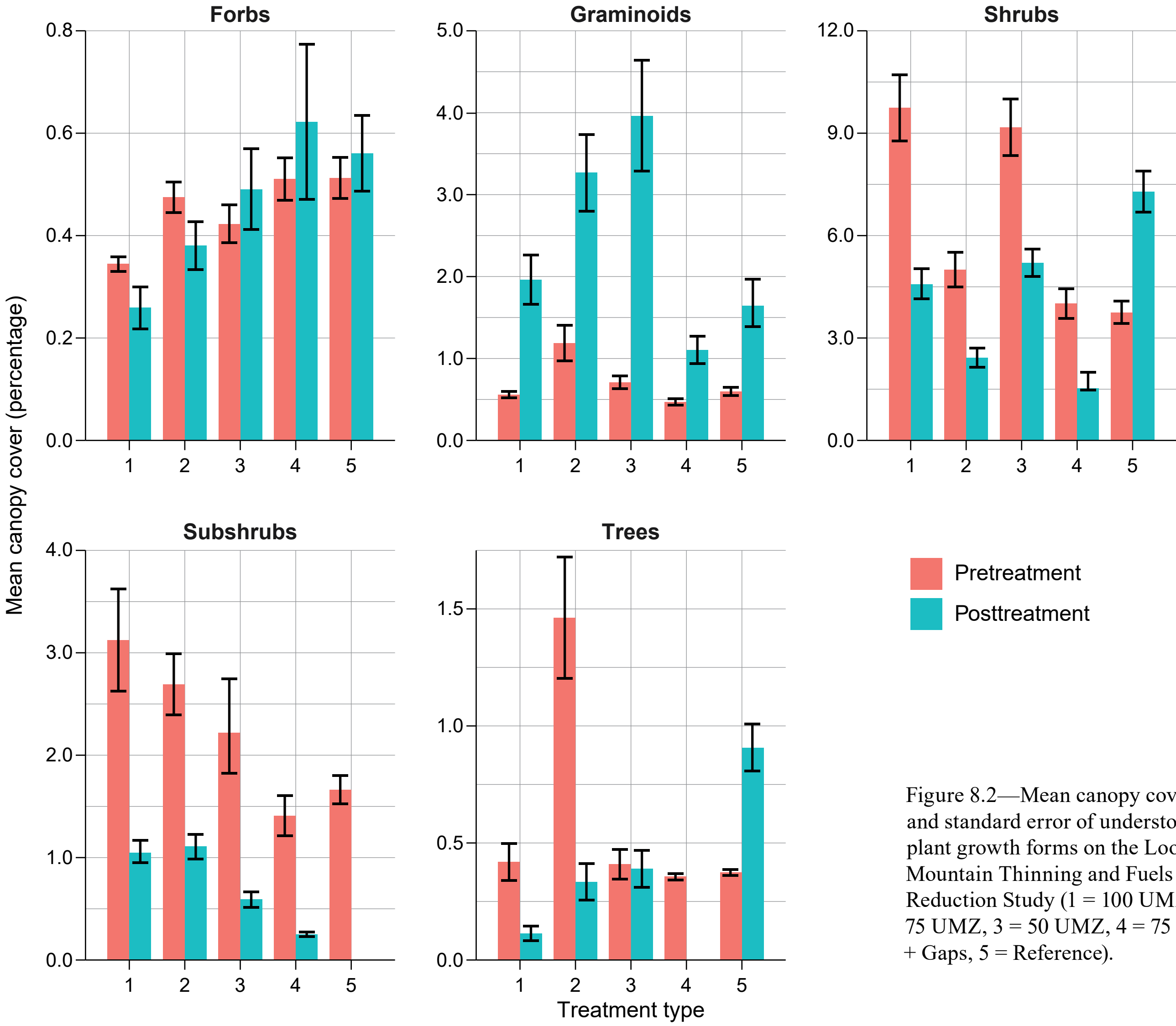

Figure 8.2—Mean canopy cover and standard error of understory plant growth forms on the Lookout Mountain Thinning and Fuels Reduction Study (1 = 100 UMZ, 2 = 75 UMZ, 3 = 50 UMZ, 4 = 75 UMZ + Gaps, 5 = Reference).

were harvested, tree seedlings and saplings <1.37 m in height were at high risk of active removal or incidental removal as slash during harvesting operations, especially in treatment prescriptions with smaller UMZ SDI targets.

The reduction in understory cover of ponderosa pine tree seedlings in 100 UMZ, 75 UMZ, and 75 UMZ + Gaps may be explained by the removal of small-diameter ponderosa pine according to the tree marking guide (see table 5.1). Seedlings may have been damaged during harvesting operations or killed as a result of the prescribed underburning. The increase in seedling cover observed in 50 UMZ may be due to increases in favorable understory conditions for germination (increased sun exposure and mineral soil) (Oswald et al. 1998) and seedling growth. Lodgepole pine was the first cutting preference for overstory tree removal. The preferential removal of this species coincides with the reduction in lodgepole pine seedling cover across treatments. Full sun and mineral soil are ideal for lodgepole pine germination, but high temperature or drought events often kill seedlings

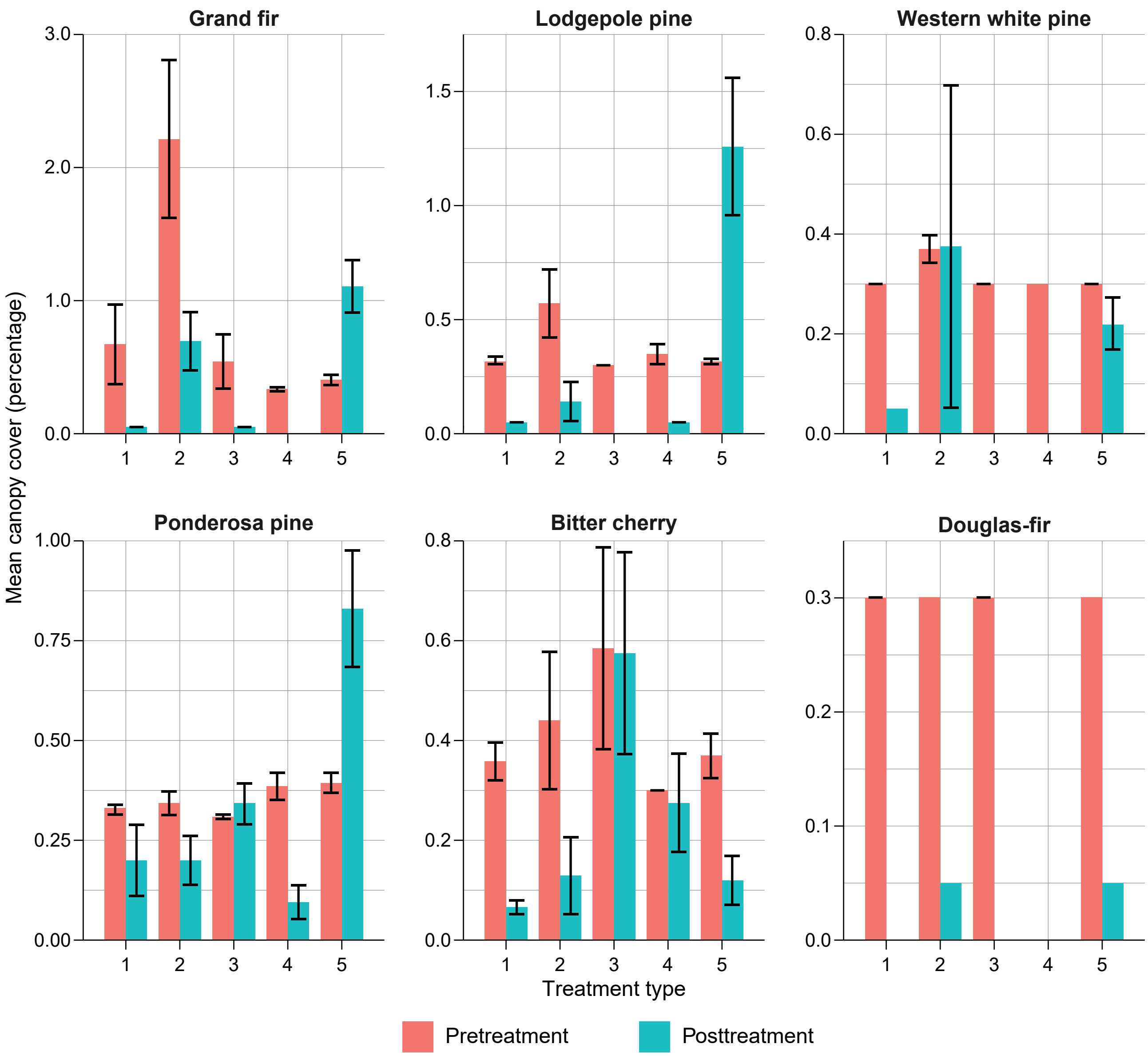

Figure 8.3—Mean canopy cover and standard error of tree seedlings ≤1.37 m in height by species on the Lookout Mountain Thinning and Fuels Reduction Study (1 = 100 UMZ, 2 = 75 UMZ, 3 = 50 UMZ, 4 = 75 UMZ + Gaps, 5 = Reference). Bars with no standard error only had one data point.

(Knapp and Anderson 1980). This may help explain the absence of posttreatment seedlings in 50 UMZ, where prescriptions of low density allowed for greater airflow, sunlight penetration, and increased probability for seedling desiccation than in other treatments. Grand fir is moderately fire resistant and was targeted for removal in the overstory. Grand fir seedling cover was reduced in all treatments. Grand fir regeneration is intermediate in shade tolerance and established beneath the canopy in 75 UMZ, potentially explaining the persistence of 0.70 percent understory canopy cover (fig. 8.3).

Bitter cherry is shade intolerant and can reproduce vegetatively by root or crown sprouting. The presence of bitter cherry seedlings in 50 UMZ and 75 UMZ + Gaps may be attributed to the facilitation of resprouting behavior with increased light availability in the understory. The retention of western white pine (≤0.38 percent) and Douglas-fir (≤0.05 percent) seedlings in less intensive treatments may be an indirect result of the reduced harvest pressure to achieve density targets in these treatments. Reduced harvest pressure can correlate with reduced understory disturbance from on-the-ground operations. Tree seedling canopy cover data for infrequently observed species—mountain hemlock, sugar pine, and Shasta red fir—are not shown in figure 8.3.

Changes in tree seedling canopy cover may also reflect individual species' light preferences (e.g., the coniferous species had greater cover in 75 UMZ, and bitter cherry had greater cover in 50 UMZ). Furthermore, observed variation in canopy cover will likely reflect variability in seed production and dispersal that may or may not be subject to treatment influences. Data collected 2 to 3 years following treatment implementation may be presenting only a partial picture of the natural regeneration and recruitment processes that will continue to play out over multiple seed years and with varying environmental conditions.

Shrubs—

Mean shrub canopy cover (maximum change = −62 percent in 75 UMZ + Gaps) and subshrub canopy cover (maximum change = −82 percent in 75 UMZ + Gaps) decreased in all treatments (fig. 8.4). Ground equipment used during thinning and fuel reduction treatments as well as prescribed fire likely disturbed these growth forms. The creation of gaps in 75 UMZ + Gaps in addition to thinning and fuel reduction treatments may have resulted in a greater percentage of area disturbance, potentially slowing or inhibiting shrub and subshrub growth in the near term.

Chinquapin (*Chrysolepis chrysophylla*) is a focal species in the LMS core vegetation study. Posttreatment mean chinquapin canopy cover was about the same for 100 UMZ, 75 UMZ, and 50 UMZ, with the lowest pre- and posttreatment cover seen in 75 UMZ + Gaps. This species reproduces vegetatively by resprouting and sexually through germination in partial shade (Mckee 1990). The similarity in posttreatment canopy cover among treatments suggests that thinning and fuel reduction treatments stimulated resprouting.

At least two species of currant (*Ribes* spp.) emerged in 50 UMZ after treatment. Because growing points of shrubs are aerial and those of graminoids are subterranean, shrubs are more likely to be damaged by the disturbance of treatment implementation. The emergence of manzanita (*Arctostaphylos* spp.) is explained by uncertainty in identification of manzanita species by field crews.

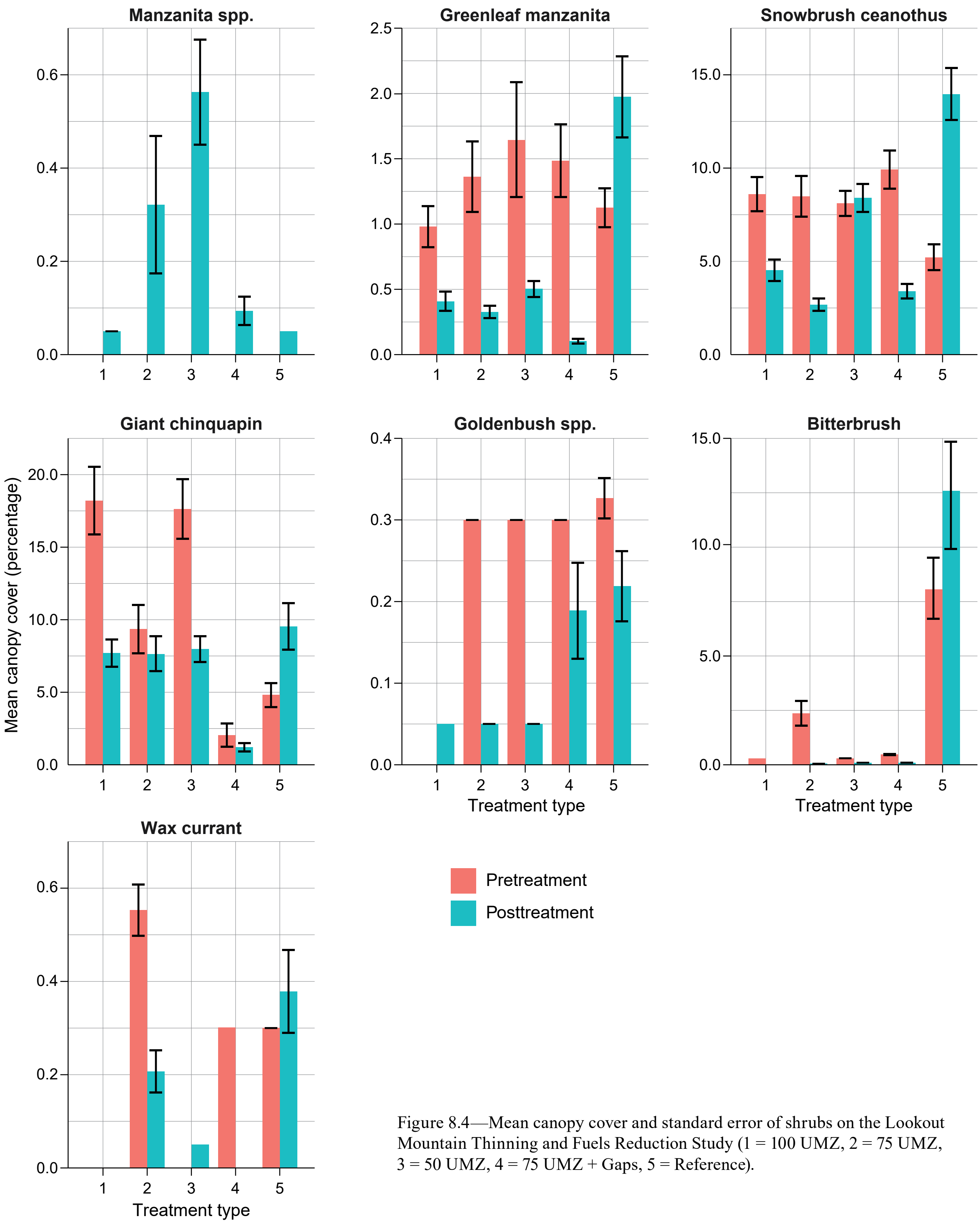

Figure 8.4—Mean canopy cover and standard error of shrubs on the Lookout Mountain Thinning and Fuels Reduction Study (1 = 100 UMZ, 2 = 75 UMZ, 3 = 50 UMZ, 4 = 75 UMZ + Gaps, 5 = Reference).

Subshrubs—

Mean canopy cover of subshrub species decreased in all treatments except for sidebells wintergreen (*Orthilia secunda*) in 75 UMZ, which increased by 25 percent posttreatment (fig. 8.5). Subshrubs are less woody and smaller in size than shrub and tree growth forms. Of note, they are less structurally robust and are therefore easier to trample during harvest operations and surveys.

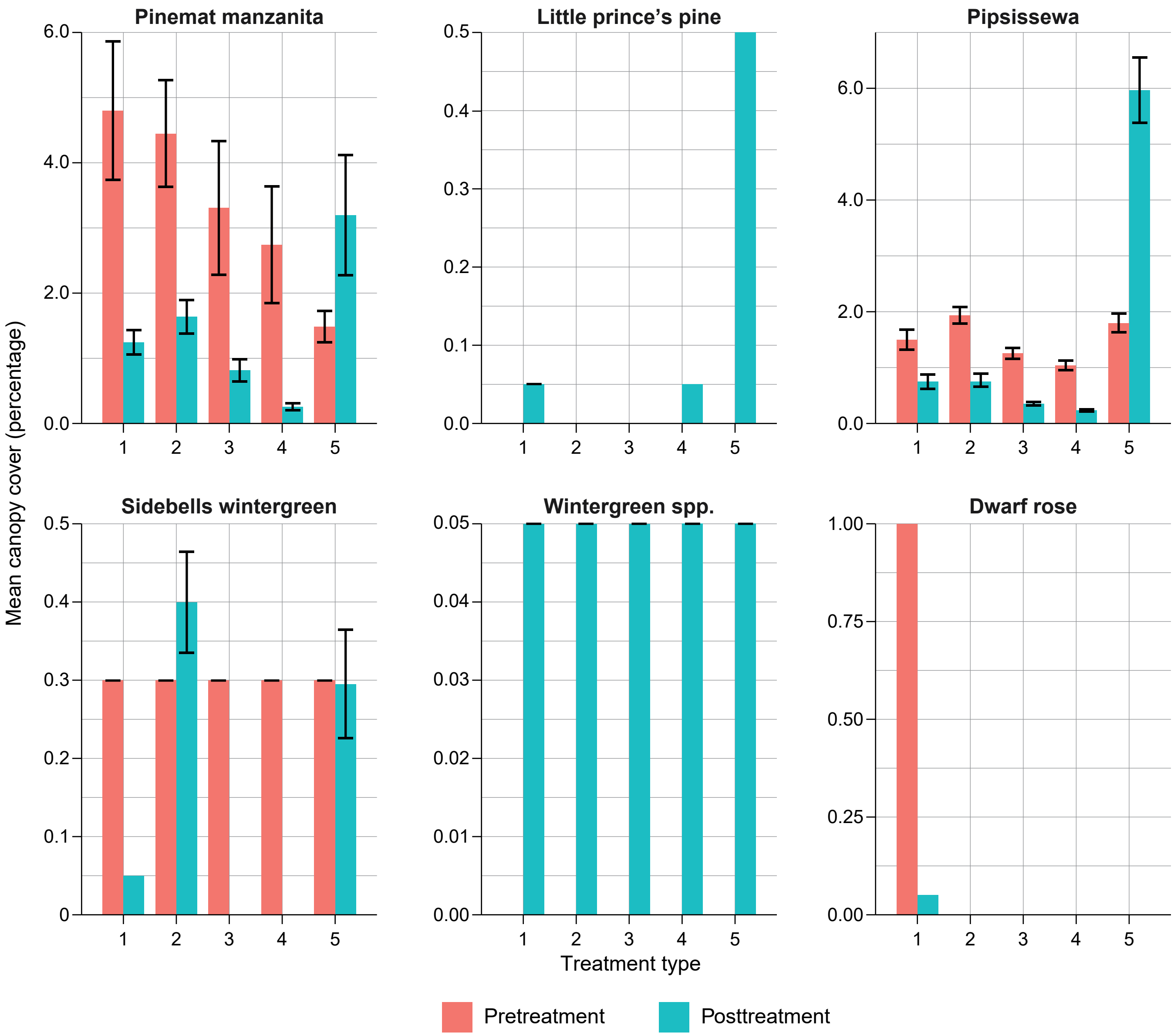

Figure 8.5—Mean canopy cover and standard error of subshrubs on the Lookout Mountain Thinning and Fuels Reduction Study (1 = 100 UMZ, 2 = 75 UMZ, 3 = 50 UMZ, 4 = 75 UMZ + Gaps, 5 = Reference).

Forbs/herbs—

Changes in forb/herb canopy cover varied by treatment (fig. 8.6). Increases or decreases in forb/herb canopy cover for each species treatment depended on numerous botanical and ecological characteristics, including regeneration strategies and light and germination requirements. Groundels (*Senecio* spp.) and thistles (*Cirsium* spp.) are considered weedy species that readily invade new areas following disturbance, so their emergence posttreatment is not surprising. Forb species that were only found in one treatment or infrequently observed are not shown in figure 8.6.

The largest change in mean canopy cover for forb/herbs was a 25 percent decrease in 100 UMZ followed by a 20 percent decrease in 75 UMZ. There was a slight increase in mean forb canopy cover in 50 UMZ (14 percent), 75 UMZ + Gaps (18 percent), and Reference (9 percent). Three genera of forbs emerged posttreatment—groundels in 50 UMZ and 75 UMZ + Gaps, thistles in all treatments, and cudweed (*Gnaphalium* sp.) in 50 UMZ.

Graminoids—

Mean canopy cover of sedges (*Carex* spp.) increased in all treatments. Many sedge species, including species found within the LMS area (e.g., long-stolon sedge [*Carex inops*] and Ross's sedge [*Carex rossii*]), are relatively drought tolerant and reproduce vegetatively in response to disturbance, so an increase in sedge canopy cover is expected after thinning. Variable changes in grass cover may also be a consequence of lumping several grass species that have differing habitat requirements and reproductive strategies into a family-level taxonomic group (fig. 8.7). Grass species were lumped to mitigate uncertainties in identification among successive crews and sedge species were lumped for the same reason.

The increase of graminoids in Reference units may be explained by inherent error in ocular estimation. However, the proportionally larger increases in 75 UMZ and 50 UMZ may be due to a generally positive response by graminoids to harvest disturbance and increased light availability in the understory following thinning and fuel reduction treatments.

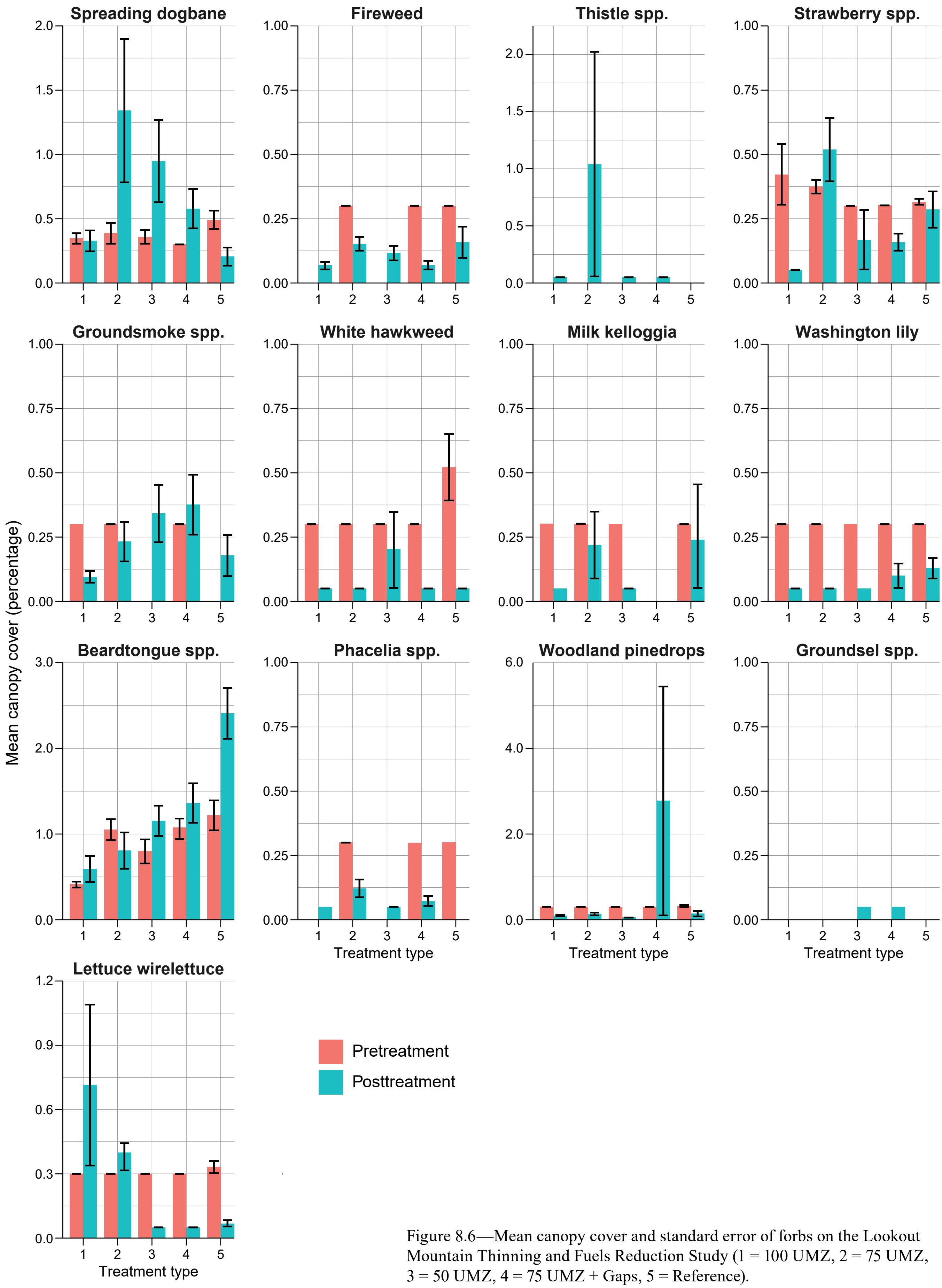

Figure 8.6—Mean canopy cover and standard error of forbs on the Lookout Mountain Thinning and Fuels Reduction Study (1 = 100 UMZ, 2 = 75 UMZ, 3 = 50 UMZ, 4 = 75 UMZ + Gaps, 5 = Reference).

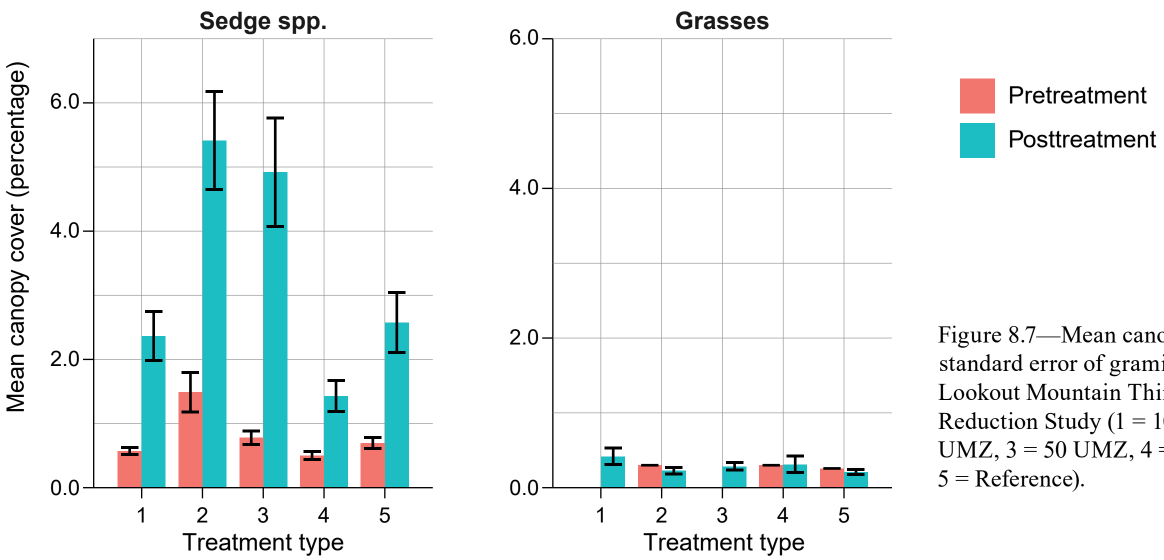

Figure 8.7—Mean canopy cover and standard error of graminoids on the Lookout Mountain Thinning and Fuels Reduction Study (1 = 100 UMZ, 2 = 75 UMZ, 3 = 50 UMZ, 4 = 75 UMZ + Gaps, 5 = Reference).

Understory Species Composition

Species Diversity

Generally, species diversity increased in all treatments and growth forms posttreatment, except subshrubs in 75 UMZ and trees in all treatments (table 8.9). The decrease in tree species diversity in all treatments was expected as harvest prescriptions were aimed at removing fire-intolerant tree species and reducing density of overstory and regenerating trees.

Unexpectedly, 100 UMZ had the highest percentage change in graminoid species diversity because 100 UMZ had few graminoids pretreatment. The disturbance of treatment implementation might have stimulated rhizomatous spreading from graminoids in neighboring plots. Graminoid seeds could have been spread by birds, small mammals, equipment, harvested materials, and field personnel.

The largest percentage increase in species diversity for both forbs and shrubs was in 50 UMZ. Three shrub species emerged in 50 UMZ posttreatment. Greater forb and shrub diversity may be associated with greater resource availability,

Table 8.9—Percentage change in Simpson's species diversity index from pre- to posttreatment averaged for all experimental units within a treatment type by plant growth form (USDA NRCS 2020) for the Lookout Mountain Thinning and Fuels Reduction Study

Treatment	Tree	Shrub	Subshrub	Forb	Graminoid
			Percent		
100 UMZ	−27	1	3	25	80
75 UMZ	−35	2	−4	25	47
50 UMZ	−55	7	0.4	31	62
75 UMZ + Gaps	−42	2	10	11	54
Reference	8	2	5	2	3

UMZ = upper management zone.

including increased growing space provided by more intensive harvest. Another possible explanation for species emergence is the stimulation of reproductive processes like sprouting or microclimatic growing conditions becoming more favorable postharvest. Seven forb species emerged posttreatment in 50 UMZ. Variable changes in subshrub species diversity likely reflect variation in species' sensitivity to disturbance and habitat requirements.

Variability

Blocks 1 and 2 are completely within the delineations of their respective plant associations according to Volland (1988). In block 3, experimental unit 35 may be more accurately described by CP-S3-12 than CW-S1-13 owing to elevational and aspect gradients. Although block 4 has been described as CP-S3-12, the delineation between CP-S3-12 and CW-S1-13 bisects several experimental units in block 4.

To better address variability in plant association types among blocks, plant association understory vegetation components according to Volland (1988) were cross-checked with observed understory vegetation data (tables 8.10 through 8.12). Simpson (2007) noted that several areas in the "mixed conifer" series, central Oregon pumice zone were not easy to identify. Inaccurate delineation of plant associations on the Lookout Mountain unit or the poor performance of Volland's classification system within this area may explain deviations from expected vegetative components according to Volland (1988).

Table 8.10—Canopy cover according to Volland (1988) and for this study for understory vegetation components for plant association CP-S3-12 (ponderosa pine/bitterbrush-snowbrush/sedge) in block 4 (n = 125 plots) of the Lookout Mountain Thinning and Fuels Reduction Study (LMS)

Species	Volland (1988) canopy cover range	Pretreatment canopy cover mean and range	Posttreatment canopy cover mean and range
		Percent	
Bitterbrush	3–35	2 (T–15)	T (T–1)
Snowbrush	3–20	12 (T–50)	7 (T–45)
Western needlegrass (*Achnatherum occidentale*)[a]	T–10	T (T–2)	T (T–2)
Long-stolon sedge (*Carex inops*)[b]	2–25	1 (T–7)	2 (T–35)
Greenleaf manzanita	Present, less than snowbrush or bitterbrush	3 (T–30)	1 (T–10)
Kelloggia (*Kelloggia galioides*)	Present	0	T (T–1)
Fireweed (*Chamerion angustifolium*)	Present	T (T)	T (T)
Dogbane (*Apocynum androsaemifolium*)	Present	T (T–1)	1 (T–10)
Strawberry (*Fragaria* sp.)	Present	T (T)	T (T–1)

[a] All grass species were lumped in the LMS dataset.

[b] All sedge species were lumped in the LMS dataset.

Note: Trace as <1 percent is given a "T," and other nondominant but diagnostic species without canopy cover (percentage) estimates are listed as "present."

Table 8.11—Canopy cover according to Volland (1988) and for this study for understory vegetation components for plant association CW-S1-13 (mixed conifer/manzanita-snowbrush/sedge-penstemon) in block 3 (n = 125 plots) of the Lookout Mountain Thinning and Fuels Reduction Study (LMS)

Species	Volland (1988) canopy cover range	Pretreatment canopy cover mean and range	Posttreatment canopy cover mean and range
	- - - - - - - - - - - - - - - - - -	Percent	- - - - - - - - - - - - - - - - - -
Snowbrush	0–40	7 (T–35)	6 (T–35)
Long-stolon sedge[a]	T–20	1 (T–5)	3 (T–40)
Greenleaf manzanita	T–11	1 (T–10)	1 (0–20)
Bitter cherry	Occasional	T (T–1)	1 (T–1)
Kellogia	Common	T (T)	1 (T–3)
Fireweed	Common	T (T)	T (T–1)
Dwarf rose (*Rosa gymnocarpa*)	Occasional	None	None
Strawberry (*Fragaria* sp.)	Common	T (T)	1 (T–3)
Pipsissewa (*Chimaphila umbellata*)	Occasional	1 (T–15)	2 (T–15)
Hawkweed (*Hieracium* sp.)	Common	T (T)	T (T)

[a]All sedge species were lumped in the LMS dataset.

Note: Trace as <1 percent is given a "T" and other nondominant but diagnostic species without canopy cover (percentage) estimates are listed as "present."

Table 8.12—Canopy cover according to Volland (1988) and for this study for understory vegetation components for plant association CW-S1-15 (mixed conifer/snowbrush/sedge) in blocks 1 and 2 (n = 225 plots) of the Lookout Mountain Thinning and Fuels Reduction Study (LMS)

Species	Volland (1988) canopy cover range	Pretreatment canopy cover mean and range	Posttreatment canopy cover mean and range
	- - - - - - - - - - - - - - - - - -	Percent	- - - - - - - - - - - - - - - - - -
Snowbrush	8–30	6 (T–30)	6 (T–40)
Western needlegrass[a]	T–10	T (T)	T (T–1)
Long-stolon sedge[b]	3–35	1 (T–20)	4 (T–85)
Greenleaf manzanita	Present, less than snowbrush	1 (T–4)	1 (T–5)
Pinemat manzanita	Present, less than snowbrush	4 (T–45)	2 (T–35)
Kellogia	Common	T (T)	T (T)
Fireweed	Common	T (T)	T (T)
Dogbane	Common	T (T–1)	T (T–1)
Strawberry	Common	T (T–2)	T (T–1)

[a]All grass species were lumped in the LMS dataset.

[b]All sedge species were lumped in the LMS dataset.

Note: Trace as <1 percent is given a "T" and other nondominant, but diagnostic species without canopy cover (percentage) estimates are listed as "present."

The extent to which mastication was a source of variability in the understory data is uncertain. The number of masticated understory tree seedlings among 30.5 cm (minimum size for mastication) and 1.37 m in height was not documented. Additionally, it is likely that some unknown amount of understory vegetation, including tree seedlings, was incidentally damaged by equipment during treatment implementation.

The extent to which microsite conditions limited access for the machinery and technicians necessary for mastication and underburning treatments is unknown. For example, areas of steep slopes or rocky outcrops falling within sampling areas were not masticated. Microsite conditions may have reduced or magnified the intended effects of underburning, which may also be a source of variability in posttreatment understory data.

Implications

The understory characteristics described above give a snapshot of the near-term compositional change associated with the experimental treatments. Continued monitoring into the future is necessary to characterize the long-term response of understory vegetation. Similar to that of overstory composition, shifting environmental factors strongly influence the understory pathway of succession. Future changes may warrant new classifications of plant associations as novel vegetation compositions and environmental conditions arise.

Coarse Downed Wood

Coarse downed wood (CDW) plays a multifaceted role in forest function. CDW buildup potentially leads to pockets of high-intensity fire. The structural diversity created by CDW provides a diversity of microclimatic conditions. Some species of trees and other plants in dry coniferous forests rely on these microclimates for germination (Harmon et al. 1986). Thus, the presence of CDW influences species composition at a site. Coarse downed wood also provides habitat for insects, small mammals, reptiles, and amphibians, whose presence can be predicted by the amount, size, and type of CDW (Thomas 1979, Harmon et al. 1986). Log volume and log percent cover were computed to describe CDW within experimental units. These metrics provide partial estimates of fuel loads, small animal habitat, and structural diversity within the LMS.

Early Observations

The posttreatment total count of CDW pieces (measured in strip plots) was 990, 1,877, 810, and 1,172 in blocks 1, 2, 3, and 4, respectively. The total count of CDW pieces (measured in strip plots) was 917, 1,130, 1,066, 756, and 980 in treatments 100 UMZ, 75 UMZ, 50 UMZ, and 75 UMZ + Gaps, and Reference, respectively. The mean length of CDW pieces across the study area was 4.2 m. The mean length of CDW pieces was 3.8, 3.9, 3.3, 3.6, and 6.7 m in treatments 100 UMZ, 75 UMZ, 50 UMZ, 75 UMZ + Gaps, and Reference, respectively.

Volume—

CDW volume was variable within treatments types (fig. 8.8). However, figure. 8.9 suggests that differences in structural and environmental gradients among blocks influenced posttreatment CDW volume.

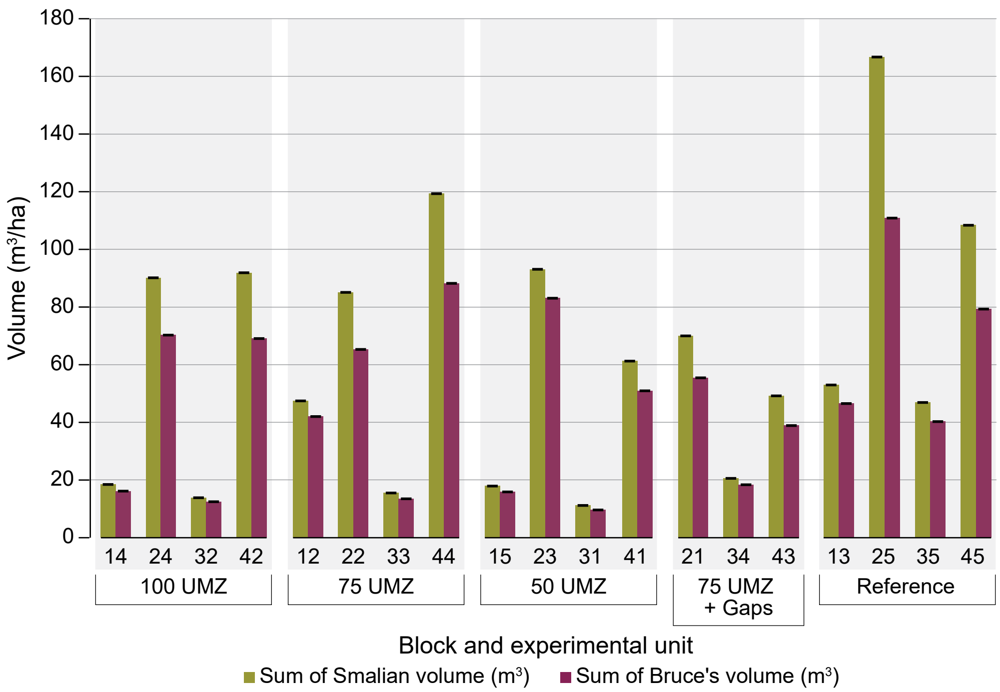

Figure 8.8—Mean volume and standard error of coarse downed wood (m^3/ha) as calculated by Smalian and Bruce's formulas on the Lookout Mountain Thinning and Fuels Reduction Study by experimental unit and treatment type. UMZ = upper management zone.

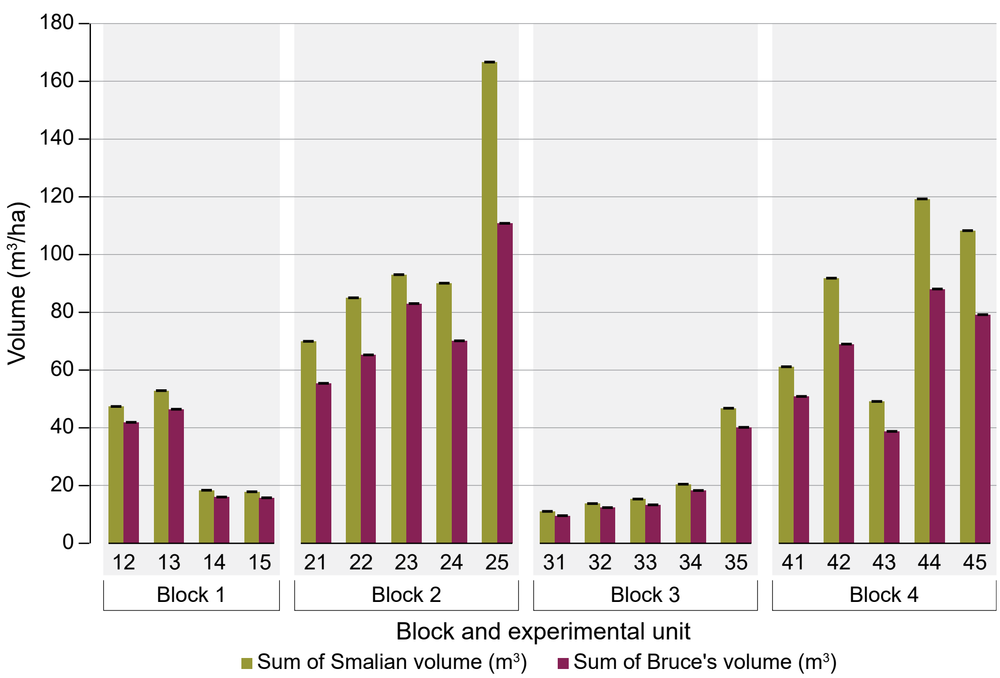

Figure 8.9—Mean volume and standard error of coarse downed wood (m^3/ha) as calculated by Smalian and Bruce's formulas on the Lookout Mountain Thinning and Fuels Reduction Study by experimental unit and block.

Percentage of cover—

Posttreatment percentage of CDW cover varied among experimental units and treatment types (fig. 8.10). Figure 8.11 suggests that differences in structural and environmental gradients among blocks influenced percentage of cover.

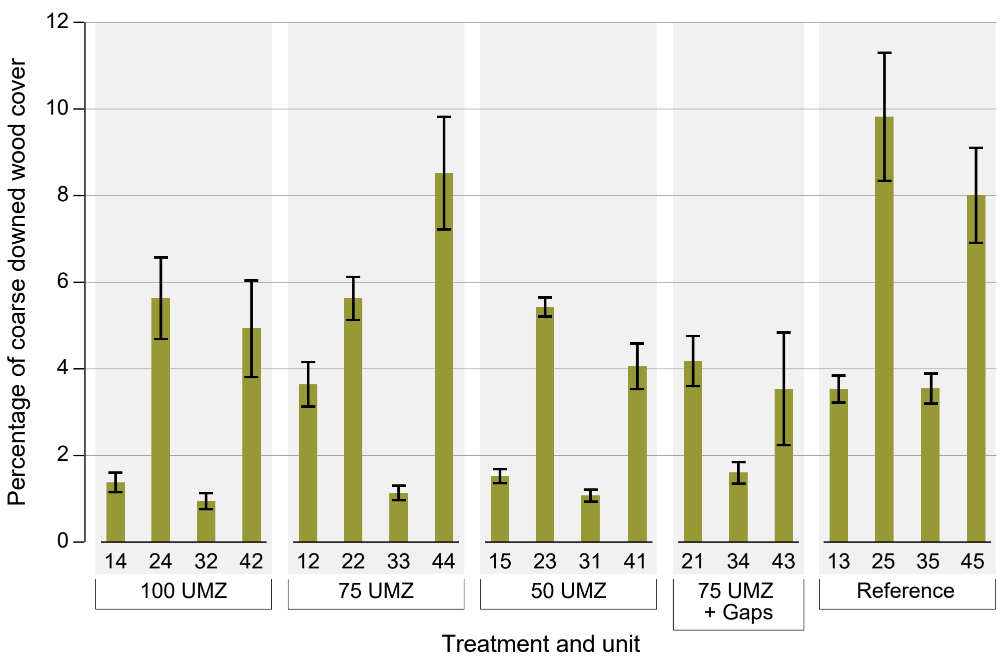

Figure 8.10—Mean cover and standard error of coarse downed wood on the Lookout Mountain Thinning and Fuels Reduction Study by experimental unit and treatment type. UMZ = upper management zone.

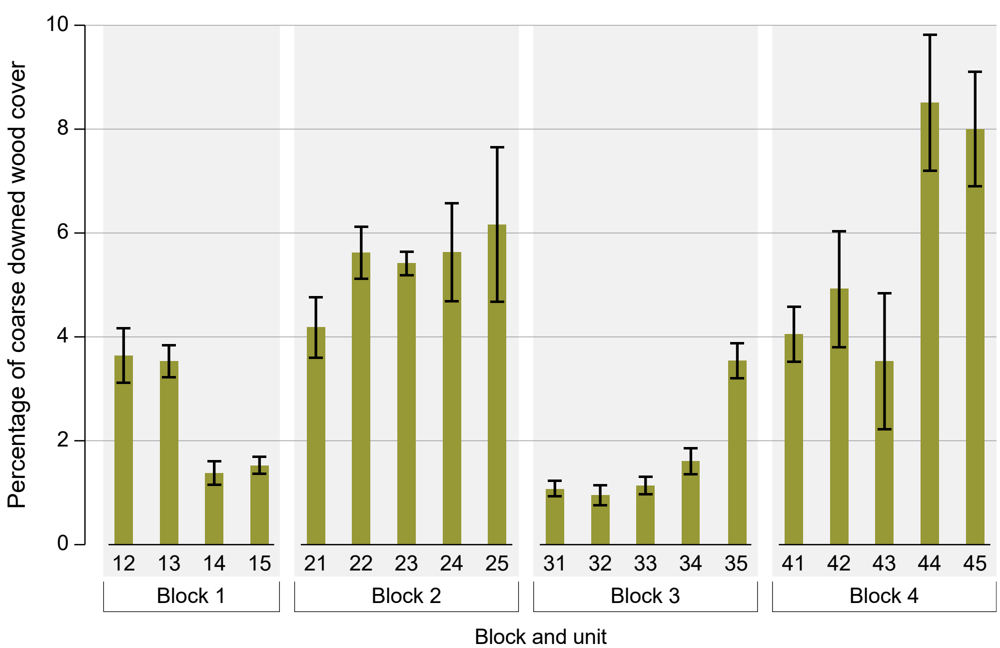

Figure 8.11— Mean cover and standard error of coarse downed wood on the Lookout Mountain Thinning and Fuels Reduction Study by experimental unit and block.

Decomposition class—

The average decomposition class of CDW pieces by treatment was 2.7, 2.8, 2.8, 2.9, and 3.1 for treatments 100 UMZ, 75 UMZ + Gaps, 50 UMZ, 75 UMZ, and Reference, respectively. See chapter 7 for decomposition class descriptions.

The average decomposition class of CDW pieces by block was 3.2, 2.9, 3.1, and 2.4 for blocks 1, 2, 3, and 4, respectively.

Variability

Patches of hotter-than-intended prescribed fire or stagnation and thus longer residence times were observed in experimental units 14, 22, 23, 31, and 33. In both situations, the volume and percentage of CDW cover were likely reduced to a greater extent than in other units. Crews on the scene of the prescribed fire noted stagnation of the burn in block 4.

Implications

Understanding the effects of CDW on wildfire and burn intensity is urgent given the recent catastrophic wildfires experienced in the Western United States. Some of the effects of CDW-associated burn intensity on soil microorganism communities are discussed in ancillary studies completed within the scope of the LMS (see chapter 10). The size and distribution of CDW can also provide information to help predict what wildlife species might be present in an area. The DecAID decayed wood advisor tool (Mellen-McLean et al. 2017) statistically summarized published wildlife and forest inventory research to provide quantifiable and testable CDW attributes in the field. The comparison below of CDW attributes found in the LMS with DecAID data summaries suggests that future surveys could focus on testing predicted presence of wildlife populations. Data on all potential wildlife species are not available in DecAID, and caveats of data usage can be found within the tool.

Mean percentage of CDW cover by treatment type within the LMS ranged from 3 (50 UMZ) to 6 percent (Reference). Mean percentage of CDW cover obtained from wildlife data in central Oregon synthesized in the DecAID tool predicts that 30 percent of the golden-mantled ground squirrel (*Callospermophilus lateralis*) population will use areas with up to 1-percent log cover, 50 percent will use areas with up to 5-percent log cover, and 80 percent will use areas with up to 10-percent log cover.

Mean diameter of CDW pieces within the LMS ranged from 25 cm (75 UMZ + Gaps) to 28 cm (Reference) for the large end and 18 cm (Reference) to 20 cm (100 UMZ) for the small end. The DecAID tool has synthesized information for a few wildlife species that use CDW in ponderosa pine forests by piece diameter. DecAID predicts that up to 30 percent of small ants will use downed pieces with a diameter up to 20 cm, 50 percent will use pieces with a diameter up to 26 cm, and 80 percent will use pieces with a diameter up to 36 cm. It predicts that 30 percent of large ants will use downed pieces with a diameter up to 18 cm, 50 percent will use pieces with a diameter up to 25 cm, and 80 percent will use pieces with a diameter up to 35 cm. DecAID predicts that 30 percent of woodpecker species will use downed pieces with a diameter up to 20 cm, 50 percent will use pieces with a diameter up to 26 cm, and 80 percent will use pieces with a diameter up to 36 cm.

CDW plays an integral role in recycling nutrients to the soil and creating favorable microclimatic conditions for germination and survival of trees. Part of maintaining a resilient forest is understanding the optimal balance of CDW amount and distribution and how thinning and fuel reduction treatments affect this balance over time. While the LMS begins to reveal the impacts of thinning and fuel reduction treatments on CDW, additional time and studies are needed.

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Chapter 9: Short-Term Responses of Bark Beetles to Thinning and Fuel Reduction

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Introduction

Bark beetles are important disturbances in western forests (Fettig et al. 2021). Some species cause extensive levels of tree mortality, as demonstrated by mountain pine beetle (*Dendroctonus ponderosae*) in several pines or western pine beetle (*D. brevicomis*) in ponderosa pine (*Pinus ponderosa*). Others are secondary agents that cause only limited amounts of tree mortality and often go unnoticed. In general, bark beetles require living phloem (the layer of cells within the inner bark that transport photosynthates within the tree) to reproduce. When bark beetle populations are low, the beetles create small gaps in the forest canopy by killing stressed trees. During outbreaks, large numbers of trees may be killed over extensive areas, often affecting many ecosystem services (Morris et al. 2018).

The type of fuel reduction treatment and the manner of implementation have different effects on the fuel matrix, which may influence the susceptibility of forests to bark beetles in different ways (Fettig et al. 2021). For example, thinning and prescribed underburning may affect the health and vigor of residual trees; the size, distribution, and abundance of preferred bark beetle hosts; and the physical environment within forests. Associated reductions in tree density may alter microclimates, affecting beetle fecundity and fitness as well as the phenology and voltinism (number of generations per unit time) of bark beetles and their predators, parasites, and competitors. Reductions in tree density can also disrupt pheromone plumes used for recruiting conspecific bark beetles to the host tree, reducing the frequency of mass attacks and levels of localized tree mortality (Fettig et al. 2007, Thistle et al. 2004). Volatiles released from trees are known to influence the behavior of bark beetles (Seybold et al. 2006, 2018). Fettig et al. (2006) showed that chipping sub- and unmerchantable ponderosa pines and depositing the chips back into treated stands increased the risk of infestation by several species of bark beetles in the Southwestern United States. The effect was due to the release of host volatiles (monoterpenes) from the chips, which enhanced attraction of bark beetles.

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Following prescribed fire, tree mortality can be immediate, owing to consumption of living tissue or heating of critical plant tissues; or it can be delayed, occurring over the course of a few years as a result of fire injuries to the crown, bole, or roots (Hood et al. 2018). Levels of tree mortality attributed to bark beetles following prescribed fire depend on numerous factors, including the tree species, tree size, tree phenology, degree of fire-caused injuries, initial and postfire levels of tree vigor, the postfire environment, and the scale, severity, and composition of bark beetle populations and other tree mortality agents in the area (Fettig et al. 2021). A common concern is that bark beetles may colonize and kill trees that were injured by prescribed fire but that otherwise would have survived (Davis et al. 2012). Following a decade of research at Blacks Mountain Experimental Forest in northeastern California (about 335 km south of Pringle Falls Experimental Forest [PFEF]), Fettig and McKelvey (2014) provided one of the more thorough assessments of the effects of fuel reduction treatments on levels of tree mortality attributed to bark beetles in ponderosa pine (and Jeffrey pine [*P. jeffreyi*]) forests. Twelve experimental units, ranging from 76 to 142 ha, were established to create two distinct forest structural types: mid-seral stage (with low structural diversity) and late-seral stage (with high structural diversity). Following thinning, prescribed underburning was conducted for half of each unit. The authors concluded that the levels of bark beetle-caused tree mortality observed following thinning and prescribed underburning did not interfere with the goal of increasing overall forest resilience (Fettig and McKelvey 2014).

The objective of this study was to determine the short-term responses of bark beetles to the Lookout Mountain Thinning and Fuels Reduction Study (hereafter LMS) treatments. Below we describe methods employed during our assessments and present some preliminary results.

Methods

Following the methods of Fettig et al. (2008), we conducted a census on each experimental unit to locate dead and dying trees by presence of crown fade. Because of the large spatial scale of the LMS, treatment implementations (see chapter 3 for treatment descriptions) and thus our posttreatment assessments were staggered across multiple years (blocks 1 and 3 in 2016, block 2 in 2015, block 4 in 2014). We tallied all recently killed trees ≥10.2-cm d.b.h. (diameter at 1.37 m in height) and identified the causal agent of mortality (table 9.1). We removed a section of bark about 625 cm^2 with a hatchet at about 2 m in height on at least two aspects to determine if any bark beetle galleries were present. The shape, distribution, and orientation of galleries are commonly used to distinguish among bark beetle species (Furniss and Carolin 1977) (fig. 9.1). Bark removal also served as a means of marking trees that had already been tallied (i.e., it took several days to weeks to complete assessment of each experimental unit). Bark removal only occurred on dead trees (with faded crowns) to avoid potentially confounding the results.

Table 9.1—Some notable tree-killing bark beetles endemic to the Lookout Mountain Thinning and Fuels Reduction Study (LMS) area (based on literature)

Common name	Scientific name	Primary host(s) within the LMS area
Fir engraver	*Scolytus ventralis*	Grand fir, Shasta red fir, white fir
Mountain pine beetle	*Dendroctonus ponderosae*	Lodgepole pine, ponderosa pine, western white pine
Pine engraver	*Ips pini*	Lodgepole pine, ponderosa pine, western white pine
Western pine beetle	*Dendroctonus brevicomis*	Ponderosa pine

Figure 9.1—A section of bark removed from a grand fir shows evidence of colonization by fir engraver.

We attributed mortality to bark beetles only when burn severity rankings on individual trees were ≤3 (i.e., we estimated prescribed fire did not directly kill the tree based on external measures of fire severity) (see Fettig et al. 2008), if applicable. Red turpentine beetle (*D. valens*) attacks are typically not considered to be a significant threat to tree health (Fettig 2016); therefore, we did not attribute tree mortality to red turpentine beetle. However, we recorded the number of red turpentine beetle attacks (pitch tubes) of each dead tree (data not shown). The proportions of trees killed were arcsine square-root transformed prior to analyses to satisfy the assumption for normality and to stabilize the variance. We checked the normality of the transformed data using a Shapiro-Wilks test and quantile-quantile plots. We also conducted analyses of variance followed by Tukey's honestly significant difference test when applicable. Analyses were conducted for the proportion of all tree species killed by all bark beetle species, the proportion of each tree species killed by all bark beetle species, the proportion of each tree species killed by each bark beetle species, and finally the proportion of each tree species killed by each bark beetle species by diameter class. We considered treatment, diameter class (10.2- to 20.3-cm, 20.4- to 30.5-cm, 30.6- to 40.6-cm, 40.7- to 50.8-cm, and >50.8-cm d.b.h.), and block in the models, and we used the stats package with R statistical software (version 3.4.4) (R Core Team 2020) via RStudio (version 1.2.1335) (RStudio 2020) to analyze the data.

Results and Discussion

We inventoried a total of 395,054 trees across all experimental units. Levels of bark beetle-caused tree mortality (all bark beetle species combined) following thinning and fuel reduction were very low (0.4 percent; 1,686 trees). No significant differences were observed among treatments ($F_{4, 11} = 2.8$, $P = 0.08$; fig. 9.2) or d.b.h. classes ($F_{4, 67} = 1.8$, $P = 0.14$). Fettig and McKelvey (2014) reported that bark beetles killed 5.6 percent of trees during a 10-year period following thinning and prescribed underburning at Blacks Mountain Experimental Forest. In their study, mortality was concentrated on high-diversity structures, on burned-split plots, and within the two smallest diameter classes (19.0- to 29.2-cm and 29.3- to 39.3-cm d.b.h.). Mortality was highest during the second sample period (3 to 5 years after prescribed underburns) (Fettig and McKelvey 2014), which suggests that increases in levels of tree mortality attributed to bark beetles may still be expected in the LMS.

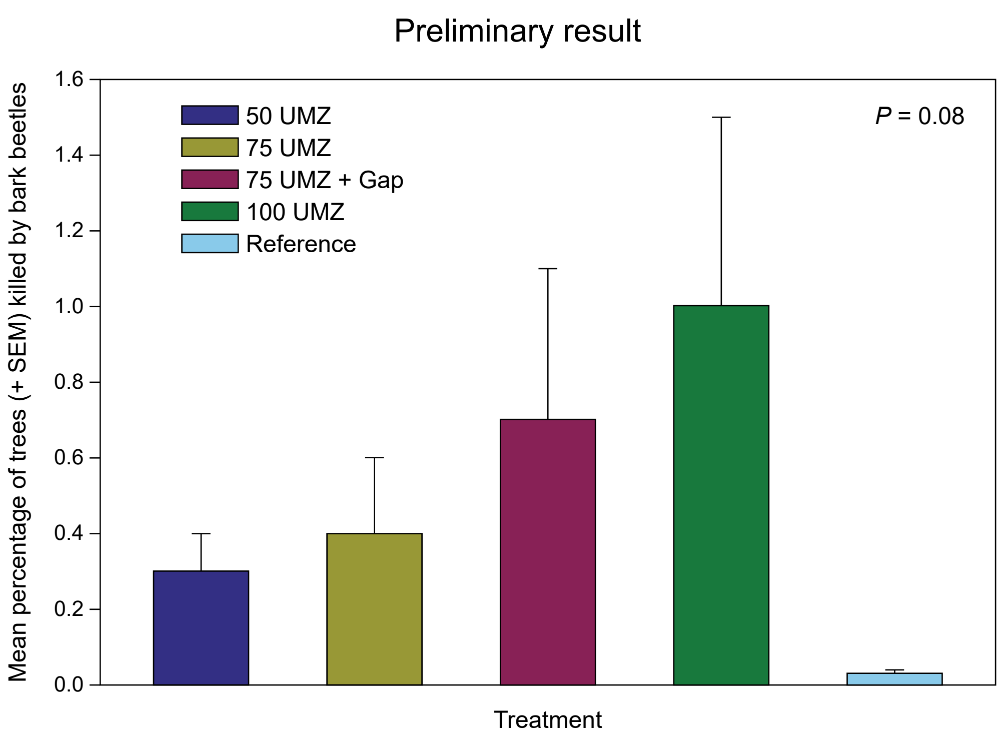

Figure 9.2—Mean percentage of trees + standard error of the mean (SEM) killed by bark beetles (all bark beetle species combined) following thinning and fuel reduction on the Lookout Mountain Thinning and Fuels Reduction Study. No significant treatment effect was observed. UMZ = upper management zone.

Western pine beetle killed the most trees following thinning and fuel reduction (839 ponderosa pines), but levels of ponderosa pine mortality attributed to western pine beetle were low, ranging from 0.03 ± 0.02 percent in Reference to 0.67 ± 0.38 percent in 100 UMZ. No significant differences were observed among treatments ($F_{4, 11} = 2.4$, $P = 0.11$). Western pine beetle is a major cause of ponderosa pine mortality throughout much of western North America (Fettig et al. 2021), and while preferred hosts are of larger diameter (Miller and Keen 1960), mortality following thinning and fuel reduction was highest in the mid-diameter classes ($F_{4, 67} = 4.1$, $P < 0.01$). Ponderosa pines in these diameter classes (30.6- to 40.6-cm and 40.7- to 50.8-cm d.b.h.) are susceptible to western pine beetle; presumably, they have bark thinner than larger ponderosa pines and were likely more stressed by the prescribed fire (Agee and Skinner 2005). In some studies, concerns about maintaining large-diameter ponderosa pines following prescribed fire owing to bark beetle infestations have been realized (Fettig et al. 2021), but this does not appear to be of concern in the LMS, at least in the short term.

Pine engraver (*Ips pini*) killed the second most trees (381), including ponderosa pines, lodgepole pines (*P. contorta*), and western white pines (*P. monticola*). While levels of pine mortality attributed to pine engraver were low, a significant treatment effect was observed ($F_{4, 11} = 3.7$, $P = 0.04$; fig. 9.3) with higher levels of mortality observed in 100 UMZ than in Reference. Pine engraver is one of the most common bark beetles in North America, but the species generally colonizes slash, saplings, and weakened trees (Kegley et al. 1997). Top killing of pines, especially ponderosa pine, is common and often facilitates colonization by more aggressive bark beetles, such as western pine beetle. Although colonization rates by pine

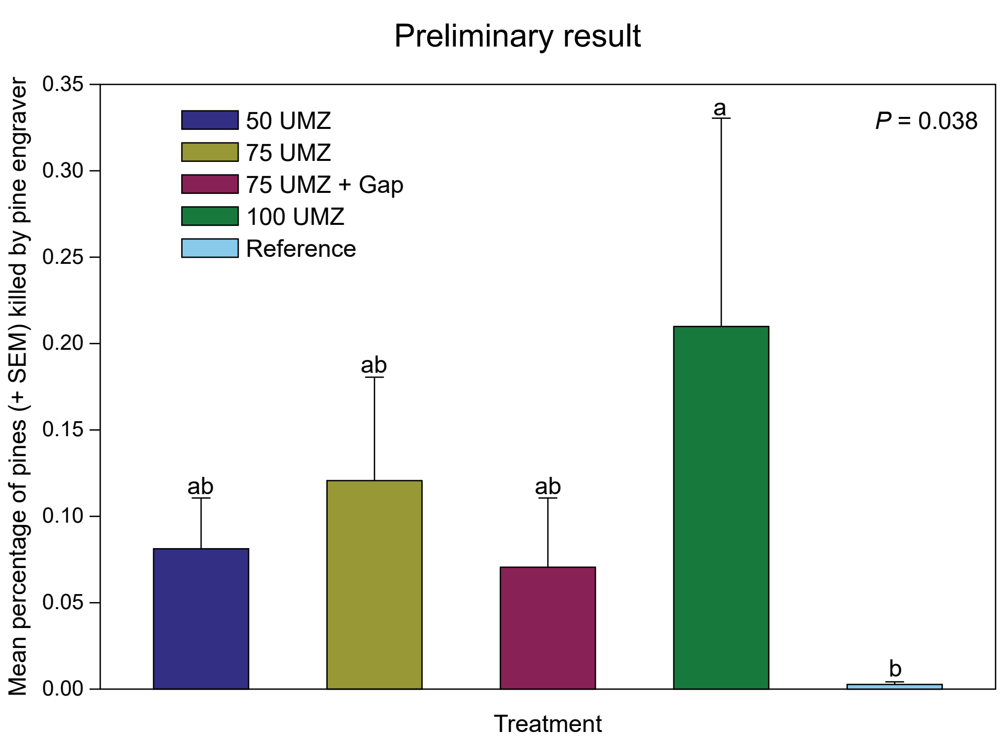

Figure 9.3—Mean percentage of pines standard error of the mean (SEM) (including ponderosa pines, lodgepole pines and western white pines) killed by pine engraver following thinning and fuel reduction on the Lookout Mountain Thinning and Fuels Reduction Study. Columns with "ab" are not significantly different, while columns with "a" or "b" are significantly different from each other. UMZ = upper management zone.

engraver are negatively correlated with tree diameter (Kolb et al. 2006), differences among diameter classes could not be evaluated in the LMS because of the highly skewed nature of these data. Fettig and McKelvey (2014) reported that engraver beetles killed few trees following thinning and prescribed underburning at Blacks Mountain Experimental Forest (0.3 percent) and that activity was limited to the first sample period (1 to 2 years following prescribed underburns).

Fir engraver (*Scolytus ventralis*) killed the third most trees (304 grand firs [*Abies grandis*]). While levels of grand fir mortality attributed to fir engraver were low, a significant treatment effect was observed ($F_{4, 9}$ = 5.1, P = 0.02; fig. 9.4), with higher levels of grand fir mortality observed in 100 UMZ than in 50 UMZ, 75 UMZ + Gaps, and Reference. Fir engraver frequently colonizes firs throughout much of the Western United States, particularly white fir (*A. concolor*). Firs of all sizes may be attacked and killed, but outbreaks are typically associated with trees stressed by drought, defoliation (e.g., by Douglas-fir tussock moth [*Orgyia pseudotsugata*]), root pathogens, or other factors (Berryman and Ferrell 1988). Fir engraver may kill large numbers of firs following prescribed fire (e.g., Fettig and McKelvey 2014), but mortality is generally concentrated in the small-diameter classes (Fettig et al. 2021). No differences were observed among diameter classes in the LMS ($F_{4, 32}$ = 0.6, P = 0.63).

The increased tree mortality observed in 100 UMZ compared to other treatments (see figs. 9.3 and 9.4, but also suggestive in fig. 9.2) is notable. A positive relationship between levels of bark beetle-caused tree mortality and stand density

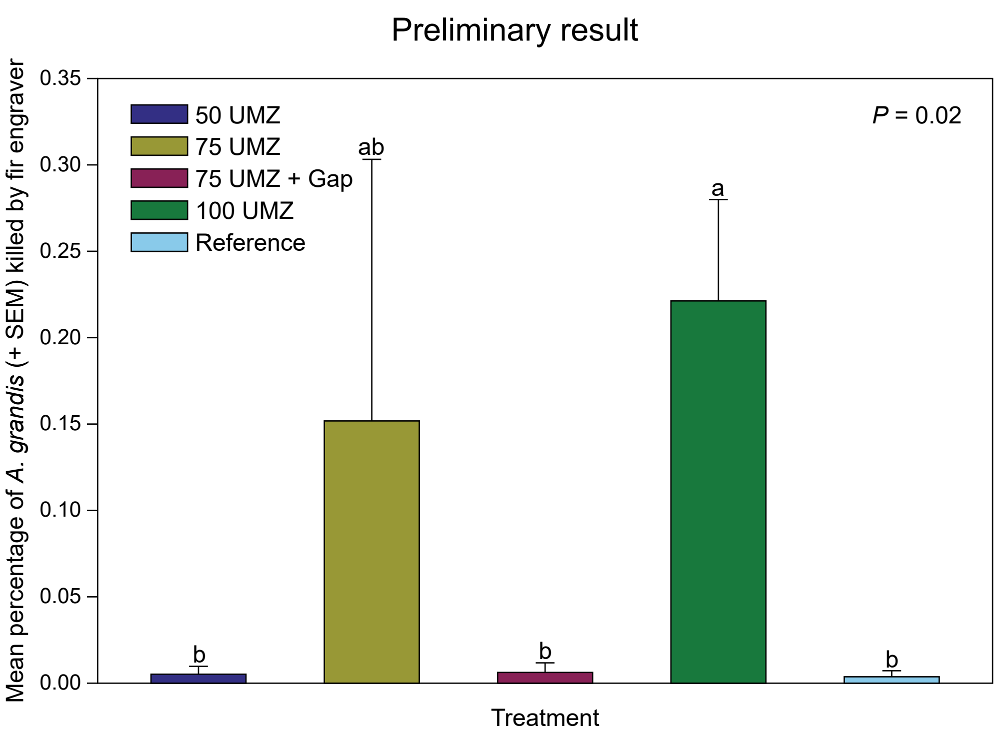

Figure 9.4—Mean percentage of grand fir standard error of the mean (SEM) killed by fir engraver following thinning and fuel reduction on the Lookout Mountain Thinning and Fuels Reduction Study. Columns with "ab" are not significantly different, while columns with "a" or "b" are significantly different from each other. Note: The arcsine-transformed data failed the Shapiro-Wilk's test; however, we proceeded with the analysis of variance based on the quantile-quantile plot. UMZ = upper management zone.

is well documented in the scientific literature for several bark beetle species (Fettig et al. 2007). As such, one might expect higher levels of tree mortality to occur in 100 UMZ owing simply to the higher residual stand densities following treatment. However, lower levels of bark beetle-caused tree mortality occurred in Reference, which had higher stand densities than 100 UMZ. This effect is likely due to an interaction between prescribed fire and stand density as the increased stress imparted on residual trees by prescribed underburning in 100 UMZ was absent in Reference. Fettig and McKelvey (2014) concluded that higher levels of bark beetle-caused tree mortality should be expected following similar fuel reduction treatments that retain larger residual stand densities, independent of the confounding effects of prescribed fire. Although most tree mortality caused by bark beetles following prescribed fire occurs during the first year or two (data presented herein), this pattern could differ in unburned areas such as represented by Reference. The reason, in part, is that unburned areas do not benefit from the positive effects of prescribed fire (such as increased growing space resulting from reduced tree density), which can increase tree vigor and reduce susceptibility to colonization by bark beetles (Fettig et al. 2007).

An important management objective at PFEF was to create stand conditions to restore tree vigor and reduce the risk of landscape-scale, stand-replacement disturbances (Youngblood 2009). Given the low levels of bark beetle-caused tree mortality observed in the LMS, we do not expect bark beetle populations to interfere with this objective in the short term. Similarly, Hood et al. (2016) found that thinning and prescribed underburning increased resistance to bark beetle infestations in ponderosa pine forests in the Northern Rocky Mountains. Furthermore, they suggested that fuel reduction and forest restoration treatments that reduce tree density increase overall resilience in the short term, while the reintroduction of fire is important for long-term resilience. Further evaluations will facilitate a better understanding of the effects of the LMS treatments, and resultant forest structures and compositions on the resilience of these forests to bark beetles. Given the effects of climate change, this could be of utmost importance.

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Chapter 10: Combustion of Large Downed Wood: Temperature-Related Changes to Soil Nutrients and Microbes

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Introduction

The suppression of wildfire for the past several decades in the Western United States has led to an accumulation of fuel buildup in dry, east-side conifer forests (Agee 1993). Historically, low-severity fire helped to reduce the dead and accumulated downed wood and vegetation while also enriching the soil by releasing nutrients bound in litter (Agee 1993, Hessburg et al. 2005). The combustion of large downed wood on the forest floor can create patches of high-intensity fire, which result in canopy openings and mineral soil beds necessary for ponderosa pine (*Pinus ponderosa*) seedlings to outcompete other vegetation (White 1985). However, the exposure of soil to severe heating, such that occurs with the complete combustion of large downed wood, volatizes soil nutrients and causes long-term damage to the function of soil microorganisms and ectomycorrhizal fungi (EMF) (Hebel et al. 2009, Neary et al. 1999, Smith et al. 2017). The functionality of these organisms is imperative for the recovery of vegetation after wildfire, and damage to them may alter the postfire plant community. Additionally, high-severity fire (e.g., high heat) can alter soil nutrient content, water availability, soil pH, and water repellency, all of which can further hinder conifer seedling establishment and growth (Certini 2005; DeBano 2000; Neary et al. 1999, 2005).

As part of the Lookout Mountain Thinning and Fuels Reduction Study (hereafter LMS), Cowan et al. (2016), Reazin et al. (2016), and Smith et al. (2016) measured the effects of soil burn intensity on soil characteristics among two conditions of burned soil by using constructed mega-log decks—parallel stacked large downed logs—and prescribed broadcast burning. The first condition, a high-intensity burn (HB), was intended to parallel soil heating during extreme pockets of wildfire (Reazin et al. 2016, Smith et al. 2016), while the second condition, a low-intensity broadcast burn (LB), was intended to replicate low severity fire typical of dry ponderosa pine forests. A third condition was unburned soil serving as a

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reference for comparison. Smith et al. (2016) compared the temperature, duration of soil heating, and depth of heat penetration of the two soil burn conditions. Cowan et al. (2016) examined the effects of soil burn intensity on soil nutrients and EMF communities. These studies begin to reveal the ways in which different soil heat load intensities alter the growing conditions for ponderosa pine in dry, east-side conifer forests. The rest of this chapter summarizes the methods and findings of Cowan et al. (2016), Reazin et al. (2016), and Smith et al. (2016).

Methods

Researchers conducted their studies within the Lookout Mountain unit of the Pringle Falls Experimental Forest (PFEF) part of the thinning and fuel reduction treatments for the core vegetation study component. The studies were implemented in treated experimental units within block 4 (units 41, 42, 43, 44; 199.51 ha) at 1340 to 1440 m in elevation, which comprised mature ponderosa pine with an understory dominated by bitterbrush (*Purshia tridentata*) and snowbrush ceanothus (*Ceanothus velutinus*). Forest soils were well- to excessively drained, loamy coarse sands and volcanic ash from 62 to 148 cm in depth with duff layers of about 3 cm in depth (Cowan et al. 2016).

On HB plots, mega-logs were constructed with parallel stacked large downed logs to simulate the naturally occurring large downed wood that would burn during a wildfire. Mega-logs were nonrandomly constructed at 12 study sites in summer 2011. A study site center was established at each mega-log site with three burn conditions stemming 10 m from the center in a pinwheel fashion. Mega-log decks cured for 2 years before ignition in spring 2013. Before ignition, the moisture content of each mega-log deck was measured, and mineral soil core samples were taken to calculate the preburn wood moisture and soil gravimetric water content (Cowan et al. 2016, Smith et al. 2016). The soil temperature at the center of each treatment of each study site was measured using thermocouple probes (Sackett et al. 1992). Ignition of the log decks occurred on 14 May 2013 using drip torches, with 10- and 100-hour fuels acting as the primary carriers of the fire.

After piles were burned, Cowan et al. (2016) sampled soil cores at 0- to 10-cm, 10- to 20-cm, and 20- to 30-cm depths and analyzed sample nutrient content, soil organic matter, pH, percentage water content, and total bulk density. Cowan et al. (2016) also planted ponderosa pine seedlings in all three soil burn conditions on all sites 8 days postburn. The seedlings were extracted 4 months later for morphological and molecular analysis of root tip ectomycorrhizas. Reazin et al. (2016) assessed soil fungal communities 1 week prior to and 1 week after the burn.

Statistical analyses of soil heating included an analysis of variance and the calculation of means and standard deviations. Statistical analyses of soil structure and contents included repeated measures linear mixed models, while EMF composition analysis included Simpson's diversity index, richness, relative abundance, and nonmetric multidimensional scaling ordinations.

Results

Smith et al. (2016) noted that an estimated 98 to 100 percent of mega-log decks were consumed across all plots and that maximum soil surface temperatures ranged from 424 to 1168 °C in the HB plots and 42 to 360 °C in the LB plots. The mean maximum soil surface (0 cm) temperature was larger in the HB treatment (759 ± 9 °C) versus the LB (107 ± 43 °C) treatment, but these max temperatures decreased exponentially at deeper soil depths (Smith et al. 2016). At the soil surface, HB plots experienced temperatures exceeding the lethal threshold to microorganism life (>60 °C) for 3.60 to 13.14 hours, while LB plots exceeded lethal temperatures for 0 to 4.68 hours (Smith et al. 2016). At 5- and 10-cm depths, all HB plots exceeded lethal temperatures for soil microorganisms (>60 °C) (Smith et al. 2016). At 5-cm depths, plots in the LB treatment were mostly below the lethal temperature threshold except for two plots, which recorded above lethal temperatures for 1.4 and 2.6 hours. Lethal temperatures were not detected at 30-cm depths for any soil burn condition.

Cowan et al. (2016) found no statistical difference in soil environment conditions or nutrient analysis results among different soil depths. There was no statistical difference found in EMF taxon richness and Simpson's diversity index among sites. However, Cowan et al. (2016) found a difference in the depth of forest floor organic material postburn, with unburned plots having the deepest forest floor, LB plots second, and HB plots third.

Cowan et al. (2016) noted that their EMF sampling protocol only captured about 50 percent of the expected taxon richness of their sites. There were 9 unique taxa found on the root tips of seedlings grown in unburned plots, 22 on those in LB plots, and 7 on those in HB plots. They found no significant difference in taxa composition among LB and HB, but they did find differences with small effect sizes when comparing both LB and HB plots to unburned plots.

Within HB plots, Reazin et al. (2016) documented a substantial turnover of soil fungal communities where dominant basidiomycetes were replaced by ascomycetes postburn. LB plots showed similar but weaker trends.

Conclusions

The combustion of mega-log decks constructed for this study produced the reddened soil typical of extreme soil heating (Hebel et al. 2009, Smith et al. 2016). Smith et al. (2016) recorded lethal temperatures at 10-cm depths under mega-log decks, but the observed temperatures were not as great as similar studies (Busse et al. 2013). The low conductivity of porous pumice soils at PFEF in the spring may explain the reduced severity of soil heat load. Moist spring soils with low porosity would favor heat transfer and potentially an increase in severity of soil heating. Additionally, there was a notable decrease in soil surface temperatures when moving from the center to the edge of the log-deck burn piles.

The capacity of forest soils to buffer soil heating in the absence of large downed wood was demonstrated by the fact that at 5-cm depths, the lethal temperature threshold for microorganisms was exceeded in only two plots for the LB treatment. Soils in LB plots mostly buffered microorganisms from the deleterious effects of soil heating. In contrast, mega-log decks created soil heating conditions that have been shown to reduce the function of microorganisms in the long term (Badia and Marti 2003, Smith et al. 2016). Natural disturbances, such as wind events or insect attack, have the potential to litter the forest floor with large downed wood, which could lead to long-term negative impacts on soil health in the event of a wildfire (Dunn and Bailey 2012).

At 0- to 10-cm depths, the high soil temperatures recorded in HB plots led to the volatization of carbon (C) and organic matter-derived nutrients. Cowan et al. (2016) explained how the degradation of C and nutrient content under these conditions may reduce EMF growth and produce water repellent soil layers. The LB plots did not produce the same negative soil impacts at 0- to 10-cm depths; below that range of depths, the capacity of the soil to buffer soil heating lessened negative impacts.

Cowan et al. (2016) results suggest that EMF can either survive or recolonize HB soils within a single growing season, likely because these fungal communities have adapted to shallow soil heating of historically frequent, low-severity fire. They speculate that heat-resistant spore banks and sclerotia, or mycelia, may be one mechanism for recolonization postfire. Spores may even require soil heating to germinate. A second proposed mechanism of inoculation is the fast rate of mycelial spread by *Rhizopogon* and *Suillus* spp., both of which were found in abundance in unburned, LB, and HB plots.

Reazin et al. (2016) results reveal that fires of differing intensities select distinct soil fungal communities with substantial community turnover. The results indicate that rapid taxonomic responses were attributable to a few fire-responsive fungi, particularly to *Pyronema* sp. and *Morchella* sp., whose frequencies increased more than 100-fold following the HB treatment. High-intensity soil burning, represented by the HB treatment, tends to be spatially confined and limited in scale, but they have the potential to generate distinct patches that may substantially contribute to fungal diversity.

The work of Cowan et al. (2016), Reazin et al. (2016), and Smith et al. (2016) suggests that a mosaic of burn severities resulting from uneven fuel distributions on the landscape may result in the necessary balance of high-intensity and low-intensity burned soils for forest function. Patches of high-intensity burned soils supported by large downed wood allow ponderosa pine seedlings to grow, while the surrounding low-intensity burned soils act as refugia for EMF, which can quickly recolonize more intensely burned soils.

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Chapter 11: Fungal Communities in Woodpecker Cavities: Preliminary Results From Posttreatment Woodpecker Surveys and Fungal Sequencing

Teresa J. Lorenz and Michelle A. Jusino[1]

Introduction

Saab and Lehmkuhl (2011)[2] established surveys to measure cavity-excavating birds pre- and posttreatment as part of the Lookout Mountain Thinning and Fuels Reduction Study (hereafter LMS). These surveys focused on white-headed woodpecker (*Leuconotopicus albolarvatus,* WHWO), which is a species of concern in dry forests of the Northwestern United States. We conducted pretreatment surveys in spring 2011. Only six woodpecker nests were documented in the pretreatment area, and none were WHWO nests. Thus, a decision was made that posttreatment monitoring on the LMS should focus on new research questions, if possible. Ideally, monitoring of cavity excavators posttreatment would explore questions of management interest that could be meaningfully examined within a small geographic area.

Currently, biologists lack information on fungi that cause wood decay for woodpecker cavity excavation in western North America. This is because sporocarps needed for fungal species identification are rare (Lorenz et al. 2015), and the genomics techniques for detecting these fungi have only been developed in the last few years (Jusino et al. 2014, Lindahl et al. 2013, Lindner and Banik 2009). Sequencing of wood decay fungi in woodpecker excavations within the Pringle Falls Experimental Forest (PFEF) was expected to provide a means for assessing what snag decomposition characteristics are associated with cavity excavation by woodpeckers. Furthermore, measuring vegetation associated with fungal occurrence in different treatments provided insights into management techniques that promote fungal colonization of snags. Such studies could be conducted at smaller spatial scales than studies of woodpecker nest occupancy. Accordingly, a decision was made to sample wood decay fungi at Lookout Mountain (a smaller area within the PFEF) in 2017 while also surveying the LMS treatments for woodpecker nests. This chapter presents preliminary results on occurrence of avian cavity excavators posttreatment at the LMS area and fungi sequenced from their nest cavities.

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[2] Saab, V.A.; Lehmkuhl, J.F. 2011. Draft study plan: Impacts of experimental thinning and fuel reduction alternatives on white-headed woodpecker ecology at the Pringle Falls Experimental Forest, Oregon. 35 p. Unpublished document. On file with: U.S. Department of Agriculture, Forest Service, Pacific Northwest Research Station, Forestry Sciences Laboratory, 3200 SW Jefferson Way, Corvallis, OR 97331.

Methods

As noted above, pretreatment surveys for cavity-excavating birds in 2011 focused on the WHWO. Past studies in central Oregon indicated that this species is most commonly found below elevations of 1600 m in this region;[3] pretreatment woodpecker surveys were restricted to elevations below 1646 m on Lookout Mountain.[4] In 2017, we also restricted surveys to elevations below 1646 m to enable comparisons with the 2011 survey effort.

We searched for woodpecker nests during three posttreatment visits on 10 through 12 May, 25 through 28 May, and 7 through 9 June 2017. We focused nest searches on four woodpecker species: WHWO, black-backed woodpecker (*Picoides arcticus*, BBWO), hairy woodpecker (*Leuconotopicus villosus*, HAWO), and northern flicker (*Colaptes auratus*, NOFL). We noted sapsucker (*Sphyrapicus* spp.) nests when we encountered them but did not actively search for their nests.

To search for nests of focal species, we walked transects spaced about 250 m apart throughout all LMS experimental units that were surveyed for woodpeckers in 2011. We conducted surveys between sunrise (around 0500 hours) and 1600 hours. When we encountered territorial or breeding behavior in focal species, we paused transect surveys and observed woodpecker behavior until a nest was located or until the bird was lost from view for about 30 minutes or more, at which point we resumed surveying transects. We followed birds off the transect line whenever necessary to track them to nests but did not approach nests until after the nesting season, unless nests were found unexpectedly while walking through the experimental units.

We alternated the order that transects were surveyed each day to ensure that all portions of the LMS were visited in morning and afternoon hours. Our focal species are typically territorial and nest >300 m from conspecifics. Once we located nests for each species in a particular area, we would not resurvey the area unless we detected breeding behavior of other focal species in the area. However, if we detected breeding behavior but did not immediately locate nests, we repeatedly visited and searched until a nest was located or the breeding season ended.

Woodpeckers may initiate excavation at many cavities in a breeding season but eventually only complete and use one cavity for laying eggs and raising young. We omitted cavity starts from sampling. We considered cavities complete if adults were observed excavating a cavity that was large enough for them to enter fully

[3] Frenzel, R.W. 2004. Nest-site occupancy, nesting success, and turnover-rates of white-headed woodpeckers in the Oregon Cascade Mountains in 2004. 36 p. Unpublished report. On file with: Audubon Society of Portland, 5151 NW Cornell Road, Portland, OR 97210.

[4] Saab, V.A.; Lehmkuhl, J.F. 2011. Draft study plan: Impacts of experimental thinning and fuel reduction alternatives on white-headed woodpecker ecology at the Pringle Falls Experimental Forest, Oregon. 35 p. Unpublished document. On file with: U.S. Department of Agriculture, Forest Service, Pacific Northwest Research Station, Forestry Sciences Laboratory, 3200 SW Jefferson Way, Corvallis, OR 97331.

or if nestlings were observed inside cavities at a later date. We did not use cavity inspection cameras to view nest contents and confirm egg laying because inspection cameras are associated with decreased nest survival (Newlon and Saab 2011). During vegetation sampling in August and September 2017, we measured cavity depth and confirmed that all sampled cavities were indeed complete excavations.

Vegetation Sampling

In August and September 2017 and after the woodpecker nesting season, we returned to nests to measure vegetation characteristics. For safety reasons, we also restricted our sampling to nests that could be sampled from the ground or from climbing ladders and in which the nest snag appeared stable. We omitted nests from sampling if they were too high to be accessed by ladders (n = 3) or if the snag appeared unstable while setting up a climbing ladder (n = 2).

At all accessible snags, we measured cavity height, tree height, and diameter at breast height (d.b.h.). We noted the presence of visible fungal fruiting bodies (sporocarps) and woodpecker foraging evidence on the bole of nest snags. We also visually estimated the proportion of limbs, branches, bark, and top missing from each snag and the proportion of each bole that was blackened by fire. Wood hardness at each nest was nondestructively measured following Matsuoka (2000) and Lorenz et al. (2015). We measured the surrounding tree and snag density (trees/hectare), and visually estimated canopy cover (with a spherical densiometer) and the shrub and woody debris cover on the ground surrounding each nest snag. We measured cavity sill width, horizontal depth, and vertical depth with a ruler. For making comparisons with unexcavated snags, we paired each nest snag with a randomly selected “control” snag within 75 m. Control snags matched attributes of nest snags but lacked cavities.

All aforementioned features were measured at control snags, except cavity width and depth because control snags lacked cavities. For comparison with nests, we measured wood hardness on control snags at the same height and orientation as its paired cavity. Each nest and paired control snag were marked with a tree tag placed in a nearby stump or log, and we took nine photos at each site. Attributes of nest and control snags were compared using paired sample t-tests; significance was reported at the $\alpha = 0.05$ level.

Fungal Sampling and Sequencing

For sequencing of wood decay fungi, we aseptically collected a sample of wood from all nest and control snags included in vegetation sampling. First, we collected a sample of wood shavings from inside nest cavities with a sterile sharpened spoon following Jusino et al. (2014). We used the spoon to scrape inside the cavity above the nest chamber, a region also called the cavity dome (Jusino et al. 2014). Second, we used a sterile 9-mm wood drill bit to aseptically collect wood shavings from the bole of each nest snag 2 to 5 cm above the nest cavity opening. To collect this sample, we drilled horizontally into each nest snag using a cordless drill with the drill bit parallel to the ground. We collected two wood samples from this region. The first sample contained wood extracted as we drilled from the bark surface to about 3 cm deep. The second sample was collected from wood extracted as we drilled from 3 to 10 cm deep. We kept these samples separate (and sterilized between samples). Our first sample (0 to 3 cm deep) simulated sill or cavity entrance wood. Our second sample (3 to 10 cm deep) represented wood in the nest body region, or cavity chamber wood.

We collected a third sample from the bole of control snags. As noted above, control snags matched attributes of nest snags but lacked cavities. As a result, we could not collect a spoon scraping from control snags. For comparison with nest snags, we sampled wood from control snags at the same height and orientation as its paired nest using the sterile wood drill bit. Spoons and drill bits were sterilized in the field between every sample using a 20-percent sodium hypochlorite bath, isopropyl spray, and flame sterilization.

Shavings were transferred from nest cavities (obtained via spoon scraping) immediately to sterile 1.5-mL tubes containing sterile cell lysis solution (CLS) (Lindner and Banik 2009). Upon collection and until we returned to the field station, tubes were placed on ice, where they were frozen at −18 °C. All samples were then maintained at −18 °C until deoxyribonucleic acid (DNA) extraction, polymerase chain reaction (PCR), and sequencing in September and October 2017. To identify fungal taxa at nest snags and paired control snags, we used ion torrent semiconductor sequencing with 400 base-pair technology. We included a custom mock community control in every run to parameterize downstream bioinformatics, which were run using AMPtk (Palmer et al. 2018). Identification of fungal taxa was carried out to the furthest extent possible based on sequences in GenBank®[5] (NCBI) and UNITE (Kõljalq et al. 2005).

To compare species composition, we used nonparametric multidimensional scaling (NMDS) in the vegan package of R (Oksanen et al. 2012) to plot and visualize differences in fungal community composition. We used nonparametric permutational multivariate analysis of variance tests (Anderson 2001) to test for significant differences in fungi communities as a function of sampling source (cavity spoon scraping, nest snag bole sample, or control snag bole sample).

[5] Genbank® is a genetic sequence database of all publicly available DNA sequences as part of the International Nucleotide Sequence Database Collaboration and can be found at https://www.ncbi.nlm.nih.gov/genbank/.

Results

Woodpecker Surveys

We surveyed 30 km of transects for woodpeckers and located 35 nests within the LMS experimental units (fig. 11.1). The most common species nesting in experimental units were HAWO (n = 13 nests) and WHWO (n = 10 nests), followed by BBWO (n = 5 nests) and NOFL (n = 5 nests). There were two additional locations on the experimental units where breeding behavior of WHWO was observed, but no nests were located. We suspect in these two cases WHWO attempted to nest, but we were unable to find the nests before nest failure.

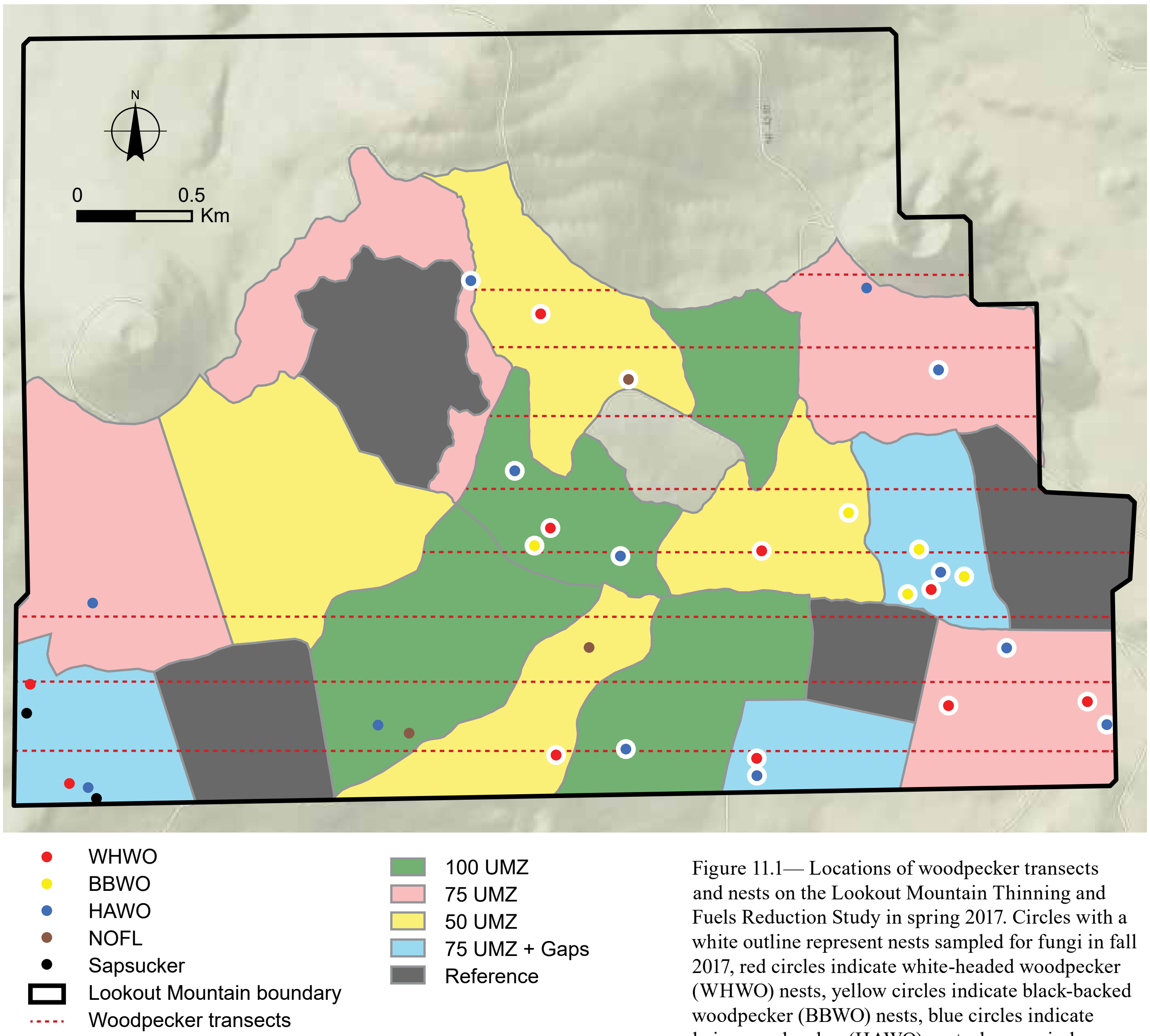

Figure 11.1— Locations of woodpecker transects and nests on the Lookout Mountain Thinning and Fuels Reduction Study in spring 2017. Circles with a white outline represent nests sampled for fungi in fall 2017, red circles indicate white-headed woodpecker (WHWO) nests, yellow circles indicate black-backed woodpecker (BBWO) nests, blue circles indicate hairy woodpecker (HAWO) nests, brown circles indicate northern flicker (NOFL) nests, and black circles indicate nests of Williamson's and red-breasted sapsucker nests. UMZ = upper management zone.

Vegetation Surveys

Nests occurred in all harvest treatments, but no nests of focal species were found in Reference units (table 11.1, fig. 11.1). Most nests were excavated in ponderosa pine (*Pinus ponderosa*) snags (80 percent) and were easily accessible by climbing ladders (80 percent of nest cavities were <5 m high). Median cavity height was 2.7 m (mean 3.5 m, table 11.2). Diameter of nest snags averaged 50.5 cm and was similar to control snags ($\bar{x}$ = 57.4 cm; $t_{23} = 1.22$, $P = 0.2332$). The only vegetation attribute that differed between nest and control snags was hardness of wood within the tree's interior, measured 3 to 10 cm inside the bole (table 11.2; $t_{23} = 9.55$, $P <$ 0.0001). Average hardness of wood at nests was 1.7 nm compared to 6.2 nm for control snags.

Table 11.1—Occurrence of nests for all focal woodpecker species[a] and white-headed woodpecker (WHWO) specifically by treatment year and type on the Lookout Mountain Thinning and Fuels Reduction Study (LMS) in 2017

	Occurrence of nest		
LMS treatment	**All focal species**	**White-headed woodpecker**	**Area surveyed[b]**
	Number (proportion)	*Number*	*Hectares*
Treatment year (burn/thin):			
2013/2011	10 (0.30)	4	199
2014/2012	6 (0.18)	2	165
2015/2013	17 (0.52)	4	310
Reference (no burn/thin)	0 (0.00)	0	111
Treatment type:			
100 UMZ	8 (0.24)	1	221
50 UMZ	7 (0.21)	3	184
75 UMZ	8 (0.24)	2	158
75 UMZ + Gaps	10 (0.30)	4	111
Reference (no burn/thin)	0 (0.00)	0	111

UMZ = upper management zone.

[a] Focal species were WHWO, hairy woodpecker, black-backed woodpecker, and northern flicker.

[b] Estimate of area within each treatment type surveyed for focal species. This is not equivalent to the total area treated within each unit or treatment type at Lookout Mountain.

Table 11.2—Mean and standard deviation of site characteristics for woodpecker nest snags compared to control snags (that lacked woodpecker cavities) on the Lookout Mountain Thinning and Fuels Reduction Study in 2017

Site characteristic	Nest snags (n = 24)	Control snags (n = 24)
Diameter at breast height (cm)	50.5 ± 18.8	57.4 ± 28.5
Cavity height (m)	3.5 ± 2.8[a]	NA
Snag height (m)[b]	15.5 ± 14.2[a]	26.3 ± 11.6
Height of char on bole (m)	8.4 ± 6.7	5.2 ± 6.0
Cavity wood hardness (newton meters)[b]	1.7 ± 0.8	6.2 ± 2.2
Cavity "sill" wood hardness (newton meters)	3.9 ± 2.7	3.1 ± 1.8
Percentage canopy cover	17.2 ± 11.6	17.5 ± 12.3
Percentage coarse woody debris cover on ground	6.6 ± 6.0	3.7 ± 3.3
Percentage fine woody debris cover on ground	10.2 ± 7.6	7.5 ± 7.7
Percentage shrub cover	31.3 ± 24.9	29.0 ± 23.7
Number of snags with conks present on bole	7	7
Number of snags with woodpecker forage sign on bole	24	22

NA = not applicable.

[a] Cavity and tree height were estimated for all 35 nest sites; other characteristics in this table were only measured at the 24 trees also sampled for fungi.

[b] Indicates this characteristic differed at $\alpha < 0.05$ in a paired-sample t-test.

Fungal Sampling and Sequencing

We sequenced 840 fungal taxa from our samples. Fungal communities differed among samples taken from woodpecker cavities, boles of the nest snags, and boles of control snags (fig. 11.2; $r^2 = 0.35$, pseudo-$F = 18.4$, $P < 0.0001$). Fungal communities were most diverse in cavity samples with a median of 59 fungal taxa detected per cavity (and 336 total taxa unique to cavity samples), compared to a median of 13 for boles of nest snags (with 43 unique taxa) and 32 for boles of control snags (with 123 unique taxa). The most commonly occurring taxa within excavated cavities were not wood decay fungi but rather fungi in class Eurotiomycetes (found in 75 percent of cavities) and *Cladosporium cladosporioides* (found in 71 percent of cavities; table 11.3). Half of cavities (54 percent) contained *Rhizopogon* spp., an ectomycorrhizal symbiont of conifers that may have been brought into cavities by rodents. These taxa were not present in samples from boles of nest or control snags.

Fomitopsis pinicola was the most prevalent wood decay fungus sequenced from woodpecker nest snags (found in 58 percent of nests and in both cavity and bole samples). Nest snags containing *F. pinicola* were taller ($\bar{x} = 24$ m) than those lacking *F. pinicola* ($\bar{x} = 6.7$ m; $t_{22} = 3.15$, $P = 0.0046$) and typically had intact tops. A t-test for differences in snag diameter was marginally nonsignificant ($t_{22} = 1.82$, $P = 0.0829$; $\bar{x} = 56$ cm for *Fomitopsis*-infected nest snags and $\bar{x} = 42$ cm for non-*Fomitopsis* snags), indicating a slight trend for *F. pinicola* to be present in larger snags. There was evidence that nest snags containing *F. pinicola* were spatially clustered at the LMS area based on a posthoc Ripley's K-function (fig. 11.3).

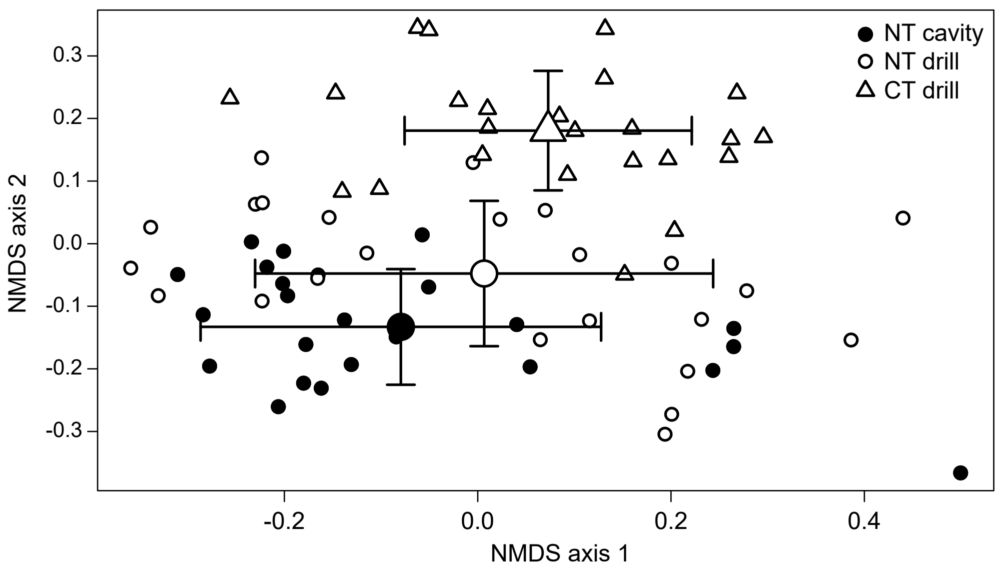

Figure 11.2—Nonparametric multidimensional scaling (NMDS) ordinations of the communities of fungi sequenced from within woodpecker cavities (NT cavity), from boles of woodpecker nest trees (NT drill), and from boles of control snags not excavated by woodpeckers (CT drill) on the Lookout Mountain Thinning and Fuels Reduction Study. The larger circles and triangle represent the means of the points on the two NMDS axes, and the bars represent one standard error from the mean.

Table 11.3—The most prevalent fungal taxa (present in at least 40 percent of samples) sequenced from the bole of nest snags (n = 24), inside woodpecker nest cavities (n = 24), and from the bole of control snags (snags lacking woodpecker excavations; n = 24) on the Lookout Mountain Thinning and Fuels Reduction Study in 2017

Sequence identities, percentage similarities, accession number, and classification of fungal operational taxonomic units (OTUs)	Proportion of sites containing each taxa
Samples from bole of nest snags, for snags not containing *Fomitopsis pinicola*	
GS|99.3|KM493073;k:Fungi,p:Basidiomycota,c:Microbotryomycetes,o:Sporidiobolales	0.50
US|0.9368|KM493531;k:Fungi,p:Basidiomycota,c:Microbotryomycetes,o:Sporidiobolales	0.50
GS|99.2|KC171331;k:Fungi,p:Basidiomycota,c:Tremellomycetes,o:Tremellales, f:Tremellaceae,g:*Cryptococcus*	0.50
GS|99.6|KX100391;k:Fungi,p:Ascomycota,c:Leotiomycetes,o:Helotiales, f:Vibrisseaceae,g:*Phialocephala*	0.40
GS|99.7|KM493763;k:Fungi,p:Basidiomycota,c:Cystobasidiomycetes	0.40
GS|97.0|KM494067;k:Fungi,p:Basidiomycota,c:Tremellomycetes,o:Tremellales	0.40
Samples from bole of nest snags, for snags containing *Fomitopsis pinicola*	
GS|100.0|EU218884;k:Fungi,p:Basidiomycota,c:Agaricomycetes,o:Polyporales, f:Fomitopsidaceae	1.00
Samples from inside woodpecker nest cavities	
GS|100.0|KT220144;k:Fungi,p:Ascomycota,c:Eurotiomycetes	0.75
GS|100.0|JQ936096;k:Fungi,p:Ascomycota,c:Dothideomycetes,o:Capnodiales, f:Cladosporiaceae,g:*Cladosporium*,s:*Cladosporium cladosporioides*	0.71
GS|100.0|JF749180;k:Fungi,p:Ascomycota,c:Dothideomycetes,o:Capnodiales	0.63
GS|100.0|EU218884;k:Fungi,p:Basidiomycota,c:Agaricomycetes,o:Polyporales, f:Fomitopsidaceae	0.58
GS|100.0|KU134841;k:Fungi,p:Ascomycota,c:Taphrinomycetes,o:Taphrinales, f:Protomycetaceae,g:*Saitoella*,s:*Saitoella complicata*	0.54
GS|100.0|KP152486;k:Fungi,p:Ascomycota,c:Dothideomycetes,o:Dothideales, f:Dothioraceae,g:*Sydowia*,s:*Sydowia polyspora*	0.54
GS|98.8|EU552113;k:Fungi,p:Ascomycota,g:*Coniozyma*,s:*Coniozyma leucospermi*	0.54

Table 11.3—The most prevalent fungal taxa (present in at least 40 percent of samples) sequenced from the bole of nest snags (n = 24), inside woodpecker nest cavities (n = 24), and from the bole of control snags (snags lacking woodpecker excavations; n = 24) on the Lookout Mountain Thinning and Fuels Reduction Study in 2017 (continued)

Sequence identities, percentage similarities, accession number, and classification of fungal operational taxonomic units (OTUs)	Proportion of sites containing each taxa
GS|99.6|KT219601;k:Fungi,p:Ascomycota,c:Leotiomycetes,f:Myxotrichaceae	0.54
GS|100.0|HM123665;k:Fungi,p:Ascomycota,c:Sordariomycetes	0.50
GS|98.3|KP411571;k:Fungi,p:Ascomycota,c:Leotiomycetes,f:Myxotrichaceae, g:*Pseudogymnoascus*,s:*Pseudogymnoascus pannorum*	0.50
GS|100.0|GU083211;k:Fungi,p:Ascomycota,c:Leotiomycetes,f:Myxotrichaceae, g:*Pseudogymnoascus*	0.50
US|0.8142|KP891819;k:Fungi,p:Ascomycota,c:Eurotiomycetes,o:Chaetothyriales	0.46
GS|100.0|JX136606;k:Fungi,p:Ascomycota,c:Dothideomycetes	0.46
US|0.8271|KM056294;k:Fungi,p:Ascomycota	0.46
GS|99.2|UDB020417;k:Fungi,p:Ascomycota,c:Eurotiomycetes,o:Eurotiales, f:Trichocomaceae,g:*Penicillium*	0.46
GS|100.0|KY102338;k:Fungi,p:Ascomycota,c:Saccharomycetes, o:Saccharomycetales,g:*Candida*,s:*Candida ponderosae*	0.46
US|0.8853|KU062445;k:Fungi,p:Ascomycota,c:Dothideomycetes, o:Capnodiales,f:Mycosphaerellaceae	0.46
US|0.8474|KP891704;k:Fungi,p:Ascomycota	0.46
SS|1.0000|KF983527;k:Fungi	0.46
GS|100.0|HQ914255;k:Fungi,p:Basidiomycota,c:Agaricomycetes, o:Boletales,f:Rhizopogonaceae,g:*Rhizopogon*,s:*Rhizopogon salebrosus*	0.46
GS|100.0|LC203712;k:Fungi,p:Basidiomycota,c:Tremellomycetes, o:Tremellales,f:Tremellaceae,g:*Naganishia*,s:*Naganishia friedmannii*	0.42
US|0.8255|KU062190;k:Fungi,p:Ascomycota,c:Leotiomycetes	0.42
GS|99.6|KT692578;k:Fungi,p:Ascomycota,c:Leotiomycetes,o:Helotiales, f:Sclerotiniaceae,g:*Botrytis*,s:*Botrytis cinerea*	0.42
GS|100.0|KY103298;k:Fungi,p:Ascomycota,c:Saccharomycetes, o:Saccharomycetales,g:*Debaryomyces*,s:*Debaryomyces vindobonensis*	0.42
Samples from control snags	
US|0.9387|KU164637;k:Fungi,p:Basidiomycota	0.46
GS|97.9|HM595536;k:Fungi,p:Ascomycota,c:Leotiomycetes,o:Helotiales, f:Hyaloscyphaceae,g:*Lachnellula*	0.42
US|0.8078|KY105766;k:Fungi,p:Basidiomycota,c:Microbotryomycetes	0.42

Fomitopsis-infected nest snags had lower fungal diversity than snags lacking *F. pinicola*. The median number of fungal taxa in *Fomitopsis*-infected nest snags as 6 (mean = 12), compared to a mean of 25 taxa for samples not containing *Fomitopsis*. Compared to its prevalence in nest snags, *F. pinicola* was sequenced from only two control snags (8 percent). Both of these control snags were within 200 m of a nest snag containing *F. pinicola*, further indicating spatial clustering of this taxa at the LMS area.

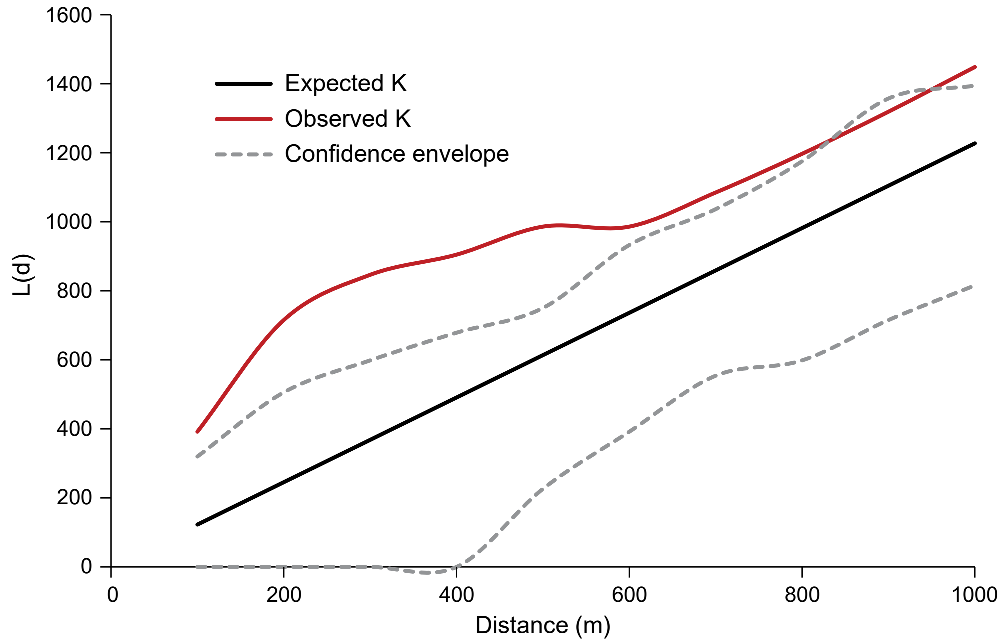

Figure 11.3—Plot of expected and observed Ripley's K functions for analysis of spatial clustering of woodpecker nests containing *Fomitopsis pinicola* on the Lookout Mountain Thinning and Fuels Reduction Study, with confidence envelope from 99 permutations. L(d) is the diameter within which the number of nests would be distributed if the distribution was random. Observed K that is greater than the confidence envelope indicates significant spatial clustering at the associated distance.

Discussion

Our posttreatment surveys revealed many more woodpecker nests posttreatment compared to pretreatment. Pretreatment surveys conducted in 2011 located only six nests for all cavity excavators within the treatment area and no WHWO nests. During posttreatment surveys, we found 35 active nests across all species and 10 nests of WHWO. This represents a fivefold increase in woodpecker nest occurrence. Although formal hypothesis tests are not possible because of small sample sizes, these results suggest that cavity excavators responded positively to treatments implemented as part of the LMS.

Focal species represented both generalist and specialist guilds and forage on a variety of substrates; NOFL and HAWO are habitat generalists that commonly forage on the ground (NOFL) or in dead wood (HAWO), while BBWO and WHWO are habitat specialists that forage predominately on dead wood (BBWO, WHWO) and by bark and foliage gleaning (WHWO) (Elchuk and Wiebe 2003, Lorenz et al. 2016, Murphy and Lehnhausen 1988). These woodpeckers are alike mostly in their use of snags for nesting. Many studies have shown they are attracted to postfire habitats during the breeding season, presumably because burned forests contain high densities of snags (Raphael and White 1984, Saab and Dudley 1998, Tarbill et al. 2015). Thus, it is likely that woodpeckers were attracted to treatments in the LMS because of the availability of fire-killed snags in the experimental units.

In western coniferous forests, research suggests that woodpeckers select snags for nesting because snags are more likely to contain decayed wood for cavity excavation compared to live trees. This is true even for snags that lack visible signs of decay or fungal sporocarps (Lorenz et al. 2015). Fungal fruiting bodies are rare on woodpecker nest snags in the Northwest, and therefore the wood decay fungi that cause wood softening for cavity excavation are not known with certainty. This is the first study to use genomic techniques to identify wood decay fungi at woodpecker nest snags in Oregon. We found *F. pinicola*, present in 58 percent of nest snags, was the most prevalent fungal taxon sequenced from nest snags. In contrast to nest snags, *F. pinicola* was present in only two control snags (9 percent).

F. pinicola is a common wood decay fungus worldwide. It is classified as a brown rot fungus, meaning it primarily consumes cellulose and hemicellulose in wood rather than lignin (Pelaez et al. 1995, Rayner and Boddy 1988). It contains antimicrobial and antifungal abilities that can suppress growth of competitors on dead wood (Dresch et al. 2015, Guler et al. 2009, Keller et al. 1996) even in hot and dry environments, such as the extreme heat characteristic of forest fires (Carlsson et al. 2014). In addition to being a strong competitor, *F. pinicola* is considered a generalist species and predominately colonizes dead trees (Rayner and Boddy 1988).

In several past studies, *F. pinicola* has been used to create woodpecker nest snags via artificial inoculation (Bednarz et al. 2013, Brandeis et al. 2002, Bull and Partridge 1986). In all these studies, snag inoculation with *F. pinicola* largely failed to create woodpeckers nest snags. For example, Bednarz et al. (2013) and Bull and Partridge (1986) collectively inoculated 430 trees with *F. pinicola* in the Northwest to encourage cavity excavation but found no woodpecker nest cavities in these snags 5 to 9 years later. In contrast, on the LMS, snags infected with *F. pinicola* were excavated by woodpeckers 2 to 4 years after tree death. There are many possible reasons why past artificial inoculation efforts have failed to attract woodpecker cavity excavations. *F. pinicola* may not be an important decayer, even at the PFEF. As noted above, the presence of fungal DNA in our samples provides no information on the decay activities of a given taxon. Our sequencing only provides information on presence and absence of fungal taxa. It is possible other fungi were responsible for wood softening in the PFEF snags. Another hypothesis is that in past studies, priority effects in artificially inoculated snags prevented *F. pinicola* from becoming established and decomposing wood.

It is also possible that *F. pinicola* is an important decayer only under limited conditions. At the LMS, we found *F. pinicola* predominately in larger ponderosa pine snags killed as a result of treatment activities. Smaller diameter snags, broken-topped snags, and cut stumps with nests (indicative of tree death before treatments) did not contain *F. pinicola*. Thus, *F. pinicola* was limited to a subset of snags with unique characteristics on the LMS. Moreover, snags infected with *F. pinicola* on the LMS were spatially clustered, indicating pockets of infection, rather than widespread occurrence of this fungus on the LMS. It is also noteworthy that during concurrent sampling of both ponderosa pine and Douglas-fir (*Pseudotsuga menziesii*) snags in central Washington, *F. pinicola* was detected in <10 percent of 49 sampled nest snags (Lorenz 2022).[6] Nest snags are smaller on average on these Washington sites, providing support for the idea that *F. pinicola* is more influential in decay of large snags. But even large ponderosa pine snags in Washington have lacked *F. pinicola*. Overall, we conclude that while *F. pinicola* was common in nest snags excavated in large ponderosa pine snags on the LMS, it is not necessarily an important decay fungus (for woodpecker excavations) throughout the Pacific Northwest.

To resolve some of these uncertainties, additional analyses of wood samples collected from the LMS are planned, including tests of cellulobiase activity following the methods of Garcia et al. (2016) and tests of dilute alkali solubility (Schilling et al. 2015). After analyzing results from these tests, we hope to have information on dominant rot type in each sample, providing insights into decay pathways and taxa causing decay. Also, sequencing of 146 additional samples from sites in central Washington in winter 2018 will shed light on the prevalence of all fungal taxa across a larger geographic area. Our conclusions from this preliminary report may be modified as these new data become available.

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Chapter 12: Summary of Impacts of Experimental Thinning and Fuel Reduction Alternatives on Prey Species of the Northern Spotted Owl

Peter Singleton and John F. Lehmkuhl[1]

Introduction

One component of the Lookout Mountain Thinning and Fuels Reduction Study (hereafter LMS) was to evaluate the effect of silvicultural treatments on the primary prey of the northern spotted owl (*Strix occidentalis caurina*) (NSO), a threatened species that uses the dry forests of the eastern Cascades as critical habitat. Silvicultural treatments intended to improve or create NSO habitat sometimes conflict with treatments intended to restore stand resilience to disturbances (e.g., drought, wildfire, and insect attacks) in ponderosa pine (*Pinus ponderosa*) dominated ecosystems. NSO habitat is complexly structured, multilayered, closed-canopy, mixed-conifer forest (Gaines et al. 2010). Dry ponderosa pine-dominated forest restoration involves fuel reduction and thinning treatments that reduce structural complexity and canopy closure (Lehmkuhl et al. 2004, Youngblood 2006).

The NSO relies on small mammals as a food source, particularly northern flying squirrels (*Glaucomys sabrinus*) and bushy-tailed woodrats (*Neotoma cinerea*). Chipmunks are not primary prey of the NSO, but they occupy similar habitat as the northern flying squirrel and bushy-tailed woodrat. Siskiyou chipmunk (*Tamias siskiyou*), yellow-pine chipmunk (*T. amoenus*), and golden-mantled ground squirrel (*Callospermophilus lateralis*) abundance can be used as a proxy for NSO prey abundance, but these species play another important role in the dry coniferous forests of the eastern Cascades. *Tamias* spp. act as both predators and dispersers of ponderosa pine seeds, simultaneously helping and hindering the establishment of new ponderosa pine seedlings as well as seedlings of other understory shrubs. The objective of this study was to quantify how changes in forest structure are related to changes in the small-mammal community that includes the primary NSO prey species and other similar-sized mammals. This study will help managers understand how silvicultural treatments influence small-mammal communities and prey availability for the NSO.

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Methods

We conducted pretreatment mammal sampling during fall 2011 and posttreatment mammal sampling during September and October 2017. Detailed descriptions on the LMS experimental design and treatment implementations can be found in chapters 3, 6, and 7. For simplicity, we modified the LMS treatment descriptions and classified stands as heavy thin, moderate thin, or no thin/Reference based on observed field conditions. We collected pretreatment data on 12 total experimental units comprising 3 Reference units (15, 25, and 35), 5 moderate thin units (14, 32, 33, 34, and 24), and 4 heavy thin units (21, 22, 23, and 31). We collected posttreatment data on 10 total experimental units comprising 4 Reference units (11, 12, 25, and 35), 2 moderate thin units (14 and 32), and 4 heavy thin units (21, 22, 23, and 31). Experimental units in block 4 were not part of the study because they were logged before or during the initial small mammal sampling period.

We used a mark-recapture approach to estimate the density of the focal species in 10 sampling units. Specific trapping methods and protocols for this study are documented by Lehmkuhl and Conway (2011)[2] and Singleton (2017).[3] We followed standard trapping methods developed by Carey (1991) and implemented by Lehmkuhl (2006). We placed Tomahawk Live Trap® model 201 wire-mesh traps on 8 by 8 or 7 by 9 trap sampling grids at 40-m intervals.[4] Each trapping unit sampled an area of about 11 ha. Traps were baited with a mixture of rolled oats, peanut butter, and molasses. We covered the traps with a waxed-paper milk carton and provided a nest box with wool to insulate the trap. We placed traps either on the ground at the base of a tree or mounted on a tree trunk about 1.5 m above the ground. Trap placement alternated between ground and tree trunk across the trap grids. We generally opened traps on the Monday and closed them on the Friday of each week during the trapping session and checked them daily. We attempted to run traps for 3 weeks to provide three 4-night trapping subsessions, with a 3-night break between each subsession, for a planned total of 12 nights of trapping.

We identified all captured animals to species and noted their condition. Individuals of species other than the project focal species were immediately released after recording species and condition. We weighed focal species individuals and individually marked them with numbered ear tags (National Band and Tag Company, style 10051). Carcasses of animals that died in traps were collected and frozen for subsequent preparation of study skins or other analysis.

[2] Lehmkuhl, J.F.; Conway, A. 2011. Lookout Mountain arboreal rodent capture summary. Unpublished report. On file with: U.S. Department of Agriculture, Forest Service, Pacific Northwest Research Station, Wenatchee Forestry Sciences Lab, 1133 North Western Avenue, Wenatchee, WA 98801.

[3] Singleton, P. 2017. Lookout Mountain spotted owl prey sampling fall 2017 progress report. On file with: U.S. Department of Agriculture, Forest Service, Pacific Northwest Research Station, Wenatchee Forestry Sciences Lab, 1133 North Western Avenue, Wenatchee, WA 98801.

[4] The use of trade or firm names in this publication is for reader information and does not imply endorsement by the U.S. Department of Agriculture of any product or service.

We also recorded nonoperational traps (i.e., traps that were missing, damaged, or stuck open for the night) when the traps were checked. The U.S. Department of Agriculture, Forest Service, Research and Development, Institutional Animal Care and Use Committee (IACUC proposal no. 2017-015) reviewed and approved our capture and handling procedures. Trapping was conducted under an Oregon Department of Fish and Wildlife Scientific Taking Permit (No. 125-17).

We report two measures of animal abundance for each unit in this report: minimum number known alive (MNKA) and density estimates from spatially explicit capture-recapture models (SECR). MNKA is simply the number of unique individuals of each species that we captured in each unit. Although it is informative to specifically document how many animals were captured in each unit, it is also important to note that the MNKA measure is sensitive to trapping effort and provides a biased estimate of animal abundance because more animals are likely to be captured in units with more capture effort. In contrast, SECR analysis provides an unbiased estimate of animal density based on trapping effort, animal capture histories, and locations of captures (Royle et al. 2014). SECR analysis uses a hierarchical maximum likelihood modeling framework to calculate the probability that an animal will be captured relative to its previous capture locations and observed movement distances for that species. The spatial component of SECR analysis provides a derived estimate of animal density that is not possible with traditional mark-recapture analysis.

Results

Pretreatment

Weather during the trapping period (20 through 27 September 2011) was ideal, with warm sunny days and cool, but above-freezing nights. We captured 895 animals (both new and recaptures), mostly Siskiyou chipmunks (58 percent) and golden-mantled ground squirrels (21 percent) (table 12.1). There was some uncertainty

Table 12.1—Summary of captures and recorded animal condition for all species captured during fall 2011 mammal sampling as part of the Lookout Mountain Thinning and Fuels Reduction Study

		Capture type	
Species	**Scientific name**	**New**	**Recapture**
Bushy-tailed woodrat	*Neotoma cinerea*	5	2
Douglas squirrel	*Tamiasciurus douglasii*	25	0
Golden-mantled ground squirrel	*Callospermophilus lateralis*	78	110
Mountain cottontail	*Sylvilagus nutallii*	3	0
Northern flying squirrel	*Glaucomys sabrinus*	40	12
Siskiyou chipmunk	*Tamias siskiyou*	268	254
Snowshoe hare	*Lepus americanus*	8	0
Yellow-pine chipmunk	*Tamias amoenus*	68	22

about the field identification of yellow-pine chipmunks (10 percent of catch) because yellow-pine and Siskiyou chipmunks are similar in coloration and weight (there were many juvenile chipmunks captured). Dissection and examination of bacula of dead Siskiyou chipmunks (i.e., trapping mortalities) in the lab confirmed they were correctly identified, but we had no dead yellow-pine chipmunks to confirm that species' field identification.

Accounting for recaptures, we captured 483 individual arboreal rodents, not counting 8 captures of snowshoe hares (*Lepus americanus*) and 3 captures of mountain cottontails (*Sylvilagus nuttallii*). Combining both species of chipmunk, we captured 336 individual chipmunks, with an average of 22 individual chipmunks per unit (table 12.2). We caught 40 individual flying squirrels (4.5 percent of all new captures) in 9 of the 12 units, with an average of 2.7 flying squirrels per unit (table 12.2). We captured only five individual bushy-tailed woodrats in three units. We captured 25 Douglas squirrels (*Tamiasciurus douglasii*) (5.2 percent of new captures), or about 1.7 Douglas squirrels per unit (table 12.2).

Among trapping units, flying squirrels were most abundant in the mesic block 2 units, with an average of 5.8 flying squirrels per unit, nearly twice the overall average. In contrast, the dry or brushy units in blocks 1 and 3, including unit 31, which had an exceptionally high abundance of flying squirrels, averaged only 1.6 flying squirrels per unit. Excluding unit 31 from that group yielded an even smaller average of 1 flying squirrel per unit. In addition to habitat conditions, it is possible that low abundance of flying squirrels in those units could be a function of predation by the pair of NSO that was detected consistently in the north end of unit 33 over the spring and summer. Trapping grids in blocks 1 and 3 were within 1.5 miles (about 2.4 km) of the mapped NSO activity center. However, the flaw in that conjecture is that units 14 and 31 were both about equidistant from the NSO activity center yet had 0 and 5 flying squirrels captured, respectively; and units 34 and 35 had fewer flying squirrels than unit 31 despite being farther from the NSO activity center.

Table 12.2—Number of individual animals (minimum number known alive, MNKA) of each target species that were tagged in each unit in 2011 mammal sampling as part of the Lookout Mountain Thinning and Fuels Reduction Study

	Unit												
Species	**14**	**15**	**21**	**22**	**23**	**24**	**25**	**31**	**32**	**33**	**34**	**35**	**Total**
Bushy-tailed woodrat	0	0	0	4	0	2	0	0	0	0	1	0	7
Golden-mantled ground squirrel	2	1	0	3	82	0	6	3	3	8	71	9	188
Northern flying squirrel	0	3	7	9	4	6	11	8	2	0	2	0	52
Siskiyou chipmunk	44	62	29	56	54	44	36	24	92	54	15	12	522
Yellow-pine chipmunk	4	0	1	2	0	10	8	8	6	25	19	7	90

Note: MNKA is biased by trapping effort.

Posttreatment

Weather conditions compromised the effectiveness of trapping during the first week. Cold and snowy conditions during the first week led to minimal data collection because we prioritized releasing animals from traps and closing traps. We deployed traps and operated units for a total of 3,520 trap nights between Friday, 22 September and Friday, 6 October. However, some traps were not operational because they were missing, damaged, or simply stuck open. We noted nonoperational traps on 161 occasions (4.6 percent of total deployed trap nights), for a total of 3,359 operational trap nights. Several instances of bear damage to traps were recorded during week two when up to one-third of the traps in one unit (32 traps) were damaged and needed to be replaced (recorded on 29 September). We employed a trap usage mask, or a set of grid cells representing habitat around all traps that is potentially occupied, in the SECR analysis to identify missing traps and incorporated that information into our animal density estimates.

We recorded a total of 1,543 animal capture events, representing nine mammal species and incidental captures of two species of birds (table 12.3). Animals were recorded as alive and in good condition for 91 percent of captures, in poor condition for 4 percent of captures, and dead for 5 percent of captures (table 12.3). For frequently captured species, Douglas squirrel had the highest capture mortality rate (50 percent), and golden-mantled ground squirrels had the lowest (1 percent).

We captured and individually marked a total of 770 study focal species individuals—433 Siskiyou chipmunks, 205 golden-mantled ground squirrels, 107 yellow-pine chipmunks, 21 northern flying squirrels, and 4 bushy-tailed woodrats (table 12.4). The MNKA for Siskiyou chipmunks ranged from 99 individuals in unit 11 to 6 individuals in unit 22. The MNKA for golden-mantled ground squirrels

Table 12.3—Summary of captures and recorded animal condition for all species captured during fall 2017 mammal sampling as part of the Lookout Mountain Thinning and Fuels Reduction Study

		Capture type		Condition			
Species	Scientific name	New	Recapture	Alive	Poor	Dead	Mortality rate
Bushy-tailed woodrat	*Neotoma cinerea*	4	1	5	0	0	0
Douglas squirrel	*Tamiasciurus douglasii*	28	0	12	2	14	0.50
Golden-mantled ground squirrel	*Callospermophilus lateralis*	215	203	410	5	3	0.01
Mountain cottontail	*Sylvilagus nutallii*	1	0	1	0	0	0
Northern flying squirrel	*Glaucomys sabrinus*	23	3	21	1	4	0.15
Siskiyou chipmunk	*Tamias siskiyou*	536	385	834	45	42	0.05
Snowshoe hare	*Lepus americanus*	1	0	1	0	0	0
Yellow-pine chipmunk	*Tamias amoenus*	129	10	121	6	12	0.09
Dark-eyed junco	*Junco hyemalis*	1	0	1	0	0	0
Gray jay	*Perisoreus canadensis*	3	0	2	0	1	0.33

Note: Capture types were new captures of untagged individuals or recaptures of tagged individuals. Animal condition was recorded as alive, poor, or dead, and the mortality rate was calculated as the proportion of all captures that resulted in the death of an animal.

Table 12.4—Number of individual animals (minimum number known alive; MNKA) of each target species that were tagged in each unit in 2017 mammal sampling as part of the Lookout Mountain Thinning and Fuels Reduction Study

	Unit										
Species	**11**	**12**	**14**	**21**	**22**	**23**	**25**	**31**	**32**	**35**	**Total**
Bushy-tailed woodrat	0	3	0	0	0	0	0	0	0	1	4
Golden-mantled ground squirrel	3	0	28	39	36	17	7	33	21	21	205
Northern flying squirrel	8	0	0	0	0	0	7	0	1	5	21
Siskiyou chipmunk	99	41	37	13	6	7	78	6	51	95	433
Yellow-pine chipmunk	1	9	19	8	13	5	5	18	24	5	107
Total	111	53	84	60	55	29	97	57	97	127	770

Note: MNKA is biased by trapping effort.

ranged from 39 in unit 21 to zero in unit 12. The MNKA for yellow-pine chipmunks ranged from 24 individuals in unit 32 to 1 individual in unit 11. We captured northern flying squirrels in four of the units (11, 25, 32, and 35), with a maximum MNKA of seven individuals captured in units 11 and 25. We captured bushy-tailed woodrats in two units—three individuals in unit 12 and one individual in unit 35.

Sample sizes and recapture rates from the fall 2017 LMS trapping data provided good estimates of density based on SECR analysis for Siskiyou chipmunks and golden-mantled ground squirrels (table 12.5). Yellow-pine chipmunks had very low probability of capture ($g0 = 0.01$), resulting in large confidence intervals around the density estimates for this species. The SECR movement parameter, σ, indicated that movement patterns estimated from observed movement between traps were similar across the three species (σ about 25 to 30). Siskiyou chipmunk density ranged from <1 individual/ha (in units 22 and 31) to >10 individuals/ha (in units 11, 12, and 35). Golden-mantled ground squirrel density ranged from <1 individual/ha (in units 11, 12, and 25) to >3.5 individuals/ha (in units 21, 22, and 23). Total captures and recapture rates for northern flying squirrels and bushy-tailed woodrats were not adequate to estimate density based on a stand-alone analysis of the fall 2017 data.

Discussion

This preliminary analysis suggests that densities of the focal species were different in areas that had different silvicultural treatments (table 12.5). For example, Siskiyou chipmunks were at their highest density in Reference units (10 to 13 animals/ha) and lowest density in heavy thin experimental units (1 to 3 animals/ha). Conversely, golden-mantled ground squirrels were at their highest density in heavy thin experimental units (3 to 5 animals/ha) and lowest density in Reference units (<1 animal/ha). Yellow-pine chipmunks were generally at their lowest density in Reference units (<2 animals/ha); however, patterns for yellow-pine chipmunk density in moderate and heavy thin units are unclear because of low capture rates.

We detected northern flying squirrels in three Reference units (with 8 or 9 nights of trapping effort) and one of the moderate thin units that was adjacent to a Reference unit. Flying squirrels were not detected in any of the heavy thin units. We captured bushy-tailed woodrats in two Reference units, including unit 12, which only had 2 nights of trap effort, suggesting that woodrat density may be relatively high in that unit. Bushy-tailed woodrats were not detected in any of the thinned units.

Timing of fieldwork for small mammal trapping in the eastern Cascades presents substantial challenges, particularly at higher elevations where weather can change quickly. The snowstorm that we experienced during our first scheduled week of trapping in 2017 affected our data collection efforts and contributed to larger capture mortality rates than we experienced during other sampling efforts. However, the previous week (4 through 8 September) was very hot, and nearby wildfires presented a different suite of safety concerns for the field crew and challenges for trapping. In the end, we were fortunate to have planned for 3 weeks of trapping rather than the 2-week trapping sessions that we have conducted in some of our other study areas. The extended trapping period allowed us to make up for time lost to the snowstorm.

Comparing the results of this round of sampling to results from our other study sites is beyond the scope of this report. We have not yet conducted SECR analysis for chipmunks and ground squirrels from other sampling efforts. Comparison of these density estimates to pretreatment sampling or estimates from other study sites will require further analysis. We expect that our ability to estimate animal density will improve as we combine analyses across study sites and leverage the additional

Table 12.5—Unit-level density estimates (Est, individuals/hectare), standard errors (SE), and lower and upper 95-percent confidence levels (LCL and UCL, respectively) of the density estimates for Siskiyou chipmunks, golden-mantled ground squirrels, and yellow-pine chipmunks from spatially explicit capture-recapture (SECR) analysis as part of the Lookout Mountain Thinning and Fuels Reduction Study

	Siskiyou chipmunk				Golden-mantled ground squirrel				Yellow-pine chipmunk			
Unit	Est.	SE	LCL	UCL	Est.	SE	LCL	UCL	Est.	SE	LCL	UCL
11	13.02	1.26	10.77	15.74	0.08	0.10	0.01	0.56	0.73	0.66	0.16	3.33
12	11.17	1.81	8.16	15.30	0.15	0.19	0.02	1.04	11.85	5.99	4.66	30.18
14	5.54	0.89	4.05	7.57	2.65	0.50	1.84	3.81	10.46	4.34	4.79	22.86
21	2.02	0.58	1.17	3.50	3.89	0.64	2.83	5.35	5.43	2.79	2.10	14.03
22	0.96	0.41	0.43	2.14	3.86	0.64	2.80	5.33	8.83	3.99	3.79	20.57
23	3.39	1.34	1.60	7.15	4.44	1.10	2.76	7.17	12.97	7.96	4.28	39.30
25	9.80	1.06	7.93	12.11	0.55	0.21	0.26	1.15	1.83	1.08	0.63	5.33
31	0.82	0.35	0.37	1.84	3.23	0.55	2.32	4.51	10.08	4.22	4.58	22.17
32	5.99	0.81	4.61	7.79	1.66	0.36	1.09	2.52	7.80	2.97	3.79	16.05
35	10.34	1.06	8.46	12.63	1.72	0.36	1.14	2.59	1.65	0.96	0.57	4.78
g0	0.08	0.01	0.07	0.10	0.12	0.01	0.10	0.14	0.01	0.01	0.00	0.03
sigma	25.69	0.78	24.22	27.26	31.56	1.34	29.04	34.30	30.20	5.98	20.56	44.37

Note: Estimates for back-transformed SECR model parameters λ (g0) and σ (sigma), detection model components that address probability of capture relative to an animal's center of activity and observed movements among traps, are also shown.

data to provide better estimates of recapture probability and movement distance, particularly for northern flying squirrels and bushy-tailed woodrats.

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Chapter 13: Landscape Understory Microclimate Responses to Overstory Thinning and Understory Fuel Reduction Treatments

Paul D. Anderson and Jonathan Burnett[1]

Introduction

A premise of the Lookout Mountain Thinning and Fuels Reduction Study (hereafter LMS) is that the study landscape is characterized by significant climatic gradients that influence ecosystem composition and dynamics. The presence of climatic gradients arises from differences in elevation and aspect associated with Lookout Mountain, a volcanic cinder cone rising about 540 m above the surrounding terrain. Climatic gradients are important drivers of forest composition and structure at various scales (Sarr et al. 2005). Given the relatively consistent soils and subsurface geology of the Lookout Mountain unit, climate variation interacting with disturbance may be the predominant driver structuring forest plant communities. The experimental design of the LMS reflects the climatic gradients at Lookout Mountain; blocks within experimental units were delineated by elevation and aspect to account for presumed spatial and temporal variation in thermal and precipitation regimes.

The array of plant associations (Volland 1988) mapped at Lookout Mountain reflect a coarse elevation stratification of ponderosa pine (*Pinus ponderosa*) and mixed-conifer series and finer scale stratification of associations driven by local topography, soils, and moisture regime. Conceptually, plant associations are the plant communities that would establish if successional sequences progressed under existing environmental conditions without human intervention. The long-term stability of climatic conditions is unlikely, and human management intervention is increasingly viewed as important to the development of fire resistance in these dry, fire-prone forests in the absence of frequent, low-severity fires characteristic of the pre-Euro-American fire regime. Therefore, it is of interest to better understand how current and future environments interact with silvicultural restoration treatments applied at a landscape scale to shape forest composition and structure. This understanding will require characterization of dynamic climate and site conditions at multiple spatial and temporal scales.

The LMS focuses on overstory thinning treatments that differ in the proportion of retained overstory trees as well as the removal of small clusters of trees to create discrete canopy gaps. Furthermore, all treatments included the removal of understory vegetation. Collectively, these changes to forest structure alter local

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environmental conditions (microclimate and microsite) directly by increasing transmittance of radiation and air flow and indirectly by altering air temperature, humidity, soil temperature, and soil moisture (Anderson et al. 2007, Chen and Franklin 1997, Parker et al. 2004). Where studied previously in mesic forests west of the Cascade crest, partial overstory removal has generated transitory changes in microclimate and microsite (Anderson et al. 2007, Chen et al. 1996). Gap creation in mesic forests has also been associated with fine-scale microclimate and microsite variations with distance from the gap edge—both toward the center of the gap and into the surrounding forest—that influence vegetation composition and development (Gray et al. 2002, Heithecker and Halpern 2007). For the drier east-side forests represented by the LMS, it is of interest to determine the nature and extent of microclimate variations given the presumed slower canopy growth response and compounding effect of understory treatment by fire or cutting.

Climate surface models (e.g., PRISM: Daley et al. 2008, Daymet: Thornton 2017, ClimateNA: Wang et al. 2016) have been developed to provide spatially explicit representation of mesoclimate variation at the Earth's surface by interpolation of data generated from regional, broadscale weather stations. However, such models have relatively coarse resolution (e.g., 800-m grid, PRISM) or generate scale-free estimates as a secondary derivative of the 800-m gridded estimates (e.g., ClimateNA). Furthermore, estimated surfaces do not account for the influence of forest cover and therefore do not reflect finer scale variations or near-surface understory microclimate conditions. In this study, we used a set of air temperature sensors dispersed across the LMS area to generate data from which to interpolate understory microclimate at 50-m resolution. We explored the resultant microclimate surface maps to assess the effects of the various thinning and fuels reduction treatments on the understory environment.

Several near-term and longer term research objectives were addressed. For the near term, we sought to better understand how microclimate varies spatially and temporally within and among treated stands. We also wanted to know how long alterations persist in microenvironments associated with thinning and gap creation given subsequent canopy and stand growth dynamics. Over longer time periods, we hope to learn if the restoration treatment effects on vegetation, insects, and other organisms vary predictably with the study area's weather or climate.

Methods

We established microclimate monitoring stations at 62 locations throughout the 1190-ha experimental landscape (fig. 13.1). In each of the 19 experimental units, we placed two stations in conjunction with permanent vegetation sampling plots and one station in a random location. In addition, we placed five stations in additional locations associated with the original pre-northern spotted owl (*Strix occidentalis caurina*) (NSO) study layout that were retained when sampling unit boundaries were moved to accommodate a pair of NSO. Each station consisted of a shaded

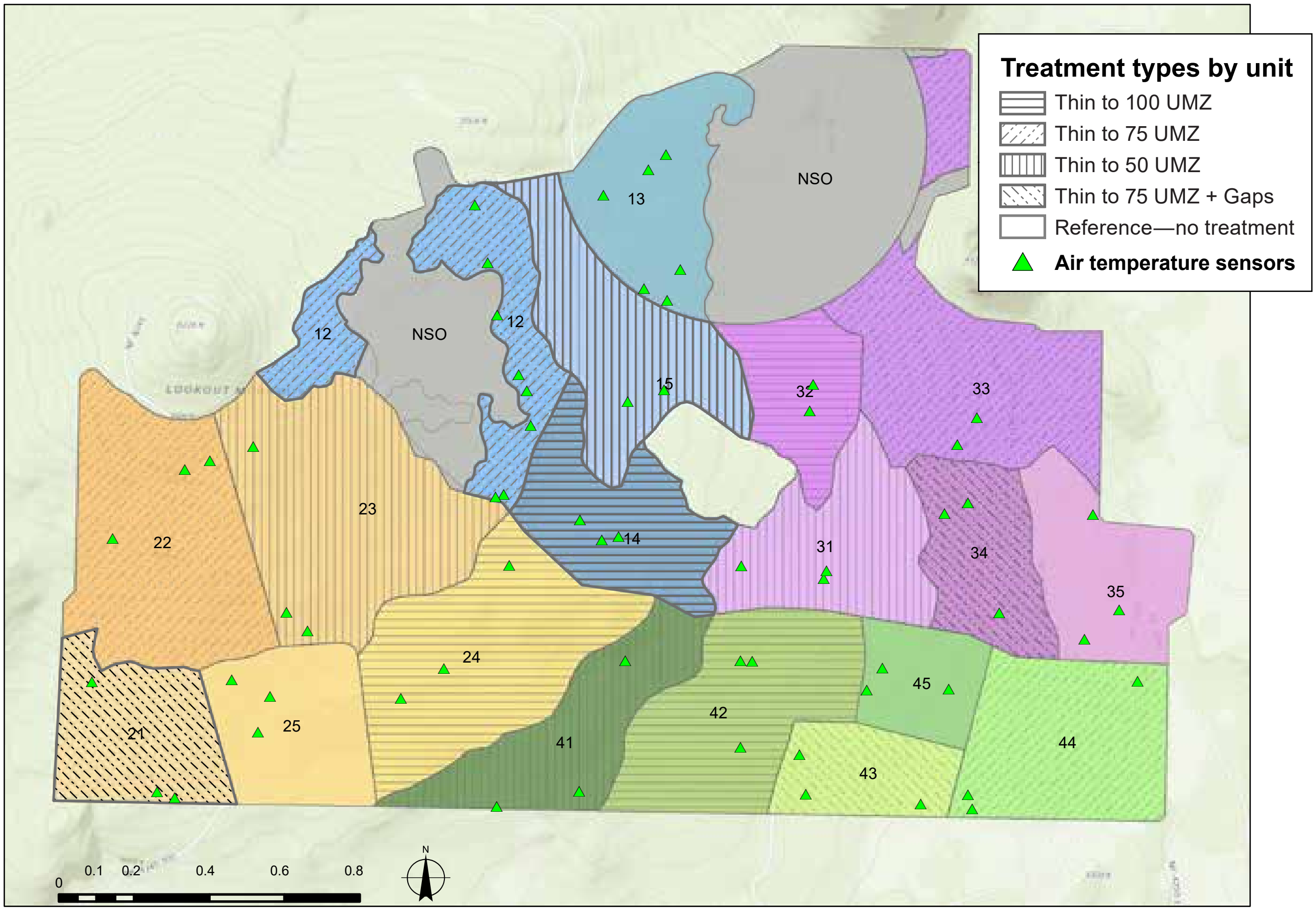

Figure 13.1—Lookout Mountain Thinning and Fuels Reduction Study landscape illustrating spatial relationships among the experimental blocks (delineated by color group), experimental units (delineated by polygon and color), and climate stations. UMZ = upper management zone. NSO = northern spotted owl.

temperature sensor suspended 1 m above the soil surface and a similar temperature sensor buried in the soil at a depth of 15 cm.

Sensors were either single-channel temperature loggers or dual-channel temperature/humidity loggers (iButton DS1922L, iButton DS1923, respectively; Maxim Integrated, Inc., San Jose, California). Deployed to record with 8-bit resolution, the sensors had 0.5 °C and 0.6 percent accuracy for temperature and relative humidity, respectively. Air temperature was recorded at 30-minute intervals, and soil temperature was recorded every 60 minutes. Air temperature loggers recording twice per hour could store about 3 months of data before downloading; we rotated out sets of sensors for downloading and calibration four times per year. There were too few dual-channel temperature/relative humidity loggers available for the full study design, which resulted in an incomplete representation of relative humidity data. We therefore did not analyze relative humidity data.

We screened sensor data for errors, which included removing observations during periods when any sensor was covered with snow, knocked from its post, or missing its cover. After screening for errors, we averaged within-day temperature data into discrete hourly intervals. Subsequently, we extracted minimum and maximum daily temperatures for each day and station. From these daily metrics, monthly mean minimum (Tmin) and maximum (Tmax) temperatures were estimated for each station and used in the modeling efforts described below. We developed microclimate surface maps, geospatial representations of Tmax and Tmin at 1 m above the ground surface through interpolation of 2013 through 2016 sensor data from 54 through 62 locations. We interpolated data using generalized additive models (GAM) with elevation, heat load index (McCune 2007), and normalized difference vegetation index (NDVI) as location covariates (Burnett and Anderson 2019). Modeling was performed in R (R Core Team 2017).

Results

Modeled point estimates for a 50-m grid across the entire LMS resulted in interpolated microclimate surface maps that show spatially and temporally varying patterns of daily mean maximum (fig. 13.2) and daily mean minimum (fig. 13.3) air temperatures. Models for all years revealed that air temperature trends were dependent on complex interactions of forest cover, site physiography, and time (Burnett and Anderson 2019), and the trends produced interpolations closer to observed conditions than readily available, coarse-scale models. Furthermore, the GAM interpolations were characterized by less bias and error than interpolations developed from either PRISM or Daymet interpolations (table 13.1), both of which produced estimates that were generally cooler than actual observations across all 4 years.

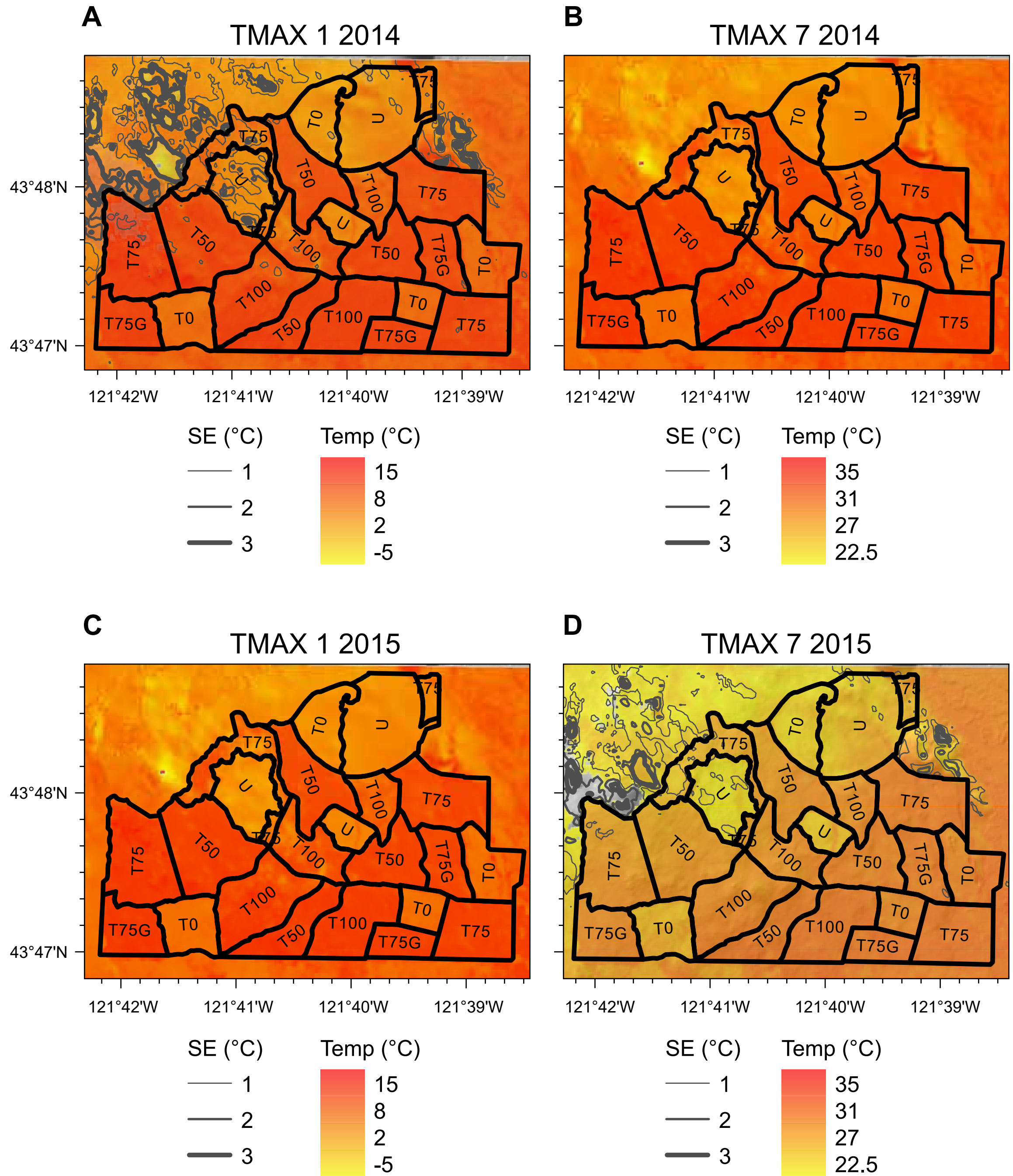

Figure 13.2—Maximum daily temperature (Tmax) interpolation grids by month for (A) January and (B) July 2014 and the same months for 2015 (C and D) and associated standard error (SE). Gray areas had temperatures above or below temperatures captured by the established scale; these areas were effectively removed to enhance visual interpretation. Experimental units of the Lookout Mountain Thinning and Fuels Reduction Study are outlined and annotated by the specified treatment. U = Not included in LMS; T0 = Reference; T50 = 50 UMZ; T100 = 100 UMZ; T75 = 75 UMZ; and T75G = 75 UMZ + Gaps.

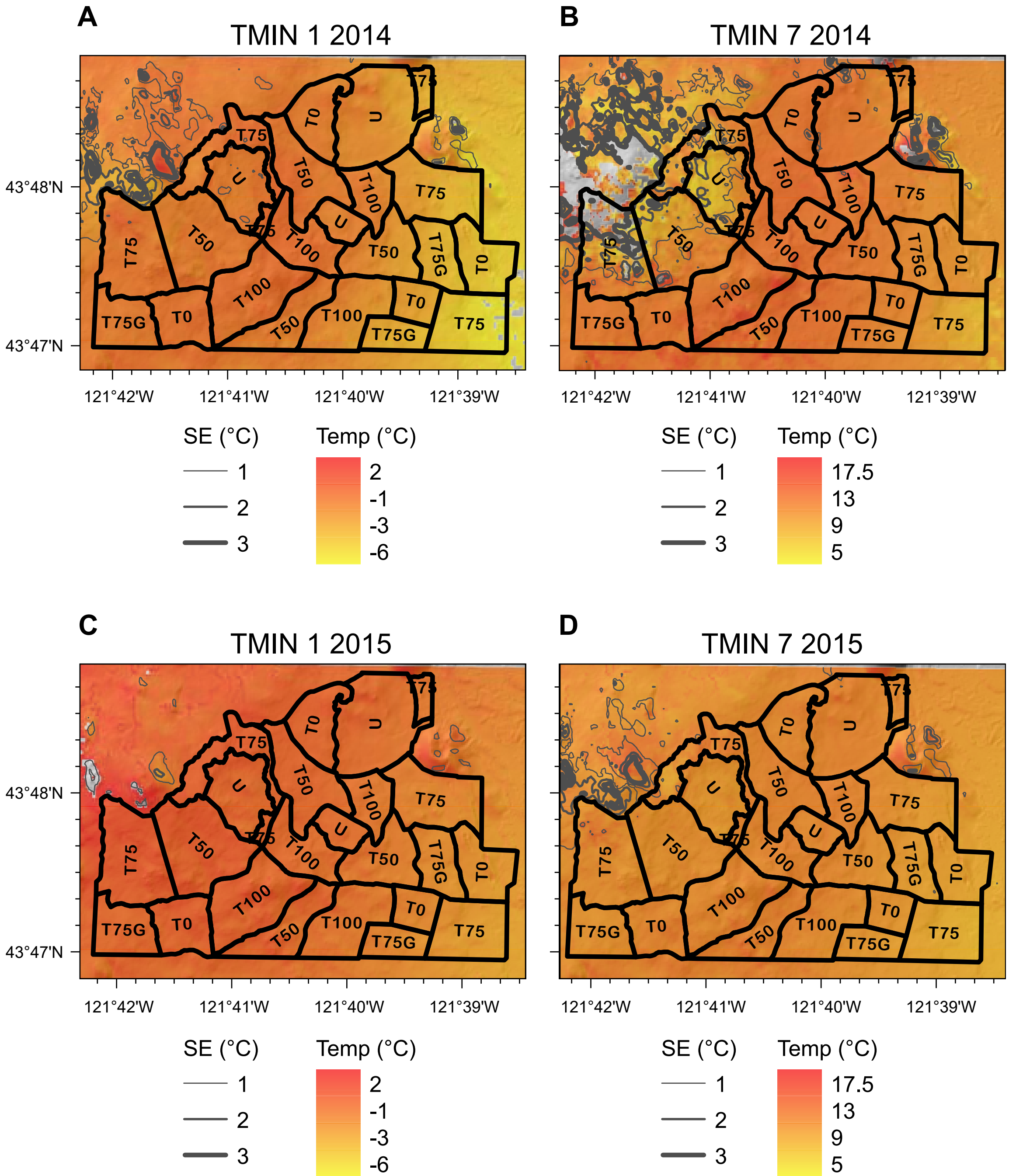

Figure 13.3—Minimum daily temperature (Tmin) interpolation grids by month for (A) January and (B) July 2014 and the same months for 2015 (C and D). Gray areas had temperatures above or below temperatures captured by the established scale; these areas were effectively removed to enhance visual interpretation. Experimental units are outlined and annotated by the specified treatment. U = Not included in LMS; T0 = Reference; T50 = 50 UMZ; T100 = 100 UMZ; T75 = 75 UMZ; and T75G = 75 UMZ + Gaps.

Table 13.1—Precision (root mean square error, RMSE) and accuracy (bias) of annual gridded estimates of daily mean maximum (Tmax) and daily mean minimum (Tmin) air temperatures on the Lookout Mountain Thinning and Fuels Reduction Study as estimated by the generalized additive models (GAM) of local observations relative to estimates generated by Daymet and PRISM models

		RMSE			Bias		
Variable	Year	GAM	Daymet	PRISM	GAM	Daymet	PRISM
		Degrees Celsius					
Tmax	2013	1.649	2.801	3.316	0.029	−1.725	−2.367
	2014	1.404	2.394	2.956	0.044	−1.518	−2.143
	2015	1.609	2.530	3.081	0.002	−1.632	−2.194
	2016	1.280	2.871	3.142	0.004	−2.071	−2.244
Tmin	2013	0.541	1.903	1.645	0.018	−1.494	−1.324
	2014	0.586	1.454	1.276	0.023	−1.079	−0.920
	2015	0.578	1.546	1.380	−0.011	−1.178	−0.973
	2016	0.576	1.304	1.578	0.009	−0.994	−1.232

Model performance varied within months and between seasons. Examination of the root mean square error (RMSE) of model interpolations at the month scale (fig. 13.3) suggests that model estimate error varied more within months than between months. Tmin generally had less absolute error as well as less variation in error (generally <0.5 °C) than Tmax. Interpolation error in Tmax tended to be greater in winter-spring months. In addition to high variation in microclimate, contributing factors may include sensor errors and some months having more operational climate stations than others.

We analyzed subcanopy air temperature interpolation maps for trends in the context of differing canopy structures arising from five fuels mitigation treatments. In both winter and summer months, all treatments resulted in elevated daily mean maximum air temperature (Tmax) relative to Reference units (figs. 13.4 and 13.5). The variation in Tmax was large within treatments; thus, there was no clear indication of Tmax differences among harvest intensities. Tmax differences associated with harvests tended to be 1 to 2 °C greater in winter than in summer.

Discussion

Our research demonstrates the ability to develop higher resolution understory microclimate surface maps of value to many aspects of ecosystem monitoring in forested environments. The primary advantage of locally derived GAM models over readily available meteorological downscaling models is they account for the influences of forest canopy, in contrast to canopy surface or bare-ground estimates derived by tools such as PRISM and Daymet.

As expected, the microclimate surface models were effective in reflecting the spatial variations associated with the topographic and vegetation structure covariates. Heat load index captured the influence of slope and aspect as coarsely

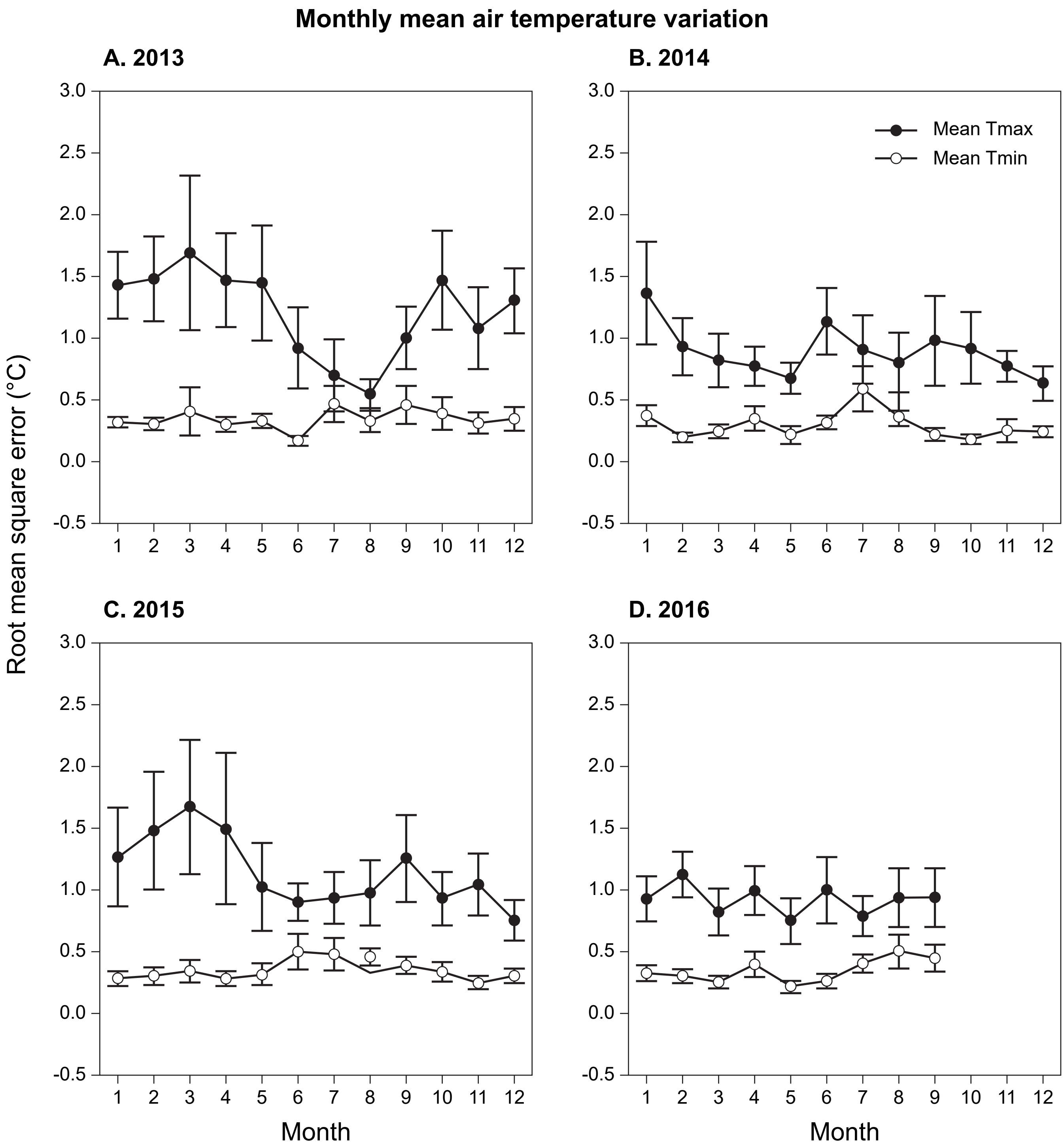

Figure 13.4—Root mean square error for model of daily mean maximum (Tmax) and daily mean minimum (Tmin) air temperatures by month in (A) 2013, (B) 2014, (C) 2015, and (D) 2016. Error bars are 95-percent confidence intervals estimated using nonparametric bootstrapping.

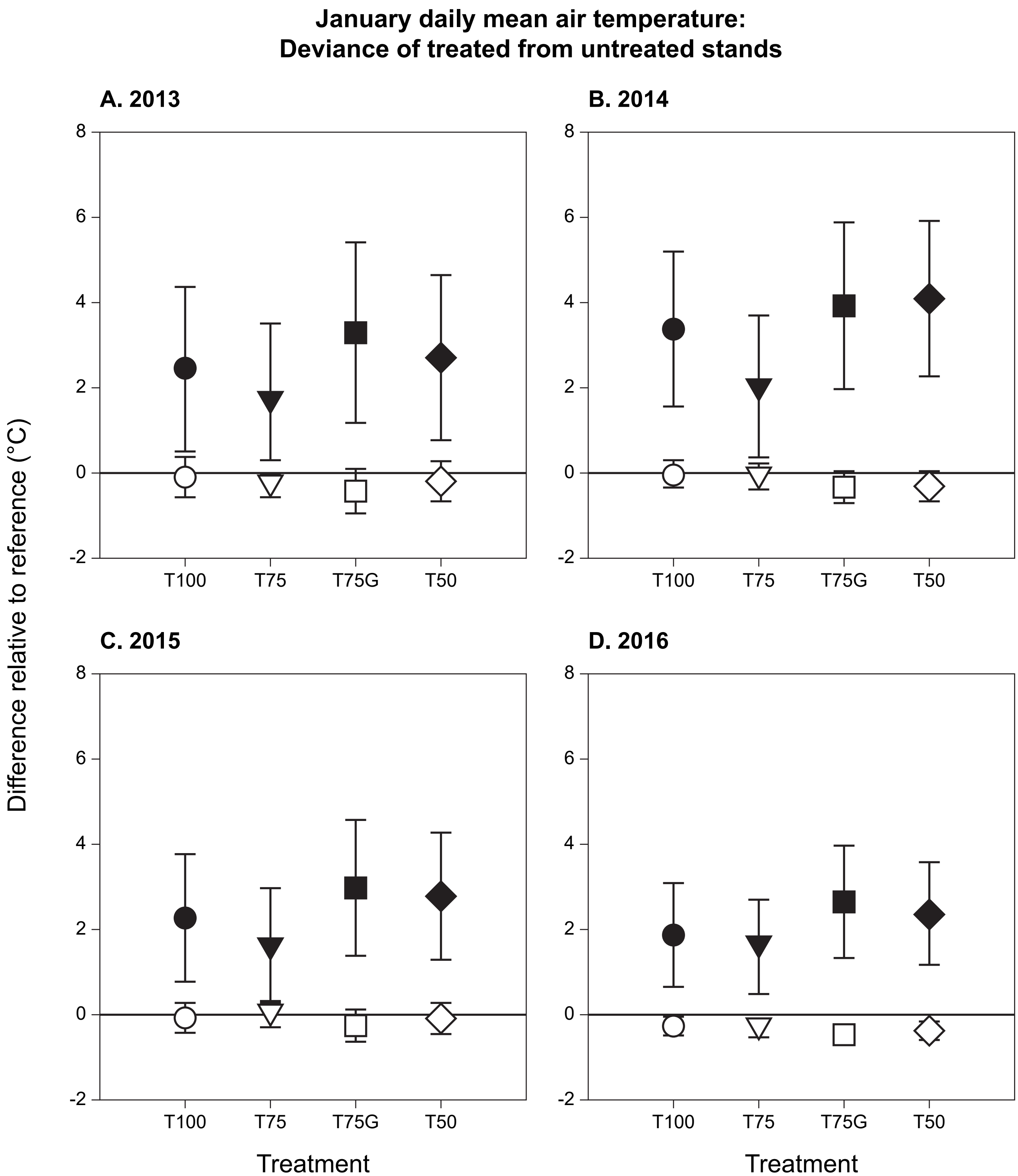

Figure 13.5—Daily mean maximum (Tmax [closed symbols]) and minimum (Tmin [open symbols]) air temperatures in January by treatment as the deviation from Reference stands on the Lookout Mountain Thinning and Fuels Reduction Study in (A) 2013, (B) 2014, (C) 2015, and (D) 2016. T100 = 100 UMZ; T75 = 75 UMZ; T75G = 75 UMZ + Gaps; and T50 = 50 UMZ.

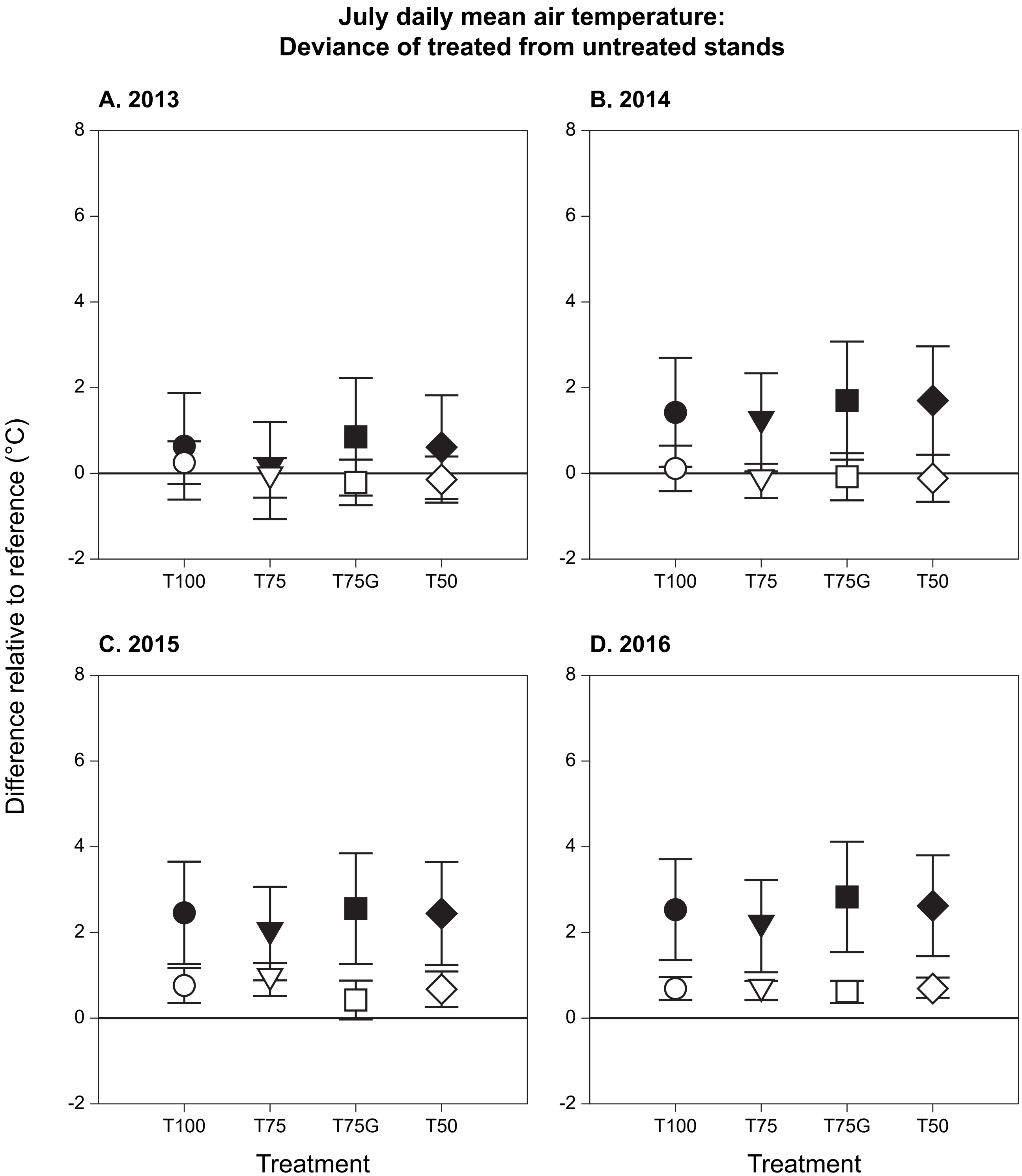

Figure 13.6—Daily mean maximum (Tmax [closed symbols]) and minimum (Tmin [open symbols]) air temperatures in July by treatment as the deviation from Reference stands for 2013 through 2016 on the Lookout Mountain Thinning and Fuels Reduction Study in (A) 2013, (B) 2014, (C) 2015, and (D) 2016. T100 = 100 UMZ; T75 = 75 UMZ; T75G = 75 UMZ + Gaps; and T50 = 50 UMZ.

influenced by elevation. NDVI effectively provided a fine-scale index of vegetation cover, both overstory and understory where visible from above, which is particularly important in estimating Tmax.

In application, the dispersed microclimate sensors are relatively easy to maintain although they do require personnel on the ground with regularity, and the sensor installations are subject to physical disturbance by animals or other factors. In this instance, we used sensors with relatively low data-storage capacity—about 4,900 recorded observations, which limited the deployment period and determined the degree of maintenance effort. The relatively low-cost sensors were affordable in the required abundance but lacked the automated data telemetry capabilities of more expensive temperature loggers.

Our sensor deployment strategy addressed two goals: (1) to provide measures (n = 2) of the microclimates tied directly to the permanent gridded vegetation monitoring plots established within each experimental unit, and (2) to provide broad representation of the microclimates of entire experimental units. Experimental units varied in size and therefore were not consistently represented by the gridded vegetation sampling area. To address these two goals, we distributed the 62 sensors across the entire LMS area, with greater density in the 6.025-ha vegetation sampling areas embedded within each experimental unit. We have not explored the sensitivity of modeling to alternative sensor densities or alternative deployment strategies that address modeling sensitivity to the influences of specific covariates. However, our results indicate that simple analysis of the point estimates derived from the limited sensor deployment would likely have resulted in nontrivial differences in interpretation of treatment influence, relative to what we inferred from the gridded estimates modeled over the entire experimental units.

Given the anticipated dynamics of understory and overstory structure and composition, there is continued interest in determining the sensitivity of the modeling approach to representation and detection of longer term temporal and spatial patterns of fine-scale microclimate variation. For instance, there are anticipated near-term influences in rapidly developing shrub and herbaceous understory vegetation that may affect near-surface temperatures through direct shading and the mitigation of bare-soil surface heat reflection. Longer term overstory canopy densification owing to crown expansion is likely to further mediate understory temperatures through decreasing incoming solar radiation and insolation of nighttime heat loss. These changes in vegetation density, stature, and composition will interact with and reflect changes in mesoclimate for central Oregon including a general trend for increasing annual temperatures and potentially more frequent or persistent soil moisture deficits (Chmura et al. 2011).

A primary motivator for the suite of thinning and understory fuels reduction treatments explored in this study is the hypothesis that exclusion of fire for more than a century in these older, predominantly ponderosa pine and mixed-conifer forests has resulted in accumulation of fuels that increase the probability of high-intensity burning if ignited. There is an emerging body of evidence suggesting that intact, older-aged forests tend to burn at lower intensity—presumably in part owing to lower temperatures and higher humidity in the understory of closed, high-canopy forests (Frey et al. 2016, Lesmeister et al. 2019). In contrast, other studies suggest that in the drier, fire-prone forests, thinned stands have been effective in reducing fire intensity (e.g., Prichard et al. 2010). The initial posttreatment increase in near-surface air temperatures observed in this study might suggest a higher probability of understory fuel desiccation and flammability but less likelihood for fire to be carried into and through the dispersed forest canopy. However, other studies have indicated that the impacts of thinning on microclimate and associated windspeed and fuels moisture are not significantly impactful unless canopy cover is decreased to very low levels such as in a group selection opening (e.g., Bigelow and North 2012). Scenario modeling of fire behavior has demonstrated a complex relationship among influences of overstory thinning, microenvironment, fire behavior, and fire intensity and suggests that caution is warranted in making generalizations (Banerjee 2020).

Future Work

Microclimate data were collected continuously from 2011 through autumn 2019. Analyses to date have focused on the initial development of microclimate geospatial surface mapping for 2013 through 2016, capturing the initial posttreatment conditions (Burnett and Anderson 2019). The first priority for future work is performing additional analyses to characterize the full temporal dynamics of the microclimate surfaces through 2019, 5 years posttreatment. A second priority is to assess the geospatial deployment of microclimate monitoring locations to best capture data that can be most effectively interpolated to maximize microclimate surface precision and to minimize error for the long-term establishment of a microclimate assessment network.

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Part IV: Moving Forward

Chapter 14: Summary and Future Work

Lorelle M. Sherman and Paul D. Anderson[1]

Summary of Research Objectives

This establishment report was designed for use by those interested in the implementation of the Lookout Mountain Thinning and Fuels Reduction Study (hereafter LMS), the early observations of the core vegetation study, and the progress in addressing ancillary research questions posed by additional principal investigators. The establishment report focuses on near-term changes in forest vegetation as measured by pre- and posttreatment data collected from 2011 through 2016. Because longer term forest responses are yet to be realized, most of the intended core vegetation study objectives have not been met at this time. A discussion of progress toward meeting the research objectives follows.

Objective 1

Objective 1 is to determine near- and longer term responses of forest vegetation dynamics (structural changes, species' range shifts, density-dependent competition, and disturbance) to thinning and fuel reduction in ponderosa pine (*Pinus ponderosa*) stands.

What is apparent from the data is that the vegetation structure, both overstory and understory, within experimental units was substantially altered by harvest and subsequent mastication and burning. The suite of prescribed silvicultural treatments reduced the stand density index (SDI) and basal area (BA) in all treated experimental units (excluding Reference units). Thinning and fuel reduction treatments changed the composition and structure of understory vegetation, but the change varied by plant growth form and treatment type. Shrub and subshrub canopy cover decreased in all treatment types and graminoid canopy cover increased in all treatment types, on average. Tree seedling and herb/forb canopy cover response varied by treatment type. Variability in posttreatment density metrics was introduced by pockets of higher-than-intended burn intensity and microsite features, such as rocky outcroppings and slope and aspect. Although this report outlines near-term observations, formal conclusions will require statistical analyses. Longer term responses will be revealed by analysis of vegetation data in future years of data collection.

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Objective 2

Objective 2 is to evaluate the effects of thinning and fuel reduction treatments on long-term susceptibility of forest vegetation to fire, insects, wind, and environmental change (reduced water availability, increased heat load, "hotter droughts" sensu Griffin et al. [2014]), and other forest disturbances.

The goal of this objective is to refine management options for restoring resistance and resilience of dry, east-side conifer forests to stressors and disturbances. Determining the resistance and resilience of forest conditions created by thinning and fuel reduction treatments to bark beetles will require intermittent data collection over the long term. However, in the near term, the removal of fire-intolerant species and stand-density reduction may have reduced potential crown spread during fire events, as well as tree-to-tree competition for resources, resulting in increased forest stand resistance and resilience. Inducing bark beetle infestations in residual ponderosa pine following prescribed fire was a concern; however, work done as part of chapter 9 indicated that treatment-induced bark beetle infestations were not a major issue in the LMS. The response of bark beetles to thinning and fuel reduction treatments did not significantly differ among treatments, and tree mortality from bark beetle infestations was very low across all experimental units. A lag in tree mortality resulting from bark beetles may occur in coming years but likely at very low rates.

The alignment of the LMS's silvicultural prescriptions with the forest conditions of the Lookout Mountain unit pre-Euro-American settlement at the patch scale warrants a discussion. The prescriptions of this study were designed to decrease stand density, eliminate fire-intolerant tree species, and restore stands to fire-resilient, ponderosa pine-dominated forests. According to dendrochronological studies for central Oregon (Merschel et al. 2014, 2018), large ponderosa pine-dominated stands that have persisted since pre-Euro-American settlement increased in stand density after the initiation of fire suppression and exclusion in the late 19th century. These stands were often at lower elevations in hot, dry microclimates with southwest-facing slopes compared with other stands in the pumice plateau and pumice plateau basins east of the Cascade crest, and shade-tolerant species like grand fir (*Abies grandis*) were absent. At intermediate elevations with higher annual precipitation and lower temperatures in the pumice plateau and pumice plateau basins, ponderosa pine dominated the forest overstory while successional shade-tolerant species grew in the understory, especially on northwest-facing slopes. Frequent fire historically kept this forest type in a ponderosa pine-dominated state, though grand fir was sometimes a codominant species in cool, moist microclimates regardless of fire frequency (Merschel et al. 2018). Merschel et al. (2018) identified a historical forest type dominated by shade-tolerant trees, such as grand fir, that has persisted in this condition at the patch scale to present day. These cool, moist, and often northeast-facing sites have likely not diverged drastically from their historical state.

Records show that the transition from shade-intolerant tree species to shade-tolerant tree species began after broadscale fire suppression and exclusion (Merschel 2014, 2018). High-elevation sites with northeast-facing slopes (e.g., experimental unit 12) may not be divergent from their historical conditions and may continue to support a mixed-species conifer forest. However, high-elevation sites with southwest-facing slopes may be managed to return to a ponderosa pine-dominated state through the LMS treatments. Some studies predict that ponderosa pine may experience increased physiological stress and ultimately replacement by other tree species (e.g., juniper [*Juniperus occidentalis*]) in hot, dry environments associated with increased water deficit and drought in the future (Coops et al. 2005, Minott and Kolb 2020). If the prescribed treatments in the LMS increase resilience of ponderosa pine forests at sites presently defined by cool, moist microclimates at intermediate and higher elevations, these sites may become important refugia for ponderosa pine in the context of changing environmental conditions.

Assessment of Progress Toward Research Objectives

To assess progress toward the LMS research objectives, we summarize early observations and discuss how well the implementation of treatments achieved the intended effects. Our discussion is framed by three research questions that address forest structure and composition as well as effects of thinning and fuel reduction treatments in the overstory and understory.

Question 1

Did the thinning and fuel reduction treatments accomplish the desired forest structure and composition?

Overstory structure—

The response of overstory tree SDI was, in part, driven by the plant associations used to determine overstory density targets. In plant associations CW-S1-15 and CP-S3-12 (see table 8.1), the application of both thinning and underburning may have resulted in densities lower than prescribed by the target density calculated as a percentage of the upper management zone (UMZ) (see chapter 3 for treatment descriptions). In CW-S1-13, the same combination of thinning and underburning treatments resulted in SDI values that were within 3 percent of the target SDI of the UMZ, except in 100 UMZ and Reference units, where densities were higher than prescribed.

Generally, the BA resulting from treatments was larger than the target BA, except in experimental units 33 (75 UMZ) and 14 (100 UMZ), where burn intensity was higher than intended, and in unit 34 assigned to the 75 UMZ + Gaps treatment.

The application of thinning and fuel reduction treatments resulted in substantial removal of large-diameter trees, predominantly fire-tolerant pine species, to achieve density targets. Similar applications of thinning and fuel reduction treatments on forest lands outside of experimental forest designations would likely be subject to policies limiting the removal of large-diameter pine.

Overstory composition—

All treatments eliminated tree species considered to have low fire tolerance in the dry, east-side forest community and reduced the number of moderately fire-tolerant tree species in the overstory. Small numbers (≤1 tree/ha) of grand fir and mountain hemlock (*Tsuga mertensiana*) were retained in the least intensive thinning treatments to reach density targets. In the most intensive treatment, only ponderosa pine was retained in the overstory.

Block 1 had the highest species diversity of conifers before treatment implementation, likely because it contained cooler and moister microclimate conditions than other blocks, which favored a mix of conifer species. The abundance of conifers in block 1 led to the highest retention rate of species and number of individual trees other than ponderosa pine, including 242 grand fir. Block 1 had the highest species diversity of all the blocks posttreatment as well.

In the specific case of the LMS, the removal of large-diameter, fire-tolerant ponderosa pine was a reflection of the landscape being dominated by ponderosa pine before treatment implementation.

Understory structure and composition—

The decreased average canopy cover of shrubs and subshrubs in all treatments aligns with the desired reduction in the abundance and vertical distribution of flammable understory fuels. Mastication of vegetation and slash disposal during harvest operations were drivers in achieving the desired understory structure. The emergence of several forb/herb species, including nonnatives, and the increase of graminoids in all treatments may also be attributed to harvest operation disturbance.

Question 2

What was the variability in thinning and fuel reduction treatment effects within and among blocks and within and among treatments, and were there unintended effects of underburning and thinning (e.g., fire hotspots and increased upper canopy load)?

Overstory—

Multiple sources of variability likely played a role in meeting target overstory tree density goals. Large variability among sample plots within a treatment may be attributed to a combination of factors, including differences in preexisting spatial distribution of overstory trees, overlapping of sample plots with gaps created as part of 75 UMZ + Gaps treatments and higher-than-intended burn intensity in some areas. Prescribed underburning or subsequent insect attack on fire-injured trees may be linked to greater-than-anticipated localized mortality of overstory trees.

Differences in environmental gradients, including elevation, slope, and aspect, may explain observed variation in treatment effects on overstory composition. For example, the diverse mix of overstory conifer species in block 1 likely contributed to a diverse set of posttreatment conditions.

Understory—
Experimental unit delineation for the LMS was informed by preexisting plant association maps. Simpson (2007) observed that Volland's (1988) plant associations are not well mapped in some areas of dry, east-side mixed-conifer forest. Apparent inconsistencies in the observed understory species composition with designated plant associations suggests that the original maps may not have been delineated at a resolution fine enough to reflect understory vegetation distribution at the experimental unit scale.

There was uncertainty in the extent to which mastication was a source of variability. An unknown amount of understory vegetation was damaged by equipment and harvesting. Additionally, microsite conditions, including steep slopes and rocky outcroppings, may have limited access for machinery and technicians in some areas.

Question 3

In what ways did treatments affect understory vegetation and coarse downed wood (CDW) characteristics?

Based on preliminary data, there was an apparent reduction in most species of understory tree seedlings, shrubs, and subshrubs in all treatment types. Evaluating treatment effects will require statistical analyses. The response of forb/herb species appears variable by treatment type. This is expected given the diverse suite of regeneration strategies and light and germination requirements within the forb growth form. At least two species of weedy forbs emerged in several experimental units posttreatment.

Differences in structural and environmental gradients among blocks likely influenced posttreatment log volume and percentage log cover of CDW. Treatment influences were variable for both metrics.

Planned Future Maintenance

Overstory

Future overstory harvest treatments will be essential to realign the stand densities to target SDI values as established in this plan for experimental units with 100 UMZ, 75 UMZ, and 50 UMZ treatments. Additional treatments will need to occur to continue the trajectory of stand development into a multicohort structure as initiated in the 75 UMZ + Gaps treatment. The timespan predicted for when additional treatments will be needed is 40 to 50 years postimplementation.

The timing of future thinning treatments will depend on the rate of overstory canopy closure, understory development, and connectivity among the understory and overstory. Rapid canopy closure or accumulation of fire-intolerant species or the establishment of ladder fuels may warrant more frequent treatments. Treatment of understory vegetation through mechanical measures or by prescribed fire may be considered independently of future thinning operations given understory treatments can be implemented with limited collateral impacts to overstory density and composition.

The impacts of future overstory treatments may conflict with the maintenance of northern spotted owl (*Strix occidentalis caurina*) (NSO) habitat structure. The NSO favors closed-canopy, mixed-conifer forests, which are inherently less fire resilient than open-canopy forests dominated by ponderosa pine and other fire-tolerant species. Foraging habitat encompasses a broad range of stand conditions, from high-quality, closed-canopy nesting habitat to more open conifer forest. Therefore, foraging habitat structure and prey abundance may be maintained under a broad range of future overstory and understory treatment prescriptions.

Understory

Additional future mastication or underburning treatments will be needed to reduce surface and ladder fuels. The initial LMS plan suggests that this occurs at 20-year intervals. Because of the rapid reestablishment of understory fuels in some units, additional understory treatments will likely be needed at a shorter return interval. A preburn mastication treatment will be applied where understory fuels have accumulated to unsafe levels for prescribed underburning.

The length of time between future understory maintenance treatments will depend on the rate of understory reestablishment and fuels loading as well as logistical constraints. The seasonality of future underburning will depend on the potential burn intensity resulting from changing environmental factors in combination with fuel load. Broadcast burns can also be planned and scheduled according to desired retention rates for CDW and creating or improving wildlife habitat.

The LMS was implemented over multiple years and with several field crews. Differences in annual temperatures, precipitation, and other environmental variables among years inherently introduced variability into the understory vegetation data. Differences among observers can be a major source of variability in long-term vegetation studies, showing up as species misidentification and pseudoturnover in understory species composition (Morrison 2016). Sampling plots will be rechecked in 2021 for these types of errors.

Recommendations for Future Work

Several ancillary studies are embedded within the broader context of the LMS and demonstrate how principal investigators have already used the treatment framework to address additional issues and research questions relating to forest management. The LMS (along with other previous work at the Pringle Falls Experimental Forest) serves as an excellent foundation for many innovative studies as new management concerns arise with shifting values and objectives in applied forestry, social perceptions, and regulatory requirements. Principal investigators with research questions that fit within the scope of the LMS treatments prescribed as part of the core vegetation study are encouraged to use this long-term, landscape-scale experimental design.

The ancillary studies discussed in chapters 9 through 13 offer a baseline for those who wish to collect additional data and track response over time. The continued study of bark beetle response to the thinning and fuel reduction treatments (chapter 9) is essential to fulfill the second research objective of the LMS. It will also be imperative for informing adaptive management of dry, east-side conifer forests. Though the white-headed woodpecker (*Leuconotopicus albolarvatus,* WHWO) work reported in chapter 11 focuses on fungal pathways of decay for nest cavities, future studies on woodpecker population dynamics in relation to snag recruitment and longevity are recommended. A spatial analysis of snags would be useful for occupancy surveys of woodpeckers and other wildlife species that use snags.

Collaborators have noted that an assessment of natural conifer regeneration by treatment type would be useful for informing how thinning and fuel reduction may or may not achieve long-term recruitment and structural goals. Additionally, measuring the reaccumulation of fuel will determine needs for future retreatment in dry, east-side, pine-dominated forests. Future reports on the forest response to thinning and fuel reduction treatments will benefit from a spatial analysis of large tree distribution pre- and posttreatment. For example, the 75 UMZ + Gaps treatment intended to preserve any preexisting clumping of large trees after implementation, but the success of this objective cannot be realized without a spatial analysis of large trees pre- and posttreatment. This analysis would also inform how the spatial distribution of large-diameter trees in relation to gridded sample plots affected the variability in BA among sample plots.

The LMS now serves as an excellent foundation for determining the long-term influence of changing environmental factors interacting with thinning and fuel reduction treatments. Given the relatively consistent soils and subsurface geology of the Lookout Mountain unit, variation in environmental factors interacting with disturbance may be the predominant driver structuring forest plant communities.

Microclimate data have been collected continuously from 2011 through autumn 2019. Analyses to date have focused on the initial development of microclimate geospatial surface mapping in 2014 and 2015, capturing the initial posttreatment conditions (chapter 13). The first priority is performing additional analyses to characterize the full temporal dynamics of the microclimate surfaces through 5 years posttreatment. A second priority is to assess the geospatial deployment of microclimate monitoring locations to best capture data that can be most effectively interpolated to maximize microclimate surface precision and to minimize error for the long-term establishment of a microclimate assessment network.

It is generally understood that thinning and fuel reduction treatments are effective means of reducing stand density and thus risk of catastrophic wildfire; however, the interaction of these treatments with novel environmental circumstances is unknown. Thinning has been advocated for reducing tree mortality caused by bark beetles in western conifer forests (Fettig et al. 2007, Oliver 1995). Long-term, landscape-scale studies, such as the LMS, are needed to identify how thinning and fuel reduction treatments affect the amount and distribution of bark beetle-induced mortality.

The maintenance and resilience of understory vegetation in forest ecosystems is less visible by still highly important, as it suggests trajectories of forest type change. Understanding interactions between long-term plant community responses to silvicultural treatments and shifting environmental factors is crucial for identifying and predicting effects on forest processes, such as conifer regeneration, nutrient cycling, and soil retention.

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Species List

Trees

Common name	Scientific name and authority
Bitter cherry	*Prunus emarginata* (Douglas ex Hook.) D. Dietr.
Douglas-fir	*Pseudotsuga menziesii* (Mirb.) Franco
Engelmann spruce	*Picea engelmannii* Parry ex Engelm.
Grand fir	*Abies grandis* (Douglas ex D. Don) Lindl.
Jeffrey pine	*Pinus jeffreyi* Balf.
Lodgepole pine	*Pinus contorta* Douglas ex Loudon
Mountain hemlock	*Tsuga mertensiana* (Bong.) Carrière
Ponderosa pine	*Pinus ponderosa* Lawson & C. Lawson
Shasta red fir	*Abies magnifica* A. Murray bis var. *shastensis* Lemmon
Subalpine fir	*Abies lasiocarpa* (Hook.) Nutt. var. *lasiocarpa*
Sugar pine	*Pinus lambertiana* Douglas
Western white pine	*Pinus monticola* Douglas ex D. Don
White fir	*Abies concolor* (Gord. & Glend.) Lindl. ex Hildebr.

Shrubs

Common name	Scientific name and authority
Bitterbrush	*Purshia tridentata* (Pursh) DC.
Giant chinquapin	*Chrysolepis chrysophylla* (Douglas ex Hook.) Hjelmqvist
Goldenbush	*Ericameria* Nutt. sp.
Greenleaf manzanita	*Arctostaphylos patula* Greene
Manzanita	*Arctostaphylos* Adans. sp.
Snowbrush ceanothus	*Ceanothus velutinus* Douglas ex Hook.
Wax currant	*Ribes cereum* Douglas

Subshrubs

Common name	Scientific name and authority
Dwarf rose	*Rosa gymnocarpa* Nutt.
Little prince's pine	*Chimaphila menziesii* (R. Br. ex D. Don) Spreng.
Pinemat manzanita	*Arctostaphylos nevadensis* A. Gray
Pipsissewa	*Chimaphila umbellata* (L.) W.P.C. Barton
Sidebells wintergreen	*Orthilia secunda* (L.) House
Wintergreen	*Pyrola* L. sp.

Forbs

Common name	Scientific name and authority
Beardtongue	*Penstemon* spp.
Cudweed	*Gnaphalium* spp.
Fireweed	*Chamerion angustifolium* (L.) Holub
Glaucous beardtongue	*Penstemon euglaucus* English
Groundsel	*Senecio* spp.
Groundsmoke	*Gayophytum* A. Juss. spp.
Lettuce wirelettuce	*Stephanomeria lactucina* A. Gray
Milk kelloggia	*Kelloggia galioides* Torr.
Phacelia	*Phacelia* Juss. spp.
Spreading dogbane	*Apocynum androsaemifolium* L.
Strawberry	*Fragaria* L. spp.
Thistle	*Cirsium* Mill. spp.
Washington lily	*Lilium washingtonianum* Kellogg
Woodland pinedrops	*Pterospora andromedea* Nutt.
White hawkweed	*Hieracium albiflorum* Hook.

Graminoids (grasses and sedges)

Common name	Scientific name and authority
Long-stolon sedge	*Carex inops* L.H. Bailey
Ross' sedge	*Carex rossii* Boott
Sedge spp.	*Carex* spp.
Western needlegrass	*Achnatherum occidentale* (Thurb.) Barkworth

Other plant

Common name	Scientific name and authority
Dwarf mistletoe	*Arceuthobium* M. Bieb spp.

Mammals

Common name	Scientific name
Bushy-tailed woodrats	*Neotoma cinerea* Ord, 1815
Douglas squirrel	*Tamiasciurus douglasii* Bachman, 1839
Golden-mantled ground squirrel	*Callospermophilus lateralis* Say, 1823
Long-legged myotis	*Myotis volans* H. Allen, 1866
Mountain cottontail	*Sylvilagus nuttallii* Bachman, 1837
Northern flying squirrels	*Glaucomys sabrinus* Shaw, 1801
Pallid bat	*Antrozous pallidus* LeConte, 1856
Siskiyou chipmunk	*Tamias siskiyou* A.H. Howell, 1922
Snowshoe hare	*Lepus americanus* Erxleben, 1777
Yellow-pine chipmunk	*Tamias amoenus* J.A. Allen, 1890

Birds

Common name	Scientific name
Black-backed woodpecker	*Picoides arcticus* Swainson, 1832
Dark-eyed junco	*Junco hyemalis* Linnaeus, 1758
Flammulated owl	*Psiloscops flammeolus* Kaup, 1852
Gray jay	*Perisoreus canadensis* Linnaeus, 1766
Hairy woodpecker	*Leuconotopicus villosus* Linnaeus, 1766
Lewis's woodpecker	*Melanerpes lewis* G.R. Gray, 1849
Northern flicker	*Colaptes auratus* Linnaeus, 1758
Northern spotted owl	*Strix occidentalis caurina* Merriam, 1898
Sapsucker	*Sphyrapicus* S.F. Baird, 1858 spp.
White-headed woodpecker	*Leuconotopicus albolarvatus* Cassin, 1850

Insects

Common name	Scientific name
Douglas-fir tussock moth	*Orgyia pseudotsugata* McDunnough, 1921
Fir engraver	*Scolytus ventralis* LeConte, 1868
Mountain pine beetle	*Dendroctonus ponderosae* Hopkins, 1902
Pandora moth	*Coloradia pandora* Blake, 1863
Pine butterfly	*Neophasia menapia* C. Felder and R. Felder, 1859
Pine engraver	*Ips pini* Say, 1826
Red turpentine beetle	*Dendroctonus valens* LeConte, 1860
Western pine beetle	*Dendroctonus brevicomis* LeConte, 1876

Fungi

Common name	Scientific name
Brown rot	*Fomitopsis pinicola* (Swartz) Karsten
Comandra blister rust	*Cronartium comandrae* Peck

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U.S. Metric Equivalents

When you know	Multiply by	To get
Nanometers (nm)	25,400,000	Inches
Millimeters (mm)	0.0394	Inches
Centimeters (cm)	0.394	Inches
Meters (m)	3.28	Feet
Square centimeters	0.15500031	Square inches
Square meters (m^2)	10.76	Square feet
Hectares (ha)	2.47	Acres
Newton meters	0.73756	Foot pounds
Degrees Celsius (°C)	1.8 °C + 32	Degrees Fahrenheit

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