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Density Management in the 21st Century: West Side Story

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4–6 October 2011, Corvallis, Oregon

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Abstract

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Since adoption of the Northwest Forest Plan (NWFP) in the early 1990s, there has been a fundamental shift in forest management practices on federal lands in western Oregon and Washington. Commodity-driven clearcut regeneration harvests have given way to thinnings intended to enhance development of late-successional forest habitats and to conserve important aquatic and riparian ecosystems. *Density Management in the 21st Century: West Side Story* presents abstracts and peer reviewed papers from a regional conference highlighting more than twenty years of research related to forest thinning in the NWFP area. Presentations from the BLM Density Management and Riparian Buffer Study provide a focal point, with presentations from additional studies providing for a more complete overview of the west-side thinning science. The contributions are organized as five topical themes: The Regional and Landscape Context for Density Management in the Northwest Forest Plan Era; Implementation and Influences of Density Management in the Terrestrial Ecosystem; Riparian and Aquatic Ecosystems and their Responses to Thinning and Buffers; Socioeconomics and Operations; and Thinning and Adaptation. Collectively, the contributions summarize many important forest dynamics and ecosystem responses to partial overstory removals. Interactions between aquatic and riparian ecosystem conservation measures and upland harvest are emphasized. Targeting resource management practitioners, decision-makers and researchers, the collected works provide a reference to the current and future roles and issues of density management as a tool for forest ecosystem management.

Keywords: Aquatic Conservation Strategy, biodiversity, density management, forest structure, ecosystem processes, late-successional habitat, northern spotted owl, public perception, riparian buffers, stream temperature, thinning, wildlife.

Editors

Paul D. Anderson is a supervisory research forester, Pacific Northwest Research Station, Forestry Sciences Laboratory, 3200 SW Jefferson Way, Corvallis, OR 97331; pdanderson@fs.fed.us.

Kathryn L. Ronnenberg is a research information assistant, Pacific Northwest Research Station, Forestry Sciences Laboratory, 3200 SW Jefferson Way, Corvallis, OR 97331; kronnenberg@fs.fed.us.

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Density Management in the 21st Century: West Side Story

Paul D. Anderson
and
Kathryn L. Ronnenberg,
Editors

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Two Decades of Learning about Thinning in the Ecosystem Management Era

Paul D. Anderson

Introduction

Adoption of the Northwest Forest Plan (NWFP, USDA and USDI 1993, 1994) in the early 1990s signaled a major shift in forest management on federal lands in western Oregon and Washington. The Plan reflected composite concerns raised by various resource managers, conservation groups, scientists, and the general public about the sustainability of production-oriented forestry as practiced from the early 1900s. Specifically, the plan addressed concerns regarding clearcutting, the harvest of old-growth trees and loss of related habitat, and threats to more than 1,000 potentially sensitive species (Thomas et al. 1993; USDA and USDI 1993), including regionally iconic salmonids and the Northern Spotted Owl (*Strix occidentalis caurina*). As new objectives such as ecosystem function and biodiversity became higher priorities, federal land managers strove to define new silvicultural practices that would provide ecologically sustainable alternatives to clearcutting and old-growth harvest while still providing for wood production and economic benefits. As a result, partial overstory removals —mostly alternative thinning methods—have replaced clearcutting as the predominant form of harvest on federal lands over the past two decades.

Two concepts that crystallized in the 1990s have been important to the development of contemporary thinning practices: 1) existing old-growth stands with large trees, complex crowns, and multiple canopy layers often developed at lower densities than those typical of current young, unthinned stands (Tappeiner et al. 1997); and 2) retained down wood, snags, and large older trees with complex crowns perform important functions in providing habitat for late-seral flora and fauna in young, managed stands (Franklin et al. 1997). When aligned with an objective to accelerate development of late-successional forests, these two concepts provide the rationale for thinning young, dense stands in unconventional ways to increase structural and compositional variability.

In addition, the 1990s ushered in an era of more holistic aquatic-riparian conservation and restoration in western forests, with a new emphasis on watershed values. The Aquatic Conservation Strategy (ACS) outlined in the Northwest Forest Plan conceptually shifted the aquatic management emphasis from individual stream-reach conditions to larger-scale watershed conditions, in accordance with evolving concepts of streams functioning as branched networks (Fisher 1997). In this regard, the ecological importance of small headwater stream reaches emerged as potentially significant. Small streams drain typically 70 to 80 percent or more of a watershed in the Northwest (Gomi 2002). Hence, they encompass significant habitat within a watershed context, but also potentially contribute significantly to downstream conditions as well as providing ecological subsidies to uplands (Baxter et al. 2005). Riparian reserve designations of the ACS extended protections to small non-fish-bearing streams by codifying interim streamside buffer widths. Within riparian reserves, harvest activity was not precluded, but must contribute to, or not retard, the attainment of ACS objectives. However, harvest, predominantly

Paul D. Anderson is a supervisory research forester, USDA Forest Service, Pacific Northwest Research Station, Forestry Sciences Laboratory, 3200 SW Jefferson Way, Corvallis, OR 97331; pdanderson@fs.fed.us

commercial thinning, occurred on only about 2 percent of total riparian reserve area in the first decade of ACS (Reeves et al. 2006).

Application, refinement, and validation of northwest forest ecosystem management concepts has occurred concurrently in local operational trials by practitioners and in formal studies conducted as research-management collaborations. Given 20 years of learning from research and operational experiences, there is much to discuss about thinning as a forest management tool relevant to local programmatic applications and broader regional strategies and policies.

Workshop Purpose & Scope

In October 2011, the principal partners of the Density Management and Riparian Buffer Studies of western Oregon (DMS, Cissel et al. 2006), the Bureau of Land Management, Forest Service, and Oregon State University, convened a 3-day conference, entitled *Density Management in the 21st Century: West Side Story*, as a means to present and discuss findings of nearly twenty 20 years of density management research. In a narrow sense, the conference provided a capstone to a first phase of monitoring, analysis, and inference of the varied terrestrial, riparian, and aquatic ecosystem responses to the initial implementation of the DMS experimental treatments in the late 1990s. While various facets of the DMS were intentionally featured, the conference organizers recognized that the science related to thinning within the Northern Spotted Owl range of western Oregon and Washington has been developed through several important studies and in various disciplinary, ownership, and interest-group contexts. A broader intent of the conference was to highlight this wide array of thinning-related science to stimulate a discussion of what we know and understand, or have yet to resolve, about thinning as a silvicultural practice to meet contemporary and future management objectives. We purposefully sought representation from other west-side studies for comparison and synthesis. Stimulation of meaningful dialogue was dependent on broad representation of the researchers who generated information and the practitioners and decision-makers who translate the science into outcomes of ecological and social importance. To this end, the conference included 40 oral and 16 poster presentations by more than 65 authors, including decision-makers, practitioners, and researchers from 11 different government, private industry, and academic institutions.

Proceedings Overview

We present a number of papers and abstracts which highlight the conference themes. These themes include: The Regional and Landscape Context for Density Management in the Northwest Forest Plan Era; Implementation and Influences of Density Management in the Terrestrial Ecosystem; Riparian and Aquatic Ecosystems and Their Responses to Thinning and Buffers; Socioeconomics and Operations; and looking forward, Thinning and Adaptation. Collectively, the papers and abstracts illustrate the degree to which the realm of thinning extends beyond a narrow focus on trees to broader forest ecosystem considerations. To view the oral presentations, visit the interagency ECOSHARE website: *http://ecoshare.info/products/completed-workshops/thinning-workshop/*

To set the context for contemporary thinning, Kenneth Ruzicka and coauthors provide an overview of the origins of the Density Management and Riparian Buffer Study in relation to the Northwest Forest Plan. The importance of public perceptions of forest management activities in determining social acceptability is addressed by Robert Ribe. At the regional landscape scale, trends in harvest intensity and the relationships to land ownership patterns and ecological gradients are addressed by Robert Kennedy and coauthors and by Janet Ohmann.

Application of thinning and subsequent

vegetation responses are addressed in several papers. Klaus Puettmann and coauthors describe understory species abundance and diversity responses as observed in the DMS and the Young Stand Thinning for Diversity Study (Poage and Anderson 2007). Peter Gould and Constance Harrington describe the development of new models of understory tree development. Kyle Dodson and coauthors address tree mortality and snag recruitment in the DMS. Mark Harmon addresses the fundamentals of thinning effects on carbon stores, and Julia Burton and coauthors discuss trade-offs between development of diverse understory vegetation communities and the accumulation of carbon stores. Daniel Luoma and Joyce Eberhart look below the soil surface to characterize mycorrhizal responses to the level and pattern of green-tree retention harvests as implemented in the Demonstration of Ecosystem Management Options (DEMO) study (Franklin 1999). Heather Root and Bruce McCune describe potential benefits to lichen diversity associated with retention of large legacy trees and decreased stand densities in young stands. The variable pathways which can lead to old-growth structures are described by Tom Spies and Robert Pabst. The effectiveness of alternative density management approaches to placing young stands on trajectories toward late-seral structure is the focus of John Tappeiner and of Mike Newton.

The impact of contemporary density management practices on enhancing terrestrial habitat for a variety of organisms was an important conference focus. From the perspective of the Northern Spotted Owl, Todd Wilson and Eric Forsman discuss the relationships between thinning and habitat suitable for flying squirrels, the principle prey base in portions of the owl's range. John Cook and Rachel Cook discuss the importance of understory vegetation nutritional value as a determinant of possible Elk (*Cervus elaphus*) forage benefits of thinning. The finding that different taxonomic and functional groups of birds benefit differentially from thinning is discussed by Joan Hagar.

The contemporary aquatic and riparian issues associated with forest management strategies at longer-term, landscape scales are framed in the context of integrated watershed analysis by Gordon Reeves. Kelly Burnett outlines concepts of intrinsic potential—the persistent characteristics of streams that define the potential to provide quality fish habitat. The hydrogeomorphic processes responsible for delivering wood to streams, and therefore realized fish habitat are reviewed by Dan Miller. Influences of thinning and riparian buffer width on aquatic and riparian attributes at a finer reach scale include works addressing microclimate and stream temperature responses (Bianca Eskelson and coauthors; Paul Anderson and coauthors; Jeremy Groom), and stream wood recruitment (Mark Meleason and coauthors; Paul Anderson and coauthors). Deanna Olson summarizes DMS findings on the impacts of thinning and alternative riparian buffer widths on amphibian habitats and species abundances and assemblages. The responses of invertebrates, mollusks, fishes and small mammals to alternative streamside buffers in the presence clearcutting in western Washington are presented by Peter Bisson and coauthors and Martin Raphael and Randall Wilk. Jason Dunham and coauthors describe fish responses to harvest in a paired watershed study. Robert Danehy and Sherri Johnson distill many of the salient lessons into four principles of stream ecology applicable to forest management.

From an operations perspective, Kurt Steele draws attention to the practical challenges associated with applying variable density thinning. David Marshall addresses the implications of contemporary thinning practices for growth and yield. Loren Kellogg and Steven Pilkerton discuss the forest engineering issues of harvest planning requirements, production costs, and stand damages associated with contemporary thinning operations.

Looking to the future, Deanna Olson and Kelly Burnett define new concepts for providing habitat connectivity in headwater-dominated landscapes. Possible implications of climate change on stream flow, thinning, and riparian buffer needs are addressed in a poster by Julia Burton and coauthors. The impact of thinning on the resilience of wildlife habitat to climate variation is discussed in a poster by Andrew Neill and coauthors.

Personal Perspectives

The conference papers serve well to illustrate the science underlying current harvest practices as well as some of the larger policy questions surrounding forest management. However, as pointed out by Mike Haske (Deputy State Director for Resources, USDI BLM) and Jerry Ingersoll (Forest Supervisor, USDA Forest Service Siuslaw National Forest) in their reflections on the presentations, unanswered questions remain, some of which are science-oriented and many that are social or political. In the following, I take the opportunity to address from my perspective as a scientist some of the points of discussion that occurred outside of the presentations and are not necessarily addressed in the proceedings papers.

Thinning isn't rocket science, but...

Principles of thinning and density management have been established over centuries of "conventional" silviculture practice in Europe and more recently in North America and elsewhere. So what is novel about the contemporary application of thinning in western Oregon and Washington? For me the answer is driven by both why and how we thin. If we change the "why" we thin, the "how" we thin changes also. When the primary objective of silviculture has been meeting wood-production objectives, thinning served primarily to harvest slower-growing trees, some destined to die, and to reallocate site resources to fewer stems of the most economically productive species, thereby promoting increased stand vigor and growth. Two common consequences of conventional thinning practices have been increased uniformity of forest structure and composition, and removal or delay in the development of dead wood as snags or down wood to meet decadence and habitat functions. Variable densities, skips and gaps, retention of minor species and snags, active creation of snags and down wood, and underplanting are elements that have been explored in contemporary density management prescriptions at the stand level. The fundamental ecological and physiological principles underpinning thinning effects on residual trees remain the same, but the interactive influences of increased spatial variability and structural complexity may alter stand development. Our experiences with contemporary thinning practices are still too few and too recently adopted to state with certainty that the broadened array of ecological objectives is being met.

Have we chosen the correct references from which to gauge observed thinning effects?

A concern raised by some conference field trip participants was that many of the contemporary thinning studies base inferences on the responses of thinned young stands relative to responses for similarly young but untreated stands: in effect, a comparison between active management and a passive, no-treatment alternative. While this is perhaps a statistically sound approach, are young, unthinned stands the most relevant standard, given the common objective of developing late-seral attributes? There are few if any 30- to 50-year-old stands developing under natural, unmanaged conditions that can provide contemporary references to the managed stands that are the current target of thinning. Further, it has been difficult to locate late-seral stands to serve as valid references to desired conditions. During initial planning of the DMS, representative late-seral reference stands were sought, but where located, the late-seral stands tended to occupy settings incomparable to the younger managed stands to be thinned—they were often at higher elevations, were forest fragments, and did not include streams. In the absence of site-specific late-seral reference stands, there are a limited number of published studies that provide definitive stand structure and composition characteristics either qualitatively (e.g., Bauhus et al. 2009) or quantitatively (e.g., Poage 2005), that can provide late-seral benchmarks. However, published characterizations were derived retrospectively, and reflect an unknown range of variability in developmental trajectory.

The earliest of the contemporary west-side thinning studies was implemented in 1992 and 1993 on the Siuslaw National Forest (Poage and Anderson 2007). The study has generated data from 17 years of post-treatment monitoring; thus, we are only 15 to 20 percent of the way to observing the planned late-seral objective. The developmental path that these recently treated stands will ultimately take in approaching late-seral conditions is both empirically unknown and relatively poorly modeled. Regardless of these uncertainties and the lack of clear standards, our research studies may better serve managers by focusing on thinning response trends as deviations from the desired late-seral conditions rather than deviations from untreated, young-stand references.

Morticulture—the management of death and decay

Critical to managing for late-successional habitats is an improved understanding of how alternative silvicultural practices interact with processes of tree death and decay and how silviculture can be used to augment or enhance those processes. Over the past several decades our ecological understanding of decadence and its importance to habitat and biogeochemical processes has increased substantially, but translation of the fundamental knowledge into coherent goals is lagging. There is likely consensus that decadence is needed; however, questions of how much, what form, where, and when continue to be a subject of debate. We have some valuable tools such as DecAID (Mellon et al. 2002) that describe the landscape variability of terrestrial snag and down wood abundances as well as rudimentary wood abundance-species use relationships, and NetMAP (Benda et al. 2007), which can quantify the probability for delivery of wood to stream channels. However, it remains a challenge to quantify the incremental gain in ecological benefits associated with incremental increases in snag and down wood abundance. Increasingly, simulation modeling is being used to explore trade-offs among benefits associated with alternative silvicultural approaches at stand or landscape scales. Our readily accessible forest-growth simulation models (e.g. Forest Vegetation Simulator; ORGANON) were developed from relatively strong tree-growth data, but unfortunately they prove to be rather poor platforms for projecting tree mortality and down wood recruitment and persistence. Improvement of these simulators will require that we invest in the analysis of the empirical snag and down-wood data accruing in our long-term silviculture experiments to provide more robust functions for model improvement. Further, uncertainties in stochastic mortality events may require novel coupling of stand-level simulators with larger-scale process or disturbance models to better project snag and down-wood dynamics at multiple scales of ecological importance.

Riparian buffers and the ACS

Thinning as conducted in the DMS, other studies, and operationally on federal lands is a much different disturbance than clearcutting when it comes to potential harvest alterations of riparian habitats, including microclimate and stream temperature. The NWFP interim guidelines for riparian reserves were relatively conservative, and we have learned that in some contexts such as small headwater streams, key ecological functions such as shading or habitat for aquatic and riparian dependent vertebrates can often be fulfilled with narrower buffers. However, as we consider more intensive thinning or retention harvests, or other ecosystem functions and their interactions, questions remain regarding the efficacy of various buffer configurations. At a larger scale, the intent of the ACS was to provide a watershed focus to assessments of stream condition—particularly, recognition that natural dynamics will result in periods of poor condition at the fine scales of stream or stream reaches, and therefore trends at the watershed scale may be more meaningful for assessing overall landscape condition and restoration efficacy. There is a need to reconcile a large-scale ACS approach with management actions and condition criteria applied at the finer stream-reach scale. Until we can effectively think about important aquatic and riparian management issues in the context of integrated, multi-scaled ecosystems, conflicts between management toward large-scale outcomes and finer-scale regulatory objectives are likely to recur.

What is the future for thinning?

Whether on BLM or Forest Service lands, thinning has become a very important silvicultural tool. In the Pacific Northwest Region, thinning and fuels-reduction treatments targeting forest restoration objectives account for about 97 percent of the Forest Service vegetation management budget (Jeff Walters, Region 6 Director of Natural Resources, U.S. Forest Service). Perhaps the continued predominance of thinning is uncertain, both in terms of continued availability of suitable stands for thinning and in terms of the limited ability of thinning to provide some desired landscape elements such as early-seral habitat. I am not overly concerned about either issue as a determinant to whether or not thinning continues to be an important silvicultural tool into the future. For example, recent discussions proposing increased regeneration harvest on federal lands using green-tree-retention (Franklin and Johnson 2012) as a means to increase the abundance of early-seral forest do not preclude restoration thinning. Rather, variable-density thinning conceivably can be an important intermediate treatment in silvicultural systems featuring legacy retention, natural regeneration, and longer rotations or cutting cycles. However, long production-cycle strategies involving the eventual harvest of large trees may continue to be constrained by a lack of social acceptance. At larger spatial and temporal scales, a very large palette of silvicultural approaches and management strategies is available to meet diverse ecological and societal objectives. Given that forests are complex systems with varying capacities to respond and adapt to disturbance or stressors, uncertainty will be the norm that justifies using a variety of management approaches.

In my opinion, the greatest challenge facing forest resources management has long been, and will continue to be, the development and fulfillment of the implicit social license permitting management of public and private forest lands to sustainably meet the breadth of needed ecological and societal services. As a scientist my intent, similar to that of my peers, is to objectively deliver information and tools that are used to inform forest management options and policy. Although scientists may suggest the management implications of their work, it is intended that considerable room be allowed for natural resource managers to exercise innovations in application as they integrate among themes and prioritize their own goals. To this end, it has been personally rewarding to serve as one principal investigator for the Density Management and Riparian Buffer Studies, and to have participated in the planning and conduct of the conference which is captured in the individual papers and abstracts that follow in this volume.

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Photos, facing page—Top: View from Green Peak toward the Willamette Valley. Photos by Paul Anderson. **Center left:** Variable-width buffer at Green Peak. Photo by Paul Anderson. **Center right:** Unthinned two tree-height buffer at Keel Mountain, with Andy Neill for scale. Photo by Paul Anderson. **Bottom left:** Stream wood and Western Skunk Cabbage (*Lysichiton americanus*) at Keel Mountain. Photo by Mark Meleason, USFS. **Bottom right:** Edge of federal and private forest industry ownerships, Green Peak. Photo by Paul Anderson.

Section 1. The Regional and Landscape Context for Density Management in the Northwest Forest Plan Era

The Intertwining Paths of the Density Management and Riparian Buffer Study and the Northwest Forest Plan

Kenneth J. Ruzicka, Jr., Deanna H. Olson, and Klaus J. Puettmann

Abstract

Initiated simultaneously, the Density Management and Riparian Buffer Study of western Oregon and the Northwest Forest Plan have had intertwining paths related to federal forest management and policy changes in the Pacific Northwest over the last 15 to 20 years. We briefly discuss the development of the Northwest Forest Plan and how it changed the way forest policy was developed in the region. The concurrent conceptualization and implementation of the Density Management and Riparian Buffer Study within this new management framework resulted in a proof-of-concept for adaptive management approaches outlined in the Plan, especially relative to riparian and upland restoration practices. The Density Management Study serves as a model for integrated knowledge discovery and adaptive management within the context of the federal forest plan. The future of the study appears to be similarly interconnected with the interagency plan for federal lands management.

Keywords: Forest thinning, western Oregon, Bureau of Land Management, Forest Service.

Introduction

The Density Management and Riparian Buffer Study (DMS) originated in 1993 as a long-term, operational-scale experiment to investigate silvicultural techniques intended to accelerate development of late-successional and old-growth forest characteristics in western Oregon (Cissel et al. 2006). To accomplish this, the study tested alternative thinning prescriptions that were not yet tested or established in the scientific literature at the time of study establishment. The upland thinning design of DMS was conceived before finalization of the Northwest Forest Plan (hereafter, the Plan) (USDA and USDI 1994a), but was implemented within the new policy framework created by the Plan. Although not an initial goal of the study, the implementation of the DMS identified ways that federal agency land managers in western Oregon, in particular the U.S. Bureau of Land Management (BLM), could adapt to work within the new policy framework and more quickly achieve Plan goals, especially in reserved land allocations.

By examining the DMS study, we explore the intertwining threads of policy changes, knowledge discovery, and new management questions asked during development and implementation of the Plan. We briefly discuss the history of the Northwest Forest Plan and how

Kenneth J. Ruzicka, Jr. is a doctoral student, Department of Forest Ecosystems and Society, Oregon State University, Corvallis, OR 97331, kenneth.ruzicka@oregonstate.edu; **Deanna H. Olson** is a research ecologist, USDA Forest Service, Pacific Northwest Research Station, Forestry Sciences Laboratory, 3200 SW Jefferson Way, Corvallis, OR 97331; **Klaus J. Puettmann** is Edmund Hayes Professor in Silviculture Alternatives, Department of Forest Ecosystems and Society, Oregon State University, Corvallis, OR 97331.

it changed the way forest policy was developed. We show how the DMS arose within the new management framework, and finally discuss how the DMS has affected federal forest management at district and forest levels relative to regional management policy. It appears certain that western Oregon forest managers will continue to employ adaptive management to adjust to new challenges. The future of the DMS appears to be similarly interconnected with the Plan.

The Northwest Forest Plan: Abbreviated Synopsis

The listing of the Northern Spotted Owl (*Strix occidentalis caurina*) as a threatened species under the U.S. Endangered Species Act in 1990 (55 FR 26114 26194) was neither the beginning nor the end of the struggle between old-growth forest preservation and timber management. Regional scientists had been concerned about loss of owl habitat since the early 1970s (Forsman 1975; Yaffee 1994). Although the spotted owl was the poster child for the forest-management conflicts coming to a head in the early 1990s, numerous scientists had been researching forest fragmentation effects on many different species, characterizing species-habitat relationships, and conducting species risk assessments (e.g., Ruggiero et al. 1991; Thomas et al. 1993). In addition to threatened species concerns, an emerging paradigm shift toward ecosystem management with multiple resource and restoration objectives came into conflict with conventional management practices on federal forests in the Pacific Northwest (Yaffee 1994). Historical timber harvest practices were geared toward wood commodity production and implemented to "get the cut out" (Thomas 2008; Williams 2009). Clashes between old-growth forest preservationists and federal land management agencies climaxed with a series of lawsuits that prevented timber sale and harvesting activities throughout the range of the Northern Spotted Owl (Thomas et al. 2006). These lawsuits in part defined a new era for natural resource management in the U.S. where the management of public land was decided by legal determinations made by judicial courts (Thomas 2008).

In the polarized forest management climate of the early 1990s, President Clinton assembled the Forest Conference in 1993 to discuss the different social, economic, and environmental issues concerning old-growth forests. He ordered federal land management agencies to draft a balanced, long-term policy that would direct the management of federal forests in the Pacific Northwest in the range of the spotted owl (USDA and USDI 1994b).

President Clinton mandated that the "policy must:

- Never forget the human and the economic dimensions
- Protect long-term health of our forests, our wildlife, and our waterways
- Be scientifically sound, ecologically credible, and legally responsible
- Produce predictable and sustainable levels of timber sales and nontimber resources
- Make the federal government work together and work for you" (FEMAT 1993).

The Forest Ecosystem Management and Assessment Team (FEMAT), which included representatives from the BLM, U.S. Forest Service, Fish and Wildlife Service, National Park Service, National Marine Fisheries Service, National Oceanic and Atmospheric Administration, and Environmental Protection Agency, was created to craft this difficult policy. The final report was released in July of 1993 (FEMAT 1993) and was adapted into the Northwest Forest Plan in 1994 (USDA and USDI 1994a; 1994b). Numerous detailed discussions of the Plan are available (e.g., see USDA and USDI 1994a; Yaffee 1994; Marcot and Thomas 1997; Thomas 2008).

The Northwest Forest Plan started a process toward implementing ecosystem-scale, science-based forest management (Thomas 2008). One hallmark of the Plan was that for the first time, scientists led development of land management alternatives and played a key role in crafting federal policy (Bormann et al. 2007). FEMAT consisted of a team of scientists with expertise in various aspects of forest management and its economic, ecological and social impacts. The team was led by the “Gang of Four plus Two”—wildlife and fisheries researchers from the Pacific Northwest aided by a cadre of hundreds of consultants (FEMAT 1993). This unique set of expertise manifested itself in a new management approach. The Plan was the first effort to manage a forest ecosystem at the regional scale across an expansive area of ~10 million hectares (24.5 million acres) using a science-based, multi-resource approach. The original FEMAT design conceptually ensured that management actions taken on a local scale would not impair ecosystem function at a regional scale (FEMAT 1993). Fine-scaled management elements were later integrated into the regional Plan, providing for a multi-scaled management framework. “Survey and manage” requirements were added by President Clinton’s Council on Environmental Quality to manage sensitive species with restricted distributions that were not otherwise protected by the Plan. Fine-scale survey and manage requirements combined with coarser-scale land-use allocations (LUAs) were intended to facilitate Plan implementation without triggering additional environmental regulations for species protections. With these requirements in place, the intent of the Plan was to assess projects by their outcomes as part of the interconnected regional ecosystem rather than assessing management impacts only on individual stand-level land parcels (Diaz 2004).

The integrated management of a regional ecosystem required that federal agencies, which had previously operated as fragmented administrative units, become partners. However, integrated management led to several operational challenges. Previous forest management actions had been planned through the local district personnel, including foresters and silviculturists, and were approved based on District or Area plans for inventory and timber harvests. When the first comprehensive meetings between federal and state land managers and researchers were convened to discuss the spotted owl and old-growth management, participants lacked standardized approaches among agencies to assess multi-ownership forest blocks. For example, region-wide maps of forest resources were not available, and local or agency-specific mapping conventions made existing smaller-scale maps incompatible (Yaffee 1994). Plan implementation was delayed as procedural issues were worked out.

The Density Management and Riparian Buffer Study as an adaptive learning platform

Much of what the FEMAT discussed was founded in research conducted in the 1980s to 1990s to address characterization of old-growth forest (e.g., Franklin and Spies 1991), and to understand its development (Spies and Franklin 1988). Other researchers sought to understand how to manage young Douglas-fir (*Pseudotsuga menziesii*) plantations for eventual old-growth characteristics. Dr. John Tappeiner, a Senior Research Forester of the Bureau of Land Management Cooperative Research Unit and Oregon State University Professor, wanted to investigate new restoration management practices on BLM lands. Contemporary studies such as forest thinning in the Black Rock Forest Research Area (Del Rio and Berg 1979) or the Young Stand Thinning and Diversity Study (Hunter 1993) were designed to contribute to this topic. Those studies provided excellent information on the growth of forest stands, especially the relationships between tree and stand characteristics and stand density. However, they did not represent the variability that would occur in the overstory and understory at the stand level (J. Tappeiner, emeritus professor, Oregon State University College of Forestry, personal communication). The need for studies with broadened scope, such as the DMS, was apparent as other large-scale silvicultural experiments were also implemented during this time (Poage and Anderson 2007). The DMS treatments were designed to be implemented at the large-stand scale (50- to 300-ha treatments), with study sites encompassing the range of managed stand conditions more representative of western Oregon managed forests.

The Riparian Buffer Study component was added to the initial density management study objectives in 1994 (Cissel et al. 2006; Olson 2013). This component addressed the "interim" provision in the NWFP regarding riparian reserves. The riparian reserve land-use allocation in the Northwest Forest Plan was defined to provide specific conservation and mitigation for aquatic and riparian resources. The interim riparian reserve widths (ranging from two site-potential tree heights for fish-bearing stream reaches to one site-potential tree height for intermittent streams) were intended to be adjusted after watershed and project-specific analyses of aquatic-riparian resources. The riparian buffer aspect of DMS was designed specifically to test various widths of riparian buffers in conjunction with upland thinning according to Plan Standards and Guidelines (USDA and USDI 1994b), as well as possible management options within riparian reserve boundaries (Hohler et al. 2001; Cunningham 2002; Diaz and Haynes 2002).

Thus, by 1994, the DMS had become the first stand-scale test of joint riparian and upland management prescriptions in the Pacific Northwest, and sought to address knowledge gaps identified during Plan development. In summary, the DMS was implemented to answer two general questions: 1) How well do alternative thinning pathways accelerate the development of late-successional forests? and 2) What are the effects on aquatic-riparian resources of riparian buffers of varying widths in conjunction upland thinning?

The DMS presented logistical and organizational challenges that echoed other hurdles in implementing the Plan. Processes developed during early DMS implementation were later used to address questions and concerns about Plan implementation. For example, DMS site selection and study implementation processes helped to frame later discourse about procedures for forest management planning under the Plan. During study-site selection for the DMS, several criteria were weighed. These included geographic representation, forest type and condition, and land-use allocation as described in the Plan on lands owned by the BLM. In the early 1990s, much of the federal landscape (52 percent of BLM holdings) in western Oregon consisted of Douglas-fir plantations younger than 80 years old (Cissel et al. 2006), with 35 percent of BLM holdings younger than 40 years (Muir et al. 2002). Thinning projects in young Douglas-fir stands had already been proposed on some national forests and BLM resource areas, and hence areas suitable for the study were easily found. Some administrators expressed doubt about the efficacy of the heavy thinning prescription proposed in the variable-density treatment (Cissel et al. 2006), and local natural-resource managers raised concerns about the use of treatments within the range of the spotted owl or the Marbled Murrelet (*Brachyramphus marmoratus*). On the heels of the forest management paradigm shift signaled by the entire Plan, which included a new set of land-use allocations for the federal lands of the Pacific Northwest, field-level personnel also raised concerns about potentially locking up lands that were intended to be managed for later regeneration harvests ("matrix" land-use allocation) into a long-term study (C. Thompson, BLM, personal communication). Others held the opinion that the study aims might be irrelevant because the BLM had been thinning stands for a number of years and knew its effects on stand development, and also because with a new [federal] administration, public land-use policy would revert to what it had been (J. Tappeiner, emeritus professor, OSU College of Forestry, personal communication). Eventually, the scope of the study was narrowed, when the Medford

BLM district was dropped from the study because the forest types and stand conditions in this subregion were deemed to be too different from the other BLM ownerships to warrant inclusion in the study. DMS sites were selected from four western Oregon BLM districts, including seven stands in lands allocated as matrix and three stands in late-successional reserves (LSR). Three additional sites on LSR-designated lands managed by the Siuslaw National Forest were selected specifically for implementation of the Riparian Buffer Study component, with elements of the moderate upslope thinning (Cissel et al. 2006). DMS study sites were among the first forest management projects implemented by field units after finalization of the Northwest Forest Plan, and in many cases, set the stage for later Plan management-unit implementation.

The study was conducted at an operational scale, addressing many logistical issues that arose later in other thinning operations. The DMS treatments were designed to diversify stands with gaps and, leave islands, as well as variable-density thinning. Consequently, sale-preparation crews had to be trained in new methods to select trees for harvest to achieve a heavier and more spatially variable thinning, compared to more conventional thinning operations. Marking crews were quick to adapt to the challenge, with most stands being adequately marked to meet prescriptions with a single effort (C. Thompson, BLM, personal communication). Additionally, methods for marking riparian buffers also needed to be developed. Researchers from the PNW along with BLM personnel developed approaches at DMS sites that were subsequently used along headwater streams in other areas of western forests. These have been general guidelines, however, and many districts still implement a buffer width of one or two site-potential tree heights for riparian buffers as proposed in the Plan (USDA and USDI 1994a). Study site coordinators from the BLM were also able to resolve concerns about sensitive species and microhabitats by adjusting how leave islands were placed. Although random selection of upland treatment units and reaches for buffer treatments was preferred, site-specific delineation adjustments to address implementation barriers allowed the DMS to act as a model for resolving project conflicts at multiple scales (Olson et al. 2002). Loggers and other equipment operators adjusted to felling and maneuvering trees in and around riparian reserves and leave islands.

Furthermore, socioeconomic considerations of implementing the DMS were projected relative to both operational and implementation costs. To make DMS sales profitable, patch openings and off-site parcels were included in some DMS sales, as they increased harvest amounts and market values (Olson et al. 2002). Wood harvest as a result of DMS implementation at individual BLM study sites was estimated to range from ~1 to 8.5 mmbf (million board feet; 2359 to 20 058 m^3) from site-specific project areas ranging from 73 to 162 ha (Olson et al. 2002). DMS harvest volumes exceeded those of conventional thinnings, largely due to inclusion of patch cuts. As with any profitable harvest on BLM-administered land at the time, the DMS harvests provided timber benefits to the economy of local counties (Olson et al. 2002). The DMS demonstrated that despite various hurdles, complicated management prescriptions can be implemented in a way that provides silvicultural and operational learning opportunities. Subsequent timber-project planning efforts benefited from the lessons learned from DMS implementation.

In contrast, reviews of the success of adaptive management in the Northwest Forest Plan have mostly identified short-comings in adaptive management areas. Problems with the application of adaptive management generally lie in the lack of a learning framework for management and a failure to close the learning loop even after monitoring is completed. Adaptive management areas included flexible provisions to address the need for development of innovative management approaches and integration of site-specific contexts into local projects. Surveys of federal land managers who have operated designated adaptive management areas identified factors limiting successful implementation, such as regulations, especially the Endangered Species Act, a lack of support from higher-level administrators, and a lack of cooperation between researchers and managers (Stankey et al. 2003). Other criticisms such as risk aversion, poor record keeping and budget misallocation also were identified as hurdles to implementation (Bormann et al. 2007). The primary failures of adaptive management were the lack of integrating learning opportunities from project inception through to final monitoring and the testing of alternative strategies. These aspects would heighten the efficacy of adaptive management approaches, and go beyond the collection of just monitoring data (Bormann et al. 2007).

However, there were some bright spots in implementing adaptive management, such as the creation of a formal monitoring program (Haynes et al. 2006), increased stakeholder involvement (Stankey et al. 2006), and increased respect and cooperation between researchers and decision makers (Bormann et al. 2007). Suggestions to improve the way adaptive management areas function included defined leadership and support roles, continuity, standardized documentation, and a commitment to learning (Stankey et al. 2003, 2006). After receiving the interpretive report (Haynes et al. 2006), federal regional managers developed and approved a formal adaptive management framework, and refocused questions for future efforts. This completed the cycle of adaptive management for the first 10 years of the Plan. The DMS sets forth a model of how a more formal question-focused learning effort can be carried out and fit within a learning environment across multiple land allocations.

Application of the DMS

The DMS study is an example of effective active management and research in support of the Plan. In line with the objectives of the study, the DMS has: shown that variable-density thinning, including gaps and heavy thinning, moved canopy heterogeneity closer to old-growth conditions (Wilson and Puettmann 2007); demonstrated that 15-m buffers largely maintained the stream microclimate within the upland thinning context (Anderson et al. 2007); and identified headwater-dependent aquatic species that may warrant consideration during forest management, for their habitat maintenance and landscape connectivity considerations (Olson et al. 2007; Olson and Burnett 2009, 2013; Olson 2012). Although set up primarily to answer basic questions about forest and riparian management, the study has included additional studies on bryophytes, lichens, songbirds, macroinvertebrates, and other natural-resource elements (Cissel et al. 2006). Over 100 research papers and technology-transfer products have resulted from the study (*http://ocid.nacse.org/nbii/density/pubs.php*). Additionally, through field trips, presentations, and outreach programs such as "Teachers in the Woods", over 5000 people were exposed to the DMS or visited the study sites from 2003–2006 (USDI, unpublished data). The study has provided an opportunity for prolific knowledge discovery, and has become a platform for diverse discussions regarding forest management.

From the outset, the DMS was designed to be relevant to forestry operations. Many of the original study designers from the BLM and PNW are still involved in the study. The result of this continuity has been a steady stream of management-relevant products and face-to-face workshops to disseminate the results of the study. In this way, the DMS has been, by design, not only relevant to the researchers, but has intimately involved managers and resource specialists as well. As a result of effective technology transfer, the DMS study has had a direct effect on the way forest thinning projects are planned by field management units. Research produced by the DMS has been cited in several recent Environmental Assessments to provide evidence that the planned management is scientifically sound. BLM districts with DMS experience as well as Forest Service projects in western Oregon have used the DMS in project planning. Projects in the Salem BLM district like the Gordon Creek Restoration Project (USDI BLM 2009) cited the district's experience with the DMS as evidence for the effectiveness of variable-density thinning treatments. The Highland Fling Project used the DMS to predict effects on riparian species (USDI BLM 2010). The Forest Service also uses results from the DMS. The Siuslaw National Forest used results from the DMS to predict the response of the understory to project actions in the environmental assessment for the Salmon/ Neskowin Project (USDA FS 2011).

In addition to the technology transfer provided by the DMS, direct alliances among DMS researchers, field managers, and natural-resource specialists have aided in bridging science and application to on-the-ground forest thinning and riparian management. The combined effort from the DMS and other large-scale silvicultural experiments provides a wealth of useful information for project planners and implementers (Poage and Anderson 2007; Anderson and Ronnenberg 2013). The sustained flow of technology transfer, including science publications as well as summary articles and workshops geared for managers, is an essential part of applied research programs. Furthermore, DMS has been used as a demonstration project for broader learning opportunities. University classes, natural-resource specialist training workshops, and national to international forestry and natural-resource specialists have visited the sites in order to understand the relevance of the variety of treatments established there. Flyers, brochures, and Web-site materials have allowed further outreach to the broad forestry community. DMS concepts have been presented to forestry organizations such as the Society of American Foresters and the International Union for Forest Research Organizations, raising the profile of DMS research applications. Finally, the DMS is an example of the evolving concept of adaptive management. Findings from the first 10 years of the study have led to "phase 2" treatments, with heavier thinning treatments and further-modified riparian buffers. Phase 2 of the DMS now tests the application of an additional thinning harvest at the sites, conducted within the existing treatments, and includes a case study of thinning without a riparian buffer. Additional response variables could be investigated within this new context to expand the learning paradigm offered by the overarching study template. If DMS sites can be retained as designated research areas beyond the scheduled end point of current studies, they could continue to be used to refine the ideas of adaptive management in the future.

Although the DMS has shown its relevance to individual projects conducted by local administrative units, it has had less impact on recent regional planning efforts like BLM's Western Oregon Plan Revision (WOPR; USDI BLM 2008) or national-level planning such as the recently proposed Planning Rule Revision from the Forest Service. The WOPR included provisions to increase timber harvest on BLM lands in western Oregon. Increased harvest would be accomplished by increasing the amount of land where harvest was allowed, reducing the width of riparian buffers and streamlining the process to approve higher thinning intensities. Although directly relevant to many WOPR issues, DMS findings were seldom referenced (USDI BLM 2008), with exception of microclimate results. The DMS findings can contribute more to science-based forest management decisions in western Oregon; closing this apparent gap is a challenge to the DMS researchers and their partners who directly navigate this science-management interface. This is an area for further development as DMS outreach to managers and decision-makers continues into the future.

Whereas the vacated WOPR was unsuccessful in changing provisions in the Northwest Forest Plan on the regional level, the Forest Service is working to alter how forest management decisions are being made at the local level. In the last decade, the Forest Service has tried to implement policies to make it easier to conduct thinning projects or other forest management activities. A sample of these policies includes President G.W. Bush's Healthy Forests Initiative in 2003, and the 2008 Planning Rule Revision (USDA FS 2008), which was suspended in 2009. A new final planning rule was recently released (*http://www.fs.usda.gov/planningrule*), and includes national-level process guidance for programmatic planning at the unit (national forests and grasslands) and regional scales, to set the context for local project and landscape-scale decision-making. It also emphasizes learning while planning and direct stakeholder involvement in collaborative assessments. Local project National Environmental Policy Act (NEPA) compliance will need to be constrained within regional and individual national forest plans, but will most likely be less prescriptive than the Northwest Forest Plan (B. Bormann, USFS, PNW Research Station, personal communication). The proposed Planning Rule Revision also emphasizes consideration of climate change and carbon storage in management decisions, and increasing public participation in project planning (USDA FS 2011). Approval of the revision will increase opportunities for DMS findings to significantly inform project planning, especially in west-side national forests. Continued development of informational products geared toward field managers will be important for DMS technology transfer to stay current. Bridging science and management across organizational scales of local areas to regions presents new challenges for studies such as the DMS and their proponents.

Finally, the DMS and density management research must stay relevant in the face of trends of declining research support from the federal government. With the DMS, the BLM has shown that it can incorporate research and sound management practices using agency personnel and agency land. By involving researchers and management personnel from multiple agencies, field offices, and districts, management projects have been funded and implemented to benefit all parties. New research paradigms may be needed to expand research capacity, because funding may be lacking in an economically austere environment. However, by partnering researchers with specialists at field units who manage lands, new projects can be conceived and implemented by a suite of collaborators, not by scientists alone: DMS has been a model in this regard. The DMS has shown how multiple agencies can work together and solve problems at study, design, implementation, and information delivery stages (Olson et al. 2002).

Several ongoing and emerging research questions included or brought about by the DMS still need attention. Further study is needed on the management of young stands for late-successional and old-growth characteristics, including the age-range within which management toward the development of these characteristics can be effective; on how to manage fuel loads; and on how to work within the "80-year rule" (Tappeiner 2009). There is also a need to use matrix lands, especially those controlled by the BLM, for biodiversity management research, especially on early-seral habitat (Spies et al. 2007). Many questions remain about the efficacy of riparian reserve widths with upland regeneration harvests. Most managers do not vary from widths prescribed in the Plan, although studies from western Washington including those by Bisson et al. (2013) and Raphael and Wilk (2013) are contributing to this arena. In recent years, the initial success of the DMS has carried the momentum to other inquiries, such as the interplay between upslope treatment and water availability in riparian areas (Burton et al. 2013; K. Ruzicka, unpublished data) and modeling how climate change will affect the trajectory of the stands (K. Ruzicka, unpublished data). These projects rely on the long-standing relationships among Oregon State University, the Forest Service Pacific Northwest Research Station, and the BLM. Cooperation between researchers and land management agencies is more important than ever.

Many collaborative land-management research projects, including the DMS, can improve their communication and partnering efforts with regional to national interest groups. Local managers of BLM and Forest Service lands understand the importance of research findings for the development of science-based management proposals. They can apply research results directly to management plans, including citing appropriate publications in NEPA documents. The absence of integration of DMS findings in recent national and regional management planning efforts suggests that research communication and advocacy to higher-level managers and policy-makers needs more attention. Project scientists and their science managers need to consider development of more effective mechanisms to actively engage regional administrators in dialogues about the outcomes and impacts of their work for regional-to national-scale management and policy development. Ensuring the integration of science findings at higher administrative scales would result in improved science-based policy decisions. Science communication across geographic scales of natural resource management and administration, and across agencies, makes this a complex task. Without integrated communication, including both the processes and the personnel such liaisons would entail, the outcome and impact of valuable studies such as DMS are not fully realized. In an era of constantly emerging issues, such as global climate change, invasive species, disease, and fire and pest management, science communication across administrative boundaries has never been more important.

Conclusion

The Northwest Forest Plan was the first regional ecosystem management plan, and tested the logistical, operational, and cooperative capacities of land managers in the region. The Density Management Study originated in an atmosphere of new forest and riparian management questions, with important policy implications proposed to be answered through adaptive management. Over a decade of research in adaptive management has identified that the opportunity for learning must be realized at all stages of the project for management to be successful. The DMS also has provided a set of lessons-learned for how to successfully plan, implement, and communicate stand-scale research to a wide audience. With climate change and other emerging natural resource issues on our horizon, increased uncertainty in research funding, and anticipated changes in forest policy, the importance of the DMS will only increase. The DMS can serve as a model for integrative studies conducted jointly by collaborative partners from diverse research communities and land-management agencies. Furthermore, DMS study sites can continue to be relevant as documented locations that provide opportunities to efficiently address new, emerging forest-management questions. The DMS remains a golden opportunity for cutting-edge forest research in the future.

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Public Perceptions of West-side Forests: Improving Visual Impact Assessments and Designing Thinnings and Harvests for Scenic Integrity

Robert G. Ribe

Abstract

Perceptions of public forests' acceptability can be influenced by aesthetic qualities, at both broad and project levels, affecting managers' social license to act. Legal and methodological issues related to measuring and managing forest aesthetics in NEPA and NFMA decision-making are discussed. It is argued that conventional visual impact assessments—using descriptive pictorial qualities against a naturalistic scenery standard—have limitations as legal evidence, in addressing other popular aesthetic values, and helping public participation in planning processes. But such descriptive assessments do have merit: they are similarly perceived by diverse people, they describe landscape attributes that managers can manipulate, and they are strongly related to the public's broad-trust perceptions of forests' acceptability. Evidence-based guidelines are offered for the production of scenic quality in Pacific Northwest west-side forests. These guidelines are derived from extensive studies of forests and perceptions in the region, and estimate and interpolate average public perceptions of average forest conditions. They inform in-stand perceptions related to forest density measurements and regeneration harvest prescriptions, of percent and pattern of tree retention, and of retained down wood. Other guidelines related to percent and pattern of retention and harvest unit design apply to vista views of harvests. These guidelines can assist planners and managers in designing forest treatments, implementing wholesale forest plans to maintain broad-trust acceptability perceptions, more reliably meeting scenic integrity standards, and making more accurate visual impact assessments at regional and project scales.

Keywords: Forest visual impacts, scenery management, timber harvest design, social acceptability, public participation.

Introduction

Perhaps in an ideal world, public forest planners and managers would only need to make decisions to optimize the technical achievement of well-defined natural-science-based objectives. Recent experience suggests that it is also important to manage forests' social acceptability to avoid popular and antagonistic perceptions of forest plans and projects (Bengston 1994). This can help forestry professionals regain and maintain their "license" to plan and execute projects with professional discretion. This paper seeks to provide practical aesthetic theory and evidence-based guidance to forest planners in estimating visual impacts and keeping forestry projects from going past "tipping points" of acceptability.

Perceptions of "social acceptability" are often not the same as those that forest managers encounter in public meetings, hearings, or other legal contexts. They are more intuitive, emotional, and holistic (Hansis 1995; Wyatt et al. 2011). They are not primarily motivated

Robert G. Ribe is a professor, Department of Landscape Architecture and Institute for a Sustainable Environment, 5234 University of Oregon, Eugene, OR 97403; rribe@uoregon.edu

to reflect the needs of a particular individual, community, corporation, or interest group. Such perceptions can be positive or negative, and become powerful when they are widely and deeply felt. They are not always formed according to criteria defined by elites, stakeholders, activists, or lawyers. A well-defined legal story usually lies behind forest management conflicts. But behind that story often lie aesthetic perceptions reinforced by political narratives, typically related to winners and losers, justice and trust in government (Tindall 2003). Together these can create adverse perceptions of forest management (Ford et al. 2009). Such shared perceptions can rely on cognitive simplifications of the complex world of forest policy and planning more than on the technical details of forest science, policy, litigation, or implementation (Shindler et al. 2002; Tindall 2003; Olsen et al. 2012).

Public agencies gain acceptance of forest management at two social scales. The first can be called "broad-trust acceptability", and encompasses regional or national populations and relates to many people's trust in an agency to serve public interests and take good care of forests. The second can be called "project-level acceptability" and is more focused on the details, trade-offs, and conflicts of interest that are manifest in "hot spot" projects, area plans, and the rules and policies that drive these. Gaining and maintaining license to manage land and harvest timber requires both kinds of acceptance. The spotted owl controversy involved a loss of broad-trust acceptability and regional/national loss of social license. Forest management projects or forest plan revisions held up by local opposition and legal/administrative challenges involve loss of project-level acceptability (Blahna and Yonts-Shepard 1989; Trosper 2003).

Social acceptance of forest management is driven by perceptions at both these scales (Shindler et al. 2002). Perceptions that drive broad-trust acceptability occur among members of the "general public" who are extensively interested in public forests because they serve generally understood values. These concerns are not typically intensely focused on particulars or places but are influenced by ideological or intuitive perceptions of public landscapes and by stories and social narratives found in news media and social networks (Allen et al. 2009; Clement and Cheng 2011). Perceptions that drive detailed project-level acceptability are mainly those of smaller numbers of activists and stakeholders focused on specific, conflicting agendas and values. Perceptions of power relationships and justice typically frame these perceptions because they are seen to weaken or strengthen these competing goals (Cheng and Mattor 2006; Daniels and Walker 1995). Technical, functional, and spatial details can matter much more in negotiating solutions that gain acceptance among conflicting parties.

Forest managers who design thinnings and harvests are familiar with these facts of political life. They are also aware of the general importance of aesthetic forest perceptions (Ribe 1989), but the role of aesthetics in decision-making is challenging (Shindler et al. 2002). They face at least three problems: (1) how to predict the aesthetic impact of forestry projects in advance for the various required environmental impact assessments; (2) how to go about managing the larger public forest landscape to maintain broad-trust acceptability; and (3) how to understand and communicate about forest aesthetics in local negotiations and public participation processes aimed at gaining project-level perceptions of acceptability across diverse constituencies. A basic, simplified discussion of practical aesthetics to address these problems is sketched below. Then, evidence-based recommendations are offered to aid forestry project design.

Practical Forest Aesthetics

The National Environmental Policy Act of 1969 (NEPA) promulgated a decision process that emphasizes scientific analysis (Bartlett 1986). The original NEPA process requires few and inadequate opportunities for public participation, and these are constrained and located in the process so as to be unlikely to influence the design of alternatives or to yield useful and accountable perceptions of projects' acceptability (Hourdequin et al. 2012). Congress did, however, require assessments of visual impacts, while recognizing that this might fly against the science-based rationalism of the rest of the law. NEPA states: "…assure for all Americans… aesthetically…pleasing surroundings…and… develop methods and procedures…which will insure that presently unquantified environmental amenities and values may be given appropriate consideration in decision making…" The National Forest Management Act of 1976 affirmed this mandate with respect to forest landscapes: "…cuts… shaped and blended to the extent practicable with the natural terrain."

In principle, such visual impact assessment methods must pass evidentiary due process tests to avoid being deemed an inadmissibly arbitrary and capricious basis for public decisions (Tassinary et al. 2010). This arguably amounts to passing the same tests of reliability and validity applied to other scientific assessments required by NEPA (Palmer 2000; Palmer and Hoffman 2001). The U.S. Forest Service addressed this requirement for visual assessment methods (Smardon 1986) by adoption of the Visual Management System (VMS, USDA FS 1974), now modified as the Scenery Management System (USDA FS 1995). A pragmatic core approach to making purportedly reliable and valid visual assessments in these methods is to restrict forest aesthetics to formalistic descriptions of pictorial aesthetic qualities. These must conform to a normative standard of naturalistic scenery (Selman and Swanwick 2010; Sheppard 2001).

Visual assessments based upon naturalistic form, line, color, scale, contrast, etc., were assumed to have intuitive "face validity" (Litton 1972; Wohlwill 1976). This is the first-impression, 'primary aesthetic quality' of forest landscapes (Ingarden 1973). These have been assumed to be reasonably reliable if made systematically by trained experts (i.e., landscape architects) using standard procedures (Litton 1968; Smardon 1986). Reliance on such experts also has the advantage of producing assessments related to the form of landscapes managers can actually control. It also has the advantage of avoiding assessments based on hearsay public opinion "driven" by various value agendas, which can define their own aesthetic qualities (Parsons and Daniel 2002). These might range into diverse, "special", and potentially contentious or contradictory ad-hoc conceptions and perceptions of aesthetic value different than those specified by NEPA. NEPA was crafted to avoid creating an intractably irresolute political process and to keep pure judgments and decision authority in the hands of objective experts and officials with democratically vested authority (Bartlett 1986).

NEPA-compliant methods of assessing the aesthetic impacts of forest management, i.e., the SMS, have been legally accepted as "best available professional practice." This is because (1) they have been formally adopted by agencies; (2) some sanctioned method is required and the institutional cost of changing those methods would be prohibitively high even if they were found legally invalid; (3) lawyers and judges instinctively avoid contesting aesthetic measurements as a risky, cost-ineffective "can of worms"; and (4) demonstrably valid and reliable, widely-applicable alternatives, not requiring slow and expensive methods, have not been proposed. Nevertheless, the reliability and validity of SMS scenic impact assessment methods have not been demonstrated (Palmer 2000), and they have remained essentially unchanged for more than 30 years, while hundreds of scientific studies of landscape perception have been published.

Studies of visual-aesthetic perceptions of landscapes frequently confirm that public perceptions of scenic aesthetics can be predicted by descriptive formalistic or cognitive sense-making attributes of landscape scenery (Ode et al. 2008; Ryan 2005). The exact degree of these relations varies in different landscape settings, cultural contexts, and with different research methods (Stamps 1999). Other studies confirm that perceived naturalism is a powerful, positive aesthetic value among most people everywhere, perhaps in "hard-wired" ways (Ulrich 1979), with psychological health benefits (Thompson 2011; Grinde and Patil 2009).

An advantage of expert visual impact assessments using naturalistic, pictorial standards is that this simple, face-valid concept of aesthetic quality is strongly related to public perceptions of broad-trust acceptability (Carvalho-Ribeiro and Lovett 2011). Ribe (2002) showed that perceptions of scenic beauty versus acceptability were strongly correlated among large samples of people with diverse environmental attitudes in the Pacific Northwest (fig. 1). This is consistent with other findings from the region (BCMF 1996; Kearney 2001; Kearney et al. 2011). Ribe (2006) also found that scenic beauty was the third-strongest predictor of informed perceptions of forests' overall acceptability (behind perceived habitat quality and economic value) in the same region (fig. 2). This is similar to other regional findings (Bradley and Kearney 2007; BCMF 2006; Brunson and Shelby 1992). The formalistic, pictorial naturalistic aesthetics specified by NEPA can therefore serve as a guidepost, among other considerations, in designing forest management plans and projects aimed at maintaining broad-trust acceptability.

Predicting Scenic Impacts of Forestry Projects

The problem remains of how visual assessments can be made more valid and reliable. When it comes to forests of the west-side Pacific Northwest, empirical evidence is available to serve as one major basis for more accurate visual impact assessments. Appropriately designed research can offer a strong contribution to legally

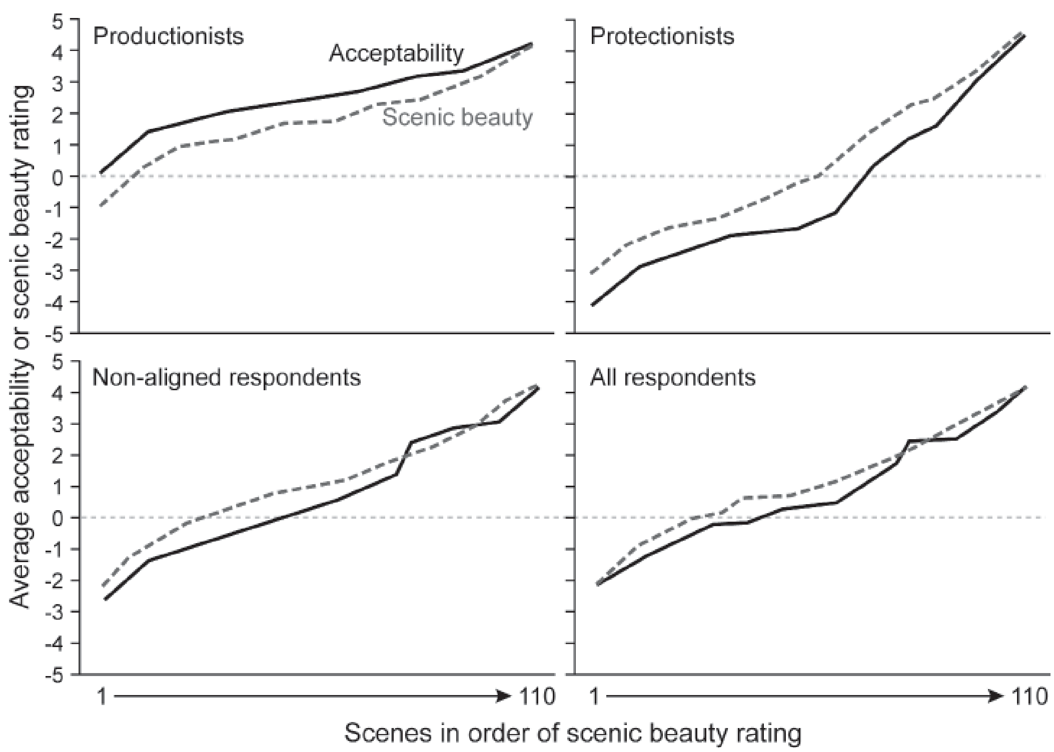

Figure 1—The relationship between uninformed acceptability ratings of 110 photos of west-side forests and uninformed scenic beauty ratings of the same photos among people with different environmental attitudes. Adapted from Ribe (2002).

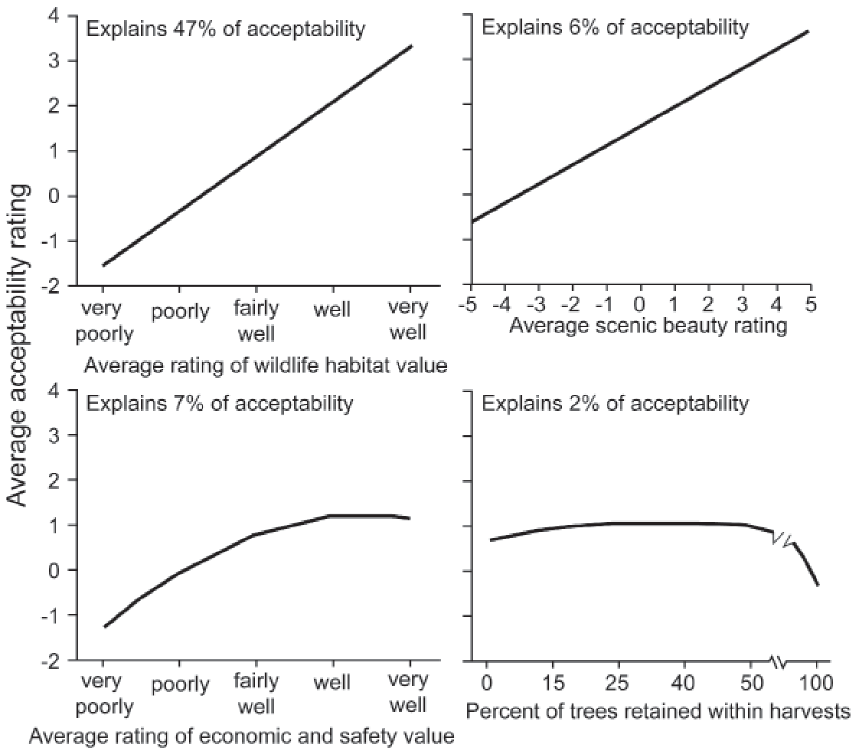

Figure 2—The relative strength and functional form of perceptions of information and of photos' beauty as these were associated with average overall acceptability perceptions of forest treatments in west-side forests. Adapted from Ribe (2006).

valid and reliable visual assessments of forest changes, using methods explained in detail by Tassinary et al. (2010).

Research cannot set the normative standards for visual impact assessment. These are socially defined and expressed through democratic due process of law and administrative procedures, i.e., NEPA and the SMS. But research can measure perceptions against these legally specified aesthetic concepts and standards. Representative samples of people's aesthetic perceptions can be elicited for well-sampled forest scenes. Ratings of legally appropriate scenic qualities can be statistically related to forest attributes depicted in the photographs. Such ratings of photographs correspond closely to perceptions in the field (Trent et al. 1987).

To understand how research can inform visual impact assessments of public forests, readers who are not landscape architects first need to understand the basics of SMS scenic standards. The SMS divides national forest landscapes into sub-areas where different desired "scenic integrity level" standards are applied (USDA FS 1995). These are based upon the beauty of each area's natural scenery, how visible the area is (hidden or seen at greater distances), how many people view it, whether what they are likely to be doing there is more sensitive to scenic quality, and how much aesthetic value local residents and recreators place upon each area. These integrity levels are conceptually defined and range from "very low" to "very high."

A critical problem with making admissibly valid and reliable visual assessments lies in predicting which scenic integrity level a thinning or harvest will conform to after it is finished. If a landscape architect is actively involved in designing a project, they can shape the cut unit as seen in a vista view to minimize visual impact and affect the scenic integrity level it should achieve. But a shortage of landscape architects may mean that this task is only performed for more sensitive projects. When silviculturists, engineers, and biologists design projects, they have no qualifications to make visual impact assessments, and landscape architects or outdoor recreation planners may be expected to provide post-hoc, conjectural visual impact assessments. If these assessments are only made for in-stand views, none of these design or non-design professionals may have a clear enough idea what the forest landscape will look like after logging operations. It may be impractical to control the visual consequences of logging operations to the degree needed to predict final scenic integrity levels using SMS procedures with high confidence, either soon after logging or over the long term.

When extensive programs of forestry projects are planned, in specific or as-yet-unspecified locations (e.g., as the product of a forest plan or long-term timber sale program), there is typically little basis for valid and reliable descriptive assessments of "ungrounded and un-designed" projects using SMS methods. This wholesale decision-making is where full EIS assessments are most often required and where strong visual impact assessments are most needed and potentially subject to scrutiny.

Research can assist in improving visual impact assessments in both these situations. It can't explain the full variability in perceived impacts as a consequence of the detailed or accidental forest scenery produced by each and every forestry project. But it can inform reliable and valid predictions of average perceived visual impacts by an average member of the public in relation to scenic integrity standards. This can be done in relation to attributes of forest structure that managers intentionally manipulate in their prescriptions.

Practical Extra-NEPA Planning-process Aesthetics

Pro-forma NEPA- and NFMA-based decision processes have encountered political and legal challenges. Successful appeals have created precedents, case law, rules, and statutes that can be demanding of decision-makers and contradictory or confounding. These have empowered stakeholder and activist groups more than NEPA and similar organic planning laws arguably intended. One solution forest management agencies are pursuing is to engage in more intensive and meaningful public participation (Arnstein 1969) that promotes collaborative learning and collective decision-making (Daniels and Walker 1996). These aim to negotiate decision design and the trade-offs needed to arrive at a solution acceptable among multiple interests. This approach seems to be headed for effective use (e.g., Gordon et al. 2012) and codification in the "new forest planning rule" for national forests (USDA FS 2011).

Aesthetics' role in this emerging new forest planning process will be potentially important and interesting. Some brief speculative comments are offered here. There are many kinds of aesthetic experiences and qualities that might become salient or powerful in participatory planning. Philosophers and aesthetic theorists make very weak and subtle distinctions between aesthetic perceptions that occur in a passionate cognitive mode versus dispassionate claims of intrinsic or inherent value (Santayana 1896; Richards 2001; Kiester 1997). Forest planning activists tend to have passionate values based upon strong aesthetic experiences and consequent long-standing forest affections (Buijs 2009; Brown and Raymond 2007). These typically motivate their participatory agendas. Examples are perceptions of the beauty of a river basin rich in healthy salmon habitat; the pleasing social and economic vitality of a rural town supported by a strong timber industry; the harmonious ecology of an old-growth forest; or the sublime historical process by which resource development has over time inexorably, powerfully, and exploitatively lifted untold numbers of people into the middle class and provided them with greater freedom. Such aesthetic perceptions underlie spoken or unspoken claims of transcendent, intrinsic values that ought to be preserved in and of themselves as wonderful, ingenious, and vulnerable (sui-generis).

Obviously, such strong aesthetic perceptions behind claims of intrinsic value can be in conflict. Indeed, they form a powerful basis for social conflicts that block forest decision-making and enliven attempts at participatory planning. Assessing and pursuing these kinds of aesthetic qualities in forests is quite different from the much safer, broadly shared (common-denominator) goal of producing as much naturalistic, formalistic pictorial landscape as possible, as Congress has directed. Many planning participants are embarrassed to voice aesthetic concerns. They seem "too religious", outside the technical aspects of decision-making that civil democratic discourse emphasizes, or divorced from the most relevant social narratives of who has power, gets what, and pays the price in political decision-making. Claims of intrinsic aesthetic value are vulnerable to the retort: "So what exactly do we decide to do to satisfy your aesthetic needs and all the other competing intrinsic (aesthetic) values?" Conventional visual impact management arguably avoids this problem by focusing on "superficial" appearances that are subject to manipulation in the landscape rather than in people's minds.

Naturalistic pictorial aesthetics may not be a major player in many public participation negotiations, as they can be too narrow and irrelevant to the most salient values in a conflict. Obvious exceptions will occur when important economic values flow from local exceptional visual aesthetic resources. But care should be taken not to sacrifice naturalistic pictorial visual landscape qualities when negotiating collectively acceptable solutions. This can put broad-trust acceptability perceptions at risk. Pictorial qualities may not be "at the table", but they still matter.

It may not be a waste of time to seek and propose specially crafted conceptions of aesthetic quality that cut across or transcend participants' competing, intrinsic aesthetic affections. Some of the ecological aesthetic ideas behind ecosystem management can be of this kind. These can seek to appreciate how active resource management can aesthetically resonate with the function of natural systems in ways most people perceive as intrinsically good (Gobster 1999; Carvalho-Ribeiro and Lovett 2011). Within a public participation process, it may be possible to promote shared aesthetic perceptions of the quality of the participation process itself. This might be a means to promote a sense of common purpose, and of the "intrinsic" value of an agreement that to some extent supersedes its privately perceived aesthetic inadequacies. But don't expect such internal process aesthetic perceptions to be shared outside the group.

Evidence-Based Aesthetic Decision Assistance for Pacific Northwest Forestry

This section provides evidence-based aesthetic decision guidance to forest planners regarding the density and structure of managed forests, particularly in the Pacific Northwest. The aims are (1) to help keep forest projects from going past "tipping points" of broad-trust acceptability perceptions that can jeopardize forest managers' social license; and (2) to more accurately predict average visual impacts of forestry projects that use the "standard" logging procedures used in the research studies from which the findings came. These impacts can be improved to some unverified extent by use of aesthetic forestry techniques, such as those advocated by Klessig (2002), the University of New Hampshire Cooperative Extension (1993), and others.

"Broad-trust tipping point" guidelines are offered based upon where source studies estimated that scenic integrity tends to be perceived to cross from low to moderate level. The aim is not to suggest that all public forestry projects should follow these guidelines. Biological and economic goals will sometimes necessitate otherwise. Instead, the suggestion is that if a preponderance of projects seen in the landscape meet or exceed the guidelines, then broad-trust acceptability perceptions should be maintained. The focus is on the author's studies of perceptions of west-side forests of Oregon and Washington. Only the most useful results of these studies are extracted and sketched. Refer to the original publications for details of methods and all findings.

In-Stand Views

Several studies are reviewed here that investigated the production of scenic beauty against measures of forest structure and types of forest treatments, including unmanaged forests. These studies investigated only treatments in forests at least 40 years old, and only the scenery found within six months after treatments. Improvements do occur in scenic beauty in the early “green-up” years after regeneration harvests, but these are not as appreciable to the general public as might be hoped for (Kearney et al. 2011).

Mean changes in average scenic beauty perceptions attributable to different forest treatments were measured via public surveys where many groups of respondents rated numerous matching forest photographs of pre- and post-treatment conditions (Ribe 2005a). The resulting aesthetic changes are shown in fig. 3, without regard to differences in the initial scenic beauty of the pre-treatment forests. The resulting sequence of magnitudes of scenic change is similar to that found by Bradley and Kearney (2007). More intense treatments tend to produce greater reductions in scenic beauty, particularly if one accounts for the extent of open clearcut areas produced within harvest units, such as in aggregated retention harvest patterns.

Density Management and Scenic Beauty

Ribe (2009) found that the density and basal area of mature forests’ structure are significantly related to average in-stand perceptions of scenic beauty. These results, described below, were derived from large samples of photographs of diverse pre- and post-treatment forests as rated by a large sample of residents of western Washington and Oregon.

A best-fit polynomial regression function (fig. 4) relating forests’ density to scenic beauty estimated on an interval scale (Ribe 1988) was graphed against U.S. Forest Service Scenery

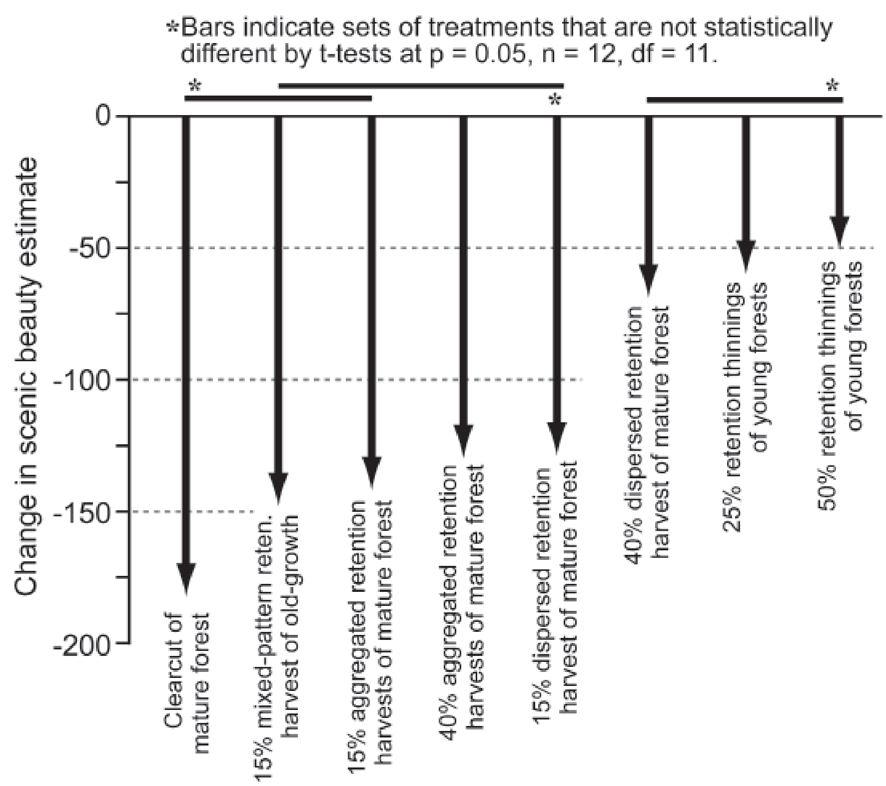

Figure 3—Comparisons of changes in average perceived scenic beauty ratings attributable to different forest treatments. From Ribe (2005b).

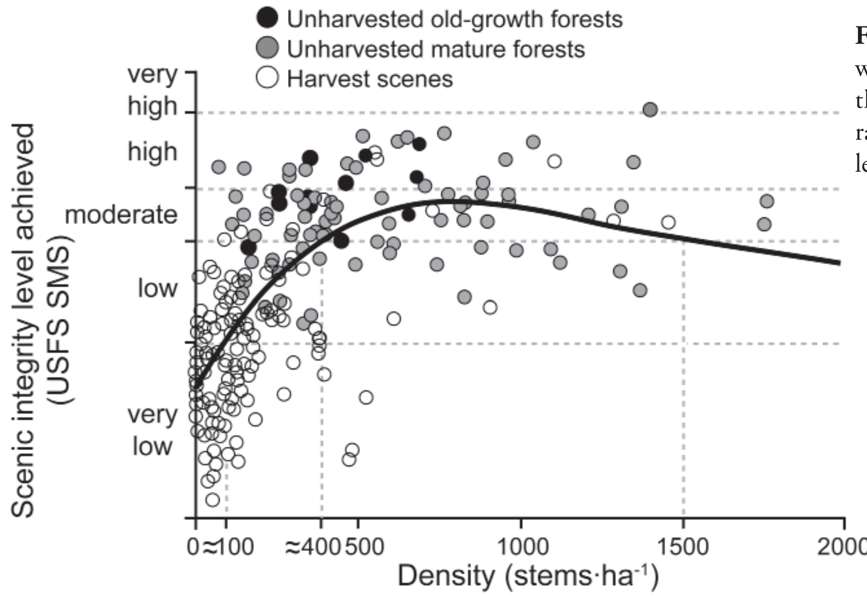

Figure 4—The relationship between west-side forests' stem density and their average in-stand scenic beauty ratings interpreted by scenic integrity levels. Adapted from Ribe (2009).

Management System integrity levels (USDA FS 1995). On average, forests with 400 to 1500 stems >5 cm·ha^{-1} tend to achieve the highest perceptions of scenic beauty, with the peak at about 750 stems·ha^{-1}. Very low, potentially adverse, aesthetic perceptions will tend to result from in-stand views of forests with less than 100 stems·ha^{-1}. The "broad-trust tipping point" threshold is about 400 stems >5 cm·ha^{-1}.

A best-fit polynomial regression function relating forests' basal area (BA) to in-stand scenic beauty perceptions (fig. 5) suggests that, on average, forests with 120 to 170 m^2·ha^{-1} BA will tend to achieve very high scenic integrity characteristic of the most beautiful old-growth forests. Forests with more than 170 m^2·ha^{-1} BA will tend to achieve high scenic integrity, likely due to the inclusion of larger trees within dense forests with broad diameter distributions; while those with 50 to 120 m^2·ha^{-1} BA will also tend to achieve high scenic integrity, likely due to moderate densities of mainly moderately large trees. Very low, potentially adverse, aesthetic perceptions will tend to result from in-stand

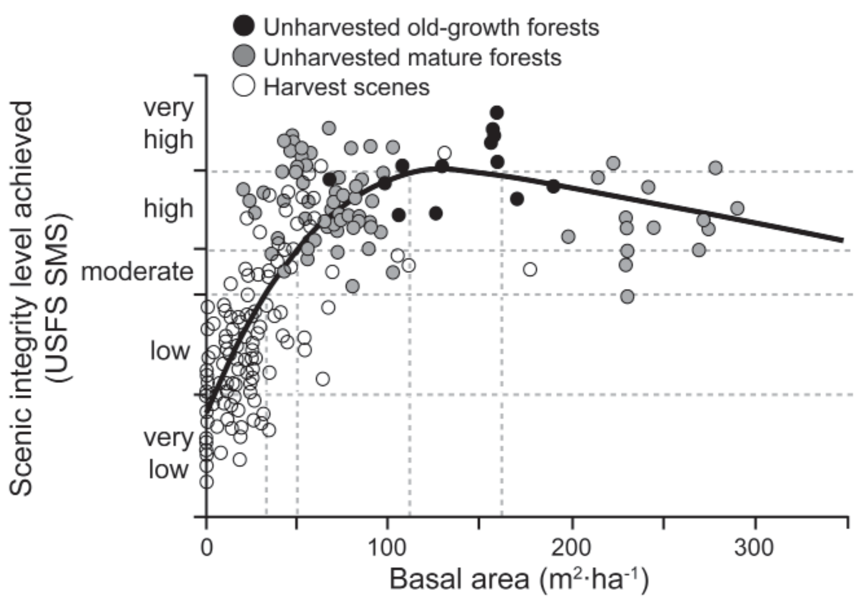

Figure 5—The relationship between west side forests' basal area and their average in-stand scenic beauty ratings interpreted by scenic integrity levels. Adapted from Ribe (2009).

views of forests with less than about 30 $m^2 \cdot ha^{-1}$ BA. The "broad-trust tipping point" threshold is about 50 $m^2 \cdot ha^{-1}$ BA.

For regeneration harvests seen from within, the relation between post-treatment retention levels and scenic integrity levels also accounted for levels of down wood (fig. 6) (Ribe 2009). Harvests with 40 percent dispersed retention and low down wood can achieve, on average, as much scenic integrity as pre-harvest forests. Other 40 percent retention harvests will tend to produce more marginal, moderate to low scenic integrity. Harvests with 15 percent aggregated retention and high down wood will tend to produce very low scenic integrity, comparable to clearcuts. Other 15 percent retention harvests can be expected, on average, to produce low scenic integrity. Ribe (2006) found that informed acceptability perceptions (as opposed to uninformed scenic beauty) were not significantly affected by high down wood levels. This suggests that public education should improve broad-trust perceptions of forests with high down wood in spite of their low scenic quality.

The same study (Ribe 2009) statistically interpolated average scenic integrity levels other than those produced at the 15 percent and 40 percent levels photographed and presented to respondents. This model has high margins of error (fig. 7). The key finding there is not so much the exact average scenic integrity level predicted for any retention level, but the interaction of favoring dispersed retention patterns and low residual down wood in producing scenic beauty, particularly at moderate (15–70 percent) retention levels. The broad-trust harvest retention levels are at about 15 percent for dispersed tree patterns with low down wood; about 30 percent in aggregated patterns with low down wood; about 35 percent in dispersed patterns with high down wood; and about 55 percent in aggregated patterns with high down wood.

Scenic Beauty of Harvests in Vista Views

Perception of scenic beauty in vista views (fig. 8) appears to be related to the design of the shape of harvest units rather than down wood (Ribe 2005b). These results suggest that aggregated-retention harvests will tend to produce low to very low scenic integrity, similar to clearcuts, irrespective of retention level. Dispersed retention patterns are more effective at mitigating scenic impacts in vista views than harvest shapes. At 15 percent retention, dispersed as opposed to aggregated retention patterns will tend to produce low, rather than very low, average scenic integrity. At 40 or 75 percent

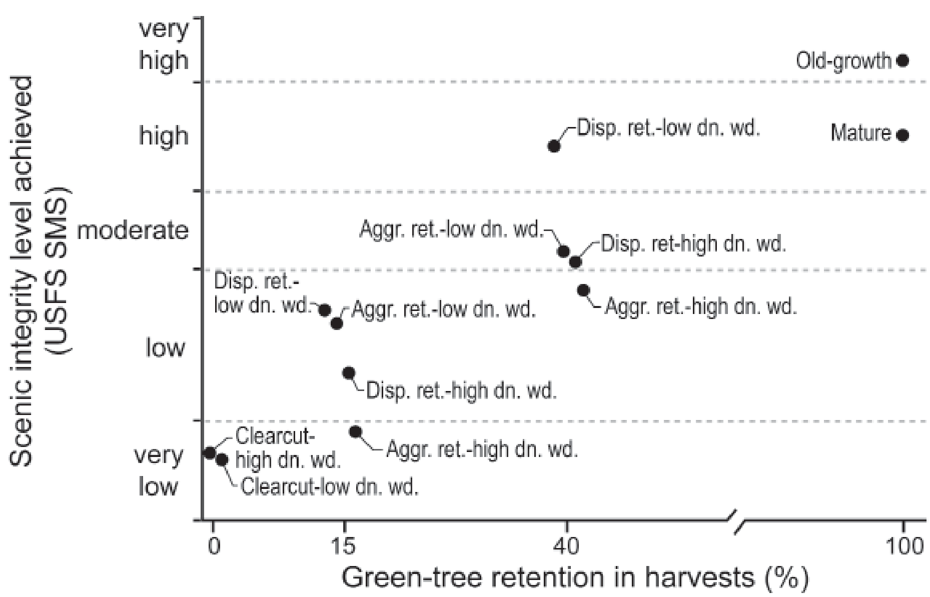

Figure 6—The relationship between west side forest regeneration harvests' variable tree retention levels and patterns, as well as down wood levels, and their average in-stand scenic beauty ratings interpreted by scenic integrity levels. Adapted from Ribe (2009).

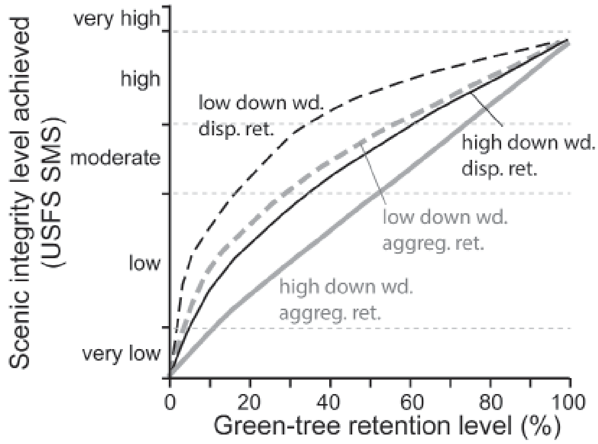

Figure 7—An estimated general statistically interpolated model of perceived scenic integrity levels in relation to a continuum of regeneration harvests' tree retention levels, sorted by retention pattern and down wood level. Adapted from Ribe (2009).

retention, dispersed patterns will tend to produce high to very-high scenic integrity, instead of the low to very low integrity levels associated with aggregated patterns. The "broad-trust tipping point" is 25 percent retention for dispersed retention harvests, irrespective of unit design. All aggregated retention harvests, including traditional clearcuts, fall below the "broad-trust tipping point" of scenic integrity. This perception is unlikely to be significantly mitigated by public education (Bliss 2000; Hansis 1995). However, Ribe (2006) found that informed acceptability perceptions of 15 percent dispersed retention harvests were substantially higher, albeit still a bit negative, than for clearcuts. This suggests that where ecological or economic goals call for cutover areas within cutblocks, if a few trees are left standing there (and remain so until greenup), then informed public observers may see these as evidence of visible forest stewardship, even if they otherwise see such areas as ugly.

Guidance for Predicting Average Visual Impacts of Forestry Project Designs

Standard SMS practice prescribes that if a forestry project will produce a visual change that conforms to the "expected or desired" standard at its location, then it is assessed as having a "low" visual impact. If it is predicted to fall one integrity level short of the local standard, it is assessed as a "high" impact. If it is expected to produce a visual change more than one level below the local standard it is assessed as having a "very high" impact. If projects may exceed the standard they are typically assessed as a "very low" impact. Average, evidence-based, estimated scenic integrity and visual impact levels achieved by forest and harvest structures described above are summarized in table 1. The SMS-derived visual impact of these designs will depend on the scenic integrity level standard set for the corresponding area of landscape, as described earlier.

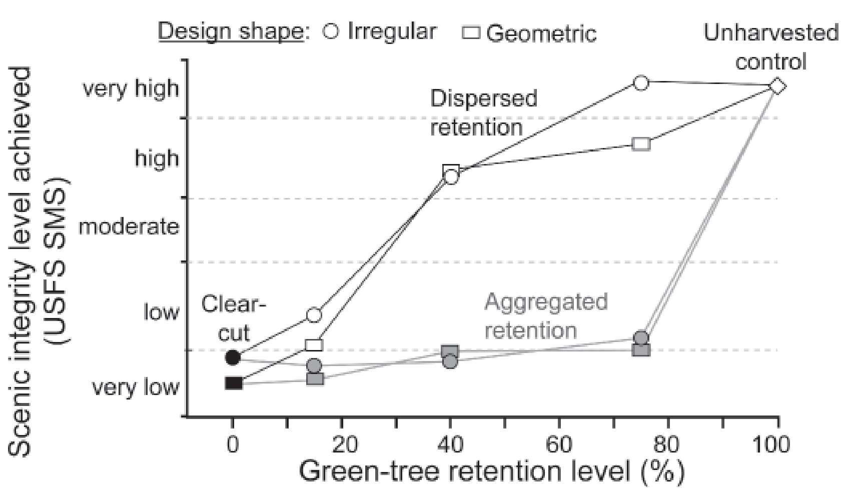

Figure 8—The relationship between average, vista-view scenic beauty ratings of forests and regeneration harvests interpreted by scenic integrity levels, as sorted by retention level, retention pattern, and harvest unit shape-type. Adapted from Ribe (2005a).

Table 1—Summary of evidence-based guidelines for estimating scenic integrity levels achieved by different forest conditions and regeneration harvest prescriptions; followed by a guide to corresponding visual impacts as related to project site scenic integrity standards. These can be used to predict expected average perceived integrity levels for the average west-side forest in assessments of multiple harvests in forest plans or for single, not-visually-controlled harvests. Levels can be improved when specific projects are designed or contextually assessed by landscape architects.

Designing Forests to Meet Scenic Integrity Standards					**Predicting Estimated Average Visual Impacts**				
If one or more of these criteria are met, then the integrity level in the left column is estimated to be met for an average forest seen by an average member of the public.					Estimated visual impact resulting if a project area's scenic integrity standard is a column header and the forest meets criteria in a row in the left matrix.				
	In-stand Views			**Vista Views**	**Scenic Integrity Standard**				
Scenic integrity level achieved	**Forest density (stems >5 $cm \cdot ha^{-1}$)**	**Basal area ($m^2 \cdot ha^{-1}$)**	**Forest type or harvest design**	**Forest type or harvest design**	**Very low**	**Low**	**Moderate**	**High**	**Very high**
Very high	700-1000	120-170	Old-growth forest	75-100% disp ret 95-100% agg ret	very low	very low	very low	very low	low
High	700-1000	50-120 or >170	Mature forest or: 40-100% disp ret, low dn wd; 60-100% disp ret, high dn wd; 60-100% agg ret, low dn wd; 75-100% agg ret, high dn wd	35-75% disp ret; 90-95% agg ret	very low	very low	very low	low	high
Moderate	400-700 or 1000-1500	30-50	20-40% disp ret, low dn wd; 35-60% disp ret, high dn wd; 30-60% agg ret, low dn wd; 55-75% agg ret, high dn wd	25-35% disp ret; 85-90% agg ret	very low	very low	low	high	very high
Low	100-400 or >1500	15-30	2-20% disp ret, low dn wd; 5-35% disp ret, high dn wd; 3-30% agg ret, low dn wd; 10-55% agg ret, high dn wd	15-25% disp ret, rectilinear shape; 0-25% disp ret, designed shape; 40-85% agg ret, any shape	very low	low	high	very high	very high
Very low	<100	0-15	0-2% disp ret, low dn wd; 0-5% disp ret, high dn wd; 0-3% agg ret, low dn wd; 0-10% agg ret, high dn wd	0-15% disp ret, rectilinear shape; 0-40% agg ret, any shape (designed shape might help 0-15%)	low	high	very high	very high	very high

Discussion

Designing forest treatments and regeneration harvests is always complex and fraught with conflicting goals and trade-offs. These goals are rooted in social values often instigated by aesthetic experiences and fundamentally understood by aesthetic sensibilities (Grob1995). The search for good forestry decisions can be understood as a search for "elegant" solutions that optimally achieve and express physical, biological, perceptual, and social functions. In a sense, such decisions can aesthetically transcend particular technical goals, biological criteria, or narrowly conceived social and aesthetic values. This is arguably a goal of ecosystem management, but this paradigm could be more explicitly attentive to broadly conceived aesthetics as they affect public perceptions (Lessard 1998). Forests that express ecological health by emulating ecological processes and attributes might gain public approval (Olsen et al. 2012), even when not of high scenic quality, by virtue of exhibiting "visible stewardship" (Sheppard 2001) to gain perceptions of their "ecological aesthetic" quality (Gobster 1999).

The challenge for forest managers is to design forest treatments that express visible stewardship and also public education programs that broaden public appreciation of ecological aesthetics among more people. If agency claims of ecological aesthetic value, as opposed to scenic value, are to gain legal validity, they have to be similarly perceived by at least a majority of the relevant public. Meanwhile, naturalistic scenic aesthetics are already similarly perceived by most people (McCool et al. 1986; Magill 1992), and are required by law and established visual impact assessment standards and methods.

The best approach seems to be the simultaneous pursuit of a mix of forests with naturalistic scenic quality and ecological aesthetic qualities (especially both at once), along with education programs seeking to increase appreciation of the latter. Over time, the socially acceptable "recipe" of these two aesthetic types may shift toward more ecological aesthetics, but the public's desire for naturalistic scenic quality is powerful, indeed likely hard-wired, and will not go away (Parsons and Daniel 2002; Thompson 2011). Managers should attend to the creation of ecological features like micro-habitats for insects, amphibians, and fungi, forests that serve as refugia or riparian or interior habitats, key habitat elements for wide-ranging sensitive vertebrate species, or the stabilization and enrichment of soil. These efforts should also pay attention to locally applicable scenic integrity standards and favor project designs, whenever possible, that tend to achieve these in the ways described above. Where scenic integrity standards are less constraining on forest attributes, projects should often, and whenever possible, seek to avoid crossing the scenic "tipping points" described above to maintain broad-trust public license to manage forests.

Acknowledgments

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A Satellite's View of Recent Trends in Forest Harvest Intensity in the Pacific Northwest

Robert E. Kennedy, Zhiqiang Yang, Justin Braaten, Warren B. Cohen, Peder Nelson, and Eric Pfaff

Extended Abstract

As a silvicultural treatment, forest harvest through thinning may serve multiple management goals at the site level. To the extent that these goals affect processes or species populations that span landscapes, their cumulative impacts can only be assessed if landscape-wide patterns of thinning are monitored. This is particularly relevant when policy or management patterns change abruptly for large land ownerships, as was intended with the implementation of the Northwest Forest Plan (NWFP) in 1994.

Satellite images provide consistent views of large landscapes and thus have the potential to monitor forest harvest, but these have typically focused only on stand-replacing harvest. In the Pacific Northwest, imagery from the Landsat family of satellites has long been used to quantify such harvest (Cohen et al. 2002), to estimate its impacts on carbon flux (Cohen et al. 1996), and to evaluate its potential impacts on critical habitat (Healey et al. 2008). Mapping partial harvest, however, is more challenging. If the approximate location of such harvest is known, the mathematical signal of partial harvest is discernible in the signal of the imagery (Healey et al. 2006), but the challenge is separating that relatively subtle signal from background noise.

Recent improvements in data access, computing power, and algorithms, however, have led to more robust mapping of partial harvest. The US Geological Survey recently opened the Landsat data archive to free access (Woodcock et al. 2008), and also simplified its delivery of the imagery, making management of large data sets practical. Combined with new algorithms that tap this archive (Kennedy et al. 2010), separation of partial harvest from background noise is now both mathematically and logistically tractable (fig. 1).

In support of the effectiveness monitoring for the NWFP (*http://www.reo.gov/monitoring/*), we developed yearly maps of forest disturbance, including partial harvest, for the entire NWFP area for the period from 1985 to 2007 (Kennedy et al. 2012). These maps are available in digital form at *http://landtrendr.forestry.oregonstate.edu/content/download-data*.

For the purposes of evaluating partial harvest patterns, these data provide information on the estimated magnitude of vegetation removal occurring in each 30-m cell in the map. These magnitude quantities are satellite-based reflections of reality and thus are not strictly biophysical quantities that equate precisely with proportional basal-area removal or other similar quantities familiar to silviculturists. In particular, the quantities are expected to have different biophysical

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Robert E. Kennedy is assistant professor, Department of Earth and Environment, Boston University, 675 Commonwealth Ave., Boston, MA 02215; kennedyr@bu.edu; **Zhiquiang Yang** is a research associate and **Justin Braaten** is a faculty research assistant, Department of Forest Ecosystems & Society, Oregon State University, Corvallis, OR 97331; **Warren B. Cohen** is a research forester, USDA Forest Service, Pacific Northwest Research Station, Forestry Sciences Laboratory, 3200 SW Jefferson Way, Corvallis, OR 97331; **Peder Nelson** is a senior faculty research assistant, and **Eric Pfaff** is a faculty research assistant, Department of Forest Ecosystems & Society, Oregon State University, Corvallis, OR 97331.

meaning in different forest types (e.g., Douglas-fir vs. Lodgepole Pine [*Pseudotsuga menziesii*, *Pinus contorta*]). Nevertheless, within a given forest type and region, they have utility to distinguish among harvests of different severities.

Grouping these data into relative intensity classes of high, medium, and low, and separating by ownership, temporal trends in harvest practice over time can be documented. In the Cascades province of the NWFP in Oregon, for example (fig. 2), overall area harvest fell after court injunctions in the early 1990s, and the harvest intensity since then has been focused on lower-intensity harvest types. Private lands, on the other hand, saw an eventual rise in harvest rate, with an increasing proportion of that harvest being high-intensity harvest. Similar summaries can be replicated over the entire NWFP area.

Keywords: Thinning, satellite mapping, Northwest Forest Plan.

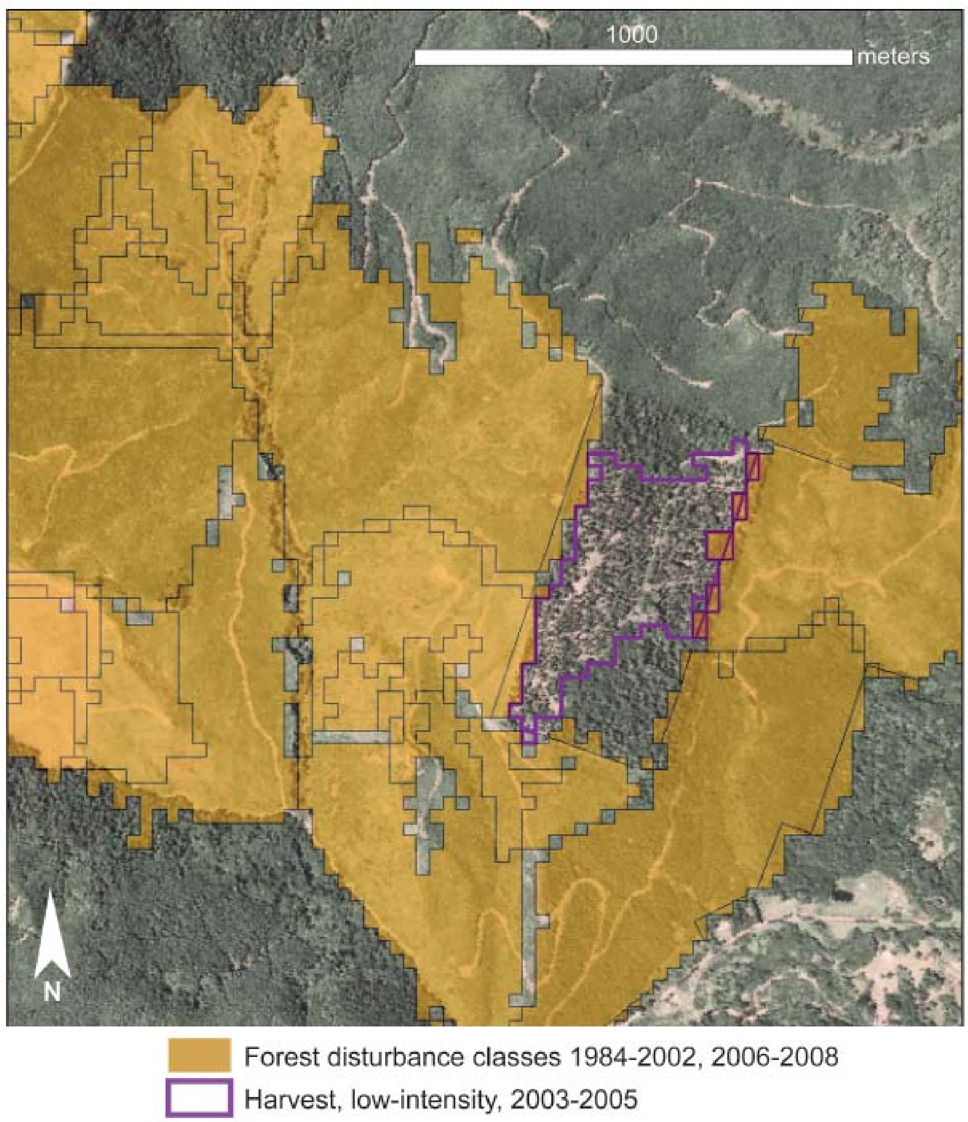

Figure 1—A partial harvest (purple outline) within a matrix of prior clearcuts (yellow-shaded areas), overlaid on a high-resolution air photo for reference. LandTrendr algorithms applied to 30-m pixels of Landsat imagery were used to detect disturbance in individual cells, which were then aggregated into a single patch delineated here.

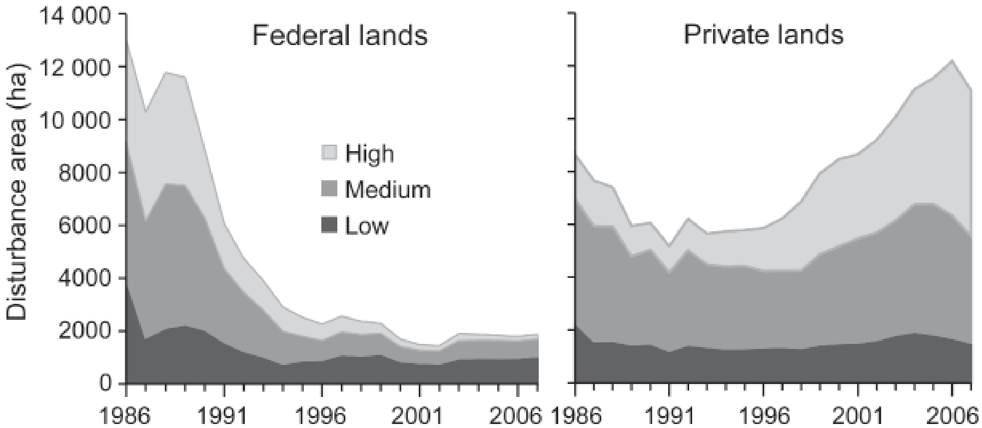

Figure 2—Harvest rates over time by relative intensity in federal (a) and private (b) ownership classes in the Cascades province of Oregon in the NWFP area. As expected, harvest rates fell on federal lands in the early 1990s, with a greater proportion of harvest in low and medium intensity classes. On private lands, rates increased from the mid 1990s onwards, with an increasing proportion of that rate occurring in higher-intensity harvest.

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Landscape Context for Density Management: Implications of Land Ownership and Ecological Gradients

Janet L. Ohmann

Presentation Abstract

Density management is implemented at a local (stand) scale, but is based on conservation goals that address a broader landscape. Although regional conservation efforts such as the Northwest Forest Plan (NWFP) focus primarily on public lands, all land ownerships and allocations contribute unique benefits over the regional landscape that need to be considered as context for federal land management. In addition, whereas the NWFP emphasizes older forest and associated species, an ecosystem and multi-species management approach encompasses the full sequence of forest development. Recent results from NWFP Effectiveness Monitoring for late-successional and old-growth forest provide detailed maps of forest composition and structure over all ownerships, and for all stages of forest development, including trends over the NWFP period (1994 to present). More than two-thirds of older forest (overstory conifers >20 inches [50.8 cm]) was federally owned, whereas most early-seral forest was on nonfederal lands. However, early-seral stands have developed primarily following timber harvest and lacked the structural diversity typical of natural stands. Over the NWFP period, harvesting removed about 13 percent (491,000 acres [198,700 ha]) of older forest on nonfederal lands, whereas the loss of older forest from federal lands (about 200,000 acres [80,937 ha]) was attributed primarily to wildfire. Overall, the monitoring data suggest there was a slight net loss of older forest on federal lands since the beginning of the NWFP, so losses of older forest to wildfire apparently were roughly balanced by recruitment. However, recruitment was most likely through incremental stand growth over the 20-inch [50-cm] threshold, or from understory disturbances (including thinning) that eliminated smaller-diameter trees and increased average stand diameter, rather than from increases in stands of much larger and older trees. Land ownerships and allocations also were unevenly distributed across regional environmental gradients. Federal forests generally occupied higher elevations and lower-productivity sites, and a substantial portion of most of the mid- to high-elevation forest types were contained within reserves and managed for conservation objectives. In contrast, several forest types, such as oak woodlands, occurred predominantly at low elevations on nonfederal lands where they are managed for a variety of objectives, and are not well-represented in reserves. I will discuss potential implications of the monitoring results for stand-level management and for the conservation of older forests and associated species across the region.

Keywords: Old growth, late-successional forest, early-successional forest, forest monitoring, change detection, Pacific Northwest, Northwest Forest Plan, conservation planning.

Janet L. Ohmann is a research forester, USDA Forest Service, Pacific Northwest Research Station, Forestry Sciences Laboratory, 3200 SW Jefferson Way, Corvallis, OR 97331; johmann@fs.fed.us

Photos, facing page—Top left: Leave island at Green Peak, photo by Paul Anderson. **Top right:** Regeneration at the Yachats STUDS site, Siuslaw National Forest. Photo by Paul Anderson. Center left: Chanterelles, *Cantharellus* sp. Photo by Paul Anderson. **Bottom left:** Dan Mikowski, USFS, at North Soup Creek, Feb. 2009. Photo by Paul Anderson. **Center right:** Patch opening at Keel Mountain, April 2005. Photo by Paul Anderson. **Bottom right:** Two Oregon Slender Salamanders (*Batrachoseps wrighti*) and a rarely-seen nest of this species in a down log. Photo by Loretta Ellenburg, USFS.


Section 2: Implementation and Influences of Density Management in the Terrestrial Ecosystem




Short-term Responses of Overstory and Understory Vegetation to Thinning Treatments: A Tale of Two Studies

Klaus J. Puettmann, Erich K. Dodson, Adrian Ares, and Carrie A. Berger

Abstract

The Density Management Study and Young Stand Thinning and Diversity Study were initiated to investigate whether alternative thinning treatments can accelerate the development of forests toward late-successional structures. An overview of overstory and understory vegetation responses indicates that the magnitude and direction of thinning effects initially varied among structural stand components. Average crown length, diameter growth, and seedling and sapling density increased rapidly after thinning. However, diameter growth of the largest trees was less responsive to thinning, at least in the short term. Overall shrub cover was initially reduced by thinning, which was likely a consequence of harvesting damage. Thinning tended to increase overall species diversity by benefitting early-successional species, and had little influence on late-successional species diversity. This trend appeared to reverse itself quickly, as early-seral vegetation declined within a decade. It appears that silvicultural treatments may best target specific stand structural components, e.g., large trees, crown length, or an understory shrub layer, rather than attempting to achieve all late-successional structural elements simultaneously. Increasing variability at smaller and larger spatial scales may be an option to achieve multiple management goals.

Keywords: Thinning, late-successional habitat, Douglas-fir, overstory, understory vegetation, regeneration.

Introduction

The dispute about management of federal forest land resulted in the Northwest Forest Plan in 1994, which emphasized development of late-successional habitats in a context of sustainable wood production. Concurrently, new research indicated that many old-growth stands in western Oregon had originated at densities that were substantially lower than densities found in current Douglas-fir (*Pseudotsuga menziesii*) plantations (Tappeiner et al. 1997). The effect of thinning on tree growth and timber production had been well researched (e.g., Marshall and Curtis 2002). Lacking, however, was information on the potential for thinning (or uneven-age management) to balance wood production with ecological concerns, such as wildlife habitat and biodiversity. Of specific interest to federal foresters were forest structures associated with late-successional habitat features (Bauhus et al. 2009), such as large trees, a layered canopy, advanced regeneration, and well-developed

Klaus J. Puettmann is Edmund Hayes Professor in Silvicultural Alternatives, Oregon State University, Department of Forest Ecosystems and Society, Corvallis, OR, 97331; klaus.puettmann@oregonstate.edu; **Erich K. Dodson** is a faculty research assistant, Dept. of Forest Ecosystems & Society, Oregon State University, Corvallis, OR, 97331; **Adrian Ares** is project director, SANREM CRSP (Sustainable Agricultural and Natural Resource Management Collaborative Research Support Program) Office of International Research, Education and Development (OIRED) Virginia Tech, 526 Prices Fork Road Blacksburg, VA 24061; **Carrie A. Berger,** 7 Rue Dimitri Ignatieff, Lotiss CPS Receiving, Noumea, Nouvelle Calédonie 98800.

understory layers (Old Growth Definition Task Group 1986). As a consequence, several studies were initiated in the western portion of the Pacific Northwest region to investigate whether thinning young, even-aged stands dominated by Douglas-fir can accelerate the development of late-successional forest structures (Poage and Anderson 2007). We present results from the first decade of two Large-scale Management Studies (Monserud 2002): 1) the Density Management Study (DMS); and 2) the Young Stand Thinning and Diversity Study (YSTDS), which were established in the Coast Range and western Cascades of Oregon.

The main objective of this paper is to present a brief overview of findings regarding the vegetation response to various thinning treatments applied in the two studies. Specifically, we present responses of overstory trees, including tree and stand growth patterns, tree mortality, and overstory canopy closure. Understory responses include seedling and sapling densities, as well as cover and composition of understory vegetation, separated by life-form (tall shrubs, low shrubs, herbs, and bryophytes) and origin (exotic species). We present up-to-date findings from the two studies, including previously unpublished results. Findings already published are highlighted by referring to the appropriate publications, where detailed findings, methods, and discussions can be found.

Methods

Study Sites and Treatments

To represent the variety of conditions in the Bureau of Land Management (BLM) ownership in Oregon, the DMS includes study sites in the Coast Range and western Cascades of Oregon. Study details are described in Cissel et al. (2006). Seven 40- to 60-year-old stands (between 100 and 150 ha in size) that had never been commercially thinned were selected, and each stand received one complete set of treatments. Treatment units ranged from 15 to 40 ha and included an unthinned control treatment (approximately 600 trees per hectare [tph]), a high-density retention (around 300 tph), a moderate-density retention (200 tph) and a variable retention which included areas with 100, 200, and 300 tph. For interpretation of average treatment conditions, it is important to note that 4 to 18 percent of the unit in all thinning treatments was reserved in circular leave islands (0.1-, 0.2-, and 0.4-ha). In addition, 4 to 18 percent of the moderate- and variable-density treatments were cut as circular gap openings (0.1-, 0.2-, and 0.4-ha). Units with 100 tph and the gap openings were planted with a mix of conifer species. Thinning was implemented between 1997 and 2000, and data described in this report were taken 6 and 11 years after the thinning operation. More detailed results from these studies can be found in Fahey and Puettmann (2007, 2008), Wilson and Puettmann (2007), Kuehne and Puettmann (2008), Ares et al. (2009, 2010), Wilson et al. (2009), Dodson et al. (2012), and Berger et al. (2012).

The YSTDS is located on the McKenzie and Middle Fork Ranger Districts of the Willamette National Forest on the western slopes of the Oregon Cascade Range. Four stands between the ages of 35 and 50 years that were dominated by even-aged Douglas-fir were selected. Each stand received a set of thinning treatments, which included an unthinned control (C; approximately 700 tph), a heavy thin (HT; 125 tph), a light thin (LT; 250 tph), and an additional light thin, in which 20 percent of the unit was cut in circular 0.2-ha gaps (LG). The heavy thin and the gaps in the light thin with gaps were planted with a mix of conifer species. The thinning was implemented between 1994 and 1997 (mostly 1995 and 1996) and the data described in this report are from measurements 1, 3, 5, and 10 years after thinning. More detail about the study establishment can be found in Manning and Friesen (in prep.[1]). Detailed results from this study are provided by Berger and Puettmann (2007), Davis et al. (2007), Kuehne and Puettmann (2008), and

Davis and Puettmann (2009).

Both studies generally used a thinning from below, but hardwoods and other minority species were maintained wherever possible. Conditions such as slope and aspect varied widely among the study sites and treatment units, and also within the treatment units. This resulted in large within-treatment variation in tree densities and other conditions (Cissel et al. 2006). The studies were implemented using operational timber sales, and variation in terrain and logging operators led to the use of both cable and ground-based harvest systems within several individual treatments. This increased the variability within individual treatment blocks compared to the homogeneity that can be achieved in small experimental units. However, this also means that the study includes many issues associated with operational treatments and results do not have to be scaled up, as is commonly done with small experimental plots.

Measurements and Analysis

The DMS sampling protocol is presented in Cissel et al. (2006). The sampling design for over-and understory vegetation of the YSTDS is presented in Davis et al. (2007) and Davis and Puettmann (2009). For the presentation below, mean values of each treatment unit in the YSTDS were calculated by averaging the plot means, with the exception of the treatment units that included gaps. A weighted average of sub-treatment (gap, edge, interior forest) means was used to calculate these treatment unit means. Weights for each sub-treatment were based on the proportion of areas in each sub-treatment to total treatment units (Davis et al. 2007).

The statistical analysis of the YSTDS data presented below used ANOVAs with a randomized complete block model in conjunction with the Tukey-Kramer adjustment for all multiple comparisons (PROC GLM). Data aggregated to the treatment unit level (n = 16 treatment units) were approximately normal; therefore, no transformations were performed. Significance level in the analysis and figures below indicate $P \leq 0.05$. Analysis procedures for the DMS varied, but are described in detail in the publications.

Results and Discussion

Overstory Responses

Stand growth patterns showed inconsistent responses. Overall, basal area after thinning was reduced in both studies. Basal area growth for the measurement interval from 5 to 10 years after thinning in the YSTDS averaged 86, 73, and 88 percent of control for the Heavy, Light, and Light with Gap treatments, respectively (table 1). Note, that these values were not significantly different from each other, and the low basal area growth in the Light treatment is mainly due to a single site (Mill Creek). In the DMS study, basal area growth 6 to 11 years after thinning was not significantly different from the controls (Dodson et al. 2012). Such discrepancies may be due to a combination of variability in age, site quality, and associated ability of trees to respond to canopy openings. Douglas-fir can maintain stand growth under a relatively wide density range (Marshall and Curtis 2002), and little impact would be expected in the lower-intensity thinnings. Apparently, such effects in conjunction with site variability did not allow detection of statistically significant differences.

Diameter growth showed a consistent response in both studies. Average diameter growth of trees increased after thinning (table 2, also Dodson et al. 2012). This is due to a combination of the removal of slower-growing trees by thinning from below (i.e., the chainsaw effect), as well as improved growing conditions for residual trees. A more detailed analysis suggested that smaller trees especially benefitted from thinning by increasing their growth rate. The largest trees were growing faster prior to thinning and did not experience as large an increase in growth rates as smaller trees

[1]Manning, T.; Friesen, C., eds. [n.d.] The Young Stand Thinning and Diversity Study establishment report and study plan. Manuscript in preparation.

Table 1—Mean basal area ($m^2 \cdot ha^{-1}$) for all trees ≥8 cm dbh in post-treatment years 1, 3, 5, and 10. Data are from the Young Stand Thinning and Diversity Study.

		Mean basal area for all trees ≥8 cm dbh ($m^2 \cdot ha^{-1}$)			
Block	**Treatment**	1997	1999	2001	2006
Cougar Reservoir (CR)	Control	39.8	45.0	47.2	51.4
	Heavy thin	13.2	15.7	17.5	22.3
	Light thin	24.6	27.6	29.6	35.6
	Light thin with gaps	17.6	20.5	22.9	27.3
Mill Creek (MC)	Control	46.7	50.3	52.5	56.3
	Heavy thin	21.1	23.2	24.9	25.0
	Light thin	31.5	33.7	35.6	35.3
	Light thin with gaps	27.0	29.7	29.4	30.0
Christy Flats (CF)	Control	53.2	55.1	56.4	61.6
	Heavy thin	14.4	15.4	16.8	20.7
	Light thin	20.8	21.9	23.5	27.8
	Light thin with gaps	19.1	20.2	21.9	25.6
Sidewalk Creek (SC)	Control	38.7	39.7	41.0	41.5
	Heavy thin	11.8	13.3	14.6	17.6
	Light thin	20.4	22.0	23.4	25.4
	Light thin with gaps	15.4	16.8	18.0	19.4

Table 2—Mean dbh (cm) for all trees ≥8 cm dbh in post-treatment years 1, 3, 5, and 10. Data are from the Young Stand Thinning and Diversity Study.

		Mean dbh for all trees ≥8 cm dbh (cm)			
Block	**Treatment**	1997	1999	2001	2006
Cougar Reservoir (CR)	Control	24.3	25.8	26.8	28.5
	Heavy thin	31.1	31.6	32.5	33.1
	Light thin	29.0	30.7	31.8	33.7
	Light thin with gaps	29.8	31.3	32.2	31.9
Mill Creek (MC)	Control	26.8	28.7	29.9	31.4
	Heavy thin	26.9	27.8	28.5	30.3
	Light thin	27.8	28.9	29.6	30.9
	Light thin with gaps	26.9	27.5	27.9	28.4
Christy Flats (CF)	Control	25.5	27.2	29.1	30.3
	Heavy thin	35.2	36.6	38.7	40.6
	Light thin	32.8	34.6	37.0	41.1
	Light thin with gaps	32.6	35.2	37.2	39.9
Sidewalk Creek (SC)	Control	22.5	23.2	24.0	25.0
	Heavy thin	26.6	25.6	24.5	22.8
	Light thin	28.4	29.4	30.5	31.8
	Light thin with gaps	26.9	26.7	26.9	26.9

(Davis et al. 2007; Dodson et al. 2012). Quite likely, the largest trees did not experience a major improvement in growing conditions as part of the standard thinning operation. However, the more open growing conditions resulting after thinning will likely mean that growth rates of larger trees can be maintained longer than under unthinned conditions (Latham and Tappeiner 2002). As an indication of this trend, the largest trees responded with increased growth under very heavy thinnings (Davis et al. 2007) and when adjacent trees were removed as part of gap creation (Dodson et al. 2012).

In general, mortality rates were higher in the control treatments than in the thinning treatments (tables 3 and 4, Dodson et al. 2012). Most tree mortality in the controls was in the smaller diameter classes, suggesting competition-related mortality as the major factor. On the other hand, mortality rates in the thinned units were minor and did not show a size-related pattern (table 4), indicating that most trees were able to withstand the exposed conditions created by thinning. These results also imply that most snags in unthinned conditions are in smaller size classes, limiting their wildlife value. Even recruitment of small snags was reduced by thinning; thus current snag densities and expected mortality may not be sufficient to reach snag levels similar to those found in old-growth stands (Ares et al. 2012). Active snag creation may be necessary to achieve specific habitat goals, especially if these goals include larger snags (Dodson et al. 2012).

Overstory cover is an important measure of wildlife habitat, but for practical purposes overstory closure is often used as an indicator of canopy density (Jennings et al. 1999), as was done in the YTSDS (no overstory closure data were available for the DMS). Typical overstory canopy closure under unthinned control conditions in the YSTDS was around 80 percent (table 5, fig. 1). As expected, thinning reduced overstory closure to around 30, 50, and 42 percent of the controls in the Heavy, Light,

Table 3—Mean trees per hectare for all trees ≥8 cm dbh in post-treatment years 1, 3, 5, and 10. Data are from the Young Stand Thinning and Diversity Study.

		Mean trees per hectare, all trees ≥8 cm dbh (tph)			
Block	**Treatment**	1997	1999	2001	2006
Cougar Reservoir (CR)	Control	753	741	723	697
	Heavy thin	151	165	172	199
	Light thin	312	312	309	327
	Light thin with gaps	221	229	237	266
Mill Creek (MC)	Control	655	614	594	589
	Heavy thin	283	290	292	304
	Light thin	415	408	408	425
	Light thin with gaps	346	357	327	378
Christy Flats (CF)	Control	869	791	717	724
	Heavy thin	133	131	129	139
	Light thin	207	198	188	185
	Light thin with gaps	1498	184	178	180
Sidewalk Creek (SC)	Control	792	771	747	733
	Heavy thin	165	191	216	288
	Light thin	277	279	276	290
	Light thin with gaps	225	241	244	267

Table 4—Conifer and hardwood (≥5 cm dbh) percent mortality by block and treatment. Data were collected in 2006, ten years post-treatment and are from the Young Stand Thinning and Diversity Study.

		Mortality of trees ≥5 cm dbh (percent)	
Block	**Treatment**	Conifers	Hardwoods
Cougar Reservoir (CR)	Control	4.0	10.6
	Heavy thin	0.3	0.0
	Light thin	1.1	0.0
	Light thin with gaps	1.6	1.1
Mill Creek (MC)	Control	2.2	4.0
	Heavy thin	1.3	4.8
	Light thin	0.5	0.9
	Light thin with gaps	1.3	0.4
Christy Flats (CF)	Control	4.5	33.3
	Heavy thin	0.9	0.0
	Light thin	1.5	3.0
	Light thin with gaps	0.0	4.8
Sidewalk Creek (SC)	Control	1.4	2.5
	Heavy thin	0.0	0.0
	Light thin	0.9	0.0
	Light thin with gaps	0.4	0.5

Table 5—Mean percent overstory cover in post-treatment years 1, 3, 5, and 10. Data are from the Young Stand Thinning and Diversity Study.

		Overstory cover (percent)			
Block	**Treatment**	1997	1999	2001	2006
Cougar Reservoir (CR)	Control	82	83	89	85
	Heavy thin	25	37	37	51
	Light thin	50	62	72	73
	Light thin with gaps	40	47	55	60
Mill Creek (MC)	Control	82	76	76	85
	Heavy thin	46	54	60	62
	Light thin	62	68	75	77
	Light thin with gaps	50	57	62	71
Christy Flats (CF)	Control	79	79	71	80
	Heavy thin	17	22	27	30
	Light thin	37	44	46	56
	Light thin with gaps	36	44	45	54
Sidewalk Creek (SC)	Control	70	77	73	76
	Heavy thin	31	27	36	46
	Light thin	53	58	63	62
	Light thin with gaps	42	46	49	54

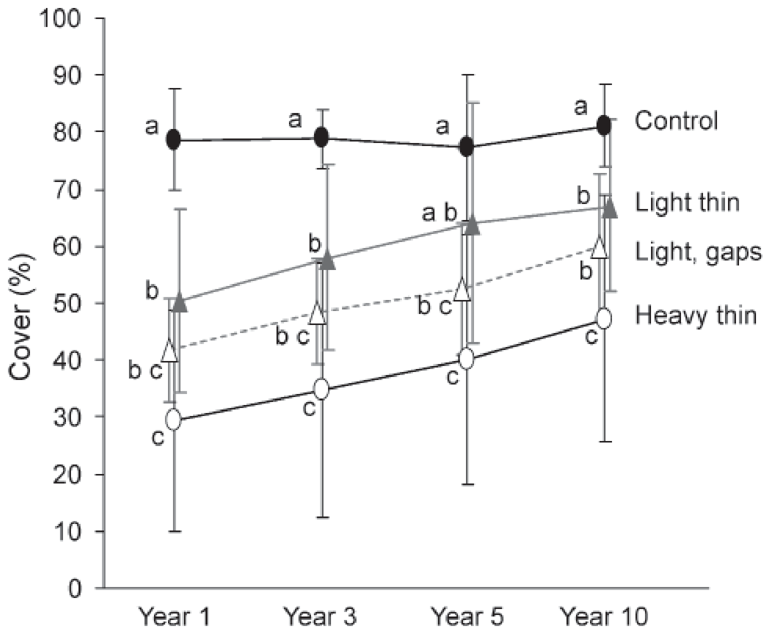


Figure 1—Overstory cover (percent) by year (includes 95 percent confidence intervals). Letters indicate significant differences among treatments (treatments with the same letter do not differ at P ≤ 0.05 level).

and Light with Gap treatments, respectively. However, the overstory recovery was quite rapid in the first decade, with average canopy closure increasing from 1 to 2 percent per year (fig. 2), a value similar to other studies in the region (e.g., Chan et al. 2006). Similarly, crown lift stopped fairly quickly after thinning (Chan et al. 2006; Davis et al. 2007). Even though overstory closure is still significantly lower 10 years after thinning, the trends suggest that long-term maintenance of open canopy conditions will require repeated thinning operations.

Tree Regeneration

Partial harvests, such as thinning, provide opportunity for establishment of a new cohort of trees through increased vigor, and thus seed production by residual trees (Reukema 1961), exposure of mineral soil that provides seed beds, and resource availability for seedlings (Nyland 2002). Generally, natural regeneration was significantly more abundant after thinning than in control treatments in both studies (fig. 3, Kuehne and Puettmann 2008; Ares et al. 2010; Dodson et al. 2012). Seedling establishment likely increased due to a combination of seedbed preparation through harvesting disturbances and increased resource availability as a result of the removal of overstory trees and damaged understory vegetation. However, short-term effects on tree regeneration did not differ among thinning treatments (Kuehne and Puettmann 2008), but the variability within treatments and among replications of treatments was high.

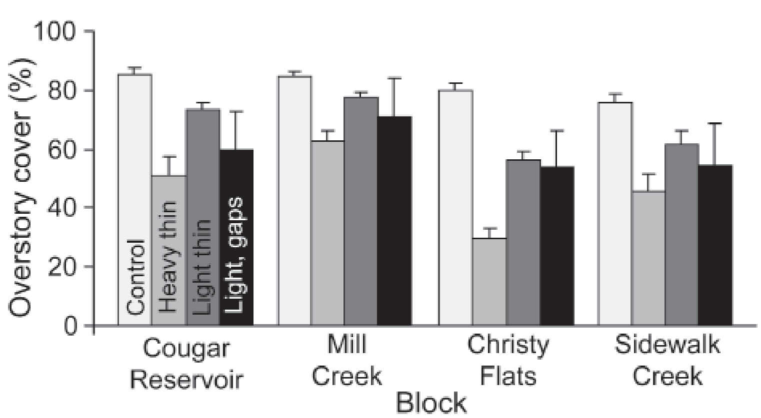


Figure 2—Ten-year post-treatment mean percent overstory cover by block (includes standard error bars).

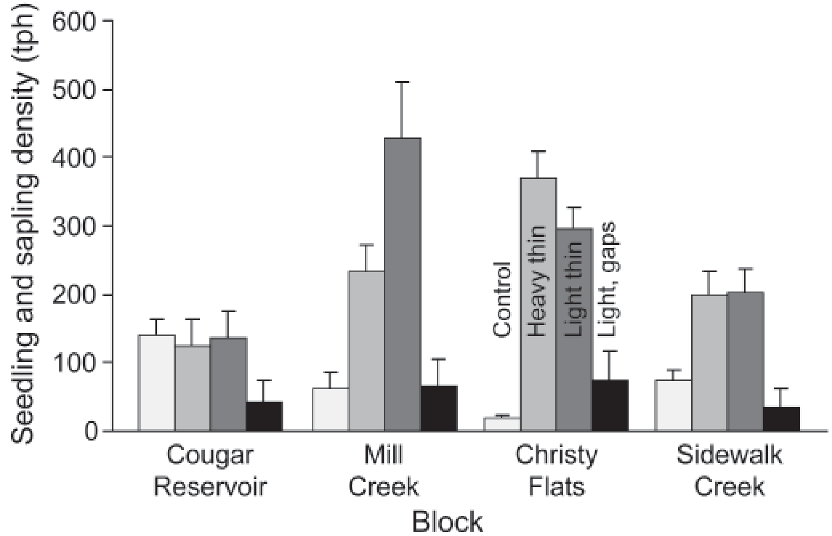


Figure 3—Ten-year post-treatment mean seedling and sampling density (tph) by block (includes standard error bars).

In some plots, the regeneration density was sufficiently high to initiate a discussion about future pre-commercial thinning (Cissel et al. 2006).

Saplings also showed higher densities in thinned units than in controls 11 years following thinning (fig. 3, Dodson et al. 2012). They likely were established prior to implementation of thinning treatments, and interpretation of these numbers needs to consider that at least some saplings were damaged during the harvesting operation (Dodson et al. in prep.[2]). Since saplings were not influenced by thinning impacts on seedbed conditions, the growing environment in thinned units appears more conducive than in unthinned controls. This has also been shown by greater growth of some species under less dense conditions (Shatford et al. 2009). The patterns in in-growth also suggest that differences in overstory density among thinning treatments will be reflected in the diameter distributions in the long run, with higher seedling and sapling densities in the more intensive thinning treatments (Dodson et al. 2012; Dodson et al. in prep.[2]).

The dominant species on the study sites (usually Douglas-fir, and in some cases Western Hemlock, *Tsuga heterophylla*) were also dominant in the seedling and sapling pool. Species richness of seedlings was higher in thinned units, and the spatial patterns suggest that the presence of "minority" species in the overstory is also reflected in the seedling pool (Kuehne and Puettmann 2008). Differential growing patterns, especially in response to competition from overstory trees, suggest that the density of light-demanding species such as Douglas-fir may decline over time (Shatford et al. 2009) unless repeated overstory thinning maintains open growing conditions. Thinning operations also damage regeneration, which can become problematic when the density of regenerating seedlings is low, e.g., in understory plantings (Newton and Cole 2006). However, five years after a second thinning, density values of natural tree regeneration were similar to units that had not been thinned a second time (Berger et al. 2012). In terms of seedling densities, the damage due to the harvesting operation during this treatment was apparently balanced out by new recruitment.

Understory Vegetation

Responses to thinning varied among understory life-forms. Tall shrubs showed a large decline right after thinning in the YSTDS to less than half of cover values found in control

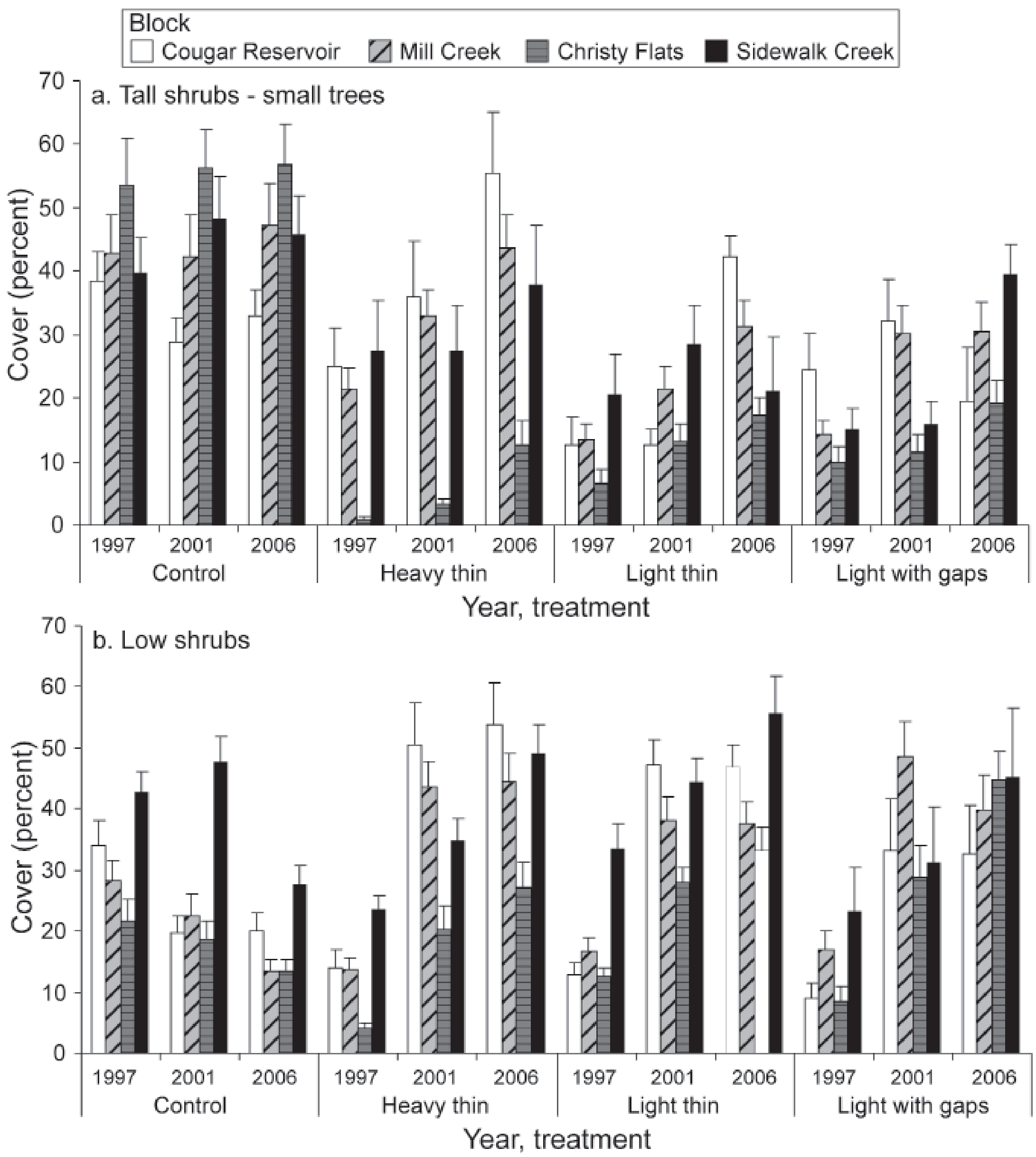
[2]Dodson, E.K.; Ares, A.; Burton, J.; Puettmann, K.J. [n.d.] Natural regeneration after thinning Douglas-fir stands in western Oregon. Manuscript in preparation for the Canadian Journal of Forest Research.

units (figs. 4a and 4b, table 6a and 6b). Since we do not have any evidence to suggest a bias in pre-thinning conditions, it appears that the harvesting operation led to physical damage of shrubs (Davis and Puettmann 2009; Dodson and Puettmann, unpublished data). Tall shrub cover was not significantly different on thinned units after a decade, suggesting that some recovery occurred. However, values remained lower than on control units.

A meta-analysis of five thinning studies in the region suggests that thinning does not consistently reduce shrub cover (Wilson et al. 2009). Instead, thinning reduced shrub cover only when shrub cover is initially high (>30 percent cover), with little effect in stands not dominated by shrubs, suggesting that mechanical damage occurs during the harvesting operations.

In contrast to shrubs, cover of herbaceous vegetation did not experience a post-harvest decline (Wilson et al. 2009, fig. 4c, table 6c). Instead, herbaceous cover increased quickly

Figure 4—Mean percent cover for a) tall shrubs/small trees, b) low shrubs, c) herbs, and d) bryophytes in post-treatment years 1, 5, and 10.



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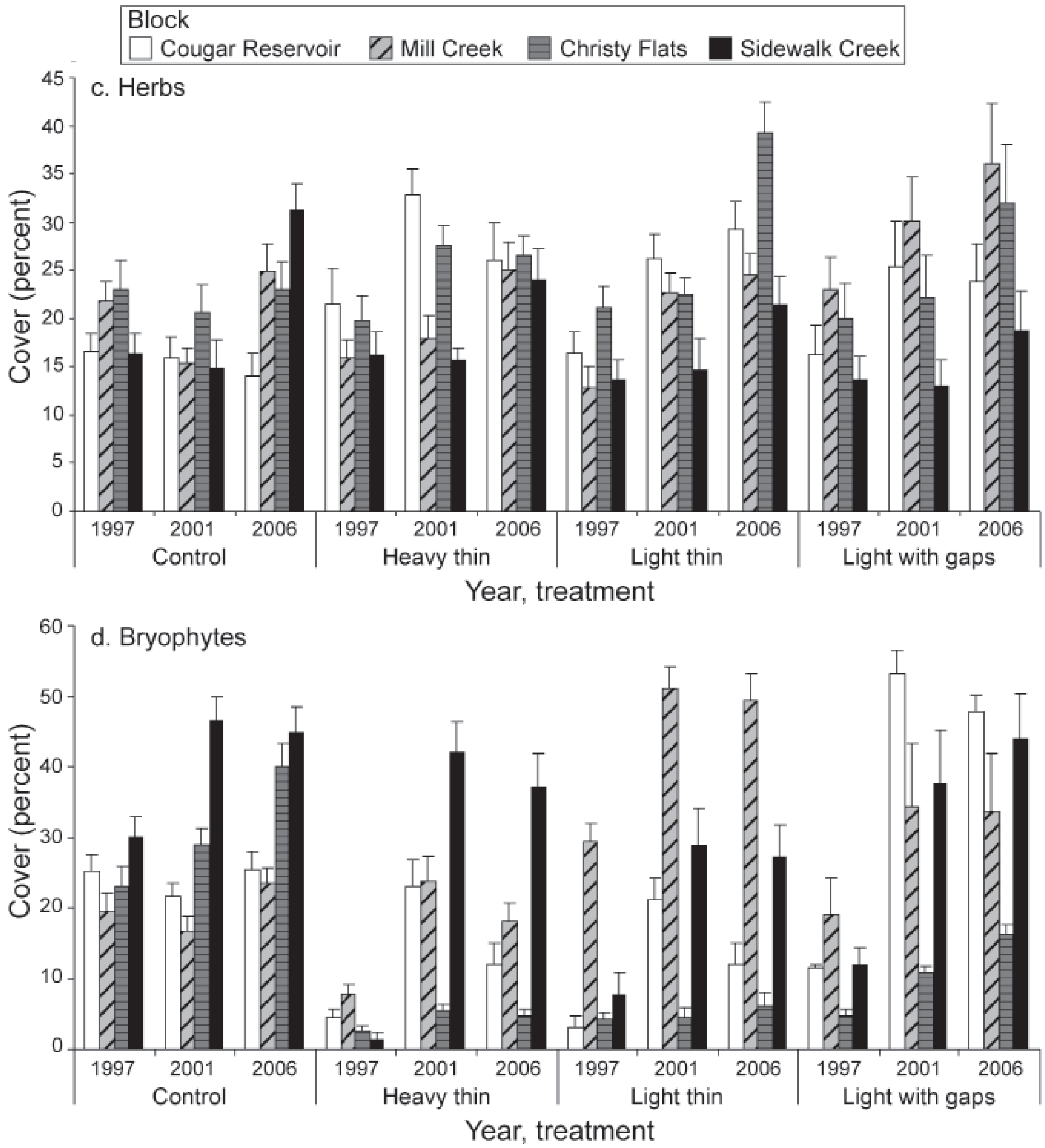

after thinning, apparently benefiting from reduced competition (Ares et al. 2009). Further increases of herbaceous cover may be limited due to increased competition, as shrubs and trees respond to thinnings. Similar to herbs, bryophyte ground cover showed an initial increase, likely covering ground that was exposed during the harvesting operation (Davis and Puettmann 2009, fig. 4d, table 6d). Over time, ground cover of bryophytes increased slightly in the YSTDS, but differences among treatments were not statistically significant. Bryophyte cover in the DMS study was significantly lower in the heavily thinned treatments than controls a decade after thinning (Ares et al. 2010).

A decade after thinning, other categories of ground cover, such as bare soil, stumps, rocks, or logs, typically covered less than 5 percent of the area in the DMS. Treatment patterns were hard to detect, but duff layers still appear to be lower in thinned units than in controls (Ares et al. 2009). The absolute cover value of litter was higher in controls, with cover in thinned units being approximately two-thirds of control value

Table 6—Mean percent cover for a) tall shrubs/small trees, b) low shrubs, c) herbs, and d) bryophytes in post-treatment years 1, 3, 5, and 10. Data are from the Young Stand Thinning and Diversity Study.

a. Tall Shrubs/Low Trees		**Tall shrub/small tree understory cover (percent)**				
Block	**Treatment**	1997	1999	2001	2006	Trend
Cougar Reservoir (CR)	Control	38.4	35.2	28.7	32.9	↓↑
	Heavy thin	25.0	36.2	36.1	55.5	↑
	Light thin	12.8	16.2	12.7	19.6	↑
	Light thin with gaps	24.3	31.0	32.0	42.3	↑
Mill Creek (MC)	Control	42.8	44.9	42.2	47.3	↑
	Heavy thin	21.3	32.2	33.0	43.7	↑
	Light thin	13.4	20.8	21.4	30.4	↑
	Light thin with gaps	14.2	22.5	30.1	31.3	↑
Christy Flats (CF)	Control	53.5	54.6	56.2	56.9	↑
	Heavy thin	0.9	2.9	3.3	12.7	↑
	Light thin	6.5	12.4	13.2	19.1	↑
	Light thin with gaps	9.9	11.7	11.6	17.2	↑
Sidewalk Creek (SC)	Control	39.8	46.0	48.4	45.8	↑
	Heavy thin	27.4	36.3	27.4	37.9	↑
	Light thin	20.6	31.3	28.6	39.4	↑
	Light thin with gaps	15.0	16.2	16.0	21.2	↑

b. Low Shrubs		**Low shrub understory cover (percent)**				
Block	**Treatment**	1997	1999	2001	2006	Trend
Cougar Reservoir (CR)	Control	34.1	29.5	19.6	20.1	↓
	Heavy thin	13.9	36.5	50.6	53.8	↑
	Light thin	9.0	22.5	33.4	32.8	↑
	Light thin with gaps	13.0	28.9	47.2	46.8	↑
Mill Creek (MC)	Control	28.4	23.1	22.6	13.5	↓
	Heavy thin	13.8	37.0	43.7	44.6	↑
	Light thin	16.9	30.6	48.5	39.9	↑
	Light thin with gaps	16.8	25.3	38.0	37.6	↑
Christy Flats (CF)	Control	21.7	16.1	18.8	13.4	↓
	Heavy thin	4.1	10.3	20.2	27.1	↑
	Light thin	8.4	18.5	28.9	44.8	↑
	Light thin with gaps	12.7	30.1	27.9	33.2	↑
Sidewalk Creek (SC)	Control	42.7	35.7	47.8	27.6	↓
	Heavy thin	23.6	34.1	34.8	49.0	↑
	Light thin	23.3	33.1	31.2	45.4	↑
	Light thin with gaps	33.6	39.8	44.6	55.7	↑

Table 6, continued—Mean percent cover for a) tall shrubs/small trees, b) low shrubs, c) herbs, and d) bryophytes in post-treatment years 1, 3, 5, and 10.

c. Herbs		**Herbaceous understory cover (percent)**				
Block	**Treatment**	1997	1999	2001	2006	Trend
Cougar Reservoir (CR)	Control	16.5	16.9	15.8	14.1	↓
	Heavy thin	21.5	30.2	32.8	26.0	?
	Light thin	16.5	21.9	26.2	29.2	↑
	Light thin with gaps	16.2	22.0	25.4	23.8	↑
Mill Creek (MC)	Control	21.8	19.2	15.5	24.8	↓↑
	Heavy thin	16.0	19.1	17.9	25.0	↑
	Light thin	12.9	20.1	22.7	24.6	↑
	Light thin with gaps	23.0	25.3	30.1	36.0	↑
Christy Flats (CF)	Control	23.1	18.1	20.6	23.0	↑
	Heavy thin	19.7	24.5	27.6	26.5	↑
	Light thin	21.1	20.9	22.5	39.3	↑
	Light thin with gaps	20.0	29.3	22.2	32.0	?
Sidewalk Creek (SC)	Control	16.4	11.7	14.9	31.2	↓↑
	Heavy thin	16.2	20.5	15.7	24.0	↑
	Light thin	13.7	15.7	14.8	21.4	↑
	Light thin with gaps	13.7	16.6	13.1	18.7	?

d. Bryophytes		**Bryophyte understory cover (percent)**				
Block	**Treatment**	1997	1999	2001	2006	Trend
Cougar Reservoir (CR)	Control	25.1	21.5	21.8	25.4	↓↑
	Heavy thin	4.5	5.8	23.1	12.1	↑↓
	Light thin	11.5	27.3	53.1	47.9	↑
	Light thin with gaps	3.0	5.0	21.1	12.0	↑↓
Mill Creek (MC)	Control	19.5	16.7	16.8	23.5	↓↑
	Heavy thin	7.8	11.7	23.7	18.1	↑
	Light thin	19.0	20.8	34.4	33.6	↑
	Light thin with gaps	29.5	24.6	51.0	49.3	↑
Christy Flats (CF)	Control	23.0	24.9	28.9	39.9	↑
	Heavy thin	2.5	2.3	5.3	4.7	↑
	Light thin	4.8	5.8	10.8	16.3	↑
	Light thin with gaps	4.4	6.6	4.5	6.1	?
Sidewalk Creek (SC)	Control	30.2	36.3	46.5	44.9	↑
	Heavy thin	1.4	21.9	42.2	37.3	↑
	Light thin	12.0	25.3	37.7	43.9	↑
	Light thin with gaps	7.7	26.6	29.0	27.3	↑

(Ares et al. 2009).

Vegetation composition showed a consistent trend toward higher species richness, which was mainly attributable to increases in early-seral species after thinning (Davis et al. 2007; Ares et al. 2009, 2010). The combination of ground disturbance and increased resource availability likely allowed establishment of these species, especially in harvested gaps (Fahey and Puettmann 2007, 2008). Species richness of mid- to late-seral species did not appear to be reduced. Notable exceptions include the reduction of obligate mycotrophs, plants in symbiotic relationships with fungi (Davis et al. 2007). This may be related to lower numbers and production of selected mushrooms after thinning (Pilz et al. 2006). Exotic species increased slightly after thinning operations, but absolute cover values typically remained under 2 percent (Davis and Puettmann 2009; Ares et al. 2009). One exception was a single treatment unit at a single site that had a nearby seed source of Scotch Broom (*Cytisus scoparius*), where cover values of exotic species increased to over 3 percent. However, it appears that within a decade the cover values of exotic species are declining, suggesting that most of these species are not invasive and expanding. In the long run, one would expect that the transitory increase in resources would be declining, but still be reflected in species composition for an extended time period, even after the overstory closure has reached high levels (Lindh and Muir 2004).

Conclusion

The short-term responses to thinning in the two studies show that questions regarding the efficacy of thinning for accelerating late-successional development do not have a simple answer. Instead, different measures of late-successional habitat appear to respond in different ways and at different time scales. For example, rapid increases in crown length and tree growth suggest that thinning is accelerating some features of late-successional structure. In contrast, late-successional attributes such as tall shrub cover and cover of late-seral species were not accelerated by thinning. Instead, they showed the opposite trends toward open and early-successional conditions, at least in the short term. Few responses varied significantly among alternative thinning treatments, suggesting that in the short term the thinning activity itself is more important than the actual choice of residual densities, or that the range of thinning treatments in these two experiments was too narrow. However, in the long run, one would expect that growth of overstory trees, tree regeneration, as well as species composition of understory vegetation, would reflect the different thinning intensities, including gap creation.

While long-term data are not available, the short-term trends in overstory development suggest that in thinned units most responses are transitory: they may be still detectable after canopy closure, but development trends will have slowed down. To maintain the benefits of an accelerated development of late-successional characteristics in thinned stands likely requires multiple entries (Del Rio and Berg 1979). Alternatively, very heavy thinning and gaps may provide sufficient conditions to accelerate late-successional habitat with a single entry. Including a variety of treatments in thinning operations, especially gaps and leave island treatments, also provides opportunities to increase the small-scale spatial variability within stands, a major characteristic of old-growth stands (Franklin and Van Pelt 2004).

The differential responses to thinning of various aspects of forest structure in these two studies suggest that managers should assess which specific structural attributes and processes are critical. Because all attributes cannot be achieved simultaneously, setting priorities is important. Alternatively, managing for heterogeneity can be used to emphasize different attributes at different spatial scales within and among stands. Tying silvicultural treatments and stand development into a multi-scale landscape plan will facilitate achievement of multiple management goals (Wilson and Puettmann 2007).

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Making the Little Things Count: Modeling the Development of Understory Trees in Complex Stands

Peter J. Gould and Constance A. Harrington

Abstract

Forest growth models are useful for asking "What if?" questions when evaluating silvicultural treatments intended to increase the complexity of future stands. *What if we thinned to level A or B? How would it affect the growth rates of understory trees? How many trees would survive?* To answer these types of questions, a growth model needs to accurately predict the growth and survival of understory trees. Some users of the Forest Vegetation Simulator (FVS) growth model have commented that model predictions for understory trees do not match their field observations or data. To study the relationships which govern growth of understory trees, we assembled a large database from silvicultural experiments and operational inventory data. This database provided an opportunity to look at the major factors that affect the growth and survival of understory Douglas-fir (*Pseudotsuga menziesii*), Western Hemlock (*Tsuga heterophylla*), and Western Redcedar (*Thuja plicata*). Tree attributes like diameter and crown ratio were the best predictors of tree growth, followed by measures of stand density and competition. We found that the potential and average growth of all three species decreased as the density of larger trees increased, but the growth of Douglas-fir was reduced the most by increasing overstory density. Similarly, competition with larger trees reduced the survival of Douglas-fir more than the other species. Western Hemlock generally had greatest growth at moderate to high levels of overstory density. Survival of Western Redcedar was the highest of the three species. Overall, we found the effect of overstory density on understory tree growth was less, but the effects on mortality were greater, than predicted in earlier versions of FVS. Incorporating these new relationships into future versions of FVS should provide forest managers with better tools to evaluate alternative management scenarios.

Keywords: Growth and yield models, Forest Vegetation Simulator, thinning, competition, under-planting, regeneration.

Introduction

In recent decades, a goal of silvicultural treatments on some forest ownerships has been to increase the complexity of future stands. Forest managers often want to take "simple" even-aged stands that are dominated by a single species and create complex stands that are spatially heterogeneous and contain multiple species, canopy layers, and age classes. Silvicultural treatments to create complex stands include variable-density thinning, thinning with "skips and gaps" (Harrington et al. 2005), and planting trees in the understory. Since we have little experience in creating these types of complex stands by design, we often turn to growth models to ask "What if?" questions. What if we alter overstory density and species? How do we know when we have enough understory trees so that

Peter J. Gould is inventory lead, Washington State Department of Natural Resources, Olympia, WA; peter.gould@dnr.wa.gov; **Constance A. Harrington** is a research forester, USDA Forest Service, Pacific Northwest Research Station, Forestry Sciences Laboratory, 3625 93rd Ave SW. Olympia, WA 98512; charrington@fs.fed.us

at least some will survive to become big trees? To answer these questions, we need a growth model that does a good job of predicting the development of understory trees.

The Forest Vegetation Simulator (FVS) is a widely used growth model in the Pacific Northwest (Crookston and Dixon 2005). We have evaluated the performance of FVS with data from research trials and found that it generally under-predicts growth and over-predicts survival of trees in an understory position. We are in the process of developing new models of understory growth and survival that will be incorporated into FVS. These models will better reflect the growth rates and survival of small trees that have been observed throughout the Pacific Northwest. In addition, the understory environment for small trees is more variable, in terms of light and other environmental conditions, in complex stands than in simpler, even-aged stands. Thus, the new models will be more sensitive to variation in overstory density and predict greater variability in growth rates than previous models. We have assembled a large data set describing small-tree growth from many different silvicultural experiments, as well as operational inventory data. Part of developing models to predict growth and survival is evaluating how different factors affect these processes. This provides an opportunity to highlight which factors are generally important in the Pacific Northwest.

Our goal for this paper was to draw together different sources of information to make some general observations about the growth and survival of small trees under different conditions. We used the database that we assembled to revise FVS to address the following questions:

1. Which set of factors best predicts the growth of understory trees?
2. How is the growth of understory trees affected by stand structure?
3. How is the survival of understory trees affected by stand structure?

Methods

The database was assembled from measurements of seven large-scale silvicultural experiments in western Washington and Oregon and from measurements of the US Forest Service's current vegetation survey (CVS) plots (USDA Forest Service 2009) (table 1 and fig. 1). These data cover the ranges of the PN and WC variants of FVS. We defined small trees as those that were <16 cm dbh (diameter at breast height) at the beginning of a growth period. We only used data from trees which were measured at least twice, so that growth increments and survival could be calculated. The period between measurements ranged from 3 to 11 years. We calculated diameter and basal-area growth increments (at breast height) for all trees that were >1.3 m tall at the beginning of the growth period. For small trees where only height-growth increments were measured, we used a height-diameter equation to calculate equivalent diameter and basal-area increments from the height increments. Growth increments were normalized to a 5-year period by assuming constant growth during the period.

Each of the silviculture experiments (full names appear in table 1) includes treatments with different levels of overstory removal and, in some cases, different spatial patterns (e.g., gaps versus uniform thinning). In five of the experiments, seedlings were planted following overstory treatments. The small trees that were selected from the ODS, STUDS, and WOTUS experiments were almost entirely those that were planted after the overstory treatment. Trees were not planted in the OHDS experiment, but the stands were old enough that our selection criterion ensured that only trees that were in an understory position at the beginning of the experiment or established afterward were selected. Stands in the Blue River and Clearwater experiments were fairly young; therefore, only trees that were planted, established naturally, or were too small to measure at the beginning of

Table 1—Silvicultural experiments that provided data for modeling the growth and survival of understory trees. Additional sources of information on these experiments can be found in the cited references.

Experiment	Understory tree development	Overstory age (years)	References
Forest Ecosystem Study Transects (FES)	natural	50-65	Carey et al. 1999
Olympic Habitat Development Study (OHDS)	natural	40-80	Harrington et al. 2005; Comfort et al. 2010
Overstory Density Study (ODS)	planted	40-70	Curtis et al. 2004; Poage and Anderson 2007
Western Oregon Thinning and Underplanting Study (WOTUS)	planted	50-55	Cole and Newton 2009
Clearwater and Blue River	most natural, some planted	15	Poage and Anderson 2007
Siuslaw Thinning and Underplanting for Diversity Study (STUDS)	planted	30-35	Chan et al. 2006; Poage and Anderson 2007

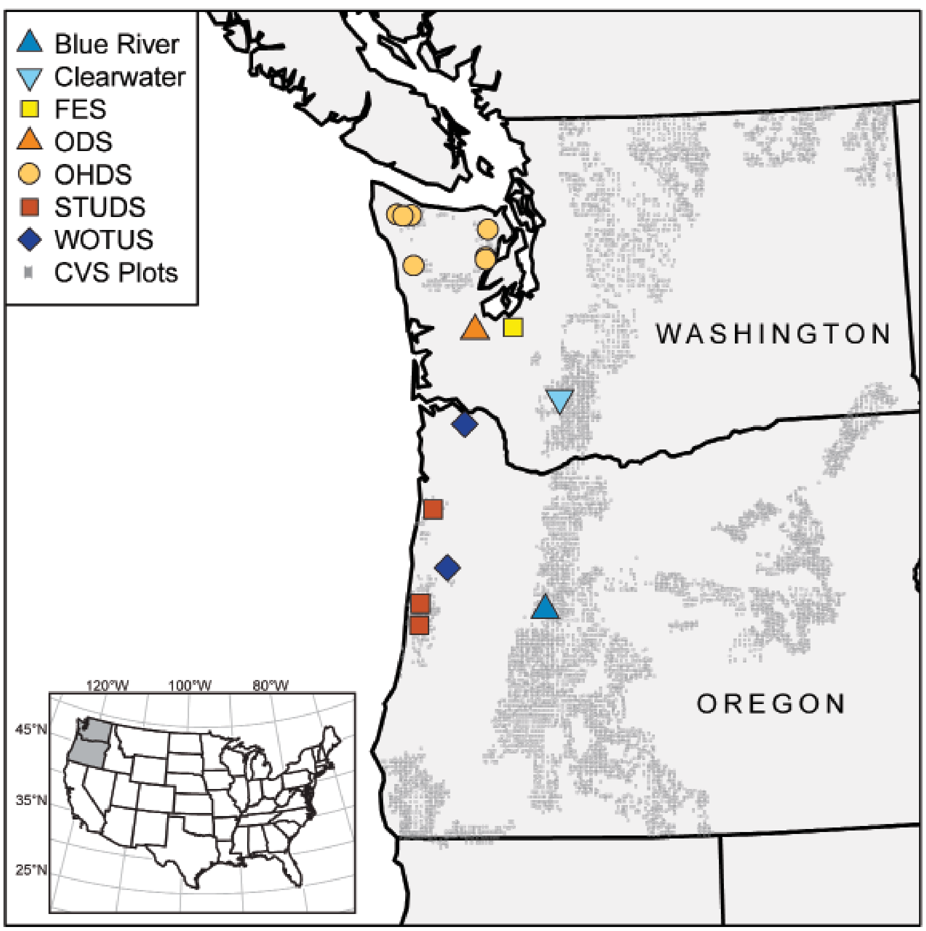

Figure 1—Locations of silvicultural experiments and CVS data in the modeling database.

the experiment (ingrowth) were selected for the database to avoid selecting overstory trees.

Small trees were selected from the CVS plots if they were <16 cm dbh at the beginning of the growth period. CVS plots were excluded from the database if there was evidence of thinning or another disturbance between measurements, leaving a total of 3988 plots. Unlike the large-scale experiments, the treatment histories and stand structures of the CVS plots were not known. We could not be sure which CVS plots were in complex stands. On the other hand, data from the CVS plots are valuable because they are representative of small-tree growth throughout much of the Pacific Northwest. For some analyses, we used only plots that had at least one tree >30 cm dbh to indicate the plot was located in a relatively complex stand. We compared results between the silviculture experiments and CVS plots to ensure our results would hold for both types of conditions.

A set of predictor variables was calculated from the database to explain small-tree growth and survival (table 2). There are three types of predictor variables:

1. Tree attributes such as dbh and crown ratio (*CR*);
2. Variables describing stand structure and the competitive environment, including basal area (*BA*) and crown-competition factor (*CCF*) (Krajicek et al. 1961); and
3. Variables describing site productivity and climate, such as site index (*SI*) and precipitation (*PPT*). Climate variables were derived from Rehfeldt's (2006) climate surfaces (Crookston 2011). Site index was measured on some of the plots and estimated for the remaining plots using nearest-neighbor imputation (Crookston and Finley 2008).

We chose to focus on Douglas-fir (*Pseudotsuga menziesii*), Western Hemlock (*Tsuga heterophylla*), and Western Redcedar (*Thuja plicata*) for this paper. Our database contained 32,534 observations for Douglas-fir, 22,593 observations for Western Hemlock, and 12,143 observations for Western Redcedar. We used the following methods to address each question:

Which set of factors best explains the growth of understory trees?

We looked at how well different combinations of predictor variables could predict 5-year basal-area growth of individual trees. We used a best-subset regression method to pick the best predictors for linear regression models that had between 1 and 10 coefficients (Lumley 2009). We repeated the subset selection process 100 times using data that were randomly selected from 25 percent of the plots each time. This iterative selection process helped to identify robust predictor variables that should work well in all of the different conditions found in the modeling data set. The predictor variables were ranked based on the numbers of times they appeared in a best regression model.

How is the growth of understory trees affected by stand structure?

The predictor variables that describe stand structure and the competitive environment are closely related, so we chose to look at only one of them, basal area of trees larger than the subject tree (*BAL*). This variable is a measure of the competitive position of individual trees, but it is also related to stand structure. For example, small trees that are growing in an understory position in complex stands will have high values of *BAL*, whereas those that are growing in gaps or even-aged stands will typically have low values of *BAL*. We looked at how *BAL* affected both average growth and potential growth by fitting cubic splines (Hastie and Tibshirani 1990). Spline fitting is a flexible modeling technique that is not constrained to produce a straight line or a particular curve shape; therefore, it allows for a detailed evaluation of the relationship between variables. Potential growth was evaluated by fitting a quantile spline (He and Ng 1999) to the 97th percentile of growth observed for each species.

Table 2—Abbreviations and descriptions for variables that were evaluated to predict the growth of understory trees.

Variable type	Variable	Description
Tree attributes	CR	Crown ratio (proportion)
	DBH	Diameter at breast height (cm)
	HT	Total tree height (m)
Stand structure / competitive environment	BA	Basal area ($m^2 \cdot ha^{-1}$)
	BA_I, BA_M, BA_T	BA of shade-intolerant, moderately tolerant, or tolerant species
	BAL	Basal area of trees larger than the subject tree ($m^2 \cdot ha^{-1}$)
	BAL/DBH	BAL divided by DBH
	BAL_I, BAL_M, BAL_T	BAL of shade-intolerant, moderately tolerant, or tolerant species
	CCF	Crown-competition factor: f(crown width, tree density) (%)
	CCF_{100}, CCF_{200}	Height above ground where CCF = 100% or 200% (m)
	CCF_D	CCF x crown depth
	CCF_R	CCF x CR
	CV_{DBH},CV_{HT}	Coefficient of variation of DBH or HT
	$DBHSUM$	Sum of tree DBH (cm)
	$HT100$	Average height of 100 largest trees per hectare (m)
	$HTSUM$	Sum of tree HT (m)
	$RELHT$	$HT/HT100$
Productivity, climate	$DD100$	Date when sum of $DD5$ = 100 (measure of spring warmth) (days)
	$DD0$, $DD5$	Degree-days <0°C, >5°C (°C·days)
	$JUNEDRY$	June temperature / June precipitation (°C·mm^{-1})
	PP, PPT_{GS}	Total precipitation, growing-season precipitation (mm)
	$PSUMR$	Ratio of summer-to-spring precipitation (proportion)
	$SDAY$, $FDAY$, FFP	Date of last spring frost, first fall frost, and frost-free period (days)
	SI	Site index (m)
	T_{AVE}, T_{MIN}, T_{MAX}	Average, minimum, and maximum temperature (°C)
	T_{COLD}, T_{WARM}	Average temperature in the coldest and warmest month (°C)

Some of the silvicultural experiments included thinning with gaps, which creates a different stand structure than uniform thinning. Where we had such information, we evaluated how gap creation affects growth by comparing growth of small trees in unthinned areas (skips), uniform (matrix) thinned areas, and gaps.

How is the survival of understory trees affected by stand structure?

Survival was evaluated across the range of BAL by examining survival rates graphically and fitting a logistic-regression equation (Schabenberger and Pierce 2002) for each species to compare average survival among species.

Results and Discussion

Which set of factors best explains the growth of understory trees?

Tree *DBH* at the beginning of the growth period was the most frequently selected variable to predict 5-year basal-area growth for each species (table 3). Crown ratio ranked second for Douglas-fir and Western Hemlock, and third for Western Redcedar. Measures of stand density or the competitive environment such as *BAL/DBH* and CCF_D (*CCF* x crown depth) were the next most frequently selected predictor variables. Some of the predictor variables related to overstory height ranked highly for Douglas-fir and Western Redcedar. The climate variables were generally not as highly ranked as those that describe tree attributes, stand structure, and density; however, at least two climate variables were found to be good predictors of growth in some cases. Climate variables related to temperature (T_{MIN} , minimum temperature, and *DD100*, a measure of spring warmth) were among the top 10 predictors for each species. Growth was predicted to increase with increasing temperature, reflecting the lower productivity of cold, high-elevation sites. Site index ranked among the top 10 predictor variables for Western Hemlock and Western Redcedar but not for Douglas-fir.

Tree attributes are generally good predictors of future growth (Hann et al. 2006). The size and crown ratio of an understory tree is a good indicator of its growth capacity following thinning (Shatford et al. 2009). Maintaining a high crown ratio may be more important to Douglas-fir than to more shade-tolerant conifers. The presence of vigorous understory trees can indicate that things are going right when trying to create a complex stand, but it is not a factor that can be readily changed by forest managers. The competitive environment of understory trees as measured by *BAL* or *CCF* has an important impact on growth and can be changed through silvicultural treatments that reduce overstory densities. We suspected that BA_T (basal area of shade-tolerant species) would be a better predictor of growth than *BA* alone, as shade-tolerant species typically allow less light to reach the understory than intolerant species (Canham et al. 1994); however, it was not a frequently selected predictor. Measures of the competitive environment that account for dominant tree height (e.g., *RELHT*, height divided by the average height in meters) and differences in crown width among species (e.g., CCF_D) were important predictors in some cases, suggesting that understory growth is affected to some degree by stand composition and vertical structure.

Changes to the understory light environment are an obvious result of reducing overstory density, but competition between overstory and understory trees for soil moisture and nutrients is also important (Devine and Harrington 2008; Harrington 2006). Plants growing below the main canopy are in a more complex competitive environment than those in the main canopy, thus, tree responses may be more variable and models to predict response may require greater complexity (Harrington et al. 2002).

Table 3—Ranking of variables for predicting individual-tree basal-area growth of small Douglas-fir, Western Hemlock, and Western Redcedar. Ranks are based on how frequently the predictors were selected using best subset regression on 100 randomly selected subsets of the modeling data. Variables definitions are in table 2.

	Frequency of selected predictors by species					
Rank	Douglas-fir		Western Hemlock		Western Redcedar	
1	*DBH*	100	*DBH*	96	*DBH*	100
2	*CR*	91	*CR*	85	*BAL/DBH*	79
3	CCF_D	71	*BAL/DBH*	77	*CR*	69
4	*HT100*	53	*CV*	59	*HT100*	52
5	*RELHT*	52	*FDAY*	47	CCF_{100}	34
6	CCF_{100}	49	*SI*	44	*DD100*	24
7	*PSUMR*	45	T_{COLD}	24	*SI*	20
8	*DSUM*	38	*DD5*	23	*JUNEDRY*	17
9	T_{MIN}	18	T_{WARM}	20	CCF_{200}	15
10	*BAL/DBH*	13	*HTSUM*	18	*BA*	15

How is the growth of understory trees affected by stand structure?

Potential and average growth rates were negatively affected by increasing competition (as measured by *BAL*) but the species responded differently to competition (fig. 2). When *BAL* was low, potential and average growth rates of Douglas-fir and Western Hemlock were about equal and both species had higher growth rates than Western Redcedar. The average growth of Western Redcedar was much lower than that of the other species at low *BAL*. The potential and average growth rates of Douglas-fir decreased more than those of the other species with increasing *BAL*. When *BAL* was greater than about 20 $m^2 \cdot ha^{-1}$, the species' ranks for potential and average growth were Western Hemlock > Western Redcedar > Douglas-fir. When we looked at data from the silviculture experiments only, we found that the species' ranks were the same, but the potential and average growth rates were generally lower.

We predicted the growth of small Douglas-fir using the current (2011) version of FVS (before our revisions) and compared it with the average growth in our data set (inset in bottom left panel of fig. 2). Observed growth was greater than predicted growth (in the 2011 version of FVS) throughout the range of our data. Predicted growth at moderate to high *BAL* dropped to less than 1.0 $cm^2 \cdot 5\ y^{-1}$ while observed growth remained around 4.5 $cm^2 \cdot 5\ y^{-1}$. Although values for both observed and predicted growth were low, the difference has a big impact on growth projections. For example, it could take more than 3 times longer for an understory tree to move into a midstory position in an FVS projection than it should based on our data. Predicted growth

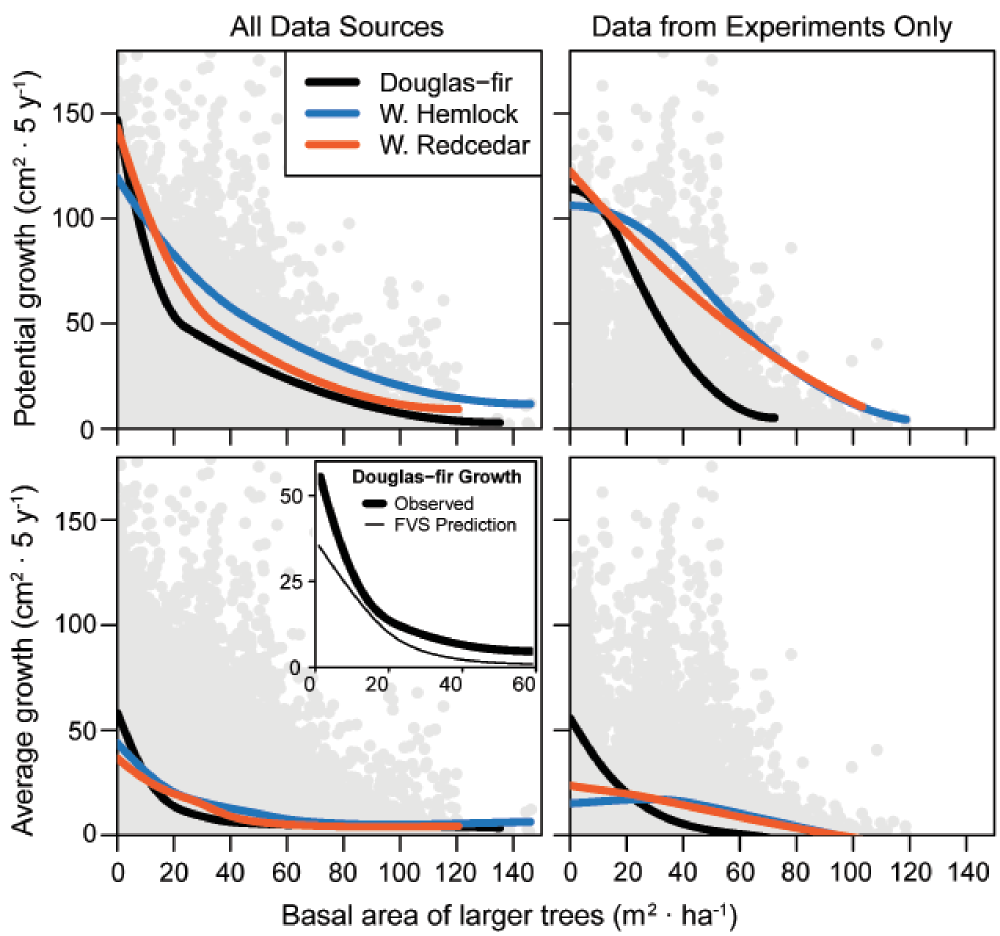

Figure 2—Potential basal-area growth (top row) and average basal-area growth (bottom row) of small trees across a range of basal area in larger trees (*BAL*). The potential growth curves show the 97th percentile of growth. Curves were fit to the entire modeling data set (left column) and to data from silvicultural experiments only (right column). The inset in the bottom left panel shows how the current version of FVS predicts the average growth of small Douglas-fir (observed growth is shown for comparison).

will more closely match observed growth in the revised version of FVS.

Our results agree with previous studies that have ranked Western Hemlock as the fastest growing species across a range of overstory densities (Chan et al. 2006; Cole and Newton 2009; Shatford et al. 2009), including complete overstory removal (Harrington 2006). Western Redcedar also showed greater potential growth compared to Douglas-fir at moderate to high overstory densities; however, its average growth was generally lower. Some of our data sets recorded high browse damage on Western Redcedar that hampered its growth (Cole and Newton 2009; Harrington 2006). The differences between average growth rates and potential growth rates highlight the variability in growth rates that can be found among trees in the same competitive position. The final FVS models will reproduce some of this variability by first predicting the average growth under a set of conditions and then adding a random component so that some trees are predicted to grow near their maximum rates.

Trees in gaps generally grew more rapidly than those in thinned and unthinned areas. Two installations of the OHDS experiment had large numbers of understory Western Hemlock within skips, matrix thinned areas, and 400-m^2 gaps (fig. 3). At the Rail installation, understory trees in gaps grew about 50 percent more than those in thinned areas and about 70 percent more than those in unthinned skips. Understory trees in both thinned areas and gaps at the Snow White installation grew about 40 percent more than those in skips.

Planted understory trees in the WOTUS experiment grew much more rapidly in gaps (600 m^2 or 1000 m^2 in size) than in uniformly thinned areas (fig. 4). Western Hemlock had the greatest growth of the three species in both thinned areas and gaps. The growth of Western Redcedar was damaged by browsing both in gaps and thinned areas.

Treatments to develop complex stands may be successful if they allow some trees to grow rapidly into midstory canopy positions, even if the average growth of understory trees is low. From this perspective, we looked at the range of growth rates within the OHDS experiment (fig. 5). Many of the trees in thinned areas and small gaps continued to grow at similar rates as in unthinned skips; however, the range of growth rates increased substantially. The range of growth rates in gaps was about twice that of unthinned skips.

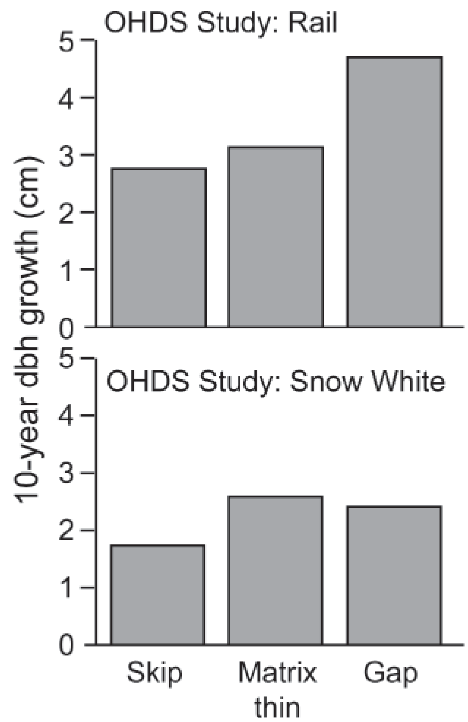

Figure 3—Diameter growth of understory Western Hemlock in skips (unthinned areas), matrix thinned areas, and gaps at the Rail and Snow White installations of the Olympic Habitat Development study.

How is the survival of understory trees affected by stand structure?

The survival of all three species was reduced in stands with high overstory densities, but the survival of Douglas-fir decreased much more than survival of the other species with increasing *BAL* (fig. 6). The mortality rate of Douglas-fir doubles between about 5 and 40 m^2·ha^{-1} of *BAL*. Western Redcedar had the greatest survival across the range of *BAL*. Survival was generally higher in the silvicultural experiments (not shown) than in the CVS plots, possibly owing to the effects of other treatments (e.g., vegetation control in some cases) and the initially high vigor of the planted growing stock. We also compared predicted survival in the 2011 version of FVS with observed survival of Douglas-fir (inset in fig. 6). Observed survival was lower across the range of *BAL* and it decreased more sharply with increasing *BAL* than predicted in the 2011 model. FVS will be revised so that predicted survival better matches the survival rates observed in our data set.

The low survival of understory Douglas-fir in particular may require forest managers to maintain relatively low overstory densities or create large gaps if they want it to survive as an understory tree. In contrast, Western Hemlock and Western Redcedar may become established and survive across a much greater range of densities. For example, we have found that Western Redcedar can survive under dense overstory conditions for decades with very little growth (e.g., Harrington and Devine 2011). We did not account for mortality that can occur after planting in the understory owing to low seedling vigor or unfavorable environmental conditions (e.g., low soil moisture) (Cole and Newton 2009). Survival may be poorer than our

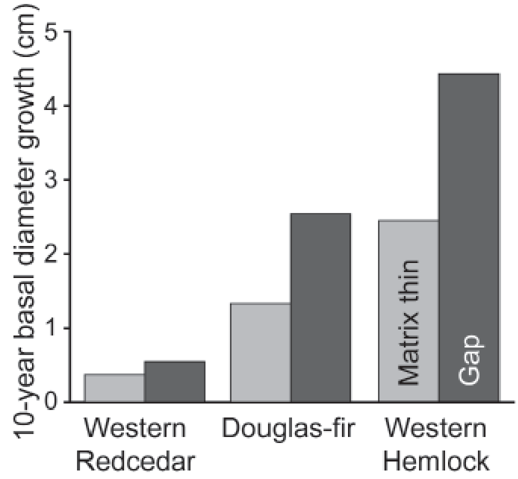

Figure 4—Basal diameter growth of understory Western Redcedar, Douglas-fir, and Western Hemlock in matrix thinned areas and gaps in the Western Oregon Thinning and Underplanting study.

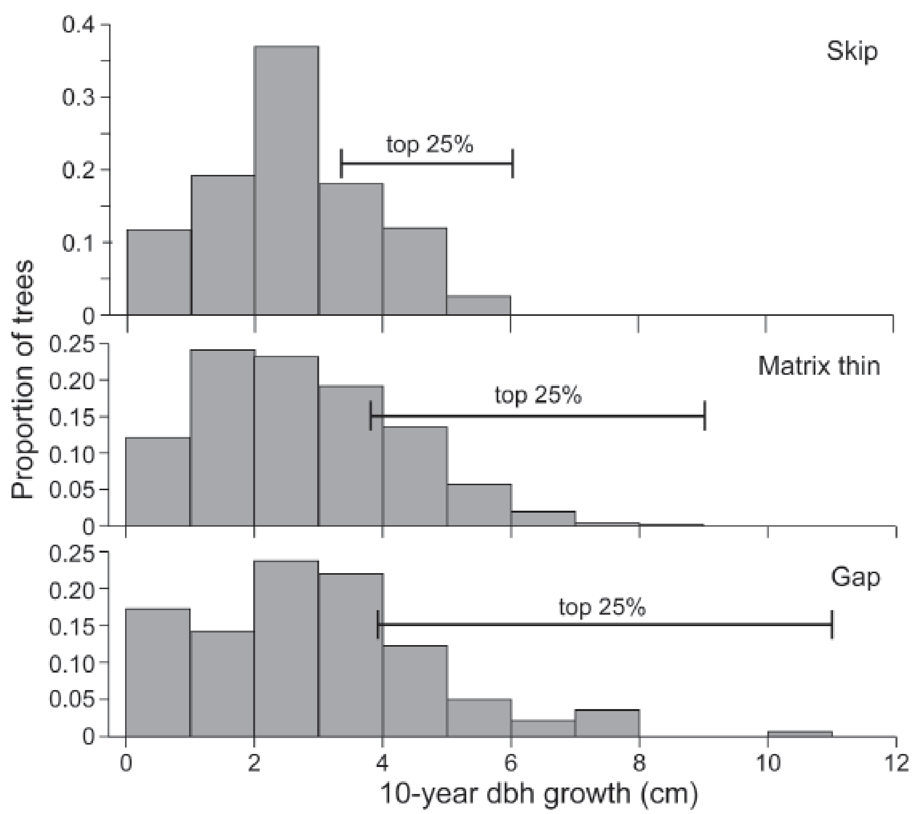

Figure 5—Range of diameter growth rates in skips (unthinned areas), matrix thinned areas, and gaps in the Olympic Habitat Development study. Thinning and gap creation greatly increased the range of growth rates among the fastest-growing 25 percent of trees, which should allow some trees to reach midcanopy positions rapidly.

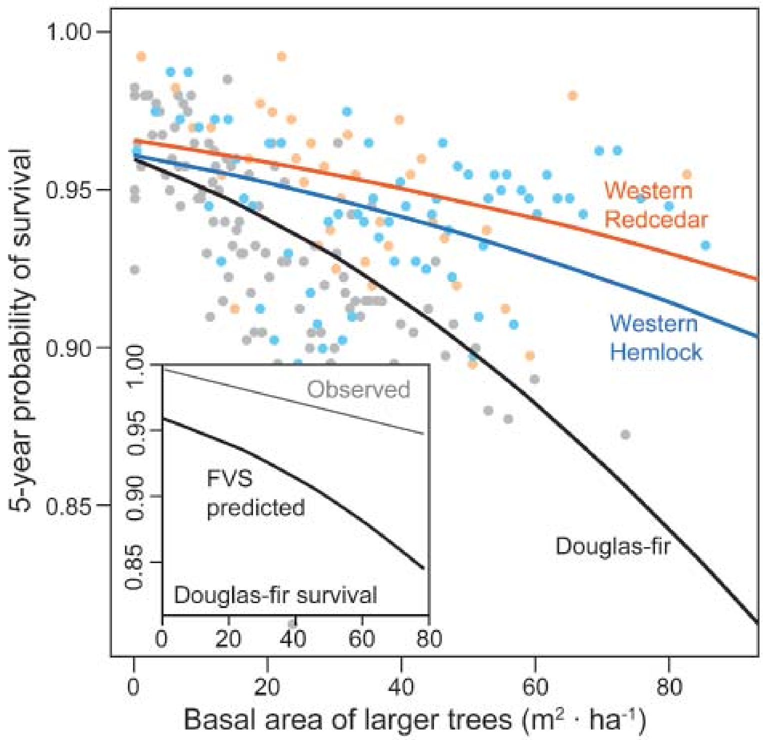

Figure 6—Survival of Douglas-fir, Western Hemlock, and Western Redcedar across a range of basal area in larger trees (*BAL*). Each point represents the survival rate of 400 trees; curves show fits of logistic regression equations. The inset shows how the current version of FVS predicts survival of small Douglas-fir.

data indicates during the first year or two after planting in some cases.

Management Implications

To create complex stands, forest managers need to create conditions that allow understory trees to establish, survive, and grow. Density management is an important tool for improving growing conditions for understory trees. Without thinning or other disturbances, most of the relatively young stands in the Pacific Northwest will maintain densities that will severely reduce the growth of understory trees (e.g., >40 $m^2 \cdot ha^{-1}$ of basal area). Uniform thinning can improve the growth and survival of understory trees, but thinning with gaps will create a wider range in understory tree growth rates, allowing some understory trees to grow more rapidly into the midstory. Treatments that create a range of overstory densities (i.e., thinning with skips and gaps) may be the best approach for creating conditions where some understory trees can thrive while meeting other management objectives.

Future Use of West-side FVS Variants

Our results will be incorporated initially into the PN and WC variants of FVS in 2012. Growth of understory trees will be predicted on a point basis, which will allow users to simulate variable thinning densities and gaps. FVS has a set of keywords (POINTREF, SETPTHIN, THINPT) that can be used to thin points to different densities or to simulate gaps. Because the variants of FVS used in the Pacific Northwest do not have full regeneration models, users will still need to add regeneration to the model runs to reflect understory planting or expected natural regeneration.

FVS can predict the average growth of understory trees under a given set of conditions, but it does not account for all factors that may be important in a particular case. Not every factor that is known to affect growth can be incorporated when developing the kind of data-based, empirical models that are used in FVS. This does not mean that these factors are not important. For example, competition with shrubs and herbaceous vegetation may strongly affect growth of understory trees, but competition with vegetation other than trees is not modeled by FVS. Forest managers will always need to use their judgment and experience to determine whether conditions are favorable for the growth of understory trees. Stands should also be revisited periodically to determine if treatments have been successful and whether FVS projections were accurate. We invite users to provide feedback to us on how well they think the models perform.

Acknowledgments

We thank Paul Anderson, Elizabeth Cole, Mike Newton, and Leslie Brodie for generously sharing data from their silvicultural experiments, Robert Brown for making the CVS database available, David Thornton for his help in assembling the research database and Erin Smith-Mateja for her help in integrating our results into FVS. We also thank the Oregon State Office of the Bureau of Land Management and the Olympic National Forest for financial support.

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Thinning Effects on Tree Mortality and Snag Recruitment in Western Oregon

Erich K. Dodson, Klaus J. Puettmann, and Adrian Ares

Abstract

Tree mortality shapes forest structural development and the resulting dead wood provides habitat for many species. However, the effects of thinning on mortality and large snag recruitment have been variable and remain poorly understood. We examined thinning effects on tree mortality at eleven Density Management Study (DMS) sites in western Oregon. Thinning reduced mortality rates for small trees, especially Douglas-fir (*Pseudotsuga menziesii*), but had little influence on mortality of large trees (≥50 cm dbh). Gaps created by harvest did not result in increased mortality during the sampling period (6 to 11 years following gap creation). Few large snags (≥50 cm dbh) were produced regardless of treatment. Thinning likely reduced ongoing suppression mortality of small trees, but in the short-term did little to provide the large snags that are characteristic of late-successional forests. Large snags may be created in subsequent planned thinning entries, but low mortality after the first thinning implementation suggests that active creation of large snags may be necessary to accelerate their development.

Keywords: Coarse woody debris, Douglas-fir, multi-cohort management, variable-density thinning, wildlife habitat.

Introduction

Tree mortality is an important ecosystem process (Franklin et al. 1987). It shapes community composition and forest structure, with direct implications for snags and down wood, which provide habitat for many species (Hayes and Cross 1987; Berg et al. 1994). Intensive management has resulted in young, homogeneous stands in the Pacific Northwest, with higher tree densities than found in many old-growth stands (Tappeiner et al. 1997). Consequently, these young stands lack many habitat features of late-successional forests, including large snags and large-diameter down wood (Spies et al. 1988; Hayes et al. 1997). Thinning has been widely proposed as a means to accelerate the development of late-successional structure (Muir et al. 2002; Bauhus et al. 2009), but its effects on mortality and snag recruitment are still not well understood. For example, thinning has been found to reduce (Williamson 1982; Marquis and Ernst 1991; Davis et al. 2007) or increase (Walter and McGuire 2004; Maguire et al. 2006) mortality rates. Thinning from below can remove small, stressed trees that are at high risk for mortality and leave a residual stand of large, vigorous trees with low mortality rates (Powers et al. 2010). Thinning also increases the resources available to smaller, residual trees, thus reducing suppression mortality (Williamson 1982; Marquis and Ernst 1991). In contrast, thinning can increase physical damage to residual

Erich K. Dodson is a faculty research assistant and **Klaus J. Puettmann** is Edmund Hayes Professor in Silvicultural Alternatives, Department of Forest Ecosystems & Society, Oregon State University, Corvallis, OR, 97331; Erich.Dodson@oregonstate.edu; **Adrian Ares** is Project Director, SANREM CRSP (Sustainable Agricultural and Natural Resource Management Collaborative Research Support Program) Office of International Research, Education and Development (OIRED) Virginia Tech, 526 Prices Fork Road Blacksburg, VA 24061.

trees due to wind, snow, and ice (Harrington and Reukema 1983; Franklin et al. 1997), with especially strong effects for trees along harvest edges (Chen et al. 1992; Zeng et al. 2004). We examined data from 11 Density Management Study (DMS) sites in western Oregon with a variety of site histories and conditions (Cissel et al. 2006) to better understand the effects of alternative thinning treatments on tree mortality.

Large snags (≥50 cm dbh [diameter at breast height]) are common in late-successional forests of the Pacific Northwest (Spies et al. 1988; North et al. 2004; Ares et al. 2012; Lutz et al. 2012), provide habitat for numerous species (Schreiber and de Calesta 1992; Barbour et al. 1997; Marcot et al. 2002), and may be critical for some species (Mannan et al 1980). Young stands may have an abundance of small snags (Carey and Johnson 1995; Van Pelt and Nadkarni 2004). However, without management intervention, more than 250 years may be required for large snag formation in natural stands (Van Pelt and Nadkarni 2004; Gerzon et al. 2011). A better understanding of large snag recruitment in thinned and unthinned forests is needed to guide management actions that seek to accelerate the development of late-successional forest characteristics.

We examined rates of mortality with and without thinning, with an emphasis on understanding effects of thinning on developing late-successional stand structure, using two components of the DMS study: initial thin (seven sites) and rethin (four sites). We use a combination of previously reported mortality results (Dodson et al. 2012) and new data analyses to address the effects of thinning on mortality. Specific research questions included:

1. Does thinning affect mortality rates of residual overstory trees?
2. Are large snags being created, and is this process affected by thinning?
3. Does gap creation during harvest alter mortality rates of nearby trees?

Methods

Study Sites

The DMS initial and rethin studies (Cissel et al. 2006) were each established on land managed by the Bureau of Land Management in western Oregon. Sites were selected to be relatively homogeneous and were generally dominated by Douglas-fir (*Pseudotsuga menziesii* (Mirb.) Franco). Common conifer species included Douglas-fir, Western Hemlock (*Tsuga heterophylla* (Raf.)) and Western Redcedar (*Thuja plicata* Donn ex D. Don). Common hardwoods included Big-leaf Maple (*Acer macrophyllum* Pursh), Red Alder (*Alnus rubra* Bong), California Hazel (*Corylus cornuta* Marsh), Pacific Madrone (*Arbutus menziesii* Pursh, *Frangula purshiana* (DC.) Cooper) and Golden Chinkapin (*Chrysolepis chrysophylla* (Douglas ex Hook.) Hjelmqvist). The initial thin sites were at least 80 ha in size and ranged from about 40 to 60 years old at the time of thinning. Rethin sites were are least 16 ha in size, ranged from 50 to 80 years old and had been thinned 25 to 30 years prior to study initiation to a density of ≥250 trees per hectare (tph). Detailed site characteristics are provided in Cissel et al. (2006) and details of tree species composition are given in tables 1 and 2.

Treatments

Initial thin treatments were implemented in a randomized-block design with one replication of four thinning treatments (including controls) applied at each of seven sites for a total of 28 treatment units. Individual treatment units ranged from 14–58 ha. Control stands were unthinned and had an average density of 565 tph. The high-density (HD) treatment was thinned to a residual stand density of 300 tph with 3 to 13 percent of the total stand retained in non-thinned circular reserves. In the moderate-density (MD) treatment, stand density was reduced to 200 tph. Additionally, 4 to 18 percent of the stand was cut in circular gap openings and 4 to 13 percent of the stand was left in unthinned circular leave islands. The variable-density (VD) treatment was designed to create the maximum spatial

Table 1—Total number of trees and number of trees dying over the period 6 to 11 years following thinning for each species or species group and treatment in the initial thinning study.

	Hardwood		Douglas-fir		Western Hemlock	
Treatment	Trees sampled	Mortality (5 years)	Trees sampled	Mortality (5 years)	Trees sampled	Mortality (5 years)
Control	266	36	3716	516	1686	142
High density	528	82	3260	195	1022	100
Moderate density	607	92	2463	189	1027	93
Variable density	475	55	2203	114	840	44

Table 2—Total number of sampled trees and the number of sampled trees that died from 1 to 11 years following thinning for each species or species group in each treatment for the rethin study.

	Cedar[1]		Hardwood		Douglas-fir		Western Hemlock	
Treatment	Trees sampled	Mortality (10 years)	Trees sampled	Mortality (10 years)	Trees sampled	Mortality (10 years)	Trees sampled	Mortality (10 years)
Once-thinned	262	26	104	15	1038	104	161	11
Rethinned	168	13	170	25	615	43	191	10

[1]Includes Western Redcedar (*Thuja plicata*) and Incense Cedar (*Calocedrus decurrens*).

variability and complexity within the stand. This treatment included both the high (300 tph) and moderate-density retention (200 tph) treatments each applied over 25 to 30 percent of the stand and an additional 8 to 16 percent of the stands were thinned to a residual density of 100 tph. Finally, 4 to 18 percent of the stand was assigned to leave islands and gap openings, respectively. Gaps and leave islands in all treatments were a mixture of 0.1-, 0.2- and 0.4-ha circular areas. Thinning was completed between 1997 and 2000. All thinning was "from below" focusing on taking out the smaller trees. Less-abundant species (typically all species other than Douglas-fir), especially hardwoods, were preferentially reserved. Details of treatment implementation at individual sites are presented in Cissel et al. (2006).

The rethin study included stands that had previously been thinned in a clumpy distribution 25–30 years before study initiation. It included just two treatments, a control (270 to 580 tph) that was not thinned again, and a second thinning to a residual tree density between 80 and 150 tph (Cissel et al. 2006; Berger et al. 2012). No gaps or leave islands were included in the rethin study.

Sampling

In the initial thin study, 77 permanent 0.1-ha circular plots were established at each of the seven sites to measure overstory trees. Plot locations were randomly selected across the entire treatment unit including gaps and leave islands, with 21 plots in each of the thinned treatment units and 14 plots in each control treatment unit at each site. All trees ≥5 cm dbh were tagged, identified to species, measured for dbh, and recorded as live or dead. Clumped hardwood trees were measured as separate trees provided their dbh was ≥5 cm. Data were collected in the summer 6 years after thinning (2003–2005) and 11 years after thinning (2008–2010), with sample years varying among sites.

Plots in the rethinning study were non-randomly established in areas where residual tree density was within 20 percent of target density and distributed to maximize coverage of the treatment unit. Spacing between plots varied among sites to establish a total of 20–25 plots per site. All overstory trees (>5 cm dbh) were tagged, identified to species, measured for dbh, and recorded as live or dead. Data were collected for each site 1, 6, and 11 years after thinning.

Statistical analyses

We selected a significance level $\alpha = 0.05$ prior to statistical analysis. All analyses were performed in SAS (SAS Institute, Cary, NC, v. 9.3). We report results from a previous study (Dodson et al. 2012) that examined the average annual mortality rate in the initial thinning study for the five-year sampling period as the number of trees that died between the first and second measurements divided by the total number of trees present at the first measurement for each treatment unit (n = 28 treatment units). For this study, average annual mortality rates were compared among treatments using a mixed model with thinning treatment as a fixed effect and site as a random effect. Post-hoc tests were conducted between each combination of individual treatments with a Tukey correction for multiple comparisons (Dodson et al. 2012). Residuals were inspected to ensure that assumptions of normality, equal variance, and independence were not violated.

Mortality for the rethin study was evaluated over a ten-year period, from 1 to 11 years following rethinning, with a mixed-model logistic regression. The analyses focused on Douglas-fir because this species was well-represented at all sites, spanned a range of initial sizes, and comprised the majority of trees (table 2). Individual trees that were alive in the first sample, one year after thinning, were assigned a "yes" or "no" based on whether they were also alive 11 years after thinning. The probability of mortality was modeled as a function of initial tree size, with treatment included as a categorical fixed effect. Site, treatment unit within site, and plot within treatment unit were included in the model as random effects to account for the nested structure of the sampling design. Model fit was evaluated visually by overlaying mortality rates for 10-cm dbh classes with predicted values. The Kenward-Roger method was used to calculate the denominator degrees of freedom due to the unbalanced sampling design.

We compared mortality rates among plots of different treatment "strata" (gaps, leave islands, and thinned areas) to examine whether small-scale treatment effects had an impact on mortality rates. Only the initial thinning treatment included gap creation, and only the MD and VD included all three strata; therefore, the analyses used only plots from these treatments. A total of 286 plots were included in this analysis, of which 71 plots included at least a portion of a gap and 32 plots included at least a portion of a leave island. Plots that included any portion of a gap or leave island were included in these strata, respectively. Gap plots included some residual trees (plots were not entirely in the gap) which were used to calculate rates of mortality. The remaining plots were considered as the "thinned" stratum for the analysis. The average annual mortality over five years (from 6 to 11 years following treatment) was calculated for each plot as described above for the initial thin study. A mixed model with stratum as a fixed factor and site as a random factor was used to analyze the effects of strata on mortality rates. A random effect was also included for treatment unit to account for nesting of plots within treatments units at a site. Post-hoc tests were conducted between the three strata with a Tukey correction for multiple comparisons. Residuals were inspected to ensure that assumptions of normality, equal variance and independence were not violated. The Kenward-Roger method was used to calculate the denominator degrees of freedom due to the unbalanced sampling design.

Results and Discussion

Douglas-fir was the dominant tree species in most stands (tables 1 and 2) and the vast majority of the large trees were Douglas-fir (Dodson et al. 2012). Douglas-fir also comprised the majority of the mortality in both the initial and rethin studies (tables 1 and 2). Douglas-fir is less shade-tolerant than the other conifer species on these

Table 3—Type III tests of fixed effects for the mixed model logistic regression for individual tree mortality in the re-thin study.

Effect	Numerator DF	Denominator DF	F-value	*P*
Treatment	1	6	9.9	0.018
dbh	1	1649	94.7	<0.001
dbh · treatment	1	103	15.3	<0.001

sites (Minore 1979; Shatford et al. 2009). This lower shade-tolerance many have contributed to higher rates of mortality for small Douglas-fir relative to other species (Dodson et al. 2012).

Thinning significantly affected mortality in both the initial thinning (Dodson et al. 2012) and rethin studies (table 3). In the initial thinning study, the VD treatment had significantly lower mortality than the control (fig. 1; Dodson et al. 2012). The HD and MD were intermediate and not significantly different from any other treatment (fig. 1; Dodson et al. 2012). In the rethin study, thinning effects varied significantly with initial tree size (table 3). Thinning reduced the probability of mortality for small Douglas-fir, but had little effect on mortality of larger trees (≥50 cm dbh; fig. 2). A similar effect was observed in the initial thinning study (Dodson et al. 2012). The decreased average annual mortality rate after thinning from below in these two component studies (Dodson et al. 2012; fig. 2) is similar to patterns observed in other Douglas-fir forests (Davis et al. 2007) and other forest types (Marguis and Ernst 1991; Powers et al. 2010). However, thinning from above can result in the opposite pattern (Marguis and Ernst 1991; Powers et al. 2010), which may contribute to some of the variability reported by previous studies.

Small trees had higher mortality rates than large trees in both the initial and rethin studies. This pattern is consistent with ongoing suppression mortality, which is common in young forests (Carey and Johnson 1995; Van Pelt and Nadkarni 2004). More than 80 percent of the trees that died across all sites and treatments in the initial thinning study were relatively small (≤35 cm dbh). In contrast, large trees (≥50 cm

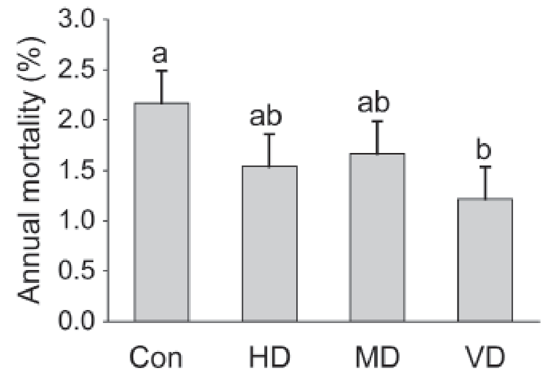

Figure 1—Annual mortality rates of all species combined in the initial thin study for each treatment for the period from 6 to 11 years following thinning. Different letters represent statistically significant differences in Tukey-HSD pair-wise comparisons. Con = Control, HD = High-density, MD = Moderate-density, VD = Variable-density.

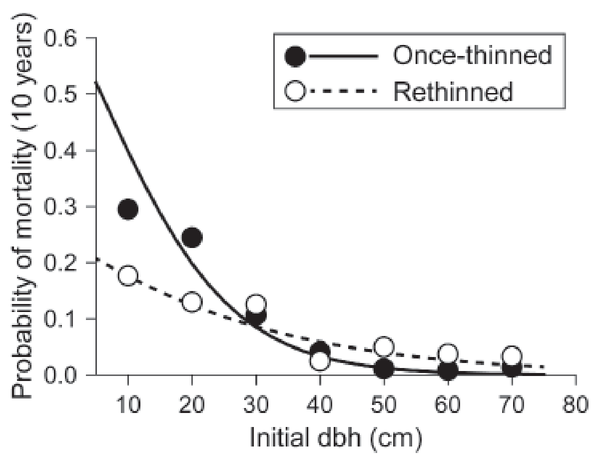

Figure 2—Probability of mortality as a function of initial tree size for a 10-year period after thinning in the once-thinned and twice-thinned units of the rethin study for Douglas-fir. Lines are predicted values from mixed-model logistic regression. Scatterplot points are mortality rates calculated from the data with trees divided into 10-cm size classes. Open circles are rethinned values, closed circles are once-thinned values.

dbh) had much lower mortality rates in both the initial thin study (Dodson et al. 2012) and rethin studies (fig. 2). For example, just one large snag per 10 ha was formed on average each year across all treatments in the initial thinning study, despite an average of about 75 large live trees per ha. These low rates of large-tree death suggest that few large snags form under current conditions, regardless of thinning treatments. This suggests that current management practices are not sufficient for the accelerated development of large snags in young forests in the short term; and indeed active creation of large snags may be implemented as part of the DMS study plan if this pattern continues over subsequent thinning entries (Cissel et al. 2006).

Average annual mortality rates varied significantly among strata (F = 3.0, P = 0.05). However, gap formation appeared to have little effect on mortality during our study period (6 to 11 years following gap creation). Average annual mortality rates on plots with gaps, which included both portions of gaps and neighboring forest interior, was 1.4 percent. This was intermediate between thinned plots (1.2 percent annual mortality) and leave island plots (1.9 percent annual mortality) and not significantly different than either (fig. 3). However, mortality rates were significantly lower on thinned plots than plots in leave islands (P = 0.046), consistent with the overall effect of thinning reducing mortality. Additionally, trees located directly on gap edges (Dodson et al. 2012) had an annual mortality rate of just under 2 percent from 6–11 years following thinning, or nearly equivalent to the rate observed in the control and leave islands. However, previous studies have reported increased mortality along clear-cut edges (Chen et al. 1992; Zeng et al. 2004) suggesting that gap size may be an important consideration. Furthermore, mortality due to physical agents such as wind is often episodic (Lutz and Halpern 2006) or may occur soon after thinning operations are completed (Chan et al. 2006), which may not have been captured in this study. Thinning effects on mortality likely also depend on the timing of thinning relative to stochastic events such as wind and ice storms. For example, if a severe storm immediately follows thinning, thinning may increase mortality, but if no such storm happens thinning may decrease mortality. The lack of an obvious increase in mortality with gap formation across the seven sites in this study, which have a variety of site histories and site conditions and were thinned in different years (Cissel et al. 2006) suggests that thinning treatments including gap creation can often be done without large increases in mortality (Roberts et al. 2007).

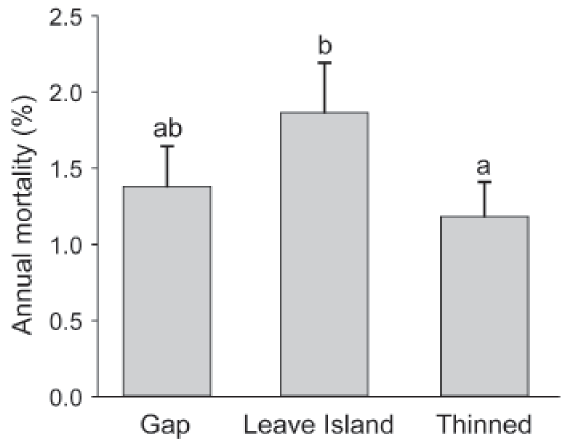

Figure 3—Annual mortality rates for all species combined from 6 to 11 years following thinning in the three different plot types in the MD and VD treatments. Different letters represent significant differences in Tukey HSD pair-wise comparisons. “Gap” plots included both portions of gaps and neighboring forest.

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Thinning Effects on Spotted Owl Prey and Other Forest-Dwelling Small Mammals

Todd M. Wilson and Eric D. Forsman

Abstract

Thinning has been promoted as a method for accelerating the development of late-seral habitat and improving the overall health and function of young forests in the Pacific Northwest. Population studies have shown early and positive responses to thinning by some small forest-floor mammals (primarily mice, terrestrial voles, and shrews). However, thinning reduces the abundance of some tree-dwelling rodents, especially Northern Flying Squirrels (*Glaucomys sabrinus*) and Red Tree Voles (*Arborimus longicaudus*), that are important prey species for Northern Spotted Owls (*Strix occidentalis caurina*). Recent studies suggest that reductions in Northern Flying Squirrel abundance following thinning may be driven by increased susceptibility to predation created by removal of critical above-ground cover. Predation, lack of canopy connectivity, and reduction in suitable nest substrates may all contribute to reduced Red Tree Vole abundance following thinning. The long-term benefits of some thinning treatments may be positive for both flying squirrels and Red Tree Voles, but may not be realized for several decades or more, as the development of a midstory layer of trees may be critical to the success of thinning in promoting habitat for these species. Additional research into the ecology of the two woodrat species (*Neotoma fuscipes* and *N. cinerea*) found in the Pacific Northwest is needed to provide a more complete understanding of the effects of forest management activities on spotted owls and their prey. It may be possible to design thinning prescriptions that lessen the short-term negative effects on arboreal rodents. Long-term goals should focus on creating more structurally and biologically complex forests across the landscape at scales and patterns compatible with the ecologies of spotted owl prey and other organisms. Joint research-management efforts to test new silvicultural prescriptions, expand current predictive models of high-quality prey habitat, and develop management strategies that consider the temporal effects of management on owl prey at the stand, landscape, and regional levels, could advance our understanding of owl prey ecology and help ensure that healthy populations of spotted owls and their prey persist on the landscape over the long term.

Keywords: Thinning, variable-density thinning, Northern Spotted Owl, Northern Flying Squirrel, Red Tree Vole, Northwest Forest Plan, old-growth forest.

Todd M. Wilson is a wildlife biologist and **Eric D. Forsman** is a research wildlife biologist, USDA Forest Service, Pacific Northwest Research Station, Forestry Sciences Laboratory, 3200 SW Jefferson Way, Corvallis, OR 97331; twilson@fs.fed.us

Introduction

The Pacific Northwest is rich in forest-dwelling small mammals, including mice, voles, shrews, tree squirrels, chipmunks, and woodrats. Collectively, these species serve as important prey for a wide variety of avian, mammalian, and reptilian predators. Small mammals also consume and disperse fungal spores essential for tree growth, help regulate invertebrate populations, disseminate vegetation through caching of seeds, nuts, and other fruiting bodies, and promote soil and carbon dynamics through excavation of down wood and soil (Carey and Harrington 2001). Thus, small mammal populations have been used as indices for the overall health and sustainability of forests in the region, including evaluation of the ecological effects of forest management activities like timber harvesting (Carey and Johnson 1995; Carey and Harrington 2001; Lehmkuhl et al. 1999; Suzuki and Hayes 2003). Four species, Northern Flying Squirrels (*Glaucomys sabrinus*), Red Tree Voles (*Arborimus longicaudus*), and Dusky-footed and Bushy-tailed Woodrats (*Neotoma fuscipes* and *N. cinerea*) dominate the diets of the federally threatened Northern Spotted Owl (*Strix occidentalis caurina*) (Forsman et al. 1984, 2004a,b; USFWS 1990). Spotted owl home-range size has been shown to decrease with increasing flying squirrel densities (Carey and Peeler 1995; Carey et al. 1992; Zabel et al. 1995), suggesting that understanding habitat needs for owl prey may be critical to the recovery of spotted owls and their habitat across the region (USFWS 2008).

One way to partition the small mammal community is by the relative degree to which each species uses the vertical strata of a forest (fig. 1). At one end of the continuum are forest-floor specialists like shrews and terrestrial voles that spend virtually all their time on or beneath the forest floor. At the other end of this continuum are arboreal (tree-dwelling) rodents like flying squirrels and Red Tree Voles that spend most of their time in the forest canopy (Maser et al. 1981, Wilson 2010). In addition, there are habitat generalists including deer mice (*Peromyscus* spp.), Trowbridge's Shrews (*Sorex trowbridgii*), and chipmunks (*Tamias* spp.) that spend considerable amounts of time on the forest floor but also regularly use forest canopies.

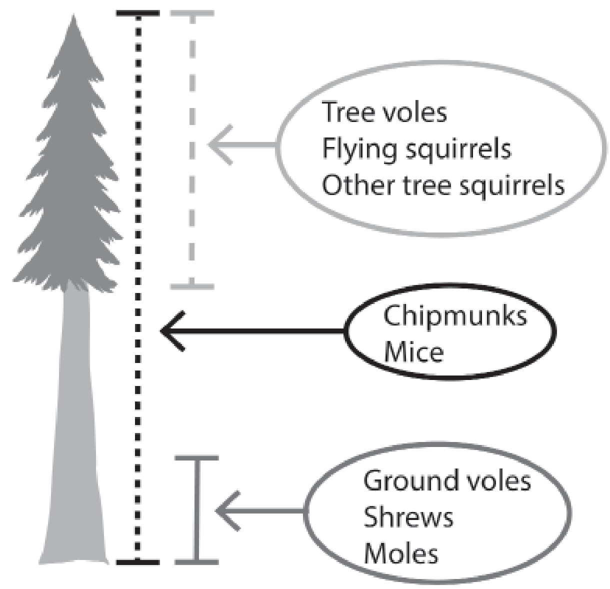

Figure 1—Illustration of the relative degree of use of the vertical component of a forest for several common rodent species in Pacific Northwest forests.

Thinning has become the management tool of choice for accelerating the development of late-seral wildlife habitat and improving the overall health and function of young forests in the Pacific Northwest. The effectiveness of thinning to either promote or maintain habitat has been demonstrated for a number of forest-floor specialists and generalists (Carey and Wilson 2001; Gitzen et al. 2007; Hayward et al. 1999; Klenner and Sullivan 2003; Suzuki and Hayes 2003; Wilson and Carey 2000). This is true in large part because the understory layer of a forest is usually the first and quickest to respond to the increase in growing space, light, and nutrients created by thinning. This understory response results in increased structure and plant diversity on the forest floor, providing food, shelter, and protective cover for small mammals. Positive responses have generally been strongest in forests that lacked substantial understory prior to thinning (e.g., young, even-aged, stem-exclusion forests with a primarily moss-dominated forest floor), but increases in abundance have also been noted for several small mammals in forests with a well-established understory prior to treatment (e.g., Carey and Wilson 2001). In this latter case, additional growth and maturation of a tall shrub and deciduous tree layer may provide additional habitat components important for some species.

In marked contrast, there is growing evidence that thinning can have negative effects on some arboreal rodent populations (Forsman 2004; Wilson 2010). For example, several studies have shown declines in flying squirrel abundance following thinning, empirically (but not necessarily statistically) higher numbers of squirrels in untreated stands compared to stands treated with thinning, or generally higher squirrel abundances in unthinned stands compared to thinned stands (Bull et al. 2004; Carey 2000, 2001; Gomez et al. 2005; Herbers and Klenner 2007; Holloway and Malcolm 2006, 2007; Holloway et al. 2012; Manning et al. 2012; Meyer et al. 2007a; Ransome et al. 2004; Waters and Zabel 1995; Waters et al. 2000; Wilson 2010). This includes arboreal rodent responses to the Forest Ecosystem Study (FES), a study specifically designed to test whether variable-density thinning (see Carey et al. 1999c) could accelerate the development of habitat for spotted owls and their prey in 50- to 60-year-old Douglas-fir (*Pseudotsuga menziesii*) forests of western Washington (Carey 2001, 2007; Wilson 2010).

The goals of this paper are to: (1) briefly highlight what is known about the ecology of each of the four major spotted owl prey species as it relates to thinning forests; (2) provide rationale for why thinning has negative consequences for some species in the near term; and (3) discuss possible management options and research directions that could benefit owls and their prey over the long term.

Flying Squirrels

Northern Flying Squirrels comprise 50 percent or more of the prey in spotted owl diets across much of Oregon and Washington (Forsman 1984; Forsman et al. 1991, 2004a), and most of the research into the ecology of owl prey in the Pacific Northwest has focused on this species. In general, two forest conditions can support relatively high numbers of flying squirrels—closed-canopy forest (old or young) with high stem density, and classic multi-layered old forest, with the latter generally providing the highest abundances (Wilson 2010).

Several habitat components have been associated with high-quality flying squirrel habitat, including understory cover, patch-level (40–80 m) changes in vegetation composition, large snags, large trees, ericaceous shrubs, high canopy cover, nearness to conifer forest, nearness to water, abundant down wood, large down wood, increased litter depth, and availability of fungi (Carey 1991, 1995; Carey et al. 1992, 1999a; Ford et al. 2004; Gomez et al. 2005; Holloway and Malcolm 2006, 2007; Hough and Dieter 2009; Lehmkuhl et al. 2004, 2006; Menzel et al. 2006; Meyer et al. 2007a, b; Payne et al. 1989; Pyare and Longland 2002; Rosenberg and Anthony 1992; Smith et al. 2004, 2005). However, these associations have not always held true across studies. For example, some forests that support high squirrel abundances have few large snags or little down wood (Wilson 2010). Likewise, some forests (including old forest) with relatively large quantities of snags, large live trees, or down wood have been found to support few squirrels (Carey 1995; Wilson 2010).

Part of the issue in associating squirrel abundance with habitat components is that most studies have only examined a limited number of forest conditions (e.g., comparing forest type A vs. B) or conducted analyzes based on broad categories (e.g., old forest vs. young forest). To help address this problem, Wilson (2010) used multivariate, structural equation models to

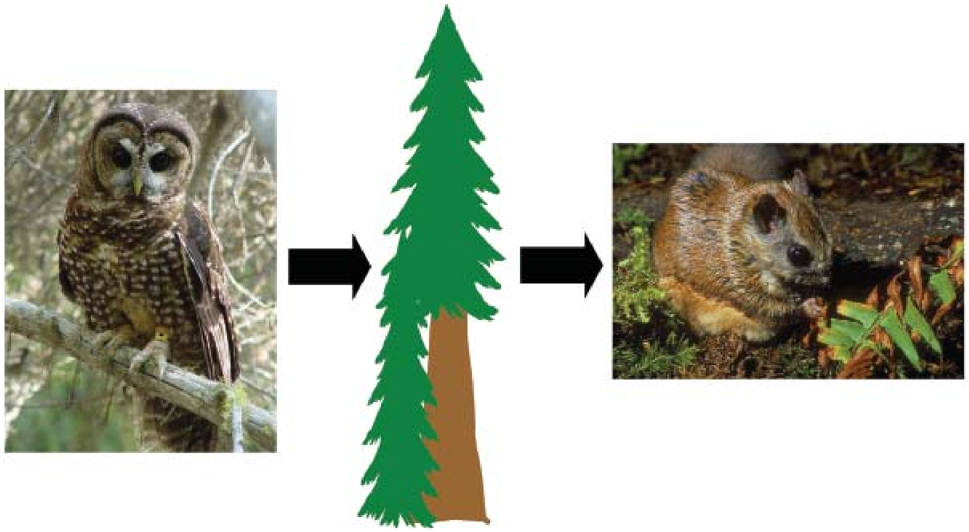

Figure 2—Forest structure, particularly in the midstory, provides the interface between predators like northern spotted owls and one of their major prey, northern flying squirrels. (Spotted owl photo by Stan Sovern; flying squirrel photo by Wes Colgan III.)

examine flying squirrel populations and habitat across 33 managed and unmanaged stands in the Puget Trough and Olympic Mountains physiographic provinces. Collectively, these stands were chosen to represent broad gradients in forest structural conditions found across low- and mid-elevation mesic forests west of the Cascade Range. Wilson (2010) found that important habitat components used in stand-level statistical models to predict flying squirrel abundance could all be associated with above-ground protective cover, and hypothesized that predation was a primary limiting factor regulating squirrels in these forests (Wilson 2010; fig. 2). Wilson suggested that structural occlusion (the degree to which physical structures inhibit detection of squirrels by predators and help squirrels escape direct attacks) was an important component of protective cover. When there is a high degree of occlusion in the midstory (e.g., high stem density, deep crowns, live crowns in the midstory layer, with few canopy gaps across the stand), it provides sufficient protection for squirrels to attain and sustain relatively high population levels (fig. 3). Even with relatively high predation rates (e.g., during spotted owl breeding years), enough female squirrels can survive each year in these forests to quickly restore populations to a relatively high level. In contrast, when there are too many gaps, too many large gaps, lack of a midstory canopy layer, or overall low stem density, squirrels succumb to predation pressure and few squirrels survive to reproduce.

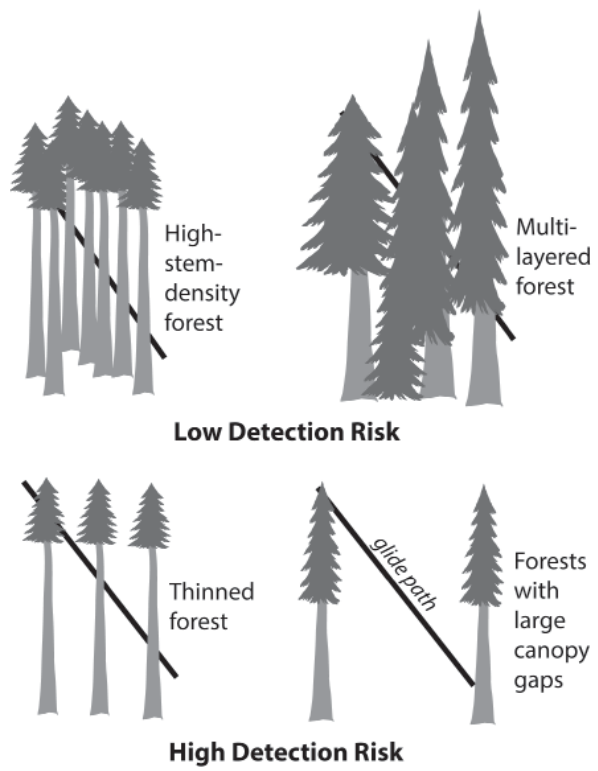

Figure 3—Black line illustrates a hypothetical glide path of a Northern Flying Squirrel and the structural occlusion created by different forest conditions that provide different levels of detection risk by avian and mammalian predators.

Most thinning prescriptions reduce occlusion in a way that appears to leave flying squirrels more vulnerable to predators than before thinning. This effect may be further compounded once the understory begins to develop in response to thinning, as abundance of forest-floor small mammals can quickly increase and, in turn, attract predators to the area that may opportunistically prey on vulnerable flying squirrels (e.g., spillover predation; Oksanen et al. 1992).

Red Tree Voles

Red Tree Voles are the most arboreal of all the rodents in the Pacific Northwest (Carey 1999). Similar to flying squirrels, the highest tree vole populations are found in old forests and forests with high stem density (Dunk and Hawley 2009). Tree voles also appear to be susceptible to increased predation brought on by reduced canopy density after thinning, as few are found in recently thinned forests (Forsman et al. 2004b). However, tree voles also have three additional ecological constraints that may make them especially susceptible to thinning. First, small trees in young forests generally have insufficient food resources (conifer needles) in a single tree to support breeding females, so individuals often forage in multiple trees surrounding their nests (Swingle and Forsman 2009). In closed-canopy forest, they can simply travel across interlocking branches to reach adjacent trees. Thinning breaks these connections, and voles must travel down (usually on bare, exposed boles) and across the ground to reach other trees. This not only increases their energetic demands, it also puts them at additional exposure to predation. Second, tree voles build nests composed of small twigs and conifer needles on platforms created by dwarf mistletoe (*Arceuthobium* spp.), epicormic branching, forked boles, and other irregularities in tree-branching patterns. Such nests need more support than the nests built of larger twigs and branches used by tree squirrels. If trees with complex limb structure, forked tops, broken boles, and dwarf mistletoe infections are removed during thinning, it may greatly restrict the ability of tree voles to find suitable nest substrates. Third, young tree voles have limited dispersal ability, and the absence of tree voles across much of northwest Oregon suggests that they may not be able to disperse across broad areas of intensively managed forest (Maser et al. 1981).

Woodrats

The Dusky-footed Woodrat and Bushy-tailed Woodrat are two of the most important species in the diet of Northern Spotted Owls, particularly in northwestern California, southwestern Oregon, and along the eastern slopes of the Olympic Mountains and eastern Cascades of Washington (Forsman 1975, 1991; Forsman et al. 2004a; Ward et al. 1998). The Dusky-footed Woodrat is a resident of California and southwestern Oregon, whereas the Bushy-tailed Woodrat occurs throughout much of the Pacific Northwest.

Sakai and Noon (1993) found that Dusky-footed Woodrats were most abundant in young (15- to 40-year-old) forests with brushy understories, with lower densities in seedling/shrub and old growth forests, and few or no woodrats in forests with merchantable timber but little understory. They hypothesized that young, brushy forest may provide important source areas for woodrats in adjacent old forests that are more suitable for spotted owls. Carey et al. (1999b) surveyed for woodrats in western Oregon and western Washington and suggested that the distribution of Dusky-footed Woodrats could be explained by a preference for dense shrub cover and the ability to consume plants toxic to other mammals. In contrast, Carey et al. (1999b) found that Bushy-tailed Woodrats were irregularly distributed, possibly as a result of their aggregated social structure (harems) that may allow predators to cue in and prey on family groups. In eastern Washington, Lehmkuhl et al. (2006b) found that large snags, dwarf mistletoe brooms, and partly decayed logs were good predictors of Bushy-tailed Woodrat abundance in dry forests. Lehmkuhl et al. (2006b) suggested that conventional fuels reduction prescriptions that reduced the amount of snags, dwarf mistletoe brooms, and logs would have a negative effect on Bushy-tailed Woodrat populations. However, there have been no subsequent studies conducted on woodrats in Washington or Oregon to further investigate these hypotheses.

Management Considerations

Thinning designed to create late-seral conditions in young forests has shown early success in promoting habitat for a number of vertebrate species. However, development of a midstory layer of trees appears important before species like flying squirrels and Red Tree Voles respond positively to thinning, and this will take time to develop. In fact, it may take several decades or longer for a midstory to develop to a point where it provides adequate protective cover and canopy connectivity. This is especially true when thinning young forests with small-crowned trees, sparse natural regeneration, and a large gap between the existing understory and the bottom of the tree canopy. The primary question, therefore, is how best to move landscape trajectories toward more complex forest while, at the same time, ensuring that the short-term negative effects on prey populations do not result in long-term negative effects for either spotted owls or their prey.

It is important to recognize that some arboreal rodents like flying squirrels have been found in fairly high numbers in forests that are relatively simple (e.g., high-stem-density young forest) or that lack habitat components often associated with flying squirrels (e.g., few snags, understory, or down wood; Wilson 2010). However, the ultimate goal when promoting owl prey habitat through thinning should be toward developing structurally and biologically complex forests that include multi-layered canopies including trees of varying sizes and species, snags, and down wood. Not only are these forests generally better habitat for spotted owls and their prey, they are the scarcest forest type on the landscape, they provide habitat for a wide array of other wildlife, and they help meet broader biodiversity goals including those outlined in the Northwest Forest Plan.

There are several specific management strategies that may help ameliorate some of the near-term negative effects of thinning on owl prey:

1. Accelerate and monitor mid-story development.

One of the primary objectives of variable-density thinning is to develop late-seral characteristics sooner by moving a forest through stages of forest development more quickly compared to leaving stands to develop as a result of natural stochastic events without management intervention (Carey et al. 1999c). However, without intentional focus on midstory tree growth, site-specific conditions can prevent or even forestall midstory development. For example, if a well-developed understory exists prior to thinning, it may outcompete new tree seedlings. If there is a limited shade-tolerant tree seed source, then midstory development may be patchy or sparse. In contrast, if the natural shade-tolerant seed source is too abundant, then regeneration competition may occur, slowing growth of the next layer of trees. Additionally, for a midstory to fully develop, new trees must grow tall enough to fill the structural void between the ground and the bottom of the existing overstory canopy. The larger this distance is, the longer it will take for a full midstory to develop.

There are a number of ways to promote a midstory. Underplanting can be considered for forests that lack a natural seed source of shade-tolerant trees. Patchy brush control could be used in conjunction with underplanting for thinning forests that have a well-established understory layer of aggressive shrub species like Salal (*Gaultheria shallon*). If shade-tolerant tree species regenerate aggressively on the forest floor, then early thinning of tree saplings can be considered. Because both thinning treatment and site-specific conditions affect development of a midstory, regular monitoring should occur to ensure that this critical forest layer continues along an accelerated path.

2. Include very young (<25-year-old) stands in the mix of stands targeted for creating late-seral forest.

High-quality habitat for spotted owl prey might be achieved most quickly by treating very young (<25-year-old) stands with pre-commercial thinning to stimulate midstory development, so that there is a much smaller vertical gap to fill with mid-story trees as compared to the gap that might develop if thinning is delayed until stands reach merchantable size. This does not preclude thinning older (25- to 80-year-old) stands, especially given their dominance on the landscape, but adding young stands could help fill a temporal niche in the presence of high-quality prey habitat across the landscape through time. Thinning at this early age is also not likely to affect existing flying squirrel populations, as they generally avoid highly-stocked Douglas-fir forests <25 years old.

3. Keep some young high-density forest on the landscape.

Although optimal spotted owl habitat is multi-layered older forest, high-stem-density forests do support owl prey and are used by owls as foraging habitat. High-stem-density forest can remain in a condition favorable for owl prey for several decades or more and could serve as a reservoir for abundant prey populations while thinned stands develop into suitable habitat over the long term. It is important to recognize, however, that high-stem-density forest can also quickly (<20 years) leave this stage, as suppression mortality and localized stochastic events (e.g., ice storms, wind, disease, insect outbreaks) remove existing trees (Wilson 2010).

4. Use defensible buffers to protect existing high-quality habitat.

Fire suppression in dry forests over the last century has created habitat that in some cases is favorable for owl prey, but such habitat is also vulnerable to stand-replacing fire events. It may be important to try to keep some of this habitat on the landscape until new suitable habitat becomes available. Strategically placed buffers with reduced fuel loads may help provide protection from catastrophic fire for these areas. Keeping such buffers as narrow as possible and with as much canopy cover as possible would be important to allow squirrels to travel and dispersal across the broader landscape and to prevent the buffers from possibly becoming predator traps.

5. Explore alternative thinning prescriptions.

It may be possible to develop new thinning prescriptions that keep moderately high populations of arboreal rodents in young forests while still achieving long-term management objectives for the stand. One such approach would be developing prescriptions that focus solely on skips (patches of trees left unthinned) and gaps (removal of patches of trees). This strategy is in marked contrast with most current prescriptions that typically thin throughout a stand (with or without delineated skips or gaps). Under such a strategy, it may be important to keep canopy-gap sizes small (100-400 m^2 per gap with <30 percent of the total stand area in gaps), and horizontal occlusion high, as there may be thresholds in the amount and extent of gaps distinguishing high- and low-quality habitat (Wilson 2010). A gap-skip-only approach may be most feasible in either young forests with sufficient bole density to provide structural occlusion at a relatively small scale (generally less than 20–40 meters), or in young mixed-conifer forest with a short, but existing multiple-layered canopy. Because there have been no demonstrated examples of the efficacy of such a strategy, this approach should be experimentally evaluated (testing size, patterning, and spatial distribution of canopy gaps) before any widespread application. A gap-skip-only approach may be particularly useful when the management goal is to reduce fuel loads as part of non-commercial restoration efforts in fire-prone forests, without necessarily wanting to accelerate tree regeneration or understory plant development. Rather than the common practice of removing the vertical component of a forest (e.g., ladder fuels) to reduce fuel loads, this approach focuses on creating horizontal patchiness to reduce fuel loads. Such a strategy may also have stronger ecological merit, in that it may more closely mimic natural fire events for some regions (Agee 1993; Harrod et al. 1999; White 1985). Depending on the size of canopy gaps, a skips-gaps-only strategy may also be suitable for developing late-seral conditions in more mesic forest, but only if gaps were large enough to accelerate mid- and understory development compared to leaving forests to develop on their own.

Future Direction

The need for sufficient structural occlusion to protect arboreal rodents from predators provides an explanation for why flying squirrels and tree voles decline in abundance following thinning, and provides a rationale for steps that might be taken to ameliorate this undesired effect. However, this hypothesis was developed from studying flying squirrels in mesic forests of western Washington, and may not reflect other limiting factors elsewhere within the range of the Northern Spotted Owl. For example, sufficient mid-story structure may be a critical component of xeric forests supporting high flying squirrel abundances in southwest Oregon, but there may also be minimum moisture thresholds needed for adequate year-round fungal production that restrict abundant populations to higher elevations, north-facing slopes, or near permanent water sources (T. Wilson, unpublished data). Likewise, predator load (predator species diversity and abundance) may also play a crucial role in regulating arboreal rodent populations for some forests. Thus there is need to further test the applicability of current empirical models using similar population and habitat analyses elsewhere. Of particular importance would be evaluating habitat in areas within and outside the range of the Northern Spotted Owl where there have been few or no population-level studies or where there are habitat conditions substantially different than those found in western Washington. For flying squirrels, these include mixed conifer forests in southwestern Oregon, high-elevation sites throughout the Oregon and Washington Cascades and Olympic Peninsula, dry forests on the eastern slopes of the Oregon Cascades and southern Washington Cascades, and throughout much of the forested area in eastern Oregon and Washington (e.g., Blue Mountain and Northern Rocky Mountain provinces; fig. 4).

For Red Tree Voles, empirical research is needed to better understand the effects of thinning on vole populations, and to determine if thinning prescriptions could be modified to allow tree voles to persist in thinned stands. There is also a basic need for understanding factors that influence the persistence of tree voles in young stands. Included in this is a better understanding of why tree voles are apparently slow to disperse through young stands, and whether or not specific management actions could improve dispersal rates.

There is also a need for further investigation into the habitat needs of both species of woodrats, given the lack of data on their distribution and habitat requirements. Recent controversy over the effects of broad-scale fire on distribution of spotted owls (e.g., Hansen et al. 2009; Spies et al. 2010) suggests that better understanding of woodrat ecology could be useful in informing such debates. For example, large stand-replacing fires would substantially reduce or eliminate flying squirrel populations, given the removal of above-ground predation cover. Such fires, however, may promote dense shrubs and early stages of forest development favorable to Dusky-

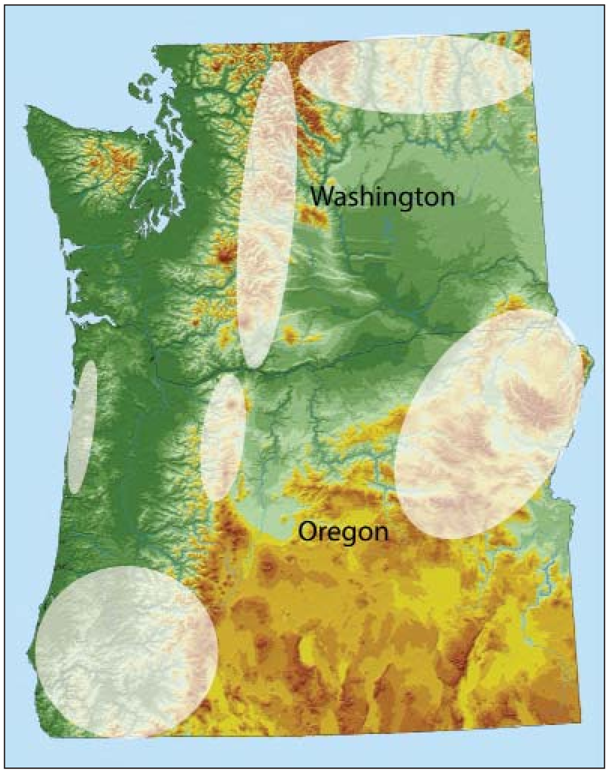

Figure 4—General areas (ovals shaded in white) of Oregon and Washington with forest types and conditions that could help strengthen an existing regional model defining high-quality habitat for flying squirrels and improve understanding of spotted owl prey ecology (Wilson 2010).

footed Woodrats, which could allow owls to persist in partially burned areas.

Finally, the continued decline in spotted owl populations throughout their range highlights the urgency of further understanding and appropriately managing for owl prey across a range of spatial scales. An important component of this is planning management activities at the landscape (e.g., fifth-field watershed or larger) and regional levels in a way that merges the spatial and temporal effects of thinning on spotted owl prey with the ecology and habitat needs of these species. Such efforts could include modeling of the: (1) current stand-level distribution of existing high-quality habitat for arboreal rodents across the region; (2) permeability of the landscape for arboreal rodents (e.g., capacity of the landscape to allow dispersal and colonization over time); and (3) projected changes in the amount and distribution of high-quality prey habitat over time under different management scenarios and stochastic events.

In summary, thinning has yet to be shown effective for promoting habitat for arboreal rodents like flying squirrels and Red Tree Voles. However, the importance of midstory cover and connectivity suggests that the trajectory of some thinned stands will eventually result in habitat conditions favorable for these species. Extra effort will be needed to ensure that the short-term negative effects of thinning on some owl prey species do not have long-term negative consequences for owls or their prey. One of the fastest and most efficient ways to move forward would be for researchers and forest managers to collaborate on projects such as testing novel silvicultural prescriptions, assessing habitat and prey abundance in different forest types, and developing landscape and regional management models and strategies for owl prey. A focused research-management effort over the next 10 years could greatly advance our understanding of owl prey ecology and hopefully result in management practices that support healthy populations of both owls and their prey in the Pacific Northwest over the long term.

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Above-ground Carbon Storage, Down Wood, and Understory Plant Species Richness after Thinning in Western Oregon

Julia I. Burton, Adrian Ares, Sara E. Mulford, Deanna H. Olson, and Klaus J. Puettmann

Abstract

Concerns about climate change have generated worldwide interest in managing forests for the uptake and storage of carbon (C). Simultaneously, preserving and enhancing structural, functional, and species diversity in forests remains an important objective. Therefore, understanding trade-offs and synergies among C storage and sequestration and diversity in managed forests is key to achieving these multiple objectives. Using the experimental framework of the Density Management Study in western Oregon, we examined the relationships among a suite of thinning treatments, above-ground carbon stocks, and understory vascular plant species richness. Six years following treatment implementation, total above-ground C declined with residual density. Total above-ground C in the high-density thinning treatment (300 trees·ha^{-1}) did not differ statistically from the untreated control treatment (~370–775 trees·ha^{-1}), and these two treatments stored 33 percent and 61 percent more C above ground, respectively, than the moderate density (200 trees·ha^{-1}) and variable density (300, 200, 100 trees·ha^{-1}) treatments. Differences among treatments were primarily related to reductions in the live overstory pool. For all treatments, C stored in the live overstory > large down wood > snags > stumps > small down wood. Coarse down wood (CDW) comprised over 30 percent of the total above-ground C storage. Most of the C in the dead pools appears to be legacy (pre-thinning) material; 50–95 percent of snags, stumps, and large down wood were in intermediate to late stages of decomposition. Between years 6 and 11 post-treatment, the overstory C increment tended to decline with lower residual density, although this trend was not statistically significant. In contrast, understory plant species richness was greater in all thinning treatments than in untreated controls. Relationships varied slightly among treatments. Moderate- and variable-density thinning treatments resulted in a negative relationship between understory plant species richness and above-ground C, while no relationship was observed in the high-density treatment and unthinned control. Results suggest that thinning increases plant species richness, implying that there is a trade-off between management for understory plant species richness and above-ground C storage.

Keywords: Biodiversity, carbon sequestration, coarse woody debris, density management, diversity, understory vegetation.

Julia I. Burton is a research associate, Department of Forest Ecosystems and Society, Oregon State University, Corvallis, OR 97331; julia.burton@oregonstate.edu; **Adrian Ares** is the director, Sustainable Agriculture and Natural Resource Management Collaborative Research Support Program, Virginia Tech, Blacksburg, VA 24060; **Sara E. Mulford** was a research assistant, Department of Forest Ecosystems & Society, Oregon State University, Corvallis, OR 97331; **Deanna H. Olson** is a research ecologist, USDA Forest Service, Pacific Northwest Research Station, Forestry Sciences Laboratory, 3200 SW Jefferson Way, Corvallis, OR 97331; **Klaus J. Puettmann** is Edmund Hayes Professor in Silviculture Alternatives, Department of Forest Ecosystems and Society, Oregon State University, Corvallis, OR 97331.

Introduction

Concerns about climate change have resulted in worldwide interest in managing forests to mitigate anthropogenic increases in atmospheric carbon dioxide (Gower 2003; Pacala and Socolow 2004; McKinley et al. 2011). Simultaneously, preserving and enhancing structural, functional, and biological diversity in forests remain important management objectives. However, few studies have examined effects of thinning on carbon (C) storage, as well as the trade-offs and synergies among managing for C storage and sequestration, and species diversity in managed forests.

The majority of above-ground C in temperate forests is stored in the live overstory: the ratio of understory C to overstory C is typically <0.005 (Ares et al. 2007). However, the majority of plant diversity resides in the understory. The ratio of understory species richness to overstory species richness ranges from 2 to 10, with approximately 80 percent of the plant species restricted to the understory (Gilliam 2007). Forest thinning can increase resources available for residual trees in the overstory as well as for understory plants (Gray et al. 2002), resulting in increases in tree diameter growth rates (Dodson et al. 2012) and understory plant species diversity (Fahey and Puettmann 2007; Ares et al. 2009, 2010). However, thinning may decrease C storage within forest stands initially and over time (Gower 2003), with the magnitude depending on the percentage of the stand harvested, harvesting interval, and spatial pattern of harvesting (dispersed vs. aggregated) (Harmon et al. 2009). As a result of these positive and negative effects of thinning on understory plant species richness and above-ground C, respectively, a negative relationship between above-ground C storage and understory diversity is predicted (fig. 1). Here we examine the short-term relationships among forest thinning treatments, total above-ground C (including live overstory, coarse down wood [CDW], snags, and stumps), and understory plant species richness. Because rates of soil C accumulation are generally low over the period of a single rotation, the fate of above-ground C is an important focus of mitigation efforts (Gower 2003).

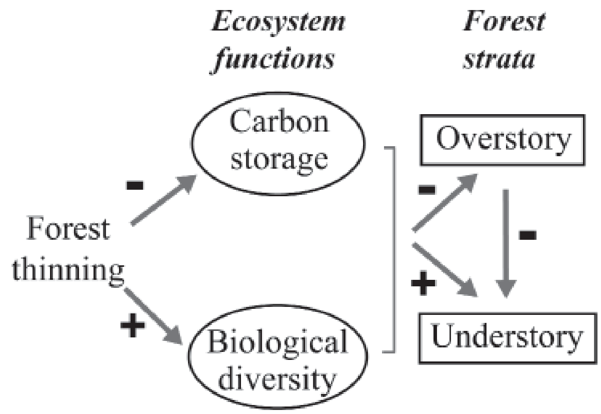

Figure 1—Conceptual diagram illustrating the contrasting effects of thinning on carbon storage (-) and biodiversity (measured as understory plant species richness) (+). Here we hypothesize that a trade-off emerges as a result of the negative effects of thinning on carbon storage in the overstory and positive effects of thinning on plant diversity in the understory.

Methods

Experimental Design

Using the experimental framework of the Density Management Study (Cissel et al. 2006), we analyzed the effects of a suite of thinning treatments on above-ground C stocks and understory plant species richness. The thinning treatments, replicated seven times across the Coast Range and northern Cascade Range of western Oregon, included three levels of residual density: 1) a high-density treatment with 300 trees·ha^{-1} (HD); 2) a moderate-density treatment with 200 trees·ha^{-1} (MD); and 3) a variable-density treatment with 300 trees·ha^{-1}, 200 trees·ha^{-1} and 100 trees·ha^{-1} (VD). In addition to an even distribution of residual trees (i.e., dispersed retention), 3 to 11 percent of the treatment unit was left unthinned in circular leave-island reserves (patches of undisturbed forest trees) in the HD and MD treatments, and 3 to 10 percent of the treatment unit was cut in circular gap openings in the MD treatment. In the VD treatment, 8.2 to 10.3 percent of the area was preserved in leave islands, and 8 to 17 percent of the area was left in circular gap openings. Gap openings and leave islands were 0.1, 0.2, and 0.4 ha in size. In addition to comparisons among the three thinning treatments, the effects of thinning were compared with unthinned control areas (CON; approximately 600 trees·ha^{-1}). Each of the seven study sites consisted of large forested stands (94 to 131 ha, with controls on 16 to 24 ha and each thinning treatment implemented on 14- and 69-ha parcels) to allow for operational application of treatments and thus avoid the need to scale-up the experimental results (fig. 2).

Field sampling

Seventy-seven permanent 0.1-ha circular plots were installed in each site to sample overstory trees. Plot centers were located randomly within treatment units (n = 21 in HD, MD and VD, n = 14 in controls) using a random point generator (Cissel et al. 2006). Plot boundaries were constrained to be ≥50 m from treatment unit boundaries and non-overlapping. Within each overstory plot, four 0.002-ha circular understory vegetation subplots were installed 9 m from plot centers in each cardinal direction. Overstory and understory vegetation measurements were taken during the summer 6 and 11 years after thinning. In each overstory plot, all trees ≥5.1 cm dbh (diameter at 1.37 m above ground) were numbered, identified by species, and measured for dbh using a diameter tape. We also measured height of 16 trees per plot (10 conifers and 6 hardwoods) using a laser hypsometer (Laser Technology, Centennial, CO, USA). In the understory subplots, we visually estimated total cover of each vascular plant species present using cover classes: 1 percent, 5 percent, and then in 10 percent increments. Overstory trees and shrubs >6 m in height were excluded from cover estimates. Species richness was calculated as the number of vascular plant species recorded in the four understory subplots (a total area of 80 m^2). Detailed vegetation measurement protocols and data analyses have been reported previously (Cissel et al. 2006; Ares et al. 2009, 2010).

Six years after thinning, all standing dead trees (snags) and three types of above-ground

Figure 2—Example of a treatment layout at one site, Bottomline. The high-density treatment retained 300 trees·ha^{-1}, the moderate-density treatment retained 200 trees·ha^{-1}, and the variable-density retained variable densities of trees: 300 trees·ha^{-1}, 200·trees·ha^{-1} and 100 trees·ha^{-1}. The density of untreated control plots was approximately 600 trees·ha^{-1}. Leave islands were maintained in all thinning treatments, canopy gaps were also created as part of the moderate- and variable-density treatments.

coarse down wood (CDW) were sampled: stumps, large CDW (≥25.4 cm in diameter and ≥0.3 m length), and small CDW (<25.4 cm in diameter and <0.3 m length). All stumps (≤1.37 m in height measured on the uphill side) within the northeast quarter-section of the overstory plot were measured for diameter and height. We measured large CDW along four 12.9-m transects connecting the centers of the understory subplots. Down wood that crossed transects multiple times (e.g., forked logs or long logs that spanned two transects) was counted as separate pieces (Harmon and Sexton 1996). Elevated dead wood at an angle >45° above the ground was not considered CDW, but was counted as a snag. For each piece of down wood, we measured the diameter at the point where it intercepted the transect line. Small CDW was measured within the understory subplots. Only pieces with their large ends inside the subplots were included in the measurement. Diameter at both ends and total length of each piece (including any part extending outside the subplot) were measured. Diameter measurements of large and small down wood pieces were taken with calipers, length was measured using a measuring tape. Decay stage was characterized using a five-class scale from 1 (least decayed) to 5 (most decayed) as per Maser et al. (1979). All snags ≥1.37 m tall and ≥5.1 cm dbh in the overstory plots were identified as such and measured for dbh.

Calculations of Above-ground C Storage

We derived above-ground C storage from biomass using the specific gravity and C-concentration values reported by Ares et al. (2007). A carbon content of 0.5 kg C·kg^{-1} was used for tree species not listed in Ares et al. (2007). Live tree biomass (bole, bark, live and dead branches, and foliage components) was estimated using measurements of dbh and height in allometric equations using BIOPAK version 2.50 (Means et al. 1994). Volumes of snags, stumps, and large and small down wood were calculated and converted to biomass and C stores by decay class using the specific gravity (mass per unit volume) and C-concentration values reported by Ares et al. (2007).

Statistical Analysis

A mixed-effects model was used to examine thinning treatment effects on above-ground C, differences in live C in the overstory between year six and year eleven post-treatment, vascular plant species richness six years after thinning, and differences in understory plant species richness between year six and year eleven post-treatment. Analyses were performed on the averages of all plots (subsamples) within a treatment unit. Thinning treatment was modeled as a fixed effect and site was modeled as a random effect in order to control for underlying differences among sites (Littell et al. 1996). Post-hoc pairwise comparisons of treatment means used orthogonal contrasts with one degree of freedom, and Tukey's HSD was used to correct p-values for multiple comparisons.

Additionally, we examined the relationship among understory plant species richness, experimental thinning treatments (HD, MD, VD, and CON), and above-ground C stocks at the plot scale using a mixed effects model (Littell et al. 1996) of fixed effects (treatments and above-ground C) and random effects (site, treatment x site). Plots were nested within a treatment unit and site. We examined main effects of thinning treatment, above-ground C, and the interaction between richness and above-ground C. In the case of a significant interaction, we tested the null hypothesis of no relationship between understory plant species richness and above-ground C for each treatment (CON, HD, MD, and VD) and then compared treatment slopes to test the null hypothesis of equal slopes between treatments (n = 6 comparisons). P-values were not corrected for multiple comparisons. We used the mixed procedure in SAS 9.2 for all statistical analyses reported here (SAS Institute 2004).

Results

Carbon Stocks Year 6 Post-treatment

By the sixth growing season following thinning, above-ground carbon (C) decreased with decreased residual density (table 1). Moderate- and variable-density treatments had lower levels of C storage than untreated control and the high-density retention treatment (difference = 78.2 ± 13.0 $Mg{\cdot}ha^{-1}$, T = -6.01, $p < 0.001$). Among the above-ground stocks, the majority of C was stored in the live overstory (~68 percent) relative to dead pools: coarse down wood, snags, and stumps (~32 percent cumulatively). Within the live overstory, the majority of C was stored in the bole wood (~72 percent of the live overstory C), although nearly 30 percent of the live overstory C was found in bark, branches, and foliage. Among the dead pools, the largest stock of C was in CDW with diameters ≥25.4 cm (large CDW) and in later stages of decomposition (classes 3–5).

Overstory C was greater in year eleven than in year six in all treatments ($p < 0.05$). Increments ranged from 12.6 to 19.1 $Mg{\cdot}ha^{-1}$, amounting to an increase of 10 to 15 percent. The difference in overstory C between year eleven and year six post-treatment increased with increases in residual density (i.e., CON > HD > MD > VD); however, while this trend was apparent, there was no evidence for significant differences among treatments statistically speaking.

Plant Species Richness Six years after Thinning

In contrast to C, understory plant species richness was higher ($p < 0.001$) in all thinning treatments than the untreated controls during the sixth growing season following treatment (table 1). Plant species richness, however, did not differ among high-, moderate- and variable-density thinning treatments ($p > 0.9$). Thinned treatments (HD, MD, and VD) contained 11.9 ± 1.4 species more than untreated controls on average (T = 8.62, $p < 0.001$). Understory plant species richness decreased from year six to year eleven in the moderate- and variable density thinning treatments, while significant changes in richness were not observed in the CON and HD treatments (table 1). On average, understory plant species declined by 4.4 ± 0.6 more species between years 6 and 11 in MD and VD relative to CON and HD treatments (T = -7.29, $p < 0.001$).

Carbon – Understory Plant Species Richness Trade-off

In year six post-treatment, species richness was related to above-ground C and thinning treatment; evidence for an interaction between treatment and above-ground C was weak ($p = 0.07$). This weak interaction resulted from the lack of a relationship between above-ground C

Table 1—Treatment averages for carbon (C) and diversity: above-ground (C) in year 6 post-treatment, differences and live overstory C between years 6 and 11 post-treatment, understory plant species richness in year six post-treatment, and differences in understory plant species richness between years 6 and 11 post-treatment. Thinning treatments = HD = high-density retention, MD = moderate-density retention, and VD = variable-density; untreated control (CON).

	Thinning treatment			
	CON	HD	MD	VD
Carbon				
†Above-ground C	269.2^{a}	223.7^{a}	168.0^{b}	168.5^{b}
Difference in live overstory C (Yr. 11 – Yr. 6)	19.1	18.2	17.1	12.6
Diversity				
†Understory plant species richness	15.9^{a}	27.3^{b}	28.2^{b}	27.9^{b}
†Difference in richness (Yr. 11 – Yr. 6)	1.6^{a}	-0.5^{a}	-4.1^{b}	-3.7^{b}

†Different superscripts indicate evidence for significant differences between treatments ($p < 0.05$).

and understory plant species richness in both the untreated control (estimate = -0.01, T = -1.1, p = 0.27) and HD treatment (estimate < -0.01, T = -0.01, p = 0.99). In contrast, in the MD (estimate = -0.02, T = -2.5, p = 0.013) and VD (estimate = -0.03, T = -3.3, p = 0.001) thinning treatments, the negative relationships between understory plant species richness and C suggest a trade-off between diversity and C at fine spatial scales in MD and VD treatments (fig. 3). There was evidence for different relationships between understory plant species richness and above-ground C in between HD and MD (albeit weak, estimate = 0.02, T = 0.96, p = 0.051) and between VD and HD (estimate = 0.03, T = 2.46, p = 0.014). In contrast, there was no evidence that the slope of the relationship differed between HD and CON (estimate = -0.001, T = -0.87, p = 0.382).

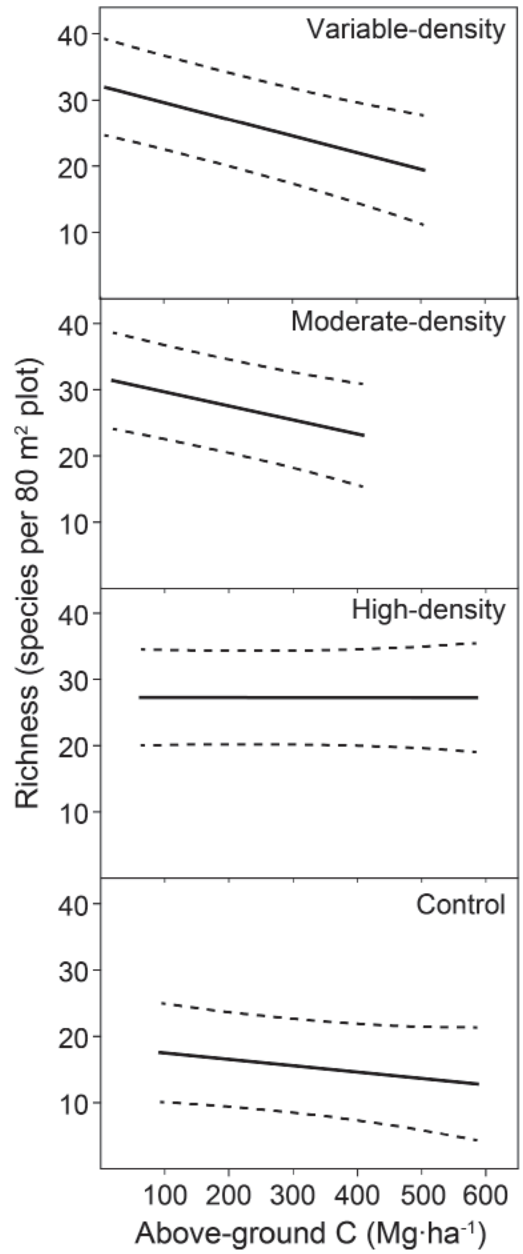

Figure 3—Relationship between understory plant species richness and above-ground carbon by thinning treatment (VD = variable-density, MD = moderate-density, HD = high-density, and CON = untreated control) 6 years post-treatment. Richness is expressed for each plot as the aggregate number of species in four subplots (80 m^2). Solid lines show model-predicted relationship; dashed lines show 95 percent confidence intervals. The random effect, site, is not shown.

Discussion

In addition to maintaining or restoring diversity and ecosystem function, forest managers are now tasked with managing forests to mitigate anthropogenic carbon dioxide in the atmosphere. Tree boles removed during timber harvesting can significantly reduce C storage within forest ecosystems (Harmon et al. 2009). Such patterns are associated with both C extraction and with transient declines in above-ground net primary productivity following harvesting (e.g., Dyer et al. 2010).

Although the live overstory comprises the largest stock of C above ground, nearly one-third of the total above-ground C at our study sites is stored as snags and coarse down wood. These proportions are likely associated with legacy material that resulted from the previous clearcutting operation, as the majority of CDW is in the large size classes (≥25.4 cm diameter) and later stages of decay. The relative importance of this portion of the CDW stock of C will decline as the legacy wood continues to decompose. Further, it appears that natural mortality rates following thinning are not sufficient to augment the size of the CDW stock (Dodson et al. 2013). CDW volumes are typically lower in harvested relative to unharvested stands (e.g., Burton et al. 2009). Therefore, management activities such as creation of down wood may be warranted to achieve C storage as well as structural goals articulated in the initial study plan (Cissel et al. 2006). Creation of down wood or snags would prevent future C deficits in CDW in late stages of decay as well as below ground in the soil.

Although not addressed here, the spatial distribution of C from large CDW was variable and not related to thinning treatments or topographic position (Julia Burton, unpublished data). Thus, managers may consider establishing a spatially heterogeneous, patchy distribution of fresh CDW. Alternative designs for down wood placement include augmenting wildlife habitat and dispersal corridors, for example orienting wood toward streams and over headwater ridgelines to increase connectivity and the dispersal of animals among riparian buffers (Olson and Burnett 2013).

While thinning resulted in reductions in C storage, it increased understory plant species richness. Furthermore, the negative relationship between species richness and above-ground C suggests that thinning results in a trade-off between plant diversity and C. As a result of the recent logging history, the majority of forests in the Coast Range are younger and in earlier, even-aged stages of structural development and have been managed at high densities to facilitate wood production. Because spatial and temporal variability in stand structure is associated with patterns of biodiversity within forests (Ares et al. 2009), recent management efforts have focused on using thinning to accelerate the development of structural heterogeneity and restore patterns of diversity within younger stands (Bauhus et al. 2009). Trade-offs between plant species richness and C therefore suggest that forest management for maximum C storage may be at odds with management for biodiversity if thinning removes C stored in live tree boles.

A number of mechanisms may contribute to the negative relationship between above-ground C and plant species richness. First, removal of live overstory C during thinning creates gaps in the canopy that function to alter the microclimate and increase resource availability above and below ground (Gray et al. 2002; Burton 2011). Such changes in the local abiotic environment interact with the traits of the individual plant species in the understory to drive changes in rates of colonization and competitive exclusion to affect the composition and structure of the plant communities (Roberts and Gilliam 1995; Burton 2011). Specifically, increases in resource availability coupled with physical disturbance of the forest floor may explain the increased levels of plant species richness (Fahey and Puettmann 2007) and coexistence of early- and late-seral species six years after treatments (Ares et al. 2009, 2010). Declines in richness overtime may be associated with increased rates of competitive exclusion of late-seral species by competitive early-seral species (Ares et al. 2010). Harvesting can facilitate the invasion of non-native or weedy plant species (Scheller et al. 2002; Aubin et al. 2007), and our previous studies have reported that introduced species richness and cover was greater in the thinning treatment units relative to the untreated controls. However, non-native species comprised a small proportion of the response of richness and cover (Ares et al. 2009). Relationships between understory plant species richness and C stored as CDW are likely more variable and complex relative to C stored in the overstory. Relationships between C and understory species richness may vary in strength and direction (positive or negative) among different pools of CDW (small vs. large pieces, early vs. late stages of decay, snags and stumps). For instance, the recruitment of fresh down wood may decrease plant species richness through its direct effects on plant survival and indirect effects on the local environment (e.g., increased shading and interference). Over time, as wood decays, such relationships may become positive as interference decreases and resource availability increases (e.g., moisture and nutrients released from decaying wood) (Campbell and Gower 2000; Spears et al. 2003). Such effects may be mediated by C storage in the overstory. For instance, CDW shading may have a negative effect on understory plant species richness in a shady understory where overstory C is high, but a positive effect in an opening (low overstory C) where it may mediate environmental extremes such as temperature. Apparent trade-offs between above-ground C and understory plant species richness reflect the net result of many complex relationships among forest structure, resource availability and the characteristics of the plants in the regional species pool (Burton et al., in press).

Finally, if forests are to be managed to mitigate global climate change by reducing atmospheric carbon dioxide, simply looking at the effects of timber harvesting on C storage and sequestration within a given stand is not the whole story, and may not reflect the net effects of thinning in mitigating the global C balance (Gower 2003; McKinley et al. 2011). As we demonstrate here, thinning can reduce above-ground C storage in the short term by removing woody biomass, reducing the input of CDW, and leading to reductions in the size of the CDW C stock. Such activities can also lead to reductions in below-ground C stocks in the organic horizon and soil (Gower 2003). However, over the life cycle of the wood (from establishment to decomposition, be it in the forest or in a forest product), thinning effects on C storage depend on whether forest products are long-lived, and on carbon-emissions during harvesting operations and processing (Gower 2003; McKinley et al. 2011).

Conclusion

Linkages between carbon (C) storage and understory plant diversity may arise from a variety of processes and mechanisms operating over different temporal and spatial scales. Our work suggests that the net effect of these interactions is a trade-off between understory plant species richness and C storage (see also Burton et al., in press). Thus, forest management aimed to increase above-ground C storage by maintaining high-density forests may negatively affect aspects of restoration and maintenance of biodiversity within stands, and vice versa. Integrating C storage goals into a larger conservation-oriented management scheme may require accepting some losses and managing trade-offs by maintaining early seral habitat within dense C-rich stands or retaining live and detrital C in regenerating early-seral stands. Alternatively, plans could separate these goals across temporal or spatial scales.

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Green-tree Retention, Ectomycorrhiza Diversity, and Ectomycorrhizal Spore Inoculum

Daniel L. Luoma and Joyce L. Eberhart

Abstract

Studies from the Pacific Northwest indicate that forest management activities can reduce the richness and abundance of ectomycorrhizal fungi (EMF). Development of management approaches to sustain these essential organisms in forests has been hampered by a lack of knowledge of EMF community structure, diversity, and spatial and temporal variability across stands and landscapes. The ectomycorrhizal fungus community, as seen on root tips, changes significantly in disturbed sites compared to nearby undisturbed, nearby forests. These changes might be due as much to environmental and biotic factors as to loss of host trees. It has been found that seedlings transplanted from mature forests into harvested gaps declined in both EMF species richness and abundance. Seedlings closest to an uncut forest edge had the highest values of both response variables. In various studies, root colonization percentage and EM diversity have been observed to decline with distance from mature trees. These outcomes and others have led several researchers to conclude that green-tree retention is important for the recovery of EM fungi in harvested sites. Inoculating bare-root conifer seedlings with spores of ectomycorrhizal fungi has been promoted to reduce transplant shock, improve growth and nutrient uptake, and improve survival of Douglas-fir (*Pseudotsuga menziesii*) out-planted on sites subjected to operational forest management. Here, from an unpublished study, we summarize the effectiveness of EMF spore inoculum to increase survival and growth. That study also compared locally sourced spore inoculum with inoculum obtained from a different ecoregion. On that particular green-tree retention site, EMF spore inoculum was largely superfluous with regard to seedling mortality or growth. It was postulated that the site had sufficient EMF inoculum, either as living mycelium or spores, to rapidly colonize the bare-root Douglas-fir seedlings that were tested. We concur with other researchers that green-tree retention can be beneficial in maintaining EMF diversity and inoculum potential on a site.

Keywords: green-tree retention, ectomycorrhizae; *Rhizopogon*; soil ecology

Ectomycorrhizal Fungi

Ectomycorrhizal symbioses are formed on about 8000 plant species (Dahlberg 2001) and a current estimate of the number of ectomycorrhizal fungus (EMF) species is 7750 (Rinaldi et al. 2008). Most of the dominant and economically important timber species in the Pacific Northwest are ectomycorrhiza (EM)-dependent, including all members of the pine, oak, and birch plant families (Pinaceae, Fagaceae, and Betulaceae) (Smith and Read 2008). Douglas-fir (*Pseudotsuga menziesii* (Mirbel) Franko) has an estimated 2000 EMF symbionts throughout its range (Trappe 1977). Douglas-fir will not grow in soil without ectomycorrhizal fungi (Trappe and Strand 1969).

Ectomycorrhizal fungus diversity is an important attribute of Pacific Northwest forested ecosystems, for instance in stabilizing below-

Daniel L. Luoma is an assistant professor, senior reseatch, and **Joyce L. Eberhart** is a senior faculty research assistant, Department of Forest Ecosystems & Society, Oregon State University, Corvallis, OR 97331; daniel.luoma@oregonstate.edu.

ground processes after disturbances (Perry et al. 1989). Seedlings associated with a high diversity of EMF may be better-adapted to disturbance as compared to seedlings with lower diversity (Simard et al. 1997). In addition, high EMF diversity seems to increase trees' competitive abilities. A laboratory study supported this hypothesis by documenting that *Pinus patula* Schiede ex Schltdl. & Cham. seedlings inoculated with two species of EMF grew taller and put on more biomass than seedlings inoculated with only one EMF species (Sudhakara and Natarajan 1997).

Ectomycorrhizal fungi also form a functional guild linking primary producers to soil systems, are important in ecosystem response to disturbance (Perry et al. 1989), and may be sensitive indicators of environmental changes (Temorshuizen and Schaffers 1987; Termorshuizen et al. 1990; Arnolds 1991; Arnebrant and Söderström 1992). Ectomycorrhizal fungi are, with few exceptions, asco- and basidiomycetes and mostly produce macroscopic sporocarps in the form of mushrooms and truffles (epigeous or above-ground fruiting bodies, and hypogeous or below-ground fruiting bodies, respectively). Sporocarps produce the spores that disseminate the species and provide for genetic recombination within and among populations. Many species, especially those that produce truffles, are also important dietary items for vertebrates and invertebrates: some small mammal species rely on them for over 90 percent of their diet (Maser et al. 1978; Hayes et al. 1986; Claridge et al. 1996; Carey et al. 1999; Jacobs and Luoma 2008).

Nursery-grown seedlings that have not been otherwise inoculated with EMF arrive at a planting site with "greenhouse-adapted" fungi forming the mycorrhizae on their roots, and some roots may be non-mycorrhizal. Species diversity is low and the fungi are not usually well adapted to the rigors of the out-planting site (Alvarez and Cobb 1977; Castellano and Molina 1989). Over time, the nursery fungi are largely replaced by indigenous EMF species (Bledsoe et al. 1982). The rate of colonization by native fungi is dependent on the inoculum potential of the out-planting site. Harsh sites tend to have low inoculum potential. Hence, applying spore inoculum of species that vigorously colonize seedling roots on harsh sites should increase growth and survival of the out-planted nursery stock (Amaranthus and Perry 1987).

The Diamond Lake Ranger District (Umpqua National Forest) has been inoculating bare-root conifer stock with spores of EMF just before out-planting seedlings onto regeneration sites. Spores from *Rhizopogon* species are a dominant component of the inoculum. The genus *Rhizopogon* is named for its typically conspicuous rhizomorphs that efficiently transport water, conferring drought tolerance to seedlings or allowing seedlings to quickly recover from drought (Parke et al. 1983a).

The inoculation treatment is intended to reduce transplant shock, improve growth and nutrient uptake, and improve survival. Successful inoculation of the out-planted seedlings, regardless of the growth response, also has the potential to affect the broader soil ecosystem. Because the EMF species that may be established by the inoculation treatment are aggressive colonists, they have the potential to alter the biodiversity of ectomycorrhizal fungi on a wide scale.

Forest Management Implications

Studies from the Pacific Northwest indicate that forest management activities can influence patterns of plant succession and reduce ectomycorrhizal fungi and forest regeneration success (Wright and Tarrant 1958; Harvey et al. 1980a, 1980b; Amaranthus et al. 1994; Waters et al. 1994). Development of management approaches to sustain these essential organisms in forests has been hampered by a lack of knowledge of EMF community structure, diversity, and spatial and temporal variability across stands and landscapes.

Several studies in northern temperate forests have examined the effects of silvicultural practices, particularly clearcutting, on the EMF community (see review by Jones et al. 2003). Most studies focused on effects of disturbance on residual fungus inoculum and on the amount and diversity of EM types on seedlings planted in situ or in greenhouse experiments (Perry et al. 1982; Parke et al. 1983b, 1984; Pilz and Perry 1984; Dahlberg and Stenström 1991; Harvey et al. 1997; Jones et al. 1997, 2002). Some studies have also examined EM diversity on seedlings planted near forest edges or aggregates of retained live trees (Kranabetter and Wylie 1998; Kranabetter 1999; Kranabetter et al. 1999; Kranabetter and Friesen 2002). As noted by Jones et al. (2003), even though the EMF community structure changes in response to clearcut logging, EM colonization of root tips remains high in PNW forest ecosystems. This led them to conclude that changes in the EMF community might be due as much to environmental and biotic factors as to the loss of host trees. The long-term effects of these changes in EMF community structure remain to be documented.

Outerbridge and Trofymow (2004) tested the hypothesis that retention of mature trees enhances colonization and diversity of EM fungi on seedlings planted in adjacent areas. It was found that root colonization and EM diversity declined with distance from mature trees. They concluded that green-tree retention was important for the recovery of EM fungi in harvested sites.

Kranabetter and Friesen (2002), working with Western Hemlock (*Tsuga heterophylla* (Raf.) Sarg.), found that seedlings transplanted from mature forests into harvested gaps declined in both EMF species richness and abundance. Seedlings closest to the uncut forest edge had the highest values of both response variables.

On a green-tree retention site, we have found that a locally-sourced *Rhizopogon* sporocarp inoculum was associated with higher initial seedling mortality (unpublished data). However, our inoculum was a biologically-active slurry of micro-organisms and was applied during a rapid transition from the cool, wet season to the hot, dry season. Nitrogen fixing and other bacteria beneficial to the mycorrhizal symbiosis are associated with both EM sporocarps and roots (Li and Castellano 1987; Li and Hung 1987; Garbaye and Bowen 1989; Garbaye 1991). We hypothesize that simultaneous strong physiological demands from the roots, shoots, and microbial community (for phosphorus in particular) could equate to transplant shock and account for the higher initial mortality. In contrast, we found that a commercial inoculum consisting of a dry spore/clay powder, and lacking an active "helper bacteria" community, may have imposed less physiological stress on the roots during the initial growing season. Garbaye et al. (1992) noted differential effects of treatments on soil "helper bacteria" in bare-root nurseries and concluded that the outcomes of such complex ecological interactions can be difficult to predict.

However after three years, neither sporocarp inoculum treatment was strongly linked to decreased seedling mortality or increased seedling growth. The level of *Rhizopogon* EM on the planted seedlings did not vary among spore inoculation treatments on the green-tree retention site (Luoma and Eberhart, unpublished report on file with the Diamond Lake Ranger District, Umpqua National Forest).

Ectomycorrhizae and Green-tree Retention

Strong correlations between the EM community structure of retained green trees and that of subsequently planted seedlings have been documented, but the role of direct, EM to EM connections via shared mycelium remains unclear (Teste et al. 2009). Since only three EM types were found on the pre-treatment nursery roots, we infer that the EM types found post-treatment were largely established from inoculum naturally present on the site, or as a result of our inoculation treatments. Our observed rapid colonization by non-nursery EM types is consistent with the few similar studies from our region (Bledsoe et al. 1982; Castellano and Molina 1989). The *Wilcoxina* EM type was found on 50 percent of the pre-treatment seedlings and 93 percent of the post-treatment seedlings (our unpublished study, discussed above), suggesting a strong presence of *Wilcoxina* inoculum on the experimental site.

In our unpublished study, two *Rhizopogon* EM types in section *Villosuli* were broadly distributed among the treatments. These taxa seem to be important within the native range of *Pseudotsuga menziesii* and where the species has been introduced in various parts of the world. Sporocarps of *Rhizopogon* section *Villosuli* were the major component of our local inoculum, and were listed as a major component of the commercial inoculum.

There was, however, no overall difference in *Rhizopogon* EM abundance among inoculum treatments. Such an outcome is consistent with green-tree retention concepts of maintaining legacy forest structure that, in turn, retains elements of biodiversity on sites (Franklin et al. 1999) including ectomycorrhizal fungi (Luoma et al. 2004; Outerbridge and Trofymow 2004; Luoma et al. 2006; Jones et al. 2008). Additionally, spores of *Rhizopogon* have been found to be relatively long-lasting in the soil, and can serve as legacy on-site inoculum (Kjøller and Bruns 2003; Bruns et al. 2009). Teste et al. (2004) out-planted seedlings that had been inoculated with a *Rhizopogon* spore slurry at a containerized nursery. They too, found no benefit associated with the *Rhizopogon* spore treatment on seedling growth or survival, two years after out-planting. They inferred that the local EM inoculation potential was sufficient to induce robust EM development.

Conclusions

On the green-tree retention site that we studied (unpublished data), use of our particular formulations of EMF spore inoculum was largely superfluous with regard to decreasing seedling mortality and increasing seedling growth. In particular, the level of *Rhizopogon* EM did not vary among fungal treatments. We concluded that our study site had sufficient legacy EMF inoculum, either as living mycelium or spores, to rapidly colonize the nursery-grown Douglas-fir seedlings that were out-planted.

A broad finding emerges from the various studies reviewed here: the EMF community, as seen on root tips, changes significantly in disturbed sites compared to undisturbed, nearby forests. Furthermore, even on clearcut sites, these changes might be due as much to environmental and biotic factors as to loss of host trees (Jones et al. 2003). The long-term effects of these changes in EMF community structure remain to be documented. We concur with Outerbridge and Trofymow (2004) and Teste et al. (2004) that green-tree retention can be beneficial in maintaining EMF diversity and inoculum potential on a site.

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Managing for Old-growth Forests: A Moving Target

Thomas A. Spies and Robert J. Pabst

Presentation Abstract

Old-growth Douglas-fir (*Pseudotsuga menziesii*) forests are a goal for conservation and restoration across millions of acres of federal forest lands in western Oregon and Washington. Where old growth currently exists, management is typically focused on protecting stands and watersheds from logging and high-severity wildfire. Where old growth was converted to Douglas-fir plantations during the 20th century, the goal is often to actively manage those areas to create ecological diversity and accelerate development of old-growth conditions. What does it mean to use old growth as a target for management? Most old-growth Douglas-fir forests are over 200 years old, and many contain trees over 500 years old. Yet existing empirical studies and our scientific knowledge of forestry are less than a century old and have limited value in projecting how silvicultural manipulations will influence ecosystem function objectives centuries into the future. Old growth as a target can mean using the structure and function of current old-growth stands as a goal for manipulating plantations. This is a challenge because current old-growth stand structure represents centuries of succession, disturbance, and climate change—pathways of development that probably will not be repeated in nature or through silvicultural manipulations. Or, more realistically, it can mean altering conditions in plantations to more closely match what we know about the structure and dynamics of old-growth stands when they were young. In this second approach, managers would manipulate structure and composition to create conditions in young stands that would be more likely to develop the general features of old-growth forests over time than if they were left alone. This approach is based on several areas of knowledge with different degrees of development: (1) a relatively well-developed knowledge of how stand density affects stand development; (2) a moderately good understanding of current old-growth forest structure; (3) a fair understanding of how current old stands developed; and (4) a poor understanding of how any given stand might develop over the next several centuries. We review what is known about current old-growth forest structure and how old-growth forests develop over time and space. We show how regional and local ecological variation can affect old-growth structure and development. We use simulation models to explore how silvicultural manipulations might affect development of old-growth characteristics in the short run. We provide some scenarios of how plantations might develop over centuries. Finally, we propose some rules of thumb to guide management of dense conifer plantations to restore ecological diversity and reset pathways toward desired old-growth structure and composition:

1. Reduce stem densities in plantations to increase rates of diameter growth to produce large-diameter trees and encourage development of large and deep crowns.

Thomas A. Spies is a research forester, USDA Forest Service, Pacific Northwest Research Station, Forestry Sciences Laboratory, 3200 SW Jefferson Way, Corvallis, OR; tspies@fs.fed.us; **Robert J. Pabst** is a senior faculty research assistant, Dept. of Forest Ecosystems & Society, Oregon State University, Corvallis, OR; rob.pabst@oregonstate.edu.

2. Use variable-density approaches to create spatial heterogeneity to promote understory development and diversity.
3. Favor hardwoods and other conifer species that would have been eliminated under intensive management for Douglas-fir timber production.
4. Plant shade-tolerant tree species where seed sources of desired species are lacking in the stand or landscape.
5. Don't use the same prescription everywhere—vary densities and frequency of entries.

Keywords: restoration, thinning, plantations, regeneration, structural diversity, species diversity, Douglas-fir, old growth.

Editors' suggestion:

Franklin, J.F.; Spies, T.A.; Van Pelt, R.; Carey, A.B.; Thornburgh, D.A.; Berge, D.R.; Lindenmayer, D.B.; Harmon, M.E.; Keeton, W.S.; Shaw, D.C.; Bible, K.; Chen, J. 2002. Disturbances and structural development of natural forest ecosystems with silvicultural implications, using Douglas-fir forests as an example. Forest Ecology and Management. 155: 399–423.

Stand Density Relationships

John C. Tappeiner, II

Presentation Abstract

Thinning stands (managing their densities) affects the development of trees and understory plants as individuals, as well as stand-level characteristics like structure, microclimate, and stand growth, habitat for various species, and fuel and potential fire severity. These characteristics and the rate of changes are affected by thinning severity—the reduction in numbers of trees, the stand density, and also by the number, sizes, species, and growth rates of the trees retained. In addition, site variables like potential for severe wind and ice/snow storms, soil depth/rooting, root diseases, and others, have an effect.

Changing stand density simultaneously affects a variety of stand characteristics. For example, thinning to low stand density favors the growth of trees with large stems, crowns, and branches, and may produce considerable volumes of commercial wood in the short term. Thinning stands to low density favors growth of large trees; it may reduce long-term commercial wood production if densities are quite low, and production of snags and dead wood by competition will probably be delayed. Thinning favors development of an understory and a multi-story stand, and generally improves wildlife habitat. However, it can lead to a dense understory that may increase risk of severe fire.

Forest stands are often variable in tree spacing, size, and species composition, even in rather uniform plantations, and the application of thinning prescriptions may need to be flexible to take advantage of this. Thinning density may vary to release understory trees, avoid damage to wet areas, or leave trees at higher density where wind throw is likely, to protect areas of snags. Presence of root disease "pockets" may require variance from the general prescription. Resistant tree species may be favored, openings may be left that will fill in with shrubs, or immune tree species may regenerate.

Large dominant and codominant trees are usually left in thinning, because they grow faster. Where the purpose of thinning is growing late-successional stands, these are the trees that are most likely to become the large trees characteristic of old-growth forests. In certain stands, some of these trees may be removed to favor minor tree species and help develop a mixed-species stand, or to produce valuable products like poles and pilings that will help pay for the costs of thinning. Several treatments may be needed to meet stand management objectives. Examples: thinning dense stands to low density may require several treatments to avoid damage from wind or ice storms that could result from a single, drastic reduction in stand density; or additional treatments may be necessary to manage the dense tree understory that can develop after thinning.

When conducting research on thinning, it is important to specify thinning objectives, and the treatments thinning methods, densities, and other characteristics used to achieve them. Often the

John C. Tappeiner, II is emeritus faculty, College of Forestry, Oregon State University; john.tappeiner@oregonstate.edu.

literature on the effects of thinning on variables other than wood production does not provide enough detail. To interpret the results or repeat the study, stand density immediately before and after thinning and for a period (5 to 10 yr) after thinning, and effects of variables like root disease, wind/ice damage, and others, are needed.

Keywords: density management, thinning severity, stand structure and composition, wood production, late-successional stands.

Editors' suggestions:

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Adapting Management of Shade-intolerant Douglas-fir Forests to Provide Maximum Diversity of Habitats Continuously and Productively

Michael Newton

Presentation Abstract

Managing overstory forest density is the key to energy that shapes the rest of the resource. The role of the tree layer in westside landscapes over long time-spans is driven by the overwhelming influence of the dominant cover in the environments of lower canopies. In even-aged stands, the manifestations of this influence vary markedly after a stand-replacing event by the stage of succession as forests recover. In the example of Douglas-fir (*Pseudotsuga menziesii*) succession, the nature of the event that removes a mature forest cover and initiates succession has an influence on initial non-coniferous cover. Abundant light near the ground allows diverse species composition in the first one or two decades, depending on stocking, that diminishes at an accelerating rate with the exponential development of crowns of the dominant conifers. When crown closure is nearly complete, a period of very low understory productivity begins that may last 20–50 years, depending on frequency of shade-tolerant conifers in the overstory, distribution of gaps in stands, or hardwood mixtures. Density management by thinning leads to soil disturbance, increase in light in understories, and development of seedlings and sprouts of understory herbs and shrubs. Residual density of overstory crowns influences the length of periods in which shade-intolerant understory species persist and over-all diversity and productivity of understory vegetation. Site quality and local overstory density after thinning are major factors in how long each component of understory cover persists. Efforts to create multi-layer canopies by thinning typically encourage understory regeneration of both Douglas-fir and shade-tolerant conifers, as well as tolerant herbs and shrubs. The persistence of each cohort of recruits is dependent on its energy needs and the duration of adequate light before the dominant canopy forecloses shade-intolerant species. In the Douglas-fir Region, expectations for forming uneven-aged stands that retain Douglas-fir may be realistic only on very poor sites, and where overstory stocking is very low or gaps are large. Requirement for gap size to recruit long-lived shade-intolerant species is now under investigation. Fast-growing hardwoods and shrubs are serious limitations on good and some very poor sites.

Where hemlock (*Tsuga* spp.) or other shade-tolerant conifers and hardwoods are present, they may regenerate and persist to form mixed stands that eventually become dominated by the tolerant species as Douglas-fir matures and senesces. The overwhelming influence of the overstory can be expected to restrict what can survive and remain viable in the stand, including Douglas-fir. In the absence of even-aged management, one may expect management of late-successional reserves to mature as Douglas-fir-dominant forests, leading eventually to hemlock or true fir (*Abies* spp.) dominance after several centuries. Maintenance of a diverse landscape dominated by Douglas-fir will likely necessitate patch-wise periodic stand replacement that re-sets early succession lest shade-intolerant species, and wildlife that depends on them, become rare or threatened. Maintenance of

Michael Newton is emeritus faculty, Department of Forest Engineering, Resources & Management, Oregon State University, Corvallis, OR 97331; mike.newton@oregonstate.edu.

high landscape diversity may require various applications of even-aged management with patch sizes large enough to support the full array of wildlife and understory plants, with cycles long enough and patches large enough to provide both early-seral and late-seral features to be present continuously on a rotating basis. Patches of disturbance may presumably be guided in size and location by desired features and continuity of future stands with habitats characteristic of old growth.

Keywords: density management, thinning, shade tolerance, disturbance, understory regeneration, landscape diversity.

Westside Thinning and Underplanting Study in 50- to 55-year-old Douglas-fir and Douglas-fir/Hemlock Stands

Elizabeth C. Cole and Michael Newton

Poster Abstract

Public agencies in the Pacific Northwest have engaged in managing for late-successional features or structure on lands which also have the possibility for high timber production. This study examined the potential for developing understory structure while maintaining a productive overstory on two sites (Willamette Valley foothills and coastal) in western Oregon. Treatments included thinning 50- to 55-year-old Douglas-fir (*Pseudotsuga menziesii*) or Douglas-fir/Western Hemlock (*Tsuga heterophylla*) stands to three or four overstory densities 30 to 65 percent below full stocking, either in a uniform pattern or by creating gaps. After thinning, plots were underplanted with three or four species with varying shade tolerances. Seedling, vegetation, and overstory measurements have been made periodically for 10–15 years after thinning. To determine potential timber yields, stand conditions after thinning were input into ORGANON and growth projected for 50 years after thinning. Tenth-year survival of underplanted seedlings varied by site and species, with the coastal site having higher survival than the Valley site. Survival at the Valley site was lowest with the highest overstory density, but density did not affect survival at the coastal site. In general, the highest-density overstories led to the smallest seedlings and the lowest-density overstories the largest. Seedlings were usually larger in gaps. Browsing affected growth of Douglas-fir and Western Redcedar (*Thuja plicata*). Natural regeneration of Western Hemlock was abundant at the coastal site and resulted in areas of hemlock "thickets", which limited light for other understory vegetation. Projections from ORGANON indicated that overstory trees are dynamic and continuing to grow after thinning. Based on these projections for the higher-density thinnings, it is likely that underplanted seedlings will have a limited potential for creating a secondary structural layer. Repeated thinnings may be necessary to insure adequate growth and survival of underplantings, but removal of overstory trees may result in damage to the understory trees. Gaps, especially the larger gaps, may provide an opportunity for underplanted seedlings to develop a secondary layer.

Keywords: Late-seral vegetation; understory structure; thinning.

Elizbeth C. Cole is a senior faculty research assistant, and **Michael Newton** is emeritus faculty, Department of Forest Engineering, Resources & Management, Oregon State University, Corvallis, OR 97331; mike.newton@oregonstate.edu.

The Theoretical and Empirical Basis for Understanding the Impact of Thinning on Carbon Stores in Forests

Mark E. Harmon

Presentation Abstract

Thinning of forests has been proposed as a means to increase the carbon stores of forests. The justification often offered is that thinning increases stand productivity, which in turn leads to higher carbon stores. While thinning of forests clearly increases the growth of residual trees and increases the amount of harvested carbon compared to an unthinned stand, there is little theoretical or empirical basis for believing that this activity increases the average carbon stores of forests. By removing trees, leaf area is temporarily decreased and carbon input to the forest via photosynthesis is also temporarily decreased. In theory, reducing the input of carbon to a forest will reduce its average carbon stores. Moreover, by increasing the amount of carbon harvested over a rotation, a greater proportion of carbon is removed, which general ecosystem theory also predicts will lower average carbon stores.

The few empirical studies that have examined long-term impact of thinning on carbon stores have indicated that as thinning intensity increases either in amount or frequency, the average long-term carbon store decreases. Simulation models indicate very similar trends. Both results are in line with predictions from general ecosystem theory. Despite the finding that thinning reduces average carbon stores relative to not thinning, this management practice may lead to increases of carbon stores compared to a system of clearcut harvesting. By replacing clearcut harvesting with a series of partial harvests, carbon stores in forests could be increased significantly.

Keywords: thinning, uneven-aged management, regeneration, structural diversity, species diversity, Douglas-fir.

Editors' suggestion:

Harmon, M.E.; Marks, B. 2002. Effects of silvicultural practices on carbon stores in Douglas-fir–western hemlock forests in the Pacific Northwest, U.S.A.: results from a simulation model. Canadian Journal of Forest Research. 32: 863–877.

Mark E. Harmon is a professor, Department of Forest Ecosystems & Society, Oregon State University, Corvallis, OR 97331; mark.harmon@oregonstate.edu.

Western Washington and Oregon Elk Foraging: Use and Nutritional Value by Vegetative Life-form

John G. Cook and Rachel C. Cook

Presentation Abstract

Recent research and modeling of Elk (*Cervus elaphus*) habitat selection and nutrition is establishing the key importance of nutritional value of plant communities (NV, e.g., digestible energy and protein of forage in ungulate diets) to habitat use and productivity of Elk herds in

western Oregon and Washington. These data show that NV of many sites can be improved via active forest management. No exhaustive studies have been conducted in western Oregon and Washington to identify relationships between thinning strategies and changes in NV of plant communities. Our work, however, suggests general patterns of NV that reflect interactions between plant community composition and forage selection capabilities of Elk and, in turn, between plant succession patterns and potential natural vegetation (PNV) zones. These have relevance for understanding and predicting effects of thinning on large ungulate nutrition and may include four patterns:

I. Thinning significantly changes plant composition in mid- and late-seral stages to include more early-seral, shade-intolerant species and (A) these species are considerably more nutritious and palatable than pre-thinning shade-tolerant species, or (B) these species are neither more nutritious nor palatable than pre-thinning shade-tolerant species.

II. Thinning invokes little change in plant composition but increases abundance of attendant shade-tolerant species, and (A) these species are nutritious and palatable, or (B) these species offer low nutrition and/or palatability.

Our foraging/nutrition research evidently offers examples of these patterns in Oregon and Washington. For example, many plant association groups may exhibit the IIB pattern in the Western Hemlock (*Tsuga heterophylla*) PNV, particularly those that support primarily Salal (*Gaultheria shallon*) and/or Swordfern (*Polystichum munitum*) understories, and in the Douglas-fir (*Pseudotsuga menziesii*) zone with Madrone (*Arbutus menziesii*), Manzanita (*Arctostaphylos* spp.), and Tanoak (*Lithocarpus densiflorus*) understories in extreme southwest Oregon. Thinning may only increase the abundance of these highly unpalatable species and thus offer little opportunity to improve the NV of these communities. In contrast, our data suggest that thinning increases abundance of a variety of shade-tolerant and -intolerant species that are palatable and nutritious in the White/Grand Fir (*Abies concolor/A. grandis*) and Mountain Hemlock (*Tsuga mertensiana*) plant associations of the Cascades (west slope) (pattern IA). Thinning in these communities may offer considerable improvement of their NV. Thinning influences on sunlight and soil moisture also may alter plant phenology, chemistry, and thus nutrient levels in ungulate forage. Integrating data of ungulate nutritional ecology and plant succession patterns eventually will be necessary for developing reliable thinning strategies to improve NV and the associated carrying capacity of landscapes for large ungulates.

Keywords: Elk, foraging, nutrition, nutritional ecology, plant composition, plant succession, thinning, ungulates.

Editors' suggestion:

Coe, P.K.; Johnson, B.K.; Wisdom, M.J.; Cook, J.G.; Vavra, M.; Nielson, R.M. 2011. Validation of elk resource selection models with spatially independent data. Journal of Wildlife Management. 75(1): 159–170.

John G. Cook is a wildlife biologist, and *Rachel C. Cook* is a wildlife biologist, National Council for Air and Stream Improvement, La Grande, OR; cookjg.ncasi@gmail.com; cookrc.ncasi@gmail.com.

Songbird Response to Alternative Forest Density Management in Young Douglas-fir Stands

Joan C. Hagar

Presentation Abstract

Thinning has been increasingly used in the Pacific Northwest to restore structural and biological diversity to densely-stocked young- to mid-aged forests that have been previously intensively managed for timber production. In the short term, thinning promotes development of understory vegetation, which in turn can increase habitat diversity for wildlife, particularly for songbirds. Thinning also has been identified as a potential approach to accelerate the development of characteristics that are typical of older forest stands. Both the immediate and long-term response of wildlife and habitat to thinning is likely to be influenced by thinning intensity and pattern. The BLM Density Management Study (DMS) was designed to compare ecological responses among various thinning intensities. The purpose of our study was to compare songbird abundance among these thinning treatments to demonstrate the range of responses that might occur as a result of different thinning practices.

We sampled breeding songbird abundance using fixed-radius point counts at four of the DMS sites (Bottomline, Green Peak, O.M. Hubbard, and Ten High). We conducted bird surveys one year before treatment and in five non-consecutive years after the first thinning, including surveys at two sites in the first season following a re-thinning treatment. Species richness increased following thinning, but response to thinning varied among species. Although several species differed in abundance between control and thinned stands following treatment, few species responded differentially to thinning intensity. The patterns of songbird abundance and species richness that we observed represent short-term responses to the disturbance of thinning; documenting bird response to variation in structural development among thinning treatments will require long-term monitoring.

Keywords: commercial thinning, forest structure, Neotropical migrants, songbirds.

Editors' suggestion:

Hagar, J.C. 2007. Wildlife species associated with non-coniferous vegetation in Pacific Northwest conifer forests: A review. Forest Ecology and Management. 246(1): 108–122.

Hagar, J.; Howlin, S.; Ganio, L. 2004. Short-term response of songbirds to experimental thinning of young Douglas-fir forests in the Oregon Cascades. Forest Ecology and Management. 199: 333–347.

Joan C. Hagar is a research wildlife biologist, US Geological Survey Forest and Rangeland Ecosystem Science Center, 3200 SW Jefferson Way, Corvallis, OR 97331; joan_hagar@usgs.gov.

Effects of Forest Thinning on Bird-vegetation Relationships in Young Douglas-fir Forests

Svetlana Yegorova

Poster Abstract

Forest thinning has been widely used in the Pacific Northwest as a technique to accelerate late-successional vegetation characteristics and promote mature forest habitats for wildlife. Thinning treatments affect tree and understory vegetation development. The effects of thinning on bird abundance and occurrence vary by bird species and change over time, presumably as a result of changes in understory and overstory vegetation after thinning. However, this assumption has rarely been tested. Vegetation characteristics, such as canopy cover, understory density and composition are frequently used in habitat modeling because they are likely to constitute both proximate cues to habitat selection and ultimate resources that contribute to fitness. However, a disturbance event, such as forest thinning, may decouple habitat selection cues from fitness variables. This study investigates the potential of such decoupling for eight bird species by examining dynamics in bird occurrence or abundance in relation to changes in vegetation characteristics within Douglas-fir (*Pseudotsuga menziesii*) stands over 10 years since thinning. Our objectives are to: 1) determine if specific vegetation characteristics account for bird species occurrence or abundance in thinned and unthinned forests; and 2) explore whether the relationships between vegetation characteristics and bird occurrence changes over time. We conducted bird surveys at 58 point-count stations during six breeding seasons in a 10-year period following forest thinning. We obtained detailed local- and landscape-level vegetation characteristics for all bird sampling points. We accounted for imperfect detection and modeled occurrence or abundance of eight avian species as a function of local and landscape-scale vegetation characteristics with generalized linear mixed models. Preliminary results suggest that for some species vegetation is a good predictor of occurrence and that relationship to vegetation remains consistent over 10 years. However, for other species these relationships vary.

Bird-vegetation relationships appear to be highly dynamic, even in a system characterized by relatively low-impact disturbance. We hypothesize that this variability is best explained by intrinsic population processes rather than plasticity in avian habitat selection.

Keywords: thinning, understory vegetation, overstory structure and composition, birds, habitat selection.

Svetlana Yegorova was a master's degree student, Department of Forest Ecosystems & Society, Oregon State University, Corvallis, OR 97331.

Lichen Habitat May Be Enhanced by Thinning Treatments

Heather T. Root and Bruce McCune

Presentation Abstract

Epiphytic lichen communities have become a focus for ecologists concerned with forest health because lichens are particularly responsive to forest management and air quality. Furthermore, they can contribute substantially to the diversity and nitrogen fixation of a stand, and play a valuable role in the food web of many forest-dwelling organisms. Can strategic thinning in *Pseudotsuga -Tsuga* forests increase biodiversity or hasten the development of late-successional features in young second-growth forests? Approximately 10 years after variable-density commercial thinning, N fixing and forage lichen species richness increased and lichen community composition was detectably different from pre-thinning data and from unthinned controls. At two sites in moist conifer forests of western Oregon, lichen community monitoring plots were established before thinning treatments; the most diverse plots in each treatment were retained as diversity islands, whereas the less diverse plots were treated in the thinning prescription. At one site, we found that lichen communities in diversity plots were quite similar to those in the surrounding treated forest, and that the proportion of *Tsuga heterophylla* in the stand was negatively associated with alectorioid and cyanolichen richness. At both sites, hardwood gaps and open-grown trees were positively associated with N fixing (cyanolichen) species richness. At the other site, surrounding plots were more like diversity "leave-islands" after thinning than before. Furthermore, thinned plots had more hardwood gaps following the thinning. These thinned plots hosted more *Bryoria*, *Candelaria concolor*, *Leptogium polycarpum*, *Peltigera collina*, *Nephroma laevigatum* and *Physcia tenella* than had been observed prior to thinning. Most of those species are hardwood associates. The forage lichen *Bryoria*, however, is associated with older remnant trees in these stands. Forage lichens may already be responding positively to the opened canopy in these stands, as evidenced by their association with plots having gaps in the canopy colonized by shrubs, and their increased abundance and frequency in 2007 in thinned plots at one of the two study sites. The retention of lichen hotspots appeared to allow rapid colonization of N fixers onto shrubs in thinned plots. Promotion of gaps in the conifer overstory that are dominated by hardwoods probably stimulated richness of N fixers and forage lichens by providing favorable substrates. We conclude that thinning treatments retaining remnants from previous cuttings, open-grown trees, and hardwood gaps have potential to favor lichen communities rich in cyanolichen and alectorioid species.

Keywords: community composition, diversity, epiphytes, forage lichens, nitrogen fixing lichens, thinning.

Editors' suggestions:

Root, H.T.; McCune, B.; Neitlich, P. 2010. Lichen habitat may be enhanced by thinning treatments in young *Tsuga heterophylla-Pseudotsuga menziesii* forests. The Bryologist. 113(2): 292–307.

Root, H.T.; McCune, B. 2010. Forest floor lichen and bryophyte communities in thinned *Pseudotsuga menziesii -Tsuga heterophylla* forests. The Bryologist. 113 (3): 619–630.

Heather T. Root is a botanist/analyst, US Forest Service Pacific Northwest Region Air Program, Siuslaw National Forest Supervisor's Office, 3200 SW Jefferson Way, Corvallis, OR 97331, htroot@fs.fed.us; **Bruce McCune** is a professor, Department of Botany & Plant Pathology, Oregon State University, Corvallis, OR 97331.

Conservation of Ectomycorrhizal Fungi: Green-tree Retention Preserves Species Diversity

Joyce L Eberhart and Daniel L Luoma

Poster Abstract

The Demonstration of Ecosystem Management Options (DEMO) project is a large, interdisciplinary study designed to test the biological and silvicultural effects of green-tree retention in Douglas-fir (*Pseudotsuga menziesii*) forests. Six treatments were replicated on six blocks in Washington and Oregon, USA: no harvest, 75 percent aggregated, 40 percent (dispersed and aggregated), and 15 percent (dispersed and aggregated) green-tree retention. We studied treatment effects on ectomycorrhizal fungi in three of the study blocks. Ectomycorrhizae (EM) were sampled by use of soil cores. Prior to tree felling, one spring and one fall set of cores were collected from each study unit. After the trees were cut, soil cores were again collected in the fall, a full year after cutting, and the following spring. A total of 432 cores were assessed. To test for treatment effects, mean number of EM morphotypes in cores, relative frequency, and species accumulation curves were compared among pre-treatment, control, moderate-thin, heavy-thin, clearcut, and leave groups. All methods of measure showed a reduction of EM in clearcut and heavy-thin treatments. The dispersed moderate-thin treatment (40 percent green-tree retention) showed less reduction in EM types, except in one block, where compaction during logging may have contributed to root mortality. Silvicultural methods that are designed to maintain high levels of biodiversity in a stand are relatively new in forestry. Through green-tree retention, EM diversity is maintained at higher levels than in clearcuts and we expect that retained trees will provide a legacy of ectomycorrhizal fungi during the development of the next stand.

Keywords: ectomycorrhizal fungi, green-tree retention, thinning, DEMO.

Joyce L. Eberhart is a senior faculty research assistant and **Daniel L. Luoma** is an assistant professor, senior reseatch, in the Department of Forest Ecosystems & Society, Oregon State University, Corvallis, OR 97331.

Effects of Varying Levels of Forest Thinning on *Tricholoma magnivelare* (American Matsutake)

Joyce L Eberhart and Daniel L Luoma

Poster Abstract

Commercial matsutake mushroom (*Tricholoma magnivelare*) harvest in the Pacific Northwest is common in Lodgepole Pine (*Pinus contorta*) forests. Insufficient ecological knowledge of the mycorrhizae and mycelium has challenged monitoring of this resource. Among many management issues are concerns over logging practices, a lack of information on the ecology and habitat requirements of American Matsutake, and the potential effects of the type and intensity of matsutake harvest on future mushroom productivity. Lodgepole Pine and mixed-conifer stands provide habitat for growing and harvesting matsutake mushrooms, but these stands have developed dense understories that can increase the risk of wildfire and disease. The Deschutes National Forest in central Oregon, USA, is implementing a plan to reduce the risk that insects, disease, and wildfire will lead to large-scale loss of forest resources. Our study will monitor the effects of the vegetation treatments (tree thinning) on shiros of *Tricholoma magnivelare* by directly sampling soil and mycorrhizae obtained from plots established throughout the project area. To minimize impact on soil hyphae, all tree-thinning experimental areas were designated for logging with snow cover present and by use of a feller-buncher to yard whole trees to landings. Four blocks were established. Each block contained three forest cover-types with treated and untreated areas. Thirty shiros of matsutake per experimental unit were identified, and four soil samples were taken from each shiro. Due to volcanic glass in these soils, it was not possible to amplify DNA using extraction techniques that require bead beating or vortexing. In response, we developed a quick, inexpensive modification of Xin's plant extraction methods. These extractions were then PCRed using unpublished primers developed by B. Bravi that specifically amplify *T. magnivelare*. To date, only pre-treatment samples have been collected. Logging over snow was completed the winter of 2010/2011, and post-treatment sampling will occur in fall 2011.

Keywords: thinning, Lodgepole Pine, American Matsutake, mycorrhizae.

Joyce L. Eberhart is a senior faculty research assistant and **Daniel L. Luoma** is an assistant professor, senior reseatch, in the Department of Forest Ecosystems & Society, Oregon State University, Corvallis, OR 97331.

Community Structure of Vascular Plants, Arthropods, Amphibians, and Mollusks in Managed Forests of the Pacific Northwest (USA)

Stephanie J. Wessell-Kelly and Deanna H. Olson

Poster Abstract

Increasing global demands on forest resources are driving large-scale shifts toward plantation forestry. Simultaneously balancing resource extraction and ecological sustainability objectives in plantation forests requires the incorporation of innovative silvicultural strategies such as leave islands (green-tree retention clusters). Our primary research goal was to determine how leave islands provide refugia for low-mobility, ecologically sensitive species in managed forests of the Pacific Northwest, USA. We examined patterns in vascular plant, arthropod, amphibian, and mollusk assemblages across five types of forest sampling units: unthinned forest (approximately 600 trees per hectare [tph]), thinned forests (200 tph), and leave islands of three sizes (0.1-, 0.2-, and 0.4-ha]) embedded in the thinned forest. Our objectives were to examine gradients in community associations to measured environmental variables, to describe differences in communities among the five types of forest, and to identify species indicative of each type of forest.

We used two multivariate community analysis methods to describe the primary environmental variables and gradients driving the species composition of these communities. First, we used non-metric multi-dimensional scaling (NMS) to ordinate sample units in species space to provide a graphical representation of plant, arthropod, amphibian, and mollusk community relationships and environmental variables. Correlations between ordination axes and environmental variables also were examined to determine the important drivers of community structure and composition for each taxonomic group. Joint plots were used to overlay environmental variables on the ordination, based on the correlations of the variables with the ordination axes. Ordinations were then rigidly rotated to maximize the loading of the strongest gradients in community variation on a single axis. Next, we used indicator species analysis (ISA) to characterize plant, arthropod, amphibian, and mollusk assemblages associated with the a priori groups of interest. These a priori groups included the five types of forest, four study sites, and two mountain ranges, the Oregon Coast Range and Cascade Range. Indicator values were calculated for each species within each group by combining information about the concentration of species abundance and faithfulness of occurrence in a particular group. Indicator values ranged from 0 (no indication) to 100 (perfect indication). A perfect indicator for a particular group was present in all sampling units for that group and occurred exclusively in that group. Our NMS results indicated strong gradients shaping vascular plant, arthropod, amphibian, and mollusk communities across study sites and across mountain ranges. In particular, microclimate conditions (relative humidity, ambient temperature, and soil temperature) seemed to be especially important in shaping species assemblages in our managed forest stands. Finally, ISA identified indicator species for the a priori groups of interest.

Stephanie J. Wessell-Kelly is the zone silviculturist/research coordinator, USDA Forest Service, Umpqua National Forest, Diamond Lake Ranger District-North Umpqua Ranger District, 18782 North Umpqua Hwy., Glide, OR 97443; swessellkelly@fs.fed.us; **Deanna H. Olson** is a research ecologist, USDA Forest Service, Pacific Northwest Research Station, Forestry Sciences Laboratory, 3200 SW Jefferson Way, Corvallis, OR 97331.

Specifically, unique indicator species were found for each of the five types of forest, four study sites, and two mountain ranges. The ecological processes that shape forest biodiversity operate over wide spatial and temporal scales. Sustaining forest biodiversity at these multiple scales requires that forest managers develop a deeper understanding of the underlying biological patterns and environmental gradients affecting forest communities. Forest management strategies guided by knowledge of these biological patterns and gradients can potentially help sustain biodiversity and community composition in intensively managed forests. Our results suggest that leave islands represent an effective silvicultural strategy for maintaining the environmental conditions needed to sustain several low-mobility taxa in intensively managed forests.

Keywords: forest leave islands, thinning effects, microclimate, Oregon.

Leave Islands as Refugia for Low-mobility Species in Managed Forest Mosaics

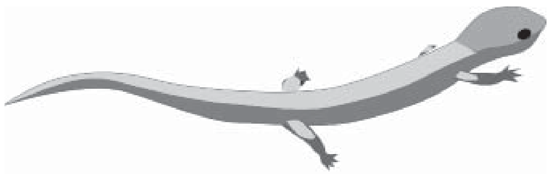
Stephanie J. Wessell-Kelly and Deanna H. Olson

Poster Abstract

In recent years, forest management in the Pacific Northwest has shifted from one based largely on resource extraction to one based on ecosystem management principles. Forest management based on these principles involves simultaneously balancing and sustaining multiple forest resource values, including silvicultural, social, economic, and ecological objectives. Leave islands, or green-tree retention clusters, have been proposed as an alternative silvicultural strategy designed to sustain the ecological integrity and biological diversity within intensively managed forests. However, pertinent questions regarding the relationship of the physical structure of leave islands to their associated microclimates, flora, and fauna remain largely unanswered. We evaluated the effectiveness of three sizes of leave islands (0.1-, 0.2-, and 0.4-ha) within a thinned forest matrix relative to thinned and unthinned forest in providing refugia for low-mobility, ecologically sensitive species one to five years following forest thinning. Specifically, we examined differences in microhabitat and amphibian, mollusk, arthropod, and vascular plant abundance and diversity with respect to the size of leave islands in managed forests. By determining habitat correlates of species and functional group occurrence, we envision that this study will provide vital information regarding aggregated green-tree retention in managed forest landscapes. Our results indicate that there are treatment effects of forest thinning and leave islands relative to microclimate and some aspects of amphibian and arthropod density, and vascular plant diversity and ground cover. These results suggest that leave islands may represent an effective sustainable forest management strategy by providing short-term refugia for some species in managed forests of the Pacific Northwest, U.S.A.

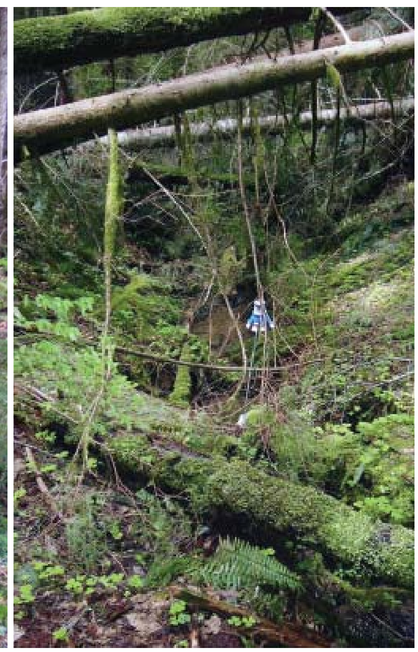
Keywords: microclimate, amphibians, mollusks, arthropods, plants.

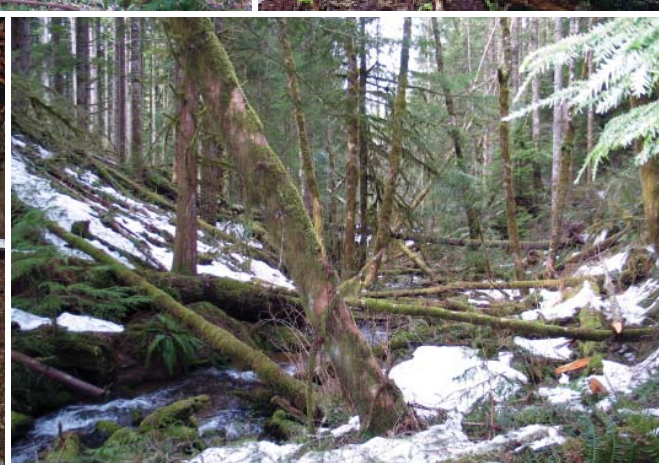
Stephanie J. Wessell-Kelly is the zone silviculturist/research coordinator, USDA Forest Service, Umpqua National Forest, Diamond Lake Ranger District-North Umpqua Ranger District, 18782 North Umpqua Hwy., Glide, OR 97443; swessellkelly@fs.fed.us; **Deanna H. Olson** is a research ecologist, USDA Forest Service, Pacific Northwest Research Station, Forestry Sciences Laboratory, 3200 SW Jefferson Way, Corvallis, OR 97331.



Photos on facing page—Top left: headwater stream initiation seep area, Keel Mountain, 2008, photo by Paul Anderson. **Top right:** Small headwater stream with a camera case for scale, Keel Mountain, 2008. USFS photo by Mark Meleason. **Center left:** Edge of variable-width buffer with stream, Keel Mountain, 2008, photo by Paul Anderson. **Center right:** Stream temperature and channel microclimate measurement station, Keel Mountain, 2008, USFS photo by Mark Meleason. **Bottom left:** Streamflow measurement, photo by Brad Catton, USFS. **Bottom right:** Instream wood at Keel Mountain. Photo by Paul Anderson.


Section 3. Riparian and Aquatic Ecosystems and Their Responses to Thinning and Buffers



Sampling and Modeling Riparian Forest Structure and Riparian Microclimate

Bianca N.I. Eskelson, Paul D. Anderson, and Hailemariam Temesgen

Abstract

Riparian areas are extremely variable and dynamic, and represent some of the most complex terrestrial ecosystems in the world. The high variability within and among riparian areas poses challenges in developing efficient sampling and modeling approaches that accurately quantify riparian forest structure and riparian microclimate. Data from eight stream reaches that are part of the Density Management Study were used in a variety of recent studies that explored sampling and modeling approaches for riparian forest structure and microclimate, and the results are summarized here. When sixteen sampling alternatives were compared based on their performance at accurately estimating the number of conifer trees per hectare, conifer basal area per hectare, and height-to-diameter ratio in headwater stream reaches, rectangular strip-plots outperformed all other plot shapes. Strip-plots oriented perpendicular to the stream generally outperformed strip-plots parallel to the stream. Understory vegetation layers form a critical component of forest ecosystems. Hence, accurate estimation of their attributes (e.g., percent shrub cover) is gaining increasing importance. Percent shrub cover was modeled as a function of distance to stream and canopy leaf area index using techniques that easily accounted for spatial dependence within and among riparian areas. The distinct ecological processes, habitats, and biodiversity of riparian areas are due in part to microclimate characteristics such as air temperature (T_{air}) and relative humidity (RH) that differ from upland forests. Improved sampling designs and predictive models are needed to characterize riparian microclimates and their response to forest management. Height above stream and distance to stream were found to be important covariates in predicting mean maximum T_{air} in riparian areas. For small sample sizes, optimized sample patterns for T_{air} outperformed systematic sample patterns. Mean maximum T_{air} and mean minimum RH are strongly correlated, and mean minimum RH can be modeled as a function of mean maximum T_{air} and other covariates such as height above stream. Mixed effects models can account for within- and among-stream reach variability in RH. Application of these results can improve the quantitative estimates and reduce the costs associated with riparian forest structure and microclimate monitoring efforts.

Keywords: relative humidity, air temperature, shrub cover, mixed effects models, copula models, optimized sampling design.

Bianca N.I. Eskelson is a research associate, Department of Forest Engineering, Resources and Management, Oregon State University, Peavy Hall 280, Corvallis, OR 97331; bianca.eskelson@oregonstate.edu; **Paul D. Anderson** is a research forester, USDA Forest Service, Pacific Northwest Research Station, Forestry Sciences Laboratory, 3200 SW Jefferson Way, Corvallis, OR 97331; **Hailemariam Temesgen** is an associate professor, Department of Forest Engineering, Resources and Management, Oregon State University, Peavy Hall 280, Corvallis, OR 97331.

Introduction

Riparian forests represent some of the most complex terrestrial ecosystems in the world, play an important role in conserving the vitality of the landscape and its rivers (Naiman and Décamps 1997), and provide special habitats for birds and other terrestrial wildlife species. Suitable riparian forest management strategies and effectiveness monitoring require detailed information about the structure of riparian overstory and understory vegetation and riparian microclimate in order to ensure provision of suitable wildlife habitat, high-quality water, and other ecosystem services. Riparian forests are extremely variable and dynamic. The high variability within and among riparian areas poses challenges in developing sampling and modeling approaches that accurately quantify riparian forest structure and riparian microclimate.

The distribution and species patterns of riparian vegetation are related to local topography (e.g., height above stream channel; Naiman et al. 2005, p. 97). Typically, early-seral species are found close to the stream channel where the environment is characterized by low nutrient and high light levels. Long-lived, shade-tolerant, woody plants are found in greater proportions at higher elevations above the wet stream channel (Naiman et al. 2005, p. 99), resulting in high variability laterally from streams. Sampling methods for quantifying riparian vegetation structure need to account for variation being greater laterally from stream to upslope than along the stream, and for the potential of differences in vegetation structure on opposite sides of the stream (Marquardt et al. 2011).

Temperature, light, wind speed, and moisture are microclimate attributes that influence plant distribution through regeneration, growth, and mortality, as well as wildlife habitat selection. High productivity and species diversity of riparian vegetation have been attributed to favorable interactions of microclimate and moisture close to streams (Naiman et al. 2005, p. 156–158). Lateral gradients in microclimate with respect to distance from stream have been characterized and tend to be nonlinear, with rates of change being greater close to the stream channel than upslope (Anderson et al. 2007; Olson et al. 2007). However, little work has been done to improve sampling and monitoring approaches for riparian microclimate attributes, and predictive models for microclimate attributes are few although they are needed to characterize riparian microclimates and their response to forest management.

Overstory and understory vegetation data as well as microclimate data collected at eight stream reaches that are part of the Density Management Study (Cissel et al. 2006) were used to explore and advance sampling and modeling approaches for riparian vegetation structure and riparian microclimate. In this manuscript, we provide an overview of the data collected at the eight stream reaches and summarize some of the studies that used the data to examine: 1) sampling methods to quantify overstory structure in riparian areas; 2) riparian shrub cover models; 3) sampling and modeling approaches for air temperature in riparian areas; and 4) modeling and monitoring relative humidity in riparian areas.

Data

Data were collected at eight stream reaches that are part of the Density Management Study (DMS; Cissel et al. 2006). The eight stream reaches are located at four DMS sites: Ten High (TH46, TH75), Bottomline (BL13), OM Hubbard (OM36), and Keel Mountain (KM17, KM18, KM19, KM21) (fig. 1). At each of the eight stream reaches, one square sample plot (72 m × 72 m = 0.518 ha) was randomly located along the stream, running 72 m parallel to the stream and 36 m upslope on each side of the plot center line that ran approximately parallel to the stream (fig. 2). Detailed overstory and understory vegetation data as well as microclimate data were collected in the 0.518-ha plot.

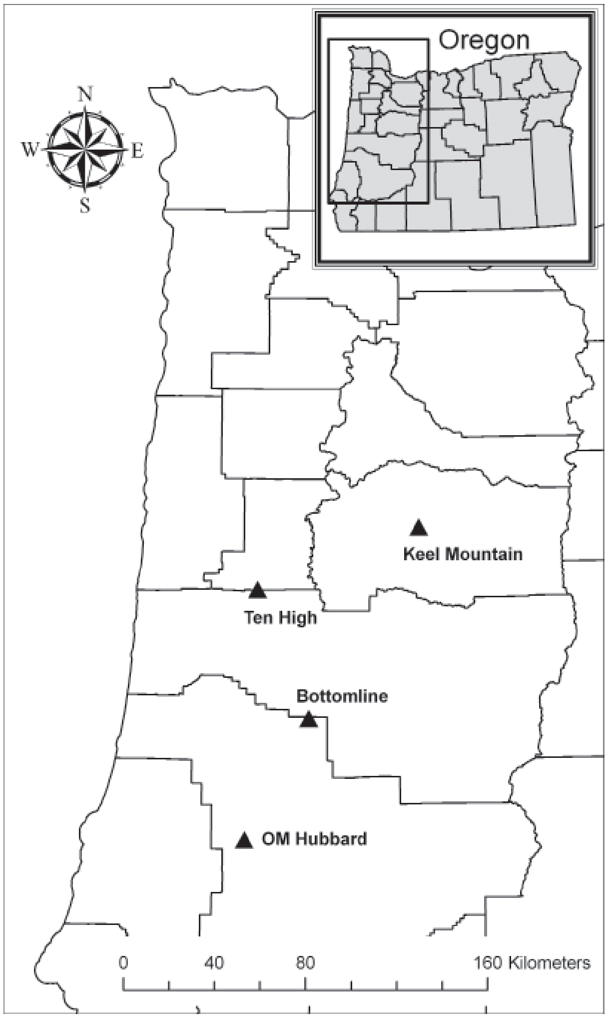
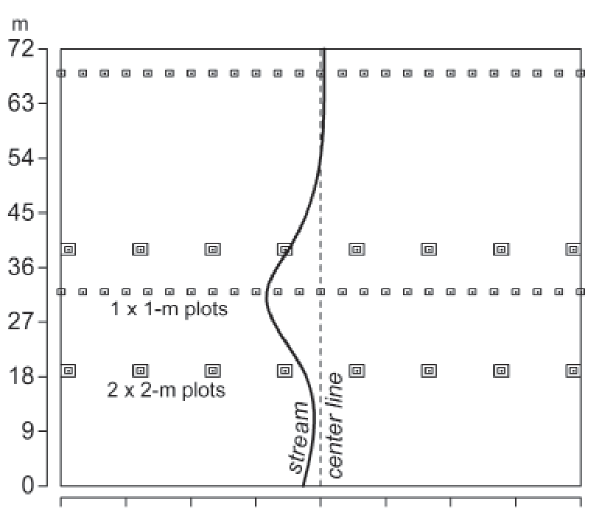


Figure 1—DMS sites where the eight stream reaches used in this study are located.



Vegetation Data

A stem map of all woody stems greater than 6.9 m tall or 7.6 cm diameter at breast height (dbh) was created for each of the 0.518-ha plots. For each mapped tree, dbh, species, condition (alive/dead), and canopy classification (dominant, codominant, intermediate, suppressed) were recorded. For live trees, the crown was classified as "live, full crown," "partially dead crown," or "dead crown," or for trees with more than one crown as "two or more crowns" or "one crown broken off." Standing dead trees (snags) were classified by size (>30 m with full bole, >30 m without full bole, >15 m and <30 m, >6 m and <15 m, <6 m). At each stream reach, 24 trees (12 on each side of the stream) were subsampled for height, crown diameter, height to crown base, and age (for details see Marquardt et al. 2011).

Information about the understory vegetation by taxonomic class (tree, shrub, fern, and forb) was visually assessed using square plots (1 m x 1 m) every three meters along two transects perpendicular to the stream (at 32 m and 68 m as measured along the center line) and every ten meters along two transects perpendicular to the stream at two random locations (<32 m, >32 m and <68 m; fig. 2). Shrubs were defined as woody vegetation less than 6 m in height. Plants less than one meter in height lacking a woody stem were classified as forbs. Percent cover of low shrubs (<1.4 m), tall shrubs (>1.4 m), forbs, ferns, and seedlings was recorded for each 1-m^2 plot, as well as the top height of the vegetative layers (to the nearest meter). On the transects with the 10-m spacing, the 1-m^2 plots were nested within 4-m^2 plots (2 m × 2 m), on which the present shrub species less than 6 m tall were recorded.

Figure 2—Layout of understory vegetation plots (1 m x 1 m and 2 m x 2 m) with microclimate sensors at plot center used in the modeling studies. Plots are arrayed along transect lines oriented perpendicular to a center line (dashed line) that runs approximately parallel to stream (solid line).

Microclimate Data and Canopy Closure

At the plot center and each corner of the 1-m^2 vegetation plots, estimates of percent canopy closure, including understory and overstory vegetation above 1-m height, were made using hemispherical detection of canopy light transmittance (plant canopy analyzer, model LAI-2000; LI-COR Biosciences, Lincoln, Nebraska). The five readings were averaged to provide a plot estimate of leaf area index (LAI, m^2 foliage·m^{-2} ground) and the diffuse non-interceptance (DIFN), defined as the proportion of visible sky.

Three-channel humidity and dual-temperature data loggers (models GPSE 101 203 and GPSE 301 203, A.R. Harris Ltd., Christchurch, NZ) were placed at each plot center of the 1-m^2 vegetation plots to record air temperature, relative humidity, and soil temperature for at least 48 hours (for details see Eskelson et al. 2011a).

Synopsis of riparian structure and microclimate studies

Sampling methods to quantify overstory structure in riparian areas

A simulation study was used to explore the performance of sixteen sampling alternatives to quantify the number of conifer trees per hectare, conifer basal area per hectare, and conifer height-diameter ratio (Marquardt 2010, 2011). Some of these sampling alternatives were designed to capture the variation lateral to the stream (perpendicular strips, alternating perpendicular strips, perpendicular strips on one stream side only), while others were designed to capture the variation parallel to the stream (strips parallel to the stream). As part of the simulation study, circular fixed-area plots were randomly or systematically distributed in the 72-m x 72-m study plot that was stem mapped at each stream reach, and horizontal line sampling (Lynch 2006) and sector sampling (Iles and Smith 2006) were employed. Each of these sampling methods was applied with two plot sizes and two sampling intensities (10 and 20 percent). The rectangular strip-plots outperformed circular fixed-area plots, horizontal line sampling, and sector sampling in terms of accuracy. Among the rectangular strip-plots, alternating perpendicular strip-plots performed best, followed by perpendicular strips, and parallel strips. Narrower strip-plots tended to be more accurate than wider strip-plots, because a greater number of narrower strips may capture the spatial variability better than fewer wider strips. Alternating perpendicular strip-plots 3.6 m wide performed best among all tested sampling alternatives (fig. 3).

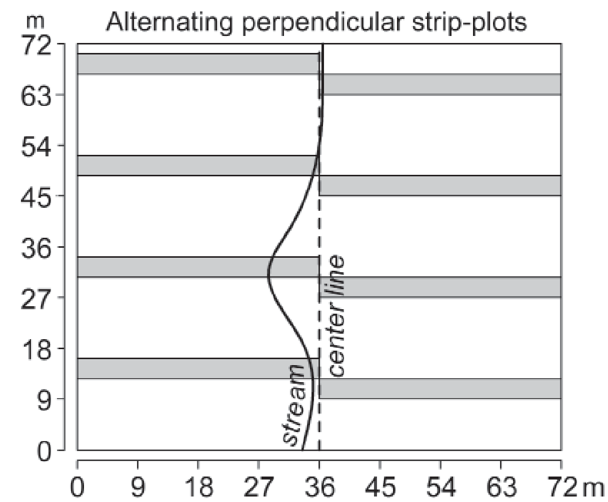


Figure 3—Example of the alternate strip-plot design (20 percent sampling intensity). The center line (dashed) of the 72-m x 72-m block was oriented approximately perpendicular to the stream (solid line).

Although the examined sampling alternatives performed quite well for quantifying conifer density and basal area per hectare in the conifer-dominated stream reaches of the study, the methods were not very accurate when used to quantify hardwood and snag abundance (Marquardt et al. 2012), which were minor components of the overall forest structure (Marquardt 2010).

Extending this work, Haxton (2010) explored the use of n-tree distance sampling to estimate density and basal area at the eight stream reaches. This method, which involves sampling the n trees nearest a sample point, performed poorly relative to variable-radius and fixed-radius plot methods that provided more accurate estimates of basal area and density, respectively. The n-tree distance method was negatively impacted by the spatially clumped distribution of trees.

Riparian shrub cover models

Understory vegetation layers contain most of the plant biodiversity in temperate forest ecosystems (Halpern and Spies 1995), and are of great importance for wildlife species (Hagar 2007). Ground and shrub cover, as well as canopy closure, also conserve bird species richness (Miller et al. 2003). Accurate estimation of understory vegetation attributes such as percent shrub cover is gaining importance, and predictive models of understory vegetation characteristics are needed (Suchar and Crookston 2010). Eskelson et al. (2011b) modeled riparian shrub cover as a function of topography and overstory vegetation, comparing three different methods: ordinary least squares regression, beta regression, and a copula model. Distance to stream and LAI were the most important explanatory variables in the models. Height above stream and interactions of height above stream and distance to stream with LAI were also significant in the models. The copula model based on the beta distribution accounted for spatial dependence within and among riparian areas. The ordinary least squares regression model resulted in underpredictions of shrub cover, while the beta regression and copula models did not. All models had low explanatory power, which was attributed to shrub cover responding to processes and conditions that occur at a finer scale than the available overstory cover variables.

Sampling and modeling approaches for air temperature in riparian areas

Riparian microclimate characteristics such as air temperature (T_{air}) and relative humidity (RH) differ from microclimate in upland forests and provide the conditions for the distinct ecological processes, habitats, and biodiversity of riparian areas. Since intensive sampling of microclimate to determine the spatial variation is impractical, we may have to rely on predictive models of microclimate characteristics. Predictive models of T_{air} can be used to characterize riparian microclimates and their response to forest management. Eskelson et al. (2011a) used kriging with external drift for point prediction of mean maximum T_{air}. Height above stream and distance to stream were found to be important covariates in predicting mean maximum T_{air} in riparian areas, with distance to stream outperforming height above stream as a covariate in stream reaches having steeply incised channels. Adding covariates that describe the over- and understory vegetation cover to the model improved the prediction results. The importance of understory and overstory vegetation cover was dependent on the predominant cover type, for example understory vegetation cover variables were more important in stream reaches with little overstory canopy closure.

Microclimate characteristics have typically been collected at set intervals along transects across stream-riparian gradients (Olson et al. 2007). Since the strongest stream effects on T_{air} and RH have been observed within 10 to 15 m of the stream channel (Rykken et al. 2007; Anderson et al. 2007) it has been hypothesized that microclimate sensor density should be increased close to the stream channel. Eskelson et al. (2011a) optimized the sampling patterns for T_{air} within the eight stream reaches using statistical methods that effectively inferred the spatial patterns (a simulated annealing search algorithm in combination with kriging). The performance of sample patterns based on optimization was compared to systematic sample patterns for which the sample points were evenly spread across four transects. The optimized sample patterns outperformed the systematic samples for small sample sizes. The optimized samples tended to have a higher density of sample points on three of the four transects, and points close to the stream on the remaining transects. This suggests that it may be advantageous to focus microclimate monitoring in riparian areas on a few transects with dense sensor deployment instead of many transects with sparse sensor deployment, and that it is important to deploy sensors close to the stream (fig. 4).

Modeling and monitoring relative humidity in riparian areas

Sensors to sample and monitor RH are more expensive than sensors that measure T_{air}. We have explored the possibility to benefit from the strong correlation between mean maximum T_{air} and mean minimum RH (fig. 5), and modeled RH as a function of T_{air}, height above stream, and DIFN. Mixed effects models accounted for the within- and among-stream-reach variability in RH. Based on a simulation study that examined different subsample sizes of RH, we found that a minimum of three to five subsamples of RH per stream reach seem sufficient for estimating the random effects for localizing the mixed-effects model (Eskelson et al. 2013). We are currently working on a model that incorporates RH measurements from previous years into the model. Application of these models can greatly reduce the costs associated with microclimate monitoring efforts.

Discussion and Management Considerations

The data set collected at the eight stream reaches that was used for the above studies is very detailed and hence provides unique opportunities to explore and advance sampling and modeling methods for riparian structure and microclimate. Microclimate, understory vegetation cover, and overstory vegetation cover are strongly linked with each other. The ability to adequately quantify under- and overstory vegetation cover is needed to improve microclimate models. However, the presence of understory vegetation is also influenced by the prevailing microclimate conditions that need to be considered, in

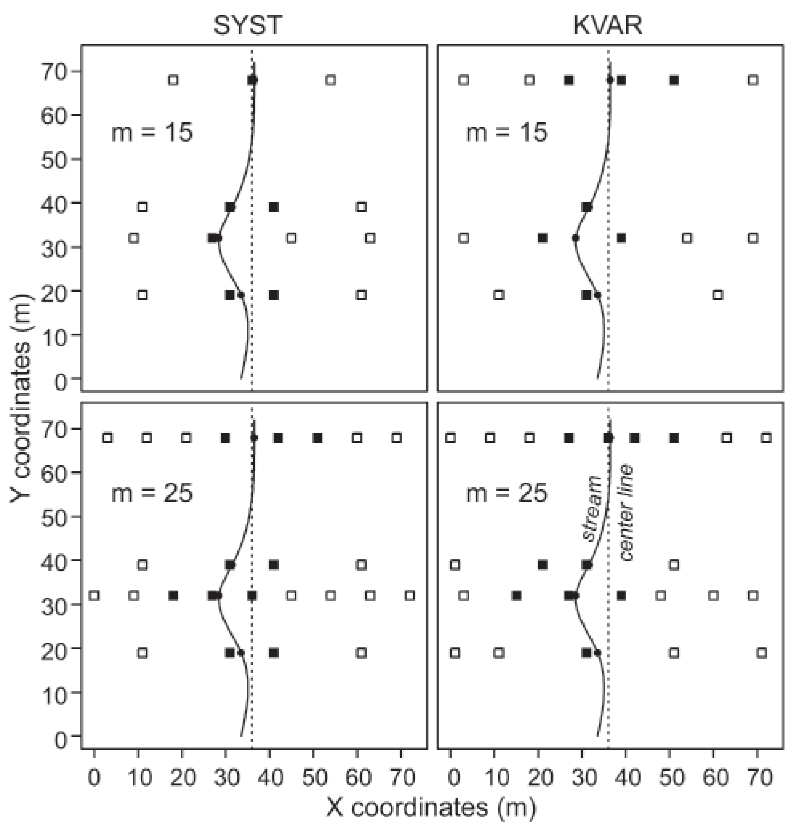


Figure 4—Illustration of optimized sampling designs for characterizing gradients in air temperature in riparian zones based on subsampling from a systematic, gridded array of potential locations (SYST) or from a statistically derived distribution of points that accounts for the inherent spatial correlation (Kriging and estimated variogram, KVAR) at two intensities of sampling—either m = 15 or m = 25 sample points. Regardless of method or intensity, the optimized sampling designs allocate more sensors to locations close to the stream, and fewer sensors upslope of the stream. Solid square, sample location within 15 m of stream; open square, distance to stream of sample location is greater than 15 m.

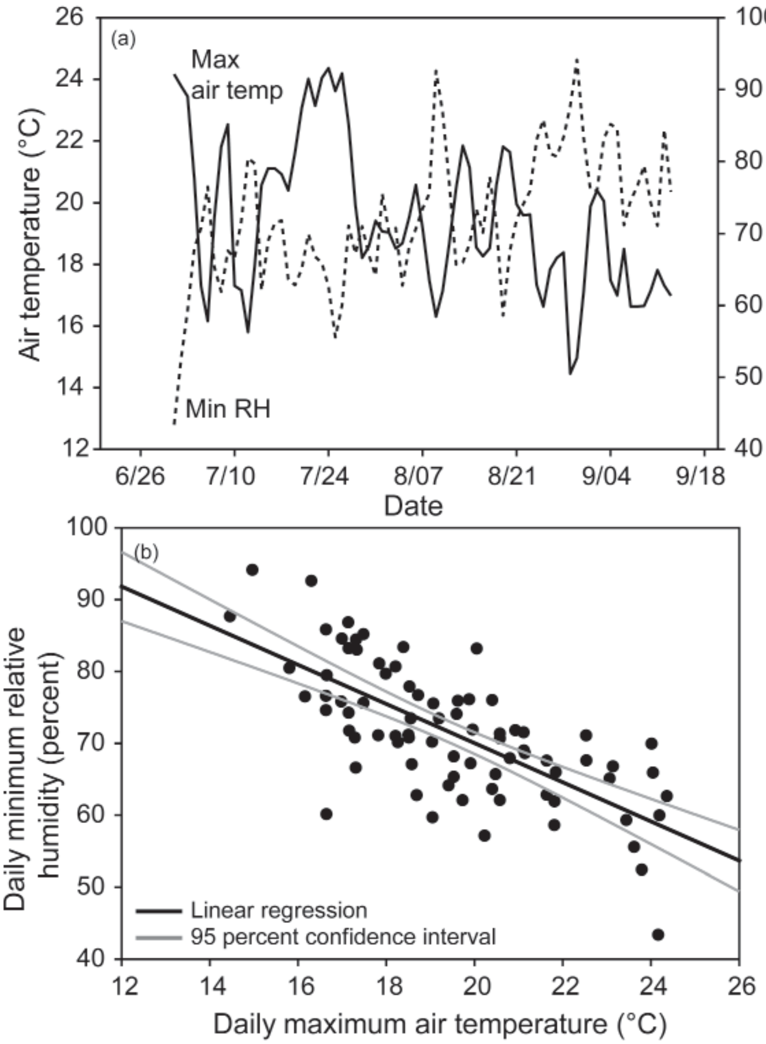


Figure 5—Relationships between summer daily maximum air temperature and daily minimum relative humidity for a stream reach (GP42): a) observed average daily values from July 1 through September 15, 2006; and b) linear regression and 95-percent confidence interval for daily minimum relative humidity versus daily maximum air temperature.

addition to the interactions between understory and overstory vegetation cover. Understanding the intricate linkages among overstory vegetation structure, understory vegetation structure, and microclimate conditions in riparian areas will allow improvement of sampling and modeling approaches.

Stream to upslope gradients of tree species composition and density have been reported (Minore and Weatherly 1994). Sampling methods in riparian areas need to be able to capture this lateral variation. We have demonstrated that alternating, rectangular strip-plots oriented perpendicular to the stream provide the most accurate estimates of conifer density and conifer basal area, and outperformed circular fixed-area plots and strip-plots parallel to the stream (Marquardt et al. 2011). However, sampling methods that quantify minor components of riparian forests such as hardwood and snag abundance still need further improvement, since the sampling approaches that work well for quantifying abundant conifers are not appropriate to quantify the abundance of rare features such as snags and hardwoods (Marquardt et al. 2012).

Improved models of understory vegetation cover in riparian areas will provide important insights into availability of wildlife habitat and food sources. Similar to the stream-upslope gradients observed for overstory tree composition and density, stream-upslope gradients for understory vegetation have been observed (Pabst and Spies 1998). These gradients can be incorporated into models, by including covariates such as distance to stream and by accounting for spatial correlation. The poor explanatory power of the shrub-cover models developed by Eskelson et al. (2011b) was attributed to the fact that overstory vegetation variables, which were used as covariates in the models, respond to processes at different scales than understory vegetation cover. Since the variability of soil conditions in riparian zones plays a major role in vegetation colonization and establishment and in determining plant productivity and diversity (Naiman et al. 2005, p. 92–93), variables describing microsite conditions, soil nutrient availability, and soil moisture may be necessary predictors in understory vegetation cover models. It should be investigated whether variables describing microsite conditions can improve the explanatory power of understory vegetation cover models.

Many of the microclimate variables are highly correlated. A good understanding of the relationships among individual microclimate characteristics may allow focusing monitoring efforts on a single variable such as T_{air} with increased sampling intensity. The development of RH models based on T_{air} measurements will greatly reduce microclimate monitoring costs by reducing necessary RH measurements within a stream reach. Developing similar models to predict soil temperature (T_{soil}) as a function of T_{air} will allow further reduction in monitoring costs. However, the correlation between T_{air} and T_{soil} is not as strong as the correlation between T_{air} and RH. Hence, model predictions of T_{soil} are expected to be less accurate than those of RH. Nevertheless, the development of RH and T_{soil} models that incorporate measurement information from previous years are expected to improve efficiency of microclimate monitoring tremendously.

Monitoring riparian forest structure and microclimate for purposes of general habitat or watershed assessment or in relation to specific forest management activities will benefit from sampling protocols that account for the distinct physical and biological attributes and processes that distinguish riparian areas from adjacent terrestrial environments. Emerging from the research summarized above, we make the following suggestions for consideration by those developing riparian monitoring protocols for conifer-dominated headwater stream reaches:

1. For vegetation characterization, employ strip sampling oriented perpendicular to the stream reaches to capture important gradients from the stream channel through the riparian zone up into the upslope terrestrial zones.
2. For a given intensity of strip sampling (proportion of total area sampled), a greater number of narrow strips per length of sampled reach may provide more accurate estimates than fewer wider strips.
3. If sampling both sides of a reach, alternating the placement of strips on either side of the reach may provide better estimates.
4. Monitor microclimate using transects running lateral to the stream reaches, with the density of sensors being higher close to the stream to capture steep near-stream gradients.
5. If both air temperature and relative humidity are of interest, consider a two-phase sampling design in which relatively inexpensive air temperature measurements are made at a higher intensity (more sample points) and relatively expensive humidity measurements are made at a lower intensity (a subset of points). The strong empirical correlations between air temperature and relative humidity can be used to enhance the estimate of relative humidity when humidity is measured at a subset of the temperature monitoring locations.
6. If important structural elements such as snags or hardwoods occur infrequently or in patchy distribution, they may be estimated using perpendicular strip-plots as previously described, but the precision of estimates is likely to be lower; more precise estimation may require different sampling strategies which remain somewhat poorly defined.

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A Simulation Framework for Evaluating the Effect of Riparian Management Strategies on Wood in Streams: An Example Using Oregon's State Forest Riparian Management Regulations

Mark Meleason, Jeremy Groom, and Liz Dent

Abstract

One objective of the Oregon State Forest riparian management strategies is to provide a long-term supply of wood to streams. We explored this objective as a case study by comparing the predicted wood loads from a riparian forest managed in accordance with Northwest Oregon State Forest Management Plan to an unmanaged riparian forest. We obtained riparian tree inventories of plots from an Oregon Department of Forestry's Riparian Function and Stream Temperature Study site. The site's overstory was measured before and after harvest conducted according to the riparian management strategies. We used the pre- and post-treatment data as initial conditions for 200-year growth simulations in the forest growth model PNW Zelig. The forest model results were then used to predict wood volume in the stream from two riparian management scenarios using the model OSU StreamWood. We found that the stream wood volumes were almost identical in the two simulations, suggesting that at least for this particular site, Oregon's state riparian regulations are predicted to provide a long-term supply of wood to streams similar to that in an unmanaged riparian forest.

Keywords: large wood, LWD, OSU StreamWood, riparian, management, simulation.

Introduction

Wood is an integral part of streams in the Pacific Northwest (Bisson et al. 1987). Wood can enter the channel from the adjacent riparian forests, by fluvial transport from upstream, and from upslope sources. Management plans for upslope forests can indirectly influence wood recruitment to streams by increasing geomorphic processes such as landslides and debris flows; however, best management practices seek to minimize these events. In contrast, management of riparian forests can directly influence the long-term supply of wood to streams, and management plans typically include this objective.

Empirical evaluation of riparian management prescriptions is difficult, due in part to the time scales involved to monitor their performance. Also, given the variation in local site conditions such as slope, aspect, forest structure, and temporal pattern in weather and stream flow, it is difficult to generalize the results to a broader scale. In addition, inferences drawn from observational studies are generally limited to the sites and time period of the study.

Simulation modeling is one useful tool to investigate the generalizable effects of riparian forest prescriptions on wood in streams. By

Mark A. Meleason is the riparian and aquatic specialist for the State Forests Division, **Jeremy Groom** is the monitoring coordinator for the Private Forests Division, and **Liz Dent** is deputy chief of the State Forests Division at the Oregon Department of Forestry, State Forests Division, 2600 State St, Building D, Salem, OR 97310; mark.meleason@state.or.us; jeremy.groom@state.or.us

definition, simulation models are a gross simplification of reality. In fact, the often-quoted phrase by the famous statistician G.E.P. Box "all models are wrong but some are useful" was restated in an expanded version that is worth repeating: "Models of course, are never true, but fortunately it is only necessary that they be useful. For it is usually needful only that they not be grossly wrong" (Box 1979).

The development of simulation models involves many trade-offs, such as the spatial extent (local or regional), the temporal scope (seconds to centuries), and the selection of processes to include or exclude. The design of a model must be closely aligned with the purpose of the model (Mankin et al. 1975). In this light, the OSU StreamWood model was designed to explore the long-term implications of riparian forest management strategies on wood in streams.

In this paper, we use simulation modeling to explore the effectiveness of the Oregon State Forest riparian management strategies (Oregon Department of Forestry 2010) in providing long-term recruitment of wood to streams. Our approach involves isolating the "treatment effect" by comparing a reference simulation to one or more simulated scenarios that differ from the reference by exactly one factor. The two simulations, treatment and reference, are identical in all respects (e.g., stream and initial forest conditions) save for the application of the treatment, which in this application is the riparian management prescription. Although the riparian management prescription is composed of numerous components (e.g., number and width of subzones, harvest protocols, and other rules), it is the overall performance of the riparian treatment that we are evaluating. Indeed, individual actions within the prescription could be evaluated in a similar manner. Our approach involved comparing total wood volume (volume of all logs that intersect at least one bank at a given place and time) from a riparian area with and without the management prescribed for Oregon State Forests. The riparian plots used in this study were measured for Oregon Department of Forestry's Riparian Function and Stream Temperature study (Dent et al. 2008; Groom et al. 2011a, 2011b; referred to hereafter as "RipStream").

Methods

Model Description

A brief overview of the model OSU StreamWood is provided here; please refer to Meleason (2001) for details. It is an individual-based stochastic model that operates at an annual time step. Tree recruitment as wood to the stream channel from the riparian forest can be provided by a forest-gap model built within OSU StreamWood, or by importing the results from other forest growth models. In our case, we modified the model to import results from the Pacific Northwest-specific growth models PNW ZELIG (Garman et al. 1992) and ORGANON (Hann 2011). For each simulation year and iteration, each riparian area was populated with trees that died in that simulation interval. Each dead tree had a known species, diameter, and height from the forest model and was randomly assigned a riparian x-y position and tree-fall angle. The riparian area was subdivided into three riparian subzones on each side of the stream. Each subzone was assigned its own width, tree-fall regime (random), and tree mortality file. Trees enter the channel if they intersect the channel given the angle of fall and distance to channel relative to their height. OSU StreamWood subjects trees recruited into stream channels to breakage upon entry. Those logs at least partially within the channel are subjected to in-channel processes of breakage, movement, and decay (fig.1). Minimum log dimension, which can be defined prior to simulation, was set to 1 m in length and 10 cm in diameter. The model runs under a Monte Carlo procedure and results are expressed as a mean and standard deviation or as a frequency distribution of wood for a given year.

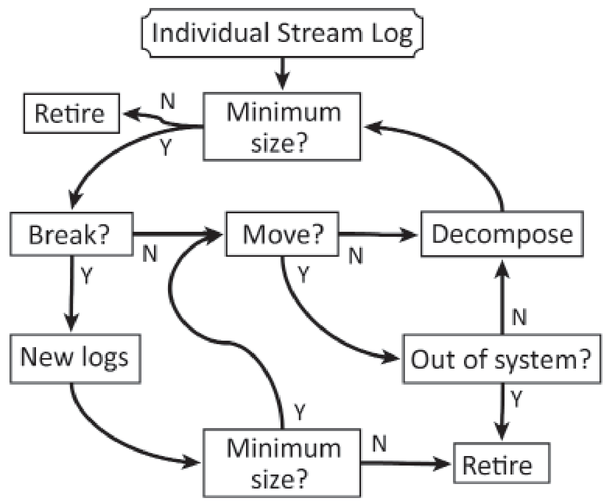

Figure 1—Fate of a log within OSU StreamWood. Trees that fall into the channel, which depends on distance to the channel and fall angle, are subjected to tree-entry breakage. For each annual time step, all logs that intersect at least one bank are subjected to a sequence of procedures representing in-channel dynamics. If the log is equal to or greater than the minimum size criteria (user-specified and set to 1-m length and 10-cm diameter for this study), then it is passed to the breakage function, otherwise it is "retired" (removed from further consideration). Breakage is a two-step function: does the log break and if so, then what are the sizes of the new logs? Next, new logs that meet minimum size criteria and logs that did not break during this cycle are subjected to the movement function. Movement is also a two-step process—does the log move and if so, how far? Logs that move out of the system are retired. Finally, logs are "decomposed" and those that meet the minimum size criteria are tallied for the results of this reach for this year. Both breakage and movement are stochastic functions that rely on uniform random numbers to determine their outcome.

Riparian Sample Sites

The RipStream study was conducted between 2002–2010 and included pre- and post-treatment surveys of a suite of riparian and stream variables at 33 sites in Oregon's Coast Range. One key objective of this study was to assess the performance of state and typical private riparian forest practices on stream temperature in western Oregon (Dent et al 2008; Groom et al. 2011a, 2011b). Another key objective was to determine whether current management approaches were effective in maintaining large wood recruitment to streams.

For the analysis reported here, we selected three sites on Oregon State Forest lands (fig. 2). The proportion of the riparian forest removed (pre- versus post-treatment) was assessed for all three sites (fig. 3), but the simulation of wood recruitment was considered for site 5301 only, which was a second-growth stand from a clearcut with a stand age of 48 years and an active channel width of 5 m. Each site consisted of an upstream reference and a downstream treatment area (fig. 4). Each of these areas contained two riparian plots, one on each side of the stream. Each riparian plot consisted of five sample zones 30 m in length (paralleled to the stream) and 52 m wide, for a total plot area of 0.79 ha (fig. 5). All trees (dbh ≥ 14 cm) were tallied within each zone. In addition, all seedlings (dbh < 14 cm) were tallied by species within six, 42-m^2 plots within each sample zone. Vegetation data were collected pre-treatment for all four riparian plots and for the treatment riparian plots in the year following harvest. For site 5301, very little windthrow was noted in the post-treatment survey, so that the standing tree survey reflected the management prescription.

The treatment reaches on the three selected study sites were subjected to harvest according to the Northwest Oregon State Forests Management Plan (Oregon Department of Forestry 2010; table 1), which defines unique management prescriptions for each of three riparian sub-zones. These sub-zones are parallel to the stream channel. Our perpendicularly defined sampling zones did not match these parallel sub-zones, so we reconfigured our riparian data to conform to State Forest harvest strategies. Since our riparian plot data included slope distance of each tree to the stream, we were able to assign trees to one of the three parallel riparian sub-zones defined by slope distance (fig. 5): streambank (0 to 8 m), inner (9 to 30 m), and outer (31 to 52 m) management zones (table 1).

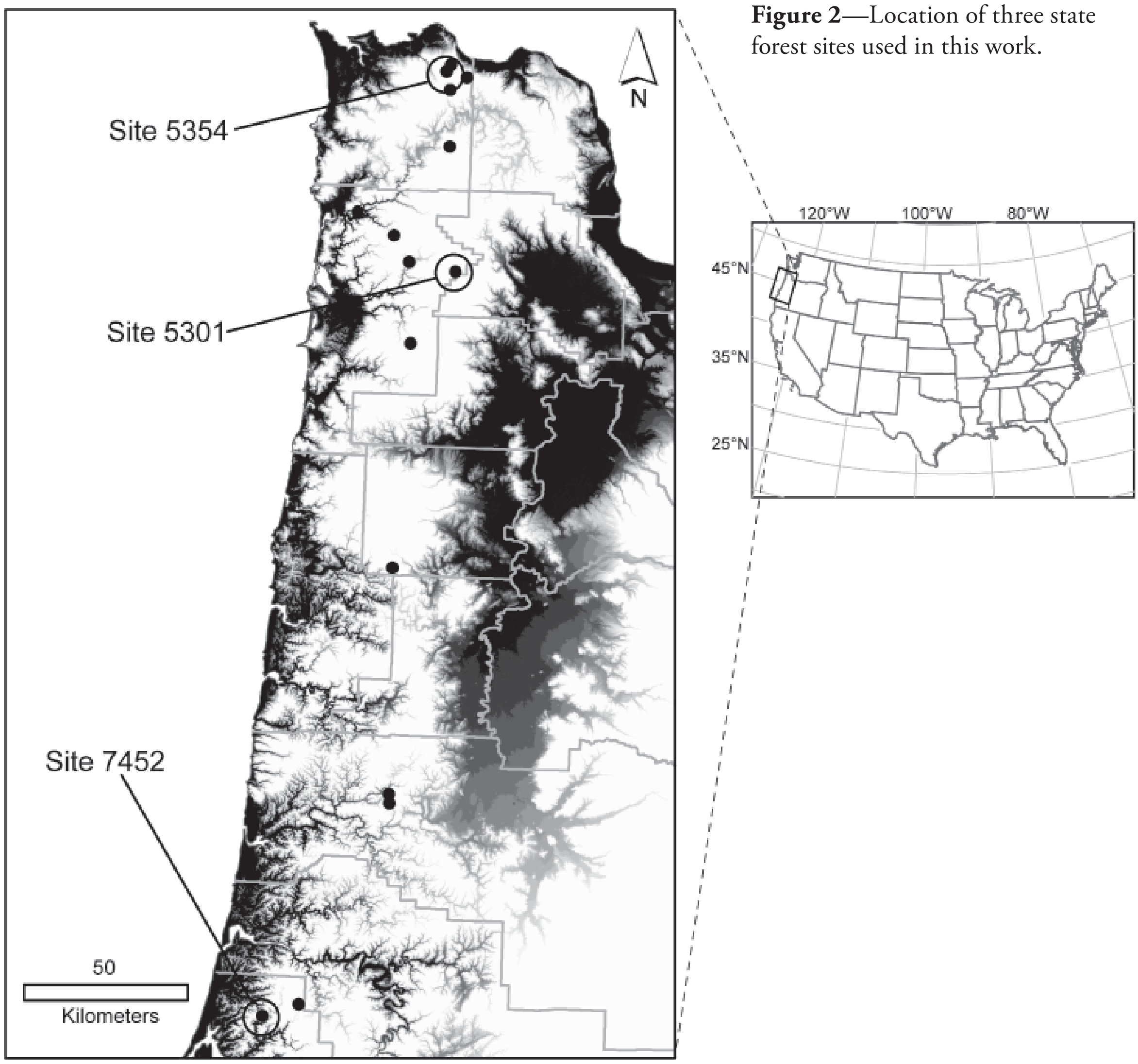

Figure 2—Location of three state forest sites used in this work.

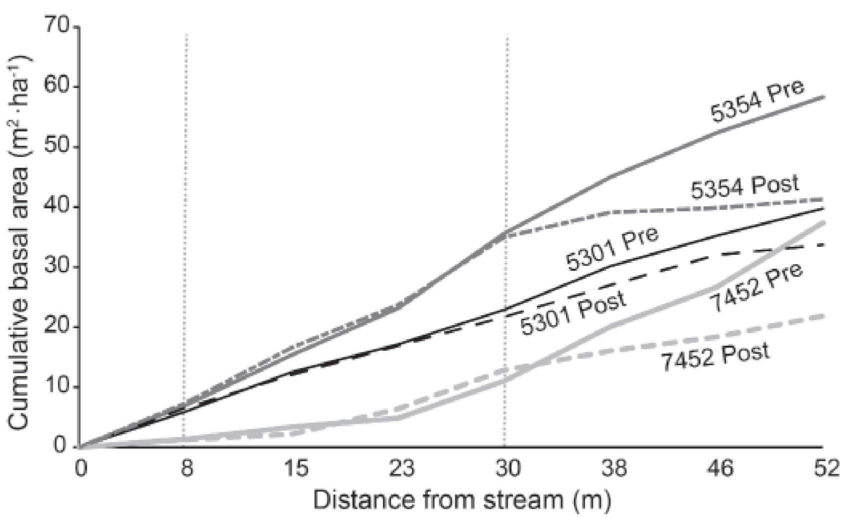

Figure 3—Cumulative basal areas by distance from stream for selected state forest harvest sites. The portion of the riparian forest removed through the riparian management strategies is the difference between the pre- (solid line) and post-treatment (dash line) curve for each site. The dark lines at 8 m and 30 m identify the widths of the riparian subzones, each of which have their own management prescriptions (table 1). Of the three sites, 5301 was selected for the simulations reported in the text.

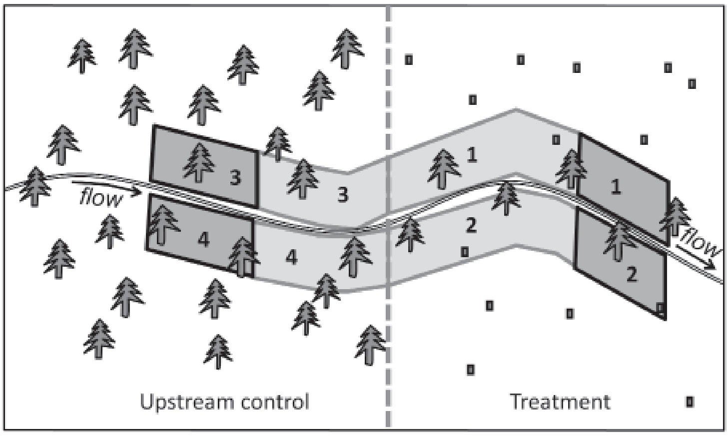

Figure 4—Harvest plot layouts and extensions. Each RipStream site had two riparian plots in the reference (upstream; plots 3 & 4) and treatment (downstream; plots 1 & 2) areas. These plots are depicted in dark grey. We extended these plots (light grey) by doubling the data in associated riparian plots to assist in modeling down wood recruitment in these systems. Upstream reference reaches were approximately 330 m in length. Treatment reach lengths varied from 300 m to 1500 m (1460 m for site 5301).

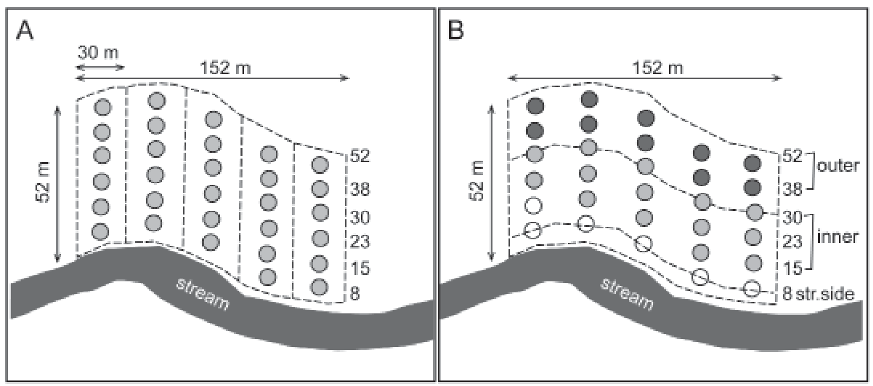

Figure 5—Conversion of the RipStream data to initial conditions used in OSU StreamWood. Riparian plot layout in the RipStream study consisted of a 52-m by 150-m riparian zone divided into five 30-m strips. All trees >10 cm dbh were measured. A total of six 0.01-acre (3.7-m diameter) shrub sub-plots were placed within each 30-m strip at 8-m intervals. Each shrub-sub plot included measurement of tree seedlings (<14 cm dbh) and vegetation cover (A) The riparian forest treatment that was applied defined three riparian management prescriptions that differed by sub-zones: 0–8 m, 8–30 m, and 30–52 m (B; table 1).

Table 1—Definition of Northwest Oregon State Forest Management Plan (Oregon Department of Forestry 2010) management zones.

Management zone	Distance to bank	Prescription
Streambank zone	0 – 8 m (0 – 25 ft)	No cutting
Inner	8 – 30 m (25 – 100 ft)	Limited entry, manage for mature forest conditions
Outer	30 – 52 m (100 – 170 ft)	Depending on streambank zone conifer density, leave 15–70 conifers / 1000 ft (305 m)

We created a tree list (with corresponding expansion factors) for each of the three riparian sub-zones per riparian plot including seedlings from the regeneration plots. These riparian sub-plots were then used as the initial conditions for the forest simulations using PNW ZELIG and ORGANON. The dead tree files produced by the forest models were then used as input to OSU StreamWood.

Simulation of Riparian Forest Growth

OSU StreamWood's prediction of wood in streams depends in part on the input from a forest model. In an effort to increase our confidence in the forest model results (e.g., basal area and tree density through time) as inputs to OSU StreamWood, we compared the two forest models, ORGANON and PNW Zelig. We used the pre-treatment sub-plots for the lower reach as initial conditions for both models. Simulations were for 100 years (100 iterations for PNW Zelig) and the results were compared graphically.

In an effort to assess the consistency between pre- and post-treatment riparian forest simulations, we compared projected basal areas from PNW Zelig. For these runs, which were used in the simulation experiment described below, the simulations were for 100 iterations of 200 years, and we compared the results graphically.

Influence of Oregon's State Forest Riparian Management Strategies on Long-term Wood Supply

The purpose of our simulations was to assess the effect that the Oregon Forest Practices for state lands have on the long-term supply of wood to streams. Our approach involved comparing stream wood volumes attributed to the pre- and post-treatment (table 1) riparian forests observed at site 5301. We simulated a 4-reach system arranged contiguously, with the two upstream plots as the reference and lower two reaches as the treatment (fig. 4). Reaches were 152 m long (width of the measured plot along the stream, fig. 5) with bankfull widths of 5 m, with no wood in the channel at the beginning of the simulation. The tree mortality output from PNW Zelig was used as input to OSU StreamWood. For the pre-treatment run, we populated the riparian plots with the 1-year pre-harvest mortality data, which consisted of 12 unique data files (3 sub-zones for each riparian plot, 2 riparian plots for each reach type, 2 reach types—treatment and reference). For the treatment simulation, we replaced the treatment reaches with six 1-year post-treatment mortality data files. Both simulations were for 500 iterations of 200 years. We compared the total wood volume in the lower reach between simulations of the pre-treatment and post-treatment riparian forests.

Validation

Our primary goal was to isolate the relative impact of a harvest treatment on total wood volume by comparing the results of two simulations that differ by management prescription. A question that arises is whether our predictions could be compared to empirical findings. Validation of our simulation results, as compared to empirical data, cannot be done directly because of the time scales involved. However, our wood volume estimates should be at least reasonable in the broader context when compared to observed wood volume estimates in streams subjected to a similar riparian management regime. To this end, we obtained wood data from the stream habitat surveys collected by Oregon Department of Fish and Wildlife as part of their Aquatic Inventories Project. These data were collected between 1998–2008 using the Aquatic Inventory Protocol (Moore et al. 2008). We obtained data on total wood volume for 142 streams that were within Oregon state forest lands and subjected to the management practice used in our treatment (table 1). We summarized these data graphically, and visually compared them to simulation results. Our intent here was to assess whether the model predictions are similar enough to the observed data to be considered reasonable.

Results

PNW Zelig and ORGANON

We visually compared predicted basal areas of standing trees through time from the two forest models for each of the three riparian sub-zones (fig. 6, inner zone not shown). The greatest divergence between the two models was for the streamside zone and least for the outer zone. The inner zone was dominated by *Alnus rubra* (Red Alder, 95 percent of the basal area) and the outer zone was dominated by *Pseudotsuga menziesii* and *Tsuga heterophylla* (Douglas-fir and Western Hemlock, 93 percent of the basal area).

Riparian Forest Simulations

Prior to treatment, the four plots from site 5301 had a mean basal area of 45 $m^2 \cdot ha^{-1}$, 544 trees per ha, and a mean stand age of 48 years. The riparian treatment reduced basal area by 15 percent in plots 1 and 2 (fig. 3). Pre- and post-treatment simulations for the sub-zones predicted similar stand development, although the pre-treatment had slightly greater basal area and number of trees through the 200-year simulation (fig. 7).

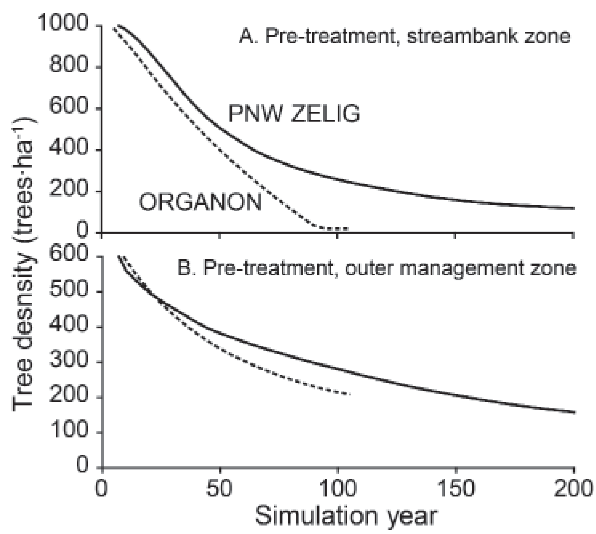

Figure 6—Comparison of ORGANON and PNW ZELIG simulations for the streambank and outer pre-treatment riparian sub-zones.

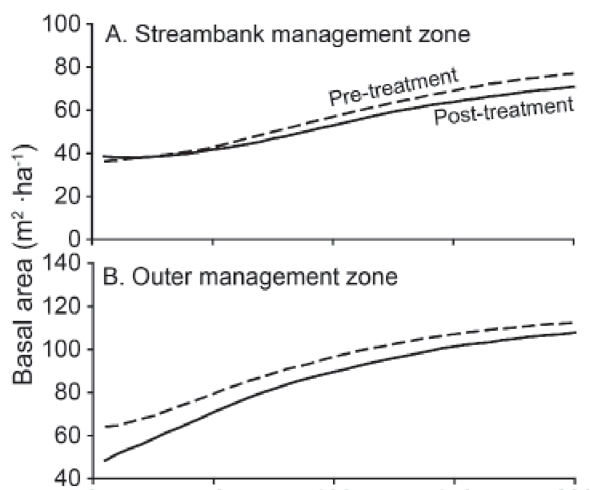

Figure 7—PNW ZELIG forest simulations of mean basal area for (a) streambank and (b) outer riparian management zones pre- and post-treatment. All simulations were identical in environmental and site conditions save for the initial tree populations. Tree mortality rates vary through time as the forest matures. Those trees that died are imported into OSU StreamWood where tree entry to the stream is simulated.

Pre- and Post-treatment Wood Volumes

Mean total wood volume through time was virtually identical between the two riparian forest management scenarios (fig. 8). Mean wood volumes increased through time, as did the variability about the mean. By year 200, the mean wood volume was 98 $m^3 \cdot 100\ m^{-1}$ (standard deviation = 33 $m^3 \cdot 100\ m^{-1}$). The coefficient of variation (standard deviation divided by the mean) ranged from 0.38 early in the simulation to 0.33 by year 200. To further explore the variability within our simulations, we plotted box plots for every 10th year of the simulation (fig. 9). Although the maximum wood volumes increased with simulation year, the minimum was consistently below 20 $m^3 \cdot 100\ m^{-1}$ for the first 100 years and mostly below 30 $m^3 \cdot 100\ m^{-1}$ for the remaining time periods for both the pre- and post-treatment simulations.

Validation

Although not directly comparable, the model results do appear to be reasonable when compared to the wood volumes from wood surveys in Oregon state forest lands (fig. 11). The median total wood volume was 25 $m^3 \cdot 100\ m^{-1}$ from the field data, which was similar to the median wood volume of the simulated reach (26 $m^3 \cdot 100\ m^{-1}$, fig. 9) at simulation year 60. The greatest difference is the narrow range in wood volumes from the simulations.

Discussion

The results of our simulation experiment suggest that the state forest management plan strategies could maintain in-stream wood in this stream as compared to an untreated stand. Additional sites would need to be examined to extend this conclusion to other state forests.

Conceptually however, these regulations (table 1) appear adequate to maintain long-term supply of wood to streams. The probability of a tree falling into the stream depends on its height

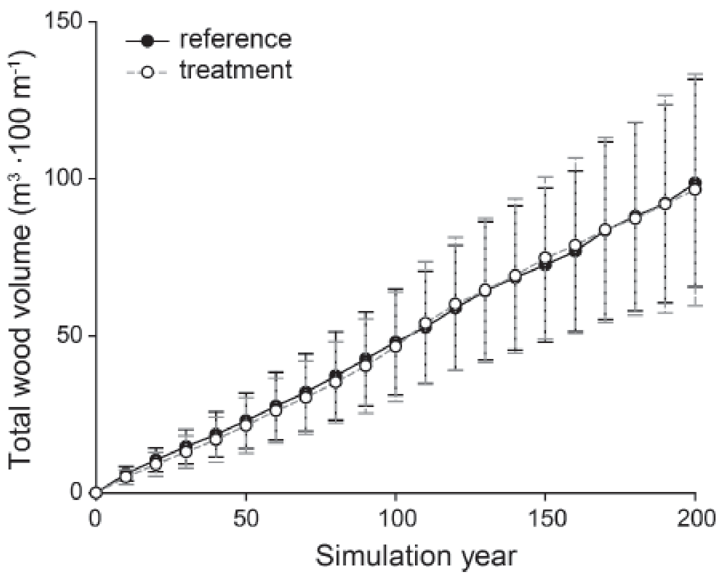

Figure 8—Mean total stream wood volume (± 1 standard deviation) for the pre- and post-treatment simulations using OSU StreamWood. All simulations were identical in stream and riparian conditions save for potential dead trees recruited to the channel, which were determined in the forest model simulations. Total wood volume includes the volume of all logs that intersect at least one bank.

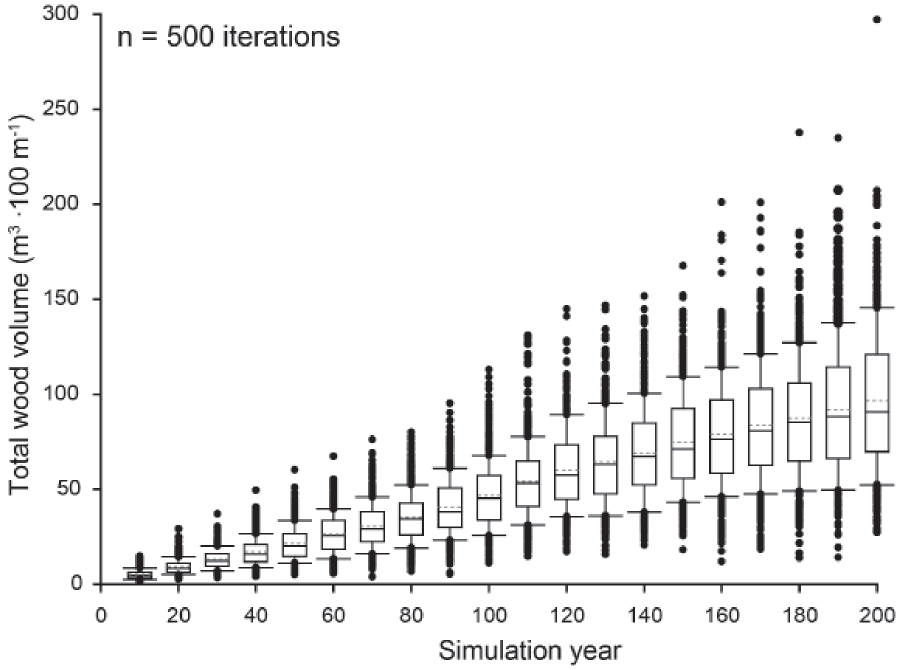

Figure 9—Post-treatment Zelig-simulated total wood volume for the downstream-most reach (n = 500 iterations). The lower and upper boundary of the box represent the 25th and 75th percentile, and the whiskers represent the 10th and 90th percentiles. The median (solid line) and mean (dotted line) are represented.

relative to its distance to the stream. For forests <200 years old, approximately 90 percent of the entry events occurred within the first 30 m from streams in both a simulation study (Meleason et al. 2002) and an observational study (McDade et al. 1990). Maximum source distance equals the maximum effective height of the tree species, although the likelihood of a tree falling and entering the stream decreases substantially with distance from the stream. Assuming a completely random tree-fall regime, approximately a third of the wood volume would be estimated to originate within 6 m of the stream and half the total volume would originate within 10 m of the bank for a 200-year-old riparian stand (Meleason et al. 2003). In the state management plan (table 1, Oregon Department of Forestry 2010), the 8-m streambank subzone is a no-cut area and could potentially contribute more than a third of the potential wood recruitment assuming a random tree fall regime.

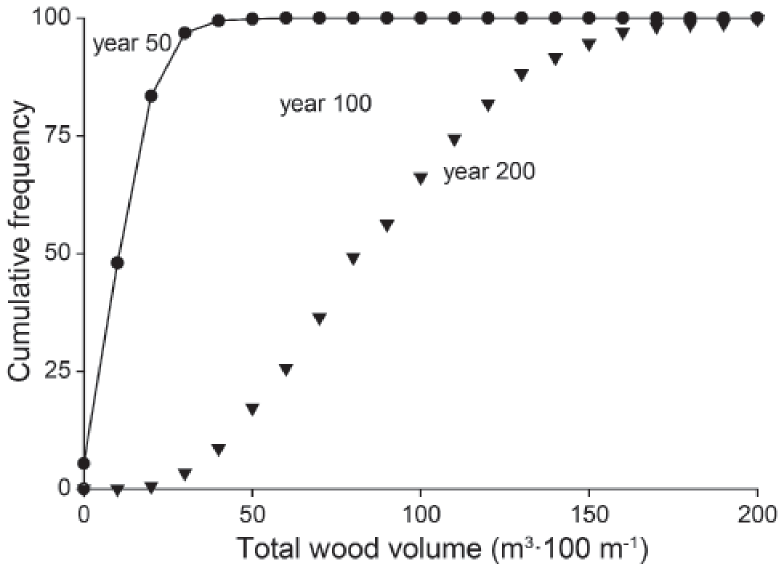

Figure 10—Cumulative frequency of post-treatment reach volumes for the lower-most reach for simulation years 50, 100, and 200 (n = 500 iterations). A cumulative frequency of 50 is the median wood load, where half the iterations were greater than and half were less than the wood volume, which corresponds to the median line in fig. 9. The median wood loads went from 20 $m^3 \cdot 100^{-1}$ m at year 50, to 45 $m^3 \cdot 100\ m^{-1}$ at year 100, to 91 $m^3 \cdot 100\ m^{-1}$ at year 200.

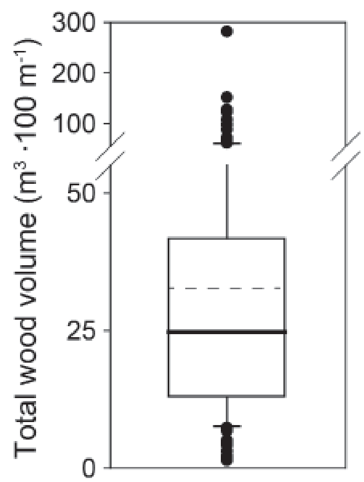

Figure 11—Wood survey results for Oregon coastal streams within Oregon State Forests collected by the Oregon Department of Fish and Wildlife (n = 124). The mean total wood volume was 33 $m^3 \cdot 100\ m^{-1}$ and the median was 25 $m^3 \cdot 100\ m^{-1}$. The lower and upper boundary of the box represent the 25th and 75th percentile, and the whiskers represent the 10th and 90th percentiles. The median (solid line) and mean (dotted line) are also represented.

The effectiveness of these regulations in providing a long-term supply of wood to streams would depend on the degree to which the harvest in the inner subzone reduces long-term recruitment. For RipStream site 5301, only 15 percent of the basal area was removed during the treatment and most of this came from the outer riparian subzone (>30 m from the stream bank). This level of harvest appeared to have very little effect on wood recruitment to the steam.

The riparian forests considered here were around 50 years old when the treatment was applied, so very large riparian trees were initially absent. The treatment had very little impact on the forest structure (fig. 3), so the reference and treatment forests that did develop through time were very similar. The performance of these regulations on older riparian forests is a topic worthy of further investigation.

Model Merits and Management Implications

The use of ORGANON as input to OSU StreamWood to address the question investigated here would be a gross misuse of that model. ORGANON was not designed to grow alder-dominated stands, such as those in our site 5301 or stands >120 years old as we have modeled here (Hann 2011). The purpose of our comparison of forest models was to assess whether they were suitable for our particular application. If the long-term projections of basal area were similar, we could have compared other forest model attributes such as mortality rates, tree size, and species composition. Ultimately, however, the goal was to compare the results of the simulation experiment (e.g., relative performance of the two riparian management scenarios on wood volume in the stream) to see if the selection of forest model influenced the final interpretation. Although ORGANON is a highly versatile model that has proven to be useful for decades, it was not suited for our particular application. Although we are not aware of an ideal forest model for Pacific Northwest riparian forests, PNW Zelig seemed to provide reasonable trajectories of stand development. Since this model has a natural seedling recruitment component, long-term simulations were possible. Other questions, such as those specific to plantation forests, might be better addressed with ORGANON. In OSU StreamWood, each riparian subzone can be associated with a unique forest mortality list, which can be produced by various forest models.

Ecological models are gross simplifications of reality. The usefulness of ecological models, given our understanding of the processes involved and data available, may be seen as "what-if" gaming (Haefner 1996). Although they will always be inadequate and limited, they do provide a means to investigate challenging questions that are virtually impossible otherwise. For example, these results provide one means of investigating the long-term consequences of various riparian management strategies on wood loading in streams. The procedure involved comparing the outcome of two simulations that are identical save for the one aspect under investigation—namely the riparian management prescription. It would be difficult to assess this question empirically, due in part to the time scales involved. In addition, observational studies are specific to the sites considered unless an adequate sample size can be randomly drawn from a population of sites. Even if this were possible, it would be difficult to isolate the treatment effect from additional confounding variables. If the catch phrase for models is "all models are wrong but some are useful", perhaps an appropriate catch phrase for observational studies is that "observational studies are relevant to a time and place; their applicability to another time or place may vary".

This simulation study illustrates that a management prescription applied to a given site has the potential to produce a range of wood volumes in the stream, and this potential range can vary through time (fig. 9). These simulations predicted an overall increase of the range of volumes over the 200-year period. Although the minimum wood volume did increase slightly through time, the majority of this increase was with the maximum wood volumes through time (fig. 9). This suggests that given the same forest structure and stream conditions, there is always a chance that a given prescription can result in a low volume of wood in a given reach. This has a direct implication for riparian forest management targeted at obtaining a desired range of wood volume in streams. A reach with a low volume of wood does not necessarily suggest that the riparian management strategy was inadequate nor does an observed high wood volume necessarily suggest that the prescription is sufficient. Rather, there is a likelihood of obtaining a given volume through time. One way of evaluating the likelihood of a given volume is to develop cumulative probability distributions for a given time period (fig. 10). For example, the simulated results suggest that the management prescription, given these forest and stream conditions, would have a 50 percent chance of obtaining wood loads of 20 $m^3 \cdot 100\ m^{-1}$ at year 50, 45 $m^3 \cdot 100\ m^{-1}$ at year 100, and 91 $m^3 \cdot 100\ m^{-1}$ at year 200 (fig.10). This approach can be used to assess the relative performance of two or more simulated management prescriptions. In this study, the two scenarios—the treatment and reference—produced almost identical results so we have chosen not to include both in figure 10.

Validation

We concluded from our coarse-level assessment that the simulation distributions of total wood volume compared reasonably with the empirical wood volumes. The greatest difference between the simulation and field data was that the range of wood volumes was narrower in the simulation results than in the field estimates. There are several aspects of our approach that did not lend themselves to direct comparison with empirical data. For example, in the simulation we held all stream (e.g., 5-m active channel width) and forest (e.g., site index) conditions constant to isolate the relative performance of the treatment—the goal of this work. The real reaches surveyed varied in both forest and stream conditions as well as other processes that we held constant, such as tree-fall regime and contribution of wood from upslope sources. Many of these factors can be addressed within the modeling framework (e.g., directional tree-fall, influence of key pieces on log mobility in larger channels) provided they are necessary for addressing the particular question under investigation.

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Riparian Buffers and Thinning in Headwater Drainages in Western Oregon: Aquatic Vertebrates and Habitats

Deanna H. Olson

Abstract

The Density Management and Riparian Buffer Study (DMS) of western Oregon is a template for numerous research projects on managed federal forestlands. Herein, I review the origins of Riparian Buffer Study component and summarize key findings of a suite of associated aquatic vertebrate projects. Aquatic vertebrate study objectives include characterization of headwater fauna and habitats, and examination of the effects on headwater-dwelling species of combined buffer-and-thinning treatments in years 1, 2, 5, and 10 post-treatment. Some treatment effects have emerged, with negative effects on bank amphibian counts occurring in treatments with the narrowest buffers 10 years post-thinning. Nevertheless, all taxa appear to be persisting at sites. Instream amphibians, in particular, appear to be highly resilient to the types of disturbances resulting from the thinning and buffer treatments of the DMS.

Keywords: riparian reserves, density management, small streams, NWFP, Aquatic Conservation Strategy Objectives.

Introduction

The Density Management and Riparian Buffer Study (DMS) of western Oregon is an overarching template for numerous research projects on managed federal forestlands, conducted collaboratively by scientists and natural resource managers with several agency and institution affiliations (Cissel et al. 2006). Herein, I provide background on the development of the Riparian Buffer Study component, and I summarize key findings of a suite of DMS projects on aquatic vertebrates and their habitats in managed forest headwaters. This review of the study context and compendium of the various aquatic-related elements may inform research and management decisions regarding riparian management options in west-side forests of the Pacific Northwest.

Background

In the early 1990s, ecological knowledge was synthesized (fig. 1) to develop the one and two site-potential tree height interim riparian reserves of the U.S. federal Northwest Forest Plan (NWFP; USDA and USDI 1993, 1994). Although these reserves were derived from the science and expert opinion of the time, many concepts had not been field-tested or were drawn from upland empirical studies (e.g., microclimate edge effects: Chen et al. 1995) because riparian studies had not yet been conducted.

Riparian reserves were conceived to contribute to the desired future conditions of aquatic-riparian portions of forest landscapes as characterized by the list of Aquatic Conservation Objectives (ACS) in the NWFP (USDA and USDI 1994; p B-11). These included maintenance and restoration of

Deanna H. Olson is a research ecologist, USDA Forest Service, Pacific Northwest Research Station, Forestry Sciences Laboratory, 3200 SW Jefferson Way, Corvallis, OR 97331; dedeolson@fs.fed.us

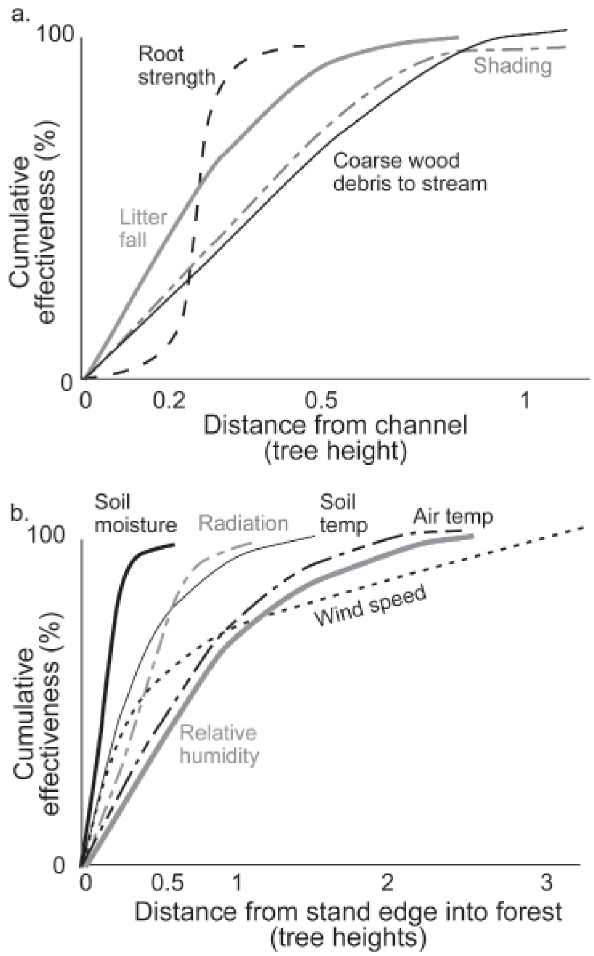

Figure 1—Concepts synthesized from a blend of expert opinion, theoretical knowledge, and empirical studies conducted in a variety of forest contexts that were integral to the development of the interim riparian reserve widths of the federal NWFP (redrawn from USDA and USDI 1993). (a) Contributing factors to stream habitat conditions were expected to be associated with distance from a stream channel. (b) Microclimate edge effects—predicted changes from a clearcut edge into the interior of an old forest stand (from Chen et al. 1995).

the physical integrity of aquatic systems, species composition, and habitats to support well-distributed populations of riparian-dependent species, and within- and between-watershed connectivity. In addition to riparian reserves, ACS objectives were to be attained through additional NWFP components including late-successional reserves, key watersheds, watershed analysis, and watershed restoration (Hohler et al. 2001). ACS objectives were written with landscape-scale language, and their application to project scales and portions of watersheds such as headwater drainages was vague. However, preliminary analyses of perennial and intermittent stream densities in watersheds across the region were conducted, which provided some baseline information about the scope of the intended riparian reserve network: intermittent streams comprised the majority of stream lengths throughout west-side forests (Appendix V–G: USDA and USDI 1993).

The riparian reserve "module" (USDA and USDI 1997) of the federal guide for watershed analysis (USDA and USDI 1995) provided a toolbox of considerations and analyses of physical and biological conditions to help forest natural resource specialists and land managers implement refinements to the interim riparian reserve measures of the NWFP as smaller spatial-scale forest planning ensued. Relative to species, benefits of riparian reserves were recognized not only for aquatic- and riparian-dependent species, but also for a host of terrestrial organisms occurring within west-side forests. Riparian reserves conceptually provided habitats for reproduction, foraging, overwintering, and dispersal of a diverse array of taxa. Before alteration of interim riparian reserve boundaries, or before decisions about management scenarios within boundaries, analysis of additional species information was suggested, so that management decisions would consider the various uses of the newly reserved habitats by an array of target species and communities of organisms. Species with certain ecological attributes were of specific interest, including species: 1) with critical habitats coinciding with riparian areas; 2) with inherently low mobility; 3) occurring over a relatively small portion of the NWFP, being "localized"; and 4) considered to be rare, found in relatively low numbers, including federally listed Survey and Manage species (USDA and USDI 1994). The module aimed to develop a systematic approach for the integration of physical and biological factors so that the interim riparian reserve widths could be potentially altered and managed. However, knowledge of the general ecology of headwater portions of forested watersheds was largely unavailable, hindering full use of the module.

The Bureau of Land Management's (BLM) Density Management Study of western Oregon (Cissel et al. 2006) was originally intended to explore thinning approaches to accelerate the development of upland late-successional forest conditions. The frequency of streams in the study areas complicated BLM site selection and potential layout of treatment blocks within sites, but also created an opportunity to test the efficacy of riparian buffers. The study was adapted to test the effects of the NWFP's riparian reserve widths, within upland thinning, on aquatic-riparian resources.

Two new no-entry riparian buffer widths (variable-width buffer, streamside-retention buffer) were added to the DMS design to address thinning within interim riparian reserve boundaries. These buffers were developed with a two-fold rationale, to: 1) stretch the concepts of the federal riparian reserves, testing narrower buffers with upland forest thinning, which was expected to be a relatively more benign disturbance compared to historical clearcutting; and 2) examine approaches to restore near-stream riparian habitats, accelerating tree growth and the development of late-successional old-growth (LSOG) conditions near streams, and growing future large down wood for instream and streamside habitats. The variable-width buffer (with a 50-ft [~15-m] minimum width on each streamside) was designed to follow ecological gradients along streams, potentially expanding with a significant change in slope gradient (becoming a steeper, more incised stream channel), seeps and slumps, or unique microhabitats or plant species, as detected at local scales during pre-project planning. The variable-width buffer was anticipated to provide advantages to natural-resource specialists and managers at the layout stage of site-level project implementation because stringent fixed distances from streams were not expected; rather, riparian edges could be uneven with varying widths. The streamside retention buffer (~20-ft [6-m] width on each streamside) was conceived as a minimum width to retain bank stability and direct overhead shading of streams.

Overall, these four buffer widths (fig. 2) were conceptually derived, with their implementation on-site reliant on the stream geometry and other site conditions and constraints (Olson et al. 2002). The study design included metrics of lateral and longitudinal distances of treatment units: laterally, perpendicular to streams, ~200 ft (~60 m) of thinning treatment was desired between the buffer edge and the ridgeline; longitudinally along streams, a stream buffer

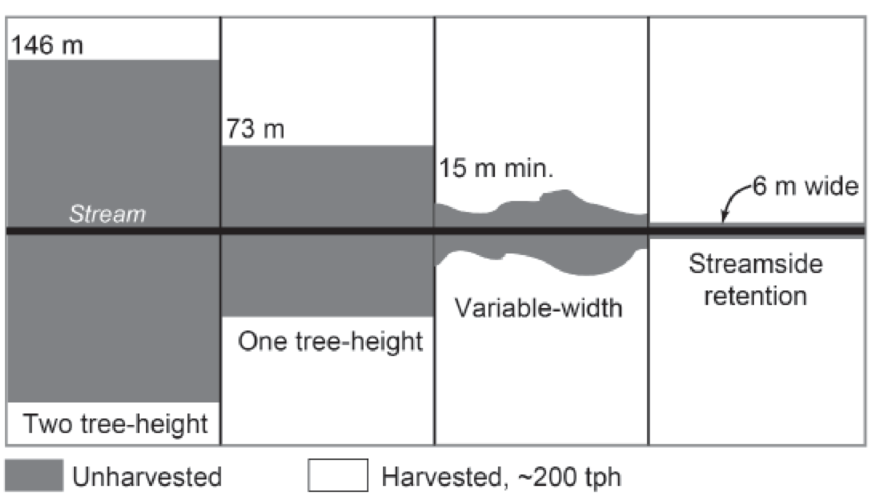

Figure 2—Riparian buffer widths examined in the Density Management and Riparian Buffer Study of western Oregon.

treatment length of about 2.5 site-potential tree heights was desired (e.g., Green Peak study site: 550 ft, ~170 m) (Cissel et al. 2006). Not all stream replicates met these exact criteria, and some stream geometries resulted in treatment units or sites being entirely unsuitable for Riparian Buffer Study implementation. Consequently, an attempt was made to prioritize replication of riparian buffer treatments within the moderate upslope thinning treatment unit, which initially reduced the overstory tree density to ~200 trees per hectare, tph (80 trees per acre, tpa) at most sites. This DMS upland thinning treatment was more intensive than standard commercial thinning densities of the early 1990s, and was established as the treatment that would be optimal to test relative to combined upland and buffer objectives for site restoration. In addition, the study design included reference streams in unthinned control units. Hence, stream geometry was considered during assignment of the moderate and control treatments within a study site. Even so, when few streams occurred at potential study sites, some sites could not be implemented, and not all buffer treatments could be planned within other sites. When stream density was high at a site, the one and two site-potential tree height buffers often extended over sub-drainage ridgelines into the adjacent stream channel drainage. This compromised the study layout criterion calling for a minimum thinned distance between the buffer and ridgeline, and hence, fewer wider buffers could be implemented. For example the two site-potential tree height buffer treatment could not be implemented at the Green Peak study site (fig. 3). The final riparian buffer study design was implemented without full replication across 12 study sites, on lands managed by BLM and the Forest Service (fig. 4).

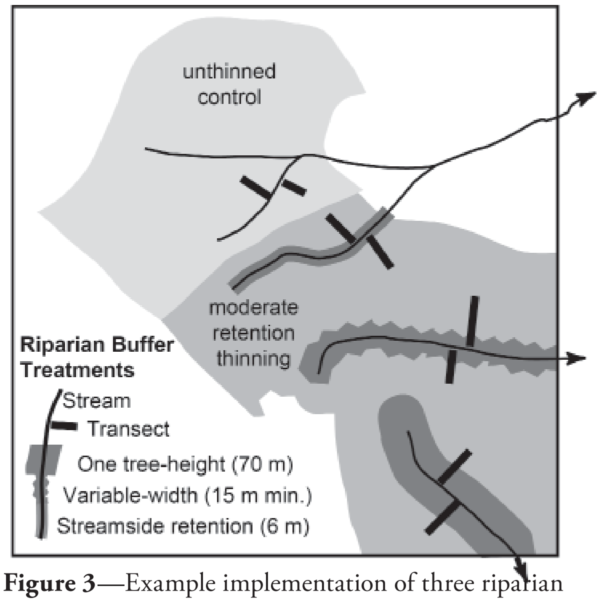

Figure 3—Example implementation of three riparian buffer widths in the moderate upslope thinning treatment at the Green Peak study site of the Density Management and Riparian Buffer Study of western Oregon. Reference streams in the unthinned control unit are shown. Transects perpendicular to streams were used for upland terrestrial salamander assessments (Olson et al. 2006; Rundio and Olson 2007) and pre-treatment down wood assessment (Olson et al. 2006). From Olson and Rugger 2007.

Upon examination of sites and stream networks proposed for use by DMS, it reinforced the fact that DMS was being implemented in headwater sub-drainages with relatively abundant intermittent streams. Because comparatively little was known about the ecology of forested headwaters at the time, research needs emerged to "down-scale" and add specificity to the ACS objectives. Characterizing site-level conditions within headwaters became a second major goal of the Riparian Buffer Study component.

At the time of the development of the DMS study, aquatic-riparian vertebrates and their habitats had become a focus of management attention in the Pacific Northwest due to three factors. First, the US Forest Service's Forest Planning Rule (Sec. 219.19 Fish and wildlife resource: *http://www.fs.fed.us/emc/nfma/includes/nfmareg.html#Fish and wildlife resource*) specified that "Fish and wildlife habitat shall be managed to maintain viable populations of existing native and desired non-native vertebrate species in the planning area." Second, there was an historical taxonomic emphasis on vertebrates for listing under the Endangered Species Act. Third, many aquatic-dependent vertebrates had a status of

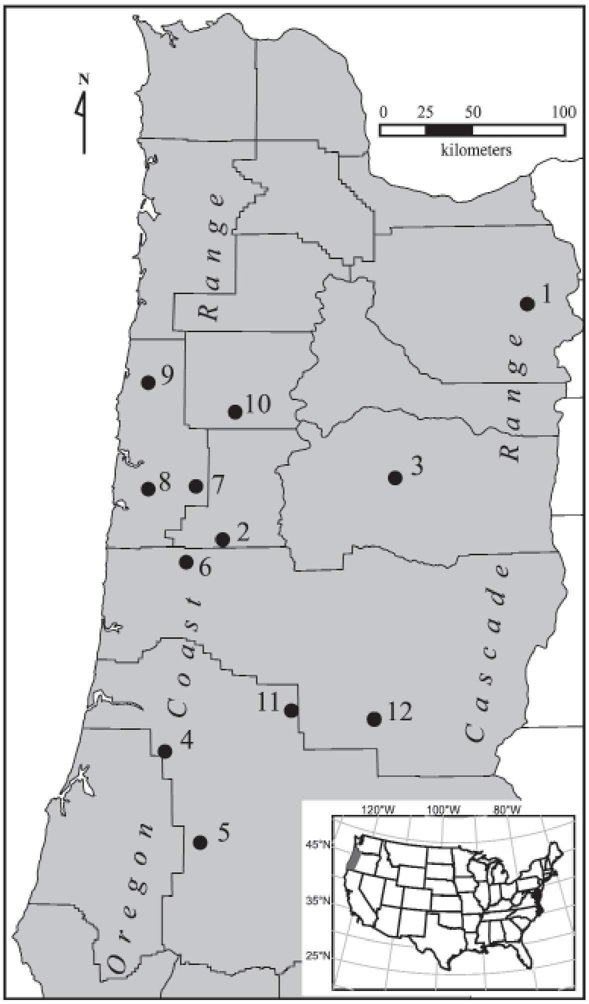

Figure 4—Riparian buffer treatments were implemented at 12 study sites in western Oregon on lands administered by the Bureau of Land Management (sites 1-6, 10-12)) and the Forest Service (sites 7-9). 1: Delph Cr.; 2: Green Peak; 3: Keel Mountain; 4: North Soup Cr.; 5: O.M. Hubbard; 6: Ten High; 7: Cougar; 8: Grant; 9: Schooner Cr.; 10; Callahan Cr.; 11: North Ward; 12: Perkins. From Olson and Weaver 2007.

general concern in the region (see Cissel et al. 2006), and many of these were amphibians. Hence, DMS aquatic ecology research has focused on amphibians and habitat attributes relevant to these animals (Hohler et al. 2001; Cissel et al. 2006).

In summary, the aquatic vertebrate component of the Riparian Buffer Study aimed to characterize the ecological values of headwaters relative to amphibians and their habitats, and assess the effects of the thinning and buffer treatments on these aquatic-dependent biota and their habitats. Key findings of analyses of headwater pre-treatment conditions and post-treatment conditions in years 1, 2, 5, and ~10 after thinning are summarized below (table 1).

Research Findings

Ecological Characterization of Headwater Streams

Animals

At DMS sites, 13 species of amphibians and 5 different fish taxa have been detected (table 1). At a single site, up to 11 amphibian species have been detected, and three sites had this species richness (OM Hubbard, Keel Mountain, North Soup Creek). Up to four fish taxa have been detected at a single site, with two sites having this diversity (Callahan Creek, North Ward).

Amphibian species found in and along streams of the DMS study reaches are organized into assemblages based on habitat associations (fig. 5; Sheridan and Olson 2003; Olson and Weaver 2007). Occurring at all DMS sites are the stream-breeding Coastal Giant Salamander (*Dicamptodon tenebrosus*) and torrent salamanders (*Rhyacotriton* species; fig. 6), the semi-aquatic stream-bank associated Dunn's Salamander (*Plethodon dunni*), the pond-breeding Rough-skinned Newt (*Taricha granulosa*), and the terrestrial-breeding Ensatina (*Ensatina eschscholtzii*).

Of this diverse aquatic-dependent forest fauna, torrent salamanders were highly associated with the uppermost headwater reaches of streams with discontinuous flow (Sheridan and Olson 2003; Olson and Weaver 2007). Furthermore, streams with spatially discontinuous flow, or "intermittent" streams, were the most frequent hydrological flow type in our study (fig. 7). Because torrent salamanders are northwest-endemic species and are species of concern, they and their stream habitats with discontinuous flow are ecological values warranting consideration relative to forest management in headwaters. These salamanders

Table 1—Amphibian and fish species occurrences at Density Management and Riparian Buffer Study sites in western Oregon.

	Study site											
Species	**1**	**2**	**3**	**4**	**5**	**6**	**7**	**8**	**9**	**10**	**11**	**12**
Amphibians												
Ambystoma gracile	x	x			x	x	x	x		x		
Aneides ferreus						x	x	x	x	x	x	
Ascaphus truei	x	x		x	x	x	x		x	x	x	
Batrachoseps wrighti			x				x					
Dicamptodon tenebrosus	x	x	x	x	x	x	x	x	x	x	x	x
Ensatina eschscholtzii	x	x	x	x	x	x	x	x	x	x	x	x
Plethodon dunni	x	x	x	x	x	x	x	x	x	x	x	x
Plethodon vehiculum	x	x		x	x	x	x	x	x	x	x	x
Pseudacris regilla	x	x	x	x	x	x			x	x	x	
Rana aurora	x	x	x	x	x	x	x		x	x	x	x
Rhyacotriton cascadae			x				x	x				
Rhyacotriton variegatus	x	x		x	x	x			x	x	x	x
Tarich granulosa	x	x	x	x	x	x	x	x	x	x	x	x
Fishes												
Oncorhynchus clarkii	x		x				x	x	x	x		x
Oncorhynchus mykiss	x											
Salmonid sp. age 0+	x		x				x	x	x	x		x
Cottus sp.	x		x					x				x
Lamprey sp. (ammocete)						x						x

Study sites: 1= Callahan Creek; 2 = Cougar Creek; 3 = Delph Creek; 4 = Grant Creek; 5: Green Peak; 6: OM Hubbard; 7: Keel Mountain; 8: Perkins Creek; 9: Schooner Creek; 10 = North Soup Creek; 11: Ten High; 12 = North Ward Creek.

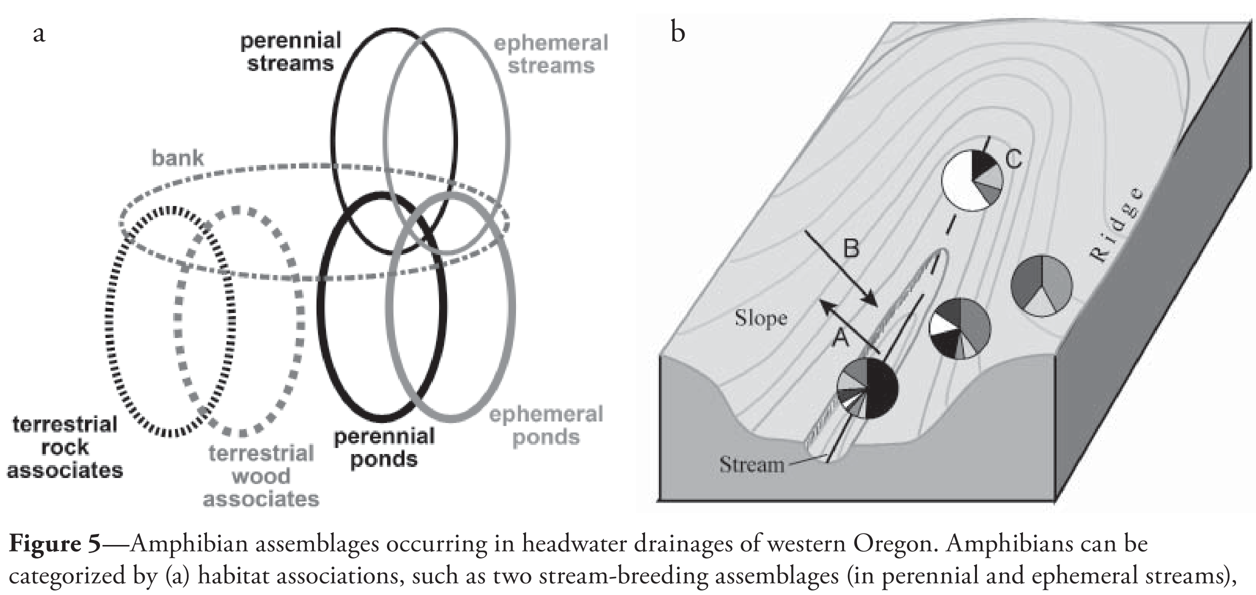

Figure 5—Amphibian assemblages occurring in headwater drainages of western Oregon. Amphibians can be categorized by (a) habitat associations, such as two stream-breeding assemblages (in perennial and ephemeral streams), two pond-breeding assemblages (in perennial and ephemeral ponds), two terrestrial-breeding assemblages (wood and rock associates), and bank-dwelling species which may include members of other assemblages; or (b) species composition by pie-charts within different portions of drainages, showing four dominant assemblages in headwaters (Sheridan and Olson 2003; Olson and Weaver 2007). Arrow A depicts direction of cool, moist microclimates from the "stream effect". Arrow B depicts direction of opposing warmer, dryer microclimates from upslope.

Figure 6—The Southern Torrent Salamander (*Rhyacotriton variegatus*) is reliant on the uppermost headwater stream reaches. (Photo by William P. Leonard.)

do not appear to be associated with larger streams (Olson and Weaver 2007). It is likely that dispersal over ridgelines rather than through the downstream aquatic network is needed to maintain connectivity among sub-populations. A torrent salamander has been documented in a pitfall trap 200 m (656 ft) from a stream (Gomez and Anthony 1996), supporting their ability to venture upslope. Although additional research is needed on this topic of overland dispersal, ongoing genetic studies of headwater amphibians in the Oregon Coast Range are confirming over-ridge connectivity among watersheds (L. Knowles and M.R. Marchán-Rivadeneira, Univ. Michigan, unpubl. data), and genetic studies in other landscapes have similarly documented both riparian movements (Spear and Storfer 2010) and upland dispersal tendencies of amphibians (Spear and Storfer 2008).

Forested uplands at our headwater study sites are habitat for a variety of amphibians requiring moist microhabitats, but not necessarily requiring standing or flowing water for any part of their life history. The upslope amphibian assemblage is distinct from instream and bank assemblages, and is largely composed of down-wood-associated species and species associated with rocky microhabitats (fig. 5a; Sheridan and Olson 2003; Wessell 2005; Rundio and Olson 2007; Kluber et al. 2008). These animals may be found within near-stream riparian areas, and the Western Red-backed Salamander (*Plethodon vehiculum*) was highly associated with near-stream areas (Kluber et al. 2008). Using mark-recapture methods to study movement patterns of terrestrial-breeding amphibians at the Green Peak DMS study site are under investigation. Short-distance movements of animals have been detected, with more movements occurring in near-stream areas. This supports the "funnel" concept, in which near-stream areas funnel animal movements along streams (Olson and Burnett 2013), possibly because either the stream acts as a barrier to dispersal or due to the cool, moist conditions near streams (Anderson et al. 2007; Olson et al. 2007; Rykken et al. 2007)

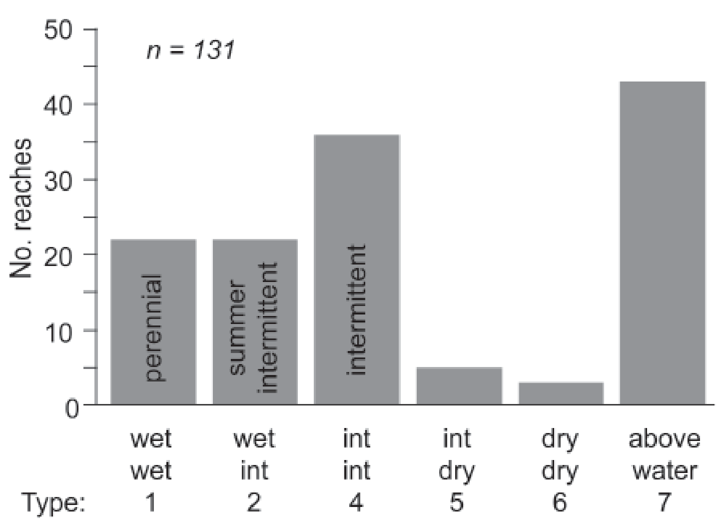

Figure 7—Spatially discontinuous streams that flowed intermittently in spring and summer were the most common stream-channel type at the Density Management and Riparian Buffer Study sites in western Oregon. Int = spatially intermittent flow. Wet = continuous flow. Dry = no water flow, but evidence of scour.

which are ideal microhabitats for many northwest amphibians.

The life history of many of these amphibian species remains largely unknown, however knowledge gained from DMS studies is incrementally adding to our understanding of the basic biology of these species. For example, the nests of two amphibian species that have been observed at the DMS sites: Dunn's Salamanders (Nauman et al. 1999) and Ensatina (Olson et al. 2006). Although most amphibian species are from instream, bank-dwelling, or upland forest assemblages, four pond-breeding amphibians (*Ambystoma gracile*, Northwestern Salamander; *Rana aurora*, Northern Red-legged Frog; *Pseudacris regilla*, Pacific Chorus Frog; Rough-skinned Newt) were found routinely in and along streams at our sites. These four species may move relatively long distances in forests, and streams and near-stream habitats likely "funnel" animals from lower in aquatic networks into headwaters (Olson and Burnett 2009, 2013). Incidental sightings of single egg masses of Northwestern Salamanders have been found in shallow pools or roadside ditches at DMS sites, confirming pond-breeder reproduction at some sites.

Various aspects of the population and community ecology of headwater forest amphibians have been addressed by companion studies at locations outside of our study sites, in order not to disturb our experimental treatments. Sagar et al. (2005) reported on the survival and growth of juvenile Coastal Giant Salamanders in small streams of the Oregon Coast Range near our North Soup Creek study site. Predator-prey interactions have been investigated among three species. Antipredator behaviors were described for Coastal Giant Salamanders in response to Cutthroat Trout (*Oncorhynchus clarkii*) presence (Rundio and Olson 2003). Southern Torrent Salamanders (*Rhyacotriton variegatus*) were unpalatable to Coastal Giant Salamanders: the giant salamanders consistently rejected torrent salamanders during predation attempts (Rundio and Olson 2001). The conceptual model of "trophic cascades" may apply to headwater stream systems. In this model, a few dominant predators (Coastal Giant Salamanders, Cutthroat Trout) control the occurrences of other trophic levels (Terborgh and Estes 2010). Further research is needed to more fully document the hypothesis that repeated predation attempts by the larger Coastal Giant Salamanders on torrent salamanders restrict the downstream distribution of torrent salamanders. Also, an abundance of sculpins (*Cottus* species) at some sites indicates that they are a potentially dominant predator in those systems, and warrants further investigation.

Biodiversity characterization at DMS sites has been conducted in companion studies. At four DMS sites, Wessell (2005) examined four taxonomic groups in upslope areas: vascular plants, mollusks, soil arthropods, and amphibians. In her comprehensive study, she found 120 vascular plant species, 3,608 mollusks of 12 taxa, 30,447 arthropods of 289 taxa, and 7 amphibian species. Her multivariate statistical analyses showed interesting associations between species assemblages and habitat attributes, microclimate metrics, sites, and mountain ranges (Coast, Cascade). Because there were no matched old-growth forest sites available in the study landscape to compare to the DMS sites, a study characterizing amphibian and plant biodiversity of unmanaged forested headwaters was conducted near one DMS site, Soup Creek, near Coos Bay (Sheridan and Olson 2003; Sheridan and Spies 2005).

Lastly, a case study of detection probabilities and occupancies of instream and bank amphibians has been conducted at Green Peak. Both giant and torrent salamanders had consistently high occupancy and detection rates, indicating that their distributions were not patchy within our treatment reaches and that our light-touch hand-sampling ("rubble rousing") methods were effective at detecting them. Greater variability in occupancy and lower detection probabilities were found for bank *Plethodon* species.

Habitat

Characterization of DMS headwater habitat conditions has focused on instream habitat typing. Along each study reach, the dimensions (width, length, depth) of fast and slow water units (e.g., pools and riffles/cascades) are recorded, in addition to the substrate type and down wood per unit. From these data, reach hydrotypes (fig. 7) were developed (Hohler et al. 2001; Olson and Weaver 2007). Published analyses have focused on assessing species-habitat associations (Olson and Weaver 2007).

Using the long-term data set (1994 to present) of multiple stream flow metrics, the hypothesis of "shrinking heads" relative to retrospective annual weather patterns is being addressed: do headwater streams shrink in size during dry years? (Burton et al. 2013a). This analysis is relevant to exploring effects of climate change scenarios on stream flow conditions, with treatment interactions also considered. Additionally, an ongoing companion study is examining how water availability in headwaters among years may contribute to forest vegetation development, again with climate change and treatment as interacting factors.

Analyses of DMS down wood metrics are ongoing (e.g., Anderson and Meleason 2009; Burton et al. 2013b). Instream down wood data have been collected during pre-treatment surveys and in years 1, 2, 5, and 10 post-treatment. Analyses include down wood patterns over time, relative to treatments, and distance-from-stream of the source tree for each instream down wood piece. From preliminary analyses, the number of wood pieces for which sources could be determined ranged from 425 to 841 per site. The average slope distance-from-stream of a source of down wood ranged from 2.6 to 9 m (8.5–29.5 ft) for 25 stream reaches, the maximum distance at four sites examined ranged from 29.4 to 40.9 m (96.5–134 ft), and the most frequent decay classes (Maser et al. 1979) of instream wood were classes 3 and 4. As these data are more fully compiled, new information will become available about down wood recruitment patterns to small streams from sites with riparian buffers and thinning.

Upslope habitat conditions relative to amphibian microhabitat requirements have been assessed during various DMS studies. At two DMS sites (Green Peak, Keel Mountain), Olson et al. (2006) reported pre-treatment percent cover of down wood which occurred in plots along transects spanning stream to upslope areas, perpendicular to streams (fig. 3). At four DMS sites (Green Peak, Keel Mountain, Delph Creek, Bottom Line) 1–5 years post-treatment, Wessell (2005) reported on upslope substrate, down wood, and forest structural components in leave islands, the moderate-density thinning treatment, and the unthinned control. At three DMS sites (Green Peak, Ten High, Schooner Creek), 5–6 years post-thinning, Kluber et al. (2008) analyzed associations of percent cover of upslope canopy, down wood, litter, forbs, and moss with distance-from-stream and treatment effects. Habitat within the near-stream survey bands differed from upslope survey bands relative to several attributes, including more down wood near streams. Thermal profiles of large- and small-diameter logs and soils in near-stream and upslope positions, in treatments and unthinned reference stands, were examined by Kluber et al. (2009). Lastly, riparian-to-upland microclimate gradients at the DMS sites were characterized by Anderson et al. (2007) and a similar study of streamside microclimates along small west-side Oregon streams was conducted by Rykken et al. (2007). Both studies documented a "stream effect" where distinctly cooler temperatures occurred within ~10–15 m (33–49 ft) of headwater streams (review: Olson et al. 2007).

Treatment effects of buffers and thinning on amphibians, other taxa, and habitat attributes

Instream and bank species

Among amphibian species occurring instream, along banks, and in upland forests, there have been a variety of effects of the DMS thinning and buffer treatments. Several caveats are needed, however. Statistical analyses have been a challenge because most species do not occur at all sites, or all reaches within sites, and many species (table 1) occur in low numbers within reaches. The most robust analyses have been possible with the more commonly occurring species, including Coastal Giant Salamanders, torrent salamanders, Dunn's Salamanders, Western Red-backed Salamanders, and Ensatina.

Instream and bank amphibians

In years 1–2 post-thinning, we found no reduction in species abundances in and along streams (Olson and Rugger 2007). This suggests that the mechanical disturbance of the thinning and the immediate effects on animals or their habitats was not substantial. Because amphibians live several years, we felt that a lag time in treatment effects on survival and reproduction warranted monitoring.

Year-5 and year-10 post-treatment results of instream and streambank animal abundances have not yet been published, hence I provide only general preliminary findings here. In year-5 post-treatment, analyses of instream and bank species were conducted with data from 68 stream reaches at 11 sites and the entire time series of data to date. Intriguing findings include reduced abundances of two streambank-occurring amphibians in year 5 compared to pre-treatment. In year-10 post-treatment, different analyses have been conducted using the entire time series of data among years (Olson et al. in press). Results included some effects on both instream and stream-bank animals, with riparian buffers and interacting factors jointly accounting for the differences in animal abundances over time. In particular, bank-dwelling salamander counts decreased over the 10-year post-thinning timeframe in treatments with the narrowest buffers. Pre-treatment abundances were consistently a factor explaining post-treatment species abundances, suggesting that animal abundances after thinning and buffer treatments are contingent on the site-specific contexts of pre-treatment abundances. The strongest take-home message emerging from our long-term instream faunal dataset is the overall resiliency of these assemblages to the treatments.

Analyses of fish species have been problematic due to their uneven distribution among sites (table 1), in addition to their uneven occupancy among both stream reaches and riparian buffer treatments within sites. Fish species have been included in exploratory analyses of year-5 and year-10 data, and no significant insights relative to treatment effects on fishes resulted.

Upland amphibians

Case studies on upland salamanders at the DMS sites tell the same story of variable responses yet apparently resilient assemblages. Although some effects on abundances have been observed, the assemblages have remained intact.

At two DMS sites, 1–2 years post-treatment, Rundio and Olson (2007) examined terrestrial salamanders occurring in transects arrayed perpendicular to streams in the moderate upslope thinning treatment (fig. 3). They reported that captures of two salamander species declined by 40 percent post-thinning at one site (Green Peak), and no treatment effect was detected at the other site (Keel Mountain). Site differences in down wood cover were hypothesized to be related to these results, with down wood potentially ameliorating effects of thinning on salamanders.

At four DMS sites (Green Peak, Bottom Line, Delph Creek, Keel Mountain), 1–5 years post-treatment, Wessell (2005) found various treatment effects of the moderate thinning treatment and leave island sizes on species and diversity metrics. For example, for amphibians, she found that species richness of amphibians was greater in unthinned than thinned forest. Effects of leave island sizes on amphibian species richness and Oregon Slender Salamander (*Batrachoseps wrighti*) abundances also were detected, with more animals in the 0.1- and 0.2-ha leave islands than in the 0.4-ha leave island (0.25, 0.5, and 1 ac, respectively).

At three DMS sites (Green Peak, Ten High, Schooner Creek), 4–5 years post-treatment, Kluber et al. (2008) found no treatment effects of the moderate thinning treatment on upslope salamanders. However, they found that distance-from-stream was associated with amphibian abundance, hence riparian buffers could play a role in upland salamander persistence at sites if adverse effects of upland disturbance did occur. Their results also suggested that rocky substrates may aid amphibian persistence in thinned forests.

Case studies ongoing at two DMS sites investigate the efficacy of artificial cover boards for long-term monitoring of the terrestrial amphibian assemblage. Using mark-recapture methods, we are documenting movements of individual salamanders between boards. At the Green Peak study site, cover boards are arrayed at different distances from the stream, and we have been examining movement patterns related to stream proximity since year 2000. At the Keel Mountain study site, cover board arrays are assessing overland movement patterns of the Oregon Slender Salamander, a species of concern in western Oregon (Clayton and Olson 2007).

In summary, although the value of the different buffer widths with thinning for amphibians is still under investigation, there is evidence that thinning near streams may be conducted at a relatively minor cost to these animals. Variable effects on species abundances are being detected, and although some species abundances have decreased, populations appear to be persisting. Given that the treatments are intended to restore riparian forests, a short-term cost for a longer-term gain of accelerated development of LSOG-forest conditions may be acceptable to land and natural-resource managers, and interested public groups. If upland areas are to be managed over the long term, continued monitoring of both instream and upslope aquatic fauna is warranted to assess population stability.

Habitat

Analyses of combined riparian buffer and upland thinning treatment effects relative to stream flow metrics and instream down wood are ongoing. Integrated multivariate analyses of additional instream habitat attributes and analyses of species-habitat relationships comparing pre-treatment to post-treatment years 1, 2, 5, and 10 data are planned.

Treatment effects on water-quality metrics have been of interest for monitoring at the DMS sites, yet despite multiple proposals over the years, we have been unable to gain sufficient funding to deploy sensors to assess sedimentation or temperature in stream reaches among sites. Nevertheless, at two case-study sites, Keel Mountain and Callahan Creek, with DMS collaborator Paul D. Anderson, instream temperature dataloggers were deployed for year-round temperature monitoring. Unfortunately, in the first winter of deployment, high flows resulted in the loss of many sensors at Callahan Creek. Since then, we have maintained a network of instream temperature dataloggers only at Keel Mountain. They are positioned at stream junctions and at the boundaries of treatment units along stream reaches. We plan to begin analyses following DMS Phase 2 data collection.

Phase 2 – Second-entry DMS Thinning Treatments

A second thinning treatment (Cissel et al. 2006) has now occurred at eight DMS study sites that include Riparian Buffer treatments: Callahan Creek, Delph Creek, Green Peak, OM Hubbard, Keel Mountain, Perkins Creek, North Soup Creek, and Ten High. This second entry was designed to reduce the overstory to near-LSOG-forest densities in order to maintain accelerated overstory tree-growth rates and further enhance understory development. The retained overstory will become the largest trees in the future forest stand. In the initial thinning thinning phase, the moderate-density treatment unit had been thinned to 200 tph; in phase 2, this unit was reduced to ~90 tph (37 tpa; with two trees destined to be future down wood and five destined for snag recruitment). Post-treatment surveys for instream vertebrates and habitats and streambank amphibians in year-1 post-phase-2 thinning will be completed in 2012 at these eight sites. At this time, there is no funding to pursue phase 2 treatment effects on upslope amphibians, with the exception that the upslope cover boards deployed at Green Peak and Keel Mountain will be monitored as time permits.

Three of the four original riparian buffer widths are continued as treatments relative to this second-entry thinning: one site-potential tree height, variable width, and streamside retention (fig. 8). The two BLM study sites at which the two site-potential tree height buffer had been implemented (Callahan Creek, Keel Mountain) are now case studies of a "thin-through" two-tree buffer, where the buffer has been thinned to ~150 tph (60 tpa), and the upland thinned to ~90 tph. There is no stream buffer in this thin-through treatment, meaning that thinning occurred up to streambanks. These case studies will assess the effects on instream resources of thinning without buffers. At one study site, Keel Mountain, the number of stream reaches with a variable-width buffer in the high-density thinning unit, initially thinned to ~300 tph (120 tpa), allowed a similar examination of thin-through buffers with unthinned reaches for comparison. These variable-width buffer reaches were thinned-through (no stream buffer) with 150 tph in both the buffer and the upland treatment unit. Although the inference drawn from case studies is limited to the sites at which the studies occur, the thin-through treatment effects will be monitored to provide insights about the utility of thin-through buffers for riparian forest restoration elsewhere.

Conclusions

The aggregate studies outlined above add to our knowledge of aquatic vertebrates and their habitats in forested headwater basins in western Oregon. Relative to ACS objectives from the NWFP, additional specificity of species and habitat conditions in headwaters has been achieved. Although analyses are still ongoing, early findings have reported relatively mixed effects of these treatments on amphibians or their habitats. The use of these riparian buffer widths in operational thinning projects in west-side forests with similar characteristics will likely have some short-term effects on specific resources, but are also likely to have long-term gains for site restoration. Continued monitoring of species-at-risk in these headwater stream locations, such as

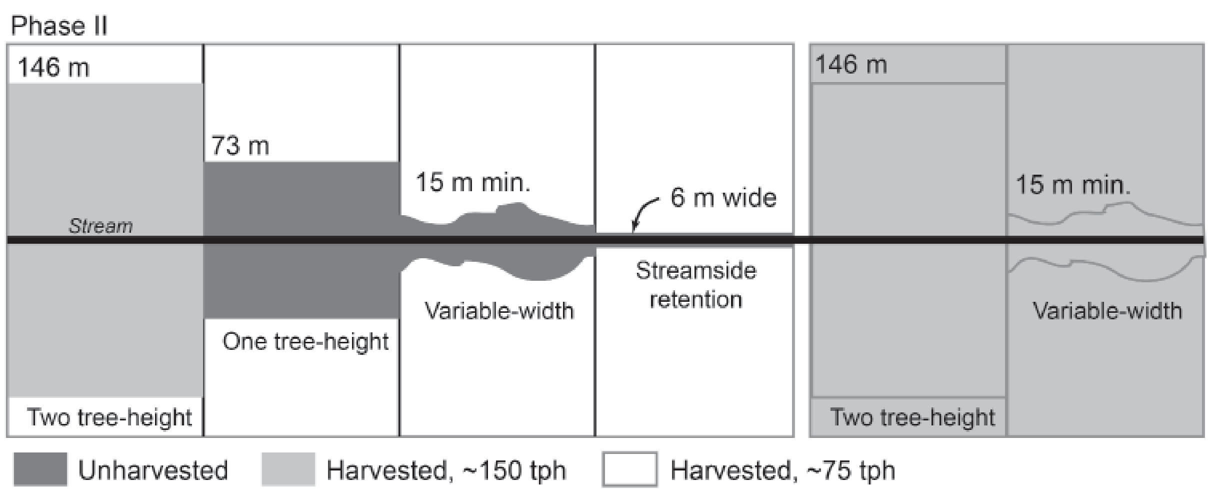

Figure 8—Riparian buffer design for Phase 2 of the Density Management and Riparian Buffer Study of western Oregon. Thin-through buffers were implemented at selected case-study reaches.

torrent salamanders, would be needed to address uncertainty about potential long-term or site-specific detrimental effects. Similarly, where neighboring private-land parcels continue to be managed using regeneration-harvest approaches, the utility of adjoining federal land parcels such as those included in this study is likely to become increasingly important for aquatic resource protection. Monitoring of selected resources into the future will help to gain knowledge about cumulative effects over time and this potential emerging role of federal lands for aquatic-resource conservation emphasis when such lands are nested within a patchwork (e.g., checkerboard pattern) of mixed land ownerships.

Ongoing research efforts are using these study sites and the overall experimental design as a template to explore emerging topics. For example: 1) knowledge gained by these studies has contributed to development of landscape-level designs for managing joint aquatic-terrestrial habitat connectivity (Olson and Burnett 2009, 2013); and 2) climate change scenarios are under study relative to headwater stream-flow and the interaction of water availability, thinning treatments, and vegetation growth. These study sites are valuable for the continued monitoring of phase 2 treatment effects, and for potential future research opportunities.

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The Riparian Ecosystem Management Study: Response of Small Mammals to Streamside Buffers in Western Washington

Martin G. Raphael and Randall J. Wilk

Abstract

One of the fundamental concepts behind the conservation strategy in the U.S. federal Northwest Forest Plan is the importance of habitat buffers in providing functional stream and streamside ecosystems. To better understand the importance of riparian buffers in providing habitat for associated organisms, we investigated responses of small mammals to various streamside management options in western Washington forests. First, we conducted a retrospective study at 49 first- to third-order streams on state, federal, and private lands on the Olympic Peninsula, Washington. Streams represented six forest and stream-buffer conditions ranging from unlogged control to young second growth with no buffers. Because of high variability among sites within each condition, we found few differences in responses of mammals to site conditions; however there were species-specific associations with stream size, gradient, and site elevation. In hopes of reducing site variability, we subsequently undertook an experimental study on state lands in western Washington along 23 first- and second-order headwater streams to evaluate small mammal populations both before and after riparian and upland forest treatment. The four treatments, grouped within blocks of adjacent catchments, included an unlogged control, and continuously buffered streams, patch-buffered streams, and unbuffered streams within upland clearcuts. We sampled small mammals in 2003 before logging and then for two years following logging. We found significant changes in abundance of most common species following logging, including a decline in abundance of Keen's Mouse (*Peromyscus keeni*) in all treatments relative to controls, and increases in abundance of the Creeping Vole (*Microtus oregoni*), Southern Red-backed Vole (*Myodes gapperi*), and Townsend's Chipmunk (*Tamias townsendii*). We found capture rates of some of the more specialized species were too low to allow conclusions about treatment effects. For these rarer species, more focused autoecological studies will be needed to better evaluate their habitat relationships along headwater streams and their potential responses to streamside buffer treatments.

Keywords: Riparian buffers, small mammals, streamside management.

Introduction

Management of streamside vegetation is a major component of the Northwest Forest Plan for federally managed lands in western California, Oregon, and Washington (FEMAT 1993). One of the fundamental concepts behind the conservation strategy in the Northwest Forest Plan is the importance of riparian habitat buffers in providing functional stream and streamside ecosystems. These buffers, as proposed by the Scientific Analysis Team (Thomas et al. 1993) and FEMAT (1993) are meant to conserve habitat conditions not only for at-risk stocks of

Martin G. Raphael is a research wildlife biologist, and **Randall J. Wilk** is a wildlife biologist, USDA Forest Service, Pacific Northwest Research Station, Forestry Sciences Laboratory, 3625 93rd Ave. SW, Olympia, WA 98512; mraphael@fs.fed.us

fish but also a diverse range of riparian-associated organisms including lichens, liverworts, fungi, vascular plants, invertebrates, and vertebrates (USDA and USDI 1994). The size of these buffers, and their placement on all stream orders including headwater and intermittent streams, has resulted in the reservation of a large portion of the landscape, ranging from 30 percent to 70 percent of the federal land base (USDA and USDI 1994). Although the size of these buffers was determined from a thorough review of existing literature, very few field data were available comparing the efficacy of alternative buffer designs at the time of their conceptual development. Understanding relationships between biodiversity and watershed function and watershed condition may lead to opportunities to better balance commodity production and protection of streamside habitat.

Much attention has focused on the role of buffers in conserving spawning habitat for salmonids and in providing habitat for streamside and aquatic amphibians. Less attention has focused on the efficacy of these buffers for conservation of small mammals. In a recent review, Anthony et al. (2003) noted that 95 of the 147 mammal species of Oregon and Washington use riparian zones to varying degrees. They classified the degree of association for each of these 95 species and found that 9 are riparian obligates (they require free water or riparian vegetation for some aspect of their natural history); 33 are riparian associates (they are found at statistically greater abundance in riparian habitats compared with uplands); 5 are early-seral species that use recently cut-over forest along streams, and 52 are non-riparian species that use riparian ecosystems but whose abundance in riparian habitats does not differ from abundance in upland habitats. Given that a large number of mammalian species use riparian forest, we undertook retrospective and experimental studies to investigate how small mammals respond to riparian buffers retained during upland forest harvest practices in western Washington.

Retrospective Study

As a preliminary effort toward better understanding the importance of riparian buffers in providing habitat of associated organisms, we initiated a retrospective study of responses of aquatic and terrestrial organisms to various streamside management options (Raphael et al. 2002). Experimental manipulation of riparian systems, evaluated both before and after treatment, is a preferable approach, but such treatments were not possible on federally managed lands during development of our study in the mid-1990s in western Washington. We gained insights using our retrospective approach of past management practices to more effectively develop a later manipulative study, detailed below.

Using a retrospective approach, sixty-two study sites were located on the west and south sides of the Olympic Peninsula, Washington, on state, federal, and private lands outside the Olympic National Park. A study site included a 300-m stream reach plus the adjacent forest area. We chose 300 m as our standard because we suspected that a shorter reach would likely reflect the effects of upstream influences rather than the channel and adjacent stands. Streams were usually first- to third-order, and <3 m across the wetted portion. We selected sites representing six forest conditions:

1. Old forest: unmanaged with intact forest on both sides of the stream;
2. Buffered old forest: old forest with adjacent clearcuts leaving buffers of 10 to 30 m;
3. Young forest: 35- to 100-year-old forest with no adjacent harvest;
4. Thinned young forest: 35- to 100-year-old forest with commercial thinning extending to the stream;
5. Buffered young forest: 35- to 100-year-old forest with adjacent clearcuts leaving 10- to 30-m-wide buffers of young forest adjacent to the stream; and

6. Early regeneration forest: 0- to 35-year-old forest on recent clearcuts with no intact buffer.

Stand types were not equally distributed across the study area because of differing ownerships and management practice histories.

Our approach was a retrospective analysis that examined existing forest condition resulting from past management practices with different intervals since logging, with and without streamside buffers. Raphael et al. (2002) sampled mammals along transects that ran parallel to the stream. We established two transects along each stream reach, one close to the stream (generally within 20 m) and one upslope, spaced 100 m apart. Raphael et al. (2002) reported results from only the near-stream transect, and those are the findings reported here because they pertain more directly to the riparian buffer portion of the study. Small mammals were trapped with Sherman live-traps. Pairs of traps were placed at 10-m intervals along transects for a total of 62 traps per transect.

Total small mammal captures averaged 5.9 individuals per 100 trap-nights; mean numbers of captures did not vary significantly among stand conditions (fig. 1). Raphael et al. (2002) found significant differences in relative abundance of the Pacific Jumping Mouse (*Zapus trinotatus*) among stand conditions, with greater abundance in younger stands than older stands regardless of buffer type. Abundance of the four other more common species did not differ significantly among sites. Capture rates of four of the five species were correlated with physical stream features. Keen's Mouse (*Peromyscus keeni*) was more abundant in smaller streams at lower elevations. The Deer Mouse (*P. maniculatus*) was also more abundant in smaller streams, but its abundance was not correlated with elevation or stream gradient. The Dusky Shrew (*Sorex monticolus*) was more abundant at higher elevations in steeper streams. Abundance of the Pacific Jumping Mouse increased with lower elevations. Abundance of Trowbridge's Shrew (*S. trowbridgii*) did not vary significantly with any of the physical attributes

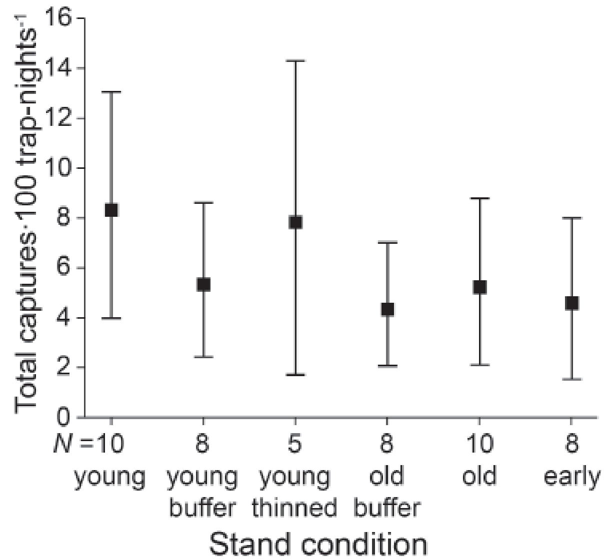

Figure 1—Mean number of captures of small mammals per 100 trap-nights in a retrospective study of existing site conditions on the Olympic Peninsula, Washington (from Raphael et al. 2002).

of streams. O'Connell et al. (2000), in their study of riparian sites in managed forests of the Cascade Range of Washington, found that total abundance and species richness of birds and small mammals using areas close to streams before any timber harvest were comparable to the number and kinds after harvest. At the species level, O'Connell et al. (2000) found that abundance of Pacific Jumping Mouse increased in buffers along streams following harvest; our data, while not conclusive, are consistent with their observation.

Experimental Study

In hopes of reducing the confounding effects of site variability, we subsequently undertook an experimental study on state lands in western Washington along 23 first- and second-order headwater streams to evaluate conditions both before and after treatment (Wilk et al. 2010). The four treatments, grouped within blocks of adjacent catchments to reduce variation in stream size, gradient, and elevation, included unlogged control streams, continuously buffered streams (strips), patch-buffered streams, and unbuffered streams within upland cut-over

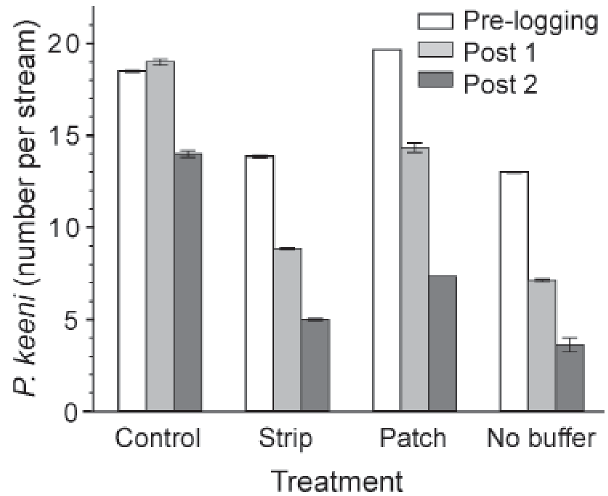

Figure 2—Mean number of Keen's Mice (*Peromyscus keeni*) captured per stream reach before and after treatment in four different stream buffer treatment types (from Wilk et al. 2010). Error bars represent 90 percent confidence interval.

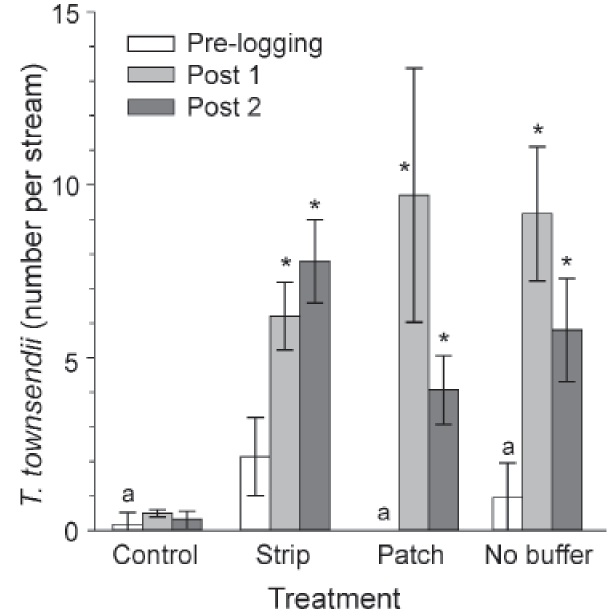

Figure 3—Mean number of Townsend's Chipmunks (*Tamias townsendii*) captured per stream reach before and after treatment in four different stream buffer treatment types (from Wilk et al. 2010). Error bars represent 90 percent confidence interval. Asterisks indicate differences in abundance between pre- and post-treatments; letters indicate a difference between treatment and control within years.

stands. We sampled small mammals in 2003 before logging and then for 2 years following logging. We sampled by using 36 live traps (half on each side of stream) spaced at 5-m intervals within 2 m of the stream. We checked traps daily over 3 successive nights in 5 bouts that were 2 to 3 weeks apart. Among all taxa with sufficient numbers of captures for analysis, we found significant changes in abundance of each species following logging, including a larger decline in abundance of Keen's Mouse in all treatments relative to controls (fig. 2), and increases in abundance of the Creeping Vole (*Microtus oregoni*), the Southern Red-backed Vole (*Myodes gapperi*), and Townsend's Chipmunk (fig. 3). We also observed increases in abundance for the Deer Mouse in continuous buffers and no buffers, and the Pacific Jumping Mouse in patch buffers. The Shrew-mole (*Neurotrichus gibbsii*) decreased in the streams with no buffers. Relative to controls, diversity of small mammals (including total captures, species richness, and species evenness) did not change significantly following logging in any of the treatments (fig. 4). Variation among sites was large, making differences among species in each treatment difficult to detect (example:

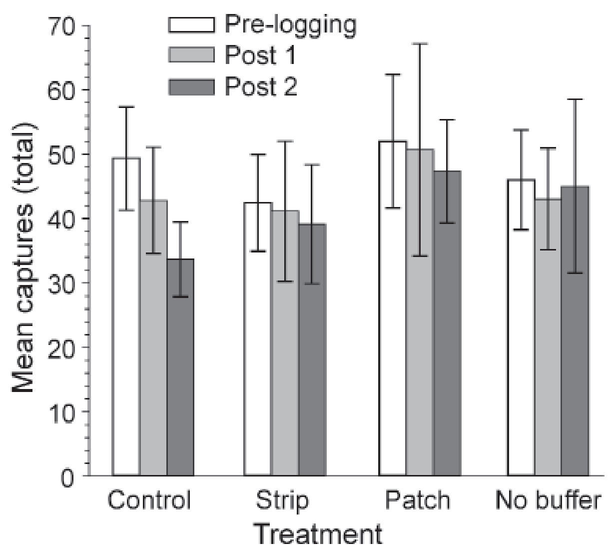

Figure 4—Mean total number of captures of small mammals per 100 trap-nights in an experimental study of stream buffer treatments on state lands in western Washington (from Wilk et al. 2010). Error bars represent 90 percent confidence interval.

Trowbridge's Shrew and Dusky Shrew that had high abundance but large SE). Differences we observed could be due to change in forest cover, not riparian influences per se. Overall, we observed that total abundance of small mammals did not differ much among treatments, but that species composition did change (fig. 5). We also found that capture rates of some of the more specialized species such as Water Shrews (*Sorex palustris*) and Marsh Shrews (*S. bendirii*)] were too low to allow conclusions about treatment effects. For these rarer species, more focused autoecological studies will be needed to better evaluate their habitat relationships along headwater streams and their potential responses to streamside buffer treatments.

Although we observed significant effects of alternative buffer designs on abundance of most of the more common small mammal species in our study, the experimental approach was challenging to implement consistently among sites. These challenges need to be acknowledged, because they may have affected results and constrain the scope of inference of our findings.

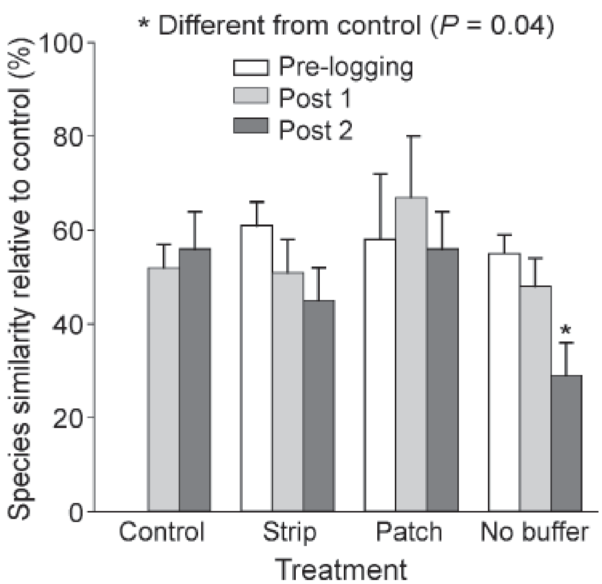

Figure 5—Changes in species composition of small mammals (percent similarity with control sites) before and after logging to create alternative buffer designs on state lands in western Washington (after Wilk et al. 2010). Error bars represent ± SE.

Treatments were not assigned randomly and the design was not balanced (not all treatments occurred in all blocks) owing to logistical constraints associated with roads and anticipated costs of treatments. Furthermore, treatments did not occur when originally planned, due to realities of timber markets driving the timing of timber sales. Implementation of timber harvest among years introduces potential time effects on small mammal populations that were not controlled among sites. Such a time effect may be reflected in our single pre-treatment year of data collection as well. Implementation of buffer design, especially strip buffers, varied noticeably, from dense shading to narrow and open. Finally, windstorms caused blowdown of trees within buffers on many of the treated sites, causing changes in habitat structure that obscured some differences among treatments. Despite these challenges, stand-scale forest experimental studies are valuable for the insights they can provide us regarding responses to anthropogenic disturbances under conditions that are largely controlled. Our mammal responses to combined riparian buffers and upland cut-over area provide such insights that can be tested elsewhere.

Overall, our results demonstrate that populations of the more common species of small mammals in our study changed following logging along streams. Changes in abundance we observed were species-specific, with some species abundances increasing and some decreasing following harvest. The changes we observed seemed to reflect change in amount of preferred forest habitat, either early-seral or late-seral, depending on species. We could not show that these changes were due to retention of streamside buffers per se. Furthermore, none of the common species we were able to sample adequately included riparian obligates as identified in Anthony et al. (2003). Our results do not mean that riparian buffers are not important for small mammals. Retention of suitable habitat along streams will contribute to the total amount and connectivity of habitat at broader scales, and thus should contribute to conservation of these species in managed landscapes. The potential role of riparian buffers could include enhanced opportunities for dispersal of populations of small mammals (e.g., Doyle 1990), but our sampling was not designed to evaluate dispersal rates. Future studies should include an explicit investigation of the role of riparian buffers as dispersal corridors, as well as more detailed investigation of population fitness of selected species in relation to buffer design.

Acknowledgments

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Evaluating Headwater Stream Buffers: Lessons Learned from Watershed-scale Experiments in Southwest Washington

Peter A. Bisson, Shannon M. Claeson, Steven M. Wondzell, Alex D. Foster, and Ashley Steel

Abstract

We present preliminary results from an experiment in which alternative forest buffer treatments were applied to clusters of watersheds in southwest Washington using a Before-After-Control-Impact (BACI) design. The treatments occurred on small (~2- to 9-ha) headwater catchments, and compared continuous fixed-width buffered, discontinuous patch-buffered, and unbuffered streams to an adjacent unlogged reference catchment. Eight treatment clusters were monitored from 2001 to 2006; four were located in the Black Hills (Capitol State Forest) and four in the Willapa Hills of the Coast Range. Logging took place in 2004 or 2005, depending on the cluster. The study streams were too small to support fishes, but catchments did harbor amphibians, aquatic invertebrates, and riparian mollusks. In addition to biota, we examined water quality, discharge, and organic matter inputs. The intent was to monitor the sites two years pre-treatment and three years post-treatment, although unforeseen circumstances caused some exceptions. Overall, results suggested that relatively small but measurable changes in ecological condition occurred in most catchments where logging occurred. Changes were most apparent in streams having no buffers. In catchments with no buffers, summer water temperature increases were largest, organic matter inputs declined, and drifting invertebrates increased or decreased depending on their trophic guild. Changes in catchments with discontinuous patch buffers were often complex and generally less detectable, and streams with continuous fixed-width buffers tended to exhibit the fewest changes in invertebrate communities and organic matter inputs relative to reference sites.

Analyses of ecological response, both physical and biological, were fraught with difficulty. Difficulties in executing the study and analyzing results include operational planning and scheduling, spatial and temporal variability, and unplanned environmental disturbances. Based on our experience, we offer suggestions for future research on riparian management in small watersheds. First, think of watershed-scale studies as interventions that are carried out so as to maximize what we can learn from them, rather than as true experiments. Second, if the study covers several locations, thoroughly characterize differences in physical and biological features among the sites before treatments are applied to avoid misinterpreting results. Third, be prepared to accommodate uncontrolled environmental disturbances (e.g., droughts, floods, wildfires, etc.) that are inevitable in multi-year investigations. Fourth, once the basic experimental layout is

Peter A. Bisson is a research fish biologist (emeritus), **Shannon M. Claeson** is a professional ecologist, and **Alex D. Foster** is a professional ecologist, USDA Forest Service, Pacific Northwest Research Station, Forestry Sciences Laboratory, 3625 93rd Avenue SW, Olympia, WA 98512-9193; pabisson@fs.fed.us; **Steven M. Wondzell** is a research aquatic ecologist, USDA Forest Service, Pacific Northwest Research Station, Forestry Sciences Laboratory, 3200 SW Jefferson Way, Corvallis, OR 97331; **Ashley Steel** is a statistician/research quantitative ecologist, USDA Forest Service, Pacific Northwest Research Station, Forestry Sciences Laboratory, 400 North 34th Street, Suite 201, Seattle, WA 98103

established, resist the temptation to switch treatments midway through the study, which will only confound analyses. Finally, when surprises occur, be flexible enough to monitor their effects to take advantage of learning opportunities.

Keywords: riparian management, buffers, headwater streams, small watershed studies, experimental design, BACI.

Pesticide Precautionary Statement

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Introduction

When the Northwest Forest Plan (NWFP) was implemented in the late 1990s, one of the most significant information gaps was whether the new buffer guidelines in the NWFP conserved sufficient amounts of riparian forest to protect fish habitat in headwater streams. The default guideline in the NWFP called for riparian buffers as wide as or wider than a site-potential tree height on each side of stream channels. While there were theoretical reasons to believe that wide buffers protected headwater stream functions and processes (Sedell et al. 1994), field studies verifying the efficacy of wide buffers were lacking. Furthermore, the NWFP's riparian conservation areas were focused on habitat protection for salmonid fishes, but questions remained about whether the guidelines adequately protected the habitats of other aquatic and riparian-associated species or the processes that provide organic matter and sediment to downstream salmon-bearing rivers.

To address the scarcity of information about the effectiveness of buffers of different width and forest age, we initiated a study in 1996 (Riparian Ecosystem Management Study—REMS) that employed a synoptic survey approach in which a large number of streams in watersheds with different logging histories and different buffer patterns were compared. The research was located on Washington's Olympic Peninsula, and included both aquatic and riparian-associated vertebrates (Bisson et al. 2002; Raphael et al. 2002). As a part of REMS, faunas were compared for small, perennially-flowing streams that included sites where no buffers were left during timber harvest, sites where narrow (<20 m) buffers were left, sites where wide buffers that had been thinned were left, and sites where intact unmanaged buffers >100 m were left. Using correlation analysis, we found that fewer fish were associated with late-seral riparian buffers; however, fish densities were strongly influenced by local stream habitat conditions such as the abundance of pools. Further, fish abundance was positively correlated with riparian characteristics associated with increased primary production —light gaps, nutrients and deciduous organic matter inputs. Stream-dwelling amphibians were positively correlated with late-seral forest in riparian zones and the amount of late-seral forest in their watersheds. Their abundance was also negatively associated with roads.

While these observations were interesting and our results were consistent with the findings of other stream investigations in the Pacific Northwest, we felt that a broad-scale survey approach relying on correlation analysis did not yield many new insights into how environmental factors related to riparian buffers directly or indirectly influenced the vertebrates of interest. To learn more about the effects of different-sized buffers on headwater streams, we undertook a second phase of REMS that applied alternative riparian buffer treatments to entire small watersheds using a Before-After-Control-Impact (BACI) experimental design. We anticipated monitoring the study catchments for two years prior to treatment and three years after treatment. We also decided to focus our research on the very smallest non-fish bearing headwater streams. The influence of riparian buffers on very small headwater streams had been little studied, and the current Forest Practices Act in Washington state does not require that buffers be left along non-fish bearing channels on state- or privately-managed lands.

This paper describes the planning, design, implementation, and results of the second phase of the REMS study. We examine the problems and unforeseen surprises encountered over the course of the study, and the lessons learned that might be applicable to future watershed-scale studies of a similar nature. The BACI approach is widespread in ecological science (Downes et al. 2002), but we experienced a number of design, implementation, and analytical hurdles in applying it to our situation. The concluding section of the paper suggests general guidelines for investigators contemplating similar studies. These lessons are largely independent of the specific results of REMS, but hopefully will be of some use in future research.

Site Selection

We experienced many of the difficulties in conducting landscape-scale experiments in AMAs (Adaptive Management Areas—see *http://www.reo.gov/ama/index.htm*) that were discussed by Stankey et al. (2003). Our original intention was to compare the NWFP default buffers for headwater streams with other buffer configurations on the Olympic Peninsula where the first REMS phase was completed. We hoped the study could take place in an AMA that was designated as a place where departures from the standards and guidelines in the NWFP could be tested experimentally. We found, however, that the Olympic National Forest was unable to implement a watershed-scale experiment that involved different stream buffers in any AMAs on the Olympic Peninsula. Two other national forests in western Washington also declined to participate in the study. Several reasons were given, including the scale at which we wanted to apply the treatments, concerns over potential environmental litigation, and conflicts between our proposed experiment and other forest management plans. Taken together, these restrictions made it impossible for the research to take place in an AMA.

We then approached the Washington Department of Natural Resources (DNR) to explore opportunities for implementing watershed-scale buffer trials in western Washington on state-managed lands. Under DNR's guidance several study areas were located in the Coast Range of southwestern Washington (fig. 1) where alternative buffer treatments could be applied to small non-fish bearing streams in headwater catchments of approximately 5 ha.

Objective and Treatments

The objective of REMS Phase 2 was to compare the ecological effects of 15- to 20-m continuous ("fixed-width") buffers on each side of the channel, discontinuous ("patch") buffers, and no buffers ("clearcuts"), with adjacent reference sites of mature second-growth forest. One to two site-potential tree height buffers in the NWFP were not included among the treatments, as these were not an option under DNR's Habitat Conservation Plan (Washington State Department of Natural Resources 2005).

Current Washington state law does not require buffers on small non-fish bearing headwater streams, i.e., those extending from the channel

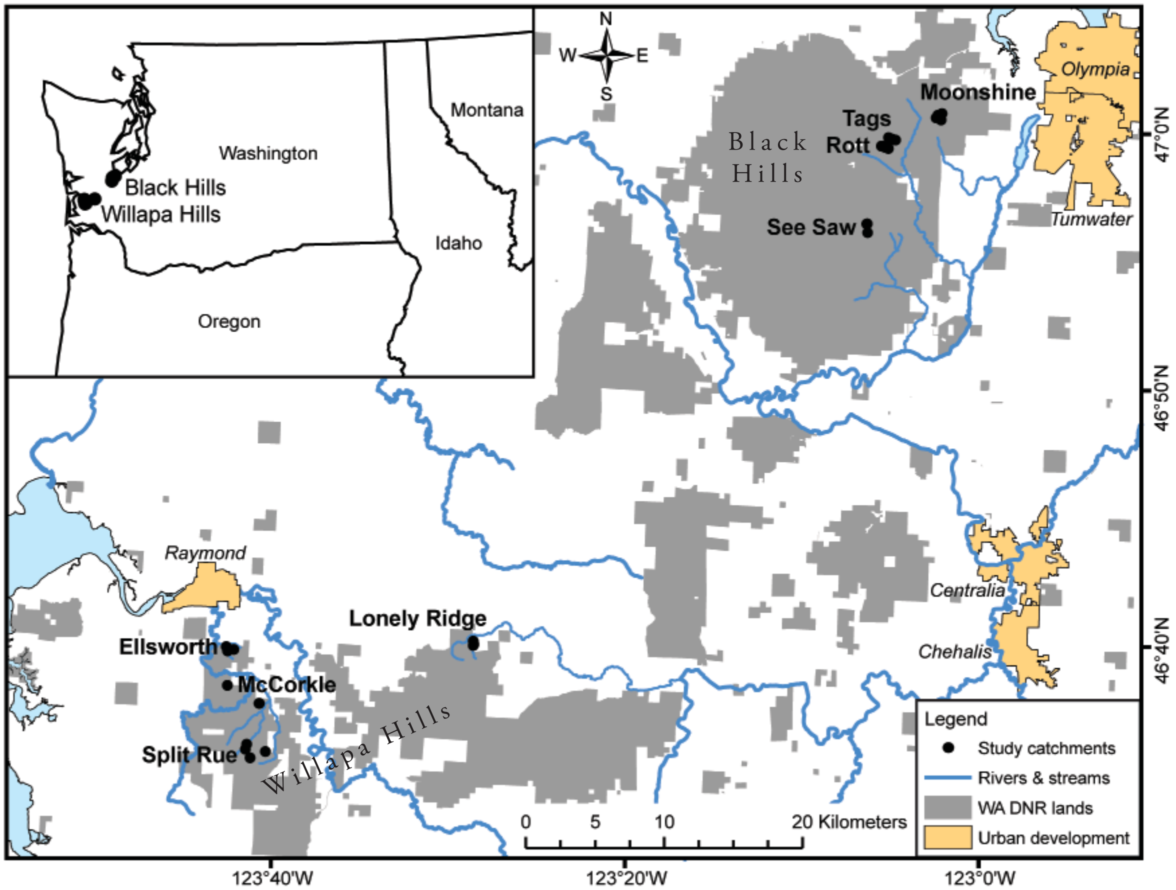


Figure 1—Location of eight clusters of non-fish-bearing headwater streams sampled in REMS Phase 2 (Riparian Ecosystem Management Study) within two regions of southwestern Washington, USA. Capitol Forest sites are located in the Black Hills area near Olympia, WA, and the Willapa Hills sites are located near Raymond, WA.

initiation point to the confluence with a larger stream. These streams, however, are protected from heavy equipment intrusion and in-channel yarding during logging operations. The fixed-width and patch-buffer treatments in our study represented alternative protection measures that would provide additional protection, beyond heavy equipment exclusion, for very small headwater streams, but these buffers would not include the relatively wide site-potential tree height buffers that were the default standard on federal lands. Thus, our study was most directly applicable to headwater streams in state and private industrial forests.

We selected clusters of small watersheds in two locations—Capitol State Forest southwest of Olympia, Washington, and the Willapa Hills, an area of mixed forest ownership which drains into Willapa Bay north of the Columbia River (fig. 1). Each location contained four clusters of headwater streams, and each cluster contained a reference catchment and a combination of treatment types (table 1). It was our goal at the outset to establish a balanced experimental design with each treatment represented in each cluster; however, this turned out to be impossible for operational reasons. Cluster size ranged from three to five catchments because timber sale plans and forest stand characteristics did not always accommodate harvesting small watersheds in the exact configuration we desired. Occasionally a cluster would possess two similar treatments and omit one treatment type. Furthermore, engineering considerations mandated that reference catchments be located at an end of a cluster in order to avoid roads and other harvest-

Table 1—Physical characteristics of eight clusters of 30 headwater catchments in the Pacific Coast Range, Washington, USA. Table modified from Janisch et al. (2012).

Cluster	Logging initiation	1st post-logging year	Catchment logging treatment	Area[a] (ha)	Channel length[b] (m)	Bankfull width[c] (m)	Channel gradient[d] (%)	Aspect	Elevation[e] (m)	Seasonally dry
Black Hills (Capitol Forest)										
Moonshine	July 2005	2006	Reference	8.5	173	1.2	36	W	393	yes
			Patch	8.5	270	0.6	32	W	287	no
			Fixed-width	2.7	176	0.4	35	W	318	yes
			Patch	4.8	176	1.8	37	W	390	no
Rott	April 2004	2004	Reference	6.0	391	0.9	29	S	246	no
			Fixed-width	7.3	403	0.8	37	S	288	no
			Clearcut	4.5	123	0.9	39	S	303	yes
			Patch	5.1	165	0.4	42	S	314	no
See Saw	September 2003	2004	Reference	6.5	173	2.3	27	N	336	no
			Clearcut	2.1	229	0.4	13	N	212	no
			Fixed-width	4.2	273	0.7	18	NW	212	no
Tags	January 2004	2004	Reference	5.5	206	—	46	NE	193	no
			Fixed-width	3.9	241	—	41	NE	203	no
			Patch	4.4	270	—	45	NE	218	no
			Patch	5.4	280	—	40	NE	230	no
			Clearcut	4.9	297	—	38	NE	234	no
Willapa Hills										
Ellsworth	February 2005	2005	Reference	1.9	111	—	17	NW	64	no
			Clearcut	3.5	255	0.7	18	SW	28	no
			Fixed-width	8.1	375	0.7	11	SW	12	no
Lonely Ridge	March 2004	2004	Reference	2.8	209	0.5	24	E	168	yes
			Clearcut	1.9	184	0.4	30	E	168	no
			Fixed-width	3.3	263	0.6	21	E	168	no
			Patch	3.1	282	1.4	25	E	169	no
McCorkle	November 2003	2004	Reference	2.7	311	1.8	24	NW	121	no
			Fixed-width	2.6	146	0.4	17	SE	110	no
			Fixed-width	3.5	155	0.5	18	S	110	no
Split Rue	May 2004	2005	Reference	6.2	229	—	26	N	225	no
			Clearcut	4.9	168	—	22	NE	205	no
			Fixed-width	8.1	480	—	21	N-NE	292	no
			Clearcut	3.4	203	—	27	SE	186	no

[a]Headwater catchment area derived from stereo pairs and ERDAS Stereo Analyst®.
[b]Channel length defined as the confluence to headwall or uppermost point of channel definition.
[c]Channel bankfull width calculated as the weighted mean of sub-segments in 2003.
[d]Channel gradient calculated as the weighted mean of sub-segments.
[e]Elevation of the channel confluence as determined from state 30-m DEM.

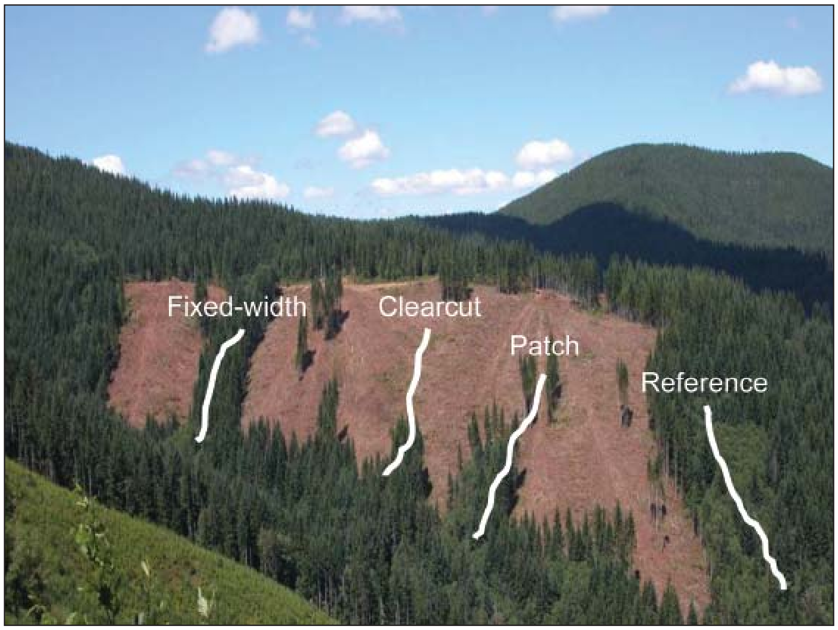


Figure 2—One of the experimental buffer treatment clusters in Capitol Forest showing fixed-width, clearcut, patch, and reference catchments, respectively, from left to right. Stream channels are shown in white. (Photo by Randall Wilk, USFS.)

related factors. For example, the reference catchments for the See Saw and Split Rue clusters were located 1.2 and 3.3 km, respectively, from the treatment catchments. Apart from these constraints, treatments were assigned randomly to individual catchments. A photograph of one of the clusters is shown in figure 2. The lower reaches of the study streams were always forested because they passed through the riparian management zone (RMZ) of the fish bearing stream into which they discharged. All measurements of treatment effects were made at, or above, the boundary of the RMZ into which the headwater stream flowed. Typical views of riparian areas after treatments were applied are shown in figure 3. The streams maintained surface flow continuously during the wet season, but the channels occasionally became intermittent during the dry season, and surface connections with the parent stream were often disrupted, although subsurface flow pathways likely remained active.

Implementation and Monitoring Surprises

We experienced several surprises during the pre-treatment, treatment, and post-treatment periods of the research. Some surprises were management-related; others resulted from natural events. In addition to being unable to establish a complete and balanced study design and to fully randomize treatment assignments, the desired one-year treatment interval was spread over two years when shifts in stumpage prices delayed several timber sales. Furthermore, the delay in harvesting some catchments took place when the regional weather pattern was changing from a multi-year warm, dry period to a cool, wet period caused by a transition from an El Niño to a La Niña weather cycle (Bumbaco and Mote 2010). The change in temperature and precipitation during the treatment period altered the hydrologic patterns of the study streams and, coupled with the harvesting delay in some clusters, added to the difficulty of sorting out treatment effects from weather-mediated changes in the parameters that were monitored.

In 2006, after the timber had been harvested from all study locations, some of the harvest units were treated with aerially applied herbicide as part of routine site preparation prior to reforestation. Although the helicopter operators exercised caution when applying the herbicide, it is likely that occasional overspray or wind drift occurred and some chemical entered the streams; however, precise wind strength and direction data

Figure 3—Post-logging appearance of typical reference and experimental treatments. (Photos by Peter Bisson.)

at the sites were not available. Because this was an unanticipated management action, we were not fully prepared to sample its effects on stream biota, and the fact that some harvest units, but not others, were sprayed made it impossible to determine if herbicide application significantly affected the results of the study.

Additional surprises occurred when severe windstorms in the winters of 2005, 2006, and 2007 caused extensive damage to some of the buffer treatments, especially at several of the Willapa Hills catchments (fig. 4). As a result of the windthrow, some fixed-width buffer treatments were altered to such an extent that they resembled patch treatments. Stream channels at sites with extensive disturbance often contained large amounts of fine sediment that entered streams when root systems were upturned. The windthrow also damaged a number of monitoring stations for forest litter and riparian-associated mollusks. Although windstorms were not factored into the original experimental design, we note that coastal watersheds in western Washington frequently experience high wind events in fall and winter, so occurrence of such storms during the REMS study was not out of the ordinary.

Finally, in November 2007 an intense storm passed through southwestern Washington and some watersheds received >30 cm of rain within 48 hours. Mass wasting was extensive throughout the Coast Range, and a large landslide and accompanying debris flow occurred in one of the patch-buffered study streams (Moonshine cluster) in Capitol Forest. The debris flow caused massive channel scouring and entrainment of riparian vegetation and destroyed existing monitoring equipment. Despite all these challenges, we feel the work yielded some valuable information with relevance to headwater stream management and it taught us several important lessons about

Figure 4—Top: extensive windthrow in a fixed-width buffer site after a windstorm in 2005. Bottom: view of the stream channel showing sediment input caused by upturned rootwads. (Photos by Peter Bisson.)

pitfalls in designing and executing watershed-scale manipulations.

Results

Results are presented for stream temperature, aquatic and riparian invertebrates and terrestrial litter inputs (fine particulate organic matter and coarse particulate organic matter). Due to time and budget constraints some samples were obtained only from a subset of study locations (table 2). Other aspects of the study were headed by other investigators and those aspects are not reported here. We briefly describe methods and statistical tests, but additional details of sampling methods and analytical techniques will appear in subsequent technical reports and publications. Below we present findings of the research thus far.

Stream Temperature

All streams were instrumented with recording temperature loggers and comparisons among treatments focused on the daily maximum temperatures during the July-August period of peak yearly temperatures (Janisch et al. 2012). Relationships between daily maximum stream temperatures in the treatment catchments and their associated reference sites during the pre-treatment phase of the study were used to predict daily maximum temperatures during the warmest part of the summer. After treatments were applied, departures from expected values were computed to determine if the treatment streams differed significantly from what was expected (H_o = no difference). Results for the first post-treatment year, which would be assumed to show the greatest treatment effects, are displayed in figure 5.

Stream temperatures in the unlogged reference catchments generally matched predictions, i.e., no significant changes occurred during the first post-treatment year. Average increases in daily maximum stream temperature were greatest in the clearcut sites, but the other two treatment types also exhibited significant thermal increases. Somewhat surprisingly, stream temperatures in the fixed-width buffer treatments exhibited a greater increase in the first post-treatment year than those in patch buffer treatments, although air temperature increases in the fixed-width buffer sites increased less than air temperatures in the patch buffers (Janisch et al. 2012). Maximum daily summer stream temperatures rose in each treatment category; however, overall temperature increases were small, the trends did not entirely match expected treatment effects (changes tended to be greater at sites with continuous fixed-width buffers than sites with patch buffers), and the responses were highly variable within treatment categories. Further analyses by Janisch et al. (2012) indicated that water temperature

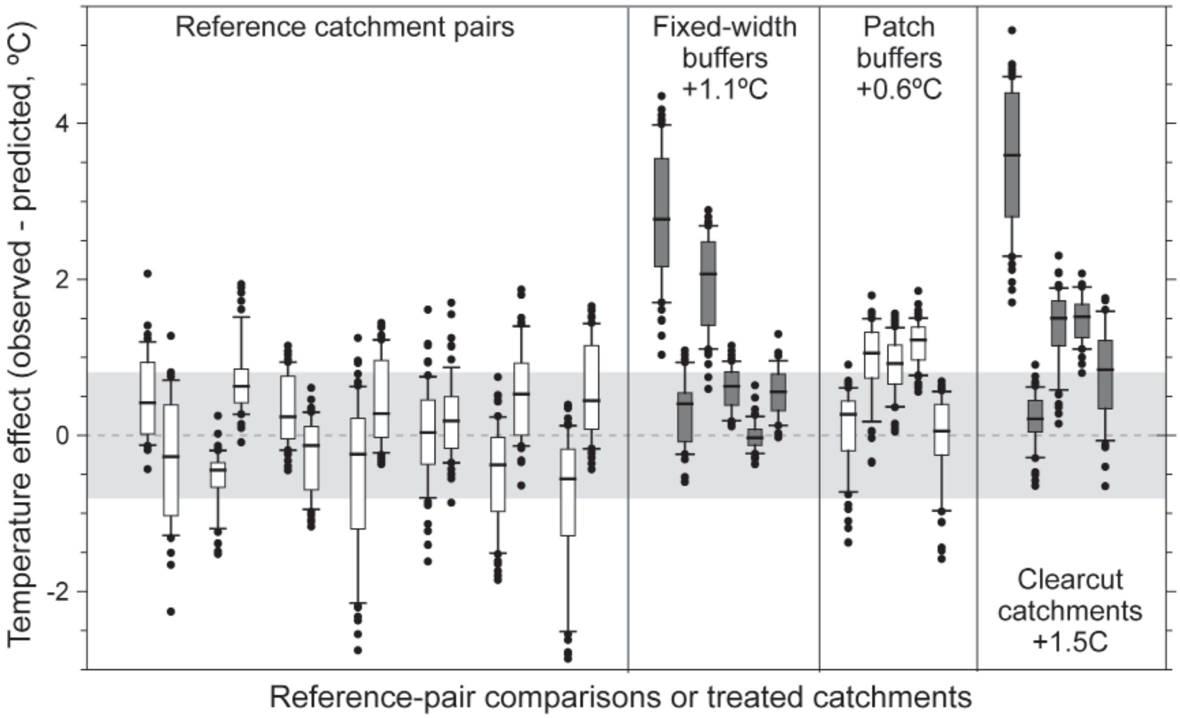


Figure 5—Changes in the maximum daily stream temperature (°C) in the first post-treatment year (or post-calibration year for reference catchments) during July and August at each site. The box and whisker plots denote the mean, quartiles, and 10- and 90-percentiles of water temperature deviations (observed minus predicted daily maximum temperature) in each catchment. Points represent more extreme daily departures from predictions in the first post-treatment year. The 95 percent prediction intervals for the daily random disturbance (in temperature variation; grey-shaded zone) was calculated as 0.00 ± 1.96 · SD of the single largest SD of all pairwise comparisons among reference catchments in the calibration year. The average change in maximum daily stream temperature for each treatment is indicated by a bold line in each box (after Janisch et al. 2012).

was strongly influenced by streambed texture, the presence of small wetlands in the catchment, and the length of channel exposed to sunlight. This suggests that, in addition to the presence or absence of forest canopy, the length of exposed stream channel and the amount of hyporheic water exchange are important factors regulating headwater stream temperatures.

Aquatic Invertebrates

The role of non-fish bearing streams in exporting aquatic invertebrates (potential food items) to fish-bearing streams is poorly understood; however, there is evidence that invertebrates exported from non-fish bearing streams provide important food resources for salmonids downstream (Wipfli and Gregovich 2002). The extremely heterogeneous substrates of the headwater channels, which consisted of a wide range of sizes of unsorted inorganic particles as well as tree roots and a variety of coarse and fine wood fragments, precluded benthic invertebrate sampling. Instead, invertebrates present in the surface water drift were sampled at gauging stations located at the downstream boundary of the headwater streams where the channel entered the riparian management zone of the larger stream into which it flowed.

Drifting invertebrates were collected from six of the eight clusters, three clusters in Capitol Forest and three in Willapa Hills (table 2). Weirs with pass-through PVC pipes were placed in each study stream so that drift nets attached to the pipes were able to capture the entire surface flow during spring, summer, and fall sampling. Drifting invertebrates were collected in 250-µm mesh tubular nets, fished for one full day

Table 2—Type of samples collected from eight clusters of headwater catchments in the Coast Range of Washington, USA. "yes" indicates that metric was sampled, "no" indicates that metric was not sampled at that particular catchment.

Region	Cluster	Treatment	Water temperature	Aquatic invertebrates	Riparian mollusks	Riparian literfall
Black Hills (Capitol Forest)	Moonshine	Reference	yes*	no	no	no
		Patch	no	no	no	no
		Fixed-width	no	no	no	no
		Patch	yes	no	no	no
Black Hills (Capitol Forest)	Rott	Reference	no	yes	no	yes
		Fixed-width	no	yes	no	yes
		Clearcut	no	no	no	yes
		Fixed-width	no	yes	no	yes
Black Hills (Capitol Forest)	See Saw	Reference	yes*	yes	no	yes
		Clearcut	yes	yes	no	yes
		Fixed-width	yes	yes	no	yes
Black Hills (Capitol Forest)	Tags	Reference	yes	yes	yes	yes
		Fixed-width	yes	yes	yes	yes
		Patch	yes	yes	yes	yes
		Patch	yes	yes	yes	yes
		Clearcut	yes	yes	yes	yes
Willapa Hills	Ellsworth	Reference	yes	yes	yes	yes
		Clearcut	yes	yes	yes	yes
		Fixed-width	yes	yes	yes	yes
Willapa Hills	Lonely Ridge	Reference	yes*	yes	no	yes
		Clearcut	no	yes	no	yes
		Fixed-width	yes	yes	no	yes
		Patch	yes	yes	no	yes
Willapa Hills	McCorkle	Reference	yes	no	no	no
		Fixed-width	yes	no	no	no
		Fixed-width	yes	no	no	no
Willapa Hills	Split Rue	Reference	yes	yes	yes	yes
		Clearcut	yes	yes	yes	yes
		Fixed-width	yes	yes	yes	yes
		Clearcut	yes	yes	yes	yes

*Stream was dry during the pre-treatment (calibration) year, so an alternative reference catchment was used for analysis.

at monthly (2003) or bi-monthly (2004–2006) intervals. Water temperature and stream discharge were measured at the beginning and end of each 24-hour period. Drift density (individuals·m^{-3} water) was calculated by dividing invertebrate export by the mean discharge of the 24-h sampling period. Invertebrates were categorized as insect or non-insect, aquatic or terrestrial, and by their functional feeding group as defined by Merritt and Cummins (1996). Drift samples were obtained in spring, summer, and fall, but snow prevented access to some of the sites in winter and no winter data are reported here. We tracked all natural log-transformed, post-year minus pre-year differences of seasonally averaged drift densities. Simple linear regression models were used to detect significant effects ($p < 0.05$) among buffer treatments, post-harvest year, and clusters. Because only three of the six clusters received a patch treatment, the sample size was too small for statistical modeling; therefore, no data from the patch treatments are presented here.

The most common terrestrial invertebrates in the drift were mites (Acari) and springtails (Collembola). Drift density proportions of terrestrial invertebrates increased significantly in clearcuts during summer compared to changes at the fixed-width buffer and reference locations (fig. 6). The proportion of terrestrial invertebrates also increased at clearcut sites relative to reference sites in fall. In spring, no significant differences were observed among the treatment and reference streams.

Mayflies (Ephemeroptera) were abundant aquatic invertebrates in all headwater streams, where they were represented by taxa adapted to scraping periphyton from the substrate (scrapers) or taxa that consumed organic detritus (collector-gatherers). No significant differences in mayfly drift densities were observed among treatment or reference streams in the spring or summer, but mayfly densities in both clearcut and fixed-with buffer treatments declined relative to reference catchments in fall (fig. 7). This was surprising because small mayflies have been reported to prosper in streams where canopy openings have favored increased periphyton production (Hawkins et al. 1983). It is possible that logging and windthrow-related fine sediment inputs

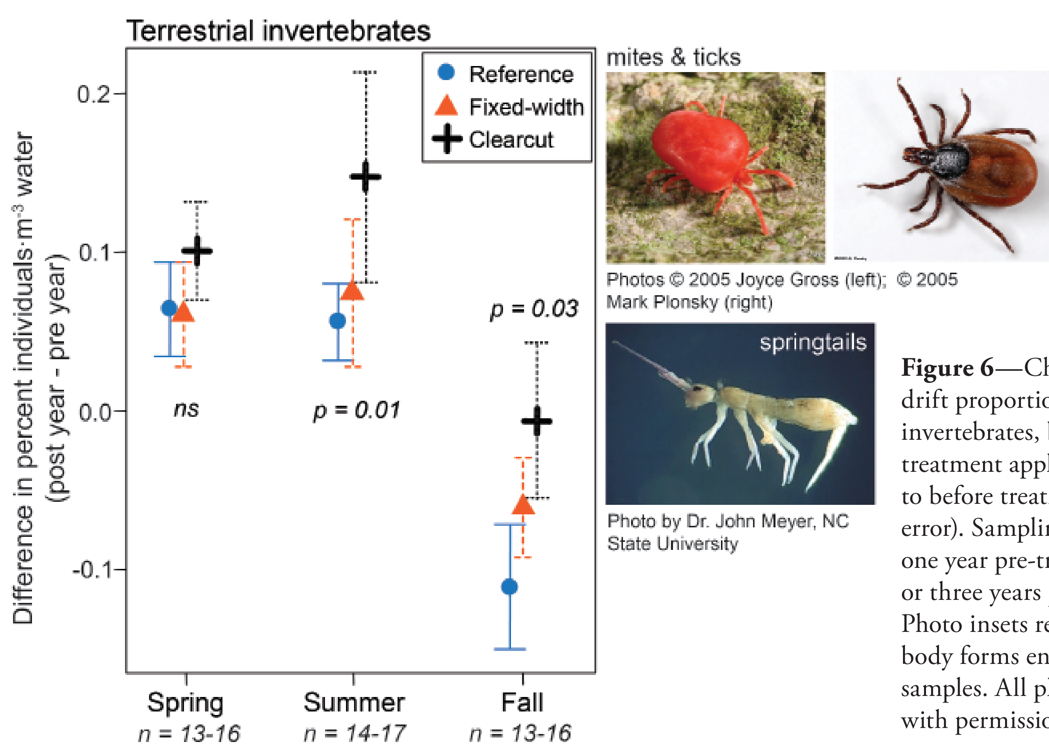


Figure 6—Change in the mean drift proportions of terrestrial invertebrates, by season, after treatment application compared to before treatment (± 1 standard error). Sampling intervals include one year pre-treatment and two or three years post-treatment. Photo insets represent typical body forms encountered in samples. All photos reproduced with permission.

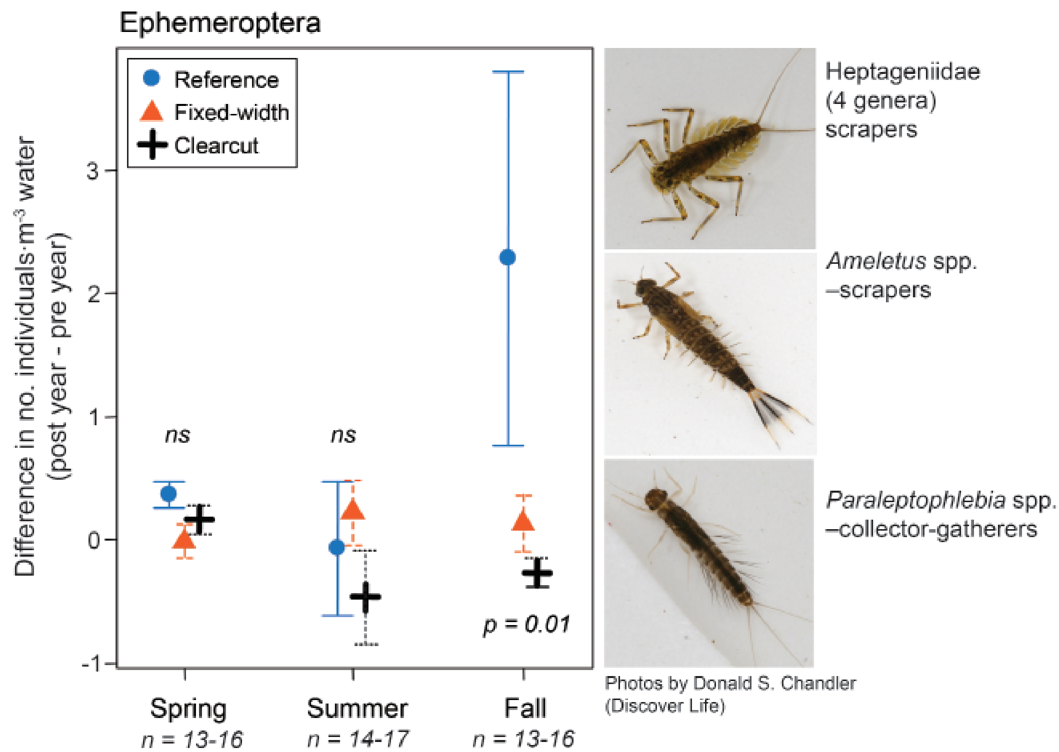


Figure 7—Changes in the mean drift densities of mayflies, by season, after treatment application compared to before treatment (± 1 standard error). Sampling intervals include one year pre-treatment and two or three years post-treatment. Photo insets represent typical body forms encountered in samples. All photos reproduced with permission.

contributed to mayfly reductions at the study locations, as taxa that occurred there are known to prefer coarse substrates.

Midge larvae (Chironomidae) are frequently the most ubiquitous invertebrates in small streams in the Pacific Northwest, and two midge groups (Orthocladiinae and Tanytarsini) were well represented in drift samples. Small-bodied midge larvae commonly enter the drift and often constitute a substantial portion of the diet of rearing salmonids (Chapman and Bjornn 1969). Midges decreased in clearcut catchments relative to the fixed-width buffer and reference catchments in the fall (fig. 8). In the spring and summer, no significant differences among treatment and reference streams were observed. Extremely high variability in spring prevented an apparent increase in midge drift densities in the treatment catchments from being statistically significant.

We classified aquatic invertebrates in drift samples into feeding guilds to determine if buffer characteristics affected functional feeding groups. Feeding guilds included scrapers (organisms that feed on periphyton), shredders (organisms that "shred" leaves and other riparian inputs), and collector-gatherers (organisms that feed on fine particulate organic material). No changes in shredder density proportions were observed in spring, summer, or fall (fig. 9). Significant declines in scrapers occurred in fall at clearcut catchments relative to fixed-width buffer and reference locations. Likewise, collector-gatherers decreased at clearcuts in fall, but not during other seasons. Seasonal density proportions of other feeding guilds, such as collector-filters or predators, did not differ among treatment or reference catchments.

Riparian Mollusks

We sampled the ground-dwelling riparian mollusk community at three of the clusters (table 2) to determine if buffer treatments affected the spring and fall densities of snails and slugs.

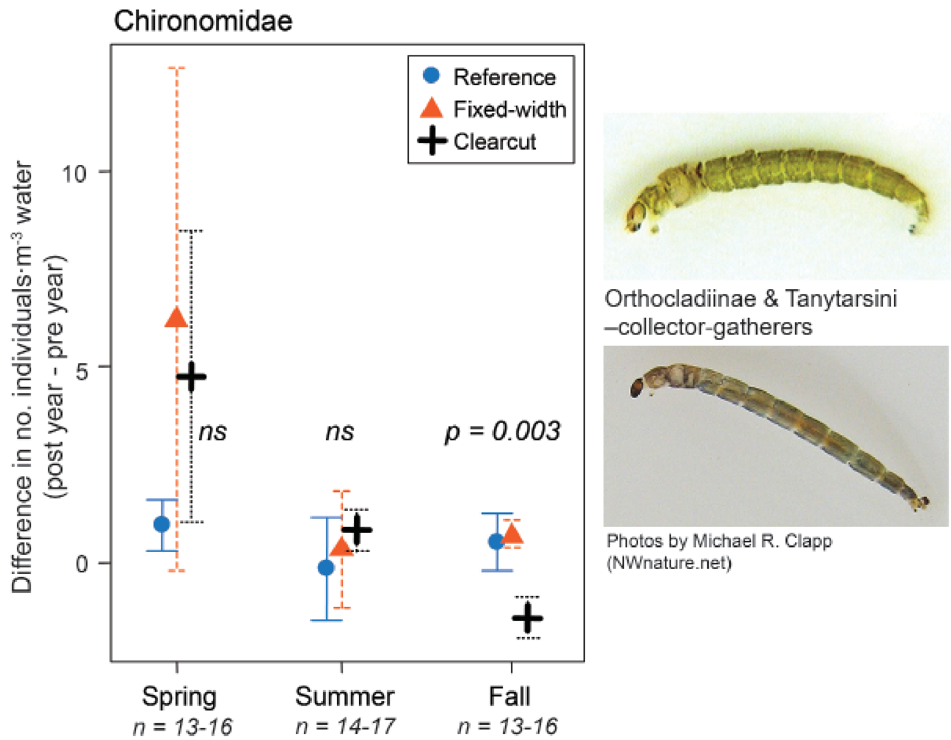


Figure 8—Changes in the mean drift densities of midge larvae, by season, after treatment application compared to before treatment (± 1 standard error). Sampling intervals include one year pre-treatment and two or three years post-treatment. Photo insets represent typical body forms encountered in samples. All photos reproduced with permission.

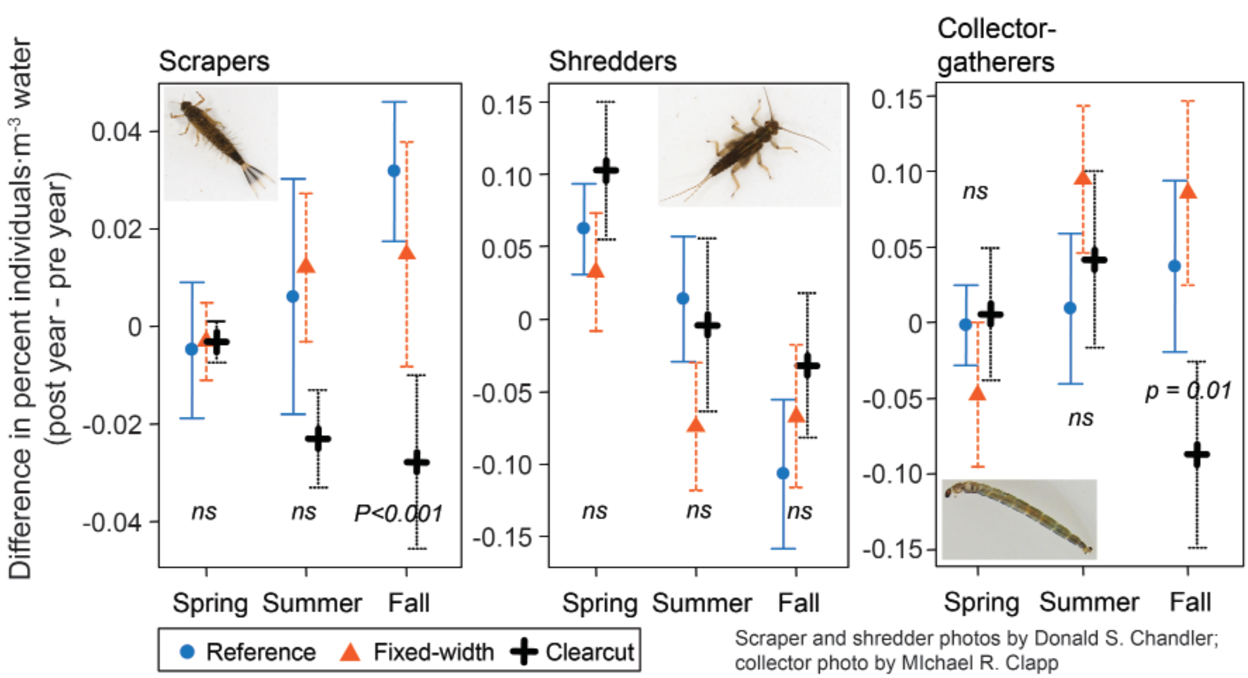
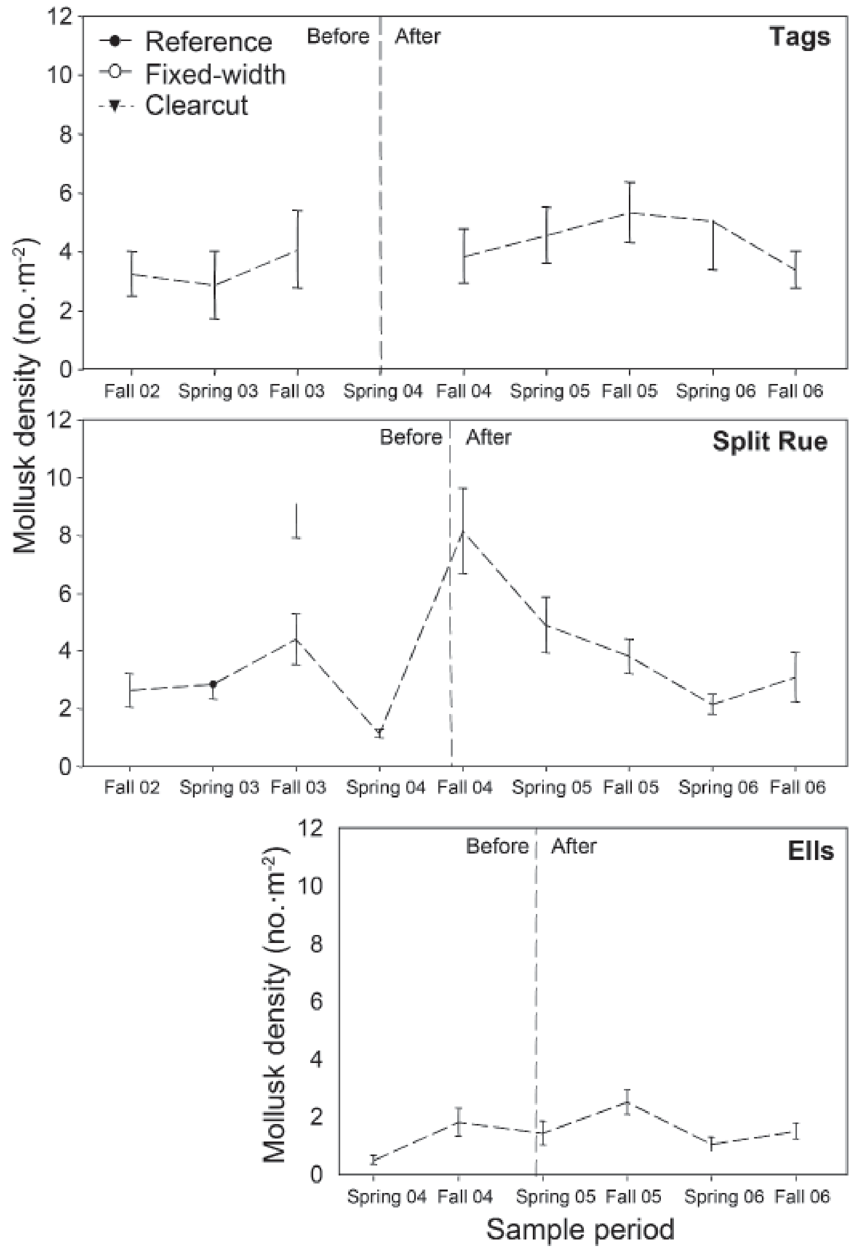


Figure 9—Changes in the mean drift proportions of scrapers, shredders, and collector-gatherers, by season, after treatment application compared to before treatment (± 1 standard error). Sampling intervals include at least one year pre-treatment and two years post-treatment. All photos reproduced with permission.

We focused on spring and fall seasons because mollusks burrow deep into the riparian forest litter in summer and some of the sites were inaccessible in winter. We used arrays of 30 x 30-cm artificial substrates made of laminated paperboard set out in evenly spaced patterns adjacent to the streams. After allowing several weeks for weathering and colonization, the laminations were peeled back and inspected under magnification for mollusks and other organisms. Total mollusk densities (all species) in three of the watershed clusters are presented in figure 10. Once again, because a limited number of catchments with patch buffers were sampled, only data for clearcut, fixed-width, and reference catchments are presented.

In general, mollusk densities between treatment and reference catchments did not change after logging (fig. 10). The reason mollusks were less abundant in catchments designated for no riparian buffers (i.e., clearcut) even before treatments were applied is not known, although we suspect that watershed features unrelated to logging (e.g., presence of seeps) played an important role in influencing them, and the

Figure 10—Mean total mollusk density (+ 1 standard error), by season, before and after treatment application (vertical dashed line) in three watershed clusters.



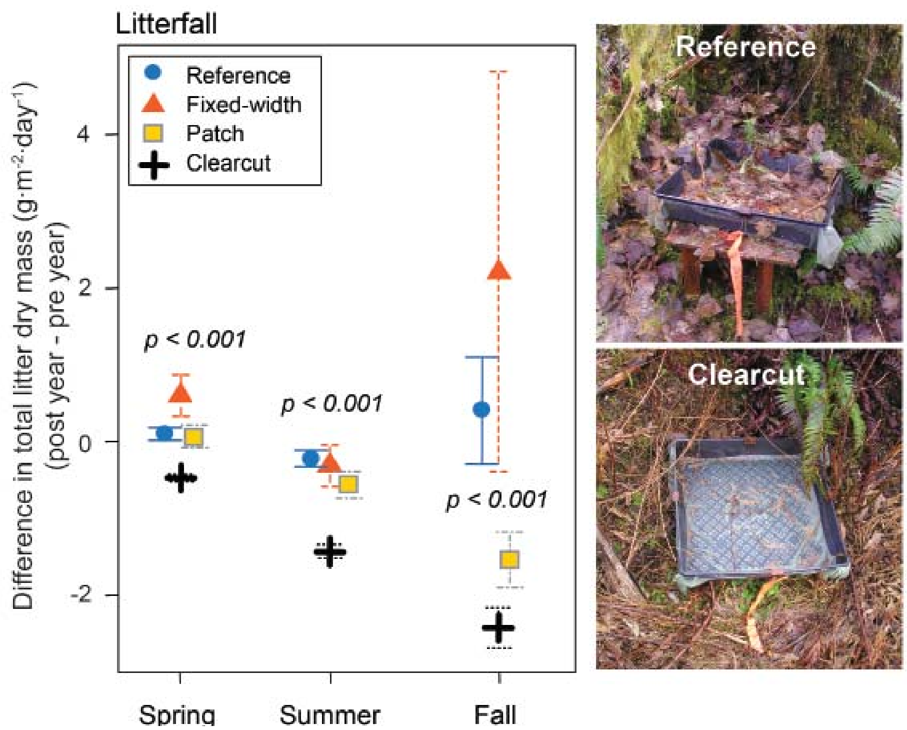


Figure 11—Changes in mean riparian litterfall, by season, after treatment application compared to input rates before treatment (± 1 standard error). (Photos by Shannon Claeson and Alex Foster.)

catchments selected for no buffer treatments happened to be less suitable for riparian mollusks by chance. As with aquatic invertebrates, the greatest divergence in mollusk abundance among treatment and reference locations occurred in fall.

Riparian Litter Inputs

Litterfall was sampled at six of the eight clusters, three in Capitol Forest and three in Willapa Hills, with an array of 40 x 40 x 10-cm traps spaced along the headwater channels. We placed litter traps 0 m, 10 m, and 20 m from the stream's edge on both sides of the channel. Monthly or bimonthly samples were collected from spring through fall (generally April through November). In the laboratory, samples were dried, sorted into various categories (coniferous litter, deciduous litter including seeds, wood, lichens, and other material), and weighed. All dry weights were divided by the sampling interval and converted to $g{\cdot}m^{-2}{\cdot}day^{-1}$. Deciduous litter inputs were greatest in fall (October–December), whereas coniferous litter inputs dominated in summer (July–September). The wood component of litterfall was variable throughout the year, and lichen and other components were generally a small fraction of total inputs.

Riparian litterfall in clearcut catchments significantly decreased to very low levels after treatment application (fig. 11), although inputs were beginning to recover in the second post-treatment year. Litterfall also declined significantly along patch-buffered catchments, but only in the fall (fig. 11). Fixed-width and patch-buffer litter inputs were highly variable post-logging, in part due to differences in windthrow among sites. No differences in litter amounts were observed at increasing lateral distances from the stream (0 m, 10 m, or 20 m).

Preliminary Ecological Conclusions

Preliminary findings suggest that biophysical changes in the headwater streams and riparian zones were detectable, but not large, after logging. Biological differences were most apparent in clearcut sites, especially in fall. An overall response to treatments was detectable, although the signal was not strong. At no streams were dramatic changes observed in the biophysical response metrics we monitored after treatment. Temperature changes were relatively minor and, by themselves, did not likely have a major impact on the biota of the headwater stream communities. Changes in the composition of aquatic invertebrate communities were observed, but we did not see major reductions in macroinvertebrate diversity or feeding guild structure after logging. It was also apparent that changes were most discernible at catchments where no buffers were left along headwater streams.

Differences between continuous fixed-width buffers, patch buffers, and reference catchments were much more difficult to detect. Because of the variability in the data and the small number of replicates, it was unclear whether these buffers provided some form of protection or whether we simply did not have the statistical power to detect differences between buffered and reference streams. For aquatic invertebrates, riparian mollusks, and terrestrial litter, differences among treatments were most apparent in fall. Concern often focuses on conditions in mid-summer, when streams are warmest, but our research suggests that monitoring in fall may be a more sensitive time to look for management impacts.

Watershed-Scale Experiments: Lessons Learned

In addition to the scientific results, we learned much about conducting experiments involving operational forest harvest at the scale of whole catchments as well as designing and implementing large-scale management research. These lessons may be useful to others considering similar types of studies.

Watershed-scale manipulations are interventions, designed to maximize what we can learn from them. They are not classical experiments. Due to numerous implementation problems, we found it impossible to achieve a complete and balanced experimental design with all treatments applied to all watershed clusters. We believe that an ideal, balanced BACI statistical design with precisely replicated treatments and controls may be impractical to achieve in a broad landscape setting. This is not to imply that careful attention should not be given to setting up a study where management actions can be evaluated, but rather to accept that implementing such studies with normal operating constraints in disturbance-prone environments may well prove impossible.

One of the major differences between a classical experiment and watershed-scale ecological experiment is the interpretation of control sites. The classical definition of a control would be a replicate that is identical to the treated replicated except that it does not receive the treatment. In lab studies or greenhouse experiments, the ability to impose control is an essential part of statistical analysis. However, at watershed scales, neither identical replicates nor rigid environmental control are feasible. Researchers must adapt the strengths of designed experiments to the realities of reference reaches or reference streams.

Logistical constraints can prevent true random allocation of experimental treatments, and therefore "control" sites are different from treated sites before treatments are administered. Even though reference and treated units in the REMS study were in close proximity, there were differences between catchments that presented analytical challenges. Where control or reference streams do not follow a similar trend pre-treatment, the use of control sites in a BACI design may actually reduce statistical power (Roni et al. 2005). In the REMS study, natural differences between reference and treated sites made it difficult to detect results that could be directly attributed to management actions. Based on our experience, it seems prudent to select reference locations as close as possible and as similar as possible to experimental treatment areas, and even then to be prepared for unanticipated variation among locations.

Natural variability can make it difficult to detect responses to experimental treatments, especially with the small sample sizes that are common in large-scale interventions. We were surprised that background levels of variability were so high, even in the absence of management actions. Temporal variability makes detection of ecological change difficult across a wide range of ecological phenomena. Temporal variability decreases statistical power, making it difficult to detect significant treatment effects, and thereby necessitating either a larger sample size or a longer measurement window, or both. Roni et al. (2003), for example, re-analyzed data from four streams in coastal Oregon and concluded that it might take more than 70 years to detect a doubling of Coho Salmon (*Oncorhynchus kisutch*) smolt production in response to habitat restoration, even using a BACI design. Korman and Higgins (1997) found a similar result. Natural variability makes detecting effects of management very difficult. This poses a particular problem for watershed-scale studies that are limited with respect to sample size. Having a large sample size and randomizing treatments across study units can improve statistical power and reduce the potential for bias.

Had we performed statistical power analyses to determine the time or sample size needed to evaluate the null hypothesis that no differences were attributable to buffer treatments, we would likely have had to either greatly expand the number of study sites or extend both the pre- and post-treatment monitoring intervals to a decade or more, and the post-logging forest would be regenerating anyway. The importance of long-term monitoring to the evaluation of environmental change has been a cornerstone of the U.S. Long Term Ecological Research network (*http://www.lternet.edu/*). This program has undoubtedly contributed to our understanding of watershed management. However, in an environment where decision-makers do not have the luxury of multi-decade time horizons for evaluating policy choices, we were essentially constrained to carry out the study in less time than was necessary to properly implement a BACI study that could accommodate natural variation in headwater areas. For similar reasons, increasing the study's sample size was logistically and financially impossible. Natural variability is a general problem in designing ecological experiments. A common tendency among scientists is to more intensively monitor a small set of study sites for a longer and longer period of time in order to truly understand patterns at a particular site; this is exemplified by the Alsea Watershed Study in Oregon and the Carnation Creek Study in British Columbia. However, significant statistical gains can be made by simply increasing the sample size, allowing the larger number of observations to absorb the natural variability so that trends across space and time can be quantified (Liermann and Roni 2008).

We had difficulty determining whether observed changes in response variables were related to buffer treatments or to long-term changes that had nothing to do with the experimental design. The use of a BACI design with reference watersheds is intended to help control for temporal variation but, when all treatments are not administered in the same year, it is impossible to completely separate the year and treatment effect. Further, if the pre-treatment period does not include conditions experienced during the post-treatment phase, the assumption of stationarity is violated and the strength of the BACI design is weakened. This was the situation in the REMS study. Staircase designs (Walters et al. 1988), in which treatments are implemented over a series of years in stepwise fashion, are intended to separate the effects of environmental variation over time from treatment effects, and to control for time-treatment interactions. These designs are also more complicated and expensive to carry out at large spatial scales. The specific questions being asked (e.g., How do buffers influence the export of invertebrates to larger streams?) will determine the most appropriate study design, given natural variability.

Experiments over large spatial scales are often confounded by unanticipated events. During the study we encountered both natural and anthropogenic events that contributed to the difficulty of detecting treatment effects. The large wind storms that struck the Willapa Hills catchments in the first two post-treatment years caused so much windthrow in some of the fixed-width and patch-buffer sites (fig. 4) that these treatments departed significantly from the original experimental design. In addition, heavy sediment inputs that accompanied the storms influenced many response variables, adding to already high levels of background variation. The landslide and debris flow that occurred in one of the Capitol Forest catchments in 2007 essentially terminated the study at that site. Finally, we did not anticipate that some of the catchments would be treated with herbicide as part of the routine site-preparation process. None of these events were factored into the original study design; however, in retrospect we believe that such surprises are not uncommon in multi-year, watershed-scale studies and may need to be accepted as an inevitable part of the research.

Scientists and managers need to work together to design, conduct, and interpret, results of large-scale studies. Scientists and managers should work together to establish realistic expectations for what can be done to maximize learning opportunities where rigid control of factors other than the variables of interest cannot be achieved. For managers, this may mean forgoing some operational flexibility to maintain as much treatment consistency as possible across study locations. For scientists, this may mean having to make some concessions in the types of treatments and the location and timing at which they are applied. Both managers and scientists should also collaborate to establish realistic expectations of results relative to the questions being asked. For example, we were naïve to think that we could fully assess the issue of headwater buffer efficacy in a 5-year study. Post-treatment changes were still occurring when the study ended, and thus our work should be seen as an examination of the short-term responses of experimental catchments in southwestern Washington to different buffer configurations.

Implement the design and stick with it until you resolve the important question(s) or until the data show that actual uncontrolled variation is so different from what was assumed during the planning phase that the design is not adequate to resolve the question. It is essential not to alter treatments in the middle of a study, even if the temptation to do so arises. In our study there was interest in salvaging trees that were blown down in the post-treatment storms. We were able to convince the managers that the additional disturbances created by salvage logging operations would only make it more difficult to evaluate post-treatment monitoring results at the affected locations. It is also important to recognize that when experimental results are simply so variable that there is no hope of answering the original questions, there may be no compelling reason to continue the study. It may take several years (or one exceptionally large disturbance) to reach this conclusion, but there is little to be gained by continuing research that cannot lead to new insights, even if treatment consistency is maintained.

Do not be afraid to investigate novel response metrics. When surprises occur, be flexible enough to monitor their effects to maximize learning opportunities. It may become apparent after a study has been initiated that adding a new metric to the suite of response variables in the monitoring plan can yield important information. Even if the metric or method is relatively untested in the context of the study questions, the benefits of incorporating something novel with the potential to shed new light on ecological processes that control system response may outweigh the risks of ignoring it. The metric may not provide useful information, and after a reasonable trial period it can be dropped. The new metric might not help answer the original questions, but instead may contribute information of different value. For example, we collected riparian millipedes along with mollusks because we found many millipedes in the artificial substrate samples. In the process, we captured several taxa that were previously unknown to science (Foster and Claeson 2011). Finally, surprise occurrences such as extreme weather events or unexpected management activities can add to the difficulty of evaluating experimental hypotheses, but they can also provide unique opportunities for learning in a setting where a monitoring program is already in place. For example, this study could have provided a useful design for addressing the question, "Are different buffer configurations prone to windthrow at different rates?" Taking advantage of these rare opportunities by investigating the effects of the event may add to the overall utility of the research.

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Applying Four Principles of Headwater System Aquatic Biology to Forest Management

Robert J. Danehy and Sherri L. Johnson

Abstract

Headwater systems, including the channel and the adjacent riparian forest, are a dominant landscape feature in forested watersheds, draining most of the watershed area, and comprising the majority of channel length in drainage networks. Being at the upper extent of watersheds, these systems are smaller and steeper than large streams, and create microhabitats that support diverse instream communities distinct from those in larger streams. Forest management can disturb headwater streams through changes in physical structure, sediment, light, and riparian detrital inputs. Locally, the extent of the buffer surrounding the system mediates disturbance intensity and responses. At broader scales, the effect of the shifting mosaic of stand ages across a landscape is less well-described. In addition, as watersheds are periodically harvested, long-term impacts of repeated canopy removal are unclear. We synthesize recent research from western Oregon and Washington focused on forest effects on headwater stream ecology. We draw on over twenty published manuscripts from Pacific Northwest (PNW) research, as well as seminal work from beyond the PNW. Findings of these studies are examined in light of four principles for managing forest near streams: *system* (the stream and riparian area as a system); *flow* (sources and duration); *disturbance* (types and frequency); and *topography* (elevation, gradient, and aspect). The interaction between local influences and landscape drivers varies among headwater systems. Nevertheless, the underlying influences on headwater system ecology are repeatedly demonstrated in recent work. Consideration of these principles in planning ongoing forest management activities can promote the stewardship of headwater systems.

Keywords: Headwater, headwater systems, flow regime, disturbance, landscape, biodiversity.

Introduction

Research focused on the aquatic biology of headwater streams has increased as ecologists recognize the importance of these ecosystems (Lowe and Likens 2005). The research stems from an interest in better understanding linkages between forest and streams, as well as headwaters and downstream processes and functions. Recent policy questions and concerns about the role of headwater systems within stream networks (Adams 2007; Leibowitz et al. 2008) have also increased interest in understanding these ecosystems.

There is also a growing recognition that headwater streams are critical components of ecosystems and landscapes (Gomi et al. 2002; Wipfli et al. 2007). They comprise the majority of stream length and drain the majority of watershed area (Leopold 1964). Given their small areal extent taken individually, the stream

Robert J. Danehy is forest watershed program manager at National Council for Air and Stream Improvement, 720 SE 4th St. Corvallis, OR 97333; bdanehy@ncasi.org; **Sherri L. Johnson** is a research ecologist for the USDA Forest Service. Pacific Northwest Research Station, Forestry Sciences Laboratory, 3200 SW Jefferson Way, Corvallis, OR 97331; sherrijohnson@fs.fed.us

and associated riparian areas which comprise the headwater system can blend into the dominant vegetation of a landscape, making them less obvious. Nonetheless, the cumulative extent of their influence, together with a clearer appreciation of the role of these systems across larger spatial scales, is increasingly acknowledged (Meyer and Wallace 2001; Richardson and Danehy 2007).

Given that small streams drain watersheds and therefore link landscapes, how they are managed has important consequences. Best management practices (BMPs) for forest management vary according to ownership. In Oregon, these range from wide buffers on federal lands, to smaller buffers on state lands, to all merchantable timber allowed to be harvested near small, non-fishbearing streams on private lands (Adams 2007). These differences in riparian management are linked to the objectives of the landowner and the goals of the BMPs. Additionally, in the PNW, headwater system BMPs were developed with consideration for their potential impact on water quality and fish populations downstream (Adams 2007). For federal buffers, the default is two site-potential trees (90 m) on each side of fishbearing streams, and narrower buffers on non-fishbearing streams (30–45 m): these can be modified with adequate justification. On private forests, BMPs are less conservative, taking into consideration the socioeconomic objective that sufficient harvestable land is available to support the economic needs of logging businesses, and thereby local communities (Adams 2007).

A major difference in riparian protection for headwater streams between federal and private lands stems from Endangered Species Act (ESA) requirements. The federal forests in the region are required to be managed in ways that directly promote recovery of ESA-listed species, including conservative assumptions about riparian conditions to protect fish and wildlife habitats (Reeves et al. 2006). Although take of anadromous fish or wildlife listed under ESA is to be avoided on private lands, riparian protection requirements on those lands are primarily intended to meet water quality criteria under the Clean Water Act (Adams 2007).

Definitions of headwater systems vary, although the upstream end of fish distribution as the lower bound is a convenient construct (Fransen et al. 2006; Moore and Bull 2004). As streams shrink to a size, a flow, or both that limits fish habitation, there are often associated changes in biotic communities (Olson and Weaver 2007). Community composition and ecological processes in these upper reaches are therefore driven by diverse interactions, both biotic (i.e., lack of fish) and abiotic (small and steep, with less stream flow) (Richardson and Danehy 2007).

Vertebrates other than fish use headwater systems, but few are fully aquatic. There can be robust amphibian populations with communities of one to as many as twelve species in western Oregon streams (Olson and Weaver 2007). The stream is a source of water and foraging habitats for most terrestrial wildlife, creating areas that are biologically rich and active (Anthony et al. 1987, 1996; Meyer et al. 2007; Moldenke and ver Linden 2007; Richardson et al. 2005a; Rykken et al. 2007a; Wilk et al. 2010). Within the aquatic portion of the headwater system, there are rich aquatic communities of macroinvertebrates and periphyton (Danehy et al. 2007; Dieterich and Anderson 2000; Herlihy et al. 2005). The composition of those communities and the abundance of taxa within those communities are determined by a suite of factors that we summarize as four principles.

We review research on headwater streams primarily from forested landscapes west of the Cascade divide in the Pacific Northwest, but include selected relevant studies from small fishbearing streams, even though these are not generally considered headwaters. We group these studies by four principles of headwater system aquatic biology to assist land managers in their understanding of the implications of management and types of impacts one might expect.

Four Principles

1. System—Riparian conditions matter. Forest-stream interactions are dominated by riparian conditions. Tight aquatic-terrestrial interactions occur for small streams. Generally, riparian detrital inputs (allochthonous) greatly exceed production from instream plants and primary producers (autochthonous production) (fig. 1).

The river continuum concept (Vannote et al. 1980) describes the types of interactions between instream aquatic biology and riparian influences as changes occur with stream size. Important drivers of aquatic biology are direct insolation and allochthonous inputs that respectively increase and decrease with stream size. Much prior research and policy focused on wadeable streams with fish, but more recent efforts have highlighted the ecology of headwater systems. Richardson and Danehy (2007), in a synthesis of the underlying factors of the ecology of headwater systems, describe how small size creates a tight coupling with the surrounding environment, how the lack of fish can change the structure of faunal communities, and how periodic low flows are common in small streams.

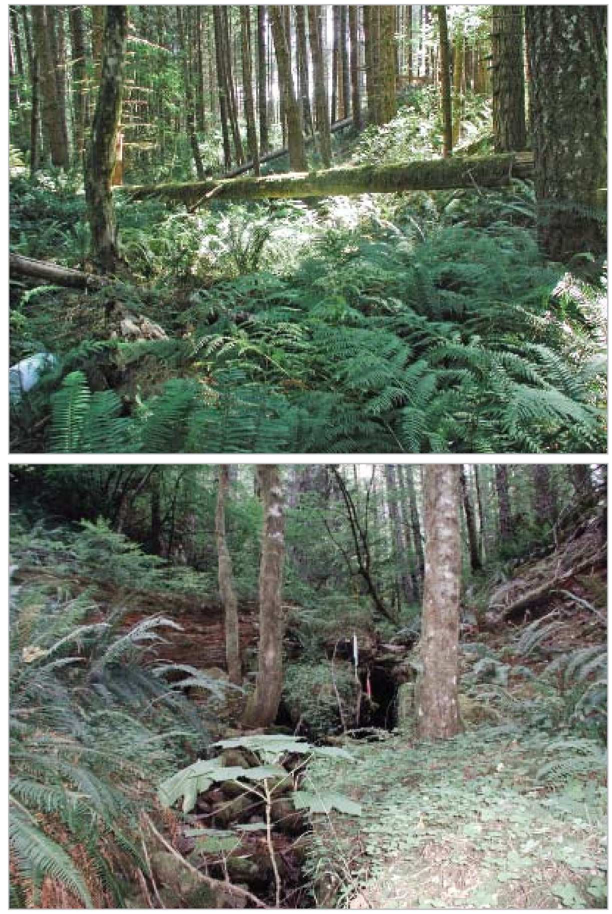

Figure 1—Examples of headwater systems in the Oregon Coast Range. Riparian vegetation is tightly woven to the stream, with the stream channel fully covered in upper image. (Photos by R. Danehy.)

The links between the flowing stream and the surrounding riparian forest are influenced by the condition of the riparian forests. Concerns over timber harvest up to the edge of the streams have led to buffer requirements across the PNW, although the size and efficacy of those buffers have been an ongoing debate (Adams 2007; Dent et al. 2008; Groom et al. 2011). Whereas fishbearing streams are all buffered to a range of requirements, and the smallest streams in some jurisdictions have small buffers, on private lands in Oregon, forest practices allow the removal of all merchantable timber along fishless streams. This typically results in a riparian area dominated by shrubs and hardwood trees such as alder (*Alnus* spp.). Consequently, headwater systems have the potential to be altered more dramatically than downstream systems. These impacts have been investigated by multiple researchers and are discussed in a recent body of literature.

More than thirty years ago, Murphy and Hall (1981) and Hawkins et al. (1982) examined effects of clearcut logging on stream communities in the Cascades, and found a suite of changes including increases in algae, aquatic insects, and salamander and fish biomass where the surrounding area has been clearcut. Those studies have more recently been corroborated by a set of studies in both the Coast Range and Cascades ecoregions of Oregon. A series of research efforts published over the last decade addresses timber harvest impacts on headwater systems (Banks et al. 2007; Cole et al. 2003; Danehy et al. 2007; Frady et al. 2007; Herlihy et al. 2005; Moldenke and ver Linden 2007; Romero et al. 2005; Rykken et al. 2007a). The research provides us with substantial information concerning the influence of changes in riparian forests on headwater biota, particularly invertebrates. While targeting forest management, those efforts also contribute to our broader understanding of headwater systems.

The results of the recent research provide substantial information on responses of headwater stream biota and their habitats following forest management and changes in riparian condition. After logging near headwaters, many researchers have noted major changes in instream physical features (Johnson and Jones 2000; Moore and Wondzell 2005). Total macroinvertebrate density and shifts in composition have been strongly correlated with the instream changes (Cole et al. 2003; Banks et al. 2007; Danehy et al. 2007). Banks et al. (2007) found that abundance and richness of emerging aquatic insects are slightly higher in clearcut reaches. Moldenke and ver Linden (2007) found more biomass and density of the insect orders Ephemeroptera, Plecoptera, and Trichoptera (EPTs) and all feeding groups except scrapers at clearcut sites. Danehy et al. (2007) examined macroinvertebrates in clear-cut, thinned, and forested stands. They found no detectable differences between thinned and forested. However, clearcut sites had higher macroinvertebrate abundance, more Chironomidae taxa, and higher biomass. Over time and with riparian regeneration, these differences were less apparent. Frady et al. (2007) compared streams through second-growth forests and old-growth sites and found no clear differences in macroinvertebrate assemblages between treatments. However, where alder had re-established in the riparian areas, they collected higher densities of shredders than in conifer-dominated reaches.

Herlihy et al. (2005) found that substrate condition and topographic features are the most important indicators of aquatic insect community composition across three western Oregon ecoregions. However, when viewed at larger spatial scales and across ecoregions, they concluded that logging was not a major driver in shaping western Oregon headwater macroinvertebrate communities; rather, the greatest variation in forested stream communities was due to substrate size, stream gradient, and altitude. Moldenke and ver Linden (2007) also found that elevation was a stronger determinant than riparian condition for insect communities.

In addition to research conducted on the role buffers have on forested headwater biota, there has been work that evaluates the links between the stream and riparian areas. This research reinforces the importance of viewing them as ecosystems with frequent exchange of energy. One area of research that was particularly insightful is the work that documents reciprocal subsidies (Nakano et al. 1999; Baxter et al. 2005) from the stream to the riparian areas. It had long been recognized that allochthonous inputs from areas outside the stream are important to instream biology. Moving upstream the ratio of allochthonous to autochthonous contribution of energy changes, which is a central tenet of the river continuum concept (Vannote et al. 1980). Starting in the 1990s, the role of marine-derived nutrients from salmon was documented by Cederholm et al. (1999) and Bilby et al. (2001); these insights reinforced the importance of salmon to forested ecosystems, particularly in the Pacific Northwest. At about the same time, in Japan, Nakano et al. (1999) and Nakano and Murakami (2001) developed a comprehensive understanding of reciprocal subsidies from the stream to the riparian forest. That research documented the importance of insects emerging from the stream for the diets of a suite of riparian predators, particularly birds and bats, and has been continued by Baxter et al. (2005). In a third-order stream, Farrand (2004) also observed that insect emergence was a major subsidy of energy to the riparian area, with only a small portion of adult aquatic insects returning to the stream. Gomi et al. (2002) and Moldenke and ver Linden (2007) documented that in fishless streams, invertebrates were a vector of energy leaving the stream and subsidizing the riparian zone. This science of reciprocal subsidies documents a cycle of energy flow from outside the stream into the stream and from within the stream to adjacent terrestrial ecosystems.

In terms of regulatory guidelines, headwater systems have been valued for their importance in supporting fish habitat downstream rather than as important biological communities unto themselves. The connection between upstream and downstream systems includes the export of heat, food in the form of drifting invertebrates, organic material, substrates, and large wood (Alexander et al. 2007; Binkley et al. 2010; Danehy et al. 2011a; Gomi et al. 2002; Hassan et al. 2005; Moore et al. 2005a; Piccolo and Wipfli 2002; Richardson 1992; Richardson et al. 2005b; Wipfli and Gregovich 2002). Those subsidies were in the form of cold, clean water to create appropriate thermal habitat for fish (Danehy et al. 2005; Dent et al. 2008; Johnson 2004), organic matter that is processed and used by fish prey (invertebrates) (Kiffney and Richardson 2010; Richardson et al. 2005a; Wipfli and Gregovich 2002), and nutrients to fuel autochthonous production (Compton et al. 2003; Ashkenas et al. 2004).

2. Flow—Presence and duration of flow are a primary driver of aquatic biota composition and abundance. In addition, non-flowing wet channels or those with hyporheic flows provide important habitats for amphibians and some insects. Springs can support aquatic communities close to ridgelines (fig. 2).

The duration of flow is a critical feature in headwater systems, as these streams may be perennial, spatially intermittent, seasonally intermittent, or ephemeral. Headwater biota often have adaptations to annual, pervasive low flow events. The adaptations to low flow vary with organisms, as recently elucidated by Walters (2011). These traits of taxa (i.e., desiccation resistance, crawling rate, armoring, size at maturity, rheophily, and habit) (Poff et al. 2006) can help us understand why some taxa survive better than others.

The source and type of flow for a headwater stream varies widely. In a central Coast Range study, all the streams had a spring source which resulted in flows continuing into late summer (Danehy et al. 2007). The sources of these streams were on average 250 m from the ridgeline. Other researchers (for example, Banks et al. 2007) found that the upper portions of headwater sites were intermittent. Yet insects continued to emerge from the streambed even as the stream dried.

The duration and magnitude of flow have strong influences on the flora and fauna of the systems. Yamamuro (2009) observed different communities and indicator taxa in runoff-

Figure 2—Headwater streams start in the upper reaches of watersheds and increase in flow volume as channels join. Throughout the year, flow can be continuous, spatially intermittent, or seasonal. B. A headwater stream arises from a small spring. (Photos by R. Danehy.)

dominated streams versus spring-fed streams in the upper McKenzie River basin. *Yoraperla* spp., a Plecopteran which was found in all streams, showed shifts in timing of life-history events, with more cohorts present in the spring-fed streams. In other perennial spring-fed headwater channels, Danehy and Bilby (2009) found clear seasonal differences in both macroinvertebrate and periphyton assemblages. The macroinvertebrate assemblage in springtime was richer, more abundant, and had higher biomass. They also found that the periphyton community was dominated by diatoms, particularly *Achnanthes lanceolata*, with greater richness of diatoms at sites with higher base flow. These differences in flow conditions are manifested across landscapes and undoubtedly contribute to biological diversity of headwater systems and forests.

Flow intermittency may be particularly important in supporting biological diversity in headwater systems. Banks et al. (2007) found that community patterns vary by season and flow duration more than flow type, which may be due to subsurface flow being present in the intermittent channels. Others, including Dieterich and Anderson (2000), Delucchi (1989) and Delucchi and Peckarsky (1989) observed substantial differences in aquatic insect communities in perennial streams versus those that go dry in the summer. Dieterich and Anderson (2000) captured more species from temporary streams (>125) than from permanent streams (100). Progar and Moldenke (2002) also noted differences in composition between temporary or intermittent streams compared to perennial streams, and found higher densities and biomass of insects from temporary streams during periods of flow. During wet seasons, these intermittent streams can be important habitats and attract species more typical of larger streams. These results suggest that insect life cycles are synchronized with annual changes in flow, but are still susceptible to changes in the timing and duration of base flow. Given what we have learned about these communities at base flow, research suggests that headwater systems could be among the first ecosystems affected should climate change create drier conditions (Mote and Salathe 2010).

3. **Disturbance**—Various disturbance processes affect headwater systems, including forest harvest, debris flows, and fire. These infrequent events shape the channel structure and biotic assemblages of headwater systems (fig. 3).

Disturbance is a natural part of the functioning of headwater systems. A variety of disturbances influence streams, and many headwater species are well-adapted or quickly recover following these episodic events. Debris flows are a well-

Figure 3—Various types of disturbance periodically reshape headwater systems. A. A stream flowing through a burned eastside forest. B. This stream channel was scoured down to bedrock by a debris flow. (Photos by S. Johnson.)

recognized disturbance to Pacific Northwest streams, particularly after extreme rainfall (Turner et al. 2010; Robison et al. 1999). In the Coast Range, May and Gresswell (2003) estimated that the mean recurrence interval of debris flows was between 97 and 357 years. Snyder (2000) suggested that in portions of the Cascade Range, return intervals of debris flows could be more frequent. Debris flow occurrences can be associated with forest management, including roads and harvest (Snyder 2000; May 2002; Swanson and Dyrness 1975; Turner et al. 2010). Fire is a less-frequent disturbance type for western Oregon (Agee 1993; Cissel et al. 1999), but when severe fires do occur, they can fundamentally change the composition and trophic organization of headwater streams. The intensity of fire, ranging from small, localized ground fires to large, canopy-replacing fires, has differing consequences for forest-stream interactions. Similar to debris flows, anthropogenic influences involving fire suppression can alter fire regimes and severity (Franklin et al. 2002; Moser and Wade 2005). Despite the potential for local and larger-scale impacts and influence by anthropogenic actions, disturbances in Pacific Northwest ecosystems are natural processes that create habitat diversity.

We also know that after disturbances, stream community recovery begins rapidly; the magnitude and intensity of disturbance influences the period of recovery (Anderson 1992; Minshall 2003). In controlled-fire experiments in eastern Washington, Mellon et al. (2008), using a suite of sampling methods (benthic, emergent, and drift sampling), found higher macroinvertebrate densities but lower diversity at burned sites. Instream communities at burned sites were dominated by chironomids. In the Yellowstone fires of 1988, Minshall et al. (2003) found similar results. After ten years, differences between reference and burned sites were still evident in headwater streams, suggesting that fire impacts decrease with increasing stream size (Minshall et al. 2003).

There have been fewer studies comparing stream communities before and after debris flows. After a debris flow in the Cascades, Lamberti et al. (1991) found that fish recolonize within a year of the event and at densities similar to pre-disturbance levels. For macroinvertebrates, Anderson (1992) and Kobayashi et al. (2010) observed rapid recovery, but major shifts in functional feeding groups after disturbance. In a longer-term study, Danehy et al. (2011b) found community differences in both periphyton and macroinvertebrates eight years after the debris flows, and predicted that those differences would remain until a full riparian canopy develops (20 or more years). Many have suggested that debris flow disturbances are essential components of creation and maintenance of fish habitat downstream (Benda et al. 2005; Bigelow et al. 2007; Burnett and Miller 2007; Reeves et al. 1995, 2003). Wood recruitment to downstream fishbearing reaches by the transport of trees from headwaters to mainstem reaches during floods and debris flows creates habitats for fish and other biota (Reeves 2003).

Floods and drought also disturb headwater systems. Floods can rearrange instream habitat and affect biota as well as disturb adjacent riparian areas (Johnson et al. 2000). Although perceived to be a major risk, actual washout of organisms by high flows is often minimal. After a high-intensity flood in a fishbearing stream, Swanson et al. (1998) observed few changes to resident fish and amphibians. Droughts can also be extreme events that influence instream biota. Classifying droughts as a disturbance depends on the timing, duration, or magnitude (Richardson and Danehy 2007). Droughts need to be distinguished from annual low flows, because low flows and flow-duration lengths are integral components of headwater system ecology.

4. Topography—Topographic differences, (e.g., gradient, aspect, elevation) contribute to biotic diversity. Different taxa occur across elevational gradients. As streams steepen, substrate coarsens and channels narrow, altering available habitats for macroinvertebrates and vertebrates, and opportunities for algal colonization (fig. 4).

Topography of headwater streams and their location in the watershed influences habitat availability, species composition, and life histories (Vannote et al. 1980; Montgomery 1999). Topography and geomorphology also strongly shape riparian vegetative composition as well as water availability, disturbance regimes (see previous section), and stream temperature characteristics (Johnson 2004). Development rate of instream organisms is linked with temperature (Li et al. 2011; Ward and Stanford 1982), which correlates with elevation and aspect. Headwater systems, with all other variables being similar, can show differing insect community compositions across elevations. Frady et al. (2007) found 70 percent overlap and 30 percent difference in macroinvertebrate assemblages in old-growth streams of similar size but at different elevations. Moldenke and ver Linden (2007) suggested that differences in macroinvertebrate community composition due to elevation were more important than silvicultural treatments. In a study of 169 randomly selected sites across western Oregon, Herlihy et al. (2005) found that topographic features of catchment slope, channel slope, and elevation are correlates of multiple environmental variables and useful in predicting macroinvertebrate diversity.

Headwater stream geomorphology in the mountains of the Pacific Northwest is often dominated by step-riffle or step-pool morphology (Jackson et al. 2002; Benda et al. 2005; Danehy et al. 2007). The distance between steps increases as gradient decreases (Bryant et al. 2007). In highly complex systems, the steps are created by small wood or exposed boulders that influence flow paths and modify the rate of flow, particularly at low flows (Kasahara and Wondzell 2003). Danehy et al. (2007) found that time of travel ranges from 0.4 to 3.4 $m \cdot min^{-1}$ in Coast

Figure 4—Across landscapes, topography provides a range of elevations, aspects, and gradients, creating a range of conditions for aquatic and riparian flora and fauna. (Photos by R. Danehy.)

Range headwater streams. At higher flows, the retentive features also serve to reduce the export of sediment and large wood. Benda et al. (2004) hypothesized that headwater streams serve as reservoirs or storage areas and transport little via episodic transport. However, Gomi et al. (2005), in their review of research on suspended sediment, concluded that small streams could transport tons of sediment annually, particularly immediately after a disturbance such as road construction.

The zone of influence of a headwater stream can be identified by the changes in microclimate with distance from the stream. In Coast Range streams, two studies found that the zone of influence was narrow, with topography having a large influence on the climatic gradients (Anderson et al. 2007; Rykken et al. 2007b). In studies of buffer treatments, the strongest gradients have been observed to occur within the first 10 m away from the stream (Anderson et al. 2007). Headwater valleys are often narrow, allowing topographic shade to augment riparian-derived shading. Topographic shading and sun exposure differs in east-to-west versus north-south aspect streams (Anderson et al. 2007; Moore et al. 2005b). While seemingly obvious, these differences do affect air and stream temperatures and insolation and therefore headwater system biota.

Management Implications

Given the tight linkages between headwater streams and their surrounding forests, forest management has the potential to influence instream communities. A reduction in riparian buffers will lead to increased insolation, which can lead to more primary production, and possibly more food for consumers and predators. However, there are also potential trade-offs when managing or making a change to one portion of a complex ecosystem. Increased insolation raises water temperatures (Johnson 2004; Moore et al. 2005a), which can increase growth rates for some taxa but can also negatively influence other taxa and processes. Changes in species composition following forest management have been observed, as well as increased biomass and richness in algae, insects, and salamanders (Banks et al. 2007; Clapcott and Barmuta 2010; Danehy et al. 2007; Moldenke and ver Linden 2007). Evaluating trade-offs and selecting management strategies that benefit one group or taxon but negatively influence others is a challenging process for managers, researchers, and policy makers.

Headwater streams and adjacent riparian areas are generally viewed by ecologists as important habitats with unique species and valuable ecosystem functions. Others may look to headwater systems for the services they provide to downstream fish and humans, or view them from a broad landscape perspective with limited individual value. When managing headwaters, it is important to consider the trade-offs among differing management activities. We know that following disturbance that removes riparian forests, headwater ecosystem processes and aquatic biota will be affected for a few years until some shading returns. Changes in resource availability (energy, nutrients, detritus inputs) also occur with disturbances. The extent and severity of a management activity or disturbance across a landscape will influence diversity and recolonization rates.

The scientific consensus is mixed on how far downstream management effects are propagated. The recruitment of large wood to create and maintain fish habitat is an example of an important connection between headwaters and downstream (Reeves et al. 1995, 2003). The processes controlling other links between headwaters and downstream continue to be examined. There are a range of findings on upstream-downstream links, including how far potential food items drift (Danehy et al. 2011a; Wipfli et al. 2007), or sediment is transported (Gomi et al. 2005; Benda et al. 2005) or thermal impacts extend (Danehy et al. 2005; Johnson 2004; Moore et al. 2005b). There is a clear need for more comprehensive study of downstream responses to upstream disturbances. Rigorous monitoring as well as new watershed studies throughout Oregon and Washington will help develop our understanding of the mechanisms and trade-offs involved in forest management along headwater streams.

Summary

Headwater streams and riparian systems are ubiquitous and diverse landscape features. The natural diversity of communities in headwaters has been further altered by a legacy of prior forest management approaches coupled with natural disturbances. Each of the four principles—*system*, *flow*, *disturbance*, and *topography*—influences that diversity. Some research suggest that aquatic production in headwater streams increases after harvest for selected taxa, these results need to be viewed cautiously, as long-term and landscape-scale effects are not well understood. To date, there have been no long-term assessments or thorough systematic evaluations of effects of disturbances or forest management on full biological diversity or on responses across multiple trophic levels in PNW headwater systems.

However, we have learned much from previous research: 1. Riparian buffer designs affect the biology of headwater systems; 2. The nature of the flow regime is a major influence on the magnitude, distribution, and diversity of aquatic biology of a headwater system; 3. Disturbance history needs to be included in evaluations of possible future conditions; and 4. Location in a landscape matters and provides essential context—steep, perennial, high-elevation streams will have very different biological potential than systems that differ in those characteristics. These four principles provide a framework and context for responses to management activities around headwater streams.

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Citation:

Danehy, Robert J.; Johnson, Sherri L. 2013. Applying four principles of headwater system aquatic biology to forest management. In: Anderson, Paul D.; Ronnenberg, Kathryn L., eds. Density management for the 21st century: west side story. Gen. Tech. Rep. PNW-GTR-880. Portland, OR: U.S. Department of Agriculture, Forest Service, Pacific Northwest Research Station: 189–202.

Integrated Watershed Analysis: Adapting to Changing Times

Gordon H. Reeves

Presentation Abstract

Resource managers are increasingly required to conduct integrated analyses of aquatic and terrestrial ecosystems before undertaking any activities. There are a number of research studies on the impacts of management actions on these ecosystems, as well as a growing body of knowledge about ecological processes that affect them, particularly aquatic ecosystems, which are used to guide the analyses. Additionally, new tools are available to assist in the analyses. These developments have advanced the potential to conduct these analyses. However, there are some critical issues that need to be included in an integrated analysis that are not readily recognized or acknowledged at this time. Two of the more important factors are space and dynamics. Watershed and landscape analyses require consideration of a different set of rules for considering context and potential effects of proposed actions than do analyses that are conducted at smaller scales. The failure to recognize these issues can result in a misinterpretation of the analyses and a failure to understand potential impacts. Watershed analysis also focuses on identifying the location of, and accessing the magnitude of, various ecological processes in the watershed. Consideration of the dynamic nature of processes that affect aquatic ecosystems is relatively new, but the implications of these dynamics are not fully recognized by many managers and regulators. In consequence of both these factors, there is a mismatch between the expectations for management of aquatic and terrestrial systems and the way in which these systems actually function that limits our ability to develop and implement new approaches to integrated management.

Keywords: integrated ecosystem analysis, temporal and spatial scale, dynamic systems, management expectations

Editors' suggestion:

Reeves, G.H.; Williams, J.E.; Burnett, K.M.; Gallo, K. 2006. The Aquatic Conservation Strategy of the Northwest Forest Plan. Conservation Biology. 20(2): 319–329.

Gordon H. Reeves is a research fish biologist, USDA Forest Service, Pacific Northwest Research Station, Forestry Sciences Laboratory, 3200 SW Jefferson Way, Corvallis, OR 97331, greeves@fs.fed.us.

Intrinsic Potential: What Is It and What Is It Good For?

Kelly M. Burnett

Presentation Abstract

Landscape characteristics are commonly invoked to explain or predict salmonid abundances and stream habitat conditions. Such characteristics may be relatively static (e.g., hill slope, rock type, and drainage density) or change over time (e.g., forest cover, land use, and road density). Ultimately, how landscape characteristics affect a particular fish species may vary with the underlying capacity of a stream to provide high-quality habitat for that species, or in other words, the intrinsic potential of a stream. Intrinsic potential is derived from reach-scale stream attributes (gradient, stream size, and valley constraint) that influence availability of the fine-scale habitat features (e.g., pools, spawning gravel, and large wood) preferred by salmonids. Collecting data on these fine-scale habitat features in all streams throughout a region is impossible with current technology and budgets. However, intrinsic potential can be mapped over large areas using digital elevation and climate data to generate a high-resolution stream network and the necessary reach-scale stream attributes. The approach and models, though targeted for salmonids, can be adapted to any stream-dwelling species for which links to reach-scale stream attributes are known.

Intrinsic potential models have been evaluated using sensitivity analysis and against field data, and so model performance and reliability are relatively well understood. Sensitivity of intrinsic potential model outputs to model inputs can vary by geographic location, by the form of the model, and by species. For example, outputs of intrinsic potential models are most sensitive to gradient for Steelhead (anadromous *Oncorhynchus mykiss*) and Coho Salmon (*O. kisutch*), but to stream size for Chinook Salmon (*O. tshawytscha*). Because reach-level data on historical fish abundances are rare, model outputs at the reach level have been evaluated against: 1) current population abundance; 2) field and expert-opinion based maps on the current distribution of spawning adults; and 3) results from the Ecosystem Diagnosis and Treatment (EDT) model. However, model outputs at the basin scale have been evaluated against historical cannery records.

Models of intrinsic potential exist for several salmonid species and regions. These include Steelhead and Coho Salmon in western Oregon, Steelhead, Coho Salmon, and Chinook Salmon in northern and central California, and Chinook Salmon in the lower Columbia River. Intrinsic potential models can benefit, and have been used in, a variety of regional land-management and regulatory efforts, such as highlighting areas that may contribute disproportionately to aquatic conservation; prioritizing locations to improve fish passage or to restore streams from the legacy of splash dams; evaluating the distribution of potential fish habitat relative to land ownership, use, and cover; establishing how much high-quality habitat may be sufficient to support a "recovered" fish species; and estimating salmonid population abundances prior to European settlement. Intrinsic potential models differ from other fish-habitat suitability models in attempting to estimate the potential to provide high-quality habitat and not the actual condition of habitat. Thus, intrinsic potential models can open windows into the past and unlock possibilities for the future.

Keywords: fish habitat, salmon, streams, restoration, models.

Editors' suggestion:

Burnett, K.M.; Reeves, G.H.; Miller, D.J.; Clarke, S.; Vance-Borland, K.; Christiansen, K. 2007. Distribution of salmon-habitat potential relative to landscape characteristics and implications for conservation. Ecological Applications. 17(1): 66-80.

Kelly M. Burnett is a research fish biologist (emeritus), USDA Forest Service, Pacific Northwest Research Station, Forestry Sciences Laboratory, 3200 SW Jefferson Way, Corvallis, OR 97331; kmburnett@fs.fed.us.

Quantifying Fish Responses to Forestry—Lessons from the Trask Watershed Study

Jason Dunham, Douglas Bateman, David Hockman-Wert, Nathan Chelgren, and David Leer

Presentation Abstract

We describe demographic processes and species interactions that influence Coastal Cutthroat Trout (*Oncorhynchus clarkii clarkii*) in small streams that are part of an effort designed to evaluate forest harvest impacts in the Trask Watershed, an industrial forest located in northwest Oregon, USA. Spatial variation in recruitment, individual growth, survival, and movement were quantified during summer low flows for four years (2007–2010). The phenology of recruitment varied substantially among sites and years. Movement during summer was limited, and varied inconsistently among sites. Individual growth and survival showed consistent size-related patterns, with variability in growth showing more consistent differences among sites in different years. Processes driving these patterns are challenging to identify, but companion studies of instream cover selection and seasonal diets of trout and other fishes suggest strong roles for predators, species interactions, and seasonal food limitations. Based on these results, we find that a process-based understanding of forestry impacts may prove more useful than traditional trend-based monitoring and impact assessments.

Keywords: forest harvest, cutthroat trout, recruitment, cover selection, food availability.

Jason Dunham is an aquatic biologist, US Geological Survey, US Geological Survey, Forest and Rangeland Ecosystem Science Center, Corvallis, Oregon, jdunham@usgs.gov; **Douglas Bateman** is a senior faculty research assistant, Oregon State University, Corvallis, Oregon, Doug.Bateman@oregonstate.edu; **David Hockman-Wer**t is a biologist, dhockman-wert@usgs.gov; and **Nathan Chelgren** is an ecologist, US Geological Survey, Forest and Rangeland Ecosystem Science Center, Corvallis, Oregon, nchelgren@usgs.gov; and **David Leer** is a Faculty Research Assistant, Oregon State University, Corvallis, Oregon, David.Leer@oregonstate,edu.

Riparian Microclimate and Stream Temperature: Thinning and Buffer-Width Influences

Paul D. Anderson

Presentation Abstract

Thinning of 30- to 70-year-old Douglas-fir (*Pseudotsuga menziesii*) stands is a common silvicultural activity on federal forest lands in Washington and Oregon west of the Cascade Range crest. Decreases in forest cover lead to alterations of site energy balances resulting in changes to understory and stream channel microclimates. Uncut vegetative buffers are commonly used to mitigate upland harvest effects on aquatic and riparian habitats and functions. To create effective buffers, we need to better understand the relationships among thinning treatments, different riparian buffer widths, channel topography, and riparian and aquatic microclimates.

As a component of the Density Management and Riparian Buffer Study (DMS) we investigated buffer width and thinning effects on riparian microclimates of headwater streams in western Oregon. Spatial variations in stand density, canopy cover, and microclimate were measured 2–4 years following harvest along transects extending from stream center upslope into thinned stands, patch openings, or unthinned stands, with riparian buffers ranging from ~5–150 m width. For treated stands, the summer mean daily maximum air and soil temperatures increased and mean daily minimum humidity decreased with distance from stream. Small headwater streams exerted a distinct cool, moist influence on riparian microclimate that extended about 10 m upslope from stream center. Thinning resulted in subtle changes in microclimate as mean air temperature maxima were ~1–4°C higher than in unthinned stands. With variable-width buffers 17 m or wider, daily maximum air temperature above stream center was less than 1°C greater, and daily minimum relative humidity was less than 5 percent lower than for unthinned stands. In contrast, air temperature was significantly warmer within patch openings (+6–9°C), and within buffers adjacent to patch openings (+3°C) than within unthinned stands. Variable-width buffers defined by the transition from riparian to upland vegetation or topographic slope breaks appear sufficient to mitigate the impacts of upslope thinning on the microclimate above headwater streams.

Intensive studies such as DMS, combined with broader-scale operational monitoring can provide much-needed empirical data and analyses to inform management and regulatory strategies to conserve important aquatic and riparian functions of headwater streams in managed forests west of the Cascades.

Keywords: riparian buffers, thinning, air temperature, streambed temperature, relative humidity, canopy cover, headwater streams, forest structure.

Editors' suggestion:

Anderson, P.D.; Larson, D.J.; Chan, S.S. 2007. Riparian buffer and density management influences on microclimate of young headwater forests of western Oregon. Forest Science. 53(2): 254–269.

Paul D. Anderson is a supervisory research forester, USDA Forest Service, Pacific Northwest Research Station, Forestry Sciences Laboratory, 3200 SW Jefferson Way, Corvallis, OR 97331, pdanderson@fs.fed.us

Stream Temperature Responses to Timber Harvest and Best Management Practices—Findings from the ODF RipStream Project

Jeremy D. Groom

Presentation Abstract

Studies over the past 40 years have established that riparian buffer retention along streams protects against stream temperature increase. This protection is neither universal nor complete; some buffered streams still warm, while other streams' temperatures remain stable. Oregon Department of Forestry developed riparian rules in the Forest Practices Act (FPA) to protect fish-bearing streams from temperature increases, but it did so with acknowledgement that its rules might be insufficient. It developed the Riparian and Stream Function project, otherwise known as RipStream, to validate the effectiveness of these rules. The FPA stipulates that for Small and Medium fish-bearing streams, no cutting is to occur within 6 m adjacent to the stream and limited entry may occur within 15 and 21 m, respectively. The Oregon State Forest Management Plan (FMP) builds upon the FPA but includes additional protections in order to meet multiple management objectives. State forest riparian protections include an 8-m no-cut zone, an 8- to 30-m zone to be managed for producing mature forest conditions, and limited harvest between 30 m and 52 m. Harvest on state forest can differ from private lands, as state forest stands may be subject to thinning instead of clearcutting. The RipStream project represents a joint effort between State and Private Forest divisions at Oregon Department of Forestry to quantify the effects of timber harvest on stream temperatures. The study includes 15 state forest and 18 private sites. Data on stream temperature, riparian vegetation, channel characteristics, and channel shading were collected at every site for two years pre-harvest and five years post-harvest. Each site included an upstream unharvested control reach and a treatment reach that was harvested after year two. All private sites were clearcut; seven out of eight state sites were thinned. By the second year post-harvest we found no change in maximum temperatures for state forests while private sites increased pre-harvest to post-harvest on average by 0.7°C, with an observed range of response from -0.9 to 2.5°C. State sites additionally demonstrated no difference in temperatures between clearcut and thinned treatments. The observed changes in stream temperature were most strongly correlated with shade levels measured before and after harvest. Treatment reach length, stream gradient, and changes in the upstream reach stream temperature were additionally useful in explaining treatment reach temperature change. Shade was best predicted by riparian basal area and tree height. These findings suggest that riparian protection measures that maintain higher shade, such as those followed in state forests, were more likely to maintain stream temperatures similar to control conditions.

Keywords: riparian buffer, stream temperature, mixed-effects, shade, thinning.

Editors' suggestion:

Groom, J.D., Dent, L., Madsen, L.J. 2011. Stream temperature change detection for state and private forests in the Oregon Coast Range. Water Resources Research. 47, W01501, doi:10.1029/2009WR009061

Jeremy D. Groom is the monitoring coordinator, Oregon Department of Forestry, State Forests Division, 2600 State St, Building D, Salem, OR 97310; jeremy.groom@state.or.us

Initial Riparian Down Wood Dynamics in Relation to Thinning and Buffer Width

Paul D. Anderson, Deanna H. Olson, and Adrian Ares

Presentation Abstract

Down wood plays many functional roles in aquatic and riparian ecosystems. Simplification of forest structure and low abundance of down wood in stream channels and riparian areas is a common legacy of historical management in headwater forests west of the Cascade Range in the US northwest. Contemporary management practices emphasize the implementation of vegetation buffers adjacent to streams, and on federal lands thinning has become a predominant form of timber harvest. The combined effects of thinning and riparian buffer width on the down wood dynamics in stream channels and riparian areas are being assessed in young, managed headwater forests of the Density Management Study of western Oregon.

The riparian buffer component of our study includes unthinned controls and four buffer configurations embedded within upslope thinning treatments, applied to 35- to 80-year-old stands at multiple locations in western Oregon. Buffer configurations include one- and two-site-potential-tree-height buffers and two less conservative configurations, streamside retention (minimum 20 foot [6 m] width) and variable width (minimum 50 foot [15.2 m] width), allowing for tree harvest within riparian areas. Our assessments include pre- and post-treatment analyses of down wood volumes and decay classes within stream channels, origin of instream down wood (distance from stream), and down wood cover along transects extending from streams into upslope forests.

Along transects extending from streams through buffers into upslope forest stands, pre-treatment mean percent cover of large down wood (>30 cm diameter) ranged from ~4–17 percent in buffers and from 5–10 percent in the upslope. Pre-treatment cover of small down wood (5–30 cm diameter) ranged somewhat less, ~4–13 percent, and also varied less than large wood both within and among treatment units. Post-treatment, 2–5 years following thinning, mean down wood cover in the buffers differed less among treatments and tended to be less variable within treatments. The decreased variation among buffer treatments was due primarily to a decrease in the maximum mean cover of small down wood in streamside retention buffers, and in variable-width buffers adjacent to patch openings.

In streams, thinning and buffer configuration only affected down wood (>10 cm diameter) volumes in decay classes 1 and 2 (fresh or slightly decomposed detritus). Down wood volume was greater in the streamside buffer treatment than in the control one year after thinning, and greater than in the control and variable-width buffer treatments 2 years after thinning.

We determined the source distance-from-stream of 2,323 down wood pieces in headwater stream channels ~10 years post-thinning along a total of 25 stream reaches at 4 study sites. Mean distance

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Paul D. Anderson is a supervisory research forester and **Deanna H. Olson** is a research ecologist, USDA Forest Service, Pacific Northwest Research Station, Forestry Sciences Laboratory, 3200 SW Jefferson Way, Corvallis, OR 97331; pdanderson@fs.fed.us; **Adrian Ares** is the director, Sustainable Agriculture and Natural Resource Management Collaborative Research Support Program, Virginia Tech, Blacksburg, VA 24060

from stream ranged 2.6 to 9.0 m, and maximum distance was 41 m. Most of this wood was in decay classes 3+4, indicative of older legacy deposition. Only 9 percent of 9,550 instream pieces observed were in decay classes 1+2, indicative of more recent recruitment.

Recent second thinning entries conducted 10–12 years following the initial harvest provide a new opportunity to spatially assess stream and riparian large wood dynamics in relation to direct inputs associated with harvest as well as the longer-term stability of the original buffers.

Keywords: riparian buffers, thinning, down wood, headwater streams, forest structure.

Editors' suggestion:

Anderson, P.D.; Meleason, M.A. 2009. Discerning responses of down wood and understory vegetation abundance to riparian buffer width and thinning treatments: an equivalence/inequivalence approach. Canadian Journal of Forest Research. 39: 2470–2485.

Thinning and Riparian Buffer Configuration Effects on Down Wood Abundance in Headwater Streams in Coniferous Forests

Adrian Ares, Deanna H. Olson, and Klaus J. Puettmann

Poster Abstract

Down wood is associated with the function, structure, and diversity of riparian systems. Considerable knowledge has been generated regarding down wood stocks and dynamics in temperate forests, but there are few studies on effects of silvicultural practices and riparian buffer design on down wood, particularly in headwater streams. We analyzed interactive effects of upland thinning and riparian buffer configuration on down wood volumes and decay classes at five managed forest sites in western Oregon. We tested the effects of two thinning treatments—an unthinned control and a moderate-density thinning with 200 trees ha^{-1}—and three riparian buffer widths: combined one and two site-potential tree heights (to 480 ft [146 m]); variable-width (50 ft [15.2 m] minimum); and streamside retention (~20 ft [6 m]), in coniferous stands dominated by *Pseudotsuga menziesii* and *Tsuga heterophylla*. Pre-harvest down wood volumes (59 to 128 $m^3 \cdot 100\ m^{-1}$ reach length) in 29 stream reaches (193 to 792 m long) did not differ among thinning treatments and stream buffer configurations. One year after thinning, down wood volumes were 5–42 percent greater than pre-harvest levels, although thinning and stream buffer effects remained non-significant. Large increases in down wood volumes with time in our headwater streams may be the consequence of severe storms or other events not directly related to silvicultural practices.

Keywords: down wood, stream habitat, timber harvest, decay class, riparian reserves.

Adrian Ares is the director, Sustainable Agriculture and Natural Resource Management Collaborative Research Support Program, Virginia Tech, Blacksburg, VA 24060; aresa@vt.edu; **Deanna H. Olson** is a research ecologist, USDA Forest Service, Pacific Northwest Research Station, Forestry Sciences Laboratory, 3200 SW Jefferson Way, Corvallis, OR 97331; **Klaus J. Puettmann** is Edmund Hayes Professor in Silviculture Alternatives, Department of Forest Ecosystems and Society, Oregon State University, Corvallis, OR 97331.

Slope Failure as an Upslope Source of Stream Wood

Daniel Miller

Presentation Abstract

Large woody debris is recognized as an important component of stream geomorphology and stream ecosystem function, and forest-land management is recognized as an important control on the quantity (and size and species distributions) of wood available for recruitment to streams. Much of the wood present in streams comes from adjacent forests, and riparian management practices now reflect our understanding of the role these forests play in modulating and maintaining stream environments. In steep terrain, slope failures also carry wood (and sediment) to streams from upslope source areas. In these environments, periodic inputs of wood and sediment from landslides and debris flows also play an important role in stream geomorphology and ecosystem dynamics.

Channel environments are naturally dynamic systems. Depending on where you are in the channel network, discharge can vary from none in the summer to bed-scouring, channel-avulsing floods in the winter. Slope failures also drive variability in this system. Deposition of wood and sediment occur at discrete points in time and space, thereby creating temporal and spatial variability in channel environments. Redistribution and decay of deposited materials over time further add to this variability, and act to hide the original source of these materials, thereby masking the role of landsliding in setting stream environments. Landslide effects thus depend on when and where you look, and can be difficult to discern if the landslide occurred some time ago. This makes efforts to anticipate the effects of landsliding very challenging. Are we interested in the short term? The long term? Are we interested in a single reach? Or effects over a basin? Observations also suggest that landslide effects depend on a host of factors, including valley geometry, channel geometry, the quantity and size of sediment and wood in the deposit, the amount of wood and sediment already in the channel, and the amount of wood and sediment that enter the channel over the lifespan of the landslide deposit.

This sets the stage for considering slope failure as an upslope source of stream wood, particularly if we are to consider in-stream wood in the context of a stream ecosystem. I will briefly review the evidence on which to base conceptual and empirical models for identifying and characterizing upslope landslide source areas, and for placing them into a channel-network context. Then I'll illustrate the data-analysis and modeling approaches that we and our collaborators have been experimenting with to identify upslope source areas for stream wood and for anticipating the in-stream consequences of management decisions in those areas. These methods span a range of complexity. At the most basic, we use digital elevation data coupled with empirical models to identify the source areas and runout tracks for landslides that could potentially carry material to specified portions of the channel network (e.g., fish-bearing streams). To gain insights to effects of management, we couple stand-growth, wood recruitment, and landscape dynamics models to estimate wood abundances over time and space.

Keywords: large woody debris, slope failure, stream wood, wood recruitment modeling.

Daniel Miller is a senior scientist, Earth Systems Institute, 6512 23rd Avenue NW, Suite 201, Seattle, WA 98117-5728; dan@earthsystems.net.

Plethodontid Salamander Population Ecology in Managed Forest Headwaters in the Oregon Coast Range

Matthew R. Kluber and Deanna H. Olson

Poster Abstract

We examined temporal and spatial patterns of terrestrial amphibian species abundances and individual movements in western Oregon managed headwater forest stands using artificial cover object (ACO) arrays. Using mark-recapture methods, we estimated the effects of species and seasonality on apparent survival rates and recapture probabilities. We captured, marked, and released over 300 individual salamanders during 18 site visits between 2006 and 2009. These captures were dominated by three plethodontid species: Ensatina (*Ensatina eschscholtzii*), Western Red-backed Salamander (*Plethodon vehiculum*), and Dunn's Salamander (*P. dunni*). We observed 64 animals move between ACOs at least once with most salamanders moving between adjacent ACOs (<5 m), and the maximum distance traveled being 31 m (Ensatina). Although total captures were evenly distributed between near-stream (<15 m from stream edge) and upslope arrays (>15 m), species distributions differed with distance from stream, and most movement occurred in the near-stream arrays (<15 m from stream edge). Using the program MARK, the annual apparent survival rate for our dominant species was 0.63; however, recapture probabilities varied among species and between site visits. Our results have implications for the efficacy of forest management approaches to address ground-dwelling species with limited dispersal capabilities in headwaters: riparian corridors are frequently used by both semi-aquatic and upland species. Stream-to-ridgeline dispersal occurs, yet is less frequent, although our ACO design likely was restricted in its utility for monitoring plethodontid salamander movements.

Keywords: dispersal, movements, riparian, survival, recapture probabilities.

Matthew R. Kluber was a master's student, Department of Forest Ecosystems and Society, Oregon State University, Corvallis, OR 97331, currently 1652 Pineland Avenue, Florence, SC 29501, matt.kluber@gmail.com; **Deanna H. Olson** is a research ecologist, USDA Forest Service, Pacific Northwest Research Station, Forestry Sciences Laboratory, 3200 SW Jefferson Way, Corvallis, OR 97331.

No Effects of Thinning With Riparian Buffers on Terrestrial Salamanders in Headwater Forests 5 to 6 Years Post-harvest in Western Oregon

Matthew R. Kluber, Deanna H. Olson, and Klaus J. Puettmann

Poster Abstract

There are emerging concerns for wildlife species associated with forested headwater systems. Given that headwater streams comprise a large portion of the length of flowing waterways in western Oregon forests, there is a need to better understand how forest management affects headwater forest taxa and their habitats. Forest management strategies that consist of only partial canopy removal and retention of riparian buffers may help ameliorate management effects on headwaters, especially relative to historical clearcutting practices. Our study investigated effects of upland forest thinning coupled with riparian buffer treatments on riparian and upland headwater forest amphibians, their ground-cover habitat attributes, and species-habitat associations. Amphibian captures and habitat variables were examined 5 to 6 years post-thinning, within riparian and upland forests thinned to 80 trees per acre with streamside-retention buffer widths (~20 ft [6m] wide, each side of streams) and variable-width buffers (50 ft [15.2 m] minimum width, each side of streams), as well as unthinned reference stands. Distance-from-stream was found to be associated with amphibian abundance. However, no treatment effects of thinning or buffer widths were found. We observed that ground surface conditions (e.g., amount of rocky or fine substrate, microclimates) likely played a role in determining the response of riparian and upland amphibians to forest thinning along headwater streams. Moderate thinning and preservation of conditions in riparian and nearby upland areas by way of variable-width and streamside-retention buffers may be sufficient to maintain suitable habitat and microclimatic conditions vital to amphibian assemblages in managed headwater forests.

Keywords: plethodontid salamanders, timber harvest, density management, riparian reserves.

Matthew R. Kluber was a master's student, Department of Forest Ecosystems and Society, Oregon State University, Corvallis, OR 97331, currently 1652 Pineland Avenue, Florence, SC 29501, matt.kluber@gmail.com; **Deanna H. Olson** is a research ecologist, USDA Forest Service, Pacific Northwest Research Station, Forestry Sciences Laboratory, 3200 SW Jefferson Way, Corvallis, OR 97331; **Klaus J. Puettmann** is Edmund Hayes Professor in Silviculture Alternatives, Department of Forest Ecosystems and Society, Oregon State University, Corvallis, OR 97331.

Log and Soil Temperature Profiles in Managed Headwater Sub-basins in the Oregon Coast Range: Implications for Wildlife Habitat

Matthew R. Kluber, Deanna H. Olson. and Klaus J. Puettmann

Poster Abstract

Down wood provides important faunal microhabitat in forests for many invertebrate taxa, small mammals, and amphibians. Habitat suitability of down wood as refugia is an increasing concern in managed forests of the US Pacific Northwest, where overstory reduction may result in both reduced down wood recruitment and increased temperatures within logs, which may make them unsuitable habitat refugia. We examined temperature regimes inside logs and soils to assess buffering capabilities against summer temperature extremes and potential habitat suitability for plethodontid salamanders. Temperature profiles of small- (0.3–0.45 m) and large- (0.7–1.0 m) diameter logs, as well as ambient soil and air temperatures, were measured in a 60-year-old forest stand under different slope positions (0–5 m and 35–40 m from stream edge) in three case studies: 1) along a headwater stream with a narrow riparian buffer (~6 m) and moderate upland thinning; 2) along a headwater stream with a wider riparian buffer (~15 m) and moderate upland thinning; and 3) along a headwater stream with an unthinned upland. Streamside and upslope maximum air temperatures experienced on all three streams during our study were near to or exceeded critical temperatures for western plethodontid salamanders (i.e., ~30°C). Streamside and upslope temperatures inside small logs, large logs, and soils stayed below critical temperatures. Our results suggest that logs of a wide size range as well as soils may provide sufficient protection against thermal extremes in uncut forests and thinned stands with limited overstory.

Keywords: Down wood, riparian, thinning, microclimate, salamander habitat.

Editors' suggestion:

Kluber, M.R.; Olson, D.H.; Puettmann, K.J. 2009. Thermal profiles of downed wood under different forest management regimes in the Oregon Coast Range and their potential impact on plethodontid salamander habitat. Northwest Science. 83: 25-34.

Matthew R. Kluber was a master's student, Department of Forest Ecosystems and Society, Oregon State University, Corvallis, OR 97331, currently 1652 Pineland Avenue, Florence, SC 29501, matt.kluber@gmail.com; **Deanna H. Olson** is a research ecologist, USDA Forest Service, Pacific Northwest Research Station, Forestry Sciences Laboratory, 3200 SW Jefferson Way, Corvallis, OR 97331; **Klaus J. Puettmann** is Edmund Hayes Professor in Silviculture Alternatives, Department of Forest Ecosystems and Society, Oregon State University, Corvallis, OR 97331.

Photos, facing page—Top: How do we keep understory regeneration standing through the 2nd thinning entry? Yachats STUDS site, Siuslaw National Forest, July 2009. **Center left:** Blowdown along a ridge, North Soup Creek DMS site March 2009. **Bottom left:** Ferns, understory, and underplanting, STUDS site, June 2007. **Lower right:** Future yarding corridors may need to be planned at the time of the first thinning to retain anchor trees. North Soup Creek, March 2009. All photos by Paul Anderson.

Section 4. Socioeconomics and Operations

Managers' Perspectives: Practical Experience and Challenges Associated with Variable-density Operations and Uneven-aged Management

Kurtis E. Steele

Presentation Abstract

Variable-density thinning has received a lot of public attention in recent years and has subsequently become standard language in most of the Willamette National Forest's timber management projects. Many techniques have been tried, with varying on-the-ground successes. To accomplish variable-density thinning, the McKenzie River Ranger District currently uses combinations of techniques such as skips, gaps, dominant tree releases, variable thinning prescriptions, designation by description (DxD), and individually marked trees within the same harvest unit. The major challenge associated with implementing variable-density thinning occurs during pre-sale. Depending on the level of variability within the harvest unit, it is estimated that it takes up to twice as long to complete a timber sale package from layout through the final contract. Issues for pre-sale resulting from variable-density thinning include: 1) more complex physical layout on the ground; 2) increased GPSing; 3) increase in cruising time; 4) increased complexity in the maps; and 5) increase in requirements of the contract.

In a recent project the McKenzie River Ranger District has also incorporated an uneven-aged management approach for Douglas-fir (*Pseudotsuga menziesii*) in many of the proposed harvest units. This has added yet another complicated element into the planning and implementation of the project. During the planning process, consideration of current and possible future logging systems as well as layout design must be analyzed early. For example, if a unit requires intermediate supports to cable log, more thought must be given to when and where the gaps are placed within the unit's rotation. If a group select (gap) is placed around the intermediate support trees and those support trees are damaged or removed by harvesting or wind throw, future cable logging opportunities may be compromised for 30–45 years. Locations of the group selects (gaps) may be critical to design prior to first implementation. The increased complexity of an uneven-aged management rotation and variable-density thinning across the landscape will also increase future data management costs. Units will need to be stratified further in stand exams and cruises to maintain quality data. Although the increased cost may not be an issue on a relatively small scale, when applied to a landscape, such as a whole project area, it may be more problematic, especially with the Forest Service's budget continuing to decline. Increased stratification will likely require more plots per acre, which in turn, requires an increase in person-hours. It is also recommended that locations of skips and gaps (group selects) should be tracked to allow for enhanced management in the future.

Keywords: thinning, skips, gaps, group select, variable-density thinning, implementation, dominant tree release, stratification, uneven-aged, designation by description, DxD.

Kurtis E. Steele is a forest silviculturist, formerly with the US Forest Service, Willamette National Forest, McKenzie Bridge, OR, now with the Chattahoochee-Oconee National Forest, 1199 Madison Road, Eatonton, GA 31024, ksteele@fs.fed.us.

Growth and Yield Considerations and Implications for Alternative Density Management Objectives and Approaches

David Marshall

Presentation Abstract

Density management through thinning is the most important tool foresters have to affect stand development and stand structure of existing stands. Reducing stand density by thinning increases the growing space and resource availability (e.g., light, water, and nutrients) for the remaining trees. This can result in increased average tree growth. More available site resources can also encourage the development of understory vegetation and trees. Alternatively, the highest amounts of stand volume (or tree biomass) growth will generally occur when the site is fully occupied. The competition levels of this high level of site occupancy will lead to self-thinning mortality of the less vigorous trees and reduced net stand growth. Reducing stand density reduces site occupancy, growing-stock, and total leaf area and generally results in reduced stand volume growth.

The impact of thinning on stand growth rates will depend on the amount of residual growing stock retained and the vigor of the trees that remain. Thinning younger stands with adequate crown and rapid height growth can be expected to build crown, leaf area, and growing stock rapidly. Vigorous trees in older stands may also respond to thinning. Higher average tree growth rates and lower mortality in thinned stands maintain rapid growth rates and flatten out the annual increment curve and push culmination of mean annual increment to older ages.

Growth-growing stock relationships have been studied in the Pacific Northwest since the 1960s and demonstrate the trade-off between individual tree growth and stand growth. Thinning to variable densities (growing space) and gaps will also follow these basic growth-growing stock principles. Understanding these relationships is important for foresters to achieve desired stand production, economic, and structural development goals.

Keywords: thinning, site resources, self-thinning mortality, individual tree growth, stand growth.

David Marshall is a biometrician for Weyerhaeuser NR Company, WTC 1A5, PO Box 9777, Federal Way, WA 98063-9777; david.marshall2@weyerhaeuser.com.

Harvest Operations for Density Management: Planning Requirements, Production, Costs, Stand Damage, and Recommendations

Loren D. Kellogg and Stephen J. Pilkerton

Presentation Abstract

Since the early 1990s, several studies have been undertaken to determine the planning requirements, productivity, costs, and residual stand damage of harvest operations in thinning treatments designed to promote development of complex forest structure in order to enhance ecological functioning and biological diversity. These studies include the Oregon State University-led cooperative Coastal Oregon Productivity Enhancement (COPE) project, the Willamette (National Forest) Young Stand Project (WYS), and the Siuslaw (National Forest) Thinning and Underplanting for Diversity (STUDS) project. Treatments focused on 35- to 50-year-old stands with density management targets included light thinning (100–120 residual trees per acre, tpa [~250-300 trees per hectare, tph]), heavy thinning (50–60 residual tpa [~125-150 tph]), very heavy thinning (30 residual tpa [~75 tph]), and light thinning with gaps. A second treatment was applied to the STUDS stands that had light and heavy treatments on the first entry.

Logging planning and layout costs were studied for the WYS project. Production and costs results for harvest operations include skyline yarding, ground-based skidding, and mechanized cut-to-length systems. Damage to the residual stand was studied for the STUDS stands.

Planning and layout efforts are beneficial to efficient and successful harvest operations. Harvest production influences related to the density management treatment were relatively small. More importantly, these studies have demonstrated important aspects of successful harvest operations for density management in planning, conducting, and assessing residual stand damage. Results of the stand damage assessments show high incidence of tree damage close to the yarding corridor / skid trail. A stand damage assessment methodology, DAMQUICK, was developed.

This presentation will (1) summarize what we have learned from the harvesting studies for successful planning and implementation of density management stand treatments, and (2) report the cable harvesting production and stand damage results from the STUDS second thinning entry.

Keywords: harvest planning, thinning costs, logging productivity, skyline yarding, tractor skidding, mechanized harvesting, cut-to-length, young stands, commercial thinning, partial cutting, timber harvesting, Douglas-fir, logging damage, riparian buffer.

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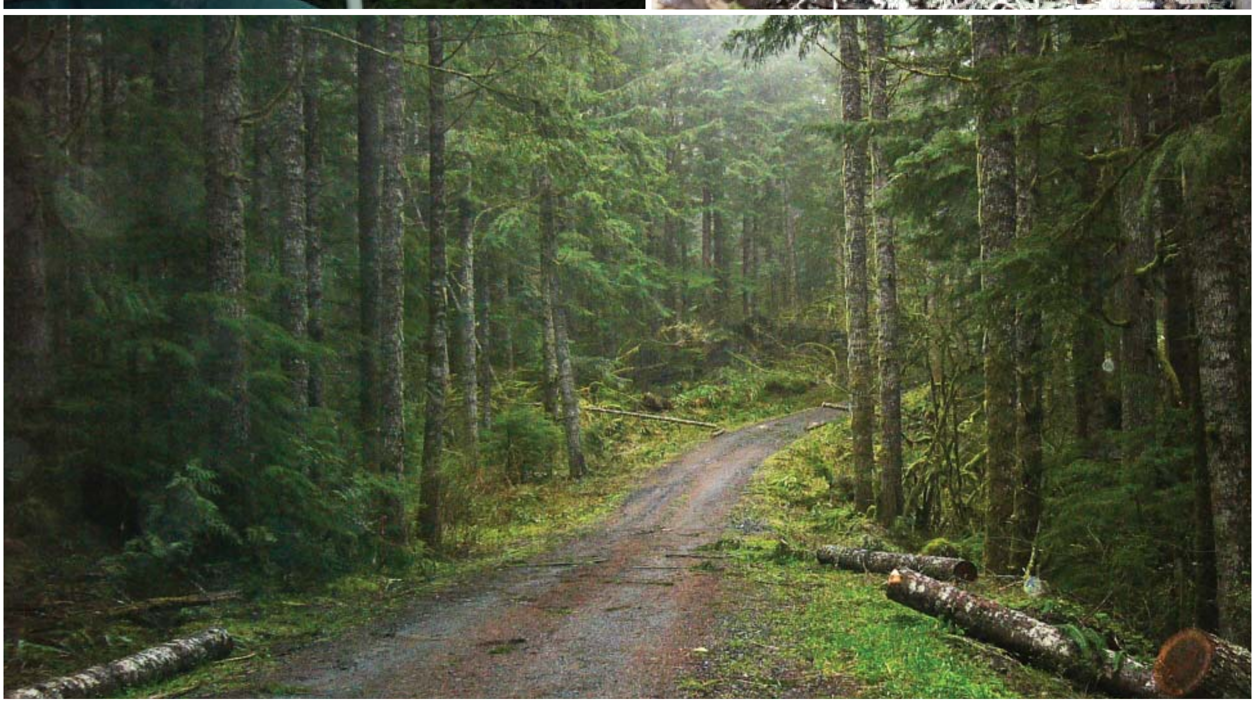
Loren G. Kellogg is a professor, and **Steven J. Pilkerton** is a research forest engineer, Department of Forest Engineering, Resources, & Management, Oregon State University, Corvallis, OR 97331; loren.kellogg@oregonstate.edu; steven.pilkerton@oregonstate.edu.

Photos, facing page—Top left: Anemone in the rain. Photo by Brad Catton, USFS. **Top right:** Coastal Giant Salamander, *Dicamptodon tenebrosus*. Photo by Loretta Ellenburg, USFS. **Center left:** Ensatina (*Ensatina eschscholtzii*). Photo by Brad Catton, USFS. **Center right:** Groundcone, *Boschniakia* sp. Photo by Brad Catton, USFS. **Bottom:** Road at Keel Mountain. USFS photo by Mark Meleason.


Section 5. Thinning and Adaptation





Geometry of Forest Landscape Connectivity: Pathways for Persistence

Deanna H. Olson and Kelly M. Burnett

Abstract

Streamside areas may be dispersal funnels or runways for a variety of species. For over-ridge dispersal, headwaters offer the shortest distance links among riparian zones in adjacent drainages. We summarize landscape designs for connectivity of habitats using headwater riparian linkage areas as the foundation for a web of landscape-scale links. We developed management considerations for placement of headwater linkage areas including: 1) providing connections between larger basins; 2) maintaining habitat connectivity in the face of climate change; 3) incorporating place-based disturbance regimes such as headwater debris-flow-prone areas; 4) targeting connectivity areas to address sensitive species conservation strongholds; and 5) accounting for geometry at the forest-stand scale of a single project or proposed timber sale, including managing habitats to connect lands on adjacent federal ownerships, by means of connecting corners of checkerboard landscape blocks along diagonals. Although our proposed linkage areas are designed to target headwater species, the resulting web of connections across the landscape is expected to benefit many forest-dependent species.

Keywords: watersheds, forest, headwaters, biodiversity, linkage areas, dispersal.

Introduction

Biodiversity retention and restoration is an emerging priority for global ecosystems. Astounding losses within major taxonomic groups have been reported nationally and internationally (41 percent of amphibians, 25 percent of mammals, 15 percent of bony fishes, 13 percent of birds: Hoffmann et al. 2010; 50–60 percent of turtles: Kiester and Olson 2011). In particular, protection and restoration of forests and forest biodiversity has become a paramount concern worldwide (e.g., Convention on Biological Diversity: *www.cbd.int/forest/*). A toolbox of management approaches has been developed to conserve forest biodiversity, largely through a mixture of fine- and coarse-grained habitat protections (e.g., United States Northwest Forest Plan: USDA and USDI 1993, 1994; Cissel et al. 1998; Lindenmayer and Franklin 2002; Raphael and Molina 2007; Lindenmayer et al. 2007) and site-specific designs to maintain or restore forest structural heterogeneity (McComb 2001; Lindenmayer and Franklin 2002; Brockerhoff et al. 2008).

Development of landscape designs to manage habitat connectivity for multiple species is an especially active research topic in forest biodiversity conservation, due to continuing trends of forest fragmentation and to an upswing

Deanna H. Olson is a research ecologist and **Kelly M. Burnett** is a research fish biologist (emeritus), USDA Forest Service, Pacific Northwest Research Station, Forestry Sciences Laboratory, 3200 SW Jefferson Way, Corvallis, OR 97331; dedeolson@fs.fed.us.

in world reforestation efforts. Managing forested landscapes for connectivity functions benefiting biodiversity requires incorporating several fundamental conservation concepts. These basic conservation tenets include identifying the critical habitats used throughout species' life histories (breeding, foraging, overwintering, and dispersal habitats), and commensurate habitat protections to ensure that these biotic functions are retained. If an organism uses different habitats through its life cycle, then maintaining connectivity among these habitats is essential to ensure its persistence. Of particular relevance is the characterization and retention or restoration of dispersal habitat. This includes the home ranges of individuals and the broader dispersal of offspring or individuals that tie sub-populations and populations together over larger areas. This broader-scale dispersal function maintains genetic variation within natural populations, which may foster resiliency needed to adapt to changing environmental conditions. The future of species may rely on our careful attention to managing for connectivity now.

Defining the adequacy of dispersal habitat in forests is a complex topic (Noss et al. 1997) and may address a variety of elements, including habitat condition, corridor sizes (length, width), and corridor redundancy (Pinto and Keitt 2008). Redundancy is especially relevant because multiple connectivity pathways can assist dispersal across landscapes by organisms in different locations and increase the probability of movement in the face of many interacting site-specific factors (microsite features, disturbances). Redundancy of habitat connectivity hedges against catastrophe, uncertainty, and stochastic processes that can affect individuals and sub-populations that vary in their movement propensities, possibly related to patch size, habitat quality, and population demography.

Low-mobility species may merit special attention devoted to the placement and redundancy of connectivity corridors, because barriers to dispersal may arise as a result of their basic biology and ecology (Raphael and Molina 2007). These species may move slowly and require refugia along corridors because it may take them years to move between optimum habitat patches. Due to a potentially longer residency time within connectivity corridors, low-mobility species may be particularly vulnerable to sub-optimal corridor conditions and stochastic processes. Hence, redundancy of connections may be critically important to increase their likelihood of successful movement across landscapes for such low-mobility species. Patches of higher-quality habitat within dispersal corridors may be used as stepping stones for such species and may be an essential aspect of their long-term persistence (e.g., Grant et al. 2010). Such stepping stones may function as habitat refugia or "stopover reserves" (Dobson et al. 1999), which promote survival of individual organisms as they move through the environment. Stepping stones may have more suitable physical habitat conditions than the surrounding area, or may allow individuals to forage to replenish energy reserves or survive harsh seasons (summer, winter) in localized refugia, from which they may disperse again later.

Herein, we synthesize our ongoing studies of the utility of headwater riparian areas as proposed connectivity corridors, or linkage areas, for dispersal of riparian-associated and low-mobility species in Pacific Northwest forests. Once designed, such headwater linkage areas may benefit many taxa. Our studies also conceptually integrate aquatic network and upland-forest habitats, functions, and processes. The combination of protections for aquatic and upland systems is providing new insights into forest ecosystem management approaches. We summarize the key considerations for the geometric orientation of connectivity pathways to assist migration of species across watersheds and across webs of connections, to maintain linked aquatic-terrestrial populations at landscape scales. Our goal here is to provide a summary of these conceptual designs, while research continues to address these issues and advance design effectiveness.

Utility of Watersheds as Redundant Landscape-scale Linkage Units

Watersheds are widely accepted units for monitoring and evaluating the effects of land use on aquatic resources (Omernik and Bailey 1997). Where their boundaries can be clearly mapped, watersheds are increasingly common units for forest management planning and conservation designs. For example, in the U.S., the Aquatic Conservation Objectives of the federal Northwest Forest Plan (USDA and USDI 1994: p. B-11), address connectivity among watersheds:

> *"Maintain and restore spatial and temporal connectivity within and between watersheds. Lateral, longitudinal, and drainage network connections include floodplains, wetlands, upslope areas, headwater tributaries, and intact refugia. These network connections must provide chemically and physically unobstructed routes to areas critical for fulfilling life history requirements of aquatic and riparian-dependent species."*

Hydrologic units (HUs), delineated by the U.S. Geological Survey (Seaber et al. 1987), are also a convenient and widely used basis for forest assessment and planning (e.g., Maxwell et al. 1995; Suring et al. 2011). The HU coding describes a hierarchical system of units nested by drainage area; larger code numbers designate smaller drainage areas. Watersheds or segments of watersheds comprise HUs. Even though the majority of HUs at each level of the hierarchy are not true topographic watersheds, such a perspective can aid biodiversity conservation designs, especially as smaller headwater basins are delineated and used for replicating protected areas (e.g., 6t^{h}-code HUs: Suzuki et al. 2008) and creation of redundant connections across landscapes (via 6th- and 7th-code HUs: Olson and Burnett 2009).

The value of using headwater basins as the premise for establishing connectivity corridors across forested landscapes is due to their habitat conditions, potential use by a variety of organisms, frequency of occurrence on the landscape, and minimization of dispersal distances (fig. 1). Olson et al. (2007) summarized some of the merits of headwater riparian habitats for species in the northwest, including providing cool, moist microclimates for interior-forest dependent organisms and aquatic-riparian associated species such as amphibians. Some taxa may use these areas due to their habitat suitability; others may respond to streams as movement barriers, and then move along banks parallel to such barriers. Streamside areas may be dispersal funnels or runways for a variety of species. For example, we have seen terrestrial salamanders (species that do not use stream or pond habitats for breeding or other life-history functions) moving predominantly through near-stream areas (D. Olson and M. Kluber, unpubl. data). Additional taxa that use riparian corridors in northwestern forests include a variety of lichens, bryophytes, fungi, vascular plants, mollusks, mammals (e.g., ground-dwelling mammals: Wilk et al. 2010), birds, and general forest-obligates that may occur in legacy forest attributes such as wolf trees along riparian buffer zones. As a minimum estimate across taxonomic groups, over 100 species were identified as likely to benefit by habitat protections of combined intermittent and perennial streams provided by riparian reserves in federal forest lands in the range of the Northern Spotted Owl (*Strix occidentalis caurina*) (table 1) (USDA and USDI 1997). Species with restricted dispersal abilities were identified for special consideration relative to utility of riparian reserves during watershed analyses under the Northwest Forest Plan (USDA and USDI 1997).

Furthermore, the high density of small streams in upland northwest forests has been widely recognized over the last 20 years, as our basic

Table 1—Species benefitting from interim riparian reserves developed for the federal Northwest Forest Plan (from table B1 in USDA and USDI 1997). Riparian reserve protection includes a one site-potential tree-height or 30.5 m (100 ft) buffer, whichever is greater, as an interim measure along all intermittent streams, and a two site-potential tree-height buffer as an interim measure along perennial streams (see USDA and USDI 1993, page III-9).

Taxonomic group	Species
Bryophytes	*Antitrichia curtipendula, Douinia ovata, Kurzia makinoana, Scouleria marginata, Tritomaria exectiformis*
Fungi	
Rare chanterelles	*Polyozellous multiplex*
Rare gilled mushrooms	*Clitocybesubditopoda, C. senilis, Neolentinus adherens, Rhodocybe nitida, Rhodocybe speciosa, Tricholomposis fulvenscens*
Rare cup fungi	*Helvella compressa, H. crassitunicata, H. elastica, H. maculata*
Jelly mushroom	*Phlogiotis helvelloides*
Moss-dwelling mushrooms	*Cyphellostereum leave, Galerina atkinsoniana, G. cerina, G. hetrocysis, G. sphagnicola, G. vittaeformis, Rickenella setipes*
Lichens	
Riparian lichens	*Certelia cetrarioides, Collema nigrescens, Leptogium burnetiae var. hirsutum, L. cyanescens, L. saturninum, L. teretiusculum, Platismatia lacunose, Ramalina thrausta, Usnea longissima*
Aquatic lichens	*Dermatocarpon luridum, Hydrothyria venosa, Leptogium rivale*
Vascular plants	*Bensoniella oregano, Botrychium minganense, B. montanum, Coptis trifolia*
Mollusks	*Ancotrema voyanum, Crypotmastix devia, C. henersoni, Monadenia fidelis salmonensis, Verspericola depressa, V. sierranus, Fluminicola* spp. nov. 1-20*, F. seminalis, Helisoma newberryi newberryi, Juga (C.) acutifilosa, J. (C.) occata, J (O.)* spp. nov. 2-3*, J. (Oreobasis) orickensis, Lanx alta, Lyogyrus* sp. nov. 1, 3, *Pyrgulopsis intermedia, Vorticifex klamathensis sintsini, V.* sp. nov. 1
Amphibians	
Riparian	*Aneides flavipunctatus, Rhyacotriton cascadae, R. kezeri, R. variegatus, Dicamptodon copei, Plethodon vandykei, Ascaphus truei*
Fish	Coho Salmon (*Oncorhynchus kisutch*), fall and spring Chinook Salmon (*O. tshawytscha*), resident and sea-run Cutthroat Trout (*O. clarkii clarkii*), resident Rainbow Trout (*O. mykiss*), summer and winter Steelhead (anadromous *O. mykiss*)
Birds	Common Merganser (*Mergus merganser*) [Marbled Murrelet, *Brachyramphus marmoratus*; Northern Spotted Owl, *Strix occidentalis caurina*]
Bats	Fringed, Long-eared, and Long-legged Myotis (*Myotis thysanodes, M. evotis, M. volans*), Hoary Bat (*Lasiurus cinereus*), Pallid Bat (*Antrozous pallidus*), Silver-haired Bat (*Lasionycteris noctivagans*)
Other mammals	Fisher (*Martes pennanti*), Marten (*Martes americana*), Red Tree Vole (*Arborimus longicaudus*)

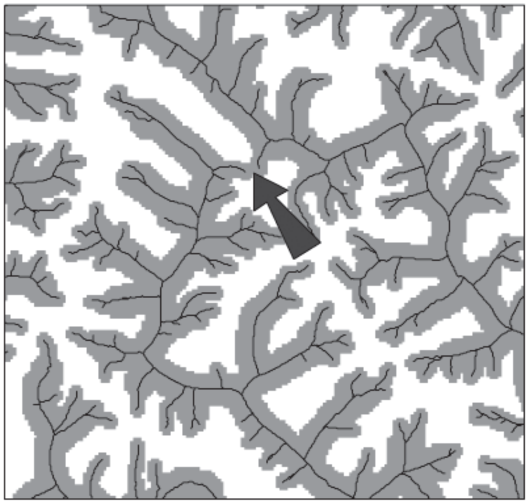


Figure 1—Example interim riparian reserve network from the US federal Northwest Forest Plan implemented in the Pacific Northwest, showing frequency of headwater streams on the landscape and the resulting one and two site-potential tree-height buffers along streams (upper left quadrant). Arrow indicates example over-ridge area where the distance between headwater riparian reserves in different watersheds is small and over-ridge connectivity may be more easily achieved. These headwater riparian areas can be used to facilitate landscape linkage area designs for organism dispersal and aquatic-terrestrial habitat connectivity functions.

knowledge of stream and forest ecology has expanded. In some areas, headwaters comprise 80 percent of a stream network (Gomi et al. 2002). This realization intersected with forest management practices when mapping of Northwest Forest Plan scenarios revealed that large percentages of watersheds were being incorporated into interim riparian reserves due to the high density of headwater stream networks (fig. 1). An additional value of using headwater drainages to plan landscape connectivity designs is that the distance from headwater streams to ridgelines is the shortest within a watershed, hence reducing travel distances for overland dispersal to neighboring stream-riparian areas or forest reserve blocks. Distance analysis tools, such as for "least-cost path" in landscape modeling (e.g., ArcGIS, Environmental Systems Research Institute, Inc., Redlands, CA), have been developed to assess distances between habitat patches. These tools would be useful for designing least-distance headwater linkage areas. Least "cost" path is a relevant term applied to the economics of animal movements, to minimize the distance moved—especially for mobility-restricted organisms. This term may also apply to the economics of forest management if identification of a dispersal corridor results in a financial cost for on-the-ground implementation or affects revenue from resource extraction in a managed forest context.

Northwest Forest Plan riparian reserves were intended as major contributors to the maintenance and restoration of aquatic conservation objectives, including aquatic network connectivity (USDA and USDI 1994). The importance of linking headwater stream functions and processes to those of downstream stream networks has been a focus of much work in the last two decades. Welsh (2011) captured many elements of the developing history of stream network theory and the role of aquatic connectivity in summarizing the conceptual frameworks of geomorphic channel processes (transfer and depositional zones), nutrient cycling (upstream marine influence via salmonid migration, downstream nutrient spiraling via down wood movements), aquatic-riparian linkages via reciprocal subsidies, and the intersection of herpetofaunal distributions with the classic stream continuum concept of taxonomic patterns that vary with stream order. As we look up the aquatic network into headwater streams and beyond, we summarize how extending riparian buffers up drainages and connecting them over ridgelines can both maintain terrestrial connectivity and functionally link aquatic-terrestrial systems.

Several conceptual designs of riparian buffer widths and patch reserves have been proposed to assist over-ridge migration of organisms within forests (fig. 2; Olson et al. 2007). Over-ridge connectivity considerations were further developed by Olson and Burnett (2009), and modeled for the Oregon Coastal Province. This model of connectivity linked every 6th- and 7th-code HU to each neighboring HU. Focusing on the Siuslaw River basin, a 4th-code HU within

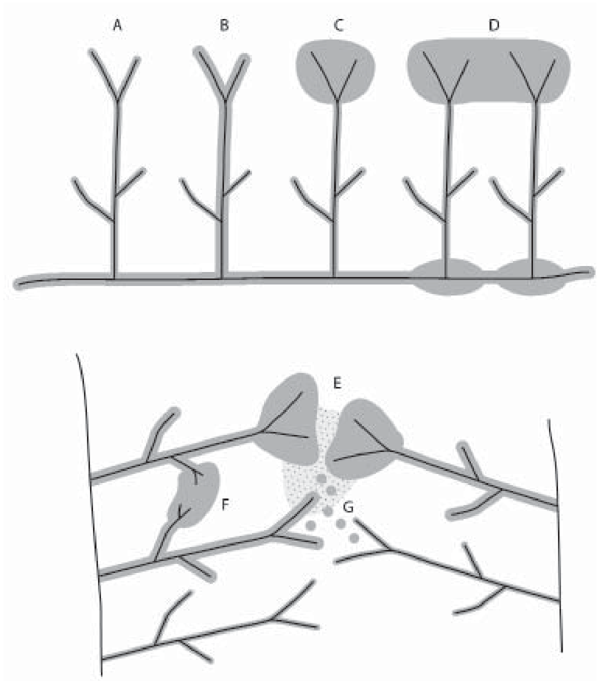


Figure 2—Headwater management considerations to retain aquatic-riparian biodiversity by stream buffers of different widths (A, B) and provide linkage areas between adjacent basins (C-G) using alternative forest management practices including uncut blocks (C, D, F), thinning (E), and leave islands (E, G) (from Olson et al. 2007).

that area, the linkage design illustrated where one over-ridge link could connect each adjacent HU (fig. 3). At the 7th-code HU scale, one link between each adjacent 7th-code watershed resulted in roughly 15 percent of headwater streams being extended and connected For the Oregon Coastal Province, this resulted in over 5,000 links, with about one link per 4.6 km^2. This is an example of redundant connectivity, essentially creating a web of connections across the landscape. Using the 6th-code HU scale, the amount of connectivity created is approximately halved, with one link per 9.3 km^2 for the Oregon Coastal Province.

There are no defined guidelines for how many links or how much habitat connectivity is necessary to maintain populations. The amount of dispersal habitat that might be needed to sustain even highly researched species, such as

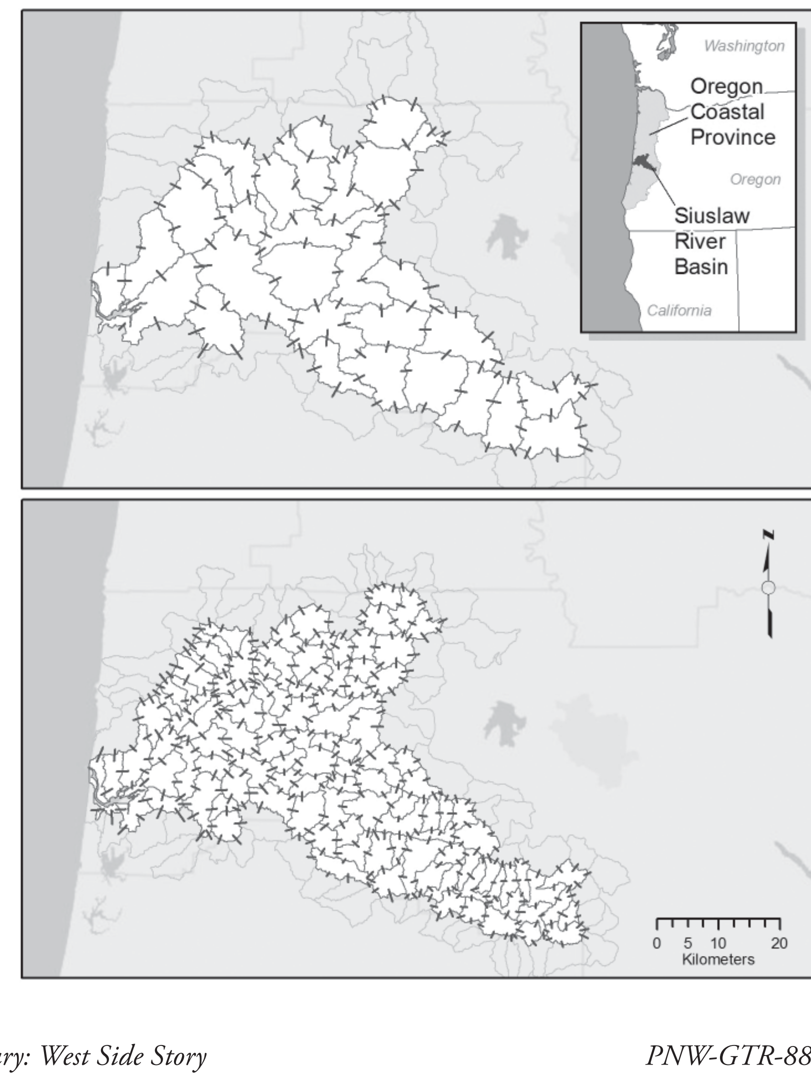


Figure 3—Linkage areas between watersheds can provide connectivity of headwater habitats across landscapes. In the Oregon Coast Range Province, the Siuslaw River basin, a 4th-code Hydrologic Unit (HU), is used to illustrate: A) a single connection between adjacent 6th-code HUs; and B) a single connection between adjacent 7th-code HUs, which results in 376 connections across the basin and if expanded to the entire province, about 5000 links within 23 000 km^2 (from Olson and Burnett 2009).

the northern spotted owl, is unknown; a "more is better" attitude prevails in the face of this uncertainty. Nevertheless, research is accruing about how much dispersal may be needed to maintain genetic diversity within and among populations. The "one migrant per generation" rule has been offered as a minimum level to reduce genetic isolation, inbreeding, and bottlenecks (e.g., Mills and Allendorf 1996). However, such a rule has many underlying assumptions that may not be supported when the complexities of natural systems are considered (e.g., Wang 2004).

Furthermore, relating effective migration rates to habitat protections in managed systems is not a straightforward exercise: if we build corridors, will they be used? Ongoing mark-recapture, radio tracking, and genetic studies are helping us to answer this question. For example, genetic connectivity analyses of stream-associated Rocky Mountain Tailed Frogs (*Ascaphus montanus*) in Idaho supported this species' affiliation with intact forested habitats: their path of connectivity followed riparian corridors in managed forests (Spear and Storfer 2010). This pattern supports the "riparian corridors as funnels" concept, but it contrasted with Coastal Tailed Frog (*A. truei*) genetic connectivity pathways in the Olympic Peninsula, Washington, which were primarily overland in areas that had timber harvest activities (Spear and Storfer 2008). Precipitation and population differences between these areas were hypothesized as accounting for these differences, as the more mesic conditions that prevail in northwestern Washington may facilitate the upland dispersal of moisture-reliant tailed frogs. Other studies (Wahbe et al. 2004; Johnston and Frid 2002; Dupuis and Steventon 1999; Nauman and Olson 2004) also found differences in riparian-corridor associations of various amphibian species in response to climate and forest conditions, generally supporting their ability to respond to microsite gradients with an apparent affiliation to cool, moist local conditions (e.g., riparian "funnels") (Olson et al. 2007). Furthermore, Spear et al. (2012) reported that Coastal Tailed Frogs track remnant tree patches in their migration pathways after the volcanic blast at Mount St. Helens, Washington. So, if we build it, will they come? The early answer is "yes, but…" —meaning that a variety of organisms appear to be occurring in or moving along pathways of retained habitats, but with geographic, taxonomic, and population-specific contexts being important considerations. A similar conclusion has recently been supported for hedgerows as corridors between woodland fragments (Davies and Pullin 2007). More research on the design of effective linkage areas will be needed. In the interim, conceptual priorities for landscape connectivity designs can be identified, and these relate directly to emerging research priorities.

Priority Areas for Habitat Connectivity

Prioritizing linkage area placement may be important to address connectivity objectives under economic constraints, and to advance research into the effective design of linkage areas. Because linking all adjoining watersheds at small HU scales may be difficult for land managers to plan and implement in the face of myriad conflicting resource objectives, priorities may guide the first steps in connecting habitats. Olson and Burnett (2009) itemized linkage area considerations at two spatial scales, landscape and drainage area (table 2). Here, we further develop five of these priority considerations:

1. "Triads," where three large basins, with limited or no aquatic connectivity, converge at their headwaters;
2. Climate change considerations including north-south, east-west, and altitudinal linkages;
3. Landslide-prone areas;
4. Species conservation strongholds; and
5. Diagonal considerations.

Table 2—Design considerations for placement of headwater linkage areas to assist migration of forest-dependent species in the Pacific Northwest (Olson and Burnett 2009). **Bold-face type** indicates new concepts discussed further in text.

Linkage Area Design Considerations	Priorities
Landscape scale	
1. Connections across large basins	**"Triads"** – headwater locations that link three adjoining basins having no aquatic connectivity.
2. Climate change migration corridors	North-south (latitudinal) dispersal routes. Altitudinal dispersal routes. Migration across ecoregion boundaries. **East-west dispersal routes.**
3. Linking landscape fragments	Connecting remnant late-successional and old-growth (LSOG) forest patches to other patches or restored habitats may aid dispersal of LSOG-associated species, especially those with dispersal limitations such as lichens, bryophytes, and mollusks; creation of connected archipelagos of patches.
4. Disturbance frequency	Correlating frequency of connections with rates of landscape-scale disturbances, natural or anthropogenic; i.e., more linkage areas in more-disturbed places.
5. **Redundancy**	Planning for multiple paths across landscapes will improve dispersal probabilities.
Drainage-basin scale	
6. Known sites for target species	Low-mobility species. LSOG-associated species. Species with status of concern. **Biodiversity hotspots** – communities. **Species "strongholds"** – priority species management areas such as key watersheds
7. Existing protections	Co-location of linkages on current set-asides (e.g., federal late-successional reserves, owl "cores", Survey and Manage species sites, botanical set asides, landslide-prone areas included in riparian reserves)
8. Short connections	For economy of space, with economic and ecological benefits, shorter connectivity corridors are preferred; ecologically, shorter distances for dispersal may reduce energetic costs for individual movements and time needed for propagules to disperse.
9. Paths of least resistance	Easier dispersal routes may be lower-gradient or lower-elevation "saddles" across ridgelines. Wind-dispersers may have least resistance in paths that follow wind directions during seasons of dispersal.
10. Risk of disturbance	Use hazard models for disturbances such as landslides, debris flows, ice/wind damage, and fire in placement of linkage areas, or in decisions about the need for redundant linkages. For example, debris-flow-prone areas may be headwater set-asides during riparian reserve delineation, and such areas may be co-located with dispersal corridors; redundant links may be considered in fire-prone areas. Mapped overlays of roads, recreation areas, human development, and mining might be avoided during linkage area delineation, when alternative locations exist.
11. Land ownership patterns	Co-location of links on federal and state lands, where possible. **Diagonal linkage areas across checkerboard ownerships.**

Each of these five considerations results in a geometric view of how connectivity webs may be arranged across landscapes.

These five considerations are not mutually exclusive; how they may interact during prioritization exercises also is developed briefly here. Although they were derived for northwest forest landscapes, these concepts may have broader utility worldwide.

1. "Triads"

In the Oregon Coastal Province, Olson and Burnett (2009) highlighted the potential importance of linking larger river basins, which have no freshwater connectivity, through existing riparian buffer networks. Over-ridge forest habitat linkages may be absent unless reserves are placed in the area. Streams in such basins may flow directly to the Pacific Ocean or into a much larger river without a forested riparian area, and so have headwaters that are functionally disconnected. Here, we examined 4th-code HUs for the Oregon Coastal province, the scale of the Siuslaw River basin highlighted above. We then looked for locations where three of these 4th-code HUs joined at their headwaters: we call this a "triad" location. For example, headwaters of the Siuslaw River, Yaquina River, and Marys River converge at Marys Peak (between Corvallis, Newport, and Waldport, OR), which would be one such triad. Only 18 of these headwater triads exist for the Oregon Coastal Province (fig. 4). We suggest that such triads be considered priorities for habitat linkage areas because these would be spatially economical for land managers to implement and potentially ecologically efficient as connections across three watershed boundaries simultaneously.

A current research priority is to empirically assess the proposed linkage-area function of landscape locations such as headwater triads. Using a genetic approach, we have sampled northwestern amphibians from headwater streams of adjacent drainages that are potentially connected across ridgelines in the Oregon Coast Range, including three adjoining headwaters in triads (such as Marys Peak). Preliminary genetic analyses of the Coastal Giant Salamander, *Dicamptodon tenebrosus*, generally support our contention of over-ridge connectivity among drainages (L. Knowles and M.R. Marchán-Rivadeneira, Univ. Michigan, unpubl. data). Previous studies have supported overland connectivity of stream-breeding amphibians (e.g., Spear and Storfer 2008, 2010), and such animals have been found up to 400 m from streams (Olson et al. 2007), but no previous published study has designed

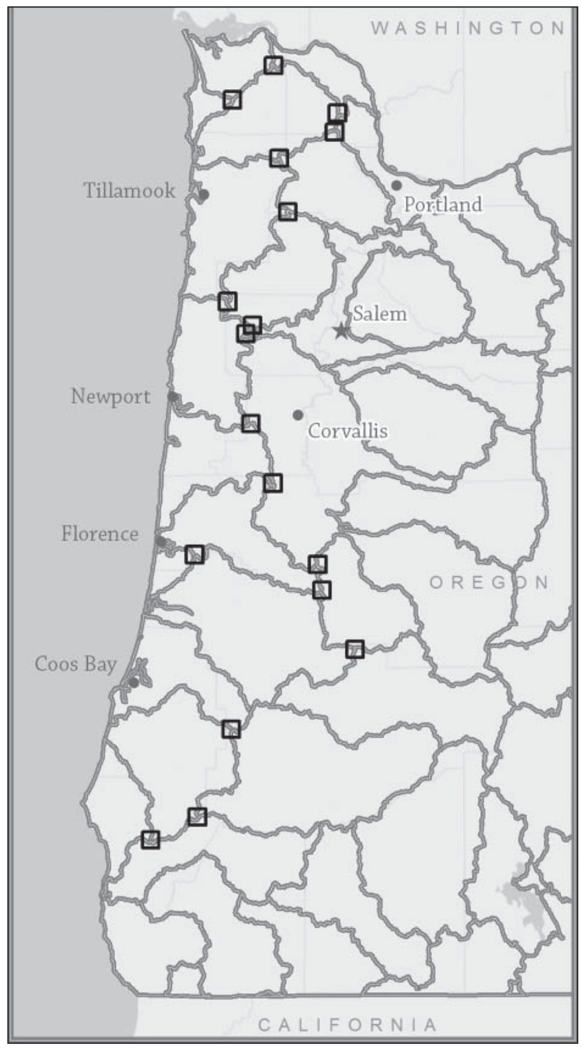


Figure 4—"Triads" are indicated (squares) where three 4th-code Hydrologic Units in the Oregon Coast Range Province meet at their headwaters. Triads are priority locations for linkage area or 'species stronghold' placement to effectively manage for species dispersal simultaneously across three distinct watershed boundaries.

sampling to specifically address headwater over-ridge connectivity among discrete drainages. This remains an information gap that could be addressed for all forest taxonomic groups, and would aid the adaptive management of the triad connectivity concept.

2. Climate Change

A second priority consideration for northwest forest connectivity is assisting migration in the face of climate change (Olson and Burnett 2009). Predicted climate change effects on northwest forest habitats include drought, insect, and fire effects on forest stands, with large conifers and high-elevation trees being vulnerable to losses (Spies et al. 2010). Aquatic habitat changes in forested landscapes are anticipated in cold-water mountain streams (Spies et al. 2010) and in headwaters (Olson and Burnett 2009). Increasing stream temperatures, with negative implications for cold-water fauna, are already apparent across the northwest (Isaak et al. 2011). Given uncertainty in the geographic specificity of climate change trajectories due to complex El Niño and Pacific Decadal Oscillation cycles, "dynamic and adaptive thinking" (Spies et al. 2010) is needed. A prudent course for linkage area placement may be to consider connected routes in north-south, east-west, and altitudinal directions within and among watersheds (fig. 5). Such consideration may allow multiple potential pathways of movement for species facing changing conditions. Pockets of suitable microhabitats for species persistence may be related to local conditions, and may occur as "stepping stones" along these linear trajectories, like beads along a string. Providing connectivity paths adjoining both riparian areas and north-facing slopes is one such example, with both near-stream areas and hill shading resulting from topographic relief providing cool, moist conditions for target species such as some late-successional and old-growth (LSOG)-associated salamanders (e.g., Suzuki et al. 2008). Landscape-scale monitoring of forest conditions and species distributions may inform

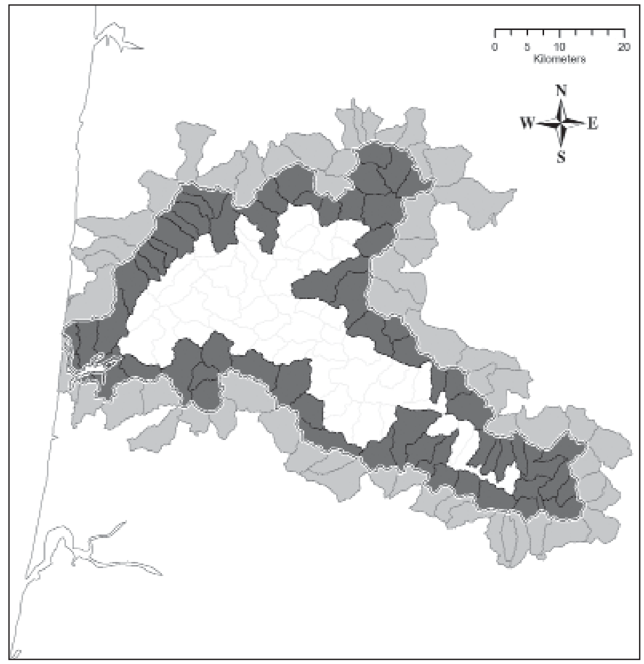


Figure 5—Designs for assisted migration of species in the face of climate change might include prioritizing headwater linkage area placement across north-south and east-west watershed boundaries. The Siuslaw River basin, a 4th-code Hydrologic Unit in the Oregon Coast Range Province, is shown highlighting perimeter sub-drainages (dark grey) where linkage areas could be made to discrete drainages to the north, south, and east.

adaptive management of likely climate change refugia for different taxa.

3. Managing Disturbances: Landslide-Prone Areas

A third priority consideration for the design of linkage areas is to integrate their placement with local disturbance regimes. Landslides and debris flows can be dominant disturbance processes affecting headwater streams in forested, mountainous regions (e.g., Benda 1990; Iverson et al. 1997). The Northwest Forest Plan directs that riparian reserves incorporate landslide-prone areas to reduce the probability that activities associated with timber harvest will alter wood and sediment inputs to streams by changing the rate, magnitude, composition, or timing of debris flows. Co-locating linkage areas with existing riparian reserves, where these include steep areas prone to landsliding, can provide economic efficiency and conservation synergy for land and resource managers. Burnett and Miller (2007) modeled differences among hill slopes and headwater channels in probabilities of initiating and transporting debris flows that deliver to fish-bearing channels for the Oregon Coastal Province (fig. 6). Those headwaters with the highest likelihood of affecting downstream areas important for fish might be high priorities for extending riparian reserves over ridgelines. Because debris flows can be important sources of large wood (May and Gresswell 2003; Hassan et al. 2005), a fundamental component of stream habitat complexity (Bilby and Bisson 1998; Gregory et al. 2003), managing these expanded riparian reserve areas to accelerate tree growth could be an additional consideration. Redundancy of connections would be important when planning ground-disturbing activities for linkage areas with a high probability of landsliding. To aid identification and adaptive management of landslide-prone areas, these areas have been mapped for many northwest forests by the NetMap interactive web-tool developed by Earth Systems Institute (*http://netmaptools.org/*).

4. Species Strongholds

"Species strongholds" are areas where biodiversity conservation is a priority, and where thriving populations can occur to anchor species persistence in the region. Retaining connectivity among species strongholds enhances the likelihood of persistence under the uncertainty of stochastic events (catastrophic fire, disease outbreaks) or emerging patterns of disturbance (climate change) that may affect any particular stronghold. Managing stronghold-to-stronghold connectivity is a fourth priority to consider in developing linkage area designs across forest landscapes. Species strongholds may be created for communities of diverse taxa at larger spatial scales by land-use allocations such as

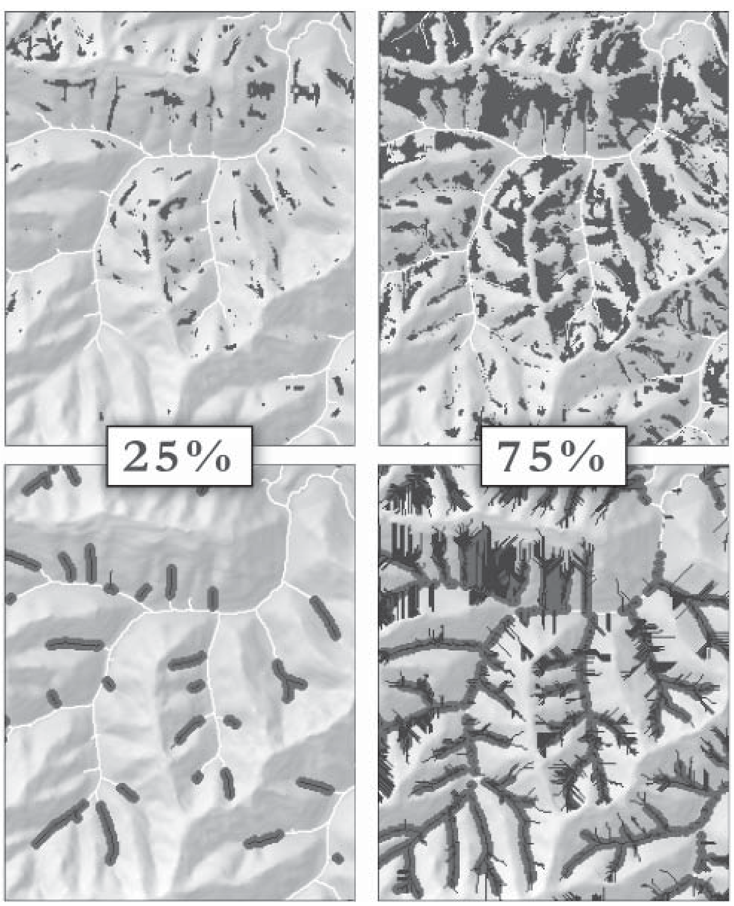


Figure 6—Placement of between-drainage links might consider other landscape-scale provisions such as management scenarios to retain the natural disturbance regime, including landslide-prone areas managed to deliver wood and sediment to streams. Top figures show models of 25 percent and 75 percent of the landslide-prone hillslopes in an example forest landscape, and bottom figures show their likely traversal paths to streams (Burnett and Miller 2007). Headwater riparian buffers of these areas provide long-term wood and sediment inputs for stream biota. From Olson and Burnett 2009.

Congressionally reserved lands (wilderness areas, national parks), or the Northwest Forest Plan late-successional reserves or key watersheds (USDA and USDI 1993, 1994). At smaller spatial scales, strongholds for a targeted species of concern may be critical habitat areas, such as caves, ponds, meadows, botanical set-asides, or areas managed for Survey and Manage species under the federal Northwest Forest Plan (USDA and USDI 1993, 1994). Riparian buffers themselves might be considered as strongholds, but here, we expand that perspective to other areas.

Developing new species strongholds is particularly important when considering connectivity issues. Three examples follow. First, areas with high "intrinsic potential," the capacity to support high-quality habitats for salmon (Burnett et al. 2007), may serve as nuclei for designing linkage areas. Intrinsic potential models have been developed and broadly applied for salmonids in the Pacific Northwest and elsewhere (e.g., Mollot and Bilby 2008; Sheer et al. 2009; Busch et al. 2011; Barnett and Spence 2011). Streams with high intrinsic potential can be identified and then targeted, as appropriate, for salmon conservation across a landscape. Such areas of high intrinsic potential are essentially "species strongholds" from which aquatic-terrestrial linkage areas can originate. Areas of high intrinsic potential for some salmonid species may occur in larger streams, but tools exist to easily identify headwater streams that feed into these both laterally and from upstream (Clarke et al. 2008).

Second, criteria for Priority Amphibian and Reptile Conservation Areas (PARCAs) are under development for nationwide application (Riley et al. 2011). PARCAs are being discussed for integration into the landscape planning processes of other entities, such as the U.S. Department of Interior Landscape Conservation Cooperatives (*http://www.doi.gov/lcc/index.cfm*), which is a partnership network to sustain America's land, water, wildlife, and cultural resources. Once established, PARCAs would function as species strongholds. Similarly, the International Union for the Conservation of Nature (IUCN) is developing criteria to identify sites of global significance for biodiversity conservation, called Key Biodiversity Areas. Such areas are synonymous with the concept of species strongholds. The additional element that we suggest is to provide connectivity among such areas.

Third, triads, as we previously described, could be ideal locations for species strongholds, as these occur at the ridgeline junction of three large basins. However, we note that triads are not established biodiversity hotspots, and are proposed here as a conceptual design.

Development of landscape-scale linkage webs from either new or existing species strongholds is needed to reduce isolation of those areas, and as possible to allow them to function as potential "source" habitats with optimal conditions that can anchor species over time and also connect to adjoining areas, in a metapopulation context. Linking dispersal pathways from strongholds up and over ridgelines to adjacent watersheds and neighboring strongholds is a direct approach that may offer a least-cost path. Relevant to our proposed headwater linkage areas concept, connecting such species strongholds to headwaters which then extend and connect over ridgelines is another consideration. Additionally, strongholds may be linked to protected riparian areas along larger streams that are subsequently extended upstream into headwaters and connected over ridgelines. Multiple connectivity pathways may be conceived. As a web of connections is considered relative to species stronghold connectivity, the previous priorities discussed above and outlined in table 2 can be overlain, including large basins and triads, linear trajectories to address for climate change gradients, and occurrence of landslide-prone areas.

Adaptive management of strongholds may need to be addressed over the long term as future conditions unfold. As applied here, the concept of a stronghold evokes less of an immovable fortress than an anchor. A species stronghold intended to anchor habitat may need to function as do real anchors on occasion, and be repositioned or "drag" across landscapes in response to changing conditions or management priorities (Olson et al. 2007). The temporal scale of strongholds can be addressed at the time of their development, and interact with the spatial scale of stronghold designs and the frequency of strongholds. For example, habitat anchors designed to drag across landscapes may be implemented more easily if they are smaller and more numerous. Olson et al. (2007) suggested considering 6th-code watersheds (HUs) as a spatial scale for amphibian habitat anchors. The anchor concept warrants testing, with a sufficient timeframe to weigh success at the landscape scale, in addition to replication. It may have greater success if it were to be implemented in areas with more resilient ecosystems.

5. Thinking Diagonally: Funnels and Chains

The geometry and land-management context of land-use and land-ownership parcels on the landscape is a final set of priority considerations for linkage-area placements that we will develop briefly here. The northwest forest landscape is a patchwork of land ownerships and land-use allocations, each with differing management priorities, which creates a complex challenge for biodiversity conservation (Suzuki and Olson 2007). During planning for large blocks of forest land, and during planning of individual projects at smaller spatial scales, managing for connectivity within and among ownership areas can be difficult due to differing priorities across boundaries. To diminish the dilemma of achieving effective biodiversity conservation in such a multi-ownership landscape, it may help to think of streams as dispersal "funnels" that serve to channel organisms along protected riparian areas, and connectivity corridors or linkage areas as "chains" functionally moving animals up and over ridgelines (Olson and Kluber, unpubl. data). Overlaying many of the previously discussed priorities can provide an integrated perspective for addressing the challenges of land-ownership/allocation geometries.

Diagonal linkage areas are of specific relevance in a landscape with a checkerboard ownership pattern (fig. 7), and in other landscape geometries that abut at corners or other edge types (Olson and Kluber, unpubl. data). Species dispersal along such diagonals might be promoted by forest management actions that retain habitat elements toward the corners of such lands. For example, weighted green-tree retention, leave islands, and directional felling of down wood (recruitment of large logs, in particular) from corners may assist migration of species along the diagonal by providing chains or stepping stones of suitable microhabitats for species refugia. Linking chains of habitat elements from corners to stream- and riparian-protected areas, especially headwaters (fig. 7B), may functionally extend and connect riparian buffers. Organisms that are funneled along riparian areas may venture through corners via these habitat chains. A chain of habitat need not extend from headwaters, but could extend from any part of a riparian buffer, or from a species stronghold, as discussed above.

It may be neither feasible nor desirable to address habitat connectivity at all corners of adjacent lands within an ownership. Similarly, when land parcels are in close proximity but do not adjoin, it may not be possible to consider linkage areas along their entire boundaries. Several additional design concepts arise and interface with ideas presented above.

First, linkage areas among land parcels might be "stream-lined" if streams align through corners (fig. 8), or connect nearby land blocks. When streams follow diagonals in a checkerboard landscape, riparian protection may more effectively promote multi-species diagonal dispersal: funnels without the added chains linking across diagonals. Streams that loosely follow diagonals, not intersecting exactly at corners, could be quite functional to assist species

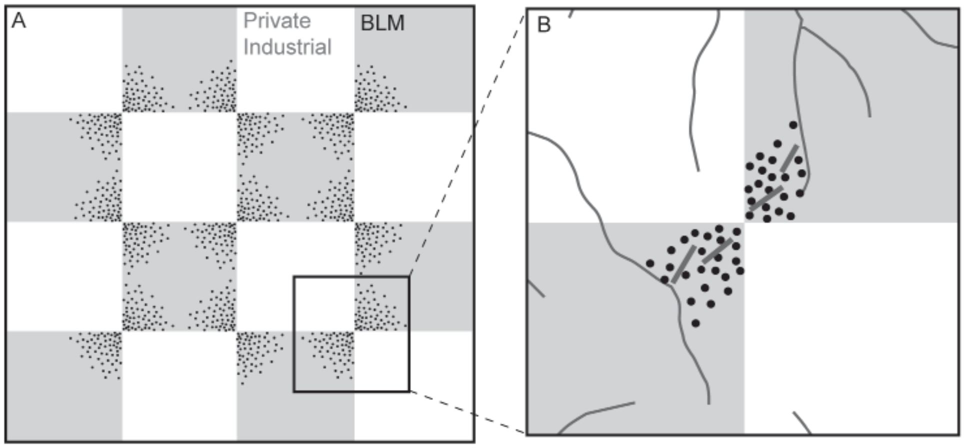


Figure 7—In a checkerboard ownership pattern, such as that created by the Oregon and California Lands Act (1937) where US Bureau of Land Management (BLM) and private industrial forestlands are intermixed, management for connectivity along diagonals may improve likelihood of species dispersal within ownerships. A: Corners are shaded to show linkage area considerations. However, routes along selected diagonals to species strongholds, reserves, or triads might be used to prioritize which corners are chosen for connectivity emphasis. B: Forest management options to facilitate species dispersal from stream corridors (which may serve to funnel species movements) to corners might include chains of habitat structures provided by green tree retention, directional log placement from corners to streams, or both. Concepts could be applied to other ownership geometries with corners or edges in proximity.

Figure 8—Connectivity designs to aid species dispersal among ownership blocks may overlay on streamside riparian management zones. This US Bureau of Land Management study site for the Density Management and Riparian Buffer Study of western Oregon (Cissel et al. 2006) shows riparian buffers extending along the full diagonal (A) as well as laterally toward an opposing corner (B), with leave islands and dispersed tree retention aiding habitat connectivity, and to a neighboring private land block (C). "Stream-lined" connectivity (A) may aid within-watershed dispersal, but overland connectivity designs (B and C) may warrant consideration to link or "chain" habitats overland between watersheds. Photo provided by Oregon Bureau of Land Management.

migration, in this regard. Streams that link disconnected parcels may similarly function to funnel organisms' movements. The context of the adjoining lands may need to be assessed, however. Managing such a stream-line to promote its potential connectivity function is a consideration, but such stream-lined connectivity does not address overland dispersal. Chains from streams to ridgelines are needed to fully integrate aquatic and terrestrial landscape connectivity functions. Collaborative management of such stream-lines and overland chains among ownerships and across land-use allocations within ownerships, remains a challenge.

Second, in multi-ownership landscapes, road densities may be higher than in single owner landscapes. An assessment of the effects of roads on species connectivity designs may be particularly important in these landscapes. In particular, paved roads or high-use unpaved roads may be barriers to low-mobility species. As roads intersect streams, aquatic organism passage may be affected, with consequences for overland connectivity. Site-specific designs can include these considerations.

Third, as hazard models of disturbances are developed for a landscape, it may be helpful to ask how hazards align with land-ownership boundaries, land-use allocations, and existing connectivity webs. For example, how are landslide-prone areas arranged relative to the geometry of lands by ownership and land-use allocation? As discussed above, can priority linkage areas be designed to overlay on landslide-prone areas that are already set-asides for riparian reserve management, and now also serve "to chain" habitats to adjoin land-ownership blocks?

Fourth, in a larger landscape context, it may be useful to know how larger basins, climate change projections, and species strongholds are arranged and whether these be used to prioritize connectivity area pathways. Can dispersal routes be conceived from streams and then through land-ownership diagonals or between ownership blocks to foster connections relative to these issues?

Multiple overlapping considerations are emerging, and a stepwise process may be needed to integrate them. Limitations may emerge due to topography, geometry of land configurations at local scales, or pre-existing conditions. For example, a dispersal barrier such as a road may need to be considered first. The existence of under-road culverts may create spatially explicit bottlenecks for connectivity planning. Routing linkage area pathways to those stream corridors and culverts may be needed to increase the odds of dispersal across the road. Culverts that act as dispersal nodes in this way could be prioritized for enhancement to provide passage for non-aquatic species. Similarly, triads and species strongholds, as discussed above, are essentially dispersal nodes. Routing dispersal routes via headwater linkage area pathways to triads and strongholds could increase the overall effectiveness of these conservation measures.

Conclusions

Forest biodiversity conservation is an ecosystem service that will continue to be addressed at local-to-landscape scales in the coming century. Retaining organisms across managed forest landscapes requires a toolbox of approaches including fine- and coarse-scale habitat protections and restoration practices, retaining or creating structural elements that are critical habitats for species, and development and management of connectivity pathways to allow gene flow. Renewed efforts to address communities of organisms as well as species of concern are called for as emerging stressors need evaluation, new knowledge is accrued, and adaptive management of existing forest plans are needed.

We review the numerous benefits of forest connectivity designs that rely on headwater linkage areas, and emphasize priorities for their placement at landscape scales. The benefits of headwater linkage areas include their likely functional role in integrating aquatic and terrestrial systems, their potential use by multiple taxonomic groups, their utility for creating webs of connections across forested lands to increase their effectiveness for biodiversity conservation, and their efficiency in minimizing both the distances that animals must move overland and the financial burdens of forest manager.

Placement of headwater linkage areas may include consideration of a variety of factors (table 2). Prioritizing linkage areas can provide a starting point for managing connectivity among critical habitat areas, suggest directional routes for dispersal among areas, or identify dispersal nodes as anchors for connectivity webs. The five priority considerations that we developed include triads that effectively link three larger basins, north-south and east-west directional routes to address climate change scenarios, linkages overlaid on management of disturbances such as landslide-prone areas, links among species strongholds, and diagonal links that route dispersal across management boundaries. These five concepts can be integrated into an overall geometry of landscape connectivity designs. Our conceptualization of headwater linkage area utility and these priority considerations are posed as hypotheses warranting further study and development. We offer these ideas with the caveat that they will not benefit all taxa in forested landscapes. Extremely rare or patchily distributed organisms with low mobility may need a finer-grained, site-by-site conservation approach (Raphael and Molina 2007).

Acknowledgments

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Headwater Stream Flow, Climate Variation, and Riparian Buffers with Thinning in Western Oregon

Julia I. Burton, Deanna H. Olson, and Klaus J. Puettmann

Poster Abstract

Headwater streams and adjacent riparian areas provide reproductive, foraging, and dispersal habitat for many forest-dependent species, especially amphibians. Although previous studies have shown that the composition of aquatic and riparian animal communities is associated with spatial and temporal patterns of stream flow, the relationships among stream flow, climate, and thinning are not well known. We characterized stream flow conditions associated with 78 headwater reaches in 13 managed forest sites in the Coast Range and western Cascade Range, Oregon, USA, tracked periodically over a 16-year period. Then, we retrospectively examined relationships among riparian buffers, stream flow characteristics, and climate variables over time. We tested the hypotheses that: 1) warmer and drier years would be associated with a shrinking of headwater streams (i.e., increased proportion of stream length in dry channels); and 2) such effects are associated with thinning upslope of riparian buffers, due to the effect of thinning to reduce interception and water uptake from the soil.

Stream flows were only weakly related to interannual climate variability. However, the relationship between climate and stream flow that was observed, although weak, supported the shrinking heads hypothesis (Hypothesis 1). Relationships between climate and stream flow classes were not related to riparian buffer treatments with upland thinning, thus no evidence was found to support Hypothesis 2. Our preliminary results suggest the effects of climate change scenarios on headwater streams, and dependent fauna, may not be mediated by harvesting within riparian zones.

Keywords: stream flow, climate change, thinning, shrinking heads, headwater streams.

Julia I. Burton is a research associate, Department of Forest Ecosystems and Society, Oregon State University, Corvallis, OR 97331; **Deanna H. Olson** is a research ecologist, USDA Forest Service, Pacific Northwest Research Station, Forestry Sciences Laboratory, 3200 SW Jefferson Way, Corvallis, OR 97331; **Klaus J. Puettmann** is Edmund Hayes Professor in Silviculture Alternatives, Department of Forest Ecosystems and Society, Oregon State University, Corvallis, OR 97331.

Thinning Impacts on the Resilience of Wildlife Habitat Quality under Climate Change in Coniferous Forests of Western Oregon

Andrew R. Neill, Klaus J. Puettmann, and Adrian Ares

Poster Abstract

To understand the impacts of overstory density reductions on resilience of forest ecosystems (i.e., the capacity of an ecosystem to maintain desired ecosystem functions in a fluctuating environment), we examined overstory basal area and understory vegetation cover and richness collected 6 years after thinning in seven 40- to 60-year-old forests dominated by Douglas-fir (*Pseudotsuga menziesii* (Mirb.) Franco) in the Oregon Coast Range and Cascade Range of western Oregon. Each site contained an unthinned control (240–1,540 trees·ha^{-1} [tph]) and three thinning treatment units (high-density = 300 tph; moderate-density = 200 tph; and variable-density with a combination of 300 tph, 200 tph, and 100 tph).

We used a mechanistic, trait based approach to quantify impacts of overstory density on understory vegetation components that contribute to wildlife habitat. Vascular plant species that produce fleshy fruits, a major food source for wildlife, were grouped as a functional group. We quantified the resilience of this function as the likelihood of being maintained under increased environmental stress related to climate change; e.g., drought duration and severity. To accomplish this, fleshy-fruit-producing species were categorized in terms of their response types based on traits that assume their ability to tolerate drought conditions.

High variability of overstory densities within and among treatments created an overstory density gradient at each site that ranged from 0–87 m^2·ha^{-1} in basal area. Average cumulative cover and species richness of species that produce fleshy fruits increased with decreasing overstory density, suggesting that thinning will increase the potential food sources for wildlife. Similarly, cumulative cover and species richness of drought-tolerant species that produce fleshy fruits also increased at lower overstory densities, suggesting that thinning will increase the likelihood that the food source for wildlife will be maintained under more droughty conditions. Changes in Pielou's evenness of species cover on a plot along the overstory density gradient were not significant for either species group, suggesting that changes are due to general increases in species abundance and richness, rather than by increases of dominant or rare species.

Keywords: stand density, plant species richness, functional diversity, vegetation response types, habitat quality.

Andrew R. Neill is a graduate student, Department of Forest Ecosystems and Society, Oregon State University, Corvallis, OR 97331; **Klaus J. Puettmann** is Edmund Hayes Professor in Silviculture Alternatives, Department of Forest Ecosystems and Society, Oregon State University, Corvallis, OR 97331; **Adrian Ares** is the director, Sustainable Agriculture and Natural Resource Management Collaborative Research Support Program, Virginia Tech, Blacksburg, VA 24060.

English Equivalents

When you know:	Multiply by:	To find:
Millimeters (mm)	0.0394	Inches
Centimeters (cm)	0.394	Inches
Meters (m)	3.28	Feet
Meters	1.094	Yards
Kilometers (km)	0.621	Miles
Hectares (ha)	2.47	Acres
Square centimeters (cm^2)	0.0155	Square inches
Square meters (m^2)	10.76	Square feet
Square kilometers (km^2)	0.386	Square miles
Cubic meters (m^3)	35.3	Cubic feet
Cubic meters	60,975	Cubic inches
Kilograms (kg)	2.205	Pounds
Kilograms	0.0011	Tons
Kilograms per hectare ($kg \cdot ha^{-1}$)	0.893	Pounds per acre
Tonnes per hectare ($t \cdot ha^{-1}$)	893	Pounds per acre
Tonnes or megagrams per hectare ($t \cdot ha^{-1}$) ($Mg \cdot ha^{-1}$)	0.446	Tons per acre
Square meters per hectare ($m^2 \cdot ha^{-1}$)	4.37	Square feet per acre
Cubic meters per hectare ($m^3 \cdot ha^{-1}$)	14.29	Cubic feet per acre
Trees per hectare (tph)	0.405	Trees per acre
Degrees Celsius (°C)	1.8 °C + 32	Degrees Fahrenheit

Diameter at breast height is abbreviated as **dbh** throughout the proceedings.

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