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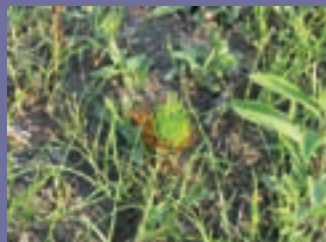


Integrated Restoration of Forested Ecosystems to Achieve Multiresource Benefits:

Proceedings of the 2007 National Silviculture Workshop

May 7-10, 2007

Ketchikan, AK



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ABSTRACT

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A primary mission of the U.S. Department of Agriculture Forest Service is multiple resource management, and one of the emerging themes is forest restoration. The National Silviculture Workshop, a biennial event co-sponsored by the Forest Service, was held May 7-10, 2007, in Ketchikan, Alaska, with the theme of "Integrated Restoration of Forested Ecosystems to Achieve Multiresource Benefits." This proceedings presents a compilation of state-of-the-art silvicultural research and forestry management papers that demonstrates integrated restoration to yield multiple resource benefits. These papers highlight national perspectives on ecosystem services, forest restoration and climate change, and regional perspectives on forest restoration and silvicultural practices to achieve multiple resource benefits from researchers and forest practitioners working in a broad array of forest types in the United States.

PREFACE

The National Silviculture Workshop is a biennial event sponsored by the U.S. Department of Agriculture Forest Service that brings together silviculturists, forest practitioners and researchers in the Forest Service to discuss emerging issues for forest management. In 2007, the workshop was held May 7-10, in Ketchikan, Alaska. The Washington office Forest Management and Vegetation Management and Protection Research Staffs, the R-10 Alaska Regional Office, the Tongass National Forest, and the Pacific Northwest Research Station (PNW) jointly hosted the workshop. The theme for the 2007 workshop was “Integrated Restoration of Forested Ecosystems to Achieve Multiresource Benefits” and the primary goal of the workshop was to present and discuss new ideas in silviculture and forest management that produce multiple resource benefits.

The 2007 National Silviculture Workshop included broad national-scale topics including ecosystem services, frameworks for restoration, climate change, and historical perspectives on restoration. Examples of these national perspectives included a keynote address by USFS Associate Chief Sally Collins on ecosystem services, and presentations on frameworks for restoration, key concepts of sustainability, and restoration efforts in a changing climate. The workshop also included regional perspectives from researchers and silviculturists working in numerous forest types and regions of the United States. Topics included restoration of old-growth forests, increasing biodiversity, improving wildlife habitat, fuel reduction, and forest insect, disease, and health issues. The presentations and proceedings papers are examples of state-of-the-art research and application in different regions of the country. These papers demonstrate the increasing importance of collaborative efforts and successful partnerships between Research and Development, National Forest System, and external partners that promote active management to achieve multiple resource forestry management objectives.

The 4-day workshop included 3 days of presentations and poster sessions and a field trip and overnight stay on Prince of Wales Island to visit natural and managed forests, cultural activities, and a sawmill. Speakers were invited to submit papers for publication, and all submitted papers were peer-reviewed by at least two referees. These proceedings include 22 papers and 8 poster abstracts from the National Silviculture Workshop.

Robert L. Deal
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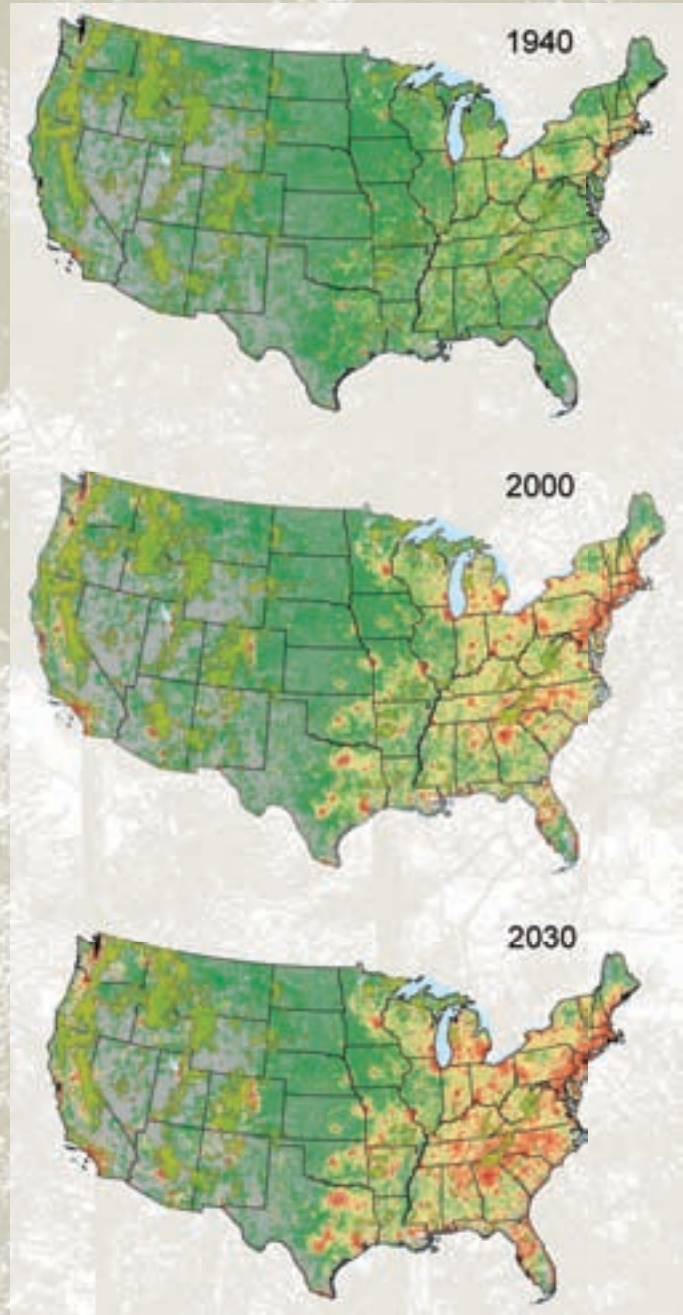


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National Perspectives on Forest Restoration

Proceedings of the 2007 National Silviculture Workshop

Caring for Our Natural Assets: An Ecosystem Services Perspective

Sally Collins and Elizabeth Larry

ABSTRACT

Global attention to climate change has advanced an awareness of human impacts on the environment. Progressing more slowly is recognition of the critical link between forest ecosystems and human welfare. Forests provide a number of societal benefits or ecosystem services, such as water purification, climate and flood regulation, recreational opportunities, and spiritual fulfillment. This paper examines an emerging perspective that describes ecosystems as natural assets that support human health and well-being. The perspective serves as both a conservation approach and an extension of ecosystem management, involving the

connection of ecosystem services to the people who benefit, in some cases with an assigned market value. We argue that the emergence of an ecosystem services perspective is timely as public interest in the state of the environment increases and natural resource managers face the reality of rapid forest ecosystem change. Forest conservation that considers the supply and delivery of ecosystem services will enhance the health and resiliency of ecosystems, engage and serve a broader public, and attract private investment and leadership in a common effort to safeguard natural systems.

Keywords: ecosystem services, ecosystem management, natural capital, climate change, human well-being.

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INTRODUCTION

The City of Los Angeles was one of the first cities in the United States to address air quality concerns. Pressured by citizens to recognize the harmful health effects of smog, the city formed the Los Angeles Air Pollution Control District in 1947 and began a public campaign to control industrial emissions. In 1959, the State of California became the first state to establish air quality standards for public health (California Air Resources Board 2007). At the same time, California state legislators were refining pesticide regulations in response to an increased use of pesticides and a growing awareness of their toxicity. Just a few years later Rachel Carson completed her landmark text *Silent Spring*, exposing the environmental and human health risks of pesticides.

For many, the publication of *Silent Spring* marked the beginning of America's modern environmental movement, leading to the first Earth Day in 1970 and a series of federal regulations for environmental protection. Carson's compelling case successfully captured the attention of the American public. She instilled a broad awareness of human life as part of the natural environment, portraying both as interconnected and equally vulnerable to the side-effects of technological progress.

Today we are witnessing the stirrings of a new environmental movement related to human health and the environment. Climate change is the focus. The potential impacts of rising temperatures and sea levels around the world are stimulating public discourse and political action on a global scale; the energy around the issue reminds us of the urgency generated by *Silent Spring* and the environmental spirit that continued to shape the movement of the 1970s. Today forests are part of the conversation. Forests are a key player in climate regulation, but more notably, forests have become a symbol of popular environmentalism and sustainability, their image displayed in part on any "green" corporate strategy or "environmentally friendly" product.

The climate change issue will continue to mobilize forest awareness and conservation efforts across the globe. As society begins to internalize the costs of unsustainable development into the future, natural resource managers have a story to tell—a story about managing and investing

in forests for the continued supply and delivery of ecosystem services.

Indeed, forests as healthy, functioning ecosystems support life and human welfare, providing important services in addition to storing carbon, such as water purification, erosion control, natural hazard regulation, and spiritual and cultural fulfillment.¹

It is this critical relationship between forests and people that can lead natural resource managers to a conservation approach or perspective that is inherently tied to the dynamic needs of society. An ecosystem services perspective involves measuring the flow of ecosystem services across a landscape and connecting these services to the people who benefit. The approach is forward looking, as conservation and restoration objectives focus on the ability of ecosystems to adapt to change and to continue to supply benefits. It helps forest management remain relevant in a time of growing concern about environmental change; forest management that considers water quality, carbon sequestration, ecotourism, bioenergy, and other benefits engages and serves a broad public. Finally, an ecosystem services perspective involves the private sector in finding ways to value and conserve ecosystems. The emergence of markets for new environmental assets, for example, presents financial opportunities for working forests that extend beyond traditional forest products. Market-based approaches to conservation have the potential to provide landowners with additional incentives to manage and continue owning forest land.

¹ Ecosystem services are commonly defined as the benefits people obtain from ecosystems. A review of the definition and its history is offered by the Millennium Ecosystem Assessment (2003) and Mooney and Ehrlich (1997).

ECOSYSTEM CHANGE AND THE LOSS OF SERVICES

For the City of Los Angeles and for Rachel Carson, the sources and impacts of smog and pesticides seemed clear and the solutions straightforward: regulated behavior would ameliorate human health effects. Today we are aware of a much more complicated picture of ecosystem change, a global network of interrelated drivers peppered with uncertainty (fig. 1).

Climate scientists agree that human activities have led to elevated atmospheric concentrations of carbon dioxide and other greenhouse gases that cause global warming, and observed concentrations are projected to increase (IPCC 2007b). Why such concern? Climate change can intensify the risk of abrupt ecosystem change for terrestrial and marine systems, affecting ecosystem structure, function, and productivity. Such change will substantially impact freshwater resources, food supplies, biodiversity, and other ecosystem services at significant social and economic cost (IPCC 2007a, MA 2005, Stern 2006).

Regional climate changes and climate variations over the past century have already had a measurable impact on our natural systems, including changes in hydrology, species distribution and range, population sizes, the timing of life cycle events, and, especially in forest systems, an increase in the frequency of fire as well as pest and disease outbreaks (Brown et al. 2004; IPCC 2007a; Logan and Powell, in press; MA 2005). Globalization and associated changes in trade patterns have contributed to some of these trends, particularly through the spread of invasive species, as have past land management practices, which have contributed to forest health problems related to fire and fuels.

Land use change is an immediate issue throughout the United States, which is experiencing a loss of privately owned forest land owing to conversion to developed uses. Over 11 percent—approximately 44.2 million acres (17.9 million hectares)—of the Nation's private forests are likely to see dramatic increases in housing development by 2030 (Stein et al. 2005). This projection doesn't take into account the already fragmented woodlots in and around urban areas that are subject to local development pressures as cities expand. Indeed, urban land in the contiguous United States is expected to nearly triple over the next

several decades, an increase in area larger than the state of Montana (Nowak and Walton 2005). Housing growth is also a key concern across the rural landscape, where rural sprawl or exurban development affects a much larger area, amplifying environmental impacts (Radeloff et al. 2005) (fig. 2). Expanding urbanization and rural sprawl affect the Nation's private forests, which compose nearly three-fifths of all forest land, as well as public lands and public land management. Forest-land conversion is a conservation challenge across a mixed-ownership landscape, impacting water quality, wildlife diversity, forest health, recreational access, and the many other benefits of open space.

The trends are complicated by their interaction at multiple temporal and spatial scales: changes in climate can affect land cover and use, for example, and changes in land cover and use will, in turn, affect climate variability (Loveland et al. 2003). It is clear, however, that, together and individually, these drivers of ecosystem change directly affect the supply and delivery of ecosystem services to the United States population and the international community. When forest land is developed or degraded we lose a range of goods and services provided, further increasing pressures on preserved areas to deliver the benefits lost.

AN ECOSYSTEM SERVICES PERSPECTIVE

As population, income, and consumption levels increase, humans put more and more pressure on the natural environment. In 2005, the United Nations commissioned a study of the extent to which human activities have altered ecosystems around the globe. Known as the Millennium Ecosystem Assessment (MA), the study catalogued and evaluated the status of a range of ecosystem services (fig. 3). The MA framework includes the most basic services from nature—provisioning services like the delivery of food, fresh water, wood and fiber, and medicine—and services that are less tangible and harder to measure but equally as critical, such as regulating, supporting, and cultural services. The MA scientists found that 60 percent of the world's ecosystem services are currently being degraded or used unsustainably; 70 percent of the regulating and cultural services evaluated in the assessment are in decline. The assessment predicts that the degradation of ecosystem services might significantly worsen during the first half of this century, substantially affecting human well-being.

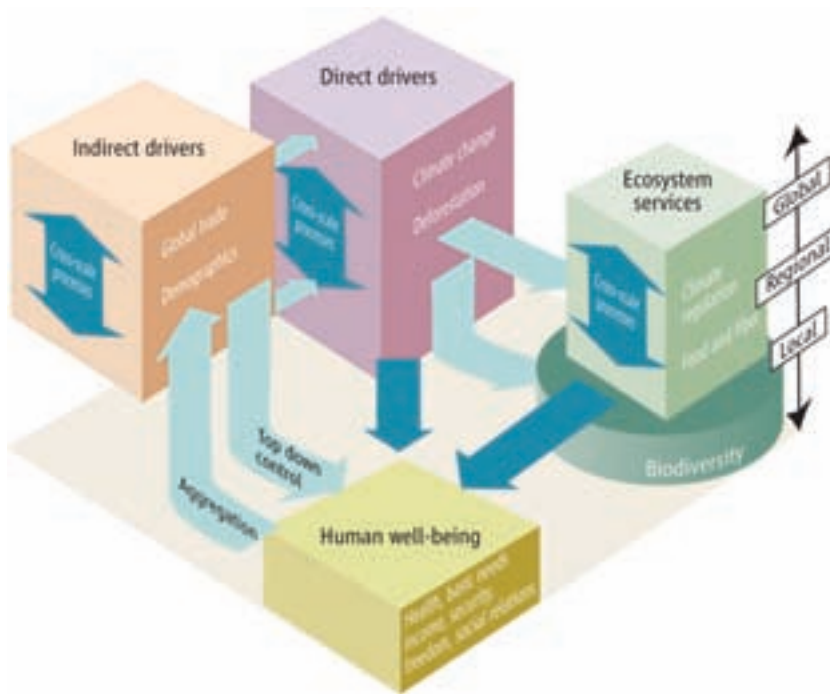


Figure 1—A conceptual framework of interactions between biodiversity, ecosystem services, human well-being, and drivers of change (MA 2005), modified by Carpenter et al. (2006) to illustrate connections among local, regional, and global scales.

ECOSYSTEM SERVICES	
	Provisioning Services Food (crops, livestock, wild foods, etc...) Fiber (timber, cotton/hemp/silk, wood fuel) Genetic resources Biochemicals, natural medicines, pharmaceuticals Fresh water
	Regulating Services Air quality regulation Climate regulation (global, regional, and local) Water regulation Erosion regulation Water purification and waste treatment Disease regulation Pest regulation Pollination Natural hazard regulation
	Cultural Services Aesthetic values Spiritual and religious values Recreation and ecotourism
Supporting Services Nutrient cycling Soil formation Primary production	

Figure 3—Ecosystem services classified by the Millennium Ecosystem Assessment (2005). The assessment evaluated the global status of provisioning, regulating, and cultural services.

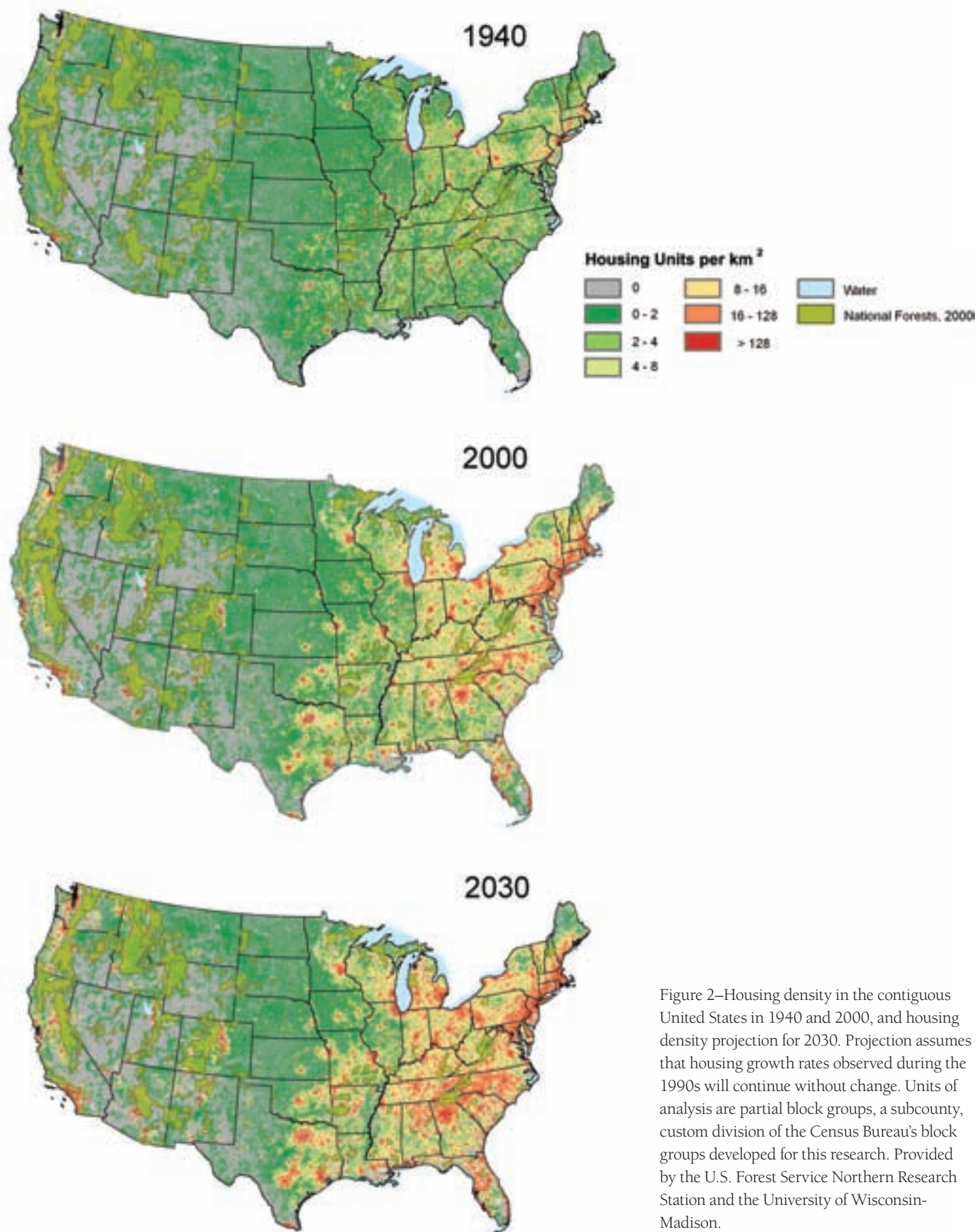


Figure 2—Housing density in the contiguous United States in 1940 and 2000, and housing density projection for 2030. Projection assumes that housing growth rates observed during the 1990s will continue without change. Units of analysis are partial block groups, a subcounty, custom division of the Census Bureau's block groups developed for this research. Provided by the U.S. Forest Service Northern Research Station and the University of Wisconsin-Madison.

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The MA framework (2003) provides a new lens through which to check the state of the environment, one that rests on “human livelihoods, health, and local and national economies” (p. 49). The findings raise important questions for natural resource managers: **Are we adequately conserving the world’s ecosystems? How can we keep pace with the growing pressures of human populations? How can forest management secure ecosystem services into the future?** We have moved into a new century with a set of conservation challenges that together seem unprecedented. We need to enhance our ecosystem management approach accordingly, with a new logic and a fresh, forward-looking perspective that can meet these challenges.

Interpretations of ecosystem management evolved throughout the early to mid-1990s in response to a continuing loss of biodiversity (Grumbine 1994). For national forest managers, ecosystem management emerged as a new approach to multiple-use, sustained-yield management that incorporated the public’s changing desires and needs. Resting on the concept of sustainability, ecosystem management is described as the optimum integration of human needs and requirements, the ecological potential of a landscape, and economic and technical considerations (Jensen and Everett 1994, Zonneveld 1988). The main principle, then, is to sustain the integrity of ecosystems (i.e., ecosystem functions, composition, and structure) for future generations while providing immediate goods and services to an increasingly diverse public (Jensen and Everett 1994, Overbay 1992).

Grumbine (1994: 34) argued that ecosystem management is an early stage in a fundamental reframing of the role of humans in nature. How, then, does an ecosystem services perspective advance this thinking? An ecosystem services perspective encourages natural resource managers to extend the classification of “multiple uses” to include a broader array of services or values; managing for water, wildlife, timber, and recreation addresses the need to sustain “provisioning” services, but land managers are also stewards of regulating, cultural, and supporting services, all of which are critical to human health and well-being.

An ecosystem services perspective encourages natural resource managers to consider the following:

Managing natural capital

Ecosystem services make up our natural life support system and are a form of natural capital. Like financial

capital, manufactured capital, and human capital, natural capital is a requisite for economic progress and human welfare (Hawken et al. 1999). Yet natural capital is usually absent from government or corporate balance sheets; in some cases, it can take the form of a liability. An ecosystem services perspective leads natural resource managers to regard landscapes as natural capital and to account for the assets they are managing. Accounting for natural assets requires measuring the stocks and flows of ecosystem services (and their indicators or appropriate surrogates) and making sure the people who rely on these assets know their value and the cost of losing them. Information relating to the status, trends, and, to the extent possible, the economic worth of ecosystem services can better inform policymakers and the public. Understanding the extent to which a forested landscape purifies the air or moderates coastal flooding, for example, can lend priority to regional investments in land management and conservation.

Urban forestry specialists have pioneered this ecosystem services approach. In urban forestry research, management, and communications, trees are regarded as natural assets that contribute to energy savings, better air and water quality, reduced stormwater runoff, local climate moderation, increased property values—even reductions in city crime and personal stress. Empowered by this information, municipalities across the country are setting tree canopy goals and investing in tree planting efforts to enhance public benefits.

Connecting ecosystem services to the people who benefit

Ecosystem management accommodates human values and uses, but management goals are structured around the protection of ecosystems. Common themes include maintaining viable populations of native species, representing native ecosystem types across their natural range of variation, and maintaining the evolutionary potential of species and ecosystems (Grumbine 1994). From an ecosystem services perspective, by contrast, management objectives are motivated by the supply and delivery of ecosystem services. Ecosystem functions are associated with a set of life-supporting services valued by humans, and management activities are designed to maintain or enhance these services. Measures of ecosystem health, then, extend beyond forest condition to incorporate the ability of an ecosystem to deliver services to a changing population.

In practice, managers first identify the ecosystem services provided by a landscape, then assess human use and dependency on these services at local, regional, and global scales (Heal et al. 2001). Key management questions driving this process are: **Who benefits from these services? Have they identified themselves as stakeholders? Are they aware of the value of these services? Are there services at risk or in decline? How can we prevent their degradation? What are the management tradeoffs?** Answering these questions across a landscape engages a broader set of disciplines, stakeholders, and decisionmakers.

Anticipating future change

Forest management traditionally assumes an historical perspective. Forestry models are based on former conditions and assumptions, and although management objectives address a desired future condition, they are often driven by past realities. The emergence of ecosystem management reflected a growing awareness of landscapes, dynamic processes, multiple objectives, and adaptive response. But the tendency to mimic historical processes and patterns remains, and management goals for healthy, functioning forests rarely incorporate change. An ecosystem services perspective leads managers to focus on a future landscape in recognition that human needs are increasing, historical patterns are being disrupted, and natural processes are challenged by climate change. Conservation aims shift to address the ability of forested landscapes to adapt to change and continue to provide ecosystem services. Within this context a “restored” ecosystem might not mirror the original landscape, but it will be a healthy, productive system capable of meeting societal needs for a broad array of ecosystem services (MA 2005).

Managing for an uncertain future is a difficult task. Resource managers and decisionmakers who take ecosystem services into account must rely on scenario building, assessments of risk and ecological tradeoffs, economic valuation, and other methods of managing uncertainty (Carpenter et al. 2006, Heal et al. 2001, MA 2005). An ecosystem services approach underscores the importance of assessing alternative management strategies and takes demographic, economic, sociopolitical, and cultural factors into account, in addition to direct drivers of ecosystem change (Carpenter et al. 2006).

Embracing a new language

Natural resource managers now have a new language to describe the benefits of forest management—a language that helps them connect a changing population to the land and the services it provides. Too often resource specialists rely on the technical language of their profession to describe their work. Technical language enhances professional credibility and enables research and application, but it can also be highly exclusive. An ecosystem services perspective moves land managers to frame a purpose that reflects a broader set of values, with greater potential to resonate with the public. It helps build bridges of understanding between different interest groups.

INVESTING IN NATURAL CAPITAL: MARKET-BASED APPROACHES TO CONSERVATION

The ecosystem services framework developed as part of the MA is effective. It explains the full extent to which people depend on healthy ecosystems—and how much they take for granted. Whereas provisioning services are valued by society—they are, for the most part, measured, counted, and fiscally inventoried—the rest are typically absent from conventional accounting. By default, regulating, supporting, and cultural services are public goods or common resources, in most cases considered free and limitless.

Without market data or evaluation in monetary terms, the contribution of these services is often misrepresented or ignored in policy formulation and decisionmaking. Measures of economic progress and wealth do not take natural assets into account, let alone the costs of environmental degradation. A devastating oil spill, for example, will increase a Nation’s gross domestic product (GDP) because each of the monetary transactions involved in its cleanup equate to positive economic growth. Similarly, an increase in a developing country’s agricultural exports may register as a growth in GDP, but social costs associated with the local loss of these goods are not weighed, nor is the decline in regulating and cultural services caused by the land conversion required to increase production.

Economists have long studied how to assign monetary value to public goods in an effort to account for environmental externalities. Economic approaches to valuing the

environment form the backbone of any natural resource economics text-book. But nature is priceless, some say; how can we capture its full value? Nature is priceless indeed; but unless a monetary value can be assigned, the importance of flood regulation, the role of nutrient cycling, or the restorative power of a scenic vista might be lost. Without investments in natural capital, life support systems are at risk. Nowhere is this clearer than on private lands, which account for almost 60 percent of the Nation's forests and are critical to the supply of ecosystem services. Because ecosystem services aren't valued financially, private forest landowners lack many incentives and resources to consider them in land use decisions (Kline et al. 2004). The result can be poor forest management, or forest-land conversion to developed uses.

The solution is not simple or clear, nor is it immediate. To help slow the loss and degradation of ecosystem services, economic and financial motivations need to incorporate a conservation objective. New technologies and new business models are needed to help integrate environmental goals into decisionmaking. Economic norms and accounting measures must be broadened. A necessary step is to align individual incentives with the collective interest.

In response to these needs, market-based conservation attempts to capture the value of natural capital and make land stewardship profitable. Markets for ecosystem services connect natural assets to beneficiaries who are willing to pay for their stewardship. In many cases, investments in ecosystem protection are more cost-effective alternatives to building new, or improving existing infrastructure designed to meet the same societal goals.

The carbon market is one example of associating a monetary value with nature's services. Other ecosystem service markets for wetlands, water quality, endangered species habitat, ecotourism, and bioenergy are also evolving—and with them, a chance to supplement traditional forest revenues and promote sustainable management, especially when used together with other conservation tools.

Although traditional conservation approaches have brought us far in safeguarding landscapes and biodiversity, we need to look beyond our own circle to find new tools, stakeholders, and environmental leaders. An ecosystem services perspective encourages us to be creative and

entrepreneurial in bringing people together to design effective solutions.

There is no better time. Growing concern about climate change is inspiring a renewed public awareness of nature and its connection to human health, an awareness characterized in part by business action and involvement. Companies are beginning to recognize the link between environmental health and their business interests; in some cases, it is their shareholders or customers who seek a corporate environmental pledge. Markets for ecosystem services have the potential to seize the enthusiasm, catalyze private investments, and organize community leadership around a shared conservation objective.

CONCLUSION

Natural resource managers have long recognized the fundamental link between nature and society. As early as 1864, George Perkins Marsh explored the causal effects of human-induced deforestation on regional climate in his seminal work, *Man and Nature*. But we haven't always articulated the returns from investing in nature's services, or the cost of losing them. We haven't explicitly served as natural asset managers. As populations develop and prosper, the full weight of human impact on Earth's resources becomes more apparent, as does the importance of managing and accounting for these resources as natural capital that supports human well-being. Connecting ecosystem services with the people who benefit can help us manage our resources more effectively and prevent their decline.

Public interest in the state of the world's resources is on the rise, and the emergence of an ecosystem services perspective could not be timelier. We have new tools for valuing ecosystems, a new language to help us impart the benefits of forest management and conservation, and, most importantly, a restored purpose for serving as nature's stewards.

Forest Service Snapshot

Ten things the U.S. Forest Service can do in the spirit of ecosystem services. Adapted from Associate Chief Collins' opening remarks at the National Silviculture Workshop in Ketchikan, Alaska (May 7, 2007)

1. Bring certainty to the ecosystem marketplace. Uniform standards, established baselines, risk mitigation, and early demonstration can accelerate and lend credibility to emerging ecosystem service markets. Our effort to help shape the 2007 Farm Bill to facilitate market-based conservation is one step in this direction.
2. Provide the most reliable and trusted information on forests for all audiences. We are helping to reconnect children to nature through education and involvement; at the same time we're informing policymakers of the positive role that forestry can play in a climate change strategy.
3. Experiment and learn on the national forests. National forest land serves as a natural laboratory for testing ideas. Demonstration projects can serve as a resource as we help private landowners benefit from market-based conservation.
4. Become market savvy. New markets for ecosystem services require an understanding of how natural assets can be enhanced through forest management. We also need to know how the demand side works—how we can attract investments in conservation and connect conservation buyers to land stewards.
5. Reduce our environmental footprint and be the environmental leader we expect others to be.
6. Lead in research that can answer critical questions about climate change and about carbon sequestration and other ecosystem services.
7. Refresh our language. Much of our vocabulary came from the production forestry era; our words often reinforce practices and a mindset that might need to evolve.
8. Rethink forest plans—what goes into them and how we consider them. Do we need to build climate change scenarios? How can we incorporate a management approach that sustains the flow of ecosystem services across the landscape?
9. Resist the impulse to jump on the ecosystem services “bandwagon” without some thinking—and resist the impulse to dismiss the ecosystem services concept as the latest in a series of attempts to redefine forestry. At the very least, we can appreciate the dialogue that the concept is stimulating.
10. Learn as much as we can. Read, share, and connect with the issues in forestry today, and consider how they relate to the agency's mission and each of our own individual programs. Learning is a key part of our work—and we have important work to do.

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A Framework for Restoration in the National Forests

Thomas R. Crow

ABSTRACT

Ecosystem restoration is an overarching and unifying theme for many Forest Service programs, e.g., invasive species; recovery from fire, hurricanes and other catastrophic disturbances; wildlife and fish habitat improvement. Yet the agency lacks consistent policies, definitions, and procedures regarding restoration. To address this need, a restoration framework was created. Two fundamental principles pervade the framework: (1) because restoration needs reflect diverse public values and transcend property boundaries, ecosystem restoration is based upon collaboration with the public and our partners; (2) scientific knowledge is essential to effect ecosystem restoration and necessarily serves as its basis. Although there are many definitions of restoration,

no single definition fully captures the concept. To supplement a definition, the framework team drafted guiding or operational principles that stress the dynamic or changing nature of ecosystems and stress the importance of considering restoration at both local and landscape levels. The science and practices of restoration are moving away from looking backward to looking forward. What conditions do we want for the future? Applications are also moving away from creating previous states and moving toward restoring ecological processes that create healthy, productive, and diverse ecosystems. These trends support the mission of the Forest Service.

Keywords: ecosystem, Forest Service, policy, restoration, science, sustainability

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INTRODUCTION

Ecosystem restoration is an overarching and unifying management objective for many Forest Services activities (Bosworth and Brown 2007). It is integral to dealing with invasive species where they threaten native biodiversity. Recovery from catastrophic fire and major disturbances such as wind or ice storms involve restoration. Projects that improve wildlife and fish habitat are restoration projects. In fact, almost any project that is conducted on-the-ground can involve restoration in part or as a whole.

The importance of restoration to a resource management agency such as the Forest Service is reflected in recent public statements by its leaders. In a September 2006 interview published in the *Summit Daily News* in Colorado, Chief Dale Bosworth noted that “restoration and recreation have replaced timber harvesting as the defining activity of the agency.” Likewise, he cited ecosystem restoration in an April 2006 Earth Day speech entitled *In the Spirit of Earth Day: Connecting People to the Land* presented at the University of California, Berkeley. In the speech, Bosworth said:

“Our focus today is on restoring and maintaining the ability of ecosystems to furnish services that people want and need. We are restoring ecosystems of all kinds, from damaged salmon and trout streams, to upland meadows and tallgrass prairies, to rangelands choked by invasive weeds, to wetlands along streams and lakes, to degraded pine and oak savannas and woodlands. Where ecosystems are in trouble, our role is to restore them to health.”

In addition to public statements made by agency leaders, the role of restoration is defined by various legislative and administrative initiatives. The National Fire Plan is one such example. It was introduced in August 2000 as a joint Departments of Agriculture and Interior response to severe fire seasons with the intent of providing essential technical and financial resources to support wildland fire management across the United States. One provision of the Plan is to conduct emergency stabilization and rehabilitation activities on landscapes and communities affected by wildland fire. Explicitly recognized as part of these emergency stabilization and rehabilitation activities are “fish and wildlife habitat restoration, invasive plant treatments, and replanting and reseeding with native or other desirable vegetation.”

The Healthy Forests Initiative (HFI) was initiated by President Bush in August 2002 in order to reduce the risks to people, their communities, and the ecosystems on which they depend from catastrophic fires. Much of the focus in the HFI is directed toward removing hazardous fuels through mechanical thinning and prescribed fire. Many of these activities can be considered restoration. The legislative complement to the HFI is the Healthy Forests Restoration Act of 2003 in which the word “restoration” is explicitly recognized in the title and where a section of the legislation deals specifically with protecting, restoring, and enhancing forest ecosystem components.

If the Chief’s comment “we are restoring ecosystems of all kinds” is correct, then many managers in the Forest Service must be practicing restorationists. Any activity that is aimed at improving forest health or enhancing native biodiversity or improving terrestrial or aquatic habitat can be arguably called restoration. But despite the aforementioned legislative and the many administrative initiatives and despite a history of practical experience, many questions remain about the concept and the practice of restoration (Wagner et al. 2000, Harris et al. 2006). If you have a conversation about restoration with a resource manager, you are likely to hear: Exactly what do we mean by restoration? Restore to what? What does restoration mean in light of global climate change? How is success defined with restoration projects? What is the balance between rebuilding past systems and attempting to create resilient systems for the future? How are the stakeholders identified and how do we involve them in restoration activities? Does the public support restoration?

These and related questions prompted the Forest Service to charter a Restoration Framework Team to provide recommendations for improving the agency’s ability to restore ecosystems on the national forests and grasslands and to provide a cohesive set of policies, definitions, and guidelines for restoration that can be applied at the national level. The purpose of this paper is to explore the commonly asked questions using the experience of the Framework Team and their recommendations provided in their January 2006 document *Ecosystem Restoration: A Framework for Restoring and Maintaining the National Forests and Grasslands*.

THE DEFINITION PROBLEM

The first challenge when considering the concept of restoration is to define it. For starters, The *American Heritage Dictionary* defines the word “restoration” as “the act of putting someone or something back into a prior position, place, or condition.” This concept of returning to a prior condition causes an endless discussion about the legitimacy much less our ability to restore dynamic systems to a prior state in a changing world. Among the many definitions that are available in the literature, perhaps the most widely accepted is that provided by the Society for Ecological Restoration – restoration is the process of assisting the recovery of an ecosystem that has been degraded, damaged, or destroyed (www.ser.org). Here the emphasis is placed on the recovery of an ecosystem without stating explicitly what is being recovered (i.e., composition, structure, function), what benchmarks should be used, or what the endpoint could or should be. It is worth noting, however, that in the recent scientific literature there is a trend towards emphasizing the restoration of ecological processes that produce healthy ecosystems as opposed to recreating previous states as measured by composition (Ehrenfeld 2000). The Society for Ecological Restoration’s definition also contains some value-laden terms – degraded, damaged, or destroyed – that cause concern among many managers. These terms are often viewed as indictments of past management practices. Alternative language such as “transformed” has been suggested, but each change brings new ambiguities.

It is perhaps naive to think that a single definition can fully capture a complex concept such as ecological restoration. After a survey of published definitions, members of the Restoration Framework Team decided to describe the concept through a set of guiding principles (Table 1). They amount to a mix of operational and defining principles that recognize ecosystems as dynamic and changing systems in which ecosystem processes are emphasized with realistic goals that have public support. In total, these “principles” provide a more complete picture of what restoration means to a resource management agency such as the Forest Service.

RETURNING TO THE FUTURE

The frequently asked question – restore to what? – begs another question. What is an appropriate reference condition or benchmark for ecosystem restoration? Several approaches have been used. A common benchmark is to use the historic range of variation to define the “boundary conditions” for restoring ecosystem composition and structure (Landres et al. 1999). So-called “natural areas” such as botanical areas, wilderness areas, and Research Natural Areas can serve as useful baselines to guide restoration of similar but damaged ecosystems. There are concerns related with both approaches. Among these are:

- Reference conditions are not always available.
- Reference conditions stress a single point in time (static vs. dynamic systems).
- Both approaches emphasize returning to the past as the term “restoration” suggests as opposed to managing for the future.

Table 1. The top 10 principles for implementing restoration projects (adapted from the Ecosystem Restoration: A Framework for Restoring and Maintaining the National Forests and Grasslands).

1. Seek and set goals for restoration that reflect societal choices through public involvement.
 2. Make operational decisions at the lowest levels in the organization.
 3. Consider restoration at multiple spatial scales.
 4. Stress restoring ecosystem processes that create healthy ecosystems.
 5. Establish restoration objectives and measure responses for the long term.
 6. Recognize that ecosystems are dynamic and that change is inevitable; avoid “static endpoint thinking.”
 7. Use multiple sources of relevant information, such as historical records, scientific studies, practical experience, and indigenous knowledge.
 8. Deal with uncertainty by using adaptive approaches to restoration and expect adjustments.
 9. Design and implement monitoring as part of restoration.
 10. Be humble and learn as you go.
-

- The concept of historic range of variation is too poorly defined or understood in most cases to provide useful guidance for applications.
- Past conditions may not be the appropriate endpoint for restoration but other endpoints might be more appropriate given climate change, species extirpation, invasive species, and social or economic factors such as patterns of land ownership or wildland-urban interface concerns.

When reference conditions are available, they provide valuable information about the magnitude and rate of change in ecosystem properties that may be useful in guiding restoration activities in similar but damaged ecosystems. Even so, recovering the full complement of components and functions from previous states is rarely possible due to the changing climate, altered disturbance regimes, species extirpation, and invasives.

MOVING FORWARD THROUGH RESTORATION

In his book *1491: New Revelations of the Americas Before Columbus*, Charles Mann (2005) argues that indigenous people in the Americas were here far longer, in much greater numbers, and created far more complex and advanced societies than is suggested in many of our history books. The impacts of their collective interventions on the landscape in the “New World” were both extensive, and in some locations, intensive. In reality, people have managed the American landscape, both North and South, for thousands of years. Mann writes:

“Native Americans ran the continent as they saw fit. Modern nations must do the same. If they want to return as much of the landscape as possible to its state in 1491, they will have to create the world’s largest gardens. Gardens are fashioned for many purposes with many different tools, but all are collaborations with natural forces. Rarely do their makers claim to be restoring or rebuilding anything from the past; and they are never in full control of the results. Instead, using the best tools they have and all the knowledge that they can gather, they work to create future environments.”

The science of restoration is moving toward restoring ecosystem processes that create diverse, productive, and above all, healthy forest, grassland, and aquatic ecosystems (see for example, Covington et al. 1997, Allen et al. 2002).

This emphasis is consistent with the Forest Service’s mission “to sustain the health, diversity, and productivity of the Nation’s forests and grasslands to meet the needs of present and future generations.” By combining this forward looking view of restoration with an emphasis on sustaining ecosystems and the social institutions they support, we are “shaping a world to live in for the future” (Mann 2005).

IS IT SCIENCE OR ART?

As mentioned in the Chief’s Earth Day speech, the focus in today’s Forest Service is on restoring and sustaining the ability of terrestrial and aquatic ecosystems to furnish services that people want and need. Managers in the Forest Service are, in fact, restoring ecosystems of all kinds, including rivers and streams that are losing their capacity to support salmon or trout, western grasslands choked with cheatgrass, riparian areas dominated by salt-cedar, mountain forests that are turning brown due to tree mortality caused by bark beetle epidemics. Just how good is the science to support these efforts?

A query of Forest Service research units in which the word “restoration” appears in the mission statement or a problem statement yields an impressive list (Table 2). The list covers a wide spectrum of disciplines (e.g., genetics, soils, silviculture, entomology, pathology, wildlife, fisheries, and physiology), ecosystems (e.g., western aspen, southern bottomland hardwoods, northern riparian forests, longleaf pine), and geographic locations (e.g., tropical ecosystems in Puerto Rico and Hawaii, agricultural landscapes in the Midwest). A query into the broader academic research community, other agencies, both state and federal, and with the ongoing efforts of NGO’s yields an even richer assortment of research information

There are, however, some obvious gaps in this work. The perspective from the social sciences and humanities are present but have received far less attention compared to ecological considerations. In their book *Restoring Nature, Perspectives from the Social Sciences and Humanities*, Gobster and Hull (2000) illustrate the importance of gaining public understanding, and hopefully, political support for restoration projects when a citizens group called ATLANTIC (Alliance to Let Nature Take Its Course) complained that trees were being cut and fires set in attempts to restore prairies and savannas in the Chicago metropolitan area.

What resulted was a barrage of negative headlines in area newspapers such as “Prairie People Compile Tree Hit List” or how about “Guru’s Forest Restoration Plans Read More Like Destruction.” This is not the type of publicity that makes for successful restoration efforts.

Another obvious need is for restoration research at multiple spatial scales (principle #3 in Table 1). Most of

the research has focused on projects at local levels, while much less is known about their cumulative impacts at the landscape and even regional scales (Hobbs and Norton 1996). The challenges associated with large-scale restoration projects are evident when considering case studies such as attempts to improve water quality in the Chesapeake Bay (Boesch et al. 2001), to restore the Everglades in south Florida (DeAngelis et al. 1998), or to reduce the hypoxic

Table 2. A selection of ongoing Forest Service research projects addressing restoration issues.

Pacific Southwest Research Station, Institute of Pacific Island Forestry (PSW-4154, Hilo, HI)

Restoration of Ecosystem Processes: Researchers are determining how to put tropical forests back on lands that were cleared for agriculture. Contact: Christian Giardina.

Pacific Southwest Research Station, Sierra Nevada Research Center (PSW-4202, Davis, CA)

Researchers are studying techniques and approaches for maintaining and restoring biodiversity in the Sierra Nevada Mountains. Contact: Peter Stine.

Pacific Northwest Research Station, Eastside Forest Health Restoration Team, (Wenatchee, WA)

Research includes landscape assessments of key changes in ecological patterns and disturbance regimes (fire, insect, disease) in eastern Washington and Oregon forests, and work with managers to restore patterns and functionality of wildlife habitats and forest structure, both from existing conditions and after large fires. Contact: Paul Hessburg.

Pacific Northwest Research Station, Forest Genetics Team (Corvallis, OR)

The research areas include studying genetic issues associated with restoring native trees, shrubs, grasses, and forbs. Contact: Brad St. Clair.

Rocky Mountain Research Station; Ecology, Paleoecology, and Restoration of Great Basin Watersheds (RMRS-4252, Reno, NV)

Mission: Increase understanding of the effects of both long-term climate change processes and more recent natural and anthropogenic disturbances on Great Basin ecosystems and watersheds, and use this understanding to devise meaningful scenarios for their restoration and management. Contact: Robin Tausch.

Rocky Mountain Research Station, Aspen Restoration in the Western United States (RMRS-4301, Logan, UT and RMRS-4451, Fort Collins, CO)

Research from a combination of research work units is providing valuable information for restoring aspen ecosystems in the western United States. Contact: Dale Bartos.

Southern Research Station, Center for Forest Disturbance Science (SRS-4104, Athens, GA)

Research is conducted on restoring southern forest ecosystems including longleaf pine, table mountain pine, and bottomland hardwoods. Soil quality and international restoration ecology issues are also considered. Contacts: John Stanturf (bottomland hardwoods, international); Research Ecologists: Ken Outcalt (longleaf pine restoration); Tom Waldrop (table mountain pine); Mac Callaham (soil quality).

Southern Research Station, Restoring and Managing Longleaf Pine Ecosystems (SRS-4158: Auburn, AL, Clemson, SC, Pineville, LA)

Research is directed at developing reliable restoration and management systems for sustaining longleaf pine ecosystems. Investigations range from physiological processes and ecological relationships to silvicultural systems and models for individual tree and stand growth. Contacts: Kris Connor (physiology and ecology), Auburn, AL; Research Ecologist: Dale Brockway (ecology and silviculture), Auburn, AL; Research Plant Ecologist: Joan Walker (ecology), Clemson, SC; Research Plant Physiologist: Mary Anne Sword Sayer (physiology), Pineville, LA; Research Plant Physiologist: Susanna Sung (physiology), Pineville, LA; Research Forester: Dave Haywood (silviculture), Pineville, LA; Research Forester: Jeffery Goelz (modeling, biometrics, silviculture), Pineville, LA.

Southern Research Station, Center for Forested Wetlands Research (SRS-4103, Charleston, SC, Santee Experimental Forest)

Mission: To provide the ecological information necessary to manage, sustain, and restore the structure, function, and productivity of wetland-dominated forested landscapes. Contact: Carl Trettin.

Southern Research Station, Center for Bottomland Hardwoods Research (SRS-4155, Stoneville, MS)

Research includes the restoration of bottomland hardwoods in the Lower Mississippi Alluvial Valley. Contact: Ted Leininger.

Southern Research Station, National Agroforestry Center, Lincoln, NE)

Research is conducted on the effective design of riparian buffers and the transfer of technology to restore watersheds and improve water quality. Contact: Michele Schoeneberger.

Northern Research Station, Ecology and Management of Riparian/Aquatic Ecosystems (NRS-4351, Grand Rapids, MN)

Studies are conducted to understand how restoration influences nutrient and carbon processing, surface water quality, and aquatic biota in northern riparian/aquatic ecosystems. Contact: Randy Kolka.

Northern Research Station, Silviculture and Ecology of Upland Central Hardwood Forests (NRS-4154, Columbia, MO)

Restoration studies are conducted in oak and pine savannas as well as bottomland hardwood forests. Contact: Frank Thompson.

International Institute of Tropical Forestry (Rio Piedras, PR)

Research aimed at accelerating forest recovery on degraded lands and management of secondary forests is being conducted in the Caribbean region, continental areas of Latin America (particularly Brazil) as well as Africa, Australia, and Hawaii through cooperative agreements. Contact: Ariel Lugo.

zone at the outlet of the Mississippi River caused by nutrient inputs from agricultural lands in the basin (Schulte et al. 2006). This lack of knowledge about ecological processes at large spatial scales is one reason why restoration projects at the scale of the Lower Mississippi Alluvial Valley or south Florida have proven to be so challenging. Other

reasons include the lack of coordinating mechanisms for working across multiple ownerships, the lack of economic incentives to engage private landowners, and inadequate funding available for restoring large areas.

There frequently is a sense of urgency relating to restoration projects because practitioners want to move quickly in

order to maintain as many possible options, e.g., restoring critical habitat for a species on the brink of extinction. In these cases, the quality of the work depends on the skill of the practitioners as they rely heavily on their experience. Without question, restoration can be successful without perfect knowledge of the target ecosystem, but there is an inherent weakness to this approach. If the science is moving toward restoring ecological processes but our understanding of these processes is incomplete, how then can success be measured? Fortunately, properties associated with ecological processes such as forest health, terrestrial and aquatic productivity, or ecological diversity are things that managers and scientists have experience in assessing and interpreting. These properties are useful surrogates for the ecological processes that improve health, productivity, and diversity and success can be measured, often times, using relative measures: Is forest health as suggested by tree mortality improving? Is diversity as measured by the number of plant species increasing?

This argument for moving forward with imperfect knowledge does not diminish the need for solid scientific information. Without the fundamental understanding of processes, restoration is limited to a series of unrelated case studies and local applications rather than a more fundamental investigation of underlying principles and mechanisms that support a broader understanding (van Diggelen et al. 2001). Without this scientific underpinning, each project becomes a new and unique challenge. The question posed as the section heading – Is it a science or an art? – is, however, a false dichotomy. As with resource management in general, restoration will always rely heavily on the skill and practical experience of the practitioner as well as the fundamental science that underpins the application. Both experience and research represent invaluable sources of information for restoration (principle #7 in Table 1).

RESTORATION AND WE THE PEOPLE

In their book chapter “The Language of Nature Matters; We Need a More Public Ecology,” Hull and Robertson (2000) pose the question: “Which nature should be restored?” In exploring this question, Hull and Robertson explore three commonly used terms – naturalness, health, and integrity – for addressing their question. There exists, however, a variety or perhaps infinite number of ways to define these

terms and each definition reflects a set of values and norms, either explicitly or implicitly, that create confusion and controversy. The point here is that determining restoration goals and defining the desired outcomes from restoration is best resolved through social discourse and involvement. A successful restoration project requires an expanded view beyond merely applying techniques – it requires social, cultural, political, aesthetic, and moral considerations as well (Light and Higgs 1996, Higgs 1997).

CONCLUSION

If restoration is viewed as part and parcel of resource management, as I assert, then the distinction between the two becomes blurred. Clearly, resource management encompasses more than restoration, but attempts to differentiate between the two are difficult when restoration activities are embedded within the broader framework provided by resource management. The fact that the principles presented in Table 1 apply to a vast spectrum of management activities on public lands supports this argument. Further, many restoration tools, such as widely applied silviculture practices, are already available. Consider, for example, that planting of pine can be accomplished in the context of establishing industrial plantations as well as reestablishing a species eliminated through exploitive harvesting a century ago (Guldin 2007). In this context, restoration involves thinking about using existing tools in new and creative ways as well as developing new tools when needed.

When making this argument, of course, the skeptic will view the restoration framework as nothing more than a ploy to repackage existing Agency programs (i.e., get the cut out) and that this is one more example of placing a new label on old behavior. Indeed, the restoration framework is nothing more than empty rhetoric unless specific operational procedures are defined and applied on-the-ground. Doing so is the challenge facing the Agency as resource managers struggle to meet increasing and often competing demands in a finite world with a changing climate.

Although it may seem counterintuitive, ecosystem restoration should be thought of as a forward-looking concept with emphasis on “shaping a world to live in for the future.” Restoration is less about restoring an ecosystem to a prior condition and more about creating diverse, productive, and healthy ecosystems. This emphasis is consistent with

the Agency's mission "to sustain the health, diversity, and productivity of the Nation's forests and grasslands to meet the needs of present and future generations." If these tenets are accepted, then restoration is about sustaining terrestrial, aquatic, and marine ecosystems and the social institutions they support. After all, they are inextricably linked.

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The Silviculture of Restoration: A Historical Perspective With Contemporary Application

James M. Guldin

ABSTRACT

In the southern United States, the turn of the 20th century saw the high-grading of virgin pine stands that left millions of acres of forestland in desperate condition. Some of these southern pine stands now support thriving forests whose patterns and processes resemble those extant before they were cut a century ago, but others do not. The success of this recovery in the southern pinery was based upon three primary elements. First, the silvics of the species had something to do with the success of their restoration; some of the southern pines have inherent ecological attributes that lend themselves to restoration, and others do not. Second, the plasticity of high-graded stands under the artful hand of the silviculturists of the day was instrumental in the recovery, partly because of the trees, and partly because of the silviculturists. Finally, major advances in silvicultural science provided astounding successes, and sometimes profound malpractice, in enabling or inhibiting the recovery. A qualitative and quantitative silvicultural review of that

history can help modern silviculturists achieve goals of integrated restoration for multi-resource benefits on public and private lands, both regionally and nationally. Key elements for contemporary silviculturists to consider are:

- 1) that restoration of process drives restoration of structure;
- 2) that successful restoration demands that a silviculturist balance the cognitive dissonance between economics and ecology;
- 3) that some tools that traditionally have been associated with intensive forestry for fiber production can help restoration prescriptions succeed at functionally meaningful ecological scale;
- 4) that a diversity of silvicultural practices among stands across a landscape is more robust than a uniformity of practice; and
- 5) that restoration will be easier in some forest types than in others regardless of the silviculturist's efforts.

Keywords: restoration, silviculture, ecology, southern pines, conservation forestry

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INTRODUCTION

During the latter half of the 20th century, timber management was the primary goal of silviculture on public and private lands. Projections in the middle of the last century anticipated society's voracious demands for wood and fiber products, and as a result, foresters on both public and private lands began to develop silvicultural practices of agronomic intensity to meet the silvicultural goal of timber production (Spurr 1979). This was done with extraordinary competence; arguably, the two dominant advances in silviculture in the last half of the 20th century were the development of genetically improved planting stock for conifers and fast-growing hardwoods, and the use of chemical amendments such as herbicides and fertilizers to promote fast growth of the desired species and inhibit the growth of the herbaceous and woody species competing with those species. Furthermore, in an undesirable outcome for the profession, silviculture came to be universally associated with timber management (Guldin and Graham 2007).

A key element to success in using timber management has been the exclusion of extraneous damaging disturbance events from forests being managed for timber production. Key to that has been the control of wildfires, which raged through cutover areas and affected the rehabilitation and recovery of the high-graded stands. Professionals in the early part of the last century were keenly interested in forest protection, and considered wildfire the single greatest threat (e.g., Chapman 1942, Reynolds 1947). After World War II, with the GI Bill promoting college education, the profession of forestry grew rapidly--especially the labor force in professional and technical positions in Federal and State forestry agencies. With this expanded pool of workers, effective fire control finally became possible. And effective it was! The Smokey Bear symbol for prevention of wildfire has become one of the most recognized advertising symbols in the world, and control of fires became, and still is, the rule of the day for Federal and State forestry organizations.

The ability of cutover understocked forest stands in the South to recover was in part the result of extraordinary efforts to implement forest management, with emphasis on fire control and silvicultural interventions to manage what remained—especially during a period such as the

Depression in the 1930s when funds were scarce and labor was inexpensive and readily available. For example, the roads, firelanes, and early buildings for the Crossett Experimental Forest in Ashley County, Arkansas, were built by Civil Works Administration field crews in 1934, working largely with hand tools, supported by the Department of Labor (Figure 1).



Figure 1— FS Photo 350876. Caption: "Typical 20-foot Forest Service road constructed with CWA hand labor. Road to be graveled and used for utilization of forest products as well as for protection and administration. Road #4 looking east from intersection of roads #4 and #6." 23 July 1937, Crossett Experimental Forest, Ashley County, Arkansas. (Photo courtesy of U.S. Forest Service, Southern Research Station)

But recovery was also due in part, and perhaps in large part, to the nature of the forest types that had been exploited. The species mixtures that recovered in these cutover stands depended upon several elements. First, some trees too small for commercial harvest survived the earlier cutting.

Second, the understocked conditions of cutover stands provided conditions suitable for new woody plants, either as sprouts, established seedlings and advance growth, or seed blown in by the wind, disseminated by animals, or otherwise brought to the site. Third, if all else failed, a new stand was established by artificial regeneration. Through these efforts, modified by infinite site-specific variations in local conditions, new forests became established and have developed to maturity. Today, across the South, thriving forests exist where cutover high-graded remnants were once common. There is a lesson in this restoration and recovery during the last century from which we silviculturists in the 21st century can learn.

For ecosystems adapted to fire, this recovery of forest condition toward the primary goal of timber production coupled with the control of fires in the woods created conditions resulting in the foremost management challenge in the 21st century—that of restoring fire to systems that are prone to burn. Fire-adapted ecosystems represent a confluence of vegetation attributes, site characteristics, and local climatic conditions resulting in ecological systems that will burn if a fire source is introduced to them. The increased fragmentation of the forested landscape places human society's infrastructural investments in homes, pasture, and communities at risk in a landscape adapted to fire, and forest stands managed in the absence of fire can be damaged or destroyed if fire occurs in an uncontrolled manner.

Coupled with this is the under-representation of two kinds of ecosystems on the landscape:

(1), ecosystems with large and old trees, since so few of them survived the high-grading at the turn of the last century, and

(2), fire-adapted ecosystems in which fire has been a regular component.

This creates management challenges, since many species of flora and fauna that require old burned ecosystems are also underrepresented on the landscape.

Thus, the rise of the 21st century poses a different set of challenges for silviculturists, especially on public lands but also in part on non-industrial private lands and even some industry lands—to restore underrepresented old, fire-adapted ecosystems in a landscape increasingly fragmented in ownership and condition. In the South, the keystone species for this restoration is the red-cockaded woodpecker

and management of this species is an important objective on forest lands in the South. Equally important ecologically is the creation of the restored habitat that benefits not only the red-cockaded woodpecker, but also a host of flora and fauna adapted to similar habitat conditions.

The tools to achieve this restoration fall squarely in the realm of silviculture. But the success of the restoration has as much to do with the species being managed as it does with using silviculture to create conditions within which the desired species can develop. Interestingly, tools developed for timber production may be important.

To explore these questions, it may be useful to consider case studies from the southern pinery, and adapt lessons learned there to other ecosystems. A subjective case-study analysis of the silvics and silvicultural conditions of three southern pine forest types may have implications and lessons for current efforts in the silviculture of restoration generally. Why focus on the southern pines? Because southern pine-dominated forests grow rapidly, rotations and cutting cycles are short relative to other forest types in the nation; the southern pinery serves as a crucible for silvicultural innovation and the evolution of silvicultural practices in the woodbasket of the nation.

CASE STUDIES FROM THE SOUTHERN PINES

Virgin pine stands across the South were logged from the 1880s through the 1920s. That harvesting and associated disturbance, especially uncontrolled burning, left millions of acres in cutover condition if not completely denuded of trees. Three forest types were harvested with greatest intensity--the longleaf pine (*Pinus palustris* Mill.) forests of the lower Gulf Coastal Plain, the mixed loblolly-shortleaf (*P. taeda* L.-*P. echinata* Mill.) pine forests of the upper Gulf Coastal Plain, and the pure shortleaf pine forests in the Ouachita and Ozark Mountains.

Stands throughout these regions were high-graded of all standing merchantable sawtimber. Cutting rules were simple—essentially, cut all pine trees to a 15-in stump, which translated to about a 12-inch d.b.h. Pines were not cut if they were culls, or below the merchantable threshold. This was the heyday of railroad logging, and the hardwoods (especially the hard hardwoods such as oaks and

hickories) growing in mixture with the pines were often cut for railroad ties, cooperage, box manufacture, or use in chemical distillation. After logging, stands were understocked, with an overstory composition of cull pines and hardwoods, pines below the merchantability threshold, and great piles of logging slash. Fires caused by harvesting activities or by settlers to clear undergrowth to promote grasses for livestock were common in these stands. Few foresters believed that these forests would recover anytime in the foreseeable future.

Yet, some of these southern pine stands now support thriving forests. The recovery and management of the loblolly-shortleaf pine forests of the upper Coastal Plain is an astounding success, with some areas now supporting the South's fourth and fifth forests. Loblolly pine and slash pine (*P. elliottii* Engelm.) have been widely planted across the lower Gulf Coastal Plain, and are a mainstay of industrial timber production across the region. The recovery and management of mountain shortleaf pine has been less effective. Pure shortleaf pine-dominated stands and mixed



Figure 2— Well-stocked longleaf pine stand, Sam Houston Ranger District, National Forests and Grasslands of Texas. (Photo by James M. Guldin)

pine-hardwood stands are widespread and have excellent growth and quality, although the recovery of a minor shortleaf pine component in upland hardwood-dominated stands is subject to debate. But unlike the other pine species in the southern forest landscape, longleaf pine has not recovered. It is now found on just a fraction of the area it occupied a century ago.

The difference in restorability or recoverability in the southern pines is a function of a number of interacting elements--the species composition, the silvics of the component species with respect to their ability to regenerate and to respond to release from suppression, the nature of the logging, and the accidental or deliberate treatment that followed. These three southern pine species and the habits they exhibit in their respective forest types offer an opportunity to speculate about the ease or difficulty of restoration, and to ponder the silvics and silvicultural attributes of each species in a subjective discussion of the potential to respond to restoration under silvicultural interventions of varying degrees.

LONGLEAF PINE ON THE LOWER GULF COASTAL PLAIN

The natural range of longleaf pine extends from southeastern Virginia to eastern Texas, and encompasses Coastal Plain, Piedmont, and mountain sites in pure or mixed stands (Boyer 1990). But longleaf is especially associated with lower Gulf Coastal Plain terrain, where it is a dominant species and at one time covered extensive areas in pure stands.

At maturity, longleaf pine stands compare favorably to other southern pines, especially with respect to straightness, quality of lumber, and yield of sawtimber volume (Figure 2). But of the three major southern pines, longleaf pine has fared the worst in the transition from cutover condition to contemporary status. An excellent summary of the conditions associated with longleaf pine harvest is presented in Earley (2004). For vast areas, cutover longleaf pine stands did not recover from high-grading, primarily because of the regeneration biology and dynamics of the species.

The slow initial establishment of longleaf pine and the difficulty associated with obtaining natural regeneration made the species impractical for management by the forest industry, especially for fiber production on short rota-

tions. Similarly, the difficulty in obtaining seed for longleaf resulted in problems associated with widespread artificial regeneration relative to other species. These issues, together with the pressing need to reforest cutover sites in the west Gulf region, led managers to plant or direct-seed slash and loblolly pine across vast areas of cutover longleaf pine sites. As a result, longleaf pine has suffered dramatic reductions in area, from more than 90 million acres of original forest to barely three million acres today (Landers and others 1995). Today, pure stands of longleaf pine are restricted to small areas along the lower Gulf Coastal Plain, and on Federal lands in Texas and Louisiana.

Longleaf pine may have existed in cutover areas as seedlings in the grass stage in cutover longleaf pine stands (Farrar, personal communication). Longleaf seedlings are difficult to distinguish from grasses (Figure 3); identification during the growing season essentially requires a taste test to identify the pine by its resinous flavor. The intermittent bumper crops of longleaf pine produce high densities of seedlings in clearly-defined age cohorts, an ecological regeneration dynamic of accumulating seedlings similar to that reported for oaks (Johnson and others 2002). After germination, longleaf seedlings remain in the grass stage and gradually increase in root collar diameter over time, under the influence of disturbance and surface fires. If they survive these influences and develop a sufficiently large root system, the seedlings break out of the grass stage and initiate height growth. Thus, it is possible that cutover areas of longleaf might have contained longleaf seedlings, but that those seedlings failed to develop because of the impacts of logging and foraging by feral hogs.

Restoration of longleaf today will be informed by several repositories of silvicultural knowledge. First, the superb



Figure 3— Longleaf pine seedling in the grass stage amid dormant grasses. (Photo by D. Andrew Scott)

work in development of the shelterwood method to regenerate longleaf pine by Southern Station scientists at the Escambia Experimental Forest in southern Alabama united attention to cone crops, understory vegetation control, and the earliest uses of prescribed fire (Crocker and Boyer 1975, Boyer 1979). Keys to the method were retaining overstory trees capable of producing cones, to retain sufficient stocking of acceptable overstory trees to optimize seedfall, and to adequately prepare the site when seedfall was forecast. After germination and establishment, seedlings remained in the grass stage for extended periods of time. Brownspot needle blight (*Mycosphaerella dearnessii* M.E. Barr) was found to inhibit emergence of longleaf pine seedlings from the grass stage, and fires were used to burn off the infected needles. However, the terminal bud of longleaf pine in the grass stage is resistant to mortality by fire because of the insulating nature of the bud scales and the protective needle whorl on the bud.

Secondly, the challenges of reforestation with longleaf have been met, especially in light of containerized planting stock and site preparation treatments that reduce the length of time longleaf seedlings remain in the grass stage (Barnett 2004). Like all southern pines, longleaf trees that have been suppressed but still retain some degree of apical

dominance and suitable crown dimensions can respond to release from competition, even at advanced ages.

However, these excellent silvicultural tools of the trade are not sufficient to restore longleaf stands if longleaf is absent at shelterwood densities or greater in the residual stand. Research in artificial regeneration of longleaf pine has been so successful is in part because it has had to be, in order to effectively develop seedlings that can be competitive and especially that can quickly emerge from the grass stage, thereby enhancing their relative competitive status against competing vegetation.

SHORTLEAF PINE IN THE OUACHITA AND OZARK MOUNTAINS

The natural range of shortleaf pine encompasses 22 states from New York to Texas, second only to eastern white pine in the eastern United States (Little 1971). It is a species of minor and varying occurrence in most of these States, typically found with other pines. But in the Ouachita Mountains of western Arkansas and eastern Oklahoma, and in the Boston Mountains and Springfield Plateau of the Ozark Mountains in northern Arkansas and southern Missouri, shortleaf is the only naturally-occurring pine.

Pine-dominated stands were and still are common in the Ouachitas and parts of the Ozarks (Figure 4).

Descriptions of presettlement conditions in the region point to a forest type that was more open than currently found, in terms of both overstory stem density and understory condition, with anecdotal comments that a person could ride a horse through the woods without losing their hat. That speaks to an openness of two kinds--an open understory condition promoting easy access for horses, and an overstory condition where low-hanging branches are uncommon or easily avoided.

The high-grading harvest of the Ouachita shortleaf is described in extraordinary detail in Smith (1986). Harvests progressed from south to north, and were conducted by running railroads along and through the long east-west valleys and ridges that characterize the region. Shortleaf pine stands came back in pure stands on south-facing Ouachita and Ozark hillsides that were initially dominated by pines, but questions persist whether shortleaf pine has returned as a minor and varying component in pine-hardwood and especially hardwood-pine stands in the Ozarks.

By and large, shortleaf pine has something of a mistaken reputation as a seed producer. It was thought to produce adequate or better seed crops on the order of every 3 to 6

years (Lawson 1990). But more recent research suggests that in the Ouachitas, adequate or better seed crops can be obtained from managed stands of shortleaf pine on the order of every other year (Wittwer and others 2003).

However, shortleaf pine has a unique attribute of seedlings relative to other southern pines first noticed by Mattoon in 1915—the seedlings, if top killed by fire, will resprout from a basal crook. The physiology of this phenomenon is not clearly understood. However, if a fire overruns a stand with shortleaf seedlings and saplings, many



Figure 4— Well-stocked shortleaf pine stand in pine-bluestem habitat restoration management area, Poteau RD, Ouachita National Forest, Scott County, Arkansas (Photo by James M. Guldin)

will resprout, and the conditions of fire scarification will promote a seedbed for new seedlings. The argument can be made that many shortleaf pine stands between 80 and 100 years old in the Ouachitas are probably of coppice origin after surface fires.

In current stands, shortleaf pine can tolerate overstocking, though at a cost of crown development. Perhaps one of the reasons why the species was thought to be a poor seed producer was based on examination of overstocked stands containing small-crowned trees in poor condition to produce an abundant cone crop. Open overstory conditions promote bigger crowns and better seed-fall, and concurrently a more vigorous understory of understory flora.

Seven decades of fire exclusion, however, in these shortleaf pine stands have had the effect of promoting persistent hardwood rootstocks of oaks and hickories. The silviculture of shortleaf pine-bluestem restoration requires the removal of this hardwood midstory to promote the desired grasses in the understory. To date, mechanical treatments and cyclic burning have not quite been sufficient to eliminate these rootstocks.

LOBLOLLY-SHORTLEAF PINE ON THE UPPER GULF COASTAL PLAIN

Loblolly pine has a broad natural range also, only slightly less broad than shortleaf pine and falling short to the extreme northeast and southwest. Throughout most of this natural range, loblolly and shortleaf pines grow together in highly productive stands (Figure 5), with loblolly dominating some mixtures and shortleaf others. The archetypal county in the US in which to show a diversity of silvicultural practices in this forest type is Ashley County, Arkansas, home to the Crossett EF. Here, both loblolly and shortleaf



Figure 5— Well-stocked uneven-aged loblolly-shortleaf pine stand, Compartment 56--Poor Farm Forestry Forty Demonstration, May 2006. Crossett Experimental Forest, Ashley County, Arkansas (Photo by Benjamin S. Glaze)

pine have historical prominence not just for the South, but the natural resource history of the Nation--because the first successful efforts to demonstrate forest sustainability through management of second-growth forests occurred in loblolly-shortleaf pine stands at Crossett, Arkansas and Urania, Louisiana (Chapman 1942).

Awareness of tree growth and the potential for economic management of second-growth stands occurred in this forest type for a simple reason—the growth of these pine stands is rapid, and easy to observe in a short period of time. Data show that well-mature stocked stands of loblolly-shortleaf pine exhibit average annual growth rates per acre of 3 square feet of basal area, 80-100 cubic feet of total merchantable volume, 400 fbm Doyle or 450 fbm Scribner. With such growth rates, high-quality sawtimber can be harvested in 50-60 years under conservative management approaches (Reynolds 1959, Reynolds 1969, Baker and others 1996, Guldin and Baker 1998, Guldin 2002).

Relatively good anecdotal evidence remains about the patterns of harvest in mixed loblolly-shortleaf pine stands of the upper West Gulf Coastal Plain, both from Chapman's work and because of the detailed descriptions of rehabilita-

tion and recovery published from Forest Service research centered on the Crossett EF, which was established in 1935 (Reynolds 1942).

Interestingly, research papers and photo captions of the day refer to mixed second growth “shortleaf-loblolly” pine-hardwood type stands (e.g., Reynolds 1947), which may refer to a plurality of shortleaf pine in mixture with loblolly pine and hardwoods. Conversely, loblolly pine dominates these stands today. The difference may be due to the different regeneration dynamics of these two pines. Shortleaf, a less prolific seed producer than loblolly, resprouts if topkilled by fire as discussed above. But loblolly pines topkilled by fire will not recover. The tactic for loblolly seems to lie in that prolific annual seed crop, which drops adequate or better seedfall 4 years in 5 (Cain and Shelton 2001).

Control of competing vegetation, in this case hardwoods, was an essential element of the successful rehabilitation of cutover loblolly-shortleaf pine stands. Early in the recovery process, hardwoods were cut for chemicalwood (an early biofuel product) and as fuel for steam generation in operating sawmills; later, as herbicides came into common silvicultural use in the 1960s, a decadal herbicide treatment was employed and still is recommended in some silvicultural prescriptions (Guldin and Baker 2002).

The need for control of competing vegetation is especially important in situations where reproduction of desired species is found as seedlings, and they are in competition with other species that are not sought but whose regeneration occurs through sprout origin. In the southern pines, this is a dynamic between pine seedlings and hardwood sprouts, which enables the effective application of herbicides that control the hardwoods with minimal effect on the pines. But rather than consider this as a ‘kill the hardwoods to release the

pines’ question, it is more properly seen in strict silvicultural semantics (though admittedly not by the public) as a ‘kill the sprouts to release the seedlings’ prescription.

Finally, one critical link in the ability of loblolly to recover from understocked cutover conditions was the ability of small suppressed trees with small crowns to respond to release from suppression, even at advanced ages. Standards developed at the Crossett EF suggest that a suppressed loblolly pine tree with a 20 percent live crown ratio, good apical dominance, and a diameter outside bark of 2 inches or greater at the base of the live crown can respond to release, and eventually develop into a codominant crown position (Baker and Shelton 1998).

COMMON THREADS IN THE RECOVERY OF THE SOUTHERN PINES

These three southern pine forest types have little that unites their successful recovery from high-grading at the turn of the last century, because the recovery varied tremendously from one forest type to the next. By and large, Coastal Plain loblolly-shortleaf pine stands fared the best, followed by pure shortleaf pine stands in the Ouachitas and

Table 1—A subjective assessment of the common silvical and silvicultural attributes that must be considered in the restoration of three major southern pine forest types.

Forest type	Frequency of seedfall	Seedling development without fire	Stem density after harvest, 1910-1930	Planting success in 1960?	Planting success today?
Longleaf pine, lower West Gulf Coastal Plain	Fewer than one crop per decade	Poor	Low	Poor	Good
Shortleaf pine, Ouachita-Ozark Highlands	Three crops per decade	Fair	Moderate	Fair	Good
Loblolly-shortleaf pine, upper West Gulf Coastal Plain	Eight crops per decade	Excellent	Moderate to high	Excellent	Excellent

Ozarks, with the poorest recovery found in the longleaf pine forest type in the lower Coastal Plain. It's possible to speculate about several reasons for this related to both silvics and silviculture (Table 1).

The most obvious factor is the frequency of seedfall, with recovery falling along the scalar of seedfall frequency. But there is probably a related effect with respect to fire exclusion. Because loblolly pine, the species with the most frequent seedfall, also has seedlings and saplings most at risk of mortality if topkilled by fire, the coincident effect of fire suppression with prolific seedfall had something to do with its successful recovery. Shortleaf pine, as Mattoon (1915) observed, had seedlings present as advanced growth and an ability to resprout if topkilled, as an adaptive trait to survive the frequent surface fires. As high-grading ended and unchecked fires were controlled, advance-growth shortleaf pine saplings may simply have responded to release, especially in pine-dominated stands. The last disturbance may also have hit shortleaf during a good seed year for the species. It's also easy to see why mixed hardwood-pine stands may be absent on the landscape today—with less fire, and fewer pines as advance growth, there would be less opportunity for pines to develop into the overstory if hardwoods dominate the site.

With longleaf, the story of seedfall is more complicated. The grass stage of longleaf is an adaptive strategy for frequent surface fires, but it renders the species difficult to see. In addition, intense fires will kill longleaf seedlings. It is likely that the heavier cut in longleaf stands relative to the other southern pines resulted in considerably more scarification, and seedfall would have been just about nonexistent from the few poor-crowned longleaf residuals after harvest. When fire was removed from the longleaf systems, the probability of longleaf seedlings escaping from the grass stage, especially when infected by brownspot, would have plummeted. Then too, the direct seeding technology that was developed to reforest the deforested longleaf stands relied on slash pine and loblolly pine rather than longleaf, largely because of seed availability (Derr and Mann 1971). Any longleaf seedlings that still persisted were not likely to survive the standard site preparation prescription used with direct seeding—a heavy cultivation treatment, either by disking or harrowing.

A second reason for the variation in recovery by these three southern pines is the degree to which a manageable residual stand was retained, a factor that generally paralleled that of the happenstance regeneration and the stocking of the initially cutover stands. Loblolly's ability to respond to release from suppression undoubtedly helped trees below the diameter limit to respond and eventually to dominate cutover sites. Shortleaf has a similar attribute, and the slower growth rates and smaller sizes of harvested trees in shortleaf probably contributed to retention of a manageable component of shortleaf pine in the mountains of the region after the high-grading occurred. Generally, longleaf pine stands - the most valuable of the three species for lumber at the turn of the last century, and also of great importance to the turpentine industry - were more heavily cut than stands in the other forest types, making recovery all the more difficult.

Finally, if a species cannot regenerate itself in desirable amounts relative to both quantity and distribution, planting will be required to obtain acceptable regeneration. The historical success of planting in these three southern pines is also correlated with recovery from high-grading. Loblolly pine was and is the easiest of the three to plant, and the rise of industrial forestry in the South relates directly to industry's proficiency in planting loblolly and to the success of the genetic improvement program over the 20th century for that species. Shortleaf seedlings are easy to grow in a manner similar to that of loblolly. But successful planting of shortleaf pine is limited by the rocky soils found in Ouachitas and Ozarks, where it is nearly impossible to insert a dibble into the ground. Planting success in shortleaf pine increased dramatically in the latter part of the 20th century with advances in site preparation practices, especially ripping to create a microeroded furrow into which seedlings could be planted; first-year survival improved quickly thereafter (Walker 1992). Conversely, technology for planting longleaf seedlings has been more difficult to develop, and also required associated advances in site preparation methods so as to get longleaf pines out of the grass stage in a timely manner (Barnett 2004). Widespread application of this recent technology to successfully plant longleaf pine will be a key to the future recovery of the species.

DISCUSSION

The pattern of recovery of forest stands from high-grading at the turn of the 20th century has lessons for silviculturists in the 21st century. Those lessons relate to applying inventive tactics to take advantage of any unique ecological attributes of the species being managed, to practicing restoration of process in a larger rather than smaller landscape, and to adapting tools for widespread industrial reforestation to a restoration context.

Successful restoration is accomplished in one stand at a time. The first step is to secure the establishment and development of the desired species in the stand being managed. If the desired species for restoration are still present on the site, managers can encourage their development to the point of maturity, and apply reproduction cutting methods appropriate for the species being managed.

For example, if the desired species has the advance growth habit, the disturbance used to recreate the restoration (such as prescribed burning) must not kill the seedlings or saplings completely, or else the process of recruitment of advance growth must begin again. But advance growth reproduction dynamics relate closely to development of new shoots from undamaged rootstocks after the shoots have been top-killed by prescribed fire, as is the case with oaks and two of our southern pines. This requires careful attention to the balance between overstory shade and understory development. It is important to manage overstory stocking to get enough light to the desired advance growth while not releasing competition too much. Generally speaking, the larger the advance growth at the time of regeneration, the greater is the likelihood of success. Irregular shelterwood reproduction cutting methods are ideal in this approach, and have been successfully applied for long periods of time in mixed loblolly-shortleaf pine stands in the upper West Gulf Coastal Plain (Zeide and Sharer 2000).

Operationally, reliance on advance growth is a less risky silvicultural tactic than relying on seedfall. Precise coordination of harvest operations with a bumper seed crop is difficult on Federal lands, given the multi-year timber sale contracts usually provided to logging contractors. Silviculturally, it is better to rely upon advance growth established prior to logging, and then prescribe site preparation or release treatments after the timber sales have been closed to

re-establish advance growth as a uniform age cohort across the site. The alternative for natural seedfall is to prepare seedbeds for seedfall after the timber sale is completed, when the next adequate or better seed crop is expected.

Reproduction cutting methods that rely upon advance growth must be initiated well in advance of the seed cut. Advance growth of a desired species should be encouraged through silvicultural interventions as much as 2 decades prior to the reproduction cutting, so as to be in place prior to the establishment of the new stand. In pines, that can often be accomplished using prescribed fire, which acts both to control competing vegetation and to scarify the forest floor so as to promote germination and seedling establishment—a prescription shown to be effective in loblolly, longleaf, and shortleaf pine. Fire must then be excluded until the seedlings are tall enough to survive it, at which time fire can be reintroduced to the restoration prescription.

Second, some measure of overstory density reduction will be required in most restoration prescriptions, especially if stands have developed through the 20th century after a turn-of-the-century high-grading. In intolerant species, opening stands in this way promotes understory development, which places a premium on the implementation of understory treatments that encourage the desired understory species of forbs and grasses rather than a preponderance of woody plants, especially those tolerant of shade. The limits within which an acceptable understory development of desired forbs and grasses can be mated to acceptable levels of woody plant advance growth for eventual use in reproduction cutting requires a certain degree of balance. It also requires that a silviculturist has some confidence that when the time comes, the desired tree species can be developed immediately after a reproduction cutting prescription is implemented.

Thirdly, successful restoration silviculture demands that a silviculturist balance the cognitive dissonance between economics and ecology. If a reduction in overstory density is prescribed, it is certainly to the advantage of the silviculturist to do so using commercial timber sales. Appropriated funds are usually less readily available than trust funds, which are based in timber sale receipts and thus represent a more liquid capital asset. If a silviculturist can use timber sales to liquidate standing volume assets in excess to the need for restoration, monies for sale area improvement

can then be collected and applied for restoration purposes through supplemental reforestation and release treatments. The bottom line is that the area that can be restored using timber sale proceeds is often much larger than can be restored with appropriated dollars. For example, suppose a restoration prescription outside a sale area costs \$75 per acre to implement. If 3 mbf per acre of volume in excess of restoration needs can be harvested at \$250/mbf, \$750 per acre will be generated from the timber sale; conservatively, half of that could be applied for restoration prescriptions within that sale area over the next 5 to 10 years. Over a 200-acre project area in a watershed under prescription development, a manager would need \$15,000 in appropriated funds to treat an area, or could draw from a pool of \$75,000 in the same area from the timber sale proceeds. Applying the timber sale program in a restoration context provides substantially more flexibility in restoring a larger number of acres at less cost to the agency than would be possible using appropriated dollars, as has been shown in the pine-bluestem restoration program on the Ouachita NF (Guldin and others 2004). The key is in defining the meaning and intent of the 'standing volume in excess of restoration' concept.

Fourth, some tools such as artificial regeneration that traditionally have been associated with intensive forestry for fiber production can be important in restoration. But modification of prescriptions and recommendations may be in order. For example, tree planting will be an important tool for forest restoration when a species is absent from the stand being restored, or in species with erratic seed production. But the practice will differ from planting where industry fiber management is a goal. A new world of opportunity opens for forest geneticists in this context, because we do not know whether families selected for rapid height and volume growth in intensively site-prepared stands will be the same families that thrive in partial shade, or where site preparation is less than complete. Then, too, there may be wisdom in variations of tree spacing when planting for restoration objectives so as to create stem patterns other than rows. A planting crew can plant 544 trees per acre on a strict 8 x 10 ft spacing, or by planting trees on a 3-ft to 15-ft spacing provided that the spacing variation is calibrated to hit the mean.

Fifth, no silvicultural practice should be arbitrarily excluded from the tools in the toolbox of restoration. One

tool more cussed than discussed in the 21st century is the use of herbicides, largely because of public distaste for the practice. However, in the southern pinery, herbicides are extremely effective in controlling competing vegetation, especially as a tool to kill sprouting hardwoods competing with pine seedlings or saplings. Midstory hardwoods that became established after the onset of effective fire control may not be desired in the restored stand, and are very difficult to remove without herbicides. Similarly, herbicides may be the only effective way to control invasive exotics such as kudzu and cogon grass that have invaded sites intended for restoration. Silviculturists must justify the use of herbicides for restoration based on achieving the ecological goals the restoration is designed to achieve—and even then, there might be considerable public opposition to the practice. However, in many instances, herbicides will be the single most effective and least costly treatment to control unwanted vegetation. This or any other practice should not be ruled out in advance of silvicultural planning.

However, restoring one stand does not restore the functional ecosystem attributes that transcend the scale of an individual stand. Restoration of ecosystems connotes a scale larger than a single stand, and thus the silviculture of restoration will require prescribing silvicultural treatments concurrently in many stands toward a common ecological goal. This requires that silviculturists expand their view from the traditional stand to the appropriate functional ecological scale. The resulting system of prescriptions for stands within the landscape should become, ecologically speaking, more than the sum of the stand-level prescriptions.

In this context, it is likely that diverse silvicultural practices among stands across a landscape will be more robust than a single textbook prescription. Thirty years ago in Region 8, reproduction cutting decisions invariably prescribed clearcutting and planting, which met important objectives of timber production and creation of early seral conditions, but generally did so in 30- to 40-acre blocks on the landscape. A greater diversity in size and intensity of early seral condition might have been obtained using a diverse even-aged and uneven-aged reproduction cutting methods, not just clearcutting, and through operations in stands of varied area. Similarly, contemporary silviculturists should resist the urge to employ a single restoration prescription, lest the restored conditions become too uniform

and fail to reflect the range of natural variability that would be desirable in restored conditions.

Finally, restoration will be easier in some forest types than in others despite the silviculturist's best efforts. The key is the ease with which regeneration of the desired species can become established and encouraged to develop. Some species such as loblolly pine could be restored by taking advantage of that bumper seedfall four years in five, with minimal capital investment. Others such as longleaf will require widespread reforestation using artificial regeneration, with large capital investment to ensure survival. Silviculturists should be increasingly aware of the opportunities for ecological interconnectedness through complementary silvicultural operations in adjacent stands, and through operations on a broad scale rather than a narrow one. The success of restoration in the Ouachita Mountains, for example, is not so much at the scale of a 40-acre stand as at the scale of a 40,000-acre south-facing ridge. This experience exemplifies the silvicultural opportunity of the 21st century—learning how to enable restoration of landscapes through functional arrangements of stands, restored one stand at a time by a silviculturist plying a time-honored craft.

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Fairly Sustainable Forestry: Seven Key Concepts for Defining Local Sustainability in a Global Ecosystem

Stephen R. Shifley

ABSTRACT

In the U.S. we increasingly restrict wood production in the name of sustainability while going abroad for a growing share of the wood we consume, even though our own forest resources per capita are far greater than the global average. The unintended consequence is that we transfer impacts (positive and negative) of our timber harvesting and wood consumption to other places. This is not sustainable in the broad sense of the word. Seven key concepts help define limits on sustainable forestry in the U.S.:

- (1) we must ensure sustained timber yield;
- (2) most people find harvesting unaesthetic and prefer not to see it;
- (3) in the U.S. we annually consume the equivalent of about 20 billion cubic feet of wood products;
- (4) the U.S. is a net importer of wood and has been for at least 90 years;
- (5) as we import wood and wood products we also export to other nations the environmental, economic, and other social consequences (both the positive and negative) associated with wood production, manufacturing, and consumption;

(6) as a natural resource, wood is generally preferable to alternative commodities; and

- (7) all the wood consumed on Earth must be produced from the 9.6 billion acres of forestland on the planet.

About 30 percent of the land mass of the earth is forested, about one-third of North America is forested, and about one-third of the United States is forested. Despite having a proportionate share of the world's forests, our national imbalance between domestic wood production and consumption annually sends billions of cubic feet of environmental consequences (positive and negative) to other nations. National Forests, for example, contain 19 percent of U.S. timberland and now produce less than 2 percent of the wood consumed in the U.S. The USDA Forest Service is rightly concerned about sustainability for National Forests, private forests, and global forests. In fact, we have separate divisions dealing with each constituency. We should think carefully about how the quest for sustainable management in any one sector affects forests elsewhere.

Keywords: consumption, growth, removals, national forest

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INTRODUCTION

Sustainable forestry is hard to define in measurable or quantitative terms, so we tend to rely on conceptual or qualitative definitions. My favorite definition is the Native American proverb, “The frog does not drink up the pond in which it lives.” It is pithy, easy to remember, and evokes a vivid mental image. And like much discussion related to sustainable forest management, the focus is more on what not to do than what to do.

The Dictionary of Forestry is more comprehensive in its definition of sustainable forest management (Helms 1998).

Sustainable forest management (sustainable forestry) (SFM) this evolving concept has several definitions 1. the practice of meeting the forest resource needs and values of the present without compromising the similar capability of future generations—note sustainable forest management involves practicing a land stewardship ethic that integrates the reforestation, managing, growing, nurturing, and harvesting of trees for useful products with the conservation of soil, air, and water quality, wildlife and fish habitat, and aesthetics (UN Conference on Environment and Development, Rio De Janeiro, 1992 [see citation for United Nations 1992]) 2. the stewardship and use of forests and forest lands in a way, and at a rate, that maintains their biodiversity, productivity, regeneration capacity, vitality, and potential to fulfill, now and in the future, relevant ecological, economic, and social functions at local, national, and global levels, and that does not cause damage to other ecosystems (the Ministerial Conference on the Protection of Forests in Europe, Helsinki, 1993) —note criteria for sustainable forestry include (a) conservation of biological diversity, (b) maintenance of productive capacity of forest ecosystems, (c) maintenance of forest ecosystem health and vitality, (d) conservation and maintenance of soil and water resources, (e) maintenance of forest contributions to global carbon cycles, (f) maintenance and enhancement of long-term multiple socioeconomic benefits to meet the needs of societies, and (g) a legal, institutional and economic framework for forest conservation and sustainable management (Montreal Process, 1993) [see citation for Montreal Process Working Group 1995].

That definition is more comprehensive in listing the components or the range of issues included in sustainable

forestry. Moreover, it emphasizes the importance of addressing multiple spatial or geographic scales from local to national to global. This definition offers little guidance for measuring or quantifying the listed dimensions of forest sustainability, but it explicitly references the Montreal Criteria and Indicators (Montréal Process Working Group 2005) and lists the widely accepted seven criteria (or dimensions) of sustainable forestry as items (a) through (g) in definition 2.

The Montreal Criteria and Indicators (Montréal Process Working Group 2005) are especially important because they also provide a list of things to measure, count or otherwise quantify in order to describe the current status of each of the seven criteria and to monitor changes over time.

Consequently in the U.S. and elsewhere there are significant, ongoing efforts to measure and monitor over time the set of Montreal indicators. The Forest Inventory and Analysis (FIA) data (USDA Forest Service 2007a) provide a remarkably detailed and highly accessible statistical profile of forest resources across a wide range of spatial scales. Moreover, FIA data have been combined with other sources of information to specifically summarize and report the conditions of U.S. forests in the framework described by the Montreal Criteria and Indicators (e.g., Carpenter et al. 2003, USDA Forest Service, 2004).

Armed with this growing body of data and standardized summaries, we now can track the way many of these important indicators of forest sustainability change over time. What we still lack in most cases is knowledge of what values of the indicators are associated with sustainable or unsustainable forestry. For example, area by forest type relative to total forest area is one of the indicators used to measure biological diversity. Forest area by cover type has changed over time in many parts of the U.S. Does that indicate a sustainable or unsustainable condition? Timber harvest, another indicator, has decreased greatly on National Forests over the past 25 years. Does that indicate sustainable or unsustainable forestry? The area burned by wildfire (another indicator) has increased dramatically over the past two decades. Does that indicate sustainable or unsustainable forestry? It is often hard to determine.

In this paper I offer my own thoughts on a quantitative context for sustainable forest management. Although it is

primarily addresses timber—the forest output that we are best able to measure—it frames a set of constraints that affects all other dimensions of sustainable forestry at local, state, regional, national, and global spatial scales. Additionally, the same concepts can be applied to other measurable dimensions of forest sustainability.

SEVEN CONCEPTS RELEVANT TO SUSTAINABLE FORESTRY

In my own thinking about forest sustainability I have been able to arrive at seven concepts that collectively help me more clearly understand how to quantify the wood commodity aspect of sustainable forestry (Shifley 2006). Each concept is fairly simple, but collectively I find them enormously instructive with regard to sustainable forestry at many different spatial scales.

Concept 1: Sustained Yield

Sustained yield is at the core of professional forest management. Forest ecosystems are not sustainable if volume or biomass losses exceed growth over large areas or long time periods. Losses can be removals for wood products or fuel, the result of land clearing, or the consequence of fire, insects or disease. Whatever the cause, if there is a net decline in volume or biomass over large areas (e.g., thousands of acres) or over long periods of time (e.g., a decade or more), there is broad agreement that the situation is not sustainable. The concept of large-scale, long-term, non-declining volume is clear, measurable, and deeply rooted in our conservation ethic.

An examination of FIA inventories of U.S. forest resources indicates that we are clearly sustainable with regard to this first tenant of forest sustainability. Although the volume of U.S. timber decreased dramatically with the great waves of industrial logging, land clearing, and European immigration that occurred in the 1800's and early 1900's, since the 1950's (the beginning of contemporary statistical forest inventories) the volume of U.S. timber increased steadily from 616 to 856 billion cubic feet (39%) (Smith et al. 2003). Over the same period the total area of timberland decreased by only one percent. This pattern of increasing timber volume over the past 50 years is consistent across all regions of the U.S.

Over the same 50-year period (1953-2002), the volume on National Forest land increased from 218 to 260 billion cubic feet (19%) while timberland area increased from 95 to 97 million acres (but varied considerably from year to year over the period) (Smith et al. 2003). The increase in timber volume was not evenly distributed geographically. National Forest timber volume more than doubled in the eastern U.S., increased by nearly 60 percent in the intermountain region, increased by 6 percent on the Pacific Coast (exclusive of Alaska), and decreased by nearly 50 percent in Alaska. In Alaska and states along the Pacific Coast the changes in timber volume have been influenced by policies and legislation that reduced the amount of timberland available for harvest. For example, between 1953 and 2002, the area in Alaska classified as timberland decreased by 40 percent (8.4 million acres). Timberland area in Washington and Oregon decreased by a combined 3.7 million acres over the same period. For all states in U.S. combined, the net loss of timberland was only 5.3 million acres for the same period because timberland area increased in the Northern and the Rocky Mountain regions of the U.S.

On the surface, at least, this is good news with respect to forest sustainability. Forest growth exceeds timber harvest and other losses to land use change or damaging agents. However, an examination of our patterns of forest growth, removals, and consumption in a broader context raises the concern that our current situation is not sustainable in a global context.

Concept 2: Timber Harvests are Unattractive and Unappreciated

As forestry professionals we understand that timber harvesting serves many important purposes such as producing commodities, maintaining biodiversity, providing specific types of wildlife habitat, and improving forest health. However, most people find harvesting unaesthetic and would prefer not to see it where they live, recreate, or travel. This attitude is often evident in public responses to proposed management activities on National Forests. Thus, decisions about where, when, and how much to harvest must have a sound scientific and social basis, because harvesting is unpopular with a large segment of the public and is likely to remain so.

Concept 3: We Consume a Lot of Wood in the U.S.

We consume about 20 billion cubic feet of wood per year in the U.S. (Howard 2003, Haynes 2003). This annual wood

consumption is equivalent to about 67 cubic feet per person and far more than the global average annual consumption of 21 cubic feet per person (Gardner-Outlaw and Engelman 1999). U.S. annual per capita consumption gradually decreased from 83 to 67 cubic feet between 1986 and 2002, but total consumption did not decrease substantially because the population of the U.S. increased over that period. Projections from the most recent Resources Planning Act (RPA) (USDA Forest Service 2007c) documents indicate that by 2050 growth in the U.S. population will drive U.S. wood consumption up to 27.5 billion cubic feet per year. That is an increase of 40 percent relative to 1996 values, even with a projected slight decline in per capita wood consumption over that period (Haynes 2003, table 11).

Concept 4: The U.S. is a Net Importer of Wood

The U.S. has been a net importer of wood for at least 90 years (Haynes 2003). We participate in the global wood market, and we constantly import and export logs, lumber, and finished wood products. For example, about one-third of the softwood lumber we consume comes from Canada (Howard 2003, Society of American Foresters 2004), and we obtain many finished wood products from abroad. At the same time we export veneer logs, wood chips, and finished products throughout world. When imports and exports are converted to their equivalent cubic feet of roundwood and compared, imports substantially exceed exports.

In 1991, net imports amounted to about 2 percent of total U.S. consumption. By 1996 they were 9 percent of consumption, and by 2002 they were 16 percent of total consumption (Howard 2003, Haynes 2003). The net balance of imports over exports is projected to increase to about 19 percent of total U.S. wood consumption by 2050 (Haynes 2003).

Concept 5: When We Import Wood We Export Consequences of Production and Consumption

As we import wood and wood products we also export to other nations the environmental, economic, and other social consequences (both the positive and negative) associated wood production, manufacturing, and consumption. This is what former Forest Service Chief Dale Bosworth said about it in 2003 (Bosworth 2003):

“Out of sight, out of mind”—that is the danger of a system that separates consumption of forest products in one place from production in another. Our system today raises serious questions of both

equity and sustainability. We need more of a dialog on how to bring consumption in the most developed parts of the world into balance with production elsewhere.”

Currently we export the consequences associated with net annual imports of about 3 billion cubic feet of wood products. By 2050 we could be exporting the consequences associated with net annual imports of nearly 5 billion cubic feet of wood products (Haynes 2003).

This fifth concept is the key concept in the list of seven. If we believe there are no positive or negative consequences associated with timber production, then the other six concepts are largely irrelevant and we could presumably meet all our current and future demand for wood by purchasing it on the global market. However, based simply on public comments related to National Forest management policies, one would be hard pressed to assert that people believe there are no social, environmental, or economic consequences associated with timber production.

Concept 6: Better to Use Wood than Most Substitutes

We could substitute other products for wood and thereby greatly reduce current and future demand for wood. However, wood is environmentally benign compared to alternatives such as steel, plastics, or concrete. Wood is abundant, renewable, recyclable, and biodegradable. It has many desirable properties for construction and manufacturing. Clearly, forests can provide numerous other commodities and amenities such as clean water, wildlife, recreation, biodiversity, and carbon sequestration while producing wood.

Compared to alternative materials it requires relatively little energy to convert wood to useful products. Total product life cycle analysis compares the total energy balance and environmental impact of wood and other construction materials from production, to processing, utilization (e.g., in a building), and eventual disposal. This research has shown wood and wood fiber construction materials to be preferable from an environmental perspective when compared to substitute materials (e.g., metal, concrete) (Lippke et al. 2004).

Concept 7: There is a Finite Area from Which the Wood We Use Must Come

All the wood consumed on Earth must be produced on the 9.6 billion acres of forestland on the planet. That acreage

changes a little from year to year due to forest clearing and afforestation, but the bottom line is that the Earth has a finite amount of forestland and many competing land uses that are incompatible with forestry. If we view U.S. forest resources within that global context, we get a new way to gauge sustainability of our own forests. By sheer coincidence, the proportion of forest in the United States is nearly identical to that of the Earth as a whole. Specifically:

- about 30 percent of the land mass of the earth is forested (Food and Agriculture Organization 2000) (fig. 1)
- about one-third of North America is forested (Natural Resources Canada 2005, Smith et al. 2003), and
- about one-third of the United States is forested (Smith et al 2003).

The analogy can be taken further for a more local view. For example, it turns out that the seven-state North Central Region of the U.S. (where I reside) is nearly one-third forested, the state of Missouri (where I reside) is one-third forested, and even Boone County (where I reside) is nearly 30 percent forested (Miles 2007). That series of statistics is enormously instructive in defining sustainable forestry in the U.S. and at smaller spatial scales. In the United States we have forest resources that are proportional in area to those found in the rest of the world. In fact, because our population is relatively low, we have the benefit of more forest per capita than the world as a whole. U.S. forestland is 2.7 acres per capita and falling; global forest land is 1.6 acres per capita and falling.

RETHINKING SUSTAINABLE FORESTRY

Sustainable forestry requires a conceptual link between the consumption and production of wood at global, national, and regional levels (Strigel and Meine 2001). This is something that we have for the most part failed to do, and for U.S. forests it has resulted in a situation that is not globally sustainable. Contemporary notions of sustainable forestry stipulate that we must be concerned about dozens of different measures of forest condition and social well being (Montréal Process Working Group 2005). However, contemporary notions of sustainability do not discourage us from creating “sustainable” forests at home by simply going elsewhere to get the wood and products we consume. This disconnect between consumption and the location of



Figure 1. U.S. forest resources in a global context. Dark shades indicate forest land. The land mass of the Earth is about one-third forested, North America is about one-third forested, the United States is about one-third forested, and Missouri is one-third forested. Sources: World map; Food and Agriculture Organization (2000), North America map; United Nations Environmental Programme (2005), Food and Agriculture Organization (2000), United States and Missouri (Zhu and Evans 1994). Composite figure follows Shifley (2006).

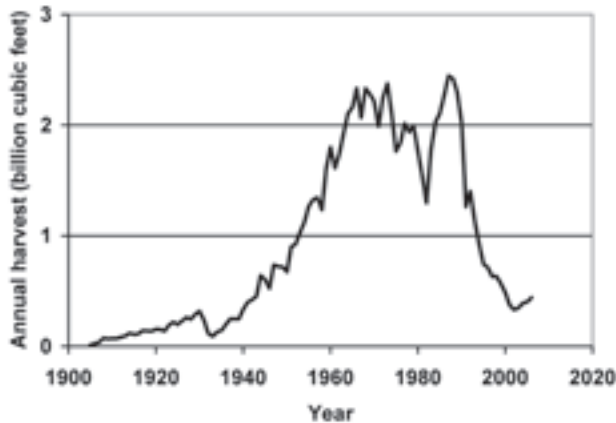


Figure 2. National Forest timber harvest 1905 to 2006. Conversion to cubic feet based on 5.2 board feet per cubic foot. Harvest peaked in 1987 at 2.4 billion cubic feet (12.7 billion board feet). In 2006, harvest was 0.4 billion cubic feet (2.3 billion board feet). Data from USDA Forest Service (2007b).

production leaves a huge void in our current notions of sustainable forestry. There is currently no social or economic penalty associated with over-consumption and/or under-production of forest products as long as we can export any associated environmental issues to the other nations that feed our demand for wood.

We expend a great deal of time and energy in this country in discussions, debates, and court battles over individual timber sales or other management actions, particularly on public lands. National Forests are focal point of much of that interaction. For the most part those discussions take place in the absence of explicit, over-arching principles regarding our national role as a global partner in sustainability. The outcome is that we increasingly restrict our domestic wood production in the name of sustainability while going abroad for the wood we consume. The unintended consequence is that we push the impact of our consumption of wood products to other places. Those impacts (both the positive and negative) go out of sight and out of mind to places where we have neither the will nor the means to ensure that local forestry practices are sustainable. Is that a sound policy for sustainable forestry given that:

- our own forest resources are every bit as abundant as on the rest of the Earth,
- our own forest resources per capita are far greater than the global average,
- and the growth of our own forests greatly exceeds harvest and natural mortality?

Can we tout our own efforts directed at forest sustainability in the U.S. if success comes at the expense of an ever increasing reliance on wood products produced elsewhere where we take little or no responsibility for the methods of production?

Sustainable forestry cannot be achieved in the U.S. by simply transferring to other nations the consequences we do not care to deal with in our own public and private forests. An integral part of sustainable forestry in the U.S. must be to balance the quantity of wood we produce with the quantity of wood we consume (on a volume equivalent basis). If we cannot do that with our proportional share of the world's forest resources, how can we expect others to do it for us?

Clearly, issues of wood consumption, wood production, and harvest levels alone cannot define sustainable forestry. However, if we fail to use those issues to guide decisions about sustainable forest management (in all its dimensions) at local, state, regional, national, and global scales, we run the risk of simply transferring impacts to someone else's forest.

SCALABLE SOLUTIONS TO SUSTAINABLE FORESTRY

An underlying premise of the proceeding discussion is that approaches to sustainability are scalable. The appropriate scale varies with the issue, but national, state, and county scales are essential. Those are the scales at which most laws, regulations, policies, penalties, and subsidies that affect forests and forest management are debated and enacted.

If we were to adopt a national goal of annually producing a volume of wood that is commensurate with our consumption, FIA statistics provide abundant information about how the nation, the 50 states, and the thousands of counties are progressing with respect to that goal (e.g., Smith 2003, USDA Forest Service 2007a). The math is easy; there are about 500 million acres of timberland in the U.S. that we can draw upon to produce the roughly 20 billion cubic feet of wood we consume each year. Note that timberland excludes forestland that is inaccessible, unproductive (e.g., due to climatic conditions), or administratively or legislatively restricted from harvesting (e.g.,

parks and wilderness). If we summarize the percent of timberland by state we get a rough estimate of how much wood each state might expect to contribute to a combined national production of 20 billion cubic feet of wood (Table 1). Policy decisions based on such goals are not easy in the face of the many competing interests for all the things that forests provide, but such goals provide an essential point of reference for sustainable forestry at state, national, and global scales.

There are minimum relevant scales for this type of analysis. For example, we can readily compute what would be required of each acre of U.S. timberland if we were to balance current domestic wood production with current wood consumption, but we don't manage individual forest acres. Rather, we manage stands which are components of forest ownerships that occur with other ownerships on landscapes that simultaneously provide many products and amenities. Thus, sustainable forestry must be approached simultaneously at multiple spatial scales. Sustainable forestry occurs hierarchically across landscapes, ecoregions, states, and nations when we measure progress in the context of specific, cumulative, scalable goals. State and national scales of reference are particularly important because those are the scales where policy, legislation, and incentives related to forest management are crafted. Moreover, if there are many instances where sustainable forest management is not practiced at the ecoregion or state scale, sustainable forestry at the National scale will be impossible to achieve.

We can look at the state of Missouri as an example. Missouri is an average state with about 2 percent of the U.S. population, and it is one-third forested. Missouri has about 15 million acres of timberland, or 2.7% of the nation's total (Table 1). Thus, as a "fair share" we might expect Missouri to contribute about 530 million cubic feet of wood towards 20 billion cubic feet of annual domestic wood consumption (Table 1). Missouri's annual removals amount to about 170 million cubic feet. Moreover, the latest on-line FIA data for Missouri show mean annual removals dropped to 120 million cubic feet while annual growth continued to increase beyond 600 million cubic feet (Miles 2007), and growth is still well below potential. Perhaps those of us who live in Missouri should be concerned about the imbalance between what we produce relative to our share of U.S. timberland.

We can look at National Forests in the same context. National Forests include 19 percent of the Nation's timberland, about 97 million acres. Using the same logic applied to Missouri or other individual states in Table 1, National Forests might be expected to produce the equivalent of 3.8 billion cubic feet of wood each year (roughly 20 billion board feet). Over the past 6 years harvest on National Forests has averaged about 380 million cubic feet cubic feet (about 2 billion board feet) (USDA Forest Service 2007b) (S. 2). That is only 10 percent of the "fair share" we might expect from an equivalent area of other U.S. timberland. This is relevant to sustainable forestry at the national and global scales.

We manage National Forests differently than most other forest land, and rightly so. They provide unique opportunities to meet a wide array of multi-resource objectives and the citizen-owners of the National Forests play an important role in guiding forest management. As an agency with commitments to sustainable management of private forests, public forests, and global forests (and separate divisions devoted to those constituencies) we need to look closely at how management decisions in one sector affect other sectors and be devoted to a joint, scalable approach to sustainable forestry across public and private ownerships, here and abroad.

CHANGING THE BALANCE

We need to be concerned about balancing consumption and production of wood products in the U.S., but harvesting more timber is not the only way to achieve such a balance. We could, for example, simply consume less wood, provided we did not replace wood with products that created adverse environmental impacts. Recycling can also be a large part of reducing net consumption of new wood. Manufacturers can change the balance of wood production and consumption by increasing the efficiency with which they convert wood into products or by engineering new products that extend the utility of a given amount of harvested timber. And it is certainly possible to increase timber productivity by elevating management intensity for selected natural forests on some sites, through greater reliance on intensive plantation management, and by increasing forested acreage through afforestation or agroforestry. When the goal is to sustainably balance wood production and wood consumption, everyone has a part in the solution.

Table 1. Current forest area, timberland area, growth, and removals by state and region with estimates of a hypothetical “fair share” of volume production if states and regions produced wood in proportion to their timberland. Based on Smith et al. (2003) and U.S. Census Bureau (2007b) with additional computations by the author

(1)	(2)	(3)	(4)	(5)	(6)	(7)	(8)	(9)	(10)
	Total land area	Forest land	Percent forest	Timberland	Percent of U.S. Timberland	“Fair share” wood production	Current Removals	Current Growth	Approximate consumption
	Thousand acres	Thousand acres	%	Thousand acres	%	Million cu ft/yr	Million cu ft/yr	Million cu ft/yr	Million cu ft/yr
North:									
Northeast:									
Connecticut	3,101	1,859	60	1,696	0.3	67	12	55	240
Delaware	1,251	383	31	376	0.1	15	8	16	56
Maine	19,753	17,699	90	16,962	3.4	673	442	402	90
Maryland	6,295	2,566	41	2,372	0.5	94	41	107	379
Massachusetts	5,016	3,126	62	2,631	0.5	104	16	97	466
New Hampshire	5,740	4,818	84	4,503	0.9	179	140	170	89
New Jersey	4,748	2,132	45	1,876	0.4	75	11	55	596
New York	30,223	18,432	61	15,389	3.1	611	141	590	1,329
Pennsylvania	28,685	16,905	59	15,853	3.1	630	216	630	856
Rhode Island	668	385	58	340	0.1	14	2	8	74
Vermont	5,920	4,618	78	4,482	0.9	178	77	190	43
West Virginia	15,415	12,108	79	11,900	2.4	473	167	510	125
Total Northeast	126,815	85,031	67	78,370	15.6	3,113	1,272	2,832	4,342
North Central:									
Illinois	35,580	4,331	12	4,087	0.8	162	69	172	874
Indiana	22,957	4,501	20	4,342	0.9	172	97	224	428
Iowa	35,760	2,050	6	1,944	0.4	77	25	41	204
Michigan	36,359	19,281	53	18,616	3.7	739	316	756	698
Minnesota	50,955	16,680	33	14,723	2.9	585	316	370	349
Missouri	44,095	13,992	32	13,365	2.7	531	168	239	394
Ohio	26,210	7,855	30	7,568	1.5	301	101	293	792
Wisconsin	34,761	15,963	46	15,701	3.1	624	347	489	378
Total North Central	286,677	84,653	30	80,346	16.0	3,191	1,439	2,585	4,116
North Total	413,492	169,684	41	158,716	31.5	6,304	2,711	5,418	8,458
South:									
Southeast:									
Florida	34,520	16,285	47	14,636	2.9	581	560	685	1,159
Georgia	37,068	24,405	66	23,802	4.7	945	1,448	1,519	593
North Carolina	31,180	19,302	62	18,664	3.7	741	958	1,160	577
South Carolina	19,272	12,495	65	12,301	2.4	489	683	945	285
Virginia	25,343	16,074	63	15,371	3.1	611	655	848	526
Total Southeast	147,383	88,561	60	84,774	16.8	3,367	4,304	5,157	3,140

Even with dedicated efforts to reduce unnecessary consumption and increase recycling, it is projected that U.S. wood consumption will increase at a rate slightly less than the rate of population increase (Haynes 2003). That does not take into account the mounting interest in using wood as biofuel to reduce net carbon additions to the atmosphere (Perlack et al. 2005). We will need to harvest and process more wood in the U.S. if our collective forest resources are to be utilized at a globally sustainable level. We are fortunate in a sense, because our projected rate of population increase will be slightly less than that for the rest of the world (U.S. Census Bureau, 2007a, b) and our

total forest area per capita will be much greater than for the rest of the world. Nevertheless, harvest levels may need to increase by about 40 percent in the next 45 years to keep pace with projected increases in U.S. population (Haynes 2003). That future scenario could lead to (Shifley 2006):

- Increased harvest—from 18 billion cubic feet currently to more than 27 billion cubic feet in 2050 (projections on total consumption from Haynes 2003, table 12).
- A more even geographic distribution of harvests, and greater visibility of harvesting practices.
- A stronger commitment to the use of best management practices.

Table 1. (Con't.) Current forest area, timberland area, growth, and removals by state and region with estimates of a hypothetical “fair share” of volume production if states and regions produced wood in proportion to their timberland. Based on Smith et al. (2003) and U.S. Census Bureau (2007b) with additional computations by the author

South Central:									
Alabama	32,481	22,987	71	22,922	4.6	910	1,299	1,460	311
Arkansas	33,328	18,771	56	18,373	3.6	730	796	896	188
Kentucky	25,428	12,684	50	12,347	2.5	490	276	384	284
Louisiana	27,883	13,812	50	13,722	2.7	545	959	834	311
Mississippi	30,025	18,580	62	18,572	3.7	738	1,150	1,105	199
Oklahoma	43,955	7,665	17	6,234	1.2	248	133	243	242
Tennessee	26,381	14,396	55	13,956	2.8	554	384	738	402
Texas	167,626	17,149	10	11,774	2.3	468	770	705	1,510
Total South Central	387,107	126,044	33	117,900	23.4	4,683	5,766	6,365	3,447
South Total	534,490	214,605	40	202,674	40.2	8,050	10,070	11,522	6,588
Rocky Mountain:									
Great Plains:									
Kansas	52,367	1,545	3	1,491	0.3	59	7	26	188
Nebraska	49,201	947	2	898	0.2	36	10	14	120
North Dakota	44,156	672	2	441	0.1	18	1	7	44
South Dakota	48,574	1,619	3	1,511	0.3	60	21	40	53
Total Great Plains	194,298	4,783	2	4,341	0.9	172	39	87	405
Intermountain:									
Arizona	72,732	19,427	27	3,527	0.7	140	14	124	378
Colorado	66,387	21,637	33	11,607	2.3	461	21	291	313
Idaho	52,960	21,646	41	16,824	3.3	668	253	635	93
Montana	93,157	23,293	25	19,185	3.8	762	168	583	63
Nevada	70,276	10,204	15	363	0.1	14	1	6	151
New Mexico	77,674	16,682	21	4,359	0.9	173	19	140	129
Utah	52,587	15,676	30	4,683	0.9	186	8	77	161
Wyoming	62,147	10,995	18	5,739	1.1	228	14	119	35
Total Intermountain	547,920	139,560	25	66,287	13.2	2,633	498	1,975	1,322
Rocky Mtn Total	742,218	144,343	19	70,628	14.0	2,805	537	2,062	1,727
Pacific Coast:									
Alaska	365,041	126,869	35	11,865	2.4	471	142	207	45
Pacific Northwest:									
Oregon	61,442	29,651	48	23,831	4.7	947	863	1,728	244
Washington	42,612	21,790	51	17,347	3.4	689	867	1,426	421
Total Pacific NW	104,054	51,441	49	41,178	8.2	1,636	1,730	3,154	666
Pacific Southwest:									
California	99,824	40,233	40	17,781	3.5	706	634	1,325	2,431
Hawaii	4,111	1,748	43	700	0.1	28	0	1	86
Total Pacific SW	103,935	41,981	40	18,481	3.7	734	634	1,326	2,517
Pacific Coast Total	573,030	220,291	38	71,524	14.2	2,841	2,506	4,687	3,227
United States Total	2,263,230	748,923	33	503,542	100.0	20,000	15,824	23,689	20,000

Notes: Column (7) is percent timberland (from column 6) multiplied by 20 billion cubic feet, the estimated annual U.S. consumption. Consumption in column (10) is 2002 population multiplied by 69.5 cubic feet per capita. Results were rounded for tabular presentation, but not during computations.

- More professionals on the ground guiding decision-making.
- Greater involvement of nonindustrial private owners in managing their forests and selling timber through forest management plans.
- Improved forest health via proactive management to reduce negative impacts of disturbance by fire, insects, disease, weather, or other undesirable agents of change.
- Matching regional forest harvest levels to the area and productivity of forest resources.
- Estimating the “right-size” for commercial forest production by state and ecoregion based on forest resources. (e.g., Table 1).
- Changing the context of local debates away from isolated battles over individual timber sales toward addressing the question “How do we sustainably

produce a quantity of wood that is in balance with our consumptive pressures and our local forest resources viewed in a global context, now and in coming decades?”

These are not quick fixes. They initiate an ongoing journey, if it is a road we choose follow. It takes time to change attitudes, to build consensus, to build infrastructure, and to educate future citizens. Our current forests bear the imprint of management decisions made over the previous century; we inherit that legacy with all its opportunities and its constraints as we move ahead.

CONCLUSIONS

This paper has focused primarily on forest sustainability in terms of wood production and consumption. Wood or timber issues alone cannot define forest sustainability. Neither can they be overlooked. The availability (or unavailability) of wood products has an enormous impact on people's well being. Wood consumption is directly linked to timber harvest, and timber harvest directly impacts forests somewhere on the Earth. This creates specific constraints and opportunities related to forest sustainability in all its other (nontimber) dimensions. Only by addressing the impacts of consumption and production of wood at local, state, regional, national, and global scales can we move forward with realistic approaches to sustainable forestry.

It is my opinion that our current course of action with regard to timber production and consumption is not sustainable in a global context. In light of our abundant forest resources, it is difficult to see how the current imbalance between our production and consumption of wood products makes us an equitable partner with regard to global forest sustainability. A first step would be to bring our aggregate national volume of timber harvest into balance with our aggregate rate of domestic wood consumption, now and for future decades. There are many complementary ways to work toward that goal while considering all the other dimensions of forest sustainability.

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Desired Vegetation Condition and Restoration Goals in a Changing Climate: A Forest Management Challenge

Linda C. Brett

ABSTRACT

The importance of forest structure, function, and processes is well recognized by the research and management communities, and is an important consideration in formulating vegetative desired conditions in forest plans. The current policy guidance and the laws underlying them implicitly assume relatively stable ecosystems; but, climate science is telling us that we are building our forest management strategies and goals on shifting sands, and that

sustainability and restoration goals will not be met if we use forest conditions that developed over the last 500 years as the benchmark for reference conditions. This paper briefly reviews information from the 2007 IPCC report and other relevant climate-related research, existing agency planning guidance for formulating desired conditions, and suggests a framework for managers to use to incorporate climate change considerations into forest plans.

Keywords: climate change, forest plans, desired conditions

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INTRODUCTION

The importance of forest structure, function, and processes is well recognized in the research and management communities throughout the world as essential components of sustainable forest management. Ecological restoration, as used by the USDA Forest Service in the Restoration Framework (2007), is defined as “the process of assisting the recovery of an ecosystem that has been degraded, damaged, or destroyed” (Society for Ecological Restoration 2004). As first defined by the Society for Ecological Restoration, and later modified by the USDA Forest Service for its own use, healthy ecosystems generally exhibit these attributes:

- The suite of species and biological structures characteristic of the reference ecosystem are present and are free from non-natives to the greatest practicable extent.
- A physical environment that is representative of the reference ecosystem or, if degraded, is still capable of supporting native species and basic processes necessary for development along the desired trajectory.
- Disturbances indicative of the historic disturbance regime continue to operate within the range of variation (disturbance types, frequencies, intensities, etc). (Restoration Framework 2007: 1-2)

In the USDA Forest Service, the legal sideboards for sustainability and restoration are set by the National Forest Management Act of 1976 (16 U.S.C. 1600). Other laws, such as the Endangered Species Act of 1973 (16 U.S.C. 1531-1536, 1538-1540) and the Clean Water Act of 1977, as amended (33 U.S.C. s/s 1251 et seq.), add to, and reinforce, the legal mandate to manage for healthy and resilient forest ecosystems. Over the past 15 years, forest plans have placed increasing reliance on the concepts of vegetative desired condition. The relevant factors in determining vegetative desired condition include carefully chosen reference conditions, inferential constructions of historic vegetation or the natural range of variation, professional judgment, and social values. The 2005 planning rule provides explicit national policy guidance on the use of desired conditions as the centerpiece of ecologically and socially sustainable forest plans (Federal Register vol. 70, 2005:1023-1061).

The existing policy guidance and the laws underlying them implicitly assume slowly changing stable ecosystems where management impacts can be readily distinguished from natural processes. Increasingly, climate science is telling us that we are building our forest management strategies and goals on shifting sands, and that sustainability and restoration goals will not be met if we use forest conditions that developed over the past 500 years as the benchmark for reference conditions (Millar [and others] 2007).

The emerging scientific consensus on global climate change presents us with many challenges. In the past forest plans have commonly said that global climate change is “beyond the scope” of the plan. Clearly the emerging information on the effects of climate change on vegetation, hydrology, disturbance regimes, plant and animal species should be considered as we revise forest plans but, the devil is always in the details. How should forest plans address climate change issues? If we cannot rely on historically derived reference conditions to provide a baseline for forest plan desired conditions, then what information should we use to establish a narrative for future forest conditions that has a sound quantifiable scientific basis? How do we facilitate restoration in changing ecosystems if the line between management induced change and changes induced by climatic shifts is blurred?

GLOBAL CLIMATE CHANGE AND FORESTS

In February 2007 the Intergovernmental panel on climate change (IPCC) released Part I of its fourth in a series of assessments of global climate change. The IPCC report represents the consensus opinion of scientific experts from around the world on global climate change. The 2007 Summary for Policy Makers (IPCC 2007) is a sobering document. Among the findings presented:

Warming of the climate system is unequivocal, as is now evident from observations of increases in global average air and ocean temperatures, widespread melting of snow and ice, and rising global average sea level. (IPCC 2007:2)

Paleoclimate information supports the interpretation that the warmth of the last half century is unusual in at least the previous 1300 years. The last time the Polar Regions were

significantly warmer than present for an extended period (about 125,000 years ago), reductions in polar ice volume led to 4 to 6 metres of sea level rise. (IPCC 2007: 9)

Average Northern Hemisphere temperatures during the second half of the 20th century were very likely higher than any other 50-year period in the last 500 years and likely the highest in at least the past 1300 years. Some recent studies indicate greater variability in Northern Hemisphere temperatures....particularly finding that cooler periods existed in the 12th to 14th, 17th, and 19th centuries. (IPCC 2007:9)

For the next two decades a warming of about 0.2 degrees Centigrade is projected for a range of emission scenarios. Even if the concentrations of all greenhouse gases and aerosols had been kept constant at year 2000, a further warming of about 0.1 degree Centigrade would be expected.(IPCC 2007:12)

The IPCC report outlines several trends of particular interest for forest land managers including:

- Warmer and fewer cold days and nights over most land areas;
- Warmer and more frequent hot days and nights over most land areas;
- The frequency of warm spells/heat waves increases;
- More heavy precipitation events;
- The areas affected by drought increase;
- There is an increase in the frequency and intensity of tropical cyclones. (IPCC 2007:9)

The implications of these trends for extreme disturbance events, insect and disease outbreaks, greater vulnerability to invasive species, and permanent long-term shifts in vegetative patterns are evident to many in the research community (Westerling [and others] 2006; Millar [and others] 2007; Salwasser 2007; Neilson [and others], 2007; Joyce and Haynes 2007; Gucinski 2007). Among the trends noted in the literature we see significant increases in wildfire activity, cold tolerant vegetation moving upslope or disappearing in some areas, reductions in the area occupied by tundra species, and the explosive expansion of mountain pine beetle into British Columbia that threatens to move into eastern areas where this species was previously unknown.

Current research shows that climate is much more variable than is commonly understood and that this variability is expressed in nested temporal and spatial scales. (Millar [and others] 2007:32-39) provides an elegant summation of natural climatic variability and its implications for forest managers. There are three key points from this research that should be considered in forest planning:

- The past climatic record clearly shows that ecological conditions change constantly in response to climate. Plant and animal species will shift even in the absence of human influence. (Millar [and others] 2007:39)
- Species ranges and demographics are expected to be highly unstable as the climate shifts. (Millar [and others] 2007:30)
- There is no easy way to precisely predict these changes at the forest planning scale, although the science community is working on meso-scale models that will assist forest managers in forecasting vegetation trends under different climate scenarios (Gucinski 2007: 48-52).

DESIRED CONDITION FORMULATION IN FOREST PLANS

Desired conditions are the centerpiece of forest plans and sustainable forest management. Objectives, management strategies, and guidelines are formulated based on desired conditions for the landscape. The National Forest Management Act (NFMA) mandates that the agency manage national forests in a way that sustains ecosystem and species diversity over time. Ecosystem diversity is defined in 36 CFR 219.6 as "...the variety and relative extent of ecosystem types including their composition, structure, and processes." The Responsible Official is instructed to evaluate ecosystem diversity in the following way:

- Identify selected ecosystem characteristics;
- Assess their natural variation under historic disturbance regimes;
- Compare that information to existing and projected future conditions. (Forest Service Handbook (FSH) 1909.12 Ch. 43).

The handbook provides specific advice on developing the natural range of variation to be used in developing desired conditions for forest plans (FSH 1909.12 43.13).

This guidance is largely premised on assumptions that we are working with stable ecosystems or, if the ecosystem is unstable, the instability is largely the result of past management. We now know that the assumption that historical climatic and ecological conditions are a good model for future conditions is fatally flawed. Further, ecological processes and conditions are not the sole consideration in devising desired conditions, economic and social conditions also play a part. Desired conditions in forest plans must be based on sound science, but they are also negotiated with the public as part of the collaborative process.

For example: restoration of mountain meadows is a common desired condition in some parts of the west. Meadows are highly valued by the agency and the public for habitat, esthetic, and economic reasons. A paleo-ecologist might tell you that meadows are not a permanent landscape feature and, given the current warming and drying trends, many meadows will likely disappear in the next several decades, whatever management interventions take place to preserve them. Does that mean that the agency should not engage in meadow restoration? No, but it does mean that we should choose our restoration goals carefully, and that we should be candid with the public about the all the ecological processes in play. It may be that by protecting key ecological features, like meadows, we buy species some time to move and adapt to changing conditions, or it may be that our efforts are better invested elsewhere. Forest plans exist to make those kinds of strategic judgments.

Since the new planning guidance was issued in 2005, a number of forests have formulated proposed plan revisions based on this guidance. As you would expect, the desired conditions in these plans often do not consider the implications of climate change and, instead, largely base vegetative desired condition on reference conditions and historic ranges of variability as per the guidance. This gap has not gone unnoticed by the public. For example, consider this comment from the 2007 Western Montana Zone proposed land management plans content analysis report (WMPZ Proposed Land Management Phase Content Analysis Report 2006). The Western Montana Zone forests include the Lolo National Forest, Bitterroot National Forest, and the Flathead National Forest.

We find the complete lack of any mention or consideration of global climate change to be a serious shortcoming

of this plan. This is especially troubling given that the plan sets specific targets for vegetation conditions, yet those conditions seem an attempt to mimic a perceived condition of the past. For example, the Desired Future Conditions call for the distribution of living and dead trees to be “consistent with historic conditions”. The same is said for tree densities and fuel conditions. If the consensus of local, national, and international climate scientists can be believed, and if the documented climate trend in the Northern Rockies continues, the future will not be like the past. (WMPZ Proposed Land Management Plan Phase Content Analysis Report 2006:30)

This comment is one of several questioning the use of historic conditions as a model for desired condition. The comments register from the full range of public opinion including both advocates for active vegetation management and advocates of letting natural processes prevail. Given that climate change is real, is affecting the natural processes that drive ecological change, and that these changes are taking place much faster than we once believed possible, the agency cannot formulate desired conditions and restoration goals without considering climate change effects.

A MODEST PROPOSAL

The Forest Service and university natural resources research communities have developed considerable information on the potential consequences of climate change to forests and rangelands throughout the United States, along with some potential adaptation and mitigation strategies for forest managers. The sweep of the vegetation change forecast from the various models is breathtaking but, unfortunately, the information cannot yet be directly applied to the local level because the current models predictive power cannot be stretched that far and remain scientifically credible. More general model forecasts of changes in forest types at the regional level may be possible and useful to planners. But absent an accurate and precise scale-appropriate model, we are left with applying the best available science on climate change effects and our own professional judgment to the information and landscape vegetation models we do have for forest and rangeland conditions, and devising management strategies based on informed supposition. Here are a few suggestions on how we might begin this process.

TREAT DESIRED CONDITION AS A WORKING HYPOTHESIS

While many forest plans portray desired conditions, management strategies, and objectives that are fundamentally invested in the historic range of variability paradigm, eastern and southern forests are sometimes less invested in this approach because so much of the “original” ecosystem elements of these forests were irrevocably altered before they were included into the national forest system. Nevertheless all forests have taken the “range of variability” approach to some degree in formulating forest plans and in doing so; make at least a passing reference to historic conditions.

Instead of expending a lot of energy on developing a detailed vision of historic conditions, I would suggest we focus on existing conditions, apply what we do know about natural processes and the natural range of variability for existing forest types, the value of ecological diversity, and informed supposition on the potential effects of climate change on these factors to devise a working hypothesis of quantifiable desired conditions for the planning period. We do know how today’s climate differs from the reference condition. We could extrapolate that shift into the future at appropriate scales.

One thing is certain; we will know more about the effects of climate change on forests and rangelands in 10 years than we know today. NFMA anticipated that periodic reevaluation of forest conditions and our management strategies would be necessary. We probably can’t anticipate all the changes that may take place in 50 or 100 years because of climate change. A 10 to 15 year planning horizon is realistic for a working hypothesis that offers some prospect of actually achieving plan objectives. Working hypotheses can and do build on one another and are likely a more realistic approach than the current desired condition formulations.

In reality the desired conditions found in some current forest plan revisions will likely be difficult to attain considering the resources we have at our disposal to make landscape level shifts in ecological complexity, structure, and function. We should remember that the landscape and planning models we currently use are useful for more than providing the required estimates of sustained yield and timber sale program quantity. Planning teams should use the models for scenario planning as they develop desired

conditions with the public. Using the models for evaluating the options is a way we can discern whether the desired conditions, objectives, and strategies are realistic and likely to be effective over the planning period given the likelihood of climate change. Developing models to build credible local climate change scenarios is also advocated by (Millar [and others] 2007: 52) in their most recent paper entitled *Climate Change at Multiple Scales*.

FOCUS ON MITIGATING THE THREATS

Uncharacteristically severe fires along with extensive outbreaks of insects and disease are two harbingers of the large scale ecological changes we can expect from climate change. Indeed there is evidence that the large scale die-off of forest vegetation posited by Nielson [and others] 2007 may have already begun. (Breshears [and others] 2005) documents the abrupt die off of nearly 90% of pinyon pine in its existing range in the southwestern United States as the result of an unprecedented warm drought cycle within last 10 years. (Westerling [and others] 2006) point to increases in spring and summer temperatures and earlier spring snowmelt as the proximate cause of the sudden increase in large wildfires in the mid-elevation northern Rockies. The authors point to climate change as the likely suspect and, in fact, the IPCC (2007) also notes these general trends in its report on global climate change. Large-scale disturbances also favor invasive species, making a relatively intractable problem more difficult to address.

I advocate focusing our desired condition and restoration goals on mitigating known threats for several reasons.

- First, public policies such as the National Fire Plan are already in place, and the public is already aware of the concerns about fire threats, insects and disease, and invasive species. These concerns are likely to grow if, or when, the threats become even more severe than they are now. There is fairly widespread public agreement that action is needed to protect forests, even if there are disagreements about strategies and the tools.
- Second, our current efforts to use thinning and prescribed fires to restore forests and reduce fuels are also excellent strategies for addressing global climate change effects on forests.

- Third, restoration of healthier forest conditions with a bias towards preserving and enhancing ecological diversity is one of the few actions we can take that will ameliorate the impacts of a warming climate and provide species the opportunity to adjust to changing climate conditions as they have done over the millennia and allows forests to store more carbon; a fact that will be increasingly important to future management.
- Fourth, according to some authors (Salwasser 2007:14-15), we may need to favor a diversity of tree species in fire prone forests, keep stocking levels lower than full site occupancy, and perpetuate and encourage carbon storage in old growth through the use of longer rotations. It may also be desirable to revise our current seed zones to anticipate warmer conditions.
- Fifth, Millar [and others] 2007 argue for the creation of “porous landscapes” meaning large landscapes with continuous habitat and few physical and biotic barriers so that species, including plants, can move and adjust to changing conditions as much as possible (Millar [and others] 2007: 53). Clearly, public forest lands are one of the few places where such continuity is possible to maintain over the long term.

Salwasser (2007) suggests: “If we want diverse, productive and resilient future forests, we need to prepare them for a warmer future. And we need to look for ways forest resources can mitigate or ameliorate undesired climate change.”

TRIAGE

Millar [and others] 2007:53 uses the term triage to refer to making reasoned management choices between the various ecosystem components that may be lost as species, plant communities, and regional vegetation respond in their individual ways to climate change. These concepts under-score the importance of restoring ecological processes, rather than focusing on trying to maintain beloved landscape features.

Returning to the earlier example of Sierran meadow restoration, we can ask ourselves if a program of removing encroaching lodgepole pine is restoring a natural process or maintaining a landscape feature. Making reasoned choices

about where to focus desired condition and restoration goals is not simple. Even if the decision-maker has a reasonably clear picture of regional vegetation trends; legal mandates, public desires, and economic considerations will all play a role in that choice. In the end we will probably expend resources trying to preserve habitats that cannot be preserved, for any or all of the reasons listed above.

MITIGATING GREENHOUSE GASES

National level policy changes on regulation of greenhouse gases are clearly on the horizon. Several individual states have already set CO₂ reduction targets in the absence of federal regulation and Congress appears to be ready to consider CO₂ reduction targets, perhaps through a cap and trade system. Public forests will likely be asked to play a role in sequestering carbon. Private entities are already offering to plant trees on public lands provided that they can claim carbon credits for doing so.

Salwasser 2007:14-16 along with numerous others, points to a number of relevant forest management strategies that could play a role, particularly in devising strategies for meeting desired condition and restoration goals. These include:

- Increase forested land area
- Manage and protect forests from fire and insects to store more carbon per acre
- Capture more carbon in wood products
- Use mill waste, woody biomass for bio-based renewable energy
- Favor wood products over more energy-demanding materials
- Reward forest landowners for ecosystem services.

Clearly some of these suggestions refer to forest policy choices that are well outside the realm of forest plans, but other suggestions are relevant, and should be considered as part of forest plan revisions.

CONCLUSIONS

Change is upon us. Climate change is probably the key evolutionary force that sets the conditions for change in species and ecosystems. We are in a time of rapid climate change where the management models that look to historic conditions as a baseline and that assume stable ecosystems will not be useful for setting desired conditions and restoration goals. While the research and models for forecasting the effects of climate change are rapidly improving, the tools and techniques to accurately forecast vegetation changes at the local forest level do not yet exist. Nevertheless we need to use the information we do have to design realistic and quantifiable desired conditions for forest plan revisions and consider climate change in our restoration strategies.

As importantly, we need to educate ourselves and the public about the likely effects of climate change on forest resources and take action to mitigate those effects where it makes sense to do so. We have the opportunity to manage our forests, grasslands, and wood products to increase carbon sequestration and to offset fossil fuel use through bio-energy, bio-products, and bio-fuels. Innovative and scientifically credible silvicultural practices will be crucial to all of these efforts, if the nation is to have healthy and resilient forests well into the future.

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Proceedings of the 2007 National Silviculture Workshop

Learning, Connecting, Reconnecting: Summary of the 2007 National Silviculture Workshop

Russell T. Graham and Theresa B. Jain

NATIONAL SILVICULTURE WORKSHOP

The National Silviculture Workshop's genesis in 1973 was the result of forward thinking individuals within the Timber Management staff of National Forest Systems. Several national issues of the 1970s including, but not limited to: 1) the passing of the National Forest Management Act, 2) the mandate that all timber removal, reforestation, and timber stand improvement on Forest Service lands be prescribed by trained and certified silviculturists, and 3) a national policy of intensive timber management on Forest Service lands provided the impetus for "National Silviculture Work Conferences" (Nelson 1974, Gillespie 1977). In 1973, a group of 70 to 80 individuals, primarily from the Division of Timber Management along with several members of the Timber Management Research Staff of the Forest Service, met in Marquette, MI to discuss the theme of hardwood management (uneven-aged management). The organizers, the Silviculture Group from the Timber Management Staff in the Washington Office, recognized the need for Na-

tional Forest System Silviculturists and their associates to become acquainted and share information often of regional and national consequence. In addition, they recognized that Forest Service Research was an integral part of this information exchange so individuals from Forest Service Research were included in the Workshops. Proceedings followed the Workshops and were initially published by the Timber Management Staff in the Washington Office. These proceedings had minimal numbers printed, minimal review, lacked a consistent format, and no library deposition. Because the proceedings contained information addressing issues relevant to the practice and evolution of silviculture, Forest Service Research and in particular, the Station co-hosting the meeting, began to publish the proceedings in 1991. As a result, the proceedings are readily retrievable from libraries, meet the highest standards (e.g., peer, policy, statistics review) for Forest Service publications, and are available from Station Internet sites.

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The early meetings were held annually and their locations moved around the country and were hosted by different Regions and associated Research Stations (Table 1). These conferences addressed various topics of local, regional, and national concern. For example, several of the meetings in the 1970s addressed how forest management would be impacted by the National Environmental Policy Act of 1969 and how timber management and silviculture could fulfill the goals of the National Forest Management Act of 1976 (Doolittle 1975, Berntsen 1977) (Table 1). Uneven-aged management and uneven-aged silviculture were common topics throughout the life of the Workshop beginning in 1973 and are still topics in 2007 (Marquis 1978). Two research sponsored workshops were devoted to the subjects partly in response to the “alternatives to clearcutting issue” and a large portion of the Workshop in Petersburg, AK (1989) was devoted to the subject (Berntsen 1977, Forest Service 1990).

Silviculture and timber management are often considered synonymous by many within the Forest Service and those on the outside (Graham and Jain 2004). Even though timber management was central to most National Forest Management in the 1970s, the recognition of other resources such as water and wildlife were frequent topics during early Workshops (Schneegas and Sundstrom 1977, Lewis 2005). In 1987 the entire Sacramento Workshop was devoted to silviculture for all resources and the 2004 Workshop was devoted to silviculture in special places (Table 1). As these examples show, the practice of silviculture within the Forest Service has been very inclusive. Other evidence of inclusiveness in the Workshops is the multitude of disciplines both from National Forest Systems and Research, including but not limited to wildlife, forest ecology, hydrology, fisheries, soils, range, economics, social, management, and policy that have made presentations and attended the Workshops. By far the National Silviculture Workshop is one of the most well attended national meetings and the 2007 meeting in Ketchikan, AK continued this standard with a diversity of attendees and overflowing attendance.

2007 KETCHIKAN MEETING

In 1905, Gifford Pinchot suggested the Forest Service strive to provide the greatest good, for the greatest number, for the longest time (Lewis 2005). The dedication and commitment of Forest Service silviculturists exemplifies this

ethic and it is showcased biannually with the planning, execution, and attendance at the National Silviculture Workshop. The 2007 meeting was a return visit of the Workshop to Alaska. The 1989 meeting, in Petersburg (93 attendees) was hosted by the Tongass National Forest, Region 10, and the Pacific Northwest Research Station. The 1989 meeting overwhelmed the commercial accommodations available and the Forest Service Family opened their homes providing rooms and meals to several attendees. Ketchikan, being frequented by cruise boats, had sufficient infrastructure to support the meeting but it too felt the meeting's presence (121 plus attendees) (McDowell 2007). The meeting planners, who did a superb job, secured the Ted Ferry Civic Center overlooking downtown Ketchikan for the Workshop.

Issues, trends, and setting

In the welcoming addresses it was obvious that the issues, trends, and concerns facing the management of southeastern Alaskan forests reflect those of forests throughout the United States (Ginn this proceedings). The forests of Alaska, and throughout the United States, are changing and the expectations of local, regional, and national stakeholders are also changing. For example, the nearly 6.9 million hectare Tongass National Forest, the largest National Forest in the United States, began its modern timber program in the 1940s by producing Sitka spruce logs for airplane construction. As the need for airplane spruce rapidly declined there was a desire locally, regionally, and nationally to provide economic stability and job opportunities in Southeast Alaska (Byers 1960, Durbin 1999). The Tongass Timber Act of 1947 authorized the Forest Service to develop long-term timber supply contracts and in 1954 the Ketchikan Pulp Company opened a mill in Ketchikan and in 1959 the Alaska Lumber and Pulp Co.'s mill opened in Sitka. These two mills anchored the forest products industry in southeastern Alaska until both mills closed in 1990s impacting the lives of many Alaskans (Durbin 1999). These changes in Alaska reflected the changes that occurred in western Oregon and Washington when the timber industry receded in the late 1980s as result of forest simplification, loss of old-growth, and protection of spotted owl habitat (Byers 1960, Durbin 1999, Donovan and others 2005, Lewis 2005).

Although timber production has decreased dramatically in the United States in the last decade, its citizens consume approximately 20 billion cubic feet per year or 67 cubic feet per person compared to a global average per capita

Table 1. The evolution of the national silviculture workshop. Various national silvicultural meetings hosted by both National Forest Systems and Research developed into what is now a national biannual meeting that began in earnest in 1979 in Charleston, SC.

Year	Location	Title/theme
1973	Marquette, MI	Hardwood management
1975	Morgantown, WV	Uneven-aged silviculture and management in the eastern United States
1976	Eugene, OR	Stocking control and tree density
1976	Redding, CA	Uneven-aged silviculture and management in the western United States
1977	Flagstaff, AZ	Silvicultural implications of Section 4 of National Forest Management Act
1978	Missoula, MT	Silvicultural examination, prescriptions and related activities
1979	Charleston, SC	Shelterwood regeneration method
1981	Roanoke, VA	Hardwood management
1983	Eugene, OR	Economics of silvicultural investments
1985	Rapid City, SD	Success in silviculture
1987	Sacramento, CA	Silviculture for all resources
1989	Petersburg, AK	Silviculture challenges and opportunities of the 1990's
1990	Wenatchee, WA ¹	Genetics/silviculture workshop
1991	Cedar City, UT	Getting to the future through silviculture
1993	Hendersonville, NC	Silviculture from the cradle of forestry to ecosystem management
1995	Mescalero, NM	Forest health through silviculture
1997	Warren, PA	Communicating the role of silviculture in management of the National Forests
1999	Kalispell, MT	The role of silviculture in the stand, forest, landscape, and beyond
2001	Hood River, OR	Silviculture odyssey to sustaining terrestrial and aquatic ecosystems
2003	Grandby, CO	Integrated restoration of forested ecosystems to achieve multi-resource benefits
2005	Lake Tahoe, CA	Restoring fire adapted forested ecosystems
2007	Ketchikan, AK	Integrated restoration efforts for harvested forest ecosystems

¹Although the genetics workshop held in 1990 was not organized as a National Silviculture Workshop it took on such a flavor as both the genetics and silviculture disciplines from the throughout Forest Service were represented.

consumption of 21 cubic feet (Shifley this proceedings). A high proportion of the wood consumed in the United States is imported though the citizens of the United States enjoy an abundance of parks, wildernesses, and other wildlands (Haynes and Horne 1997). Alaskans reflect this trend with

the majority of their wood building products being produced in the lower 48 States despite the state's abundance of forest lands. This conundrum provides context for use, preservation, and restoration of United States forests in the 21st century.

Even though the large timber markets have declined in Alaska, similar to the declines in the western United States, the concept of ecosystem services is evolving into a forest management issue, for which silviculture is essential for its success. Ecosystem services include a wide variety of benefits humans gain from ecosystems and include such things as purification of air and water, mitigation of droughts and floods, detoxification and decomposition of wastes, and the sequestration of carbon in which forests and silviculture play important roles (Daily and others 1997, Collins this proceedings). The sequestration of carbon and the evolving carbon markets are integral to reducing emissions of the greenhouse gases that cause climate change. Both National Forest Systems and Forest Service Research

will have critical roles addressing this issue in the future. For example, the National Service Center in Fort Collins, CO in cooperation with Research and Development, provided the Forest Vegetation Simulator (FVS) the ability to produce and project carbon metrics for forests (Collins this proceedings, Dixon 2002).

United States forests have changed as the result of fire exclusion, introduction of invasive species, urbanization, land-use changes, and timber harvesting. As a result, there is considerable interest in restoring forests and the Forest Service has developed a framework outlining several principles for restoration, and the practice of silviculture is critical for these restoration activities (Crow this proceedings).

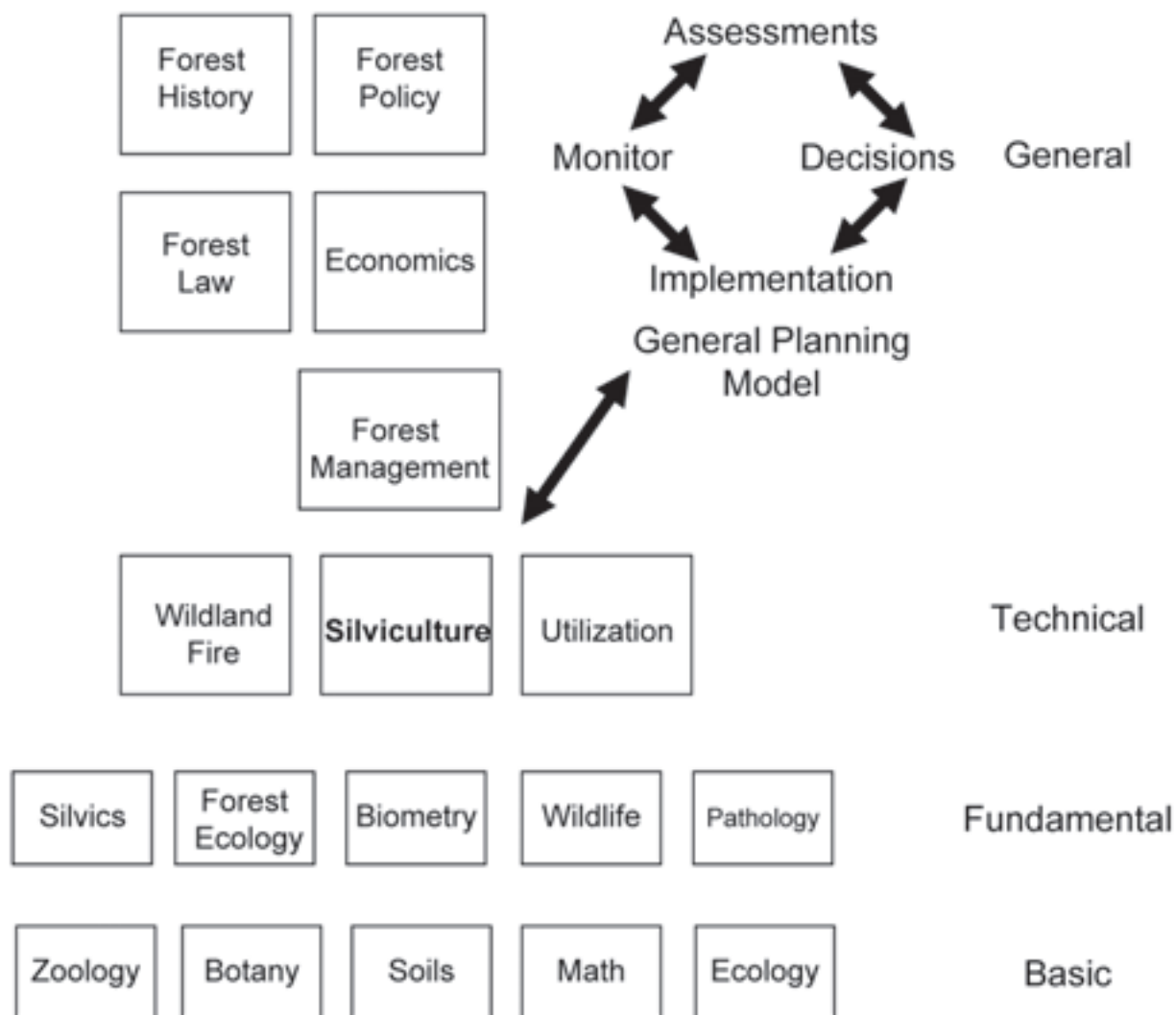


Figure 1. Silviculture is an integrative discipline incorporating basic, fundamental, and general knowledge into management frameworks (silvicultural systems) applicable for managing forests and woodlands (adapted from Nyland 2002).

There are several issues intertwined with forest restoration including, but not limited to, restoring degraded ecosystems, restoring watersheds valued for domestic water production, and restoring wildlife habitat. Restoring wildlife habitat is especially important for those wildlife species considered threatened and endangered under the Endangered Species Act. A prime concern inextricably connected to these issues and to forest restoration is the protection of communities and developments from wildfire (Graham and others 2004). As mentioned before, climate change is the overriding constraint or over-arching unknown influencing forest management policy, decisions, and silvicultural practices no matter the landowner or their objectives. For example, in Alaska there is evidence that the decline of yellow cedar is related to climate change. Specifically, warmer springs and reduced snow pack initiate premature dehardening and predispose trees to spring freezing injury (Hennon this proceedings). Even though other disciplines, and the public at large, have now (2007) brought climate change to the forefront, silviculturists have been cognizant of how it may impact silvicultural activities for years, as noted by SESCO (1990) at the 1989 National Silviculture Workshop.

Other disciplines may disagree, but silviculture is at the center of forest management (Figure 1) (Nyland 2002). Through the years silviculture has been closely allied with timber management and the production of wood crops (Evelyn 1664, Schlich 1906, Nyland 2002). As a result, the majority of the silvicultural practices were developed and honed over the years to grow trees efficiently and rapidly. As the early speakers in the Ketchikan Workshop articulated, this objective is very relevant. However starting with the 21st century exciting and challenging times are ahead for silviculture to respond to contemporary and evolving needs of forest management. More than ever people, enjoy, cherish, and often live in the forests the Forest Service manages. As such, the protection of communities, homes, and fire fighter safety are going to be of paramount importance. Developing and maintaining wildfire resistant and resilient forests and other issues such as biodiversity, carbon credits, maintaining sense-of-place, and old-growth will challenge silviculturists to be forward looking, adaptive, relevant, and responsive in the face of climate change.

Information and Knowledge

A key goal of the National Silviculture Workshop has always been for both practicing silviculturists and research silviculturists to exchange knowledge, information, and

information needs. This approach, which furthers the development of the art and science of silviculture began in mid-1600s in England (Evelyn 1664) and continuing with Schlich (1906) in the late 1800s in England and India. Over the last 34 years, the proceedings from the Silviculture Workshops traces the evolution of silviculture within the Forest Service and evidence indicates that such approaches have influenced current silvicultural thought (Nyland 2002, Tappeiner and others 2007).

A key need within all National Forests is for information and assistance in planning processes. The National Forest Management Act requires the Forest Service to prepare Forest Plans. Presently (July 2007), the rules for completing plans are somewhat uncertain, however, forest restoration will most likely be a theme for most plans along with sustainability and the maintenance of fire adapted ecosystems (Brett this proceedings). As Forests revise their plans, a suite of vegetation models are available, but they are rarely used to their full potential (Peterson and others 2007, Brett this proceedings). Other integral parts of the planning process are science use and scientist involvement during plan preparation. For example, the Tongass is in the process of finalizing their Plan and the Pacific Northwest Research Station, in 1997 and again in 2007, has been instrumental in this planning effort. Relationships between Research and National Forest Systems can transcend more than one Forest such as the Tongass. Research can be integral in developing projects by providing site specific information and models applicable for landscape planning. For example, the Daniel Boone and Bankhead National Forests partnered with the Southern Research Station to conduct large scale landscape studies using the Forest Plan and typical NEPA processes (Schweitzer and others this proceedings). This collaboration provides both National Forest and Research personnel a better understanding of mutual procedures, information needs, and most importantly proved information and protocols which benefit both organizations. Another key collaboration of Research and National Forests Systems is for Research organizations to participate and provide assistance in monitoring activities on National Forests. Monitoring has always been a part of forest management but it is often intermittent and inconsistent. For the most part monitoring protocols developed for timber production may not be adequate or appropriate for addressing many of today's management objectives such as forest restoration or maintaining wildlife habitat.

Often these types of issues cover multiple National Forests and/or large landscapes. Moreover, it has been suggested that by developing a unified monitoring strategy applicable to many Forests, comprehensive information could be assembled and progress towards restoration objectives would more likely be recognized and applied (Yaussy and others this proceedings).

In contrast to the forests of Alaska, the forests of the southern United States have been managed (harvested) since the 1600s and by the turn of the 20th century the removal of high value pines or high grading left many of them in a depleted condition. Restoration of these degraded conditions offer insight and/or principles that are applicable to the success of restoring forests anywhere. In general, restoration activities are dependent on the silvics of the species inherent to the site, the plasticity of the species, the abilities and insight of silviculturists (both current and past), and the application of evolving and developing scientific silvicultural knowledge (Guldin this proceedings). Again, from the experiences in the South it is obvious that restoration will be easier in some forest types than it will be in others regardless of the efforts of the silviculturist.

Some have suggested that climate change is impacting the size and extent of wildfires throughout the United States and elevating the level of insect and disease caused mortality in many forests (Dale and others 2001). In response to such forest changes, silvicultural strategies for restoring forests to a state in which they are more resilient and resistant to wildfires, insect outbreaks, and disease infestations have been suggested (Stanturf and Madsen 2005). In addition, invasive species (exotics) often exacerbate the impact wildfires have and controlling the abundance and distribution of invasives is also central to many restoration strategies (Graham and others 2004). For example, the introduction of cheat grass in the western United States has displaced native species, disrupted successional pathways, and made the forest and shrub lands extremely susceptible to intense and severe wildfires (Graham and others 1999). Also, throughout the West, white pine blister rust has devastated forests once dominated by 5-needle pines (e.g., western white pine, sugar pine, white bark pine). In concert with increasing forest resilience to wildfire, integrated management of western white pine and white pine blister rust has been the focal point of many restoration strategies (Jain and others this proceedings). Another example includes work

associated with developing strategies for controlling the hemlock woolly adelgid, an introduced species, in concert with activities aimed at restoring the central Appalachian and southern New England forests (Fajvan and others this proceedings).

Often forest restoration directly or indirectly implies the changing of species composition from one susceptible to fire, insects, and diseases to one more resistant and resilient (Stanturf and Madsen 2005). Through fire exclusion, grazing, and timber harvest the changes and the need for restoration are exemplified by restoring ponderosa pine and western white pine forests in the western United States and restoring Missouri Ozark forests in the eastern United States (Jain and others this proceedings, Nelson and Studyvin this proceedings, Zhang and Ritchie this proceedings). The latter restoration effort was designed to restore forests to ones containing assemblages of native species reminiscent of historical forests. This strategy was developed in collaboration with the Nature Conservancy and it, like many other efforts, recognized the role native disturbances played in maintaining and sustaining the Ozark Forests.

Linking the stand to the landscape is often the ultimate determinant of successful forest restoration. Finney (2001) suggested that forest structures could be modified and strategically placed in the landscape to disrupt a fire's progress. Jain and others (this proceeding) demonstrated how fuel treatments, even though they were small, could accomplish this task. The fuel treatments Jain and others developed and applied initiated a silvicultural system aimed at restoring the moist forests while providing structures and compositions relevant to many contemporary forest management objectives (e.g., sense-of-place, wildlife habitat, watershed protection). Another innovative fuel treatment discussed at Ketchikan was from the Bitterroot Forest where they used fuel treatments to reduce the fire hazard in a mixed conifer forest yet maintained hiking trails through an old apple orchard that people of the Bitterroot Valley cherish (Johnson this proceedings).

Riparian areas provide habitat for a wide range of game and non-game wildlife, they reduce the amount of sediment entering the stream, shade the stream, contribute woody material that increases hydraulic and structural complexity of the stream, provide habitat for aquatic and terrestrial organisms, and are integral to properly functioning water-

sheds. A very common method of addressing riparian areas in the proximity of timber harvest is through the establishment of stream management zones (buffers). These reserve areas have ecological benefits but they also have opportunity costs that are valuable information for both decision and policy makers (LeDoux this proceedings). A key to restoring watersheds, especially after disturbance, is prompt and robust vegetative regeneration (see field trip below). For example, in Alaska, mixing red alder with conifers when regenerating the rainforests following timber harvest offers prospects for increasing biodiversity and wildlife habitat abundance. Headwater streams with plentiful riparian alder had more invertebrates and supported more downstream fish biomass compared to basins with little or no riparian alder (Deal and Russell this proceedings).

Depending on the species, fish and wildlife habitat needs can be very site specific, but often habitat needs exceed stand and forest boundaries thus requiring silvicultural treatments to be relevant at these multiple spatial scales. For example, the habitat of the northern goshawk covers large landscapes often exceeding 3,000 hectares in size necessitating silvicultural treatments to transcend many different spatial scales. Although this is often recognized, integrating silvicultural activities across large areas often makes the preparation and implementation of silvicultural prescriptions more difficult. Nevertheless there are tools available to assist silviculturists in applying treatments at the site level while insuring they are relevant within the landscape (Youtz and others this proceedings). These silvicultural protocols developed for sustaining goshawk habitat represent and are presented as an uneven-aged silvicultural system.

Uneven-aged silviculture and uneven-aged management have been topics of discussion throughout the history of the National Silviculture Workshop. Probably nowhere in the United States has uneven-aged silviculture been practiced longer than in the South. For the past 70-years, selection systems have been used on the “Good and Poor Farm Forestry Forties” on the Crossett Experimental Forest producing high quality saw timber. Information from these studies has quantified forest development and produced multiple silvicultural strategies applicable for growing and sustaining old-growth characteristics of the southern pines. More importantly, these studies provide relevant information useful for treating the dry pine systems in the remainder of the United States (Bragg and others this proceedings).

Multiple canopy layers, heterogeneous horizontal structures, and a high interspersed of vegetation cohorts from young to old, are often desirable forest attributes for meeting wildlife and other contemporary forest management objectives. Most often uneven-aged silvicultural systems are a viable option for producing and maintaining these kinds of conditions (Graham and others 2007, Jain and others this proceedings, Youtz and others this proceedings). However, the conversion from even-aged structures to uneven-aged structures especially in stands developed from plantations can be problematic. It may take several entries and several decades to develop the desired forest conditions and for understory plant communities to differentiate in response to subtle differences in microclimate and light (Anderson this proceedings, Youtz and others this proceedings).

Regeneration and the disposition of high forest cover are often the major determinants for evaluating the success of silvicultural applications. High forest cover is fundamental for determining fire relations, many wildlife needs, product production, and the over-all character of the forest. Nevertheless regeneration is requisite for the success of all silvicultural systems and the abundance, juxtaposition, and species of regeneration are highly dependent on the biophysical conditions of a site and the amount and kind of natural or management disturbance to the forest floor and high forest cover (Jain and others this proceedings). Usually, canopy gaps or canopy openings are required for the regeneration of even the most shade-tolerant species, however, following abundant acorn crops, northern red oak can regenerate and develop even in the dense shade of closed-canopied forests (Dey and Miller this proceedings). Desired species with this attribute give rise to opportunities for managing the species and mixtures containing the species using a wide variety of even-aged and uneven-aged silvicultural systems.

The success of a silvicultural system is often predicated on effective and timely intermediate treatments (e.g., cleanings, weedings, thinnings). Intermediate treatments set the structure, composition, and developmental trajectories of the forest (Nyland 2002, Graham and others 2005). For example, in southeastern Alaska, clearcutting has been the dominant regeneration method and has been very successful giving rise to plentiful young forests. These young forests provide abundant opportunities for establishing studies to determine the impact thinning has on the development

of Alaskan rainforests (Bragg and others this proceedings, McClellan this proceedings) (see field trip below). Although this work was originally initiated to quantify forest growth and yield, these studies and other wide ranging plots throughout southeastern Alaska have been invaluable for understanding ecological and pathological relations and for calibrating forest development models (Poage and McClellan this proceedings). Not only in Alaska (Tongass-wide young-growth studies) but throughout the United States, these kinds of long-term studies have provided data on forest development which can inform a multitude of forest management decisions for a wide variety of objectives (e.g., wildlife habitat, sense-of-place, fuel treatments, timber production) (Bragg and others this proceedings, McClellan this proceedings, Graham and others 2005). With these manipulations of young stands not only are residual structures and compositions produced with value but they



Figure 2. Loading the ferry in Hollis, Alaska for the return trip to Ketchikan and watching whales, eagles, and sea lions on the stern of the ferry.



Figure 3. Young western hemlock and Sitka spruce forest growing on the Maybeso Experimental located on Prince of Wales Island.

also set up the forest for future intermediate treatments which often produce commercial forest products (Lowell and others this proceedings).

Field trip to Prince of Wales Island

Field trips have always been an integral part of the National Silviculture Workshop. They provide opportunities for attendees to see, touch, and learn about forests and silvicultural activities applied in them. Because the Workshop moves around the country, different forest types are visited and many different silvicultural techniques, presentations, and forest management objectives have been encountered. For example, at the 1973 meeting the attendees viewed the Dukes Experimental Forest, American Can Company lands, and areas on the Hiawatha National Forest near Marquette, MI displaying a variety of hardwood management techniques. In addition, while enjoying breakfast on a Convair 580 airplane, the group traveled to the Vinton Furnace Experimental Forest near Columbus, OH and then on to the Kaskaskia Experimental Forest near Marion, IL thereby viewing a variety of uneven-aged treatments on each Forest. This airplane travel set the standard for field



Figure 4. Terry Fifield discussing the role and importance of Totems to Native Alaskans in Craig, Alaska.

trip transportation, nonetheless school buses, tour coaches, and city transit buses are more the norm. More than once during National Silviculture Workshop field trips buses have broken down requiring alternate transportation and delaying the start of the ever important pre-dinner information exchange. For the Ketchikan Workshop a combination of busses and a ferry were used for the overnight field trip to Prince of Wales Island (Figure 2).

Prince of Wales Island offers many examples of silvicultural treatments representing timber production in Alaska. Mike McClellan hosted the first stop on the Maybeso Experimental Forest. The Forest was established in 1956 to investigate the effects of clearcut timber harvesting on forest regeneration and development, and on the physical habitat of anadromous salmon spawning areas. The watershed was treated with the first large-scale industrial clearcutting in southeast Alaska and nearly all of the commercial forest was removed from the watershed, including the riparian zone, from 1953 through 1960. The present forest consists of even-aged, second-growth Sitka spruce and western hemlock trees. The Old Tom Research Natural Area (an

old-growth forested watershed) is located nearby and it provides an old-growth control for studies in the Maybeso. Patti Krosse hosted a tour stop displaying late successional forests similar to those occurring in the natural area. This experimental forest and the surrounding area, provides abundant research opportunities for studying succession and development of Alaskan rainforests (McClellan this proceedings, Poage and McClellan this proceedings, Deal and Russell this proceedings) (Figure 3).

A theme running through the Workshop was restoration and probably no other forest attribute exemplifies this more than stream restoration. Pink, chum, and coho salmon, Dolly Varden, steelhead, and cutthroat trout are present throughout the streams on Prince of Wales. Sheila Jacobson and KK Prussian showed the attendees the efforts being made by the Tongass National Forest throughout southeastern Alaska to restore streams modified by past timber harvesting and road construction.



Figure 5. Kirk Dolstrum hosted the Viking Mill stop on the Prince of Wales Island.

Table 2. National Silviculture Workshop award recipients. At the National Silviculture Workshop beginning in 1989 in Petersburg Alaska individuals from National Forest Systems, State and Private Forestry, and Research and Development Deputy areas of the Forest Service have been recognized for their contributions to the practice of silviculture and forest management.

National Forest Systems and State and Private Forestry			Research and Development		
Individual	Title ¹	Current status	Individual	Title ²	Current status
1989 Petersburg, Alaska					
C.R. Blompquist	R-9 Reg. Silviculturist	Retired	Wyman Schmidt	INT Res. Silviculturist	Retired
John Standwerwick	R-10 Silviculture Group Leader	Retired	David Marquis	NE Res. Silviculturist	Retired
Jerry Hamilton	R-4 Reg. Silviculturist	Retired	Nelson Loftus	WO Forest Management Research	Retired
1991 Cedar City, Utah					
David Thomas	R-5 Eldorado Silviculturist	Deceased	Dean Debell	PNW Res. Silviculturist	Retired
John "Bob" Naumann	R-1 Reg. Silviculturist	Retired	Russell Graham	INT Res. Silviculturist	RMRS
1993 Hendersonville, North Carolina					
Richard Fitzgerald	WO, Asst. Director (Forest Products) Forest Management	WO	Phil Aune	PSW Program Manager	Retired
Richard Miller	WO Geneticist	Retired	David Loftis	SE Res. Silviculturist	SRS
Walter Knapp	R-6 Reg. Silviculturist	Retired			
1995 Mescalero, New Mexico					
Dick Bassett	R-3 Reg. Silviculturist	Retired	Wayne Shepperd	RMRS Res. Silviculturist	Retired
John Fiske	R-5 Reg. Silviculturist	Retired	Bill Oliver	PSW Res. Silviculturist	Retired
Bobby Kitchens	R-8, Reg. Silviculturist	Retired			
Ralph Johnson	WO FVS Group Leader FMC, Ft. Collins	Retired			
Milo Larson	R-3 Timber Management Director	Retired			
Dennis Murphy	WO Silviculturist	Retired			
1997 Warren, Pennsylvania					
Jack Amundson	R-4 Reg. Silviculturist	Retired	Clint Carlson	RMRS Res. Silviculturist	Retired
Barry Bollenbacher	R-1 Reg. Silviculturist	R-1	Carl Edminster	RMRS Res. Silviculturist	Retired
Frank Burch	WO National Silviculturist	WO	Bob Powers	PSW Program Manager, Res. Soil Scientist, Res. Silviculturist	PSW
Ken Denton	R-6 Reg. Silviculturist	Retired	Susan Stout	NE Res. Silviculturist	NRS
Susan Gray	R-2 Reg. Silviculturist	R-2			
Mike Landram	R-5 Reg. Silviculturist	R-5			
Doug MacCleery	WO Asst. Director (Policy) Forest Management	WO			
Richard Teck	WO NRIS Project Manager, FMC, Ft. Collins	WO			

¹FMC = Washington Office Forest Management Service Center, Fort Collins, CO, Reg. = regional position, WO = Washington office.

²Intermountain (INT) and Rocky Mountain (RM) Research Stations combined to form the Rocky Mountain Research Station (RMRS) (1997), the North Central (NC) and Northeastern (NE) Research Stations combined to form the Northern Research Station (NRS) (2007), and the Southeastern (SE) and Southern (SO) Research Stations combined to form the Southern Research Station (SRS) (1995), Res. = research position, WO = Washington Office.

Table 2 cont. National Silviculture Workshop award recipients. At the National Silviculture Workshop beginning in 1989 in Petersburg Alaska individuals from National Forest Systems, State and Private Forestry, and Research and Development Deputy Areas of the Forest Service have been recognized for their contributions to the practice of silviculture and forest management.

National Forest Systems and State and Private Forestry			Research and Development		
Individual	Title ¹	Current status	Individual	Title ²	Current status
1999 Kalispell, Montana					
Peter Laird	R-1 Reforestation/Nursery Specialist	Retired	Dick Tinus	SRS Plant Physiologist	Deceased
Gary Dixon	WO, FVS Group Leader, FMC, Ft. Collins	WO	Phil McDonald	PSW Res. Silviculturist	Retired
			Gary Fiddler	PSW Forester	Retired
			Jim Chew	RMRS Forester	RMRS
2001 Hood River, Oregon					
Fred Zensen	R-6 Reforestation/Nursery Specialist	R-6	Ray Shearer	RMRS Res. Silviculturist	Retired
			Jim Jenkinson	PSW Res. Plant Physiologist	Retired
			Bob Curtis	PNW Res. Silviculturist	Retired
			Nicholas Crookston	RMRS Forester	RMRS
2003 Grandby, Colorado					
Brian Ferguson	R-4 Reg. Silviculturist	Retired	Jim Guldin	SRS Res. Silviculturist	SRS
Mary Frances Maholovich	R-1 Reg. Geneticist	R-1	Kurt Gottschalk	NE Res. Silviculturist	NRS
Monty Maldonado	WO Reforestation/ Nursery /Genetics Programs	WO	Paul Johnson	NC Res. Silviculturist	Retired
Tom Tibbs	R-8 Reg. Geneticist	Retired			
2005 Lake Tahoe, California					
William "Bill" Jones	R-9 Reg. Silviculturist	Retired	Jim Barnett	SRS Res. Silviculturist	Retired
Glenda L. Scott	R-1 Reforestation/Nursery Specialist	R-1			
Tom Landis	National Nursery Specialist, State and Private Forestry	Retired			
Jim Russell	R-10, Tongass Silviculturist	R-10			
2007 Ketchikan, Alaska					
Marlin Johnson	R-3 Forestry Asst. Director	R-3	Terrie Jain	RMRS Res. Silviculturist	RMRS
Kathy Sleavin	WO FSVeg Project Coordinator, FMC Ft. Collins	WO	Steve Shifley	NRS Res. Silviculturist	NRS
Dave Evans	R-5 Lassen Silviculturist	R-5			
Bill McArthur	R-6 Reg. Silviculturist	R-6			

¹FMC = Washington Office Forest Management Service Center, Fort Collins, CO, Reg. = regional position, WO = Washington office.

²Intermountain (INT) and Rocky Mountain (RM) Stations combined to form the Rocky Mountain Research Station (RMRS) (1997), the North Central (NC) and Northeastern (NE) Research Stations combined to form the Northern Research Station (NRS) (2007), and the Southeastern (SE) and Southern (SO) Research Stations combined to form the Southern Research Station (SRS) (1995), Res. = research position, WO = Washington Office.

Traveling throughout southeastern Alaska, and in particular on this field trip, the current and historic reverence Native Americans have for forests and their intimacy with all natural resources was very evident. Totems, their display and carving are special to Alaskan Native Americans and the Workshop attendees were shown by Jon Rowan and Terry Fifield how totems are carved and displayed (Figure 4). As stated earlier the extent and impact forest industries have in southern Alaska has changed dramatically with the closing of the pulp mills. However a limited number of saw mills do harvest forests and produce wood products (Figure 5). The Forest Service is trying to support these kinds of small and essential forest industries with the development of the 2007 Plan for the Tongass.

The town of Craig is on the opposite side (western) of Prince of Wales Island from the Inter-Island Ferry dock at Hollis therefore, the field trip members traversed the width of the Island with the evening dinner and accommodations being in Craig. A local blue grass band provided music during dinner at the Craig Community Center and the lodging in Craig was creatively arranged by the Workshop planners. No bus break downs or delays getting to dinner were encountered, allowing all to have a good time at the pre-dinner information exchange and the post dinner wrap-up late into the evening. These informal information exchanges are a key element of the Silviculture Workshops and provide quality time for all of the participants to interact, make new friends, and rekindle old associations. The three plus hour ferry and bus ride to and from Craig provided high-quality time to reflect on the field trip, the role of silviculture in forest management, and the information presented at the Workshop. In addition, Jim Baichtal described the karst geology and caves of the Tongass, Terry Fifield discussed the history of the Thorne Bay Logging Camp, and Gary Lawton described the recreational opportunities on Prince of Wales. These excellent presentations on the ferry competed with the whale, eagle, and sea lion watching occurring on the stern of the ferry (Figure 2).

Workshop Banquet

The food caterers at the Ted Ferry Civic Center did an excellent job the entire Workshop and the Halibut Oscar served at the banquet was superb. Continuing the trend of understanding the native peoples of southeastern Alaska, Terry Fifield did an excellent job of showing how the Civilian Conservation Corp restored totems in Southeast Alaska beginning in 1938 and continuing until 1954. The

Ketchikan Indian Community Intertribal Dancers sang, drummed, and danced to an appreciative audience and enticed several to join the dancers on stage.

NATIONAL SILVICULTURE AWARDS

The practice of silviculture has several aspects but most importantly its aim is to sustain forests through their establishment and tending to fulfill the management objectives of the forest landowner, and in the case of the Forest Service, for the citizens of the United States (Figure 1). Forest management is a long-term endeavor as articulated by the likes of Evelyn in the 1600s, Schlich in the 1800s, and Nyland in 2002. Because of this long-term nature of forestry it takes individuals committed to the practice of silviculture and to the forests for which they are caring. At each Workshop since 1989 in Petersburg, led by the Silviculture Group in the Washington Office, individuals showing outstanding contributions to the practice of silviculture have been acknowledged (Table 2). Six such individuals were recognized at the Workshop in Ketchikan on May 10, 2007 including Terrie Jain, Research Forester, Rocky Mountain Research Station, Moscow, ID, Steve Shifley, Research Forester, Northern Research Station, Columbia, MO, Marlin Johnson, Region 3, Assistant Director for Forestry and Forest Health, Albuquerque, NM, Kathy Sleavin, Washington Office, FS Veg Project Coordinator, Ft. Collins, CO, Dave Evans, Forest Silviculturist, Region 5, Lassen National Forest, Susanville, CA, and Bill McArthur, Region 6, Regional Silviculturist, Portland, OR. These individuals exemplify the outstanding people the Forest Service has committed to providing the greatest good, for the greatest number, for the longest time.

CONCLUSION

People, experience, knowledge, and research are essential for an organization to succeed and for the Forest Service these requisites have proved to be invaluable. The biannual National Silviculture Workshop provides a forum where individuals can share information, learn, develop relations, and further their professional and personal growth. The Ketchikan meeting was a challenge for the organizers but Region 10, Pacific Northwest Research Station, and the Tongass Forest did a superb job of hosting the Workshop. The ferry and the over-night stay in Craig were integral parts of the field trip, once again raising the bar for future

field trip successes. As with all National Workshops many new faces were evident and others were conspicuous in their absence. Nevertheless, the overlap of individuals attending past meetings, led by Dick "Fitz" Fitzgerald who was on that Convair 580 for the field trip in 1973, with new attendees this year provides continuity, learning, and institutional memory. This strong tradition will continue in 2009 when Region 4 and the Rocky Mountain Research Station host the National Silviculture Workshop.

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Understory Vegetation Responses to Initial Thinning of Douglas-fir Plantations Undergoing Conversion to Uneven-Age Management

Paul D. Anderson

ABSTRACT

Since the mid-1990's, several studies have been undertaken to evaluate thinning treatments designed to promote development of complex forest structure in order to enhance ecological functioning and biological diversity. Most of these studies have focused on forest responses to one or possibly two thinning entries. In contrast, this study, the Uneven-aged Management Project (UAMP), is evaluating alternative silvicultural regimes for converting young Douglas-fir stands to mixed-species, uneven-aged condition. Conversion is to be achieved through regulation of stocking with repeated thinnings over an indefinite period.

The relatively low intensity thinning treatments applied as a first entry in the conversion process had little impact on the abundance, size, or diversity of understory vegetation. Disturbance resulted in short-term decreases in understory vegetation cover, particularly tall shrubs. However, within five years of treatment, understory vegetation abundance returned to approximate pretreatment condition. Regardless

of treatment, shrubs and ferns dominated the understory with coverage that was two-three times that of forbs and grasses. Species richness averaged near 12 species per 0.1-ha plot before treatment and five years post treatment. Community composition was dominated by a few very abundant species regardless of treatment. Post-thinning increases in conifer regeneration were consistent with the density of underplanting. Substantial increases in tree regeneration occurred in the light thinning treatment due to an increased seedling density of the shade-tolerant bigleaf maple. The persistence of planted and natural seedlings of less shade-tolerant species remains to be determined.

The general lack of understory vegetation response to the thinning treatments was likely due to the inherent resistance and resilience of the plant communities to disturbance, as well as the low intensity of disturbance attributable to the treatments. Over time, repeated frequent thinning entries may challenge this apparent community stability.

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

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INTRODUCTION

Thinning and uneven-aged management in douglas-fir

The Northwest Forest Plan established a management goal to enhance the development of late-successional characteristics in young second-growth forests on a large portion of federal lands in western Oregon and Washington. Currently there are several hundred thousand hectares of 40-to-80 year-old plantation forests on federal land (Cissel et al. 2006). These predominantly Douglas-fir (*Pseudotsuga menziesii* (Mirb.) Franco) forests have developed following large-scale disturbances associated with stand-replacing fires in the late-1800s and early-1900s and subsequent regeneration harvests in the 1950s through the 1980s. They were managed for timber production as a primary objective. Slash and other woody debris was removed prior to reforestation and subsequently competing vegetation was controlled and stands were pre-commercially thinned to enhance survival and growth of conifer regeneration. As a result of these silvicultural practices, relatively uniform, single-species high density stands have developed. These young forests typically lack the legacy features and the structural and biological complexities characteristic of late-successional forests and are poorly suited to supporting the northern spotted owl (*Strix occidentalis cuarina* (Merriam)), and other wildlife species (Carey 1995, Lindermayer and Franklin 2002).

Without intervention, these young Douglas-fir forests may remain in the stem exclusion phase of stand development (Oliver and Larson 1996) for an extended period (Andrews et al. 2005) and therefore delay the onset of mortality process required to create canopy openings conducive to development of understory vegetation and large, complex tree crowns associated with late-successional forests (Franklin and Van Pelt 2004). Retrospective analyses have indicated that many of the existing old-aged stands tended to be less dense during their early stages of stand development and therefore individual trees grew faster, maintained deeper crowns, and developed larger limbs (Tappeiner et al. 1997). Existing old-growth stands contain larger amounts of large woody debris than is present in current young forests. The less dense, more heterogeneous late-successional forests tended to develop multiple cohorts of trees lending vertical connectivity between understory shrub layers and the predominant tree canopy. Finally, natural disturbances and subsequent stand dynamics resulted in a small proportion

of trees that developed into dominants extending above the general canopy. Combined, these features of late-successional forests represent a more structurally diverse ecosystem than current young-growth Douglas-fir forests and they provide a suite of ecological and social values that differs from that of young forests.

A variety of silvicultural treatments may be undertaken in young Douglas-fir forests to enhance development of compositional and structural features associated with older-aged forests. Use of thinning to respace young stands is being widely explored both by forest managers and researchers (DeBell et al. 1997, Tappeiner et al. 2002). Variable retention thinning coupled with gap creation and leave islands have been used to increase spatial heterogeneity within stands (Cissel et al. 2006). Mechanical treatments and biological agents have been explored for creating snags. Understory planting of conifers and hardwoods has been used to establish additional cohorts of trees (Chan et al. 2006, Maas-Hebner et al. 2005).

Although many studies have been undertaken to evaluate thinning treatments designed to promote forest diversity, uneven-aged management, which maintains continuous forest cover and diverse forest structure over time, has not been extensively investigated in the Pacific Northwest, particularly in young Douglas-fir forests (Emmingham 1998, Emmingham et al. 2007). One early trial conducted in the 1930s concluded that selective cutting (individual tree and small group selection) applied to mature Douglas-fir was a failure (Isaac 1956), as regeneration of Douglas-fir was not adequately attained and there was a loss of growth and volume production in the residual stand, much of this loss associated with damage (Curtis 1998, Isaac 1956). From a retrospective assessment of 50-80 year-old Douglas-fir stands managed by selection thinning, Miller and Emmingham (2001) observed that although the volume growth of the residual stand was less than that for a more typical low thinning, growth per unit growing stock was similar and in some cases the residual overstory density was low enough to result in regeneration of the moderately shade-intolerant Douglas-fir.

With an increased emphasis on ecological function and biodiversity as management objectives on some forested lands, responses of understory vegetation to overstory thinning has become a recent focus of research (Muir et

al. 2004; Thomas et al. 1999; Thysell and Carey; Wilson and Puettmann 2007). One common tenet is that increased diversity of structure and composition of plant communities will lead to increases in faunal components of biodiversity (Lindh and Muir 2004). Wildlife respond to habitat features at several hierarchical scales reflecting their respective ranging, foraging, sheltering and social traits (Wilson and Puttmann 2007). Thinning in forest stands alters the physical environment in terms of canopy openness, and therefore visibility, access, and cover (Carey 1996), as well as light transmittance and energy balance. In addition to direct impacts on habitat suitability, these changes in physical site and associated resource availabilities may be drivers in the development of understory plant communities important to providing food, cover or nesting. A variety of bird and mammal fauna in the Pacific Northwest have been shown to require a variety of canopy and understory vegetation conditions with preferences differing among species (Hayes et. al. 1997, Hagar et.al. 2004). As a result of increased structural and compositional variation at the stand scale, forests may provide a greater range of functions for a greater number of organisms.

Objectives

The objective of this study was to determine the efficacy of three alternative thinning treatments for increasing the abundance and diversity of understory vegetation in 30-40 year-old, even-aged, Douglas-fir plantations at mid-elevation on the west-slope of the Cascade Range, central Oregon.

Specifically, the following questions related to stand development occurring during the five-year period following the initial thinning entry in the conversion process are addressed:

- 1) Did the different thinning treatments result in substantially different residual stand densities and canopy covers?
- 2) Were there differential responses to the thinning treatments in the abundance and diversity of understory vegetation?
- 3) Did the thinning treatments result in initiation of a new cohort of trees?

METHODS

Study location

The study was conducted on the H.J. Andrews Experimental Forest, a 6,350 ha 6th-field watershed on the Willamette National Forest, near Blue River, Oregon. The H.J. Andrews is located within the western Cascades physiographic province (Franklin and Dyrness 1988). Spanning elevations from 440 to 1090 m, the H.J. Andrews is forested primarily by Douglas-fir, western hemlock (*Tsuga heterophylla* (Raf.) Sarg.) and western red cedar (*Thuja plicata* Donn ex D. Don) at lower elevations and transitions to a mix of noble fir (*Abies procera* Rehd.), Pacific silver fir (*Abies amabilis* (Dougl.) Forbes), Douglas-fir and western hemlock at higher elevations. Climate is maritime with mild, wet winters and warm, dry summers. Average precipitation totals about 2,300 mm but only about 6% of that occurs in the summer months.

Sixteen stands were selected from all stands on the H.J. Andrews that were 30-40 years of age, a minimum of 8 ha in area, a mid-point elevation less than 915 m, and having no other ongoing experimental activity. The experimental stands actually ranged from 7 to 14 ha in area, 540 to 920 m elevation, 13 to 45 percent slope, 208 to 272 cm annual precipitation and 26.5 to 40.2 m site index (King 1962). Soils were generally gravelly loams or gravelly clay loams of the McKenzie River, Frissell or Carpenter series. Although Douglas-fir dominated, western hemlock, western redcedar and Pacific yew (*Taxus brevifolia* Nutt.) co-occurred in lesser abundance. Predominant plant associations, based on potential natural vegetation (McCain and Diaz 2002; Hawk 2005) included TSHE/RHMA/GASH (western hemlock/rhododendron/salal), TSHE/RHMA/MANE2-GASH (western hemlock/rhododendron/Oregon grape-salal) and TSHE/POMU (western hemlock/sword fern). In total, the study encompassed about 140 ha of approximately 611 ha of similarly aged Douglas-fir stands on the experimental forest.

The stands originated following clearcut harvest between 1950 and 1960. Two to five years following harvest and prescribed burning, the units were either aerial seeded, planted, or seeded and planted with Douglas-fir. Most of the stands were precommercially thinned in the late 1960s and early 1970s.

Experimental treatments

The experiment consisted of three thinning treatments, each applied as the first entry in a prescribed multiple entry conversion regime, and an unthinned reference treatment. The three thinning treatments included a light thinning, a light thinning with small gaps, and a heavy thinning. Due to initial variation in average tree size and density among stands, all prescriptions were based on a relativized measure of density – Relative Density (RD, Curtis 1982) – calculated for trees 12.7 cm d.b.h. and larger. The three thinning treatments are summarized below:

Light thinning consisted of a uniform thinning of stands from below toward a target density of RD 4.3 (RD 30 imperial). Stands were underplanted with approximately 550 seedlings per hectare using an even mixture of Douglas-fir, western hemlock, western redcedar, and western white pine (*Pinus monticola* Dougl.). When stands reach RD 7.2 (RD 50 imperial), they will be rethinned to RD 4.3 and thinning will occur in all diameter classes. Re-entry intervals of about 10 years are expected.

The group selection treatment consisted of creating 0.05 to 0.1 ha gaps over 10 percent of the stand and thinning the remainder toward a target density of RD 4.3 as in the light thinning. Stands in this treatment were underplanted throughout as described for the light thinning. Subsequent entries will create additional gaps, 10 percent of the area with each entry and will be triggered by the attainment of RD 7.2 in the thinned matrix of the stands. With each entry, expected to be at approximately ten year intervals, gap size will increase as a function stand height.

The heavy thinning treatment consisted of a thinning from below toward a target density of RD 2.9 (RD 20 imperial) and underplanting. A second thinning is prescribed when the combined RD of the residual overstory and the newly established regeneration cohort reaches RD 6.1 (RD 40 imperial). The second thinning will decrease the RD of the combined cohorts to RD 2.9. Subsequent entries, expected to be at 15 to 20 year intervals, will be performed to maintain the combined RD of all cohorts between minimum and maximum thresholds of 2.9 and 6.1, respectively.

Although the three treatment prescriptions were formulated to achieve the common goals of increasing structural

diversity and enhanced understory vegetation development, they differ in expectations for conifer regeneration. It was expected that the light thinning combined with understory planting would result in nearly continuous recruitment of additional tree cohorts of both planted and natural origin with a shift in species composition to the more shade tolerant western hemlock. The group selection treatment was envisioned to provide areas of canopy opening adequate to recruit the intermediate shade tolerant Douglas-fir, although the gaps themselves may become competitive environments for tree regeneration if they become occupied by shrubs. The heavy thinning was envisioned as an intermediate condition in which more persistent opening of the canopy may provide an opportunity for recruitment of both intermediate and tolerant conifers while limiting the potential for large increases in competing understory shrubs.

The 16 stands were grouped into four blocks of four stands each. The four experimental treatments were assigned at random to the stands within each block. Thus the experimental design was that of a randomized complete block. Stands were harvested between April 1 and December 1, 2000 predominantly by cable yarding systems, although there was a small portion of the study where tractor yarding was done. Underplanting occurred in the spring of 2002.

Vegetation measurements

Vegetation abundance and composition were measured using a nested sampling design. Breast height diameter (d.b.h.) of all live trees ≥ 5 cm d.b.h. and snags ≥ 10 cm d.b.h. were measured on a 0.1-ha circular plot. Percent cover by tall shrubs and small trees (< 5 cm d.b.h.) was measured by species along two 14.5-m planer transects per 0.1-ha plot. For low shrubs, forbs, ferns and grasses, percent cover by species was ocularly estimated from 0.1-m² (0.2 m by 0.5 m) subplots at eight equally spaced locations along each of the two 14.5-m transects. Tree seedlings and saplings (stems < 5.0 cm d.b.h.) within two 21.75-m² (1.5 m by 14.5 m) strip plots per 0.1-ha plot were tallied by species and height class (0.1 to 0.2 m, 0.21 to 0.5 m, 0.51 to 1.0 m, 1.01 to 2.0 m, 2.01 to 3.0 m, > 3.0 m). Canopy cover was estimated using a moosehorn densiometer at plot center and at four points located 10.5 m radial distance from plot center in the cardinal directions.

From five to 13 permanent vegetation sample plots were established in each stand. The target sampling intensity for the unthinned, light thinning and heavy thinning treat-

ments was seven percent of the stand area; for the group selection treatment it was 15 percent of the stand area. Pretreatment measurements were made in the summer of 1997 or 1998, and post-treatment measurements were made in the summers of 2001, 2003, and 2005.

Statistical analyses

Overstory tree, canopy cover and understory vegetation data were analyzed by mixed model analysis of variance. Thinning treatment was a fixed effect and block was a random effect. Pretreatment data were analyzed independent of post-treatment data. Post-treatment data were analyzed as repeated measures with block nested within treatment being the repeated subject and using autoregressive or unstructured covariance as dictated by individual data sets. Given the low number of replications in the operational treatment implementation, a probability threshold of $p=0.1$ was used to determine statistical significance.

Vegetation abundance response to treatment was analyzed by four growth-form strata (small trees, tall shrubs, low shrubs, and forbs, ferns and grasses). Seedling and sapling density data were collapsed from six height classes to four (0.1 to 0.2 m, 0.21 to 0.5 m, 0.5 to 2.0 m, and >2.0 m) and were log transformed prior to analysis. Among repeat measurement events, there were apparent crew-biased differences in ocular cover estimates for low shrubs, forbs, ferns and grasses derived from the 0.1-m² subplots. Therefore, for analyses of these data, mean percent cover for each treatment was relativized to that of the unthinned control. Specifically, for each repeat measurement and block, the differences in mean covers between each thinning treatment and the unthinned control were calculated and those differences were used as data for the repeated measures analysis of variance. In effect this relaxed the broad assumption that all crews had the same accuracy of ocular estimation; rather, it assumed that in any given year of measurement and for any block, the accuracy of estimation was the same for all treatments

Understory vegetation diversity response to treatment was assessed for the combined forb, fern, grass, low shrub, tall shrub and small tree lifeforms. Due to identification uncertainties, some putative species of the same genus were collectively recognized as a representing a single taxonomic entity (eg. two putative species of the genus *Galium*). For convenience, we include these aggregated taxa in our use of the term species. Diversity was quantified as the total

number of species (N_0) and the effective numbers of species (N_1, N_2), the latter two being abundance weighted measures interpretable as the number of abundant and very abundant species present, respectively (Hill 1973, Ludwig and Reynolds 1988). Evenness, the degree to which species comprising a community are equally represented, was estimated as the ratio of the numbers of very abundant to abundant species (N_2/N_1 ; Hill 1973). Indices of species diversity and evenness were analyzed by repeated measures analysis of variance as described above.

The relationship between overstory canopy cover and relative density by live trees >12.7 cm d.b.h. was modeled by fitting a second-order polynomial regression to stand averaged data from all treatments and measurement events. Degree of association between overstory cover and tree seedling density for each measurement period was estimated by Pearson correlation.

Analyses of variance, regression modeling, and correlation analyses were performed using Proc Mixed, SAS® v. 9.0 (SAS Institute, Inc., Cary, NC).

RESULTS

Overstory

Prior to treatment, stand density, basal area and average diameter breast height ranged among treatment units from 490 to 915 tph, 29 to 40 m²/ha, and 21 to 29 cm, respectively. Stands in the unthinned treatment had slightly greater average pretreatment density and lesser average tree diameter than stands receiving the three thinning treatments (table 1) however, differences among treatment means were not statistically significant. Relative density (Curtis 1982) calculated on a metric basis ranged among the sixteen stands from 5.7 to 7.6 (RD 40 to 53 imperial). Pretreatment relative density averaged across stands within treatments (table 1) did not differ among treatments.

Initial post-treatment measurements were made in summer, 2001, the first growing season following harvest. Density, basal area and relative density of the unthinned stands increased slightly from pretreatment (table 1). Basal area of the light thinning, group selection and heavy thinning treatment were decreased by harvest an average of 22, 21, and 33 percent, respectively. Although the residual

Table 1. Mean (standard deviation) pretreatment (1998) and posttreatment (2001) stand conditions for trees greater than 12.7 cm dbh for each experimental treatment.

Treatment		DBH	Basal area	Density	Curtis (1982)
					Relative density
			Square meters per hectare	Trees per hectare	
Unthinned	Pretreatment	25 (3.8)	33 (1.8)	682 (190)	6.6 (0.55)
	Posttreatment	25 (4.0)	37 (1.5)	710 (198)	7.2 (0.55)
Group selection	Pretreatment	29 (0.9)	38 (2.3)	550 (64)	6.9 (0.51)
	Posttreatment	31 (1.4)	30 (2.9)	391 (20)	5.4 (0.43)
Heavy Thinning	Pretreatment	26 (1.4)	33 (3.3)	566 (45)	6.3 (0.50)
	Posttreatment	29 (1.8)	22 (4.3)	311 (52)	4.0 (0.71)
Light thinning	Pretreatment	27 (2.1)	36 (2.6)	578 (72)	6.8 (0.43)
	Posttreatment	30 (3.7)	28 (2.8)	371 (70)	5.0 (0.48)

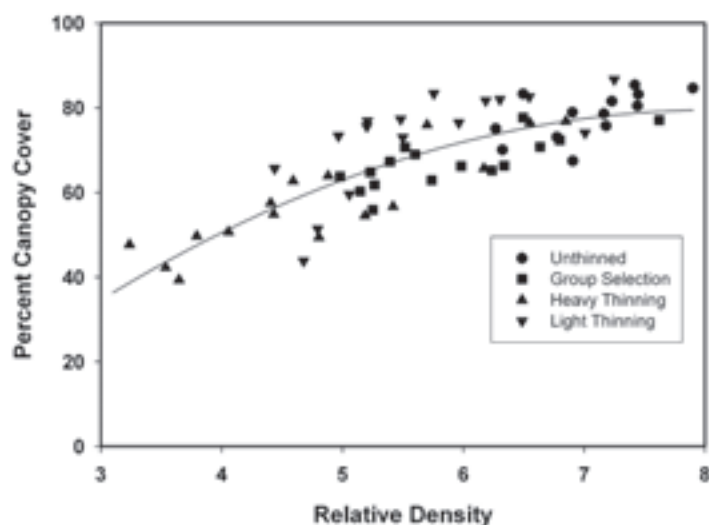


Figure 1—Percent canopy cover as a function of relative density.

relative density and stocking of the heavy thinning treatment was less than that of the light thinning and group selection treatments as intended, the residual relative densities exceeded prescribed targets for all three treatments (table 1). The negligible post-treatment difference in stocking between the light thinning and group selection treatments is not entirely surprising given that small groups were prescribed to cover ten percent of the group selection stands, with the remainder thinned similar to the light thinning treatment. The variation in pre-treatment stand conditions would make such precise treatment differences difficult to achieve in practice.

Canopy cover

Pretreatment mean percent over-story canopy cover ranged from 66 to 87 for the sixteen stands. Initially following harvest, mean percent cover for the group selection, light thinning and heavy thinning treatments was 63, 62, and 48, respectively, while percent cover of unthinned stands averaged 77. Based on stand averaged values, percent canopy cover was positively correlated with RD throughout the 1998 to 2005 monitoring period (fig. 1). Based on a second-order polynomial regression, RD accounted for 68 percent of the variation in percent canopy cover ($p < 0.0001$).

From 2001 to 2005 canopy cover increased on average 1.1, 2.9 and 2.0 percent per year for the group selection, light thinning, and heavy thinning treatments, respectively. In contrast, changes in canopy cover in unthinned stands were not detected. Although, the group selection and light thinning treatments had similar

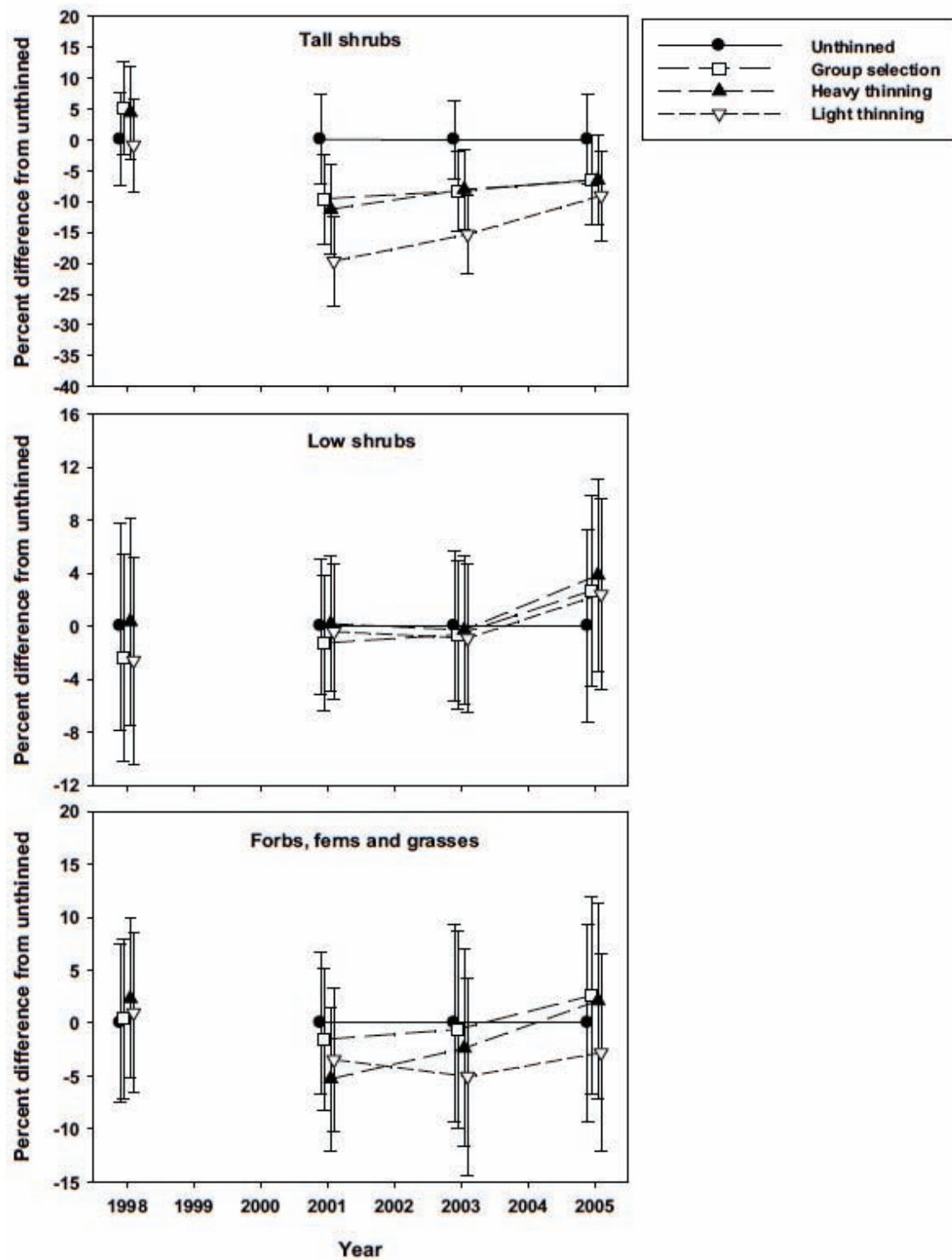


Figure 2 — Cover of tall shrubs, low shrubs, and forbs, ferns and grass vegetation strata by treatment expressed as a difference from the unthinned treatment. Error bars represent one standard error about the mean of $n=$ four replications.

average residual densities and stocking, the lesser rate of canopy expansion in the group selection treatments is likely the effect of the 0.05- to 0.1-ha group openings, which are included in the stand averaged canopy cover estimates.

Understory vegetation

Over the study, mean cover by tall shrubs in the unthinned treatment averaged 36.9 percent. Relative to the unthinned treatment, percent cover by tall shrub species was decreased approximately 11 percent in the group selection and heavy thinning treatments and by 21 percent in the light thinning treatment at the first post-treatment measurement (fig. 2). By 2005, significant growth ($p=0.058$) resulted in treated stands having tall shrub cover that was approximately four to nine percent less than the unthinned treatment suggesting a substantial recovery over the four-year period. Differences among thinning treatments in the rates of recovery or overall relative abundance were not statistically significant.

In contrast to tall shrubs, cover by low shrub species was unchanged by the harvest activity as there was no ap-

parent cover decrease relative to the unthinned stand (fig. 2). Over the study low shrub cover in unthinned stands averaged 17.9 percent. Between 2003 and 2005 there was an apparent three to five percent increase in low shrub cover by all of the thinned treatments, but the increase was not statistically significant.

Forbs, ferns, and grasses provided an average of 19.8 percent cover in the unthinned treatment. Cover by these taxa was decreased slightly during harvest as treatment means for the group selection, light, and heavy thinning treatments were approximately two, four, and seven percent less than that of the unthinned treatment in 2001 (fig. 2). Subsequently, cover in the thinning treatments increased five to ten percent ($p=0.002$) between 2001 and 2005 resulting in little difference in cover between thinned and unthinned stands.

The extent to which understory vegetation was dominated by the shrub and fern component is further evident from species-level estimates of relative cover and frequency of occurrence. Shrub species that occurred with both high

Table 2. Percent relative cover (RC) and frequency (Freq) for abundant or common tree, shrub, fern and herb species prior to treatment and five-years post-treatment. Relative cover is the percent cover of a species relative to total vegetation cover averaged over those plots on which a species occurred. Frequency is the percentage of all plots on which the species occurred.

Species	Life-form	Pre-treatment (1997-1998)								Five years post-treatment (2005)							
		Unthinned		Group		Heavy Thin		Light Thin		Unthinned		Group		Heavy Thin		Light Thin	
		RC	Freq	RC	Freq	RC	Freq	RC	Freq	RC	Freq	RC	Freq	RC	Freq	RC	Freq
Douglas-fir	Tree	5.5	9.1	6.2	12.5	6.4	19.4	7.4	9.1	3.1	13.6	2.8	7.1	16.8	2.8	1.4	4.2
Pacific yew	Tree	7.7	45.5	11.4	19.6	8.0	19.4	8.8	59.1	14.6	27.3	5.9	8.9	1.4	5.6	9.1	54.2
Western redcedar	Tree	4.4	27.3	7.1	33.9	7.1	11.1	12.3	54.5	6.1	50.0	9.5	19.6	3.9	8.3	10.9	50.0
Western hemlock	Tree	9.3	50.0	8.8	48.2	6.9	52.8	6.4	54.5	12.4	59.1	9.4	42.9	11.8	38.9	12.2	50.0
Oregon grape	Shrub	18.1	86.4	13.8	76.8	10.9	72.2	18.3	95.5	15.0	90.9	15.1	83.9	11.4	72.2	19.8	95.8
Pacific blackberry	Shrub	3.4	86.4	5.6	92.9	10.0	94.4	4.7	90.9	1.6	86.4	6.5	94.6	12.3	97.2	4.3	87.5
Red huckleberry	Shrub	3.4	36.4	4.0	28.6	3.3	38.9	6.9	72.7	3.6	31.8	2.0	21.4	2.9	30.6	6.7	54.2
Rhododendron	Shrub	17.2	36.4	14.0	44.6	15.2	30.6	9.5	50.0	17.5	36.4	13.2	48.2	14.1	25.0	9.8	45.8
Salal	Shrub	11.6	54.5	11.8	60.7	11.9	63.9	6.8	90.9	9.9	54.5	12.2	66.1	13.6	61.1	9.2	75.0
Vine Maple	Shrub	31.8	86.4	35.2	94.6	36.0	80.6	25.2	81.8	31.3	90.9	27.1	82.1	32.7	75.0	23.0	87.5
Sword fern	Fern	16.7	90.9	20.3	82.1	18.2	83.3	17.2	100.0	20.1	90.9	25.9	87.5	20.3	83.3	17.7	100
Bedstraw	Herb	1.2	45.5	0.9	58.9	0.6	61.1	0.6	45.5	0.5	40.9	1.2	67.9	0.6	61.1	1.5	70.8
Goldthread	Herb	0.7	36.4	0.7	23.2	0.7	19.4	1.4	72.7	0.4	36.4	1.0	26.8	1.0	30.6	1.1	83.3
Oregon oxalis	Herb	19.8	27.3	5.9	16.1	0.3	11.1	2.1	9.1	13.2	31.8	8.5	19.6	3.1	5.6	4.1	8.3
Redwoods violet	Herb	0.2	27.3	0.9	25.0	1.8	50.0	1.3	54.5	0.4	22.7	1.1	33.9	1.7	47.2	0.9	62.5
Wall-lettuce	Herb	0.0	0.0	0.6	7.1	0.2	2.8	0.1	4.5	0.0	0.0	0.9	30.4	0.5	22.2	0.9	20.8
Western starflower	Herb	0.9	18.2	0.5	32.1	1.8	36.1	2.4	4.5	0.4	45.5	0.7	58.9	1.3	58.3	0.3	45.8
Western twinflower	Herb	3.3	50.0	3.2	64.3	7.0	58.3	3.2	81.8	1.8	59.1	4.5	66.1	8.9	69.4	2.4	87.5

relative cover (10 to 35%) and frequency (>50%) were Oregon grape (*Mahonia nervosa* Pursh), salal (*Gaultheria shallon* Pursh), and vine maple (*Acer circinatum* Pursh) (table 2). Sword fern (*Polystichum munitum* (Kauf.) Presl.) occurred at relative densities in the range of 15 to 25% at frequencies in excess of 80%. Pacific blackberry (*Rubus ursinus* Cham. & Schlecht.) also occurred at high frequency but provided low relative cover. In contrast, on an individual species basis few herb species occurred at relative covers greater than 5%, although there were species such as goldthread (*Coptis lanciniata* Gray), western twinflower (*Linnea borealis* Torr.), redwoods violet (*Viola sempervirens* Greene), and bedstraw (*Gallium* spp.) that occurred at frequencies in excess of 50% for some treatments (table 2). Although an increase in herbaceous cover might have been expected as a result of harvest-related disturbance, only the early-

seral associate wall-lettuce (*Mycelis muralis* (L.) Dumort), demonstrated an apparent increase in frequency (table 2).

Species diversity

Prior to thinning, species richness (N_0) averaged 11.7 species per 0.1-ha plot when averaged across all treatments. In 2001, following treatment, the mean number of species declined somewhat, particularly for the light thinning and heavy thinning treatments (fig. 3). However, species richness increased significantly ($p=0.0014$) between 2001 and 2005 to return to pretreatment levels (fig. 3). In contrast to richness, the evenness component of diversity did not differ among treatments or vary over time with mean evenness index values being approximately 0.72 to 0.78 (fig. 3). These values indicate a relatively large contribution to community composition by a few species present in high abundance.

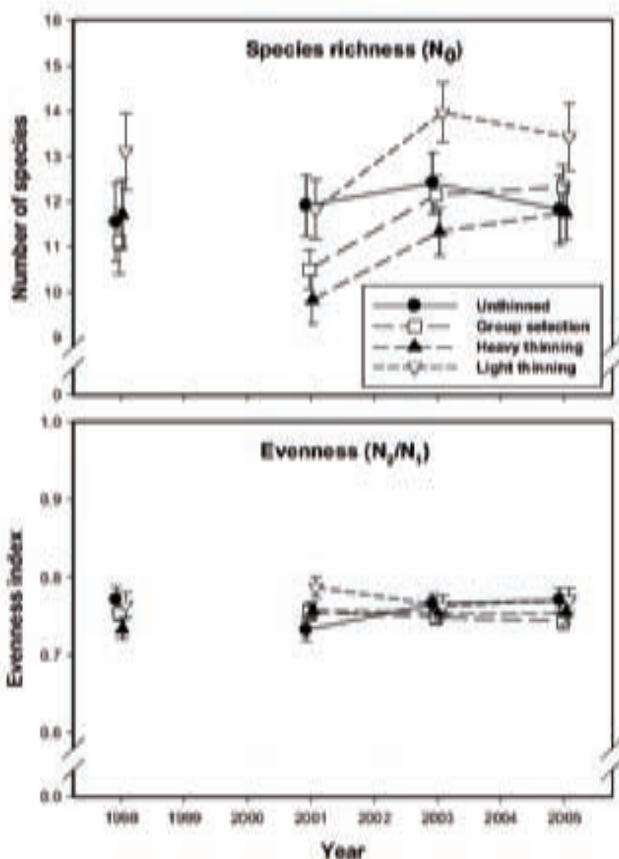


Figure 3—Species richness and evenness by treatment and year. Richness is the mean number of species observed per 0.1-ha plot. Evenness is a measure of the extent to which all species on a plot contribute to total cover. Evenness index values near 0 indicate equal contributions by all species; values near 1 indicate dominance by a few species. Error bars represent one standard error about the mean of $n=$ four replications.

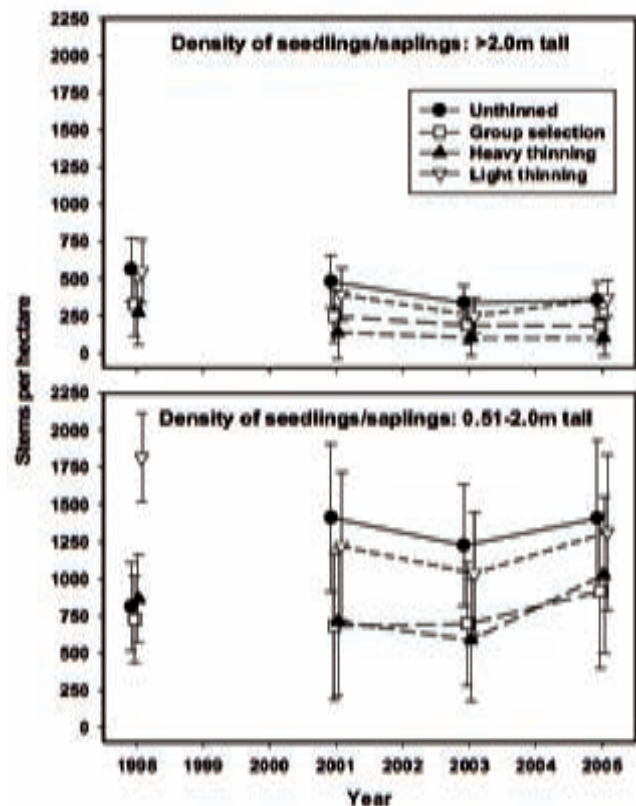


Figure 4—Density of seedlings and saplings greater than 0.5 m tall. Error bars represent one standard error about the mean of $n=$ four replications.

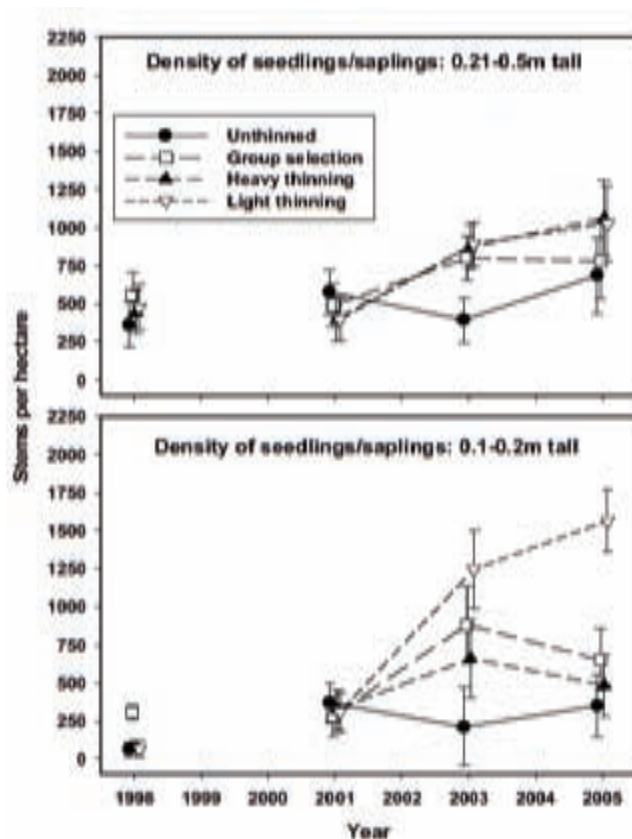


Figure 5— Density of seedlings and saplings from 0.1 to 0.5 m tall. Error bars represent one standard error about the mean of n=four replications.

Tree regeneration

The density of large saplings (trees >2 m tall and <5 cm d.b.h.), averaged over all treatments tended to decline ($p=0.060$) from about 315 tph to 250 tph over the study. Of the three thinning treatments, only the light thinning demonstrated a tendency for increased large sapling density between 2001 and 2005, as this was the only thinning treatment that demonstrated an obvious decrease in sapling density initially following harvest (fig. 4). This response was consistent with tall shrub cover which also was decreased by harvest activities to the greatest extent in the light thinning regime.

Prior to treatment, densities of saplings 0.51 to 2.0 m tall were more than twice as abundant in the light thinning treatment than in the group selection, heavy thinning and unthinned treatments ($p=0.015$). However, following harvest the abundance of saplings was diminished to amounts similar to the other treatments (fig. 4). Furthermore, there

was a large variation about the treatment mean values and differences among treatments or over time were not detected.

Densities of seedlings in the 0.1 to 0.2 m and the 0.21 to 0.5 m height classes tended to increase between 2001 and 2005, with the extent of increase differing among treatments (time by treatment interactions, $p=0.0004$ and $p=0.0006$ for the two respective classes). A minimum increase in seedling density of approximately 500 tph is expected for the three thinning treatments given that the stands were underplanted in 2002 and seedling stock would be nearly 0.2 m tall when planted. Although there was no effort to explicitly monitor planted seedlings or to distinguish planted seedlings from natural regeneration, the expected contribution of underplanted conifers to seedling density is consistent with the data in figure 5.

In addition to underplanted seedlings, there was also evident a large increase in the density of 0.1 to 0.2 m seedlings in the lightly thinned treatment (fig. 5). Although the data are not shown here, the increase in natural seedling density for that treatment was due predominantly to the occurrence of bigleaf maple (*Acer macrophyllum* Pursh).

In general the range of average abundances by year and treatment of western hemlock (650 to 1200 tph) and western red cedar (115 to 800 tph) also indicated recruitment of natural regeneration, however, the occurrence was variable and treatment effects were not statistically evident. While Douglas-fir was represented in all of the seedling and sapling height classes, its abundance (90 to 320 tph) was low relative to the more shade tolerant western redcedar and western hemlock. Among tree species with densities significantly correlated to canopy cover or relative density, western hemlock, western redcedar, and Pacific yew were positively related and Douglas-fir was negatively related, as expected given existing knowledge of species shade tolerance.

DISCUSSION

Of the three questions addressed, the most certain result was that the thinning prescriptions resulted in a variety stand conditions with a distinct separation in terms of density and canopy cover occurring between the unthinned stands and the heavily thinned stands. However, it is im-

portant to note that the thinning treatments imposed in this study are of relatively low intensity in terms of both the proportion of basal area removed and the size of gaps created in the group selection treatment (see Figure 3, Poage and Anderson 2007). For example, a study conducted in western Oregon on the Siuslaw National Forest evaluated thinning treatments having basal area removals of 51 to 86% and residual Curtis' relative densities ranging from 1.2 to 4.0 (Chan et al. 2006). The BLM Density Management Studies, also being conducted in western Oregon examined thinning treatments that initially removed 35 to 65% of the basal area in a thinned matrix and incorporated 0.1 to 0.25 ha leave islands and gaps over as much as 40% of the stand area (Cissell et al. 2006).

Although, the treatments resulted in discernable differences in density and cover, particularly between the unthinned and heavy thinning treatments, there was little evidence of substantial alterations of understory shrub and herbaceous vegetation. This lack of strong understory vegetation response in terms of composition, abundance, or size is consistent with several studies of thinning in Douglas-fir. In a recent review of seven operational-scale silviculture experiments, Wilson and Puettmann (2007) report that percent cover by shrubs and percent cover by herbs, one to seven years following thinning showed little difference across a wide range of residual basal area. In contrast, the retrospective studies of Baily et al. (1998) and Suzuki and Hayes (2003) showed in some cases large increases in shrub cover ten to 30 years after thinning. Short-term vegetation responses likely reflect the direct influences of disturbance, such as harvest damage, and transitory shifts in resource availability, whereas a longer period is required for altered canopy structure to have an impact on understory vegetation (Lindh and Muir 2004; Thomas et al. 1999; Wilson and Puettmann 2007).

Halpern et al. (2005), studying young Douglas-fir forests in the western Cascades of Washington, observed that the initial understory vegetation response to overstory removal with green tree retention was an increase in within stand variation in abundance. They hypothesized that this resulted from disturbance associated with harvest and that the influence of canopy structure on understory vegetation composition and structure would be manifest over a longer period of time. However, they also observed that the amount of disturbance was greater with lower levels of green-tree

retention (15 percent versus 40 percent retention) and with aggregated retention than with dispersed retention, which contrasts with our observations for greater reductions in cover, particularly for tall shrubs, occurring in the light thinning treatment. These differences were likely due to the overall greater intensity of harvest treatments considered by Halpern et al. However, it may be speculated that logging systems also had a role; light, evenly dispersed thinning may have had less concentration of cable yarding activity. Although fewer trees were removed relative to the heavy thin and group selection treatments, lateral skidding in the light thin treatment may have been more dispersed. As a result the light thin treatment may have impacted a greater proportion of the area.

The question of whether these treatments enhance recruitment of a new cohort of conifer trees is still unclear. Where understory conifers were present in abundance prior to treatment, the density was decreased in harvest. The small post-treatment increase in Douglas-fir seedlings in treated stands can not be attributed to either natural or planted origins because the sampling protocol did not distinguish origin of seedlings. Recent studies by Maas-Hebner et al. (2005) and Chan et al. (2006) have demonstrated in thinned stands of the highly productive Coast Range forests of Oregon that underplanted Douglas-fir and western hemlock will have high rates of survival through eight years following thinning and planting across a broad range of residual stand densities. However, they observed that underplanted western hemlock seedlings generally grew faster and developed better height:diameter ratios than did underplanted Douglas-fir seedlings at all residual densities. Given the relatively high residual overstory densities and the relative abundance of regenerating shade-tolerant conifers and hardwoods in this study, the initial results suggest that recruitment of Douglas-fir regeneration will occur in association with other tree species, rather than as a species-dominant cohort.

The rapid recovery of diversity following disturbance, as indicated in this study by the diversity metrics of Hill (1975), is a response consistent with other studies in managed forests in the western hemlock zone of western Washington and Oregon (Halpern 1989, Halpern and Spies 2005; Thomas et al. 1999; Thysell and Carey 2001). The numbers of species, particularly the total (N_0) are somewhat lower than that reported for stands of similar age in the

western Cascades of Washington, but are within the range observed (Halpern and Spies 1995). The rhododendron-salal and rhododendron-Oregon grape plant associations comprise the potential natural vegetation community in 14 of the 16 stands (Hawk 2005). Of six plant associations assessed within the H.J. Andrews, these two plant associations ranked first and second both in resistance to disturbance and in resilience to disturbance (Halpern 1988). The predominance of a shrub component that may confer the attributes of resilience and resistance, may also limit the development of an abundant and diverse herbaceous stratum (Halpern and Spies, 1995; Huffman et al. 1994, Deal 2001, Deal and Tappeiner 2002).

CONCLUSIONS

The treatments imposed in this study produced a range of stand structure conditions. Although stand conditions differed among treatments, the initial understory vegetation responses were limited to a short term decrease in the abundance of tall shrub and saplings greater than one meter height. This decrease was greatest in the lightly thinned unit, possibly because of a greater dispersion of yarding disturbance in the predominantly cable-logged harvest. The shrub-dominated plant communities observed in this study are characteristically resistant and resilient to disturbance, and are not likely to demonstrate significant, long term shifts in abundance or composition with a single harvest entry within the range of treatment intensity evaluated. The long term implications of repeated entry at relatively frequent ten to 20 year intervals, as prescribed, may challenge the resistance and resilience of these understory communities. Although planted conifers have survived through five-years following thinning treatment, the persistence and growth of planted seedlings, particularly the intermediate shade-intolerant Douglas-fir is yet to be determined. To address these long-term questions, the experimental prescriptions should be followed and the stands monitored for several decades into the future

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Proceedings of the 2007 National Silviculture Workshop

**Sustaining Northern Red Oak Forests:
Managing Oak From Regeneration to Canopy Dominance
in Mature Stands**

Daniel C. Dey, Gary W. Miller, and John M. Kabrick

ABSTRACT

Across the range of northern red oak, managers have problems sustaining current stocking of northern red oak in forests. Oak species are adapted to frequent stand disturbances that reduce the abundance of shade tolerant competitors and control fast-growing pioneer species. A widely recommended approach to regenerating northern red oak is to develop relatively large advance reproduction by manipulating stand structure to provide adequate light to oak seedlings in the forest understory. One approach to develop large oak advance reproduction is to use the shelterwood method with woody competition control. Herbicides, mechanical cutting and fire can be used to manage competing woody vegetation. Underplanting shel-

terwoods with oak seedlings can be used to supplement natural reproduction. Where landowners desire to maintain some presence of mature forests at all times, alternative approaches such as the group selection method or shelterwood with long-term retention of a low density overstory may be used to promote oak regeneration and recruitment into the overstory. Once stand regeneration has been initiated, maintaining the desired level of oak stocking in dominant and codominant crown positions can be accomplished through crop tree thinning in young stands. Sustaining oak stocking requires a regime of managed disturbances that begin before stand initiation and extend through the first 20 to 30 years of stand development.

Keywords: northern red oak, silviculture, ecology, regeneration, thinning, stand development.

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INTRODUCTION

Northern red oak (*Quercus rubra* L.) is highly valued for its timber and acorn production. It is widely distributed throughout eastern North America occurring in a variety of ecoregions (Sander 1990). Northern red oak occurs on over 16.9 million ha in the eastern United States (Miles 2007), and successional replacement of this species across the landscape would be a major ecological and economic loss. Its current abundance and distribution is linked with the history of fire and land use during the past 200± years (Abrams 1992, Dey 2002, Dey and Guyette 2000, Pyne 1982, Whitney 1994, Williams 1989). Historic disturbances were important in the expansion and dominance of oaks across a diversity of ecological land types throughout eastern North America (Nowacki and Abrams 1992, Hicks 1998). Modern oak forests originated when logging, fuelwood cutting, charcoal production, woods burning, grazing, and agriculture occurred extensively and frequently in the eastern region, completing a progression toward increasingly greater oak dominance that began when Native Americans brought fire to the region (Pyne 1982, Cronon 1983, Williams 1989, Nowacki and others 1990, Abrams 1992, Whitney 1994, Abrams and McCay 1996).

However, disturbance regimes that once favored oak species have changed. Now, oaks are being displaced by shade tolerant tree species as forest succession progresses in response to small-scale, gap-type disturbances, or by shade tolerant or intolerant tree species following catastrophic natural disturbances or timber harvesting in the absence of frequent fire (Abrams and Nowacki 1992, Hix and Lorimer 1991, Johnson and others 2002, Lorimer 1993, Pallardy and others 1988). Loss of oak species is greatest on high quality sites where oak is competing with high densities of shade tolerant species and, in particular, fast growing species such as yellow-poplar (*Liriodendron tulipifera* L.) (Loftis 1983, Beck and Hooper 1986, Schuler and Miller 1995, Weigel and Johnson 2000, Weigel and Parker 1997). Sustaining current oak stocking levels in mature forests is a common management objective and challenge to forest managers. Loftis (2004) described the two “laws” of oak silviculture as being: (1) the presence of competitive sources of oak reproduction and; (2) disturbances that provide timely and adequate release for those sources. The purpose of this paper is to review the literature on northern red oak physiology, ecology and silviculture and synthesize it to give insight

into how to regenerate mature forests and manage forest development to sustain northern red oak stocking.

OAK REGENERATION ECOLOGY AND MANAGEMENT

Sources of Oak Regeneration

It is important to consider the sources of oak reproduction when developing a prescription to regenerate northern red oak. Most oak species regenerate from new seedlings (stem and root system same age), seedling sprouts (stem age younger than root age), and sprouts from the stumps of overstory trees cut during harvesting (Johnson 1993). Oak reproduction present before the regeneration harvest is collectively referred to as advance reproduction, and the ability of trees, from seedlings to mature individuals, to contribute to stand regeneration through sexual or vegetative reproduction is known as regeneration potential (Johnson and others 2002). The goal then is to increase the regeneration potential of northern red oak sources of reproduction to maximize success in oak regeneration after harvesting.

Oak seedling and advance reproduction

Northern red oak produces good to excellent seed crops every two to five years (Sander 1990), and oak regeneration failures are not normally related to matters of seed production or germination (Lorimer 1993). During years of bumper seed crops, northern red oak-dominated forests can produce more than 600,000 acorns per hectare (Auchmoody and others 1993). Even in the low light of mature eastern hardwood forests, which often is below five percent of full sunlight (Lockhart and others 2000, Lorimer and others 1994, Miller and others 2004, Motsinger 2006, Parker and Dey in press), northern red oak acorns provide the energy necessary for first year seedling growth. Subsequent growth is usually hampered by low understory light, and within 10 years of germination most oak seedlings are dead (i.e., < 10 percent survival) (fig. 1) (Beck 1970, Crow 1992, Loftis 1988, 1990a). Any survivors are small in size.

Small (e.g., less than 1.0 cm basal diameter) northern red oak seedlings and seedling sprouts have low regeneration potential, (i.e., low probabilities of assuming dominant or codominant crown positions in the regenerating stand (Johnson and others 2002, Loftis 1990b)). The long-term dominance probability for red oak reproduction decreases

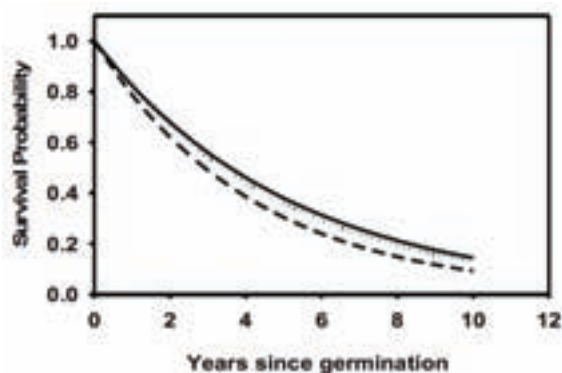


Figure 1— Survival of northern red oak seedlings in the understory of mature mesic forests in the southern Appalachians and in northeastern Wisconsin (adapted from Beck 1970 (dashed line), Crow 1992 (dotted line), Loftis 1988 (solid line)).

with increasing site quality because they are out-competed by more rapidly growing species such as yellow-poplar and black cherry (*Prunus serotina* Ehrh.) (Loftis 1990b, Tift and Fajvan 1999, Weigel 1999, Weigel and Johnson 2000). Whether northern red oak reproduction is established naturally or artificially, its regeneration potential increases with increasing initial seedling size (fig. 2) (Johnson 1984, Loftis 1990b, Spetich and others 2002). The key to successful regeneration of northern red oak is to have sufficient numbers of large advance reproduction and potential oak stump sprouts to provide adequate oak stocking at the end of the regeneration period.

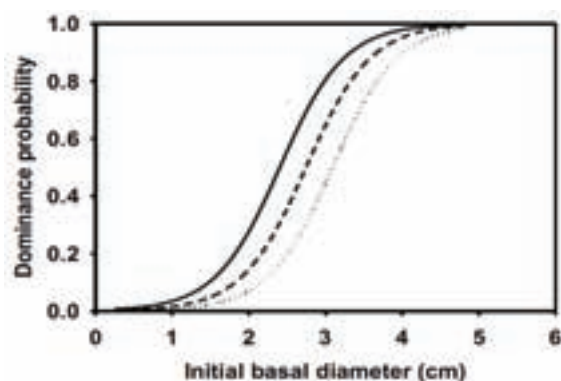


Figure 2— Dominance probability for free-to-grow northern red oak advance reproduction in relation to site quality eight years after clearcutting in the southern Appalachians (adapted from Loftis 1990b). Northern red oak site index 21 m (solid line), 24 m (dashed line), and 27 m (dotted line).

Oak stump sprouts

Another important source of oak reproduction is oak stump sprouts. They are the fastest-growing and most competitive source of oak reproduction (Johnson and others 2002). However, not all oak stems cut in timber harvesting produce sprouts. Thus, current levels of oak stocking in mature forests cannot be sustained by relying on stump sprout regeneration alone, hence the need for and importance of oak advance reproduction. The probability that a northern red oak stump produces a live sprout at the end of the first growing season is highest for smaller diameter trees (e.g., parent tree d.b.h. 10 to 30 cm), decreases with tree age, and increases with increasing site quality (fig. 3)

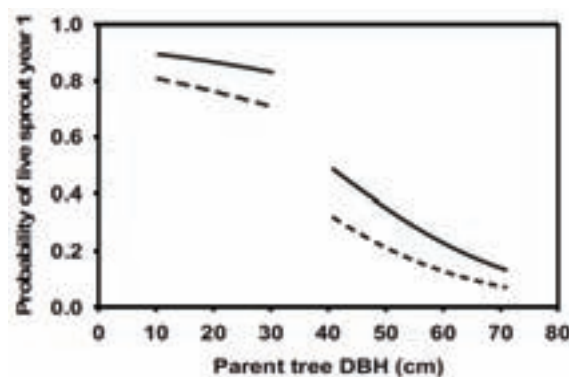


Figure 3— Probability that northern red oak stumps produce a live sprout in relation to initial d.b.h. of the parent tree, tree age and site quality one year after clearcutting in southern Indiana (Weigel and Johnson 1998b). Site index 18 m (dashed line) and 23 m (solid line).

(Weigel and Johnson 1998b). Larger trees have thicker bark that probably reduces sprouting by physically inhibiting development of vegetative buds. Also, older trees have reduced ability to produce sprouts due to physiological senescence. Trees on high quality sites may have improved vigor that initially enhances sprouting capacity.

Longer-term dominance probabilities were reported for northern red oak stump sprouts growing for 15 years in southern Indiana clearcuts by Weigel and others (2006). They defined dominance probability based on the crown class of the most dominant stem in the oak sprout clump. Weigel considered northern red oak sprouts successful, or “dominant” if they were in the dominant or codominant crown class. Northern red oak stump sprout dominance probabilities at stand age 15 decreased with increasing parent tree d.b.h. and age (fig. 4). Further, they estimated

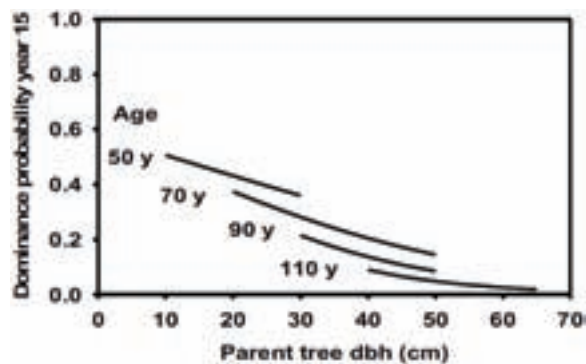


Figure 4— Northern red oak stump sprout dominance probabilities 15 years after clearcutting in southern Indiana (Weigel and others 2006).

conditional dominance probabilities of northern red oak stump sprouts and found that the longer oak stump sprouts persisted in regenerating stands the greater was the likelihood that they would remain dominant at stand age 15 (fig. 5). For northern red oak sprouts that were alive one year after harvesting, the probability they were dominant at age

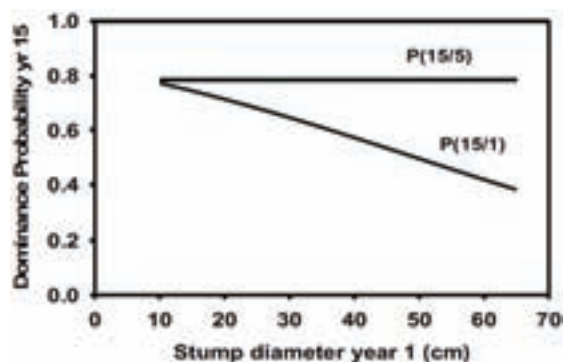


Figure 5— Northern red oak stump sprout conditional probabilities of dominance at stand age 15 in southern Indiana clearcuts – site index 18 m (Weigel and others 2006). Conditional probabilities at stand age 15 are estimated for northern red oak stump sprouts given: (a) they were alive in year 1 ($P(15/1)$), and (b) given they were alive in year 5 ($P(15/5)$).

15 decreased with increasing diameter of the parent stump. A sprout on a 10 cm diameter stump had an 80 percent likelihood of persisting as a dominant at age 15, whereas a sprout on a 60 cm diameter stump had only a 40 percent chance of remaining dominant. By stand age 5, if a northern red oak stump sprout was alive, it had a high probability of being dominant at age 15 regardless of the initial size of the parent tree. The less competitive northern red oak stump sprouts are eliminated by stand age 5.

Importance of Light

A primary cause of oak regeneration failure is low light levels (i.e., < five percent of full sunlight) in the understory of fully-stocked, mature forests; light so low that it is not sufficient to meet the respiratory demands and is prohibitive to the development of large oak advance reproduction (Abrams 1992, Gardiner and Yeiser 2006, Hodges and Gardiner 1993, Johnson and others 2002, Lorimer 1993, Lorimer and others 1994, Miller and others 2004). Heavy-shaded forest understories resulted from increasing forest structure due to changes in historical disturbance regimes. Over the past 50 years, the development of shade tolerant understory canopies has followed fire suppression, elimination of woods grazing by livestock, and changes in timber harvest practices (Lorimer 1993).

Northern red oak is intermediate in shade tolerance (Sander 1990). In many forest situations there is insufficient light to support increases in northern red oak seedling root and shoot biomass. Often, understory light is below the light compensation point for northern red oak, which is two to five percent of full sunlight (Gottschalk 1987, Hanson and others 1987). Light saturation of photosynthesis in northern red oak occurs between 20 to 40 percent of full sunlight (Ashton and Berlyn 1994, McGraw and others 1990, Teskey and Shresta 1985). Positive shoot growth in northern red oak seedlings requires more than 20 percent of full sunlight, and seedling diameter and height growth increases with increasing light levels up to 50 to 70 percent of full sunlight (Gottschalk 1994, Musselman and Gatherum 1969, Phares 1971). The physiology of northern red oak natural advance reproduction and planted bareroot seedlings is similar in forest understories. Both types of reproduction respond equally well to increases in light intensity and resource availability following shelterwood harvesting, midstory canopy removal, and other practices that reduce forest stocking and density (Parker and Dey in press).

MANAGING LIGHT IN FORESTS

Midstory Removal

Increasing light for oak advance reproduction in forest understories is accomplished by timber harvesting (e.g., clearcut, shelterwood, group selection), removing midstory or ground-level woody vegetation, or combinations of these

two approaches. The density of midstory and understory woody vegetation can be reduced by mechanical cutting, herbicide application or prescribed burning. Stems of any size can be cut with saw or ax. Herbicides can be applied by spraying foliage, basal stem, or cut stump, or by stem injection depending on tree size and chemical used. When properly done, herbicides have the greatest potential to permanently eliminate most woody competitors with one treatment (Miller 1993, Quicke and others 1996, Groninger and others 1998, Kochenderfer and others 2001). Care must be taken in applying herbicides because oaks are susceptible to any chemicals used to control their hardwood competitors. Prescribed burning, which is commonly done in the spring as relatively low intensity surface fires, can reliably girdle or kill trees less than 12 cm d.b.h.; larger trees are able to survive these fires intact, with only minor stem wounding (Barnes and Van Lear 1998, Dey and Hartman 2005, Reich and others 1990, Waldrop and Lloyd 1991). Cutting larger trees followed by burning in subsequent years is one way to reduce stand density and increase light because the shoots of small diameter stump sprouts are very susceptible to low intensity fires. Spring surface fires can kill most acorns that are not buried under the wet duff or in mineral soil, and cause high (e.g., 70 percent) mortality of oak seedlings less than three years old (Auchmoody and Smith 1993, Brose and Van Lear 2004, Johnson 1974). Therefore, burning should be delayed for several years following a good acorn crop.

Whether mechanically cut or killed by fire, stems of smaller-sized (10 to 20 cm d.b.h.) hardwoods often resprout and may need to be controlled again in the future. After harvesting, sprout growth can be great in the high light environment of clearcuts, but increasing amounts of residual overstory trees suppress growth. Hence, there is a negative relationship between overstory shade, hardwood sprout growth, and growth of other competing woody vegetation. The extent of this influence depends on the specie's shade tolerance. On good- to high-quality sites in the southern Appalachians, Loftis (1990b) recommended maintaining an intact main forest canopy while reducing stand basal area by thinning from below to control fast-growing, shade intolerant yellow-poplar while favoring northern red oak advance reproduction. He cautioned that reductions in stand basal area should be decreased (e.g., from 40 percent to 30 percent) as site quality increases (from 21 m to 27 m site index for northern red oak). The challenge is to reduce overstory stocking to increase light to oak while

not stimulating the growth of competitors.

Increasing understory light levels by midstory removal has been shown to significantly improve the survival and growth of northern red oak advance reproduction. Lorimer and others (1994) were able to increase understory light from one percent of full sunlight in undisturbed stands to seven to nine percent by reducing the density of the understory woody canopy in Wisconsin mesic oak-northern hardwood forests. They cut the stems of woody species that were taller than 1.5 m and sprayed the stumps with Tordon 101R. This level of light increased the survival and growth of planted northern red oak and white oak seedlings. In the central Appalachians of West Virginia, Miller and others (2004) reported that survival and growth of natural northern red oak advance reproduction was significantly increased after a 22 percent reduction in stand basal area, which was done by thinning from below. They found that understory light was increased from 2 to 12 percent of full sunlight when they applied herbicide by cut stump or stem injection to all tree stems greater than 0.6 m tall in the understory and all trees in the intermediate and suppressed crown classes. Similarly, Motsinger (2006), working in Mississippi River bottomland forests in Missouri, increased understory light levels to 16 percent of full sunlight and improved the survival and growth of pin oak (*Q. palustris* Muenchh.) reproduction. In Mississippi, Lockhart and others (2000) increased understory light to 30 percent of full sunlight in bottomland pine-hardwood forests by removing trees in the intermediate and suppressed crown classes, thus improving the growth of cherrybark oak (*Q. pagoda* Raf.) advance reproduction. The higher light levels achieved in this study may be largely a result of the abundance of pine in the overstory, and light's increased ability to penetrate pine foliage and crowns compared to a pure hardwood canopy of similar stand stocking. In a diversity of forest types across eastern North America, it has been demonstrated that removal of midstory woody stems benefits oak reproduction by increasing light, but light intensity after these treatments, in general, are still significantly below the light levels associated with maximum net photosynthesis and growth.

Overstory Harvesting

In general, more significant reductions in overstory canopy density are necessary to increase understory light to 30 to 70 percent of full sunlight, which is required for maximum net photosynthesis and growth in northern red oak advance reproduction. Regeneration methods that

substantially reduce overstory stocking are recommended for oak regeneration (Hannah 1988, Gardiner and Yeiser 2006, Johnson and others 1986, Loftis 1990a, 1993, Sander and Graney 1993, Smith 1993, Spetich and others 2002, Weigel and Johnson 1998a). Oak reproduction grows rapidly in group selection openings that are at least as wide as the height of the adjacent forest (Marquis 1965, Minckler 1961). The clearcutting method has been most successful in regenerating oaks in xeric ecosystems where large advance reproduction has been able to accumulate before harvesting (Johnson and others 2002). However, releasing small oak reproduction to full sunlight by clearcut or large group selection harvests on high quality sites without concurrent competition control has not been successful because oak seedlings are quickly suppressed by either well-established shade tolerant advance reproduction or fast-growing, shade intolerant species (Beck and Hooper 1986, Brose and Van Lear 1998, Canham 1988, 1989, Jensen and Parker 1998, Loftis 1983, 1990b, McQuilkin 1975, Orwig and Abrams 1994, Ross and others 1986, Smith 1981, Weigel and Parker 1997).

In a diversity of forest types, shelterwood harvesting that reduces stand stocking by more than 40 percent, or basal area by more than 50 percent, or crown cover by more than 30 percent can provide 35 to 50 percent of full sunlight to oak reproduction provided the midstory canopy is also removed (Gardiner and Yeiser 2006, Godman and Tubbs 1973, Leak and Tubbs 1983, Lorimer and others 1994, Parker and Dey in press, Sander 1979, Schlesinger and others 1993). As mentioned earlier, maintenance of higher residual shelterwood density and continuous canopy cover in the main overstory helps to control shade intolerant competitors (Loftis 1990b, Schlesinger and others 1993). Schlesinger and others (1993) found that control of taller understory woody competitors in the Missouri Ozark Highlands is not necessary to promote oak advance reproduction growth on low quality sites (i.e., site index 18 m black oak) where oak reproduction did best growing under a 40 percent stocked shelterwood. In contrast, oak reproduction on more productive sites (i.e., site index 21 m black oak) did best when stocking was left at 60 percent and woody understory competition was controlled. These Missouri Ozark ecosystems are outside of the range of yellow-poplar and really do not have a comparable aggressive shade intolerant competitor. Final removal of the shelterwood is triggered when the needed stocking of

large oak advance reproduction is present, and depends on management goals for oak stocking at stand maturity. Dominance probability models such as those presented by Loftis (1990b) or Spetich and others (2002) are helpful in assessing the regeneration potential of oak advance reproduction, and timing the shelterwood removal.

Oak Recruitment into the Overstory

For the first few years after a disturbance, tens of thousands of small seedlings and sprouts per hectare compete for the free growing space, but the winners of this competition become apparent in just a few years. As stand age increases after clearcutting, density of all trees, as well as number of dominant and codominant trees, decreases (fig. 6). In central Appalachia, there may be as many as 7,400 trees

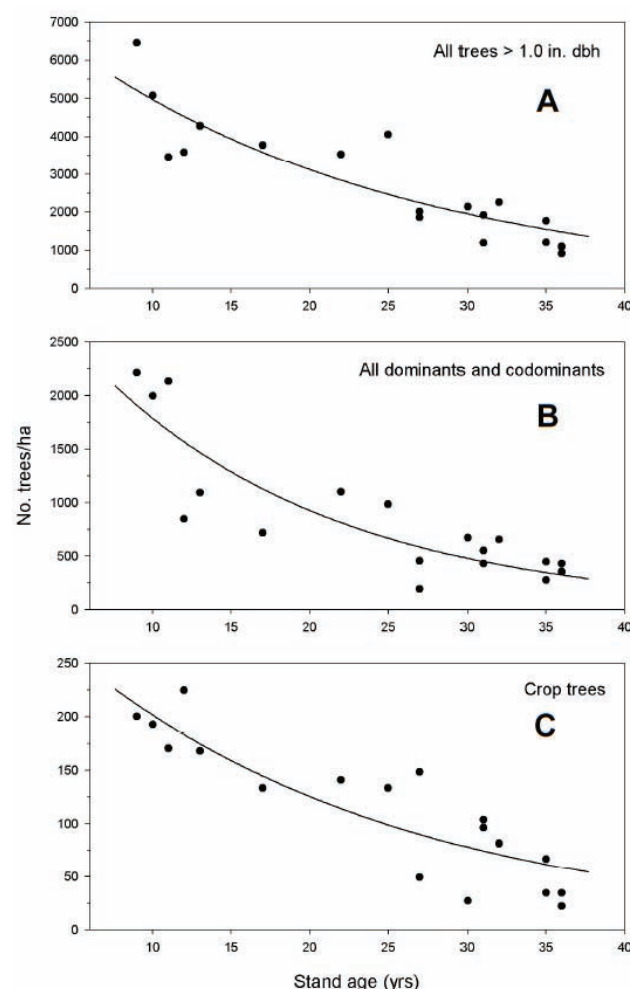


Figure 6— Number of trees per hectare of (a) all trees >2.5 cm d.b.h., (b) all dominant and codominant trees, and (c) only crop trees in 18, even-aged, mixed hardwood stands on northern red oak site index 20 to 21 m in the central Appalachians. Circles are observed densities and lines are regression estimates of average density by stand age (from Miller et al. 2007).

per hectare in a 10- to 15-year-old mixed hardwood forest clearcut (site index 20 to 21 m northern red oak), where the overstory trees are 9 to 12 m tall at canopy closure (fig. 6a) (Miller and others 2007). Although a young stand contains thousands of trees per hectare, only the canopy dominant and codominant stems can persist for many years and provide long term benefits. For example, a typical 15-year-old stand averages 3,700 trees per hectare (≥ 2.5 cm d.b.h.) (fig. 6a), of which only one-third of the trees are in competitive positions in the canopy (fig. 6b).

Without management intervention, suppressed and intermediate northern red oaks have low probability of becoming more dominant as stands progress toward maturity. Ward and Stephens (1994) studied stand dynamics in second growth northern red oak-mixed hardwood forests in Connecticut as the stands developed from age 25 to 85. Northern red oak trees were distributed among all crown classes at stand age 25 with the majority of the oaks in the intermediate and larger classes. However, the only northern red oaks to persist to stand age 85 were those that were dominant or codominant. They reported that about 75 percent of the dominant northern red oak at age 25 remained dominant after 30 years, but only about 25 percent of the codominant oak remained in the codominant class by age 55. Most of the codominant trees dropped into the lower crown classes or died. Between stand age 25 and 55 most of the intermediate and suppressed oaks died. Northern red oak trees that express dominance early are the only trees that are able to persist as dominants into maturity. Thus, stands must be thinned at an early age to favor oak so that managers have flexibility in meeting their stand stocking objectives. This is more imperative on high quality sites, or where oak is competing with fast-growing species such as yellow-poplar.

The probability of oaks recruiting into the overstory as canopy codominants and dominants is modified by site quality. As forests progress through the stand initiation stage to the stem exclusion stage (Oliver and Larson 1996), oaks have a higher chance of becoming dominant on low to average quality sites than on higher quality sites (Johnson and others 2002). Hilt (1985) observed in 29 Ohio Valley clearcuts of oak-dominated forests that the proportion of oak in the reproduction was similar for the first 14 years regardless of site quality, and that 20 to 30 percent of all trees were oaks. After stand age 15, the proportion of oak

increased dramatically to 60 percent of all trees on sites of relatively low quality (15 to 18 m site index). They were one-third of stand density on medium quality sites (18 to 21 m site index), but dropped to about 10 percent of stand density on high quality sites (21 to 24 m site index). Stocking of dominant, competitive oak can be increased even on high quality sites by thinning (cleaning) to release potential oak crop trees.

Crop Tree Thinning

Crop trees are defined by management objectives. The tree's market value, wildlife value, esthetic value, or "diversity" value determines whether or not it meets management objectives. Species and quality are key characteristics in identifying potential crop trees. Northern red oak that exhibit good bole quality (i.e., straight, branch-free boles, no cankers, no low forks, good attachment to stumps) and good vigor (i.e., healthy crown and bark development, etc.) are desirable crop trees. Only a small percentage of dominant and codominant trees qualify as crop trees (fig. 6c). A 15-year-old stand that contains about 1,200 dominant or codominant trees per hectare may have only 160 potential crop trees (Miller and others 2007).

It is also important to recognize that the number of potential crop trees declines as the stand ages (fig. 6c). Trees continue to compete for growing space and some overstory trees lose their canopy status and even die as the stand matures. Each year some potential crop trees succumb during the natural thinning process. In the absence of crop tree release treatments, stands older than 25 years often contain less than 100 crop trees per hectare (fig. 6c). In any given stand, the management objectives can be even more restrictive than the general characteristics described here. As a result, the crop tree densities in Figure 6c probably overestimate the number of potential crop trees in most stands.

The number of crop trees released within a typical hardwood stand varies depending on site quality, species composition, stand age, management objectives, and cost of pre-commercial operations. As a result, it is important to understand how hardwood stands develop and change over time so that release treatments can be applied effectively to achieve management objectives. When the stand is mature, the main canopy will contain 148 to 173 dominant and codominant trees per hectare, so there is no need to release more than this density when the stand is young. In most cases, however, the release treatment will involve far fewer

trees (fig. 6c) due to their actual availability.

Guidelines for spacing of crop trees are less precise. Hardwood stands that form from natural regeneration exhibit random patterns in their distribution of potential crop trees. It is not unusual to find a clump of three or four potential crop trees growing in close proximity or scattered 15 to 30 m apart. A general guideline is to focus on finding the best available crop trees, regardless of spacing, and provide them with an adequate release. Avoid releasing trees that do not qualify as crop trees just for the sake of achieving an even distribution, as this approach may not be an efficient use of resources. In rare cases where crop trees are abundant and dispersed throughout the stand, seeking an even distribution of crop trees is acceptable, so long as each crop tree receives an adequate release.

In young hardwood stands, the best time to apply release treatments is when the canopy begins to close and continues for about 10 to 15 years after canopy closure. The stand age at canopy closure varies with site quality. On high-quality sites, where abundant resources accelerate stand development, canopy closure can occur about age 8 to 10 years. On poorer sites, where fewer species are competitive and stand development is somewhat slower, canopy closure can occur about age 13 to 15 years. Keep in

mind that young stands contain more potential crop trees compared to older stands (fig. 6c) and thus opportunities to improve long-term species composition of the overstory are greatest in young stands that are less than 25 to 30 years old because the number of crop trees per hectare tends to diminish with stand age.

In older hardwood stands approaching large pole or small sawtimber size, there are still opportunities to culture dozens of crop trees per hectare to improve vigor, growth, and spacing of desirable trees as the stand matures. Beyond age 25 or 30 years, the number of potential crop trees can range from 62 to 99 trees per hectare, but this number will decline in coming years if such trees are not favored with crown release.

In sawtimber stands, crown release can be applied as a commercial operation, thus yielding timber sale revenue to offset other management costs while favoring selected crop trees. The problem with delaying the release until operations are commercial is that the number of remaining crop trees is relatively low in older stands and the opportunity to enhance oak stocking has long been lost. The overstory in commercial stands may include 148 to 173 trees per hectare, but usually less than half of them are desirable crop trees if the

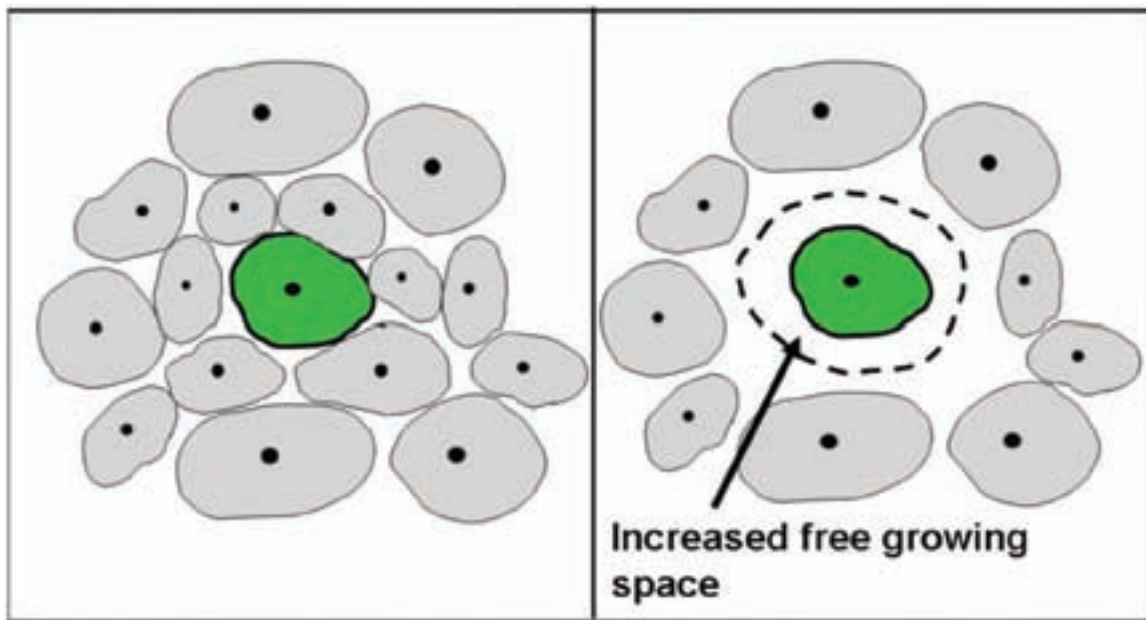


Figure 7—A crop tree crown (dark) shown from above the forest canopy. The left diagram represents a crop tree crown before release with 6 adjacent competitors. The right diagram illustrates the free growing space available when a crown-touching release is applied to remove competing trees from all sides of crop tree. (Diagram produced by J.W. Stringer and published originally in Miller and Stringer 2007).

stand has not been managed before. Ideally, release treatments should be applied when the stand is younger to retain as many crop trees, and as many oak crop trees in the overstory as the stand matures.

Cleanings can be used to deaden or remove trees that are limiting the horizontal crown expansion of an oak crop tree, thus increasing its free growing space and improving its survival probability. Generally, a crown-touching release is used to eliminate adjacent competing trees whose crowns touch that of the crop tree (fig. 7). The increase in growing space provides for an increase in direct sunlight and below-ground resources available to the crop tree. The crop tree can then develop more leaf area and a larger crown, thus increasing photosynthesis, stem diameter growth, and root growth. Improved vigor and crown size also has the potential to improve mast production of individual trees.

Thinning treatments can be used to provide various degrees of release based on the proportion of the crown that is left free-to-grow. In most cases, it is beneficial to retain trees in the overtopped and intermediate crown classes adjacent to crop trees. Such trees might be important for wildlife and aesthetics. They can also protect timber crop trees by partially shading their boles and protecting them from sunlight that can trigger epicormic branching.

Thinning Oak Stump Sprouts

Early thinning around oak sprout clumps can increase oak stocking at the end of the stem exclusion stage of stand development, especially on high quality sites. Weigel and others (2006) found an inverse relationship between oak stump sprout dominance probability and site quality within the range of 18 to 22 m site index for northern red oak in southern Indiana. Favoring oak reproduction through pre-commercial thinning gives managers greater densities of more highly competitive oak reproduction from which to work with to attain desired stocking levels at stand maturity. It is a better to have too much oak, rather than not enough, when sustaining oak is a primary management objective.

To maximize the development and competitiveness of oak stump sprout reproduction, it is helpful to thin around individual stump sprout clumps, thus releasing all the sprouts arising from a single stump. Thinning should be done early, i.e., by stand age 5. The number of oak stump sprout clumps to release follows the same recommendations given above for crop trees, and is ultimately determined by

their availability, spatial distribution in the forest, and the desired level of oak stocking at stand maturity. Not all oak stumps produce sprouts following harvesting; their sprouting capacity depends, in part, on the species, and tree size and age (Dey and others 1996, Johnson and others 2002). In general, northern red oak sprouting potential declines with increasing tree diameter and age (Weigel and Peng 2002, Weigel and others 2006). Initial sprouting frequency of northern red oak stumps increases with increasing site quality, but long-term survival and dominance of oak stump sprouts declines with increasing site quality due to the greater intensity of competing vegetation than is found on lower quality sites (Weigel and Peng 2002, Weigel and others 2006). Release of desirable oak stump sprout reproduction may not be enough to meet oak stocking goals, especially when regenerating older stands that have lower stump sprouting potential. Therefore, additional crop tree release of young oak that regenerated as advance reproduction may be necessary.

Early thinning (e.g., at age 5) of oak sprout clumps to a single stem can significantly increase tree diameter for species in the sections *Quercus* and *Lobatae* (Lowell and others 1987, 1989, Dwyer and others 1993). Johnson and Rogers (1980, 1984) estimated that 25th-year diameter of northern red oak stump sprouts can be dramatically increased by thinning the sprout clump to a single dominant stem. Thinning oak stump sprouts at any age between five and 15 years significantly increased 25th-year diameter (fig. 8).

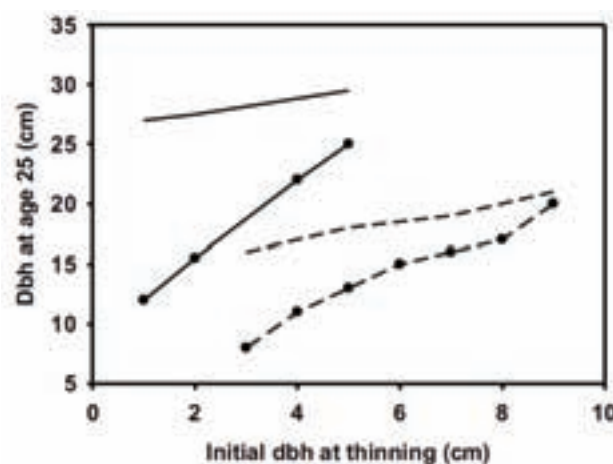


Figure 8— Comparison of estimated 25th-year diameters of northern red oak stump sprouts that were thinned to a single stem (lines without symbols) or not (lines with symbols) at ages 5 years (solid lines), or 15 years (dashed lines) (Johnson and Rogers 1980, 1984).

Initially smaller diameter oak sprouts showed the greatest increase in diameter by stand age 25. The earlier the thinning, the greater was the expected increase in 25th-year diameters. Groninger and others (1998) thinned 12-year-old chestnut oak (*Q. montana* Willd.) and scarlet oak (*Q. coccinea* Muenchh.) stump sprout clumps in Virginia clearcuts by reducing sprouts to a single dominant stem per stump and removing surrounding competitors that had crowns touching the oak crop tree. They found that thinning by chainsaw or herbicide (stem spray or stem injection with 2,4-D or triclopyr): (1) effectively controlled competing stems both within and external to the oak stump sprout clump; (2) significantly increased crop tree diameter after five years; (3) did not affect total height; and (4) herbicides did not cause unusual mortality of crop trees.

CONCLUSION

Sustaining oak stocking requires managing forest regeneration and recruitment processes to enhance oak regeneration potential and its long-term competitiveness through to stand maturity. There are three defining stages in stand development relevant to sustaining northern red oak. In the initial mature forest stage, beginning several years or more before the parent trees are removed, sources of oak reproduction must be present in sufficient numbers, size, and distribution to compete with other vegetation on the site. Oak regeneration potential at this point is largely determined by the probable contributions expected from advance reproduction and stump sprouts. In the stand initiation stage, oaks must be released by a timely and adequate reduction in the canopy such that residual trees do not interfere with the oaks as they compete for dominant or codominant status in the young stand. Once the canopy of the young stand closes, a third critical stage begins during which the young oaks in the upper canopy of the new stand must maintain their status for decades until the stand is mature. Again, upper-canopy oaks can be sustained for long periods by providing a timely and adequate crown release in the young stand.

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Restoring East-Side Ponderosa Pine Ecosystems at the Blacks Mountain Experimental Forest: A Case Study

Jianwei Zhang and Martin W. Ritchie

ABSTRACT

The ecological research project of interior ponderosa pine forests at the Blacks Mountain Experimental Forest in northeastern California was initiated by an interdisciplinary team of scientists in the early 1990s. The objectives of this study were to determine the effects of stand structure, and prescribed fire on vegetation growth, resilience, and sustainability of ponderosa pine ecosystems. Two stand structures were created with mechanical thinning on 12 units of about 100 hectares each. The low structural diversity (LoD) treatment has an even spaced, continuous canopy with few snags and few large canopy gaps. In contrast, the high structural diversity (HiD) includes many large, old trees with multiple canopy layers, abundant large diameter snags, many small canopy gaps, and some dense clumps of smaller trees. Each structural treatment was randomly assigned in each of three blocks. Each combination (unit) was split into two halves and prescribed fire was applied to one of the split plots after the completion of thinning. Because of the complexity of treatments and large experimental units, thinning and prescribed burns took 5 years to complete. Accordingly, our first post-treatment measurements were staged one year after implementation of prescribed fire. The second post-treatment measurements were completed five years after the initial measurement. In the short term,

we found that (1) tree growth and understory woody plant cover significantly responded to structural diversity, but not prescribed fire; (i) the HiD plots carried much more basal area and stems than low structural diversity (LoD) plots, (ii) both individual-tree and stand-level growth increment was higher in the LoD plots than in the HiD plots, and (iii) understory woody plant species grew back in five years with higher percentage cover in the LoD than in the HiD, (2) prescribed fire effect was significant for dead woody materials on the forest floor and bark beetle colonization; (i) debris was reduced significantly more with prescribed fire and (ii) bark beetles tended to attack trees weakened by the prescribed fire, (3) by comparing adjacent untreated stands, our HiD treatment appeared to improve health of the largest trees (DBH > 60 cm) in the stand, and (4) no significant interaction was found between structural diversity and prescribed fire. These results suggest that ponderosa pine forests can be treated to enhance stand growth and health without sacrificing understory vegetation diversity and stand productivity in the short-term.

Keywords: interdisciplinary research, ponderosa pine forest, thinning, prescribed fire, stand dynamics, understory shrubs

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INTRODUCTION

In recent decades the Forest Service has applied a multiple-use philosophy in managing the national forests and grasslands (Wilkinson and Anderson 1987). This has not only focused on the production of valuable commodities such as wood, fiber, food, water, and minerals for human use, but also emphasized recreational, spiritual, aesthetic, and educational experiences on the public lands (Kessler et al 1992). Effective management can be helped by interdisciplinary research that addresses multiple factors (Oliver 2000).

Recommendations from the National Research Council (1990) and the Forest Service Strategic Plan for Research (1990) led the Pacific Southwest Research Station to establish an interdisciplinary team to consider research needs in interior pine forests. The team began planning a large-scale ecological research project at Blacks Mountain Experimental Forest in northeastern California in 1991. Experimental design, National Environmental Policy Act (EPA) documentation, and timber sale preparation took some five years; treatments were initiated in 1996. Treatment installation was completed in October of 2000 when the prescribed burning treatment was applied to the last block.

The objectives of this long-term study are to increase our understanding of the effects of forest structural complexity and prescribed fire on health and vigor of an interior ponderosa pine ecosystem and to determine how these ecosystems can be managed for sustained resource values (Oliver 2000; Oliver and Powers 1998). Blacks Mountain Experimental Forest was chosen for this project because of the existing forest conditions with a representative late seral structure still extant. Designated in 1934 as an Experimental Forest, it is one of the few forests in the United States with more than 50 years of records on forest structure. The forest has an extensive, well-designed road and compartment system.

In this report, we will review some short-term results from this study from the initial 5-year post-treatment period. These results are primarily from manuscripts submitted to a special issue of Canadian Journal of Forest Research entitled "Interdisciplinary Research on Interior Pine Forests at Blacks Mountain Experimental Forest" submitted for publication in 2008. We appreciate all authors who allowed us to use their unpublished data.

SITE DESCRIPTION

Blacks Mountain Experimental Forest is located approximately 35 km northeast of Mount Lassen in northeastern California (40°40' N latitude, 121°10' W longitude). The forest occupies 3,715 ha on the Lassen National Forest; elevational range is between 1,700 m and 2,100 m. Soils are Typic Argixerolls, 1-3 m deep over basalt with mesic soil temperature regimes at lower elevations. Andic Argixerolls with frigid soil temperature regimes predominate at higher elevations. The climate is characterized by warm dry summers and cold wet winters. Annual precipitation is approximately 460 mm and falls primarily as snow from October through May. Mean air temperatures usually range from -9 to 29°C.

The overstory vegetation on the experimental forest is dominated by ponderosa pine (*Pinus ponderosa* Dougl. ex Laws.) with varying amounts of white fir (*Abies concolor* (Gord. & Glend.) Lindl. ex Hildebr.), incense-cedar (*Calocedrus decurrens* (Torr.) Florin), and Jeffrey pine (*Pinus jeffreyi* Grev. & Balf.) (Oliver 2000). The understory contains abundant regeneration of both ponderosa pine and white fir. The five shrub species accounting for most of the ground cover are: greenleaf manzanita (*Arctostaphylos patula* Greene), prostrate ceanothus (*Ceanothus prostratus* Benth.), snowbrush (*Ceanothus velutinus* Dougl. ex Hook.), antelope bitterbrush (*Purshia tridentata* (Pursh) DC), and creeping snowberry (*Symphoricarpos mollis* Nutt.).

EXPERIMENTAL TREATMENTS

The study covers 1,226 ha of the experimental forest and 111 ha of the adjacent national forest. Silvicultural treatments and grazing have been imposed under a completely randomized split-plot design (Figure 1), with combinations of structural diversity (high vs. low) of the residual stand and grazing (grazed vs. ungrazed) as the main plot factors and with (F) and without (NoF) prescribed fire as the subplot factor (Oliver and Powers 1998, Oliver 2000). Whole-plot experimental units ranged from 77 to 144 ha ($\mu = 111$ ha) and were blocked by proportion of species composition rather than geographic proximity because white fir and incense-cedar become increasingly abundant at the higher elevations in the forests. The two structural treatments were randomly assigned to six experimental units each. The high structural diversity (HiD) treatment was designed to leave all

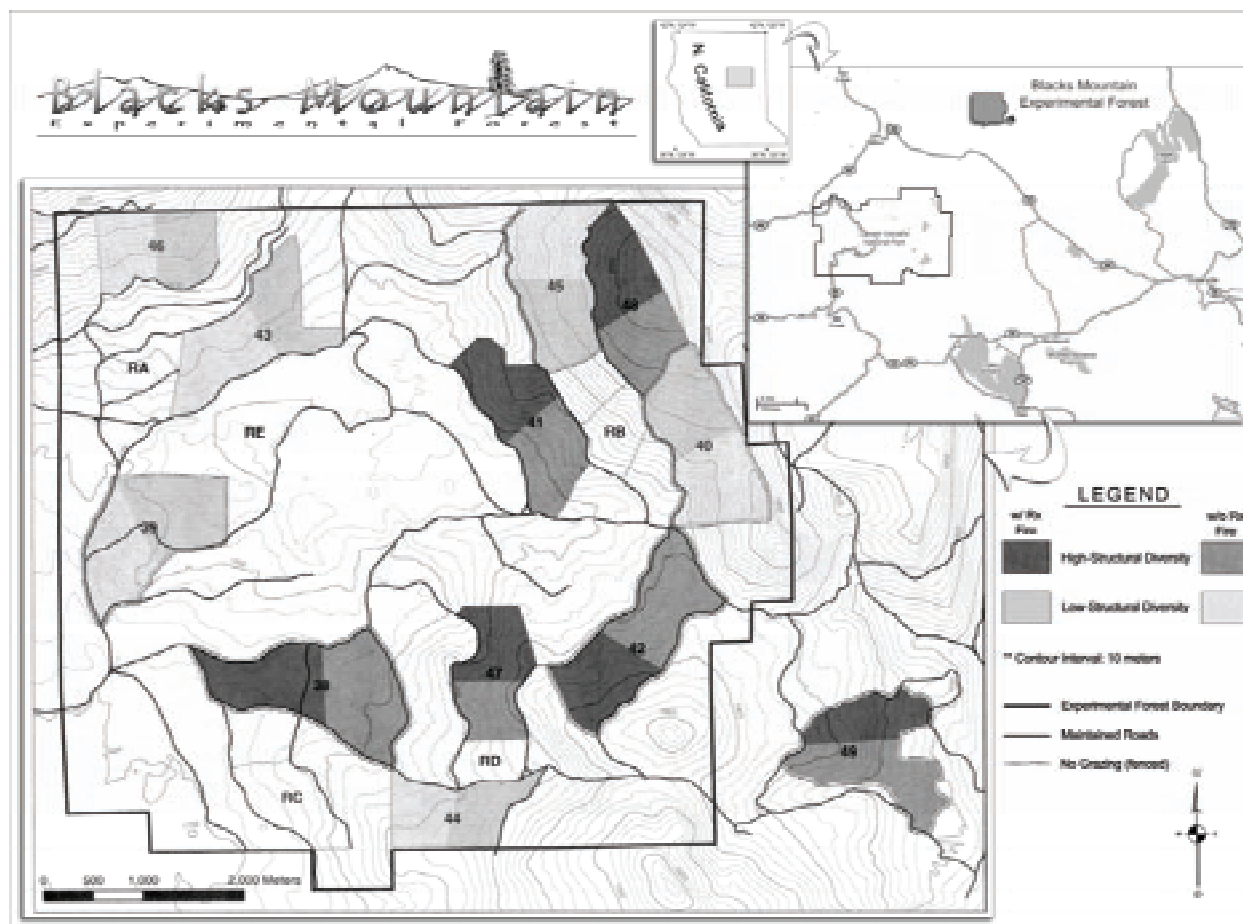


Figure 1— Location of the Blacks Mountain Experimental Forest and layout of the Blacks Mountain Ecological Project in northeastern California, USA.

large, old dominant trees, abundant snags, multiple canopy layers with dense clumps of smaller trees, and many canopy gaps and forest floor openings (Oliver and Powers 1998, Oliver 2000). The low structural diversity (LoD) treatment produced a single canopy layer of well-spaced co-dominant trees, few snags, and few large canopy gaps and forest floor openings (Figure 2). Five Research Natural Areas, which are not included in this study, are well-distributed within Blacks Mountain Experimental Forest (Figure 1) and could provide information on untreated stands.

Because of large plot size, treatment implementation took several years. Three individual blocks, each with two HiD units and two LoD units were created in 1996, 1997, and 1998 respectively. The prescribed fire was planned for the fall of the following year. Because the conditions needed to meet the burning prescription could not be met in 1998, prescribed fire for blocks two and three were

delayed to the fall of 1999 and 2000 respectively. Duff litter around existing snags was raked prior to application of prescribed fire.

Six of twelve units were fenced to exclude grazing in the summer of the following year when the structural diversity treatment was applied. After treatment installation, an interdisciplinary team of scientists observed that our grazing treatment has not been effective because grazing intensity is largely a function of proximity to water, which limits most grazing to block one. In addition, a preliminary analysis indicated that there was no significant grazing effect for all variables measured. Therefore, the grazing treatment will not be considered in the analysis, thus doubling the replications within each block.



Figure 2— Stand structure and species composition on low structural diversity without prescribed fire, low structural diversity with prescribed fire, high structural diversity without prescribed fire, and high structural diversity with prescribed fire in the pre-treatment, immediate post-treatment, and five years after post-treatment at the Blacks Mountain Ecological Project in northeastern California, USA.

SAMPLING PROCEDURES

Throughout each of the experimental units, a permanently monumented grid on 100-m centers was installed (Oliver 2000). The monuments were located by conventional survey methods. Actual location of each monument was within 6 inches. Monuments are 46-cm rebar with an aluminum cap stamped with the main plot number and Universal Transverse Mercator (UTM) coordinate. All measurements were referenced to these grid points. Among variables that have been measured, trees were most intensively measured; the detailed methods were described in Zhang et al (2008). For example, a pre-treatment tree inventory (DBH > 9.1 cm) was conducted in 1994, with a 25 % grid point sampling intensity. The purpose of the pre-treatment inventory was to collect the baseline data that was utilized to designate the block and unit layout. The post-treatment inventories were more intensive with 50% of the grid points systematically sampled in the year following the prescribed fire application (1998, 2000, and 2001). Every tree with DBH > 9.1 cm DBH has been tagged. As part of the study, the plots are scheduled to be re-measured every five years after the first post-treatment inventory. The second post-treatment re-measurements were conducted in 2003, 2005, and 2006 by blocks.

Understory woody plants, species number and cover percentage were also measured during the post-treatment survey (Zhang et al. 2008). Dead woody materials were measured prior to and post treatment (Brown 1974; F.C.C. Uzoh and C.N. Skinner, unpublished data). A census of bark beetle-induced mortality was established (Fettig et al. 2008). Because there have been many more measurements conducted in the study, we are only focusing on a few variables that have been thoroughly analyzed.

TREATMENT EFFECTS ON FOREST COMPONENTS

Dead woody fuels on the forest floor

Dead woody materials were sampled with the planar intercept method before the treatment was installed in 1994 and one year following treatment. Volume ($\text{m}^3 \text{ha}^{-1}$) of sound and decayed woods were separately calculated and converted to dry weight (Mg ha^{-1}) using method of van Wagtendonk et al (1996) (F.C.C. Uzoh and C.N. Skinner, unpublished data). Results show that prescribed fire has significantly reduced both sound and rotten woody materials on the forest floor (Figure 3). Effect of structural diversity was not significant. Although woody materials

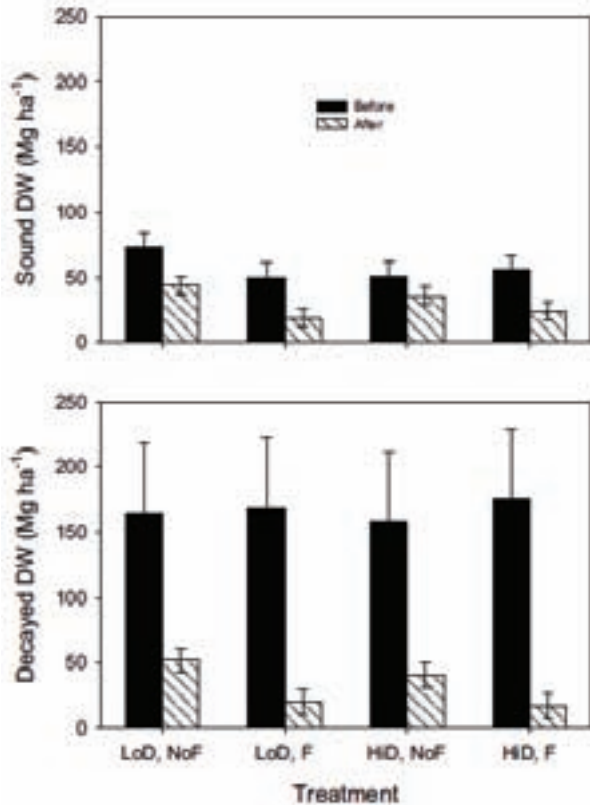


Figure 3— Mean dry weight ($\pm 1SE$) of sound and decayed dead woody materials (F.C.C. Uzoh and C.N. Skinner, unpublished data) before and after treatments were installed at the Blacks Mountain Ecological Research Project in northeastern California. LoD = low structural diversity, NoF = without prescribed fire, F = with prescribed fire, and HiD = high structural diversity.

with advanced decay were more prevalent than sound woody materials prior to treatments, the amounts were comparable after treatment due to breaking up of larger materials by harvesting machines (Weatherspoon 1983).

Understory woody plants

Using the planar-intercept method on a 100 m transect associated with sampled grid points for the post-treatment measurements, the first post-treatment inventory did not reveal any significant differences in shrub cover or number of species with respect to structural diversity or prescribed fire (Figure 4). Five years later, shrub cover was significantly higher on LoD units, but number of species did not vary with structural diversity. A potential explanation is that tree canopy cover in LoD treatment is much lower than that in HiD treatment (Vaughn and Ritchie 2005). Shrubs may have responded to this with increased growth with more sunlight and perhaps more water available under the more open canopy. However, these treatments did not

alter the seed bank or propagules of the existing species before treatment and nor did they appear to create favorable conditions for exotic species.

Stand dynamics

The thinning treatment produced two very different stand structures. The average values of LoD and HiD treatments were 10 vs. 25 $m^2 ha^{-1}$ for BA (Figure 5) and 282 vs. 513 trees ha^{-1} for number of trees (Figure 6), respectively. Based on the initial post-treatment measurements, the upper limit for diameter in LoD units was 60 cm and BA was concentrated between 20 and 40 cm DBH classes. In contrast, HiD treatment maintained trees across a broader range of DBH classes; BA was concentrated between 20 and 80 cm DBH classes (Zhang et al. 2008).

Based on the second post-treatment measurements, structural-diversity effect was still significant in stand BA and density; the average values of LoD and HiD treatments were 11 vs. 25 $m^2 ha^{-1}$ for BA and 258 vs. 487 trees ha^{-1} for

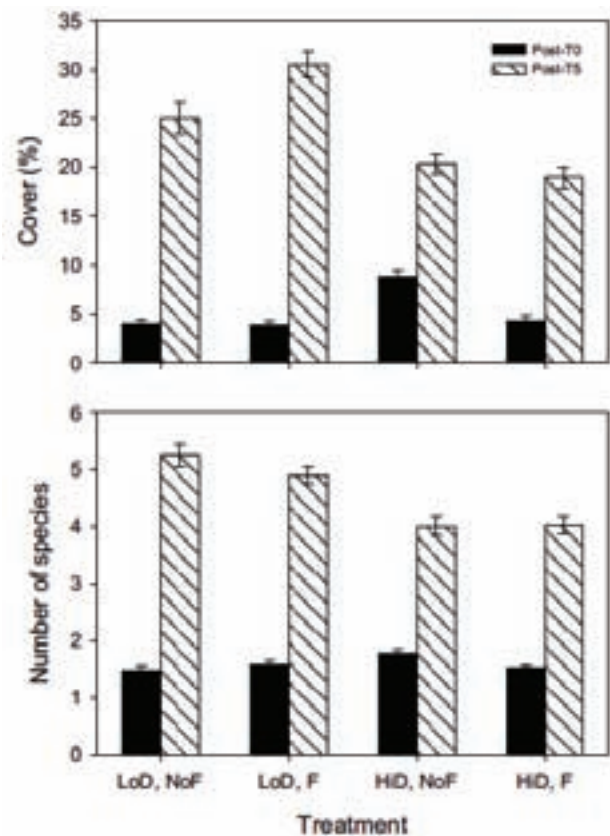


Figure 4— Overall shrub cover (%) and number of species ($\mu \pm 1se$) grown in the treatments at the Blacks Mountain Ecological Research Project measured immediately (Post-T0) and five years (Post-T5) after treatments were installed (Zhang et al. 2008). Abbreviation of x-axis label is in the legend of Figure 3.

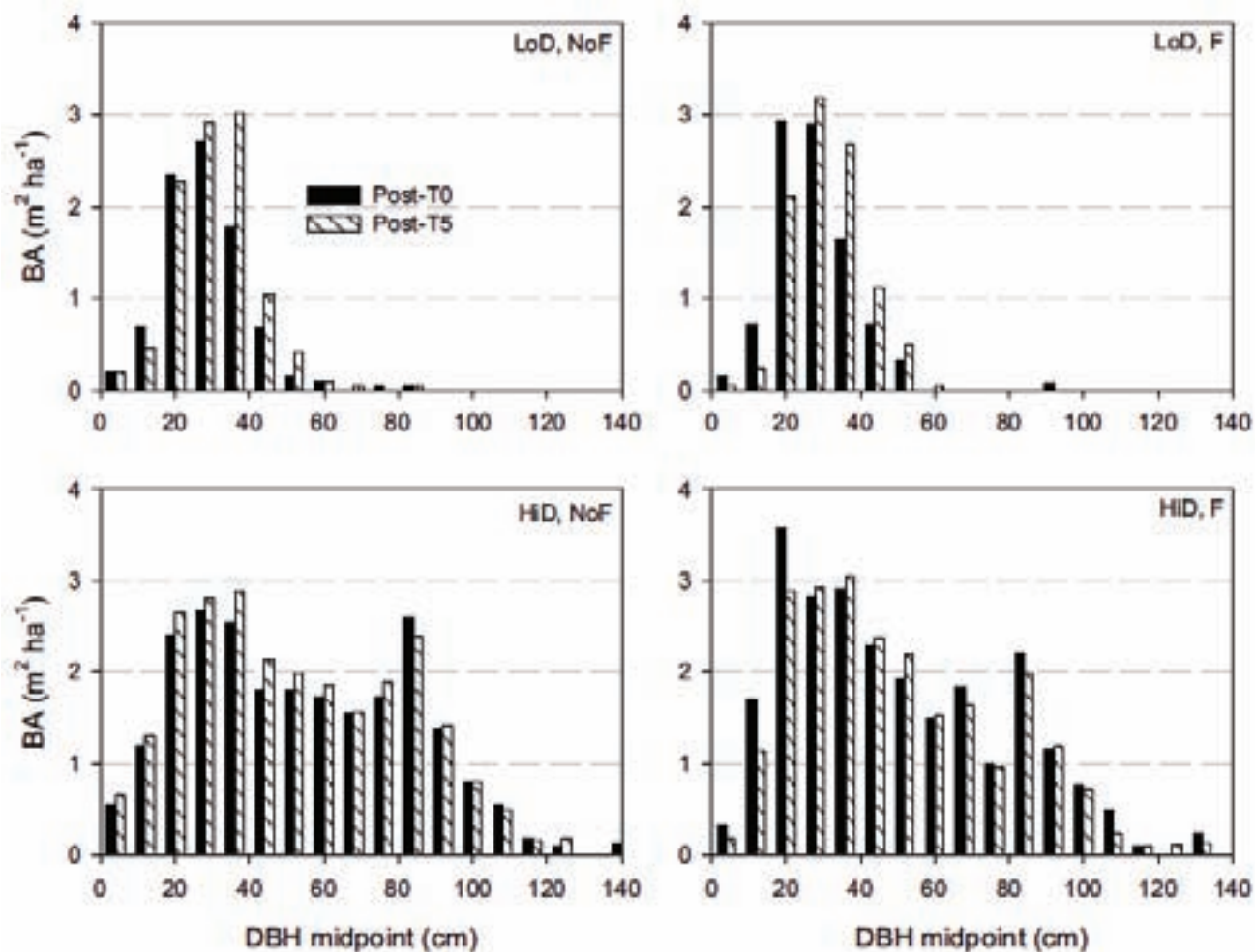


Figure 5 - Diameter-class basal area distributions for trees grown in the LoD (upper two) and in the HiD (lower two) without prescribed fire (left two) and with prescribed fire (right two) at the Blacks Mountain Ecological Research Project measured immediately (Post-T0) and five years (Post-T5) after the treatments were installed.

number of trees, respectively. Prescribed-fire effect was also significant for trees ha^{-1} . Differences were mainly related to mortality caused by prescribed fire and bark beetles. The mortality tended to occur in the smaller DBH classes (<20 cm) (Figures 5 & 6). During the five year period, mean trees ha^{-1} increased 11% (32 tree ha^{-1}) in the LoD without prescribed fire and decreased 28% (79 tree ha^{-1}) in the LoD with prescribed fire. Similarly, we found a 13% (67 tree ha^{-1}) increase in trees ha^{-1} in the HiD without prescribed fire and a 29% (145 tree ha^{-1}) decrease in the HiD with prescribed fire between the first and second post-treatment measurements. The results demonstrated that prescribed fire damaged or killed some trees. Many trees apparently alive in the first post-treatment measurements died during the next 5 years. Therefore, mortality assessments should consider both primary (directly killed by fire) and secondary

(weakened by fire and killed by insects or disease) mortality after the prescribed fire to obtain accurate estimates (McHugh & Kolb 2003).

Stand diameter (QMD) grew significantly more in LoD treatment ($0.65 \pm 0.02 \text{ cm yr}^{-1}$) than in the HiD treatment ($0.33 \pm 0.01 \text{ cm yr}^{-1}$) (Figure 7). Effect of structural diversity was not significant for PAI BA, although PAI BA was slightly higher for the LoD treatment ($0.38 \pm 0.02 \text{ m}^2 \text{ ha}^{-1} \text{ yr}^{-1}$) than for the HiD treatment ($0.34 \pm 0.02 \text{ m}^2 \text{ ha}^{-1} \text{ yr}^{-1}$). Trees surviving fire grew as much as trees without fire, confirming that low intensity surface fires may not affect tree growth (Busse et al. 2000; Ritchie and Harcksen 2005).

Stand productivity based on living trees measured at both times may be underestimated for the LoD plots rela-

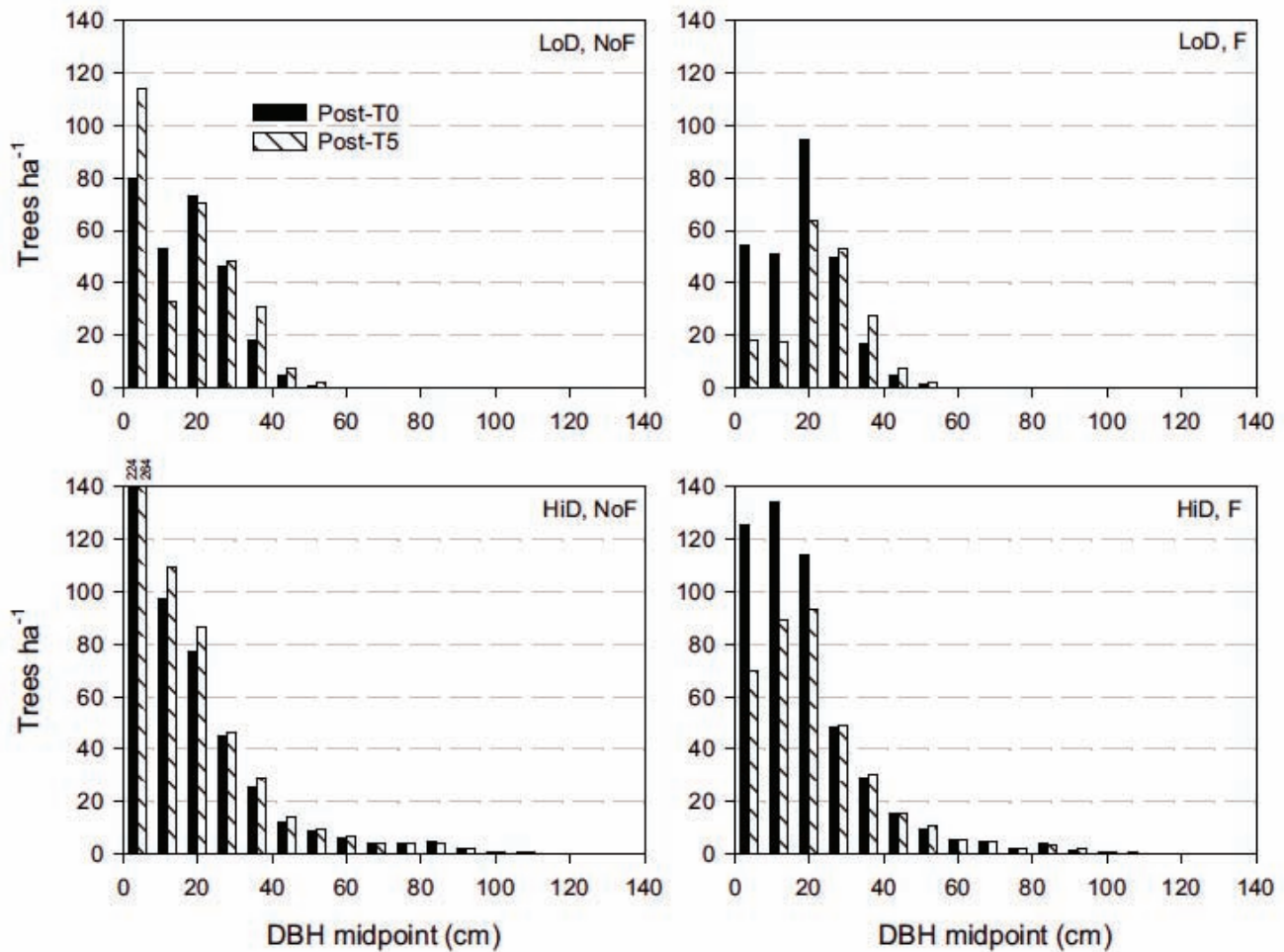


Figure 6 - Diameter-class frequency distributions for trees grown in the LoD (upper two) and in the HiD (lower two) without prescribed fire (left two) and with prescribed fire (right two) at the Blacks Mountain Ecological Research Project measured immediately (Post-T0) and five years (Post-T5) after the treatments were installed.

tive to the HiD plots if the Langsaeter's growth curve was considered (*sensu* Daniel et al. 1979, p. 318). When an old stand was thinned, annual increment in the earlier periods was strongly related to the initial stand density as found on other long-term density studies in ponderosa pine (Cochran and Barrett 1993, Oliver 1997, Zhang et al. 2006). The trend would not change until the onset of inter-tree competition occurs again (Smith et al. 1997, Zhang et al. 2006). In this regard, it is perhaps surprising that the LoD plots, which carry less than half the BA of the HiD plots, showed such elevated growth levels. If we considered growth proportion by dividing by initial BA to compare both treatments, the LoD would have showed even higher percentage growth ($0.38/10 = 3.8\% \text{ yr}^{-1}$) than the HiD would ($0.34/25 = 1.4\% \text{ yr}^{-1}$).

Large trees in the HiD

One of the objectives in restoring ponderosa pine ecosystems is to improve the health of large dominant trees by minimizing environmental stresses. Risk rating is one metric for tree health (Keen 1943; Salman and Bongberg 1942). Evaluating risk rating for the large trees (DBH > 60 cm) in the HiD units, Ritchie et al. (2008) found far fewer of these trees were rated as "high-risk" than the large trees grown on adjacent untreated Research Natural Areas (Figure 8), although the number of large trees was approximately the same on both HiD and RNA. This trend held in both burned and unburned units. Both treatments appear to improve the health of these large dominant trees.

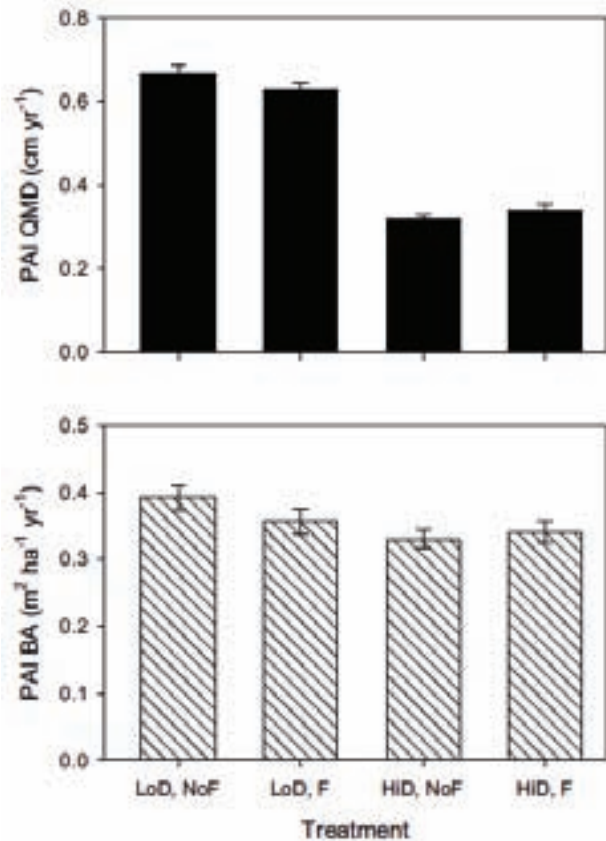


Figure 7— Average ($\mu \pm 1se$) periodic annual increment (PAI) for quadratic mean diameter (QMD) and basal area (BA) for different treatments calculated from first post-treatment measurements and second post-treatment measurements (redrawn from Zhang et al. 2008). Abbreviation of x-axis label is in the legend of Figure 3.

Bark beetle colonization

Bark beetles attack significantly more trees on plots in prescribed fire splits than on plots without prescribed fire, regardless of structural diversity (Figure 9). About 86% of bark beetle-caused mortality occurred in the prescribed fire splits, mainly in smaller diameter classes (Fettig et al. 2008). This result may explain the significantly different mortality between the first and second post-treatment measurements within the prescribed fire split. Therefore, secondary mortality may be expected even when initial fire-caused mortality is low because prescribed fire weakens trees that are susceptible to the bark beetle attack. It is not known if this same phenomenon will be observed with later applications of prescribed fire in stands with more vigorous trees and healthier crowns.

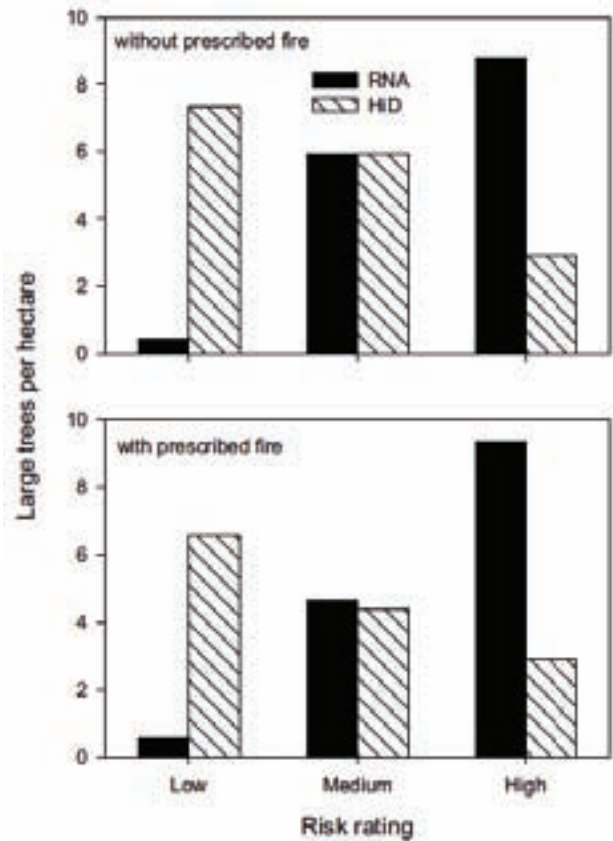


Figure 8— Large trees (DBH > 60 cm) per hectare sorted by risk rating observed from the high structural diversity (HiD) treatment and adjacent Research Natural Area (RNA) without and with prescribed fire at the Blacks Mountain Experimental Forest. Figures were drawn based on data of Ritchie et al. (2008).

IMPLICATIONS

Across northeastern California, many crowded ponderosa pine forests are threatened by severe, stand replacing fire and bark beetle outbreaks. Forest managers are very interested in using silvicultural tools to restore resilience in these ecosystems. Because the problems have resulted from decades of fire exclusion combined with inadequate stand density management in this fire-adapted ecosystem, they cannot be solved overnight. This project demonstrates a good start, by successfully using thinning and/or prescribed fire to treat these forests. The treated stands contain much fewer high-risk trees than untreated stands, suggesting that the health of the treated stands has been improved within five years. Based on results of dead woody materials and shrub cover for the initial five years, it appears that the dead woody fuels on forest floor only respond to prescribed fire (Figure 3) while the growth of shrubs is significantly associated with over-story structural diversity but not

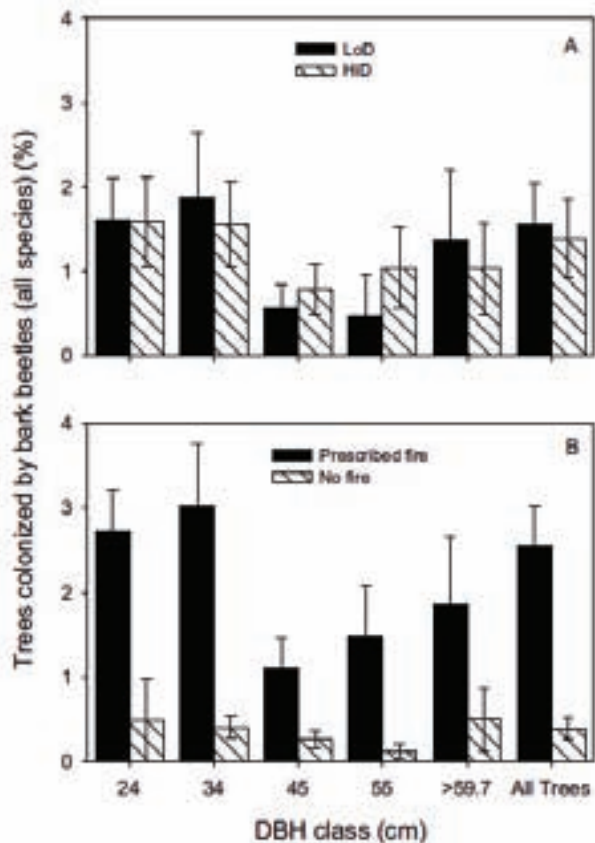


Figure 9— Mean ($\mu \pm 1se$) percentage of tree colonized by bark beetles (all species combined) by diameter class for all trees by structure treatment (A) and with and without prescribed fire (B) at the Blacks Mountain Ecological Research Project (redrawn from Fettig et al. 2008).

with prescribed fire (Figure 4). These results may explain the findings by Ritchie et al. (2007) that a combination of thinning and prescribed fire is more effective than thinning only at reducing wildfire severity.

The stand structures considered in this study are representative of two contrasting management priorities. The LoD stand is consistent with a timber production emphasis – maintaining thrifty fast-growing trees — yielding a much higher rate of growth both for individual trees and at the stand level. The HiD stand is consistent with an emphasis on maintenance of a range of tree sizes that may be more consistent with historic (pre-settlement) conditions. This increase in structural diversity comes at a significant loss in productivity. This cost may be offset by other amenities (e.g. wildlife habitat, visual resources) that they may provide. With either type of stand structure, use of prescribed fire may be beneficial in maintaining the health and vigor of these stands in the long term. However its application may reduce the occurrence of snags and woody debris, and both of these contribute to wildlife habitat.

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Silvicultural Strategies for Restoring Missouri Ozark Ecosystems on the Mark Twain National Forest

Paul Nelson and Charly Studyvin

ABSTRACT

The 1.5 million acre Mark Twain National Forest (MTNF) spans thirteen ecological subsections stretching over 200 miles of the Ozark Highlands in Missouri. Historically, this landscape contained a diverse assemblage of savanna, woodland, forest, glade, fen, spring, stream and cave natural communities. However, following European settlement, decades of domestic livestock overgrazing, logging, conversion to agriculture and fire suppression led to degradation and loss of these natural communities. This degradation resulted in loss of biodiversity, altered successional pathways, soil loss, reduced forest productivity, oak decline

and fuel buildups. Today, ecosystems and the disturbance processes that they depend upon are out of character relative to historic reference conditions across most of the MTNF. The 2005 Forest Plan allocates 29 percent of the forest, or 438,000 acres, to Management Prescriptions 1.1 and 1.2, which emphasize the restoration of globally significant natural communities. A basic restoration premise is to restore the structural vegetative condition and emulate historical disturbance processes and functions under which natural communities evolved and were adapted. Implementation strategies and silvicultural methods are discussed.

Keywords: Ozark Highlands, ecosystem restoration, natural communities, Mark Twain National Forest

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INTRODUCTION

The 1.5 million acre Mark Twain National Forest (MTNF) encompasses 9 separate units that span thirteen ecological subsections stretching over 200 miles of the Ozark Highlands. It contains unique, diverse, ecologically complex and globally distinct biological areas. Historically, this landscape contained a diverse assemblage of savanna, woodland, forest, glade, fen, spring, stream and cave natural communities that provided habitat for nearly 2,000 vascular plant species and nearly 700 vertebrate animal species including over 150 endemics (Ozark Ecoregional Assessment Team, 2003). Some 12,500 years ago, Native Americans occupied Missouri's landscape, and influenced the composition, structure and distribution of natural communities (see Wettstaed in Nelson, 2005). The vegetation at the time of European settlement was for the most part composed of essentially indigenous plants and animals that co-evolved and adapted to centuries of aboriginal disturbance regimes, especially fire (Guyette 1991, Nelson 2005). Wind and storm events, and native grazing ungulates such as deer, elk, and woodland bison also influenced the distribution, productivity, patterns and composition of vegetation making up some 65 distinct natural communities on the Forest.

By the early 1800s, Native Americans and their practices of landscape burning, hunting-gathering and farming from small villages had been largely eliminated from the Ozark region. Increasing settlement by Europeans brought industrial logging (resulting in the nearly complete unregulated highgrading of virgin timber), lead mining, charcoal production, subsistence farming, and open range grazing by domestic livestock (Smith 2003). Significant soil erosion problems occurred after the timber boom, mainly as a result of poor farming practices, overgrazing and annual burning. By the 1930s, much of the most productive soil resource was literally downstream due to severe sheet and gully erosion. Prior to European settlement the stream channels were narrow and deep, but over the years the river channels became wider, shallower and unstable, and large gravel bars developed along the stream beds (Smith 2003). The landscape the settlers depended on was exhausted by the early 1900s, and eventually some of these self-reliant folk were forced to turn to the government for assistance (Smith 2003).

The U.S. Forest Service started purchasing property in Missouri in the fall of 1933, and began the process of landscape stabilization by planting trees, instituting a grazing policy, and excluding fire. Missouri's fire prevention program was especially aggressive as state and federal agencies were convinced that burning by locals was destroying timber and wildlife habitat, and was contributing to soil erosion. However, these early efforts at management could not alter the fact that decades of abuse and disruption of disturbance regimes led to the degradation of natural communities. This degradation resulted in loss of biodiversity, unnatural succession, soil erosion, reduced productivity of timber products, oak decline and fuel buildups (USDA Forest Service 2005a). On the MTNF, eight decades of management since the 1930s have gone a long way to stabilize the landscape and soil resource, and to reestablish a productive and sustainable forest condition. But these forests are substantially different from the ecosystems that existed prior to European settlement, largely because key disturbance processes, notably fire, have been effectively removed from the landscape. As a result, ecosystems that are adapted to fire, and the important flora and fauna they contain, are dramatically underrepresented in the modern forest landscape.

In 2002 the MTNF began the process of revising the 1986 Forest Plan. Among the revision topics identified as a need for change was ecological sustainability and ecosystem health. The MTNF identified opportunity areas to conserve the best, most viable arrays of ecosystems, plants and animals. Opportunity areas were delineated using information from the following sources:

- Ozarks Ecoregional Conservation Assessment (TNC 2003)
- The Terrestrial Natural Communities of Missouri (Nelson 2005)
- The Missouri Natural Areas Program
- Partners In Flight; Ozark-Ouachita Physiographic Region
- Missouri Resource Assessment Partnership
- The Missouri Grasslands Coalition

The MTNF worked closely with the Missouri Chapter of The Nature Conservancy (TNC) to gather information on conservation areas and discuss the results of the Ozarks Ecoregional Conservation Assessment (OECA) (Ozarks

Ecoregional Assessment Team 2003). This assessment identified globally significant species, natural communities, and ecological systems in the Ozark ecoregion, and helped determine the spatial configuration that would most efficiently conserve viable examples of globally significant biodiversity features. The MTNF used the OECA to focus conservation planning and management efforts on specific opportunity areas on Forest lands, also known as “portfolio sites.” As a result of these discussions and planning, the Forest Planning Team, and appropriate ranger district personnel, identified 19 separate areas to be managed for ecosystem restoration (see Appendix A of the 2005 Management Plan for the MTNF).

These 19 areas, comprising 29% of the forest (438,000 acres), are in Management Prescriptions 1.1 and 1.2, which emphasize restoration of globally significant natural communities. The plan includes specific objectives for restoration work on 100,000 to 150,000 acres during the planning period (ten years). These objectives were formulated by identifying minimum/maximum viability targets for natural community types. Natural community types include prairies, glades, savannas, open woodlands, closed woodlands, forests and fens. A basic management premise is to restore the structural vegetative conditions and maintain (emulate) the historical disturbance processes and functions under which natural communities evolved and were adapted. Fire disturbance is emphasized because of its historical role in the region, and its influence on natural community structure and species composition.

NATURAL COMMUNITY TYPES ON THE MTNF

In Missouri state and federal land managers have been classifying, inventorying, describing and restoring Ozark ecosystems as Natural Areas for 30 years (Nelson 2005). Missouri Natural Areas, and select locations in state parks and the MTNF, have been managed for their natural qualities for at least 25 years by emulating historic disturbance processes. These sites serve as reference benchmarks, exhibiting ecological conditions that existed prior to European settlement. Using these benchmarks as desired conditions, prescriptions can be written to move the current conditions of the project area towards those desired conditions. Project monitoring also relies on these reference sites to compare species richness and other ecological parameters.

Using the reference sites and other literature on the structure and dynamics of forest ecosystems, the MTNF developed a set of parameters, which were included in the forest plan, to describe the desired condition of these communities of interest.

- A Forest is a relatively large area (typically over 10 acres) dominated by trees forming a closed canopy and interspersed with multilayered shade-tolerant sub-canopy trees, shrubs, vines, ferns, and herbs. Most forests occurred on steep north-facing slopes, deep coves and in broader river valleys thereby limiting the effects of historic fires.
- Woodlands are highly variable natural communities with a canopy of trees ranging from 30 to 80 percent closure with a sparse understory (or midstory) and a dense ground flora rich in forbs, grasses and sedges. Two types of woodlands, open and closed, are managed on the MTNF. The gently to moderately dissected plains and hills of the Ozarks were conducive to the frequent spread of historical fires on which woodlands depend (Nelson 2005).
- Savannas are fire-dependent grasslands interspersed with open-grown scattered oak trees, groupings of trees of various age, and shrubs. The tree canopy cover is generally less than 30 percent. Broad, gently dissected plains facilitated more intensive and frequent fires, which resulted in the development of savanna landscapes.
- Prairies are fire dependent natural communities dominated by perennial grasses and forbs with scattered shrubs and very few trees (less than 10 percent cover). Intense, frequent surface fires propagated across the nearly level to gently rolling Missouri plains thereby resulting in the development of treeless prairies.
- Glades are open, rocky barren areas dominated by drought-adapted forbs, warm season grasses and a specialized fauna. They occur as openings primarily on southwest-facing slopes across the otherwise wooded Ozarks landscape. Trees and shrubs occur on glades, but are not dominant unless overgrazing and/or disruption of natural fire regimes have resulted in invasion by woody species like red cedar.

Presently, many natural communities on the MTNF are out of character (see Table 1); basal areas are too high and the tree canopies are closed with associated dense shading,

Table 1. Range of Ecological parameters for respective natural communities in Management Prescriptions 1.1 and 1.2

Natural Community Types	% canopy	Basal Area	Understory	Aspect, Slope, roughness	Shrub layer	Structural age/growth stages	Ground organic layer	Percent ground cover	Patch size
Prairie	<10	NA	NA	All aspects gentle slopes,	Sparse	Grassland with few scattered shrubs and trees	Grass, sedge, and forb cover	90-100	10 to 200 acres
Savanna	10-30	<30	Scattered oaks and Shrubs	Broad ridges, all aspects, Gentle slopes	Dense, mostly scattered oaks and other shrubs	Shrub oak/pine covering 10-25% of area	Grass, sedge and forb cover	90-100, grasses dominant	50 to over 1,000 acres
Open woodland	30-50	30-50	Mixed shrubs, early seral	Southwest-facing to upper ridges, gentle to steep, plains and hills	Dense, mostly scattered oaks and various shrubs	Shrub oak/pine covering 10-25% of area, even-aged stands	Grass, sedge, and forb cover, little accumulated leaf litter	60-80 grasses dominant	100 to over 1,000 acres
Closed Woodland	50-80	50-90	Early- to mid-seral trees	Upper ridges to base of north-facing slopes, gentle to steep, hills and break	Sparse, mostly scattered oak and various shrubs	Shrub oak/pine in 5-10% of area, even-aged stands	Shallow leaf litter, mixed grasses sedges and herbs	80-100	100 to over 1,000 acres
Upland forest	80-100	80-100	Shade-tolerant shrubs and small trees	Generally north-facing slopes, steep to very steep, hills and breaks	Sparse, scattered, vines present	Oak and mixed species of variable age, small isolated gaps 1-5 acres	Moderately deep leaf litter	50-70	10 to 100 acres
Bottomland Forest	80-100	90-100	Shade-tolerant shrubs and small trees	North-facing slopes, very steep or broad-level floodplains, hills and breaks	Sparse, vines present	Multi-layered, uneven-aged, few gaps	Deep leaf litter, ephemeral herbs	50-70	10 to 500 acres
Fen	<10	NA	NA	Toe slopes, ravines and floodplains	Dense to sparse or none	Shrub thickets to open herb/sedge meadows	Shallow marly to deep muck	90-100	<100 sq ft. to 15 acres
Glade	<10	NA	Small shrubs and trees, rock outcrops	Generally southwest-facing but all aspects on igneous and White River, steep to very steep, hills and breaks	Variable	Variable exposed	Sparse to dense grass cover, mineral soil often	30-90 grasses dominant	½ to 300 acres

heavy fuel loading from deep leaf litter, and sparse ground cover. As well as being out of character with respect to the range of ecological parameters, many areas are also out of character with respect to overstory species. An estimated 4.2 million acres of shortleaf pine forest and 2.4 million acres of mixed pine-oak forest existed prior to 1880 (Liming 1946). Based on 2003 Forestry Inventory and Analysis (FIA) data across 29 counties covering the MTNF, the extent of shortleaf pine and mixed pine-oak is 545,929 acres--which equals 8% of the pre 1880 estimate. Much of the MTNF that was historically populated by shortleaf pine or shortleaf pine/white oak mixtures is now dominated by trees of the red oak group--mostly black and scarlet oak (USDA Forest Service, 2005a).

RAPID ECOLOGICAL ASSESSMENTS

Although there are 438,000 acres covered by MP 1.1 and 1.2 where restoration is the main emphasis, it may not be possible or even desirable to prescribe restoration work for all of those acres. It is especially important to focus our

efforts on those ecological areas where restoration goals are most likely to succeed. The MTNF uses "rapid ecological assessments" (REAs) to assist in identifying those areas to best do the work. The REA evaluates the ecological condition, or health, of the landscape by evaluating species richness and restorability, concentrations of habitat variety, and how close the area is to the desired condition parameters for respective natural communities. The REA process uses a combined GIS and field evaluation approach conducted by the Forest Ecologist, Forest Botanist and respective ranger district staff to rapidly evaluate (within 2-3 months) larger management areas in MP 1.1 and 1.2. Because regular and frequent prescribed burning will be required to restore most natural community types, the REA also considers the feasibility of conducting large landscape-scale prescribed burns. The REA results in a report that lists high priority sites (future project areas) with restorable natural communities, sensitive areas, threats and risks, and operational management recommendations. Once identified by the REA, projects can be planned in those high priority areas that have the most opportunity to accomplish restoration

work using timber sales, and the Knutson-Vandenberg (KV) funds that those sales generate—which promotes opportunities for far greater habitat restoration than could be accomplished with appropriated dollars alone (Guldin and Graham 2007).

MAPPING OF NATURAL COMMUNITY TYPE AND HISTORIC VEGETATION

A process to develop a detailed natural community map for respective projects in MP 1.1 and 1.2 is outlined in the implementation guide for the 2005 forest plan. This process combines GIS mapping, modeling of the landscape, GLO notes and field ground truthing. Part of this analysis process also includes comparison of the natural community types with current vegetation conditions and knowledge about a local ecoregion's historic disturbance processes. Information and data that can be used include: land survey records, direct evidence of remnant shortleaf pine stumps from the initial cut of virgin timber in the late 1800s to early 1900s, studies/inventories of natural areas, writings of early travelers, county histories, naturalist accounts (early botanists, birders, etc.), archaeological/paleontological studies, and various terrain/historic vegetation models. Armed with this information, plus a knowledge of the description for the respective natural communities (Nelson 2005), resource managers can make informed decisions about on-the-ground natural community mapping.

Forest staff also use “indicator plant species” to help determine where the respective boundaries for various natural community types fall on the ground. While plant species occurrences vary widely across ecological subsections of the Missouri Ozark Highlands, certain plant indicator species are surprisingly faithful to a respective natural community type. For example, the presence of cream white indigo (*Baptisia bracteata*), turbinate aster (*Symphytrichum turbinellum*), little bluestem (*Schizachyrium scoparium*), lowbush blueberry (*Vaccinium pallidum*) and bristly sunflower (*Helianthus hirsutus*) suggests an open woodland that would be dominated by shortleaf pine (*Pinus echinata*), post oak (*Quercus stellata*) and black hickory (*Carya texana*); similarly, plants such as the elm-leaved goldenrod (*Solidago ulmifolia*), tick trefoil (*Desmodium nudiflorum*), Boscii's panic grass (*Dichanthelium boscii*) or woodland brome (*Bromus pubescens*) indicate a closed woodland that would

be dominated by white oak (*Quercus alba*) and mockernut hickory (*Carya tomentosa*).

SILVICULTURAL STRATEGIES

A primary focus of natural community restoration is emulating critical disturbance processes on a landscape scale. The 2005 Forest Plan for the MTNF provides quantifiable objectives for natural community types that are distributed across eight ecological subsections. Thus, landscapes in need of treatment across these subsections may range from several hundred to several thousand acres. How well natural community remnants recover (assuming we already have sufficient biotic components) depends on whether managers can restore desired physical/structural conditions and distribution patterns, reverse historic damage and then replicate historic disturbance processes across areas large enough to be ecologically and functionally meaningful when restored (see McCarty in Nelson 2005). The goal of restoring landscapes is met by executing silvicultural prescriptions in each of the stands that comprise the landscape under restoration.

The silvicultural prescription to apply in a given stand depends on three key sources of information: the natural community map for the region, the desired condition for the given natural community (Table 1), and the existing condition of the given stand determined from stand examinations and inventory data. Knowing these three elements allows the silviculturist to prescribe treatments to move the area from the existing condition towards the desired condition. Simply put, most management practices fall into some combination of thinning, regeneration, planting, prescribed burning and perhaps control of exotic species. Important measures of success in restoring natural communities include species richness (especially those plant species found in the ground layer), plant and animal indicators (particularly animals adapted to variable structural conditions), overstory composition, and absence of exotic species.

A real challenge to silviculturists is whether it is more important to restore those overstory species (in the case of shortleaf pine) now removed from most woodlands, or the more sensitive diversity of ground cover plant species, many of which are species of conservation concern. The answer is that both are important and the approach depends on the overall ecological conditions of the area to be treated.

Prescribed Burning

Effective application of prescribed burning is critical in restoring natural communities, and maintaining them once they are restored. “An irrefutable body of evidence exists that the biological landscape of the Ozarks reflects the effects of millennia of frequent, low intensity, dormant season fires set by humans” (Ozarks Eco-regional Assessment Team 2003). Many other documents cited in the new Forest Plan (US Forest Service 2005a 2005b) and Nelson (2005) strongly substantiate this evidence. Fire remains vital to the form and function of all types of savannas, woodlands, glades, fens and prairies on the MTNF. The absence of fires is one of the most significant change agents of modern times, second only to the outright destruction of native vegetation due to overgrazing, habitat conversion to other uses and exotic species invasion. Further, there is no similar or substitute treatment that can replicate the complicated chemical, biophysical and ecological effects of the burning from which ecosystems and plants/animals have evolved and adapted over millennia. However, it is not feasible nor possible to apply the burning process to all ecosystems on the MTNF. It is no accident that planning for ecosystem restoration on MTNF involved prioritization of applying prescribed fire to Management Prescriptions 1.1 and 1.2, and to those high priority areas identified in the Rapid Ecological Assessment process.

The reintroduction of fire requires an understanding of fire frequency and interval, fire intensity, and seasonal and spatial patterns of fire (see discussion by McCarty in Nelson 2005 regarding Missouri fire regimes). The progression of desired change and effects of fire application for restoring woodlands is well documented in Missouri (Nelson 2005). This information can be used to develop and implement an ecologically-based prescribed burn program to restore and maintain fire-adapted Ozark woodlands, savannas and glades (Table 2). A major challenge to resource managers practicing ecosystem restoration is how to model fire in a way that mimics the different roles that an average or extreme fire event played historically. This becomes especially difficult in a rural environment now fragmented by homes, farms and livestock; thus, it becomes more challenging to plan and execute the types of prescribed burns that will achieve conditions worthy of restoring and sustaining an ecosystem while protecting other values (timber, wildlife, soils, air quality, homes, etc.).

Restoration Thinning

Among the first treatments required in savanna and woodland restoration is a reduction in stem density in both the overstory and the midstory. The definition of restoration thinning is “thinning to a specified basal area to restore the natural community type.” The basal area specified (see Table 1) for savannas and woodlands is much lower than for a regular commercial thinning, and often would fall below the lower threshold for optimal utilization of the site from a traditional timber management perspective. The advantage of a low residual basal area is that resources become available for understory flora, a key component of composition and structure of desired woodlands and savannas.

However, it may be prudent to accomplish this reduction in stem density with a two-step process. Reducing the basal area to restoration levels in a single thinning can often result in unacceptable logging damage in the residual stand, and may predispose the stand to other kinds of damaging agents such as wind-throw and/or the spread of undesirable invasive plants. In addition, excessively heavy thinning can create unintended developmental dynamics such as a flush of hardwood growth from advanced regeneration and sprouts that would overwhelm the establishment of desirable ground flora (where problematic, the advanced regeneration of undesirable sprouting can be controlled through various means including herbicide application to stump sprouts, repeated prescribed burning and/or hardwood understory reduction treatments). Also, initial burns in thinned areas with accumulated deep leaf litter may be intense enough to reduce the basal area further than desired.

If it exists on the site, the desired historic vegetation (shortleaf pine, white oak) can be favored. Along with the thinning, a burning schedule should be established to reduce the leaf litter and encourage establishment of grasses, sedges and herbs. As an example, consider an open woodland site with a basal area of 120-150 ft²/ac. The restoration prescription suggests an initial thinning of sawtimber to a residual basal area of 70-80 ft²/ac of sawtimber followed by a series of three prescribed fires at 3-4 year intervals. The second thinning in about ten years would then further reduce the basal area to about 30-50 ft²/ac.

Table 2. Estimated fire frequency by natural community based on 30 years of applied ecosystem restoration.

Natural Community	Restoration Fire Landform	Maintenance Fire Frequency	Frequency
Prairie	Broad, level Plains	1-3 years	2-4 years
Savanna	Rolling Plains (lower intensity)	1-4 years	3-5 years
Open woodland	Dissected Plains and hills	1-3 years	3-7 years
Closed woodland	Hills and Breaks	2-3 years	3-10 years
Forest	Breaks, Floodplains	5-25 years	10-25 years
Glade	Dissected hills	2-3 years	4-10 years

removal of herbaceous ground cover by historic overgrazing. Many of these black and scarlet oak stands are also overstocked and reaching the end of their life span, creating conditions ripe for oak decline. An estimated 400,000 acres of the MTNF are currently at moderate to severe risk of oak decline.

Stands that show evidence of moderate to severe oak decline may be salvaged to recover economic value that would otherwise be lost, and to help move the forest towards the desired ecological condition for the given natural community in which the stand is located. Prescriptions vary depending on the percentage of black and scarlet oak in the stand. Salvage and sanitation cutting in stands with lower percentages of black and scarlet oak may result in residual basal areas that approach those desired in restoration thinning or shelterwoods. Harvesting

The thinning and burning prescription is simple to implement if the trees being thinned can be sold commercially, and if KV funds can then be collected and used to accomplish the burning. However, if the trees are not marketable, funding the thinning and the burning can be much more difficult. Options might include using fuels dollars to execute prescribed burning that may accomplish some thinning, as well as removing the leaf litter; doing non-commercial thinning and prescribed burning using KV funds generated from a timber sale in other stands within the sale area; or using appropriated funds. We also continue to explore the feasibility of marketing small diameter, low quality hardwoods for biofuels and other products.

Oak Decline Salvage

Black, red and scarlet oak now dominate many former open woodlands once dominated by shortleaf pine, post oak and white oak (Law et. al 2002). This out-of-character dominance has contributed to an increase in oak decline, a complex condition caused by the combination of degraded soils, removal of those dominant overstory trees better suited to dry open woodlands (Kabruck et. Al 2002), and

in stands with larger percentages of black and scarlet oak to be salvaged may mimic seed trees or clearcuts. Reforestation work such as artificial regeneration with shortleaf pine, or release of appropriate advanced shortleaf pine and/or oak regeneration may be required. Note that salvage activities are not subject to limits on the maximum size of temporary openings that restrict even-aged regeneration harvests.

Regeneration

The rotation age in MP 1.1 and 1.2 is increased from that in the general forest area—from 70 to 80 years for red, black and scarlet oak, from 70 to 100 years for shortleaf pine, and from 90 to 120 years for white oak. The 2005 Forest Plan projects 34,500 acres of reproduction cutting during the 10-year planning period intended to redress the imbalance of age classes found on the MTNF. Disturbance ecology and fire scar data suggest that those natural communities most affected by moderate to high intensity fire and wind damage regenerate themselves in uniform age cohorts that can be recreated using even-aged silvicultural methods. Missouri's variable topography coupled with cyclic severe droughts likely created a landscape of variable even and

uneven-aged natural communities. Fire scar studies across Missouri showed even-age cohorts for post oak and some shortleaf pine stands likely resulting from more intense or mixed severity fires that occurred during 10-20 year drought cycles (Guyette 2007). The standards and guidelines in MP 1.1 and 1.2 allow the use of even-aged silvicultural methods to create forest openings up to 500 acres in size on the Houston/Rolla and Ava/Cassville/Willow Springs districts, primarily because the interpretation of general land office survey notes suggest patterns of larger historical barrens. Accordingly, most regeneration in savannas and woodlands is projected to be obtained using even-aged silvicultural prescriptions. Clearcutting, with scattered relict tree and reserve tree retention, will be used in natural community restoration, and planting to restore shortleaf pine will be required in many areas. Most of the uneven-aged silvicultural prescriptions employed will be practiced in upland forest natural communities.

SUMMARY AND CONCLUSIONS

In the restoration of natural communities, silvicultural prescriptions are intended to move stands from their existing condition toward a desired future condition that better achieves important ecological objectives. The silvicultural techniques practiced are similar to those routinely used in traditional forest management. However, rather than maximizing production of outputs such as timber products or specific wildlife habitats, the emphasis is on regulating structure and composition, and emulating historic disturbance process that create diverse, healthy, natural communities. Moving towards the desired condition may take 15 to 25 years for ground cover and more than 100 years on a landscape scale for the composition and structure of respective canopy characteristics. The ecosystem management approach should create diverse, productive, and healthy ecosystems, and perhaps capture 80% of Ozark biodiversity, including most species of conservation concern. Production of timber and other goods and services may be a by-product of management activities, but should not be considered an end result.

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Integrating Land and Resource Management Plans and Applied Large-Scale Research on Two National Forests

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ABSTRACT

Researchers working out of the Southern and Northern Research Stations have partnered with two National Forests to conduct two large-scale studies designed to assess the effectiveness of silvicultural techniques used to restore and maintain upland oak (*Quercus* spp.)-dominated ecosystems in the Cumberland Plateau Region of the southeastern United States. We based both large-scale studies on approved Land and Resource Management Plans (LRMP), but the projects being studied have been implemented under separate authorities. On the Bankhead National Forest, the District Ranger and staff have worked with a formal forest liaison board to gain interest and acceptance of the new Forest Health and Restoration Project to be implemented under traditional Forest Service authorities. The impetus for the Daniel Boone National Forest study was also consistent with the LRMP developed in collaboration with interested publics, but the study projects were implemented using authorities granted by the Healthy Forest Restoration Act

of 2003. In both studies, researchers are assessing response to silvicultural techniques used to control species composition and alter habitat for wildlife and restoration purposes, including intermediate treatments (e.g., thinning and prescribed burning) and regeneration harvests (e.g., shelterwood). Forest managers and researchers are interested in modeling changes in overstory cover and composition, fuel loading, residual tree health and vigor, available light, and understory regeneration. In addition, the response of the herbaceous component, the response of the avian and herpetological populations, and the response of habitats will be assessed using a multidisciplinary approach. We will use results from these studies to help forest managers monitor and predict forest response and to develop habitat and regeneration models for each forest system. We discuss the logistics, challenges, and triumphs of implementing such large-scale projects, and joint efforts for science delivery.

Keywords: silviculture, oak regeneration, national forests, fire

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INTRODUCTION

Increased human population densities, loss of the American chestnut (*Castanea dentata* L.), and a variety of land-use changes have caused current forest ecosystem composition and structure in the southeastern United States to vary widely from that thought to exist in pre-Columbian times (Levitt 2002). Attempts to restore ecosystems in the region to a former state or to manage these ecosystems in their current state will require a thorough understanding of how various environmental factors affect ecosystem characteristics.

Oak-dominated forests are the most abundant cover types in the Southern Appalachian Highlands hardwood region (Johnson et al. 2002). Silvicultural practices to regenerate hardwoods have been the subject of much research (Loftis and McGee 1993; Johnson et al. 2002; Hannah 1987; Roach and Gingrich 1968; Sander 1977, 1988). The clearcutting method, variations of the shelterwood method, and uneven-aged methods can be applied to regenerate upland hardwood stands in both research and operational settings, but these approaches have not been uniformly successful in restoring native oak species, particularly on more mesic sites (Beck 1988).

Prescribed burning and thinning are intermediate treatments that when applied in oak-dominated stands can improve forest health, increase the abundance and size of oak regeneration, and control species composition. A more site-specific method to increase oak regeneration potential is the use of herbicides to alter the light environment in the understory to enhance the survival and growth of oak seedlings (Loftis 1990). However, prescriptions for herbicide use were developed specifically for use on sites of higher than average productivity, and herbicide use on National Forests is often administratively restricted. Generally, a combination of various techniques (herbicide treatments, prescribed fire, thinning, and shelterwoods) applied at intervals (depending on vegetation response and site characteristics) may have the most profound effect on community structure and composition (Franklin et al. 2003; Loewenstein and Davidson 2002; Lorimer 1992; Nowak et al. 2002). However, the interactions of these techniques are largely unknown.

The main problems limiting the adoption and use of silvicultural techniques by National Forest System (NFS) managers are administrative constraints, social influences on management decisions, and imperfect transfer of the knowledge from researchers to forest managers. Researchers can assist NFS managers by conducting collaborative research with them and by sharing up-to-date technology. If researchers and NFS managers jointly test the effectiveness of new or modified silvicultural techniques on National Forest lands, it will likely enhance the ability of NFS managers to predict the outcomes of management practices and efficiently meet their forest management objectives.

We implemented two large-scale studies on the Daniel Boone and William B. Bankhead National Forest in the Cumberland Plateau Region of Kentucky and Alabama. The overall goal of the studies was to determine the effects of various silvicultural techniques on restoring or maintaining the oak component of the forest. The studies are still ongoing, but in the process of establishing the studies we learned many practical lessons about conducting large-scale, long-term research projects on National Forest lands.

The first study we discuss was implemented on the William B. Bankhead National Forest (BNF) in north-central Alabama. Decline and death of southern yellow pines due to the southern pine beetle (*Dendroctonus frontalis* Zimmermann) has provided impetus for this study, which examines the effectiveness of the BNF's current management techniques, as detailed in their latest Forest Health and Restoration Project (FHRP) (USDA Forest Service 2003), in restoring the oak component. The second study we discuss was implemented on the Daniel Boone National Forest (DBNF) in south central Kentucky. The forest may soon be affected by gypsy moth and the often-related disease complex, oak decline, and these impacts would reduce the ability of managers to provide expected benefits derived from the forest. This study was implemented under the auspices of the Healthy Forest Restoration Act of 2003 (HFRA) and was designed to examine the effectiveness of various silvicultural techniques in restoring and maintaining the oak component and improving forest health conditions on the DBNF. The use of these silvicultural techniques is described in the DBNF's LRMP (USDA Forest Service 2004).

The purpose of this paper is to discuss the research goals and general experimental design of each study, the challenges and successes of study implementation, and lessons learned through participation in cooperative research projects of the NFS and Research branches of the Forest Service. We hope to provide general guidelines for other large-scale replicated research projects implemented on NFS lands.

METHODS

Study Areas

William B. Bankhead National Forest

The BNF, established by proclamation in 1914, has a long history of repeated logging and of soil erosion caused by poor farming practices during the Depression era. The 180,000-acre BNF is in the Strongly Dissected Plateau sub-region of the Southern Cumberland Plateau, within the southern Appalachian Highlands (Smalley 1982). In 1975, the 12,000-acre Sipsey Wilderness Area in the north-western portion of the forest was established to preserve a natural forest environment. Today the Sipsey Wilderness totals 25,202 acres. The forest contains the Black Warrior Wildlife Management Area and the Sipsey Wild and Scenic River. Base age 50 site indices for loblolly pine (*Pinus taeda* L.), red oaks (*Quercus rubra* L., *Q. velutina* Lamarck, *Q. coccinea* Muench., *Q. falcata* Michx.), and white oaks (*Q. alba* L., *Q. prinus* L.) are 75 ft, 65 ft, and 65 ft, respectively (Smalley 1982).

Daniel Boone National Forest

Two separate tracts were merged and proclaimed as the DBNF in 1966. The area proclaimed as the Cumberland Na-

tional Forest in 1937, a narrow strip along the western edge of the Cumberland Plateau, contains the study area. The rugged eastern area is maturely dissected and characterized by narrow, winding ridgetops, steep sideslopes, and narrow winding valleys (Smalley 1986). Within the forest are 18,000 acres of designated Wilderness and 19 miles of Wild and Scenic Rivers. Base age 50 site indices for shortleaf pine (*Pinus echinata* Mill.), red oaks (*Quercus rubra* L., *Q. velutina* Lamarck, *Q. coccinea* Muench.) and white oaks (*Q. alba* L., *Q. prinus* L.) are 60 ft, 55 ft and 60 ft, respectively (Smalley 1986).

The study was implemented on the Cold Hill Area of the London Ranger District of the DBNF. The treatment units are located on the Central Escarpment (221 Hb) landtype association, which is transitional between the Highland Rim and the Cumberland Plateau (Taylor et al. 1997; USDA Forest Service 2004).

Study Design

William B. Bankhead National Forest

The BNF study employs a randomized complete block design with a 3 by 3 factorial treatment arrangement and four replications of each treatment. The treatments are three residual basal area treatments (50 ft²a⁻¹, 75 ft²a⁻¹, and an untreated control) with three fire frequencies (frequent burns every 3-5 years, infrequent burns every 8-10 years, and an unburned control; Table 1). Each treatment is replicated 4 times, for a total of 36 treatment units. Treatments are representative of management practices described in the BNF's FHRP for restoring oak forests and woodlands.

Table 1--Disturbance treatments for silviculture research on the Bankhead National Forest, Alabama

	DISTURBANCE TREATMENTS
1	No Burn and No Stand Density Reduction
2	No Burn and 75 ft ² a ⁻¹ Residual Stand Density
3	No Burn and 50 ft ² a ⁻¹ Residual Stand Density
4	Infrequent Burn (8-10 yrs) and No Stand Density Reduction
5	Infrequent Burn (8-10 yrs) and 75 ft ² a ⁻¹ Residual Stand Density
6	Infrequent Burn (8-10 yrs) and 50 ft ² a ⁻¹ Residual Stand Density
7	Frequent Burn (3-5 yrs) and No Stand Density Reduction
8	Frequent Burn (3-5 yrs) and 75 ft ² a ⁻¹ Residual Stand Density
9	Frequent Burn (3-5 yrs) and 50 ft ² a ⁻¹ Residual Stand Density

Criteria for stand selection were based on species composition, stand size, and stand age. Treatment units for the study were located on upland sites currently supporting 20 to 35-year-old loblolly pine plantations with a significant hardwood component in the understory. Treatment units were at least 22 acres in size with basal areas ranging from 80 to 140 ft²a⁻¹.

Prescribed burning was conducted during the dormant season (December-February) using backing fires and strip headfires to ensure that only surface fire occurred. Commercial thinning was conducted by marking from below smaller trees or trees that appeared diseased or damaged. Hardwoods were preferentially retained. Stand density reduction treatments were completed prior to the initiation of the burning treatments (thinning conducted from June through November).

Daniel Boone National Forest

The study design for the DBNF study was a randomized complete block design with a 2 by 5 factorial treatment arrangement with two site types (dry-mesic and dry-xeric) and five disturbance treatments (shelterwood with reserves, oak shelterwood, B-line thinning, oak woodland, and a control; Table 2). The treatments were replicated three times, for a total of 30 treatment units.

Criteria for stand selection and treatment assignment were based on several factors including administrative

constraints, proximity to road infrastructure, stand history, ownership boundaries, topography, soil type, and species composition. All treatment units were located on broad ridges, were dominated by oak species, had basal areas ranging from 70 to 150 ft²a⁻¹, and were occupied by stands between 70 and 150 years old.

The shelterwood with reserves treatment (1) will leave trees that will promote good forest health conditions and improve habitat for wildlife and plant species that benefit from open, low basal area forest conditions. In the oak shelterwood treatment (2), triclopyr ester will be applied as a thinline basal bark treatment to trees less than 3 inches dbh, of undesirable species. Trees greater than 3 inches dbh in the mid- and understories will be treated with stem injection. Undesirable tree species include red maple, which has little benefit to wildlife, and trees with unhealthy stems and/or crowns. Thinning to B-line (3), based on the Gingrich (1967) stocking chart, will reduce tree density and allow residual trees to take advantage of improved growing conditions. The oak woodland treatment (4) will be conducted by first thinning to 30-50 ft²a⁻¹ and then conducting prescribed burning every 3-5 years. White oaks will be favored as residual trees to increase hard mast production. Enhancement of spatial and vertical heterogeneity will be an objective of the treatment. The final treatment (5), a control, will not receive a silvicultural treatment, and will be used as a basis of comparisons and evaluations.

Table 2—Disturbance treatments for silviculture research on the Daniel Boone National Forest, Kentucky

DISTURBANCE TREATMENTS (X= dry-xeric; M= dry-mesic)	
1	X- No Burn and No Stand Density Reduction
2	X- Shelterwood with Reserves (10-15 ft ² a ⁻¹) Residual Stand Density
3	X- Oak Shelterwood (60-75 ft ² a ⁻¹) Residual Stand Density
4	X- B-Line Thinning following Gingrich's Stocking Chart
5	X- Oak Woodland (30-50 ft ² a ⁻¹) Residual Stand Density and Frequent Burn (3-5 yrs)
6	M- No Burn and No Stand Density Reduction
7	M- Shelterwood with Reserves (10-15 ft ² a ⁻¹) Residual Stand Density
8	M- Oak Shelterwood (60-75 ft ² a ⁻¹) Residual Stand Density
9	M- B-Line Thinning following Gingrich's Stocking Chart
10	M-Oak Woodland (30-50 ft ² a ⁻¹) Residual Stand Density and Frequent Burn (3-5) yrs

IMPLEMENTATION

We established five 0.2-acre vegetation measurement plots (BNF) or twenty 0.1-acre vegetation measurement plots (DBNF) in each treatment unit and measured plots prior to and just after treatment implementation. All plot centers are permanently marked with rebar, flagging, and GPS coordinates. We permanently tagged all trees 1.5 inches and greater diameter at breast height (dbh) with aluminum tags. We measured and recorded tree species, dbh, crown condition, tree grade, canopy cover, and tree height. In each plot, we also created a 0.01-acre plot where we enumerated regeneration (trees < 1.5 inches dbh) by species and height class. We tagged 5 representative seedlings per regeneration plot and recorded species and measured height and basal diameter (immediately above the root collar). In the BNF study, we sampled fuel loading using line transects and employed electronic recording devices and temperature sensitive paints to quantify fire behavior during burns. We revisit plots once a year near the end of the growing season to document recruitment and tree growth.

Stand selection and data collection for the BNF began in the summer of 2004, and to date, three of the four replications of thinning and burning treatments have been implemented. The final replication will be completed by the winter of 2008. Stand selection and data collection for the DBNF began in the summer of 2005 and treatments should be implemented by summer of 2008.

DISCUSSION

The goal of both studies is to increase the knowledge of silvicultural prescriptions used by forest managers to create and sustain desired habitat conditions. These two studies represent unique opportunities not only to increase the knowledge base, but also to increase collaboration and trust between the two branches of the agency. Researchers learn about how NFS operations such as harvesting and burning are conducted. NFS personnel have personal access to up-to-date research to assist in answering questions and concerns about management strategies. The synergy created by these two projects outweighs the constraints presented.

Although the studies were only recently initiated, we have already gained a great deal of practical experience about

coordinating large, long-term research projects between the research and land management branches of the Forest Service. First, we learned it can be done successfully- at least through study establishment in our case. We also learned that it isn't always easy, and sometimes things that seem obvious to one group are not at all obvious to others.

Both the NFS and Research branch personnel need to be aware of and prepared for the long time commitment needed to install silvicultural research studies implemented at the stand level. We quickly learned that installation of large-scale, replicated studies requires a significant amount of coordination and communication between NFS managers and researchers. Having designated leaders from both branches facilitates this coordination and communication. The land management plans are an excellent reference for researchers as they develop study plans. NFS must provide the initial leadership in locating potential study stands and in providing the researchers with appropriate information, such as stand history, stand characteristics, location, and any special or limiting circumstances. In our case the NFS managers are solely responsible for all the coordination and implementation of the harvesting and burning. The DBNF personnel were trained by researchers to assist with data collection, and the research work units provided the Forest funding to support these activities. All data on the BNF were collected by the research teams.

Site selection is one of the most important phases of any large-scale research project. The easiest and best procedure for choosing appropriate stands is to have a NFS representative who is intimately knowledgeable with the Forest vegetation and management history work directly with the researchers. We found that the Continuous Inventory of Stand Conditions (CISC) database maintained by each National Forest was not adequate to guide decisions on study site selection. The CISC database was either in a format not conducive to use for analysis, or it lacked information researchers needed. Consequently, substantial amounts of time were spent conducting field reconnaissance and gathering new data for potential study sites. A large time commitment and effort by all parties should be expected and detailed at the inception of the study.

Clear lines of communication are essential. Data collection for these two studies was intensive and expensive, and communication about timing of treatment implementation

is important. Researchers should know when treatments are going to be implemented so they can properly plan collection of pre-and post-treatment data. Researchers should communicate with managers the cost and importance of the data being collected so all can feel a sense of ownership and responsibility. We found it helpful to hold meetings once every 3 to 6 months to share findings to date and discuss logistics of treatment implementation and data collection.

Researchers and NFS staff often struggled with basic communication because the two groups used certain terms differently and because areas were not always referred to by names that were understood by all parties. For example, the researchers used “stand” to refer to the treatment units; the NFS staff referred to these same areas as “units” within compartments. We developed a glossary that defined terms and outlined the names of places within the study areas, including the ‘local’ names of the prescribed burn units. It is imperative that all involved practice patience and take care to explain terms.

Other complicating factors were the administrative and social constraints on management in National Forests that led to non-random assignment of treatments to selected stands. For example, it was postulated that a certain threatened plant existed in several of the study stands after stands had already been randomly assigned to a treatment. Consequently, the study design was altered post-hoc so the threatened plant would not be disturbed. That had the potential to violate underlying statistical assumptions about randomization of experimental treatment. If the possibility that this plant was present was explained prior to stand selection, some stands could have been removed from consideration for the study. However, the presence of the plant represented a fortuitous chance to study a threatened species, and the design could have been altered to test what habitat conditions the plant required. Other constraints in treatment assignment to stands included proximity to roads and houses for treatments that included commercial harvesting and prescribed burning. Researchers should be aware that treatment assignments to stands available for research on NFS lands may not be totally random, and the potential impact of that must be addressed or reconciled prior to study installation.

Communication between researchers and NFS personnel on potential timing of prescribed burns is crucial, particularly when data must be collected during the burn. The NFS personnel should provide the researchers with information on ideal burning conditions and the researchers should be “on-call” to collect burn data with less than 24 hours notice. Research personnel should obtain proper permission to allow their staff to work before or after regular work duty hours to facilitate collection of data about prescribed burns. Another constraint to data collection was that research personnel were not allowed on burn units without proper safety training and testing. It would behoove researchers who may be actively engaged in fire research to obtain all necessary training, certificates, and equipment needed to work on prescribed burning projects.

Faculty and students from Alabama A&M University are conducting additional research on the BNF plots. This research is in the areas of avian and herpetofaunal ecology, herbaceous characterizations, remotely sensed data analysis, and harvesting operations. The DBNF study also includes additional research conducted by collaborators, including faculty and students from The University of Kentucky, Eastern Kentucky University, and The University of Tennessee. These additional research projects will examine bat ecology, evaluate microclimatic attributes, and establish stand disturbance histories using dendrochronology. Collaborative research greatly benefits all involved parties, but can also greatly complicate design and implementation. We found that we needed to give extra attention to the collaborative partners to ensure that they were cognizant of on-the-ground procedures, such as road and gate usage and safety regulations.

SCIENCE DELIVERY

We found that the science delivery process begins immediately. We have already conducted field tours on both Forests. These tours provide a terrific opportunity for the NFS and Research teams to highlight their partnership, and the breadth and depth of information provided by both teams is impressive. Presentations developed by the research teams have been provided to NFS personnel for additional science delivery. Results are being presented at research conferences and at public and private meetings of lay audiences. Peer-reviewed publications will highlight research results and acknowledge the effort committed by both branches of the agency.

CONCLUSIONS AND RECOMMENDATIONS

We recommend that the following measures be taken when large-scale silvicultural research projects at the stand level are to be implemented on NFS land:

1. Hold frequent meetings and conference calls between researchers and NFS managers, particularly at the beginning of study implementation. Researchers should explain the timing and costs of planned data collection. Managers should explain logistics problems and arrangements, and constraints on implementation of proposed treatments. Take detailed notes and distribute them widely.
2. Both researchers and NFS managers should be clear at the outset about the time required to implement proposed projects. Because these types of studies are generally long-term, the NFS personnel should be prepared to protect and maintain study sites for at least 10 years and to clearly document that necessity through the inevitable personnel changes that will occur over the life of the study.
3. Designate a project leader and alternative from NSF and research. The NFS leader should be knowledgeable about Forest tree species, history, and administrative and social constraints that could affect project implementation. The research leader should be knowledgeable about experimental design, timing and costs of data collection.
4. Researchers should use the LRMP as a starting point in developing study plans. This will ensure that proposed treatments are consistent with the goals and objectives of the Land Management Plan.
5. Clarify terms used frequently, such as stand, unit, and plot. Create a comprehensive study plan with a project glossary that clearly defines both research and NFS terminology used to describe areas in study.
6. Research personnel should obtain required safety training, certificates, and equipment if they are to be on prescribed burn units during burning.
7. Research personnel should obtain proper permission to allow staff to work before or after regular duty hours when necessary (e.g. for collection of prescribed burning data).

8. Researchers should be aware that stands available for treatment may be limited and that assignment of treatments to stands may not be random, which can bias research results. Thus, they may need to revise the study or interpretation of the results accordingly.

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The Potential Role of Red Alder to Increase Structural and Biological Complexity in Even-Aged Hemlock-Spruce Stands of Southeast Alaska

Robert L. Deal and James M. Russell

ABSTRACT

Stand dynamics were evaluated in mixed red alder-conifer forests of southeast Alaska by assessing stand development, tree regeneration, total basal area, and understory biomass in 40-year-old red alder-conifer stands that developed following logging. Overall, these mixed red alder-conifer stands provided more heterogeneous structures, multiple canopy layers, and greater diversity and abundance of understory plants than typically found in similar-aged conifer stands. However, the success of alder plantations and planned silvicultural systems that include mixed alder-conifer forests in southeast Alaska are unknown. Red alder has never been planted in the region and its occurrence has been confined to areas associated with landslides, logging and riparian zones. In 2003, an operational-scale treatment that included alder plantations

was replicated at 23 sites within the Tongass National Forest. The alder treatments included a control with no alder planted, a low density planting of 50 tph (trees per hectare) and a high density planting of 200 tph in naturally regenerating 0-5 year old conifer stands. Alder survival of these stands was assessed at 1 and 3 years after planting. Early survival of planted alder seedlings was excellent, but 3-year survival has been much lower and site specific. These plantations will need to be monitored and assessed to determine the long-term potential of mixed alder-conifer stands for improving and restoring forest ecosystems in southeast Alaska. Overall, red alder may serve as an effective tool for increasing biodiversity and improving terrestrial and aquatic habitat when included in regenerating forests following timber harvesting in the region.

Keywords: Stand structure, red alder, Sitka spruce, western hemlock, understory plants, wildlife habitat

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INTRODUCTION

Southeast Alaska is a coastal temperate rainforest region with steeply rising coastal mountains and densely forested islands. These rain forests are dominated by conifer species such as western hemlock (*Tsuga heterophylla* (Raf.) Sarg.) and Sitka spruce (*Picea sitchensis* (Bong.) Carr.), and are characterized by simple species composition with complex age and size structures. Abundant annual precipitation occurs throughout the year along with occasional hurricane force winds. The significance of this climate for the forest is that moisture is generally not a limiting factor for tree regeneration, wildfire is rare, and tree wind-throw is common (Harris, 1989; Nowacki and Kramer, 1998; Kramer et al., 2001). The region's forests contain multi-aged stands created by high-frequency, small-scale natural disturbances such as wind-throw, landslides and endemic tree disease (Deal et al., 1991; Kramer et al., 2001; Hennon and McClellan, 2003). These multi-aged stands contain complex forest structures with many forest canopy layers, abundant understory vegetation, large trees, snags, woody debris, and other important ecological characteristics of old-growth forests (Franklin et al., 1981; Alaback and Juday, 1989; Franklin and Spies, 1991). The understory vegetation is also important as food for herbivores and as cover for ground foraging and nesting birds and small mammals.

Silvicultural systems using clearcutting and natural regeneration have been the dominant timber management practice in southeast Alaska forests since the 1950s (Harris and Farr, 1974). Post-harvest conifer regeneration is frequently abundant (>10,000 trees/ha) with the development of a dense new cohort of western hemlock and Sitka spruce trees. The forest canopy closes in 20-30 years followed by a dense, long-lasting stage of stem exclusion (Alaback, 1982; Deal et al., 1991). Canopy closure eliminates most herbs and shrubs (Alaback, 1982) and attempts to reestablish understory plants through thinning dense young-growth stands has led mostly to conifer regeneration (Deal and Farr, 1994) with little success in herbaceous plant colonization (Tappeiner and Alaback, 1989). These dense young-growth stands have relatively uniform tree height and diameter distributions, and notably lack the multi-layered, diverse forest structures and shrub/herb layers found in old-growth stands (Alaback, 1984; Deal, 2001; Hennon and McClellan, 2003).

This long-lasting stage of stem exclusion has significant implications for understory plants and wildlife such as Sitka black-tailed deer (*Odocoileus hemionus sitkensis* Cowan) that depend on these plants as forage (Wallmo and Schoen, 1980; Schoen et al., 1988; Hanley, 1993). For the first 15 to 20 years after clearcutting, these young-growth stands provide greater understory plant biomass than old-growth stands (Alaback, 1982); however, snow accumulation makes them much less useful for deer habitat in the winter (Kirchhoff and Schoen, 1987; Schoen and Kirchhoff, 1990). Following canopy closure, the resulting dense stands are recognized as having broadly negative consequences for wildlife habitat (Wallmo and Schoen, 1980; Hanley, 1993; Dellasalla et al., 1996; Schoen et al., 1988). Therefore, there is increasing interest in developing forest management practices that maintain or enhance biodiversity and assure long-term sustainability of wildlife and other forest resources.

Red alder (*Alnus rubra* Bong.) is the most common hardwood tree in southeast Alaska, and is frequently found along beaches and streams, on avalanche tracks, landslides and logging skid trails, and as a pioneer species with Sitka spruce (Harris and Farr 1974). In Alaska, pure stands of red alder are rare and alder occurs primarily in mixed stands with shade tolerant conifers including Sitka spruce, western hemlock and western red cedar (*Thuja plicata* Donn ex D. Don). Recent studies of mixed alder-conifer stands in southeast Alaska indicate different successional pathways following clearcutting than the development patterns of pure conifer stands. These mixed alder-conifer stands generally appear to have lower tree stocking and stand density than pure conifer stands of a similar age (40-50 years), with more open forest canopies and more heterogeneous stand structures (Deal, 1997; Deal et al., 2004). Other studies in mixed alder-conifer stands have reported increases in plant species richness and highly productive understory vegetation with biomass similar to that of old-growth stands of the region (Hanley and Hoel, 1996; Hanley and Barnard, 1998; Hanley et al., 2006). Habitat quality for small mammals in even-aged alder-conifer stands may be equal to that of old-growth forests (Hanley, 1996; Hanley and Barnard, 1999). Red alder may convey additional benefits in riparian forests. Riparian forests with some red alder appear to produce more prey biomass and food for fishes than conifer riparian forests (Wipfli, 1997; Piccolo and Wipfli, 2002). If similar processes occur in upland

forests, the presence of red alder may increase invertebrate production, providing more food for animals such as birds, small mammals, and fish.

The objective of this paper is to synthesize current information on management strategies for mixed red alder-conifer forests. First, we summarize results from a large recent study of mixed alder-conifer stands in the region, and evaluate the relationship of alder with stand structure, understory plant diversity and biomass, and the potential of alder in conifer-dominated forests for improving wildlife habitat. Second, we assess survival of red alder in young plantations and discuss some management options for increasing stand structural diversity and enhancing biodiversity in young forests in southeast Alaska.

METHODS

Mixed red alder-conifer stands

To examine the relationship between red alder composition and stand structural characteristics, we assessed several mixed alder conifer young-growth stands that naturally regenerated following harvesting. This study was conducted in alluvial valleys in the Maybeso and Harris watersheds, on Prince of Wales Island, Alaska. Nine widely separated stands were selected in the summer of 2000 for intensive sampling. These stands had been clear-cut harvested using cable logging systems about 40 years ago. The amount of alder as a proportion of total stand basal area was the main criterion for stand selection, with an increasing proportion of alder sampled across a compositional gradient ranging from 0 to 86% red alder (Table 1). Stands were selected in upland areas with non-fish bearing headwater streams and site selection included the following criteria:

- a) stands that were logged and naturally regenerated;
- b) average stand size of 5-10 ha;
- c) elevation less than 150 m;
- d) moderate slopes of 10-40 percent;
- e) no intermediate management activities (e.g., no thinning, planting or alder girdling).

A systematic grid of 20 variable-radius plots was installed throughout each stand to determine average stand composition and to ensure that sites provided a wide range of alder-conifer mixtures.

Fixed-area plots were installed at five randomly selected variable-radius plot locations in each stand. A total of 45 fixed-area plots in the nine stands were used to measure stand density, tree size distribution, patterns of tree mortality, and tree height growth patterns. Detailed sampling methodology is described in Deal et al. (2004). The variable radius plots were used to sample understory vegetation in each of the 9 stands (Wipfli et al., 2002). Total above-ground biomass was measured for herbaceous plants and methodology as described by Hanley et al. (2006). Linear regression analyses were used to test the relations between the proportion of red alder basal area and (1) understory biomass and (2) carrying capacity for deer (Hanley et al., 2006).

Red alder plantations

Following the preliminary results of the previously described study on mixed red alder-conifer stands, the Pacific Northwest Research Station and the Tongass National Forest, designed and implemented a series of operational-scale tests of young-growth management options known as the Tongass-Wide Young-Growth Studies (TWYGS). One of the treatments in TWYGS included low density and high density planting of red alder seedlings in 0 to 5 year old stands following harvesting (McClellan, 2007). The 0-5 year age class was chosen because conifer regeneration should be small enough to allow for the successful planting and survival of red alder seedlings. The objective of this alder study was to examine red alder seedling survival in conifer plantations. Also, because this was the first large scale planting of red alder in the region, early survival and growth of red alder seedlings could be assessed to determine establishment success of alder in sites where it may not be common. We assessed first and third year survival of red alder over a large geographic range and related survival with site and soil characteristics to determine alder establishment success.

Twenty three sites were selected from a pool of potential sites and selection criteria included a wide geographic range throughout the Tongass National Forest. Operational size treatments with experimental units was generally at least ten acres, elevation less than 350 m, with no intermediate treatments such as weeding or conifer planting. Experiments used a randomized complete block design, with experimental blocks laid out within a single timber-harvest unit, which was divided into three experimental units. In total, 11 sites were established in the southern

region of the Tongass National Forest (Ketchikan), 9 sites in the central (Stikine), and 3 sites in the northern region (Chatham).

For the red alder plantations, three treatments were installed that included red alder planted at a low density of 50 tph (trees per hectare), a high density of 200 tph, and a control, where no red alder were planted. Complete sets of treatments were replicated at 23 sites. The red alder planting stock was grown from seed collected in south-east Alaska and contractors planted the trees in recently harvested areas (0 to 5 years old) during the spring of 2003. Survival was assessed from stake-row survival of 100 seedlings in each treatment. Survival was assessed in

2004 and 2006 or one year and three years after seedlings were planted. Elevation, aspect, slope, site index and soil information was determined in the office using Geographic Information System (GIS) map data. Data were analyzed using SAS to determine relationships among first and third year red alder survival for the low and high density planting treatments, and for site class, elevation, slope, aspect and soil drainage. Differences among survival of treatment classes and first and third year survival were determined using Duncan class analyses.

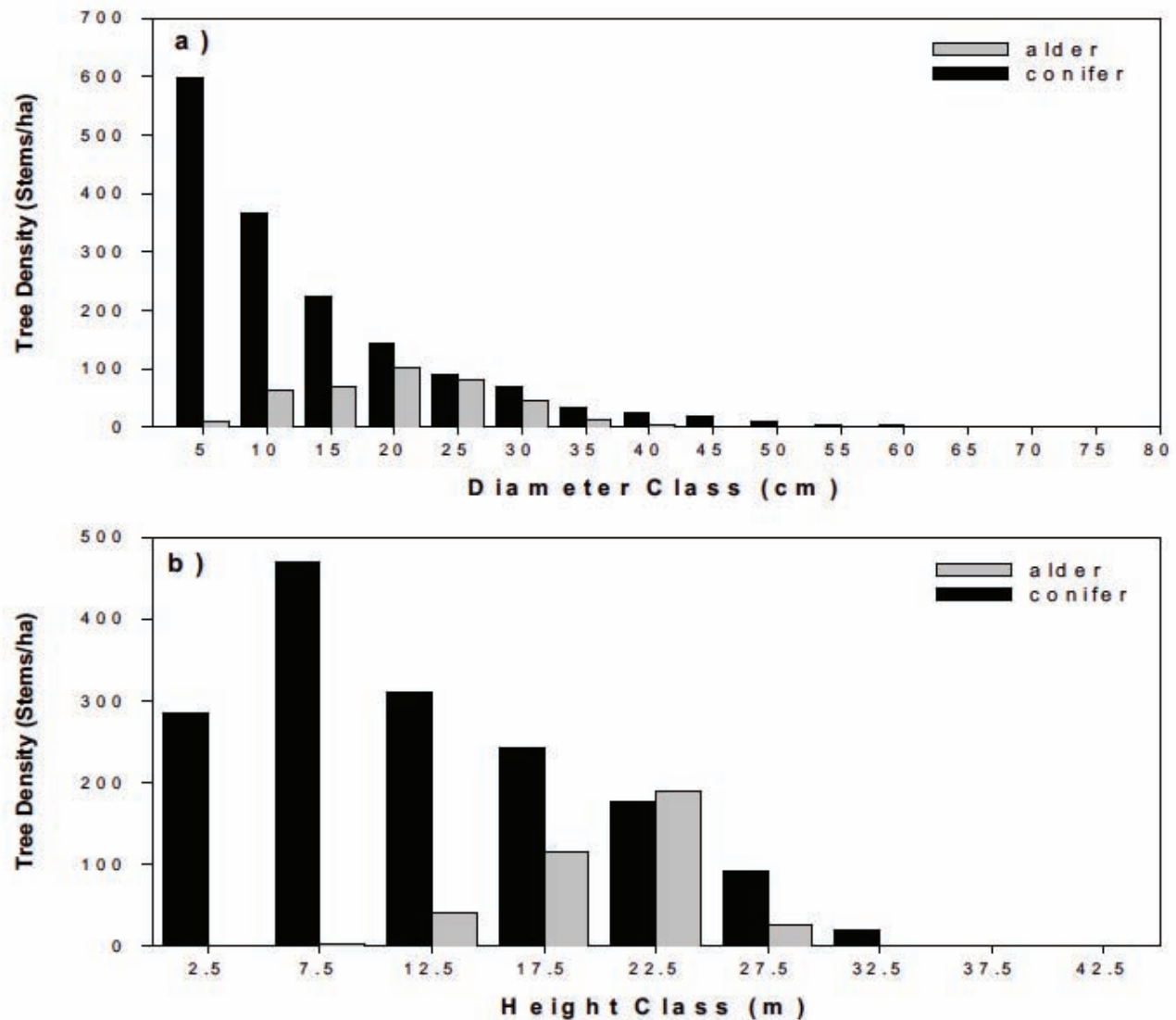


Figure 1— Average tree diameter distribution (a) and average tree height distribution (b) for alder and conifer trees at all nine research sites (modified from Deal et al., 2004).

RESULTS

Density and tree size of mixed red alder-conifer stands

Diameter distributions of trees differed between conifers and alders. Conifers had many more trees in the smallest size classes with decreasing numbers of trees in progressively larger sizes following a reverse J-shaped diameter distribution (Fig. 1a). In contrast, the diameter distribution of alders were bell shaped with most trees in a narrow diameter (20-30 cm) and overstory height (20-25 m) range (Fig. 1a & 1b). The conifers were unevenly distributed with a few taller and larger-diameter trees and several hundred trees in the smallest diameter classes. The mixed alder-conifer stands had more complex diameter distributions than pure or conifer-dominated stands. The alder in these stands provided a different tree size cohort than the conifer-dominated stands that contained numerous small diameter trees. These mixed alder-conifer stands created a multi-layered forest canopy with a few dominant overstory conifers, a mid-canopy level of red alder and a lower canopy level of small diameter conifers (Fig. 1b).

Tree species composition was related to differences in stand basal area and density of trees (Table 1). Analysis of alder composition among the nine stands showed a significant decrease in total stand basal area with increasing proportion of alder basal area ($p = 0.013$; Deal et al., 2004). Tree density, however, was not closely associated with alder composition; a slight and non-significant decline in density occurred with increasing proportion of alder basal area (Table 1; $p = 0.298$). While alder diameters were relatively similar among all of the sites, the average diameter of all

conifers significantly decreased with increasing proportion of alder basal area ($p = 0.029$). There was also no significant decrease in size of the largest trees with increasing proportion of alder in the stand; the largest diameter conifers appeared to be relatively independent of alder composition with large diameter trees found across a wide range of alder-conifer mixtures. These large conifers could provide an important structural component for riparian zone management to produce large wood for in-stream habitat.

Plant diversity and abundance of mixed red alder-conifer stands

The species richness of these mixed alder-conifer stands was high with 112 species of understory plants including 49 herbs and 15 shrubs. The lowest species richness was in the pure conifer stand; the greatest species rich stands were those stands with more even mixtures of alders and conifers. Understory canopy coverage appeared to be closely related to the amount of alder in the stand, and understory plant cover increased with increasing proportion of alder basal area. Analysis of cover for vascular plants and non-vascular plants (bryophytes) showed different relationships. Non-vascular plant cover was similar for all of the sites, varying from about 35-45%. There was no significant relationship ($r^2 = 0.088$, $p = 0.291$) between the cover of non-vascular plants and the proportion of alder basal area (Fig. 2). In contrast, a strong correlation ($r^2 = 0.701$, $p = 0.005$) occurred between increasing vascular plant cover and increasing proportion of alder basal area (Fig. 2).

Both shrub and herbaceous biomass increased significantly with increasing percentage red alder (Hanley et al., 2006). Total aboveground biomass of understory vegetation

Table 1. Stand density, species composition and soil characteristics of nine sites listed with an increasing proportion of red alder basal area.

Site	Time Since Cutting (years)	Basal Area		Species Composition				Soil Characteristics		
		Stand (m ² ha ⁻¹)	Alder (% BA)	Alder (trees ha ⁻¹)	Spruce (trees ha ⁻¹)	Hemlock (trees ha ⁻¹)	Other (trees ha ⁻¹)	Geomorphic Position	Geomorphic Deposit	Soil Type
Upper Good Example	42	59.6	0	0	493	819	0	Mountain Flank	Colluvial/Alluvial	Karta/Tolstoi
Cedar 1&2	40	50.6	3	24	705	2533	488	Mountain Flank	Colluvial	Tolstoi/Rock outcrop
Lower Morning	42	48.0	16	348	601	753	468	Mountain Flank	Colluvial/Alluvial	Tolstoi
Lower Good Example	42	60.8	18	451	843	689	27	Mountain Base	Colluvial/Alluvial	Tolstoi/Karta
Mile 22	40	54.4	28	392	265	1415	0	Mountain Flank	Alluvial	Tolstoi/Karta
Big Spruce	40	47.8	33	291	303	672	57	Mountain Flank	Colluvial/Alluvial	Tolstoi/Rock outcrop
Group Photo	38	45.5	39	560	519	801	0	Mountain Base	Alluvial	Tolstoi/Karta
Alluvial Fan	42	46.0	64	680	645	333	27	Mountain Base	Alluvial	Tolstoi/Karta
Brushy	40	38.0	86	708	275	389	240	Mountain Flank	Colluvial/Alluvial	Tolstoi/Karta

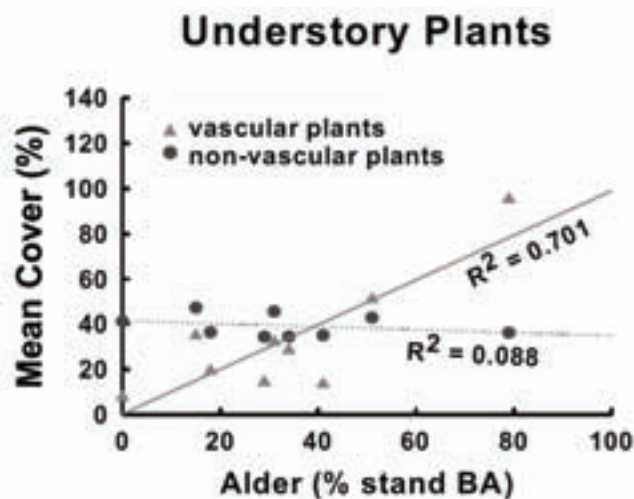


Figure 2— The mean canopy cover of vascular and non-vascular plants as a function of the proportion of red alder basal area (BA) for all nine research sites.

ranged from 10 to 616 kg ha⁻¹ across the 9 stands and was highly correlated with percentage red alder basal area in the stand (Fig. 3a, $r^2 = 0.743$, $p < 0.001$). This correlation of herbaceous biomass with increasing percentage red alder

was consistent throughout the range of stands studied. The significant increase in plant biomass during the summer had pronounced positive impacts on the available forage for deer. Carrying capacity for deer in summer increased significantly (Fig. 3b, $r^2 = 0.846$, $p < 0.001$) from 0.05 to 122.18 deer days per hectare with increasing percentage red alder (Hanley et al., 2006). However, there was no significant relation between carrying capacity and percentage red alder in winter (Fig. 3b, $r^2 = 0.246$, $p > 0.10$). This is probably related to loss of available understory biomass during the winter in these younger forests.

Red alder seedling survival in plantations

First year survival of seedlings was high with an average of 97% survival in the low density treatment, 96% survival in the high density treatment and an overall survival of 96% (Table 2). Third year survival declined markedly with an average of 60% survival in the low density treatment and 74% survival in the high density treatment (Table 2). Overall, third year survival averaged only 68% and was significantly lower than first year survival ($p < 0.001$). Third year survival was higher in the high density treatment and

Table 2. Red alder seedling survival at 23 sites by planting density after one year and three years. Site index and soil characteristics were determined from GIS mapping units.

Site	Alder Seedling Survival				Spruce Site Index 50 yr (m)	Soil Characteristics			
	First Year Survival (%)		Third Year Survival (%)			Geomorphic Position	Geomorphic Deposit	Soil Type	Drainage
	50 tpa	200 tpa	50 tpa	200 tpa					
Thorne Bay 3	100	100	90	95	22	Smooth hillside	Till/Coluv./Resid.	McGilvery	Well
Thorne Bay 2	100	100	60	85	24	Smooth hillside	Till/Coluv./Resid.	Tohtoi-McGilvery	Well
Thorne Bay 1	100	100	60	100	30	Smooth hillside	Till/Coluv./Resid.	Karta-Tohtoi	Moderate-well
Thorne Bay 5.	100	100	20	40	23	Rolling hills	Resid./Coluv.	Wadleigh-Kegish	Poor
Gatchi	90	100	65	60	15	Rolling hills	Organic/Till	Maybese-Kaikli	Poor
Naukati	95	95	75	85	27	Rolling hills	Coluv./Resid.	Ullor-Sarkar	Well
Shahen	100	100	65	90	28	Smooth mtslopes	Till	Karta	Moderate-well
Yahuk	95	90	65	90	30	Smooth hillside	Till/Coluv./Resid.	Karta silt loam	Moderate-well
Chin Point 3	100	96	60	72	23	Smooth hillside	Coluv./Resid.	Kupreanof-McGilvery	Well
Chin Point 2	100	90	33	40	26	Smooth mtslopes	Coluv./Organic	Kupreanof-McGilvery	Well
Chin Point 1	100	94	40	38	30	Smooth mtslopes	Coluv./Resid.	Voon-Traitors	Moderate-well
George Islet	100	94	60	60	30	Smooth hillside	Coluv./Resid.	Voon sandy loam	Moderate-well
Newlywed	NA	100	NA	100	16	Smooth hillside	Till	Nakwaina	Poor
Nemo	NA	100	NA	96	23	Smooth mtslopes	Coluv./Resid.	Kupreanof-Mooman	Well
Portage Bay	100	96	57	76	26	Smooth hillside	Coluv./Resid.	Kupreanof-Mooman	Well
Big John	92	84	75	60	26	Smooth hillside	Coluv./Resid.	Kupreanof-Mooman	Well
Duncan Canal	92	96	60	68	30	Smooth hillside	Till/Coluv.	Karta silt loam	Well
Appleton	92	88	55	60	24	Broken mtslopes	Till	Yakobi mucky loam	Poor
Barnof	96	100	70	53	25	Broken mtslopes	Till	Yakobi mucky loam	Poor
Humpback	95	80	75	75	26	Broken mtslopes	Coluvium	Mikof loam	Somewhat-poor
Etolin 2	NA	100	NA	76	25	Smooth mtslopes	Coluv./Till	Mikof-Mooman	Well
Etolin 3	NA	100	NA	84	18	Smooth mtslopes	Residuum	St. Nicholas loam	Poor
Etolin 4	NA	100	NA	92	24	Smooth mtslopes	Coluv./Resid.	Kupreanof-Mooman	Well

survival was significantly higher compared with the low density treatment ($p = 0.019$).

Site index, aspect, elevation and slope were not strongly related with alder seedling survival for any of the treatments and none of these variables were significantly correlated with seedling survival. Soil drainage appeared to be related to third year survival and some of the poorly drained sites had low survival (Thorne Bay S. and Baranof, Table 1). However, some of the well drained sites also had low survival (Chin 1 and Chin 2) as well as very high survival (Thorne Bay 3 and Nemo). Overall, we found no significant relationship between soil characteristics and seedling survival. However, it is important to note that soil types were determined from GIS mapping units and not confirmed in the field, and it is possible that soil type of planting areas were incorrectly identified. Information on elevation, aspect and slope were probably accurate but soil characteristics, soil type and drainage were less certain and need to be confirmed in the field before making any conclusions about soils and seedling survival.

DISCUSSION

Role of alder for increasing stand diversity and understory plant abundance

In younger forests, mixed alder-conifer stands provided different tree size distributions and more complex forest structures than are typically found in pure conifer stands of the same age. Pure conifer young-growth stands in this coastal region are typically very densely stocked (DeMars, 2000) and have relatively uniform tree height and diameter distributions. Our results suggest that red alder can provide different tree size structures with multiple canopy layers in mixtures with conifer stands (Fig. 1). These multiple canopy layers may be the most important structural attribute of mixed alder-conifer stands and may distinguish these stands from predominantly even aged conifer stands of the same age. The presence of alder also did not significantly reduce the size of the largest conifer trees in the stands (Orlikowska et al., 2004; Deal et al., 2004). These conifers will continue to grow in diameter and could provide an important source of large wood for instream fish habitat (Bryant, 1985; Orlikowska et al., 2004).

The mixed alder-conifer younger stands had more complex stand structures than similar aged conifer stands and these structures have important management implications for forest wildlife resources. Researchers have reported broadly negative effects of clearcut logging on wildlife habitat in southeast Alaska (Schoen et al., 1988; Wallmo and Schoen, 1980; Hanley, 1993). The principal problem is that dense conifer regeneration and canopy closure result in an understory with few herbs, few shrubs and little forage for as long as 150 years after initial canopy closure (Alaback, 1982, 1984; Tappeiner and Alaback, 1989). An important finding of this study is the reduced number of small diameter conifers found in the more evenly mixed alder-conifer stands (Deal et al., 2004). The lower stocking of these small diameter trees may allow more light to reach

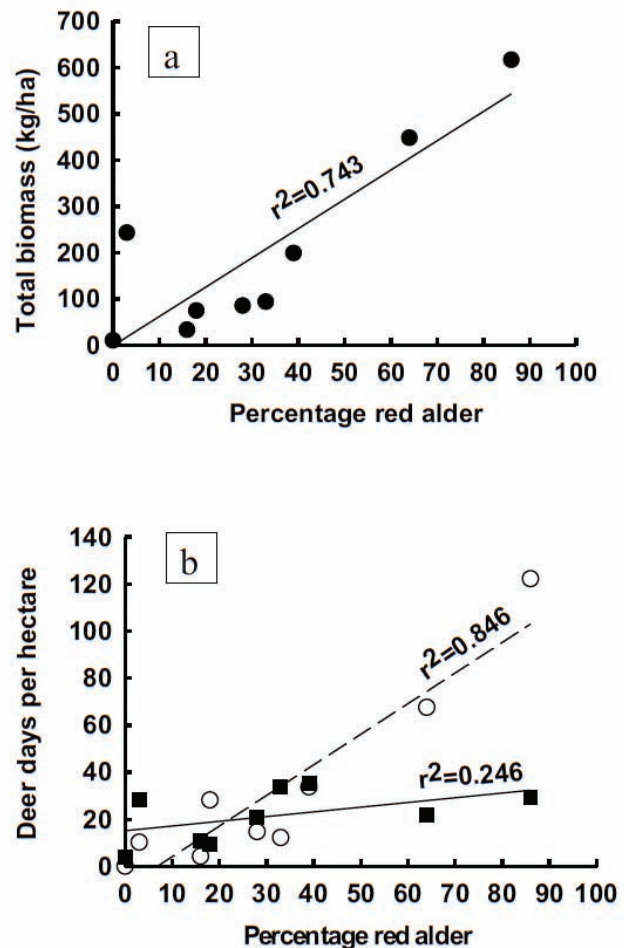


Fig. 3. Total aboveground understory biomass (a) and deer days per hectare (b) of food resources for black-tailed deer during summer (circles) and winter (squares) as a function of the proportion of red alder basal area for the nine research sites (modified from Hanley et al., 2006).

the forest floor that could help maintain understory plants. Recent studies have documented a correlation of herbaceous biomass with increasing percentage red alder throughout the range of stands studied (Hanley et al., 2006). This is important from both a biodiversity and a forest management perspective, because the herb component is difficult to maintain through secondary succession of even-aged stands following clearcutting in southeast Alaska (Alaback, 1982; Hanley, 1993), including precommercial thinning (Doerr and Sandburg, 1986; Deal and Farr, 1994). Summer food value and deer carrying capacity increased with increasing red alder because the quantity of both forbs and shrubs increased with increasing red alder (Fig. 3), however, in winter, most of these red alder-associated species are either senescent (forbs and ferns) or of very low nutritional quality (Hanley and McKendrick, 1983). Therefore, while mixed red alder-conifer stands might improve summer habitat for deer significantly, its potential benefits for winter habitat are somewhat limited.

The effects of mixed red alder-conifer stands for other wildlife species may differ from those for deer. Wildlife-habitat consequences of red alder in young-growth conifer stands differ seasonally and for different wildlife species, depending on the wildlife species' habitat requirements (Hanley et al., 2006). Red alder stands with similar understories to those with the highest percentage red alder stands of this study (Hanley and Hoel, 1996) provided year-around habitat for Keen's mouse that was of equal quality to that of both upland and riparian old-growth forests (Hanley and Barnard, 1999). Other studies conducted in riparian zones and in aquatic ecosystems showed that red alder increased the abundance of food for fish in its immediate vicinity and in downstream reaches (Piccolo and Wipfli, 2002; Wipfli and Musslewhite, 2004). Within red alder stands, vegetation for both invertebrate and vertebrate herbivores was increased by the presence of red alder, an observation consistent with previous work in southeastern Alaska (Hanley, 1996; Hanley and Barnard, 1998; Deal, 1997; Wipfli, 1997; Piccolo and Wipfli, 2002). Streams at the higher end of observed red alder densities delivered about four times more invertebrate biomass than streams canopied predominantly with conifers (Wipfli and Musslewhite, 2004). Also noteworthy was the trophic influence that upland red alder had on more distant habitats. Fish habitats downstream of red alder-dominated reaches were supplied with invertebrates (food for fish) and organic

detritus (food for invertebrates) from upstream headwaters (Wipfli and Gregovich, 2002; Piccolo and Wipfli, 2002) and aquatic food webs downstream of red alder received more prey than those below conifer-dominated reaches (Wipfli and Musslewhite, 2004).

Establishment of red alder plantations

In 2001, a large adaptive management study in the region known as the Tongass Wide Young Growth Study (TWYGS) was implemented to assess the effects of different management practices in young-growth stands throughout southeast Alaska including the planting of red alder (McClellan, 2007). Although there are retrospective studies on red alder-conifer stands (Deal, 1997; Hanley and Barnard, 1998; Deal et al., 2004; Hanley et al., 2006), red alder has never been planted in the region, and information on the growth and survival of red alder will be very useful. Assessment of alder plantations showed that early survival was quite high but third year survival declined to 68% and long-term survival of these alder plantations is a concern. However, these results are not particularly surprising considering that plantation sites were selected without consideration of potential establishment problems for red alder. Red alder can be difficult to grow successfully in plantations under better growing conditions in Oregon and Washington and red alder is generally considered less forgiving than conifers when planted on the wrong sites or with improper silvicultural practices (Dobkowski 2006). Many alder plantations initially failed in the Pacific Northwest and it was not until specific site quality information was developed (Harrington 1986) that alder plantations became successful. Although often considered a weed species, alder has specific growing conditions, and poor drainage, frost, drought stress and exposure can negatively affect establishment success (Dobkowski 2006). Harrington (1986) identified 14 soil and site properties important for predicting site index for red alder including geographic position, topographic position, soil moisture, aeration and soil fertility. It is likely that several of the plantation sites in the TWYGS study are poor alder sites and long-term establishment success is questionable. The highly variable third year survival data for these alder plantations is evidence of some potential problems for alder establishment. An important future need is to determine soil and site properties of these plantations and to identify information for better site selection. The preliminary information on alder establishment success will be useful to identify future sites for alder plantations.

In summary, research indicates that inclusion of red alder in conifer-dominated young-growth stands helps maintain understory plants and provides forage for deer and small mammals. Results also show a clear linkage with improved invertebrate diversity in aquatic systems. Well planned silvicultural systems that include a mixture of red alder-compositions could provide trees for timber production and improve other forest resources that are often compromised in pure-conifer young-growth forests in the region.

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Restoration of Northern Rocky Mountain Moist Forests: Integrating Fuel Treatments From the Site to the Landscape

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ABSTRACT

Restoration and fuel treatments in the moist forests of the northern Rocky Mountains are complex and far different from those applicable to the dry ponderosa pine forests. In the moist forests, clearcuts are the favored method to use for growing early-seral western white pine and western larch. Nevertheless, clearcuts and their associated roads often affect wildlife habitat, water and soil resources, and scenic values. To address this issue, we are applying and quantifying integrative silvicultural methods and systems that are applicable for regenerating and growing these, and other early seral species, while maintaining forest characteristics that are relevant to many contemporary forest management objectives. The silvicultural options we developed maintain multiple tree densities, a variety of canopy cover, and enhance old-forest attributes and most importantly, the harvesting, mastication, grapple piling, and prescribed fire

treatments we applied will modify both wildfire intensity and burn severity. We found that the heterogeneous forest structures we created, even with small openings (average size 2.6 ha) and the minor proportion of the landscape (3 percent) treated, would alter a wildfire's progression, flame length, and fire type, according to FlamMap and FARSITE wildfire simulations. This analysis showed the placement, juxtaposition, and location of treatments within the landscape would disrupt a hypothetical wildfire's progression under weather conditions that occurred during one of the worst fire seasons (1967) in the northern Rocky Mountains. We found that masticating fuels after harvest in these multi-species and highly variable forest conditions was as cost effective as grapple piling the fuels and offered additional benefits.

Keywords: silviculture, uneven-aged management, site preparation, wildfire simulation

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INTRODUCTION

The forests of the Inland West have changed significantly in the last 100 years. Fire exclusion, timber harvest, and the introduction of white pine blister rust (*Cronartium ribicola* Frisch.) have altered forest composition and structure from conditions that occurred historically (pre 1900) (Covington et al. 1994, Jain and Graham 2005, Neuenschwander et al. 1999, Steele et al. 1986). Forests once dominated by shade-intolerant and fire-tolerant species such as western white pine (*Pinus monticola* (Dougl. ex D. Don), western larch (*Larix occidentalis* Nutt.), and ponderosa pine (*Pinus ponderosa* P. & C. Lawson) have been replaced by shade-tolerant and fire-intolerant species such as grand fir (*Abies grandis* (Dougl. ex D. Don) Lindl.), western hemlock (*Tsuga heterophylla* (Raf.) Sarg.), and western redcedar (*Thuja plicata* Donn ex D. Don) (Jain and Graham 2005, Neuenschwander et al. 1999). Likewise, in many areas once dominated by ponderosa pine, fire suppression and timber harvest have created dense multi-storied stands of Douglas-fir (*Pseudotsuga menziesii* (Mirbel) Franco) and grand fir (Covington et al. 1994, Steele et al. 1986).

Historically in the moist forests, insects and diseases collectively with wind, ice, and snow interacted with fire to regenerate and regulate the development of these forests (Jain and Graham 2005). In concert, these disturbances and the resulting regeneration, development, and mortality created and maintained highly heterogeneous forests occurring at a variety of mosaics over time and space. As such, old-forests did not occur continuously across the landscape and a series of disturbances was required at the appropriate time and place for them to develop (Camp et al. 1997, Franklin et al. 2002). Nevertheless, before 1900 there were many expanses containing abundant and old western white pine, western larch, ponderosa pine, and Douglas-fir trees that fueled the timber based economies of the northern Rocky Mountains (Graham 1990, Hann et al. 1997, Jain and Graham 2005). These historic conditions create the desire of many to restore these forests to the grandeur they once had before the days of intensive timber management (Fins et al. 2001, Graham and Jain 2005, Neuenschwander et al. 1999).

In the Inland Northwest of the United States, clearcutting provided a way to remove the large amount of residual defective trees (e.g., grand fir, western hemlock) through

burning and to eradicate *Ribes* (the alternate host of blister rust) (Davis and Moss 1940, Matthews and Hutchison 1948). This regeneration method used for the production of western white pine became the norm even after *Ribes* control ceased (Fins et al. 2001) and scientific evidence indicated that western white pine does not need full sunlight to regenerate and develop (Graham et al. 1983, Haig et al. 1941, Jain et al. 2004). Although economically sound, applying clearcut methods and other historical harvesting systems that require short skidding distances (e.g. Idaho Jammer) and high road densities can affect wildlife habitat, soil and water quality, and other characteristics that society values (Bliss 2000, Harris 1984). Given these circumstances, to restore these forests, we designed a selection system to manage western white pine and other early seral species. This system, which we call the free selection silvicultural system (Graham and Jain 2005, Graham et al. 2007), was designed to create environments similar to those created by fine-scale disturbances (e.g. insect, disease, weather, surface fires) that removed single to small patches of trees.

Site preparation and surface fuels reduction

The preparation of sites for seed germination, seedling establishment, or planting is fundamental to all silvicultural systems because it helps define future forest composition and subsequent development (Eramian and Neuenschwander 1989, Graham et al. 1989, Graham et al. 2005, Haig et al. 1941, Nyland 2002). The use of fire is becoming increasingly difficult to apply due to concerns over air quality from smoke, the risk of fire escape, and the increased presence of structures in the urban wildland interface (Berry and Hessel 2004, Shindler and Reed 1996). Grapple machines are a common technique for piling debris, which offers more latitude for treating debris yet protects the soil resource. In addition, these machines are able to work in complex forest structures and on relatively steep slopes (≈ 40 to 50 percent) (Graham et al. 2005). A new emerging treatment for site preparation and treating harvest debris is machine mastication. These machines chunk, shred, grind, break, or in other ways reduce the size and character of standing and down material. As with grapple piling these machines can work in complex forest structures and on relatively steep slopes and leave a variety of forest floor conditions. There is a need to evaluate the economical and ecological effects of mechanical (grapple and mastication) and prescribed fire methods as to their efficacy in favoring heterogeneity in the future development of vegetation while decreasing fire hazard.

Because of the complex forest structures and forest floor conditions left after the above-mentioned silvicultural treatments, the traditional regular spacing of planted trees described in planting contracts may not be appropriate. Regular spacing of seedlings at 3 by 3 m or 4 by 4 m intervals is common for achieving timber management objectives and easily translated into contracts (Graham et al. 2005). However, naturally regenerated trees seldom occur with regular spacing and the historical fine-scale disturbances that regulated the structures and compositions of moist forests rarely maintained uniform forest conditions; rather, trees would grow in small groups with interlocking crowns and often with one or more species in a group. To foster the development of these heterogeneous forest structures and compositions, it is more appropriate to plant trees in groups and clumps of more than one species. Furthermore, if there is a wide variety of overstory densities and compositions combined with a mosaic of forest floor conditions, regularity in planting is impossible. Incorporating attributes such as variation in species and density based on forest floor and overstory conditions into language suitable for planting contracts requires innovative thinking to insure regeneration goals are accomplished.

Fire hazard and fuel treatments

Canopy bulk density, canopy base height, and amount and condition of surface and ladder fuels influence both fire intensity and burn severity (what remains after a fire). Their influence is important in all forests but appears to be most prominent or at least most recognized in the dry forests (Graham et al. 2004). In the moist forests, these metrics are also important but because the sites are productive, large amounts of surface, ladder, and crown fuels develop quickly after harvesting or other disturbances. Jain and Graham (2007) and Jain et al. (2006) recognized the potential for short longevity in fuel treatments in these forests and identified thresholds and estimates of uncertainty associated with forest characteristics such as forest developmental stage, canopy base height, and tree density and their influence on burn severity in moist forests after wildfires. Even though some of these forest characteristics have not been included when designing and implementing silvicultural systems and prescriptions, they are relevant for creating and maintaining forests resilient and resistant to wildfires and damaging burn severities.

Another component when managing forests for wildfire fuel is to disrupt or alter fire progression and provide or en-

hance suppression opportunities. Finney (2001) provides a conceptual way of placing fuel treatments within landscapes and shows the proportion of the landscape needing treatment to alter fire progression. Similar to the forest metrics that influence burn severity, the location and juxtaposition of fuel treatments can also be important for planning and executing silvicultural systems. FARSITE and FlamMap, two fire simulation tools, offer opportunities to evaluate the effectiveness of fuel treatments for disruption of a wildfire's progress within a landscape (Finney 1998, 2006).

The practice of silviculture, being an integrative discipline, is the proper construct to incorporate and evaluate objectives such as forest restoration and the creation and maintenance of old-forests, heterogeneous and complex forest structures and compositions, and fire resilient forests into an operational silvicultural system. These objectives are values important to many stakeholders. Integral to such an evaluation is disclosing the costs and logistics of implementing such a silvicultural system, which many would consider extremely complex and difficult to apply. Therefore, we designed this study to implement and evaluate silvicultural methods potentially valuable for restoring moist forests. The salient components of the study reflected in the previous discussion are to:

- 1) create forests with canopy openings that reflect fine-scale disturbances, but leave a high proportion of forest cover;
- 2) create forest conditions that would alter both fire intensity, spread, and burn severity if a wildfire were to occur;
- 3) evaluate different techniques applicable for treating slash and preparing planting and germination sites in highly complex and diverse forests; and
- 4) evaluate the logistics and costs of implementing such integrative and complex silvicultural systems using National Forest Systems procedures and contracts.

METHODS

Study area

The study is located on the Priest River Experimental Forest (PREF) on the west slope of the Selkirk Mountains in northern Idaho (fig. 1a). PREF (2573 ha) contains upland slope angles range from 20 to 60 percent and is dominated (two-thirds) by mixed conifer (e.g., western hemlock,

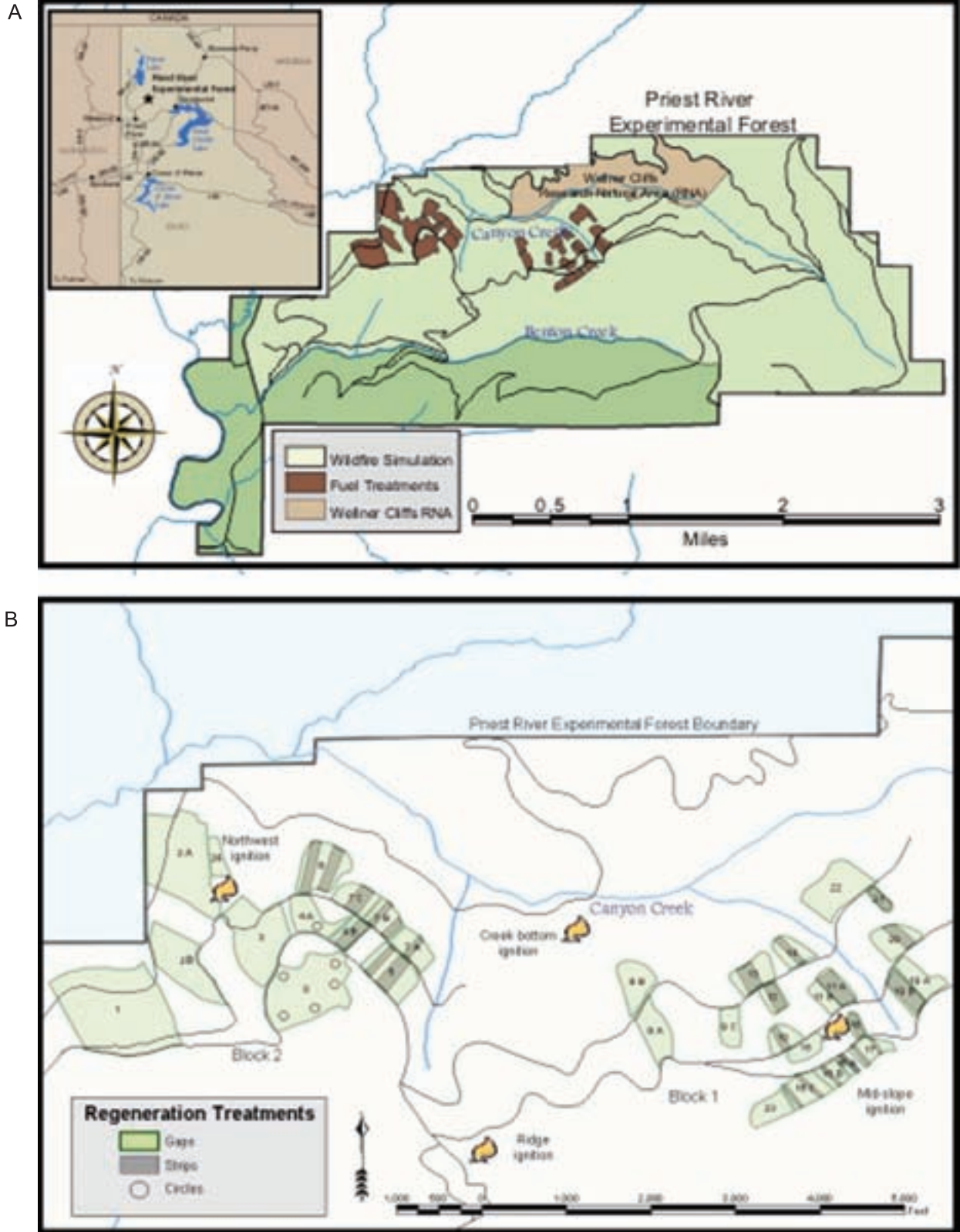


Figure 1— The study area is within the Canyon Creek drainage at Priest River Experimental Forest (a). We used a combination of different width strips and opening sizes to create diversity in forest structure within and among the harvested sites (b). To evaluate efficacy of treatment juxtapositions, we conducted a wildfire simulation using FARSITE and FlamMap within the area bordered by Benton Creek to the south and in the Canyon Creek drainage (light green area on 1a). Locations of orange flames indicate the four ignition points we used in the wildfire simulation: stream bottom, mid-slope, ridge, and northwest (b).

western redcedar, Douglas-fir, ponderosa pine, western white pine, grand fir) forest, ranging in age between 130 and 150-years old (resulting from a fire ca. 1860), with small portions having trees over 200 years old (Wellner 1976). Three primary drainages (Benton Creek, Canyon Creek, and Fox Creek) dissect PREF east to west, which drain into Priest River at 655 m in elevation (fig. 1a). As a result, primary topography faces north and south with small tributaries further dissecting slopes, creating a diversity of aspects with some areas facing east and west. The study area lies within the Canyon Creek drainage (fig. 1b).

The climate of PREF is transitional between the Pacific Coastal and Continental types. The Pacific climate favors moderate winter temperatures with January being the coldest (-4°C , 25°F), with the lowest temperature recorded since 1911 was -38°C (-36°F). Summers at PREF are sunny and dry with July being the warmest (18°C , 64°F) and the highest temperature ever recorded was 40°C (103°F) (Finklin 1983). Annual precipitation averages 817 mm with the majority (60 percent) in the form of snow in fall and winter, with accumulations in the valley bottoms averaging 49 cm. At high elevations (1455 m), precipitation averages 925 mm, with snowfall accounting for more than 50 percent of the total and snow accumulations averaging 132 cm. This snow pack can occasionally remain well into June.

SILVICULTURAL PRESCRIPTIONS

Management objectives and vision

The objectives and vision of the treatments were to create or begin developing forest compositions and structures that would enhance current and future research opportunities at PREF utilizing a variety of silvicultural methods. In addition, we wanted to create forest conditions resilient to insect, disease, wildfire, and climate change. To increase resilience, we wanted high amounts of early- and mid-seral tree species (e.g. western white pine, western larch, ponderosa pine), while simultaneously reducing the current abundance of grand fir, western hemlock, and Douglas-fir, without clearcutting (Everett et al. 1994, Hann et al. 1997, Harvey et al. 1999, Jain and Graham 2005). Because there is always wildfire potential at PREF, we wanted to modify the fire behavior and burn severity if a wildfire were to burn through Canyon Creek. To alter the potential fire behavior and the burn severity a wildfire could cause, we wanted to introduce heterogeneity within the vegetation (e.g., vegeta-

tion composition and structure, surface and ground fuel composition, structure, and distribution). Before treatment, Canyon Creek had areas of vegetation uniformity, coupled with pockets of high amounts of tree disease, and lacked the plurality of early seral species.

Current condition

The east-west orientation of the Canyon Creek drainage allows north- and south-facing aspects to dominate the topography (fig. 1a). Western hemlock habitat types, consisting of dense and continuous tree canopies, prevail on the northerly aspects. The southerly aspects, where western redcedar and grand fir habitat types dominate, are drier and the tree canopy cover tends to be more discontinuous compared to cover on the northerly aspects, (Cooper et al. 1991). The forests in Canyon Creek originated from a major wildfire that occurred nearly 160 years ago and historically contained a diverse mixture of conifers, with an abundance of western white pine. However, during the last century white pine blister rust killed many white pines (fig. 2). Western larch, the other historically dominant species, is present throughout the drainage, yet tends to be small in diameter (< 50 cm) and nearly all larch crowns in the drainage are heavily infected with dwarf mistletoe (*Arceuthobium laricis* Piper). Douglas-fir is also well represented in all canopy layers (fig. 2c), with the Douglas-fir beetle (*Dendroctonus pseudotsugae* Hopkins) especially active on southerly aspects. Dominant and co-dominant western hemlock, grand fir, and western redcedar trees are intermingled through the stands (figs. 2a, b, and d), especially on the northerly aspects, with the mid and low canopy layers containing dense (1000's to 10's of thousands of trees/ha) assemblages of all three of these species, often occupying small canopy gaps. The root diseases *Phellinus weirii* and *Armillaria spp.* are very prevalent throughout the area and Indian paint fungus (*Echinodontium tinctorium*) is very common in the boles of western hemlock and grand fir. The moderate to high amount of insect and disease activity in Canyon Creek, along with isolated wind and ice damage has contributed to moderate (37-74 Mg/ha) amounts of coarse woody debris dispersed across the forest floor and canopy gaps that have been readily occupied by regeneration of late seral species.

Desired future condition

The desired forest condition within Canyon Creek is to insure there is a representation of all the potential forest species and structures, with an emphasis on increasing western

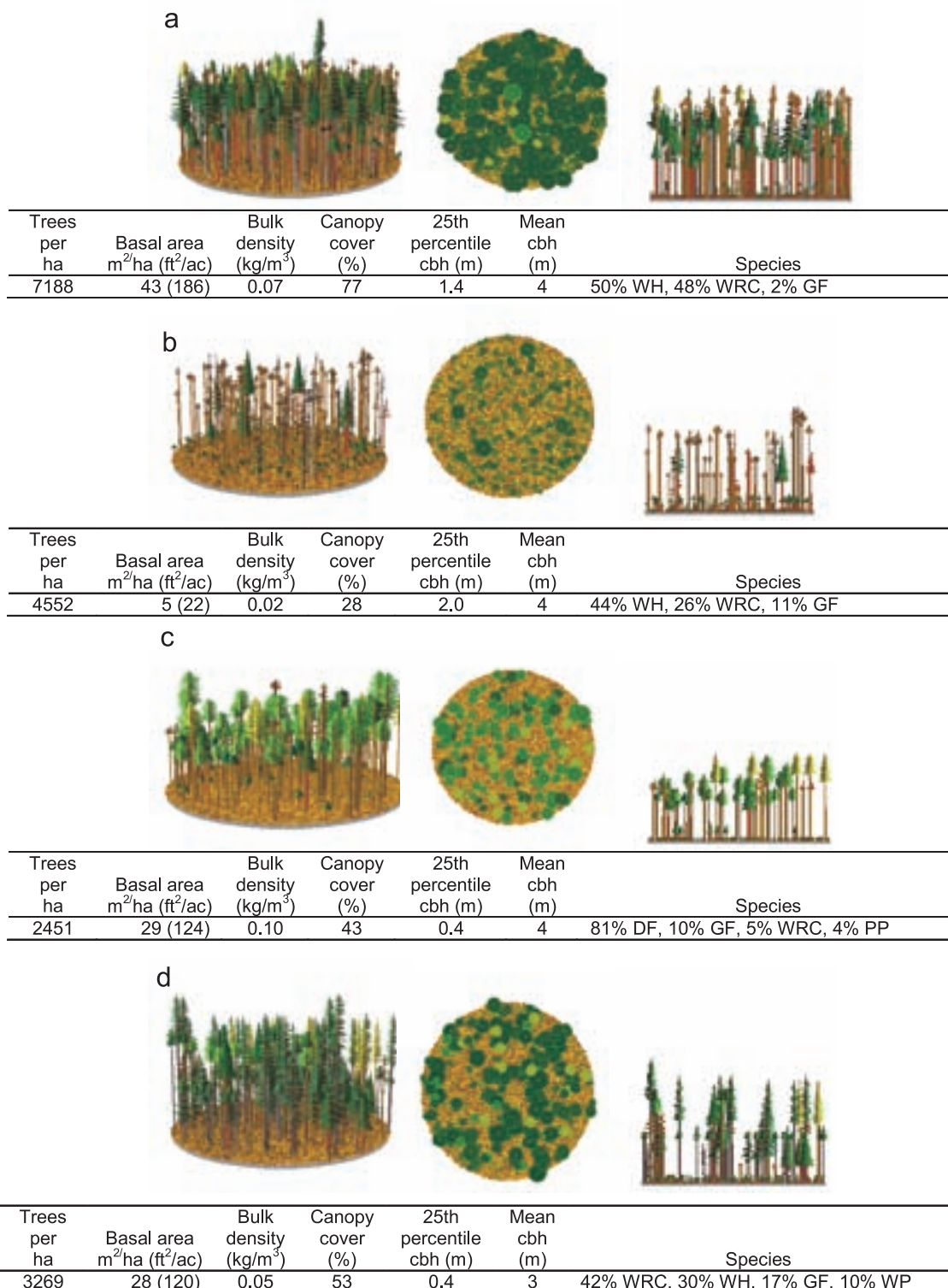


Figure 2— Before treatment variation in forest structure and composition. Western hemlock (WH) and western redcedar (WRC) dominated north-facing aspects with small amounts of grand fir (GF) and western white pine (WP) (a, b, d). Douglas-fir, grand fir, and a few ponderosa pine dominated west-facing aspects (c). The table provides forest structure characteristics shown in figure. All sites contained a diversity of canopy cover (28 to 77%), basal area (5 to 43 m²/ha), trees per ha (2400 to 7200), and canopy bulk densities (bulk density) (0.02 to 0.10 kg/m³). Average canopy base height (cbh) was 4 m; however, at least 25 percent of the observations on some sites had a cbh equal to or less than 1 m.

white pine abundance so that it becomes the primary cover type (defined as sites that contain a plurality of white pine) (Eyre 1980). The desired multi-species compositions and complex forest structures will provide abundant research opportunities not found in other areas of PREF, created by traditional even-aged management. By decreasing the abundance of suppressed and diseased trees and featuring dominant western white pine, western larch, and ponderosa pine where they occur, the resilience and resistance of the vegetation on PREF to native and introduced disturbances will increase (Hoff et al. 1976, Jain and Graham 2005). Also desired is the presence of large and old trees on PREF in addition to old-forest characteristics, which includes large trees, snags, and down logs. In all structural and compositional conditions, it is our goal to have high heterogeneity at all scales, from 0.01-12 ha.

Experimental design

Canopy dynamics (e.g., opening and gap creation along with tree and canopy growth) and forest floor conditions determine how a forest regenerates and develops. For regeneration treatments, we used a randomized complete block design. The design contained two blocks and within each block, six regeneration treatments were replicated a minimum of 3 times (table 1, fig. 1b). We were unable to cross all regeneration methods and slash treatments, because the overstory composition prevented us from using prescribed fire for treating the slash in several treatment units. For example, in areas where our free selection system created small (< 0.2 ha) canopy gaps among western hemlock and western redcedar, a prescribed fire would likely kill these fire intolerant trees. On such sites, we either grapple piled, masticated, or in some isolated areas, jackpot burned the surface fuels (table 1). Therefore, for slash treatments we used a second independent randomized complete block design. The design contained two blocks and four slash treatments and an untreated control, which were independently placed within the regeneration methods and each slash treatment was replicated 2 to 14 times (table 1). We recognize the combination of size and slash treatment influences forest regeneration and development. However, this interaction occurs at a far smaller spatial extent than a large regeneration harvest (Jain et al., 2002). Therefore, testing the interaction between canopy opening and slash treatment will be micro-site specific, using fisheye photography to quantify canopy cover and forest floor characteristics. Because of the forest conditions present in Canyon Creek, we have only one site where we used free selection to enhance the old-

forest characteristics of the site (site 22) (table 1 fig. 1b). We will evaluate this as a case study to determine if we create conditions that reflect old-forest structure.

Prescription implementation

Prior to implementing free selection in Canyon Creek, we conducted a thorough reconnaissance of the area to identify places containing high levels of disease, presence of western white pine, or containing or potentially containing old-forest structures. We identified operational constraints (access, slope angles), and limitations and preferences for different slash treatments. Using this information, we located 40 treatment units predicated by slope angles, current forest conditions, and access. Even though free selection and the vision guiding it framed the treatments we applied, we used strips of different widths (16-62 m) and circles (0.4 ha) to delineate areas (areas of disease, insect attack) where we ensured openings greater than a canopy gap were created (≥ 0.4 ha) (table 1) (Jain et al. 2004). As such, throughout the study area (271 ha) we created a wide variety of canopy openings and forest floor conditions (table 1).

Marking guide

In general western white pine, western larch, and ponderosa pine were the preferred species to favor in all settings and in particular near or within gaps and canopy openings (strips and circles) intended for regeneration (Jain et al. 2004). Heterogeneity, groups, and clumps of trees were preferred and tree marking was to have minimal impact on the inherent clumps of trees that often contained multiple species and even snags. Dominant or co-dominant trees ≥ 40 percent crown were favored; however, we also left western white pines with <40 percent crown, even those with dead tops. We removed intermediate and suppressed western white pines, depending on what the current condition presented, particularly if they had minimal seed potential. Large western larches (>46 cm) with full and mistletoe free crowns were preferred and larches ≥ 61 cm in diameter were often left, even with poor (<20 percent) crowns, for snag recruitment. Douglas-fir, grand fir, western redcedar, and western hemlock trees were frequently favored if their crowns were full (>40 percent), they were co-dominant or dominant trees, and when western white pine, ponderosa pine or western larch could not fulfill the desired conditions outlined in the objectives and vision for Canyon Creek. Trees of all species were eligible to become potential snags and subsequent large log recruitment, with western redcedar having the highest priority, followed by

Table 1. Regeneration and slash treatment summary. The experimental design included two blocks and within each block three replications of different width strips and circles. The objective, current condition, and the use of western white pine (WP) management thresholds determined opening size. Western white pine thresholds for regeneration establishment is between 25 and 54 percent canopy opening, competitive advantage (comp. adv.) begins at 55 percent, and free-to-grow begins at 92 percent. Favored species are western white pine (WP), western larch (WL), ponderosa pine (PP) and western redcedar (WRC). Slash treatments were replicated a minimum of five times across the entire study area and include grapple pile (grapple), prescribed fire (pres. fire), mastication (mast.), jackpot burn, and no treatment (NT).

Block	Unit no.	Regeneration treatment	Unit size (ha)	Strip or circle (ha)	Rep.	WP threshold (+ is above threshold)	Canopy opening (%)	Tree species planted	Slash treatment
1	17_1	16.0 m strip	1.0	0.10	1	Comp. adv.	55	WP, WL	Grapple
1	17_2	16.0 m strip	1.0	0.14	2	Comp. adv.	55	WP, WL	Grapple
1	16c1	16.0 m strip	1.1	0.15	3	Comp. adv.	55	WP, WL	Grapple
1	18	31.0 m strip	0.4	0.26	1	Comp. adv. +	70	WP, WL	Pres. fire
1	10	31.0 m strip	1.0	0.27	2	Comp. adv. +	70	WP, WL	Mast.
1	14	31.0 m strip	1.1	0.33	3	Comp. adv. +	70	WP, WL	Mast.
1	20	47.0 m strip	1.6	0.48	1	Comp. adv. +	85	WP, WL	Grapple
1	12	47.0 m strip	0.8	0.46	2	Comp. adv. +	85	WP, WL	Mast.
1	21	47.0 m strip	0.4	0.34	3	Comp. adv. +	85	WP, WL	Grapple
1	19	62.0 m strip	1.2	0.94	1	Free-to-grow +	100	WP, WL	Pres. fire
1	11a	62.0 m strip	1.1	0.83	2	Free-to-grow +	100	WP, WL	Pres. fire
1	13	62.0 m strip	1.6	0.64	3	Free-to-grow +	100	WP, WL	Grapple
1	8_1	0.4 ha circle	6.6	0.40	1	Free-to-grow	92	WP, WL, PP	Mast.
1	8_2	0.4 ha circle	6.6	0.40	2	Free-to-grow	92	WP, WL, PP	Mast.
1	8_3	0.4 ha circle	6.6	0.40	3	Free-to-grow	92	WP, WL, PP	Mast.
1	23	<0.2 ha gap	1.3	<0.20	1	Establishment	25 - 40	WRC, WP	Mast.
1	15	<0.2 ha gap	0.9	<0.20	3	Establishment	25 - 40	WRC, WP	Mast.
1	9a	<0.2 ha gap	2.0	<0.20	4	Establishment	25 - 40	WRC, WP	Mast.
1	9b	<0.2 ha gap	1.8	<0.20	4	Establishment	25 - 40	WRC, WP	Jackpot
1	9c	<0.2 ha gap	1.1	<0.20	5	Establishment	25 - 40	WRC, WP	Mast.
1	22	<0.2 ha gap	3.3	<0.20	6	Old forest	20 - 30	WRC, WP	Grapple
2	16a	16.0 m strip	0.2	0.15	1	Comp. adv.	55	WP, WL	Mast.
2	16b	16.0 m strip	1.4	0.14	2	Comp. adv.	55	WP, WL	NT
2	16c2	16.0 m strip	1.1	0.21	3	Comp. adv.	55	WP, WL	Grapple
2	5_1	31.0 m strip	2.8	0.34	1	Comp. adv. +	70	WP, WL	Pres. fire
2	7a	31.0 m strip	1.4	0.21	2	Comp. adv. +	70	WP, WL	Mast.
2	7b1	31.0 m strip	1.3	0.28	3	Comp. adv. +	70	WP, WL	Mast.
2	7b2	47.0 m strip	1.3	0.24	1	Comp. adv. +	85	WP, WL	Grapple
2	6_1	47.0 m strip	2.8	0.70	2	Comp. adv. +	85	WP, WL, PP	Pres. fire
2	6_2	47.0 m strip	2.8	0.68	3	Comp. adv. +	85	WP, WL, PP	Pres. fire
2	5_2	62.0 m strip	2.8	0.74	1	Free-to-grow +	100	WP, WL	Mast.
2	4b	62.0 m strip	0.9	0.67	2	Free-to-grow +	100	WP, WL	Mast.
2	7c	62.0 m strip	1.4	0.58	3	Free-to-grow +	100	WP, WL	Pres. fire
2	4a	0.4 ha circle	1.7	0.40	1	Free-to-grow	92	WP, WL	Mast.
2	8_4	0.4 ha circle	6.6	0.40	2	Free-to-grow	92	WP, WL, PP	Mast.
2	8_5	0.4 ha circle	6.6	0.40	3	Free-to-grow	92	WP, WL, PP	Mast.
2	1	<0.2 ha gap	11.9	<0.20	1	Establishment	25 - 40	WRC, WP	Grapple
2	2a	<0.2 ha gap	7.7	<0.20	2	Establishment	25 - 40	WP, PP	NT
2	2b	<0.2 ha gap	3.9	<0.20	3	Establishment	25 - 40	WP, PP	NT
2	3	<0.2 ha gap	5.1	<0.20	4	Establishment	25 - 40	WRC, WP	Jackpot

western larch, ponderosa pine, western white pine, and western hemlock.

When we encountered favored species with short crowns or heavily infected with disease, openings most often were created within the range of competitive advantage (55 percent) and free-to-grow (92 percent) thresholds and, if available, dominant western white pine and disease and mistletoe-free western larch were featured (Jain et al. 2004). In some locales within Canyon Creek, because of slope steepness (>40 percent), we applied free selection in strips to accommodate skyline yarding systems. Nevertheless, these strips contained different canopy densities and had a variety of widths; while still addressing the canopy opening thresholds identified by Jain et al. (2004). However, strip placement had to fall within the context of the current forest conditions (areas where a strip could be located without removing favored species and composition) and within the context of fulfilling the short- and long-term conditions outlined in the vision. We ensured that near the strips, we increased tree density slowly (e.g., feathering of the edges) to minimize any artificial appearance of the forest (straight boundaries). Single species to multiple species groups and clumps (or both) of trees tend to occur naturally in the moist forests as do single trees. We maintained this character in the forest by avoiding the creation of geometrically uniform openings (e.g. squares or rectangles), even when canopy openings were of sufficient size for western white pine to grow freely (>0.4 ha).

Harvesting

We used both ground based and skyline yarding systems, depending on the slope angle and residual tree densities (figs. 3a and b). We used the skyline system on slopes exceeding 40 percent, and contrary to normal sets, where yarding follows the slope profile, we yarded diagonal to the slope. This yarding technique yields different forest and forest floor conditions than yarding following the slope profile. As such, the visual impacts were minimal. In addition,

this corridor configuration readily created residual tree spacing and an appearance very compatible with our application and conditions we wanted to create using free selection. On slopes <40 percent we used both conventional rubber tired skidders and processors (cut-to-length) to do the harvesting. The operator made the choice of machines, with residual tree density influencing the decision.

Slash treatments

There were three slash treatments, grapple pile followed by pile burning, mastication, and prescribed fire. Prior to grapple piling the slash, hand crews cut undesirable standing trees and cut down long material in smaller pieces (slashing). Two mid-sized excavators equipped with hydraulic thumbs, at \$82.50/hr, piled the material after slashing on 10 treatment units (table 1, fig. 3c). The piling occurred during July and crews burned the piles in fall of 2006. During the pile burning and especially in the area in which we used free selection to enhance old-forest character (site 22), we



a) Ground based yarder



b) Skyline yarder



c) Grapple piler



d) Masticator

Figure 3— Various equipment and yarding techniques used to harvest sites and conduct slash treatments. The topographic relief required two yarding techniques: (a) ground based equipment included skidders (pictured), processors, and forwarders; (b) we used a slack skyline to harvest the steeper (40%) slopes. We used (c) grapple and (d) mastication equipment for mechanical fuel treatments in addition to prescribed fire (broadcast and jackpot).

Table 2. Weather from August 26 through September 2, 1967. During the FARSITE and FlamMap simulations we used minimum and maximum temperature, and minimum and maximum relative humidity during the Sundance wildfire of 1967 from the Priest River weather station. Dead fuel conditioning period occurred from August 26 until September 1, prior to ignition, and the burn period was from 0400 on September 1 through 2300 on September 2.

Month	Day	Minimum temperature hour	Minimum temperature C° (F°)	Minimum relative humidity (%)	Maximum temperature hour	Maximum temperature C° (F°)	Maximum relative humidity (%)
Conditioning period							
August	26	0500	0 (32)	16	1600	31 (87)	98
	27	0400	10 (50)	34	1600	23 (74)	94
	28	0500	3 (38)	23	1400	32 (89)	98
	29	0600	6 (42)	14	1400	34 (94)	99
	30	0400	7 (45)	16	1500	34 (94)	99
	31	0500	6 (41)	12	1600	37 (98)	99
Burn period							
September	1	0500	6 (42)	13	1400	33 (92)	98
	2	2300	7 (44)	30	1400	26 (78)	97

allowed the fires to creep, creating a mosaic of soil burn severities. To treat the slash through mastication, we used a large ($\approx 36\,364$ kg) excavator with a rotary mastication head, that cost \$300/hr. Hand slashing was not necessary because the machine treated both standing and down material (18 units) (table 1, fig. 3d). Even though the machine was large it had the capacity to operate on steep (45 percent) slopes and its ground pressure was light (4 g/cm^2). There was minimal soil displacement and compaction as it only traversed a 21 m wide strip once while masticating. With this machine we tried to create a variety of piece sizes with the smallest being the size of a basketball or football so as to maintain the wood integrity (outer bark, inner bark, cambium, sapwood, and heartwood) in contact with the forest floor enhancing the decomposition of these materials (Graham and Jain 2007).

After hand slashing the unwanted standing material, District crews conducted spring prescribed fires in seven of the treatment units starting in 2007. These fires tended to be very low intensity with short (<1.0 m) flame lengths and were conducted when the lower duff moisture contents exceeded 100 percent. At these moisture contents, forest floor organic material consumption is minimal, preserving the microbial and nutrients intrinsic to them, yet providing a burned over surface ideal for conifer seed germination (Graham et al. 2005, Haig et al. 1941). In addition to the spring prescribed fires, jackpot burns will remove concentrations of slash in the fall (2007) after wetting rains in two units. These units are inaccessible for machine piling and

grapple piling and inappropriate for using prescribed fire because western redcedar and western hemlock dominate the residual trees (figs. 2a, b, and d). In the units where Douglas-fir, western larch, or ponderosa pine are dominant, we used prescribed fire (fig 2c).

Wildfire simulation

To determine effects of created stand structures and our increase in fuel heterogeneity, we simulated several wildfires using FARSITE and FLAMMAP (Finney 1995, 1998, 2006, Stratton 2004). FARSITE models fire progression and rate of spread while FlamMap predicts whether a particular site will favor a particular fire behavior such as

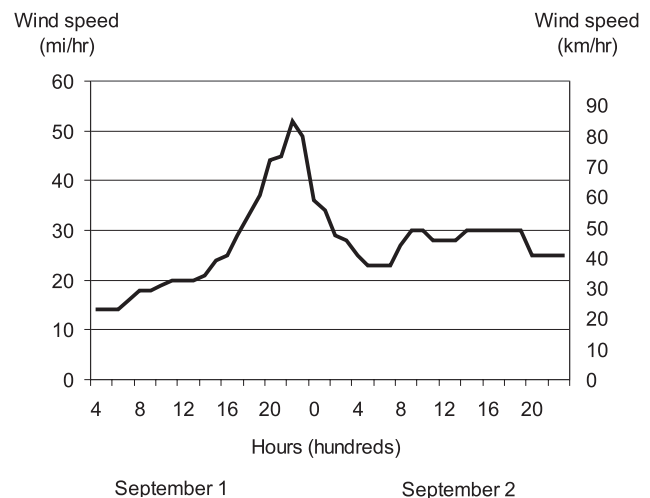


Figure 4— Wind speeds used during the FARSITE simulation. We used wind speeds from September 1 and 2, 1967, which was the same period when the Sundance wildfire had the fastest rate of spread. In the FlamMap simulation, we used a wind speed of 82 km/hour.

Table 3. FARSITE simulation parameters and fire behavior options used to test the effectiveness of treatments for altering fire behavior.

Model input/output parameter	Parameter selected	Fire behavior options	Option selected
Time step (min)	30	Enable crown fire (Finney method)	Yes
Visible step (hrs)	4	Embers from torching trees	Yes
Perimeter resolution (m)	30	Enable spot fire growth	No
Distance resolution (m)	15	Ignition frequency	1.00
Units	metric	Ignition delay	0

a surface, passive crown, or active crown fire. The wildfires we simulated burned a large portion of PREF including the southern aspects in Benton Creek and the entire Canyon Creek drainage (fig. 1a). To populate this landscape with the appropriate data, we used a combination of LiDAR (light detection and ranging), stand exams, and research plot data. These data included canopy cover (percent), tree height, canopy base height, canopy bulk density, elevation, aspect, slope, and fuel model. The research staff familiar with PREF assigned the appropriate fuel model to each stand using descriptions from Anderson (1982) and Scott and Burgan (2005). LIDAR, remotely sensed data, was adjusted using plot data in regressions; thus the LIDAR data ultimately populated the area (each stand) with the appropriate information needed to conduct the FARSITE and FlamMap analysis.

Using long-term weather records from PREF (since 1911), we identified an extreme weather event out of several years (1919, 1926, 1939, 1967, 1991, and 2003) to use in the FARSITE and FlamMap simulations (table 2). We selected August 26 through September 2, 1967 which was hot (23–36° C, 74–98° F) dry (relative humidity, 12–34 percent), and windy (6 m wind speeds were > 81 km/hr) and coincided with the period when the Sundance fire burned near PREF (Anderson 1968, Finklin 1983) (table 2, fig. 4). The fire burned over 22 662 ha and on September 1, 1967, moving sometimes at a speed 10 km/hr. Selecting the period of the Sundance fire gave us a realistic and extreme condition to evaluate our free selection system in influencing wildfire intensity and burn severity. In addition to the input data, we selected parameters and fire behavior options in FARSITE and FlamMap when running the simulations (table 3).

The distance between the forest surface and the canopy base height is a major forest characteristic manageable with fuel treatments. This characteristic also tends to be very

sensitive for predicting crown fire potential. In addition, this attribute is highly variable and is typically calibrated using expert knowledge when using FARSITE. For example, when simulating wildfires using FARSITE and the results do not reflect a probable outcome (crown fire), canopy base height is often “tweaked” to produce expected results. To avoid this subjective modification of canopy base heights, we used four different canopy base heights (CBH) in our FARSITE simulations. Rather than using, one CBH value in our simulations we arrayed the data and used the 1st and 3rd quartiles values along with the median CBH obtained from the FFE-FVS tree list (Dixon 2002). We also used a CBH calculated with FFE -FVS where the program defines CBH as the height where canopy bulk density exceeds 0.011 kg/m³ (Reinhardt and Crookston 2003). Because there are so many trees < 0.3 m in Canyon Creek, we used this value rather than the default of 2 m for calculating CBH. These stand level measurements of CBH combined with the LIDAR data allowed us to estimate for the entire area these four CBH values.

To evaluate how the juxtaposition of fuel treatments may influence a fire’s behavior and progression we used four different and realistic ignition points in our simulations and selected these points prior to locating them in relation to the treatments we applied in Canyon Creek (fig. 1b). We chose one ignition point near the bottom of Canyon Creek and one in the northwest corner of PREF. We chose a mid-slope ignition point near a well-traveled road, as well as a ridge ignition point. By establishing this range of ignition points, we ensured a variety of simulated fires (e.g., head, backing, or flanking) would encounter the treatments we established.

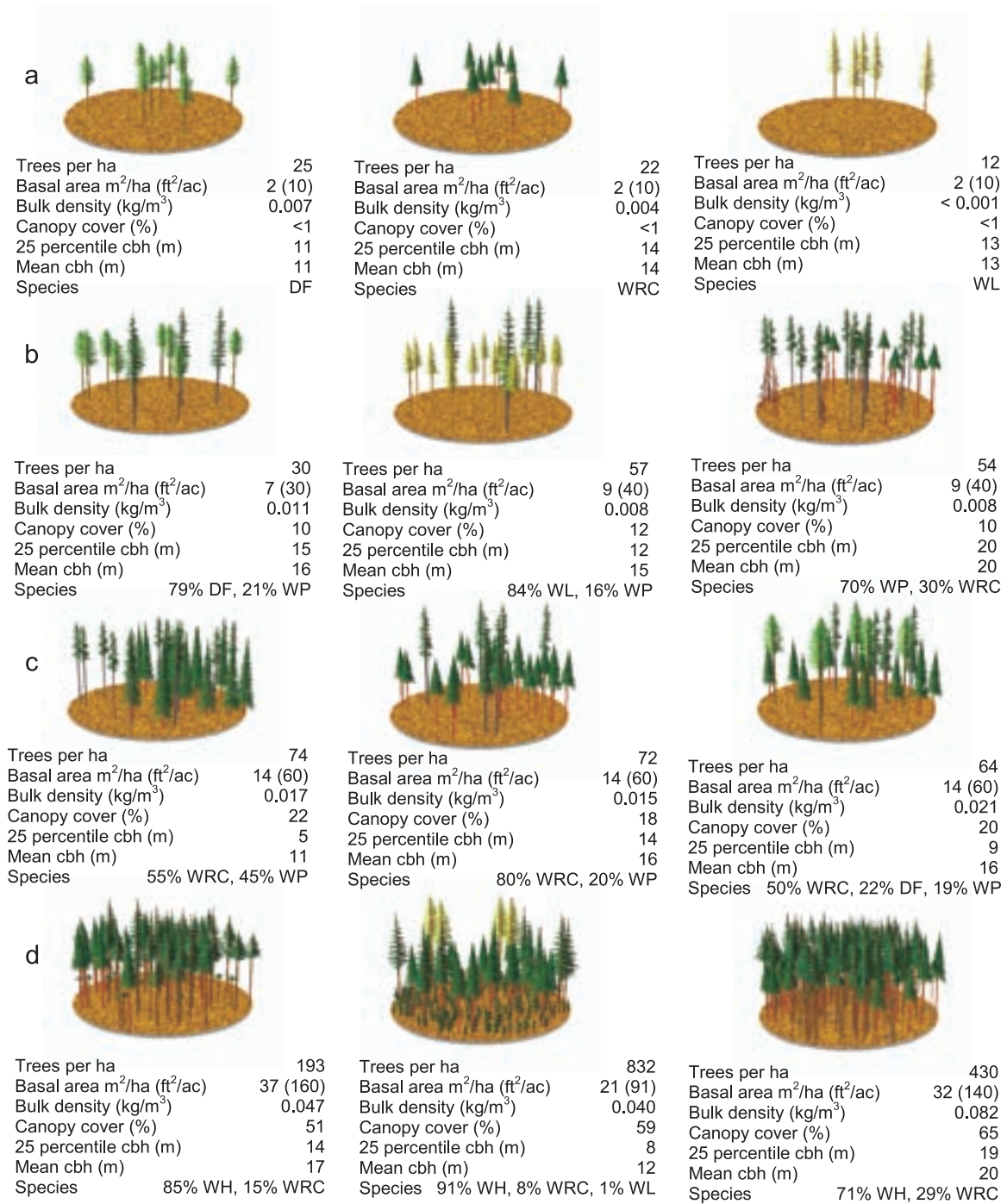


Figure 5— After treatment variation in forest structure and composition. Each series of examples reflect observations that occurred within the (a) 25th, (b) 50th, (c) 75th, and (d) 100th quartile in canopy cover. The treatments maintained considerable variation in canopy cover and species composition while favoring the abundance of western white pine (WP), western larch (WL), western redcedar (WRC), and Douglas-fir (DF) (a, b, and c). Only the sites that contained the highest densities favored western hemlock (WH) as the primary species (d). In all cases, we increased canopy base height to disfavor the potential for crown fire.

Table 4. Slash treatment costs for grapple piling. Approximately 17 ha were grapple piled on 9 units. Costs included actual work (hand slashing and grapple piling), administration (admin.) costs, and pile burning (not in table). Pile burning cost an average of \$106.00/ha bringing total average cost to \$1877/ha. Variation in cost and productivity (prod.) was due to number of trees needing slashing, steepness of slope, and avoiding residual trees.

Unit Number	Area treated (ha)	Slope (%)	Debris distribution			Costs/ha (US dollars)				
			Before (Mg/ha)	After (Mg/ha)	Change (%)	Hand slash	Grapple pile	Admin.	Total	Prod. (ha/hr)
7b1	1.3	32	137	63	-54	312	532	65	909	0.92
5_1	2.8	35	90	38	-57	517	696	53	1,267	0.70
20	1.6	31	72	54	-25	128	1,047	128	1,303	0.46
13	1.6	32	148	99	-33	495	805	70	1,370	0.60
1	6.1	55	N.d. ^a	N.d.	N.d.	786	694	104	1,583	0.70
22	0.8	26	117	109	-7	290	1,346	244	1,880	0.36
21	0.4	28	114	74	-35	732	1,088	133	1,953	0.44
16	1.1	41	80	94	18 ^b	766	1,341	508	2,615	0.36
17	1.1	43	N.d.	N.d.	N.d.	3,377	1,073	459	4,909	0.46
Mean (ha)	1.8	36	108	76	-30	768	852	151	1771^c	0.10^d

^a N.d. indicates no fuels information available for these units.

^bWoody debris increased because of redistribution of large wood.

^cAverage per hectare costs = total of each (individual cost*unit hectares)/total hectares

^dAverage (hectare) production = total hectares/172 hours

RESULTS

Forest structures and species composition

Prior to treatment, the density of trees in Canyon Creek varied from several hundred to several thousand of trees per hectare (fig. 2). Western hemlock, western redcedar, Douglas-fir, and grand fir dominated the sites with a presence of western white pine, ponderosa pine, and western larch (fig. 2). Canopy cover ranged from 28 percent (fig. 2b), exemplified by small crowned western redcedar to large crowned (77 percent) western hemlock and western redcedar (figure 2a). Multiple canopy layers were the norm with the intermediate and suppressed crown classes dominated by western hemlock and western redcedar. However, small amounts (< 10 percent canopy cover) of western white pine (fig. 2d), ponderosa pine (fig. 2c), and western larch (not shown) were present in the dominant canopy class.

Our application of free selection created a variety of forest structures and compositions (fig. 5). The 1st quartile (the first 25 percent of the observations when ranked from low to high residual canopy cover) of the distribution of canopy cover occurred predominantly in areas where we removed numerous trees with thin crowns and abundance of disease (purposely located strips) (fig. 5a). These areas

contained <1 percent canopy cover consisting of up to 12 through 25 trees/ha of western larch, or western redcedar, or Douglas-fir, or western white pine (not shown) with residual basal areas in the range of 2 m²/ha. In the second quartile (26 through 50 percent) of observations, the canopy cover ranged from 10 through 12 percent and ranged from 30 through 57 trees/ha consisted of western larch and western white pine, creating a maximum basal area of 9 m²/ha reminiscent of a shelterwood (Nyland 2002) (fig. 5b). The maximum canopy cover for the third quartile (51 through 75 percent) of observations was approximately 20 percent comprising 14 m²/ha of western redcedar and western white pine and in some cases Douglas-fir (fig. 5c). Canopy cover in the fourth quartile (76 through 100 percent) of observations, tended to be >50 percent and basal area ranged from 21 through 37 m²/ha of western hemlock and western redcedar (fig. 5d). Contrary to traditional seed-tree and shelterwood harvesting, regular spacing among trees was non-existent; rather, trees were grouped, clumped, and irregularly distributed across the sites (fig. 5). On all sites, average canopy base heights ranged from 11 through 20 m with the 25th quartile ranging from 5 through 19m.

Table 5. Slash treatment costs for mastication. Approximately 15 ha had activity fuels masticated. Costs are associated with administration (admin.), and actual work (masticate). Variation in cost was due to steepness of slope and leaving residual trees.

Unit number	Area treated (ha)	Slope (%)	Debris distribution			Costs/ha (US dollars)			Residual trees/ha	Prod. (ha/hr)
			Before (Mg/ha)	After (% change)		Masticate	Admin.	Total		
				>8cm	<8cm					
8_4	1.6	7	76.7	-54	54	914	109	1,022	32	1.95
12b	0.4	7	83.2	-32	175	1,010	121	1,131	32	1.78
8_2	1.6	2	46.6	-32	124	1,030	123	1,153	32	1.73
14	0.6	2	105.8	-23	150	1,078	128	1,206	35	1.66
8_3	0.6	5	42.1	-64	27	1,097	131	1,228	32	1.63
12	0.7	10	87.7	-51	110	1,112	133	1,245	32	1.61
8_1	1.8	5	53.8	-41	92	1,112	133	1,245	32	1.61
10	0.9	5	78.2	-23	54	1,221	145	1,366	27	1.46
4b	0.7	12	61.9	-37	66	1,279	152	1,431	31	1.41
23b	0.5	49	114.1	-63	-4	1,363	162	1,525	24	1.31
9a	1.3	15	129.6	-65	90	1,426	169	1,595	45	1.26
23	0.7	10	114.1	-63	-4	1,474	176	1,651	30	1.21
7a	1.1	5	75.8	-32	224	1,479	176	1,656	35	1.21
4a	0.9	5	77.8	-53	-23	1,566	186	1,752	38	1.14
15	0.7	17	54.7	-54	99	1,571	186	1,757	25	1.14
9c	0.4	44	105.6	-47	136	1,619	193	1,813	45	1.11
14b	0.3	30	124.0	-54	24	1,634	196	1,830	40	1.09
7b1	0.2	10	96.6	-32	66	1,972	234	2,207	35	0.91
16a	0.1	49	84.1	-72	-14	3,374	401	3,775	50	0.52
Mean (ha)	0.8	10	37.8	-47	99	1,297	156	1,453 ^a	34	0.23 ^b

^aAverage per hectare costs = total of each (individual cost*unit hectares)/total hectares

^bAverage (ha) production = total hectares /65.34 hours

Slash treatments

One-hundred and eighty two hours were spent grapple piling 16.8 ha with an additional 520 hours spent hand slashing the material before it was piled (table 4, fig. 3c). The highest piling costs occurred in areas (unit 17) with abundant standing trees that required slashing. Often these costs were confounded with high administrative costs due to crew and machine operator supervision. When slope angles approached 45 percent, the machine operator needed more finesse, thus slowing the machine. In addition, with these steep slope angles, slashing crews tended to become more readily fatigued, slowing productivity and increasing costs. Nevertheless, the machines piled an average of 0.1 ha/hr with slashing, piling, and administrative costs together averaging \$1771/ha (table 4). The following fall, crews spent 108 hours burning the piles, which cost an

additional \$106/ha, resulting in a total of \$1877/ha for treating the slash by machine piling.

A little over 65 hours was spent masticating 15 ha (table 5, fig. 3d). By masticating, the objective was not to remove the material but to change its consistency, size, and distribution and bring the material in contact with the forest floor to favor its decomposition (Graham et al. 2005). Material ≥ 7.6 cm in diameter consisting of logs, limbs, and unwanted standing trees was converted to small chunks and wood pieces < 7.6 cm. For example, mastication reduced total fuel loadings (tons/hectare) of wood ≥ 7.6 cm in diameter by an average of 47 percent; however, there was a two-fold increase in the abundance of small material (table 5). There were a few sites, particularly in unit 4a, where the small material was also reduced by 23 percent.

The mastication machine treated an average 0.23 ha/hr, with an average cost of \$1453/ha including administrative costs (table 5). Masticating slash on steep slopes (>50 percent slope angle) required additional care by the operator to maintain machine stability, which slowed production and escalated costs (i.e. unit 16a at \$3775/ha) (table 5). Even though the machine was capable of working in areas with abundant standing trees, these residual trees prevented the operator from freely swinging the boom and required operator diligence not to damage them, also decreasing productivity and increasing costs. In general, the machine operator, after instruction, made all decisions on what live trees to cut and how the fuels were to be treated but an on-site administrator observed and directed the mastication when necessary. As such, the mastication treatment created

a wide variety of standing tree compositions and structures as well as a multitude of forest floor conditions.

In very similar physical and forest conditions, the masticator treated more area on the average in one hour (0.23 ha/hr) when compared to the grapple piling (0.10 ha/hr) (tables 4 and 5). Moreover, it was less expensive (\$1453/ha) to masticate the slash than grapple pile it (\$1877/ha), as the grapple piling required slashing, piling, and pile burning while mastication treated the slash in one step. However, mastication does not remove material but it instead changes its structure and distribution. Because mastication temporarily increased the amount of fine fuels, it also increased the fire hazard at least in the short-term (Graham and Jain 2007). However, the retention of woody material on sites will conserve nutrients and provide organic materials to the forest floor (Harvey et al. 1987). Slope steepness and abundance in the residual tree density tended to affect both mastication and machine piling equally, slowing productivity and increasing costs.

Wildfire Simulation

Influence of canopy base height

We used a mid-slope ignition point to evaluate four different CBHs; the FFE-FVS CBH calculation, the 1st quartile, median, and 3rd quartile CBHs (figs. 1b. and 6a). The FFE-FVS calculation produced the lowest canopy base heights, followed by the 1st quartile estimate of CBH, which was 2 m. Estimating CBH using the medians produced a CBH of approximately 4 m and the CBH for the 3rd quartile was estimated at 6 m. We used these different estimates of CBH to determine their influence on a wildfire's development, if it were to burn in Canyon Creek, without any of our treatments in place in the summer of 1967 (fig. 6b). Our simulations showed that if the median and 3rd quartile canopy base heights were used to estimate CBH, the overall size and rate of spread of a fire was less when compared to the rate of spread and size of a fire using the 1st quartile and FFE-FVS estimated CBHs. There was little or no difference in fire size and rate of spread between the median and 3rd quartile CBH. In contrast, there were some minor differences in fire size and rate of spread using CBHs estimated by FFE-FVS (more aggressive fire behavior) and with CBH estimated using the 1st quartile (fig. 6b). Based on these results, we selected the 1st quartile for the wildfire simulation.

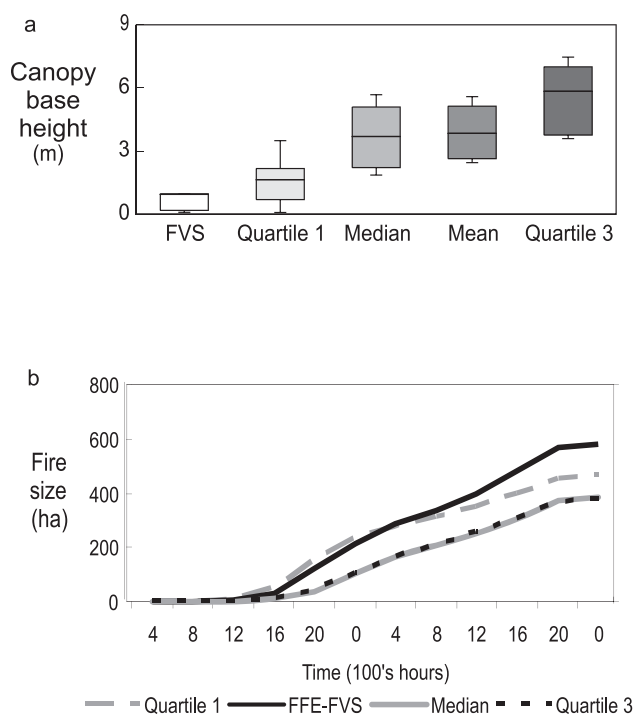


Figure 6—Influence of canopy base height (CBH) on fire size. We selected four canopy base height values from our observations to determine which canopy base height should be used in our model simulations. Fire and Fuels extension of the Forest Vegetation Simulator (FFE-FVS) value is the CBH when canopy bulk density exceeds 0.011 kg/m³ (Reinhardt and Crookston 2000) (a). We used a frequency distribution to identify the average, 25th, 50th, and 75th quartile. Average CBH and the 50th quartile had similar distributions; therefore, we used the 50th quartile in our evaluation. Although the 50th and 75th quartiles were different heights, fire size was similar for both values (b). The FVS calculated value burned the largest area followed by the CBH value reflected at the 25th quartile (b). For this analysis, we used the 25th quartile value to evaluate the efficacy of treatment juxtaposition in the FARSITE and FlamMap simulations because it appeared to reflect expected fire behavior.

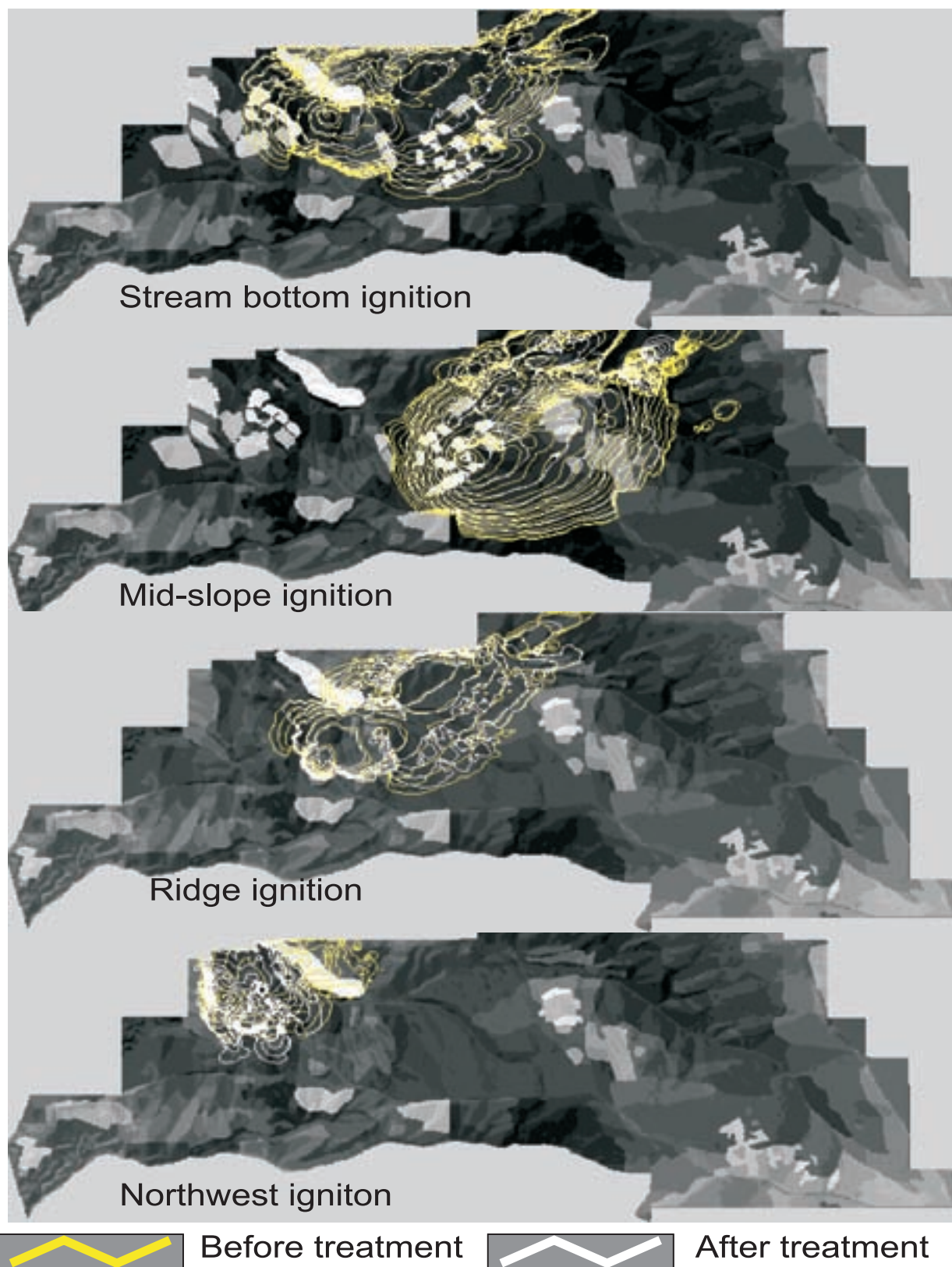


Figure 7— FARSITE comparison of fire progression before (yellow) and after (white) treatment. We evaluated fire progression using four ignition points (stream bottom, mid-slope, ridge, and in the northwestern part of the study area) (see fig. 1b). For illustration, the treated landscape is in the background. Although total fire size did not decrease substantially, fire progression changed when burning occurred through treated areas, by circling around units and changing fire shape.

Table 6. The number of hectares burned by fires before and after treatment started from different ignition points (fig. 1) using weather from 1967 in FARSITE simulations.

Time	Ignition points				Fire type		
	Stream bottom	Mid-slope	Ridge	Northwest corner	Surface	Passive	Active
	-----Hectares burned-----				-----Percentage of area-----		
Before treatment	334	478	288	137	68	35	< 1
After treatment	382	446	280	122	71	32	0

Fire progression with and without treatments

Using FARSITE and FlamMap to simulate fires burning in Canyon Creek during very dry and hot conditions, the treatments, although small (many <2 ha), influenced the progression and behavior of the fires during the 1967 weather for four selected ignition points (fig. 7). The yellow contour lines reflect the fire's progression prior to the treatments and the white contour lines show the fire's progression after the treatments. The contour lines represent the distance the fire moved in a four-hour period. Because there were high winds in 1967 (32, 48, and 88 km/hr) from the west, the fires progressed to the northeast (fig. 7). Pronounced spotting (although we did not allow them to grow) occurred when there were no treatments compared to the simulations after treatments, particularly with the northwest ignition, mid-slope ignition, and stream bottom ignition. The ridge ignition showed a similar elliptical pattern in both the pre- and post-treatment simulations but the post-treatment simulation burned less area (280 ha) compared to pre-treatment (288 ha) (fig. 7, table 6).

Of the four ignition points tested, fire size decreased by 12 percent after the treatments using the northwest ignition (fig. 7, table 6). This ignition occurred within the middle of the treated area and treatment placement prevented the fire from burning outside the treated areas within the 43 hrs we simulated the fire. As a result, by treating the area we burned 122 ha compared to the 137 ha we burned if the area was untreated (table 6). Unlike with the other ignition points, with the stream bottom ignition point a larger area (382 ha) burned after the treatments compared to the pre-treatment fire size (334 ha). However, the backing fire slowed when the fire burned through the western block of treated units (fig. 7). The progression of both the before and after treatment fires using the mid-slope ignition point was similar. However, fire size after treatments was slightly less (446 ha) compared to the simulated fire before the

treatments (478 ha) (table 6). Moreover, the treatments did not allow the fire to progress to the northeast to the same extent as the pre-treatment simulation. When we ignited the fire on the ridge, the fire was more susceptible to winds, but the treatments did slow fire growth, particularly on the northwest side of the treatments (fig. 7).

FlamMap simulates fire behavior characteristics for a given forest structure, topography, and weather scenario. There is no contagion among the pixels, the information provided reflects only one point in time, and there is no specific ignition point (Finney 2006). The point in time we selected in 1967 was September 1 at 2000 hours, the period with the highest wind gusts (fig. 4). Although we only had 3 percent of the Canyon Creek drainage treated, the treatments modified the fire behavior, changing the treated sites from a passive crown fire to a surface fire (fig. 8). The northwest corner of the area burned in the simulations included a small area where an active crown fire occurred (typically a very rare event) before the treatments. With the treatments in place, this active crown fire fragmented this area; thus, the treatments created a diversity of forest structures minimizing the amount of active crown fire. Flame lengths of fires simulated before the treatments were 1.5 to 3.0 m while fires simulated after the treatments had flame lengths 0.3 to 0.6 m in the areas treated (fig. 9).

DISCUSSION

Applying the free selection system to restore Inland Northwest moist forests

The objective was to apply treatments in Canyon Creek to enhance research opportunities at PREF and to create forest conditions resilient to both native and introduced disturbances. We wanted to create heterogeneous forest structures and compositions and reduce the amount of

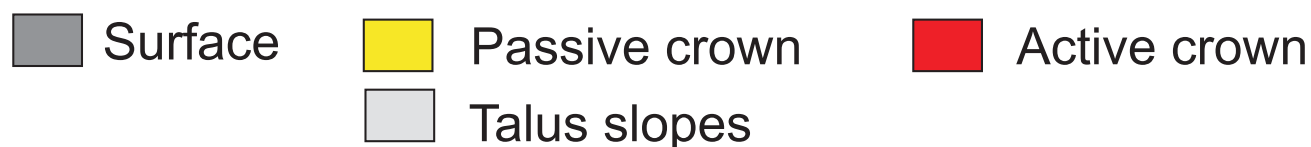


Figure 8— Fire activity before and after treatment from the FlamMap simulation. A FlamMap simulation evaluates individual characteristics within each pixel independently at a given point in time and determines its potential fire activity. The arrows show treatment locations. Crown fire was abundant across the landscape without treated areas (orange and yellow). Fire activity changed from a crown fire to a surface fire within treated sites (grey).

suppressed trees, mistletoe infected western larch, dead and dying western white pine, and ladder fuels. Moreover, we wanted to rejuvenate a stagnate forest and increase the presence of western white pine and other early-seral species. The key to our implementation was using our vision, future accessibility, current conditions, and desired future

conditions to develop silvicultural prescriptions (Graham and Jain 2005). Through adaptive management, we will determine the timing, extent, and intensity of the next entry as we continue to monitor the development of the forests in Canyon Creek to determine if they fulfill the conditions we outlined in our objective. Most importantly, the condi-

tions we created and those that develop will offer abundant research opportunities now and in the future.

Free selection was adaptable at incorporating new information such as adding findings from Jain et al. (2002, 2004)

that describe canopy opening thresholds for establishment (40 percent), competitive advantage (55 percent) and free-to-grow (92 percent) (table 1). Incorporating these western white pine canopy opening thresholds not only facilitated regeneration opportunities but also will be useful for de-

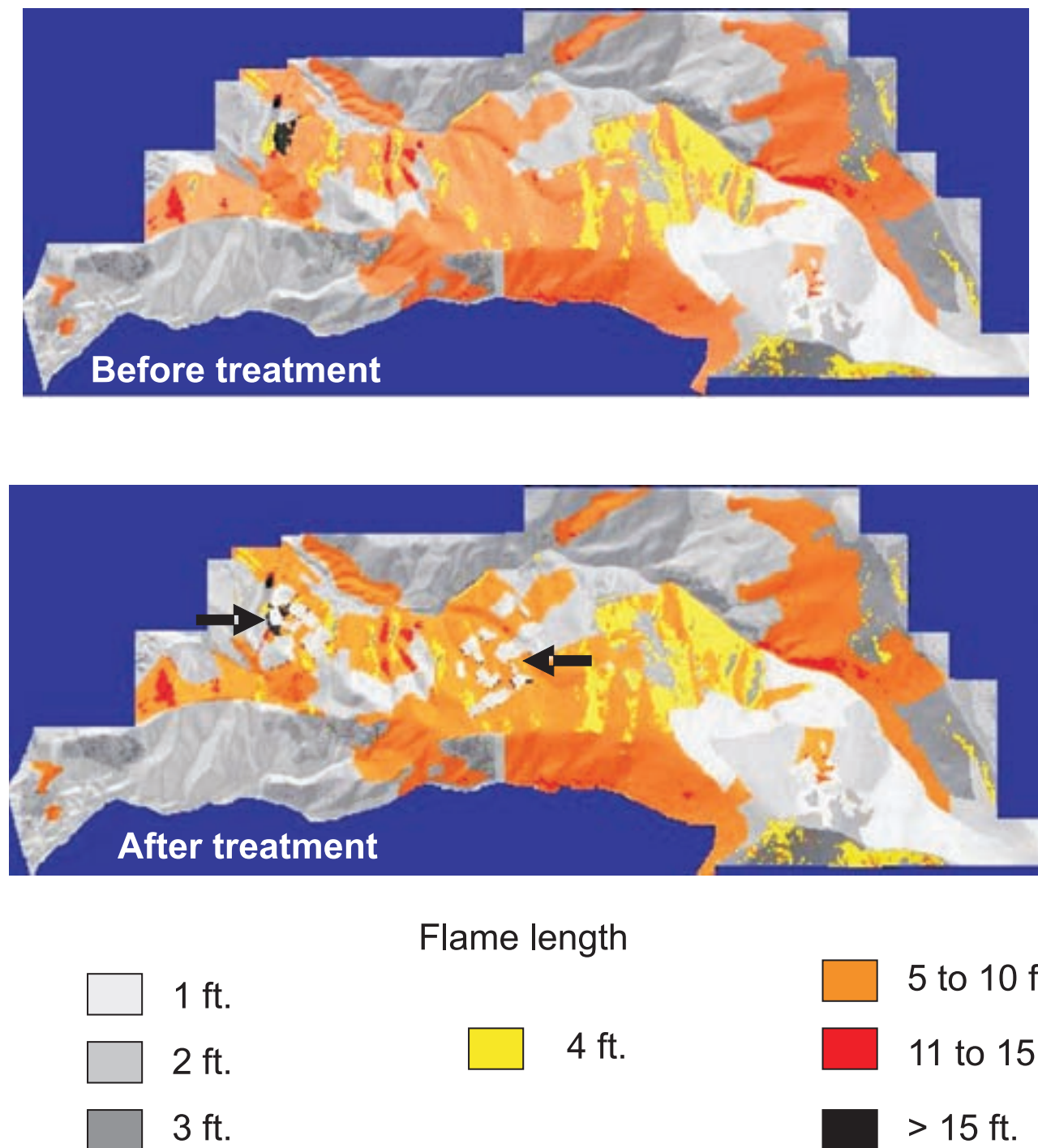


Figure 9—Flame length before and after treatment from the FlamMap simulation. A FlamMap simulation evaluates individual characteristics within a particular pixel independently and determines its potential flame length. The areas prior to treatment had potential flame lengths of 1.5 through 3 m and on the northwest corner had > 15 m flame lengths. After treatments, the flame lengths were 0.3 m (arrows).

termining residual canopy densities in future entries (e.g., overstory removal, cleanings, and weedings) to insure the vegetation of Canyon Creek develops as desired--as would be necessary in all selection systems (Marquis 1978, Meyer 1943). In addition, applying the thresholds created a diversity of forest conditions that reflect environments created by small-scale disturbances (e.g., weather, wind, fire).

The results from using free selection created a fine scale mosaic (0.05 ha) that contained a diversity in canopy cover (fig. 5), surface vegetation composition and structure, coarse woody debris amounts and structure, and a variety of forest floor conditions, ranging from undisturbed to masticated to burned with some mineral soil exposed. These conditions should produce abundant natural regeneration of groups and clumps of forbs, shrubs, and trees of all species in addition to the artificial regeneration of western white pine, ponderosa pine, western larch, and western redcedar planted in micro-sites to ensure their establishment and abundance. This mosaic would develop in response to the wide range of operational environments reflected in this first entry of a selection system.

Grapple piling versus mastication

Mastication costs appeared to be very high (\$300/hr), leading to the conclusion that it is more expensive than typical slash treatments (prescribed fire or grapple piling). However, this is not always the case and mastication is a superb slash treatment alternative in areas where trees are not resistant to fire. In addition, a masticator can remove undesirable standing material, treat surface fuels, and prepare planting or seed germination sites in one operation. In contrast, grapple piling, or for that matter any kind of slash piling, requires slashing of unwanted standing and surface materials, piling the slash, and the subsequent burning of the piles, and often months to 10's of months can transpire between operations, delaying planting. In addition, by not removing both fine and coarse woody debris from the site, we conserve the organic and nutrient materials. However there are disadvantages to masticating slash and a major one is that slash is not removed but it is redistributed toward the fine fuel classes thus temporarily increasing the fire hazard (2 to 3 years) (Graham and Jain 2007).

Wildfire simulation

Although FARSITE and FlamMap are easy to use, calibration of the models requires people who are knowledgeable in fire behavior. Moreover, preparing the input data can be

labor intensive, requiring GIS skills and appropriate data available that reflects the variability across a landscape, can be a major limitation. Although the models can simulate fires in forests with high amounts of variation, the data are rarely available with sufficient resolution to take advantage of these abilities. For example, although we conducted the simulations using 15 m cell size, the data we used could only be summarized to a "stand" with all of the cells within a stand having the same values. Therefore, rather than identifying variation in fire progression and behavior within a stand, we were simulating the variability in fire progression and behavior across stands. We did have LiDAR data (30 cm resolution) that could create a very complex landscape but did not have enough spatially explicit sample plots to take full advantage of the LiDAR data.

We determined that the FFE-FVS and 1st quartile CBH created similar results compared to the results from simulating the median or 3rd quartile CBH (fig. 6). Based on these results, we selected the 1st quartile crown base height to conduct our evaluation. These results indicate that determining CBH from associated attributes is an alternative to depending on only expert knowledge to determine CBH. To obtain a quartile from the original data, we used the tree list from FFE-FVS using the TREELIST keyword (Dixon 2002) and then a spreadsheet to calculate the different CBH quartiles. The default minimum height threshold for calculating CBH or canopy bulk density in FFE-FVS is 2 m which excludes all shorter trees. Because we had so many short trees <0.3 m, we used the CANCALC keyword and adjusted the minimum height to 0.3 m. Thus, CBH determined by FFE-FVS produced simulated fire results similar to the 1st quartile CBH. This finding emphasizes that when using a model, a basic understanding is required of how it works and that further evaluation of the minimum height CANCALC keyword and its impact on the FFE-FVS calculation of CBH is warranted.

Wildfire suppression opportunities due to treatment juxtaposition

In this study, we had two objectives concerning the placement of the treatments. One was to increase the heterogeneity of forest structure so that if a wildfire were to occur, the fire would create a heterogeneous burn severity (what a fire leaves behind). We also wanted to alter potential fire behavior to provide opportunities for fire suppression. Based on our simulations and information on the relation between forest structure and burn severity, we believe we

achieved both objectives within the short-term (estimate of 30 years) (Jain and Graham 2007, Jain et al. 2006, Graham et al. 2004, Peterson et al. 2005, Scott and Reinhardt 2001, Stephens 1998, Stephens and Moghaddas 2005a, 2005b, Stratton 2004). Although our treatment units were small, the juxtaposition of the units (partially overlapped) contributed to our success in modifying fire behavior (Finney 2001). If the treatments were overlapped as the fire moved through, the rate of spread slowed and created delays while the fire flanked around the unit sides (fig. 7).

Under the 1967 weather, the presence of the treatments did not decrease the ultimate size of the fires substantially (table 6); however, the window of opportunity to connect the fuel breaks safely, became substantially larger. When a fire ignition occurred in the stream bottom, the juxtaposition of the treatments would provide an opportunity for fire crews to connect the fuel breaks and increase the likelihood of slowing or stopping the fire. Although the mid-slope ignition started where only a few treatment units affected fire growth, the treatments were effective at reducing the extent of the fire run. Moreover, the backing and flanking fires through the west side of the eastern treatments would allow suppression forces to have a greater opportunity to take safe action on the east flank and connect the treated fuel breaks. With the ridge top ignition, although more susceptible to the winds, both treatment blocks did slow fire growth. Moreover, when the simulated fires burned through these units using FlamMap, the flame lengths were 0.3 m, providing excellent suppression opportunities. This is because fires with flame lengths <1.2 m can generally be attacked at the back or flank of the fire directly by firefighters using hand tools and handlines thereby increasing the ability to actively manage the wildfire (National Interagency Fire Center, 2005).

We were pleased that the treatments were located within high fire risk areas (crown fire potential). We purposely located cuttings in locations to remove suppressed or diseased trees or both and the results from our FlamMap simulations show the same areas were also high risk for crown fire. Although our units were small and contained clumps of trees with interlocking crowns, they influenced burn severity. This impact is most likely related to the high (>5 m) canopy base heights and the slash treatments (surface fuel reductions). In addition, the high diversity of forest floor conditions we created (e.g., burned surfaces, masticated

woody debris, exposed mineral soil, untreated areas) would result in a multitude of different soil burn severities (Jain et al. 2006). This diversity will provide many opportunities for seed germination (new and seed stored in the duff) and tree planting, creating a forest reflecting characteristics similar to historical conditions (Marshall 1928). If such fires were to occur, they would kill many western white pine, western hemlock, and western redcedar, but many would also survive, as would the large ponderosa pine and western larch. Although there is always, a risk of wildfire in Canyon Creek, the treatments through our simulation analysis would provide opportunities for suppression and the remaining forest structures would provide vectors for the forest to recover.

Challenges toward implementing the free selection silvicultural system

Study implementation offered several challenges to Forest Service District personnel, beginning with the Environmental Assessment (EA). The objective of any harvesting activity on an experimental forest is to provide conditions for research. Therefore, there were only two alternatives to evaluate in the EA; either conduct the study using the study plan (harvest) or do not conduct the study (do not harvest). The silvicultural prescriptions did not describe specific stand metrics such as opening size, residual canopy cover, and species compositions, making it difficult for the wildlife biologist, hydrologist, botanist, and other specialists to make informed judgments as to the impacts the treatments would have on other resources. To alleviate this challenge, we spent considerable time in the field with the EA Team Leader and specialists discussing the silvicultural prescription, and used many examples showing what Canyon Creek would look like after the treatments. Even though there were challenges in developing the EA and implementing the study, through our collaborative efforts we found solutions to the issues as they arose. Through these collaborative efforts, other interested groups did not appeal the decision, and implementation is progressing.

During completion, the arrangement of the units and the tree-marking plan was also challenging. This required many hours of reconnaissance to find all the locations to fulfill these objectives. When implementing the marking guides for the free selection, we found that persons with a forestry background (many years of experience or a degree in forestry) and with considerable tree-marking experience were able to grasp the concepts quickly. On the other hand,

people who had tree-marking experience but non-forestry backgrounds who were accustomed to marking guides such as traditional seed-tree, shelterwood, or thinning prescriptions often described by basal area, stand density index, tree spacing, or other metrics had difficulty understanding the concepts used in free selection.

The objective was to create a diversity of forest floor conditions (planting and seed germination sites), requiring a variety of slash treatments, often within one treatment unit. For example, unit 7 (≈ 5 ha) had grapple piling, mastication, and prescribed fire distributed throughout the unit. To coordinate our efforts, we used a pamphlet containing photographs of each treatment area within a harvest unit and identified the research objectives, slash treatments, and planting needs associated with the treatment area. This enabled personnel from Research and National Forest Systems to work from a common understanding. The guide provided a way to document changes and points of discussion as the study progressed.

Planting

Two of the goals when planting were to have variation in density and species. We achieved variation in planting density by having non-plantable sites, incorporating normal expected mortality, and using differences in overstory canopy cover to guide the planting concentration. We achieved the variation in species as a function of overstory canopy cover and composition. With the cooperation of the contractor and crew leader, who understood which species favor a particular environment (e.g. western larch under low canopy cover versus western redcedar under high canopy cover) we were able to plant the desired species in the appropriate locations.

To accomplish the reforestation objectives, the District used a tree spacing (3×3 m) requirement, with either site preparation, full planting, or no site preparation interplanting for most of the units, with per hectare cost being the same for each site preparation type. For the units that required a little more finesse or thought as to where to plant trees (i.e. units 1, 2, and 24) at varying densities, it was more appropriate to pay by the hour. These units required tree planters to look up and assess the canopy opening coverage and size (minimum 50 percent canopy opening and 0.1 ha in size) to ensure competitive advantage was attainable. Although the cost per tree was higher (50 percent) when paying by the hour, the cost per hectare planted,

was lower (35 percent). Although it is not always easy to make comparisons, due to size and number of openings and unit size, a crew has to walk through, based on our experience during implementation, the type of planting used in this study area can be a viable option for future applications. Moreover, the planting was successful due to a clear understanding of the objectives and good communication and trust between the District's Contracting Officer's Representative and the Contractor's crew leader.

CONCLUSION

Baker (1934) suggested that silvicultural systems are often confused with rigidity, but they are flexible based on management objectives and the forest conditions. Gifford (1902) and Schlich (1904) developed silvicultural strategies intended for harvesting and managing forests containing irregular structures and complex compositions. Therefore, the free selection concept we have developed and applied for restoring the moist forests is not new, but rather an extension of the many ideas expressed by other silviculturists. Ultimately, what we implement, test, accomplish, and display at PREF may or may not be appropriate in other places, different times, or to meet other objectives. However, the concepts, ideas, results, and thought processes we used and discovered when implementing the study are very applicable to other forest management and silvicultural decisions in other places with other objectives. The practice of silviculture has a proud heritage of being responsive to the needs of society. However, inherent in all of us is the resistance to change. Therefore, the challenge regarding the practice of silviculture is to develop methods to meet today's (wildlife, fire hazard) and tomorrow's (carbon sequestration, climate change) societal objectives. In meeting these challenges, some of our tried and true methods, systems, and procedures may need to be changed, which can be difficult. However, we have a long history and knowledge base from which we can draw on and this study added to this knowledge. The mutual learning by all involved with the study and their open mindedness of trying something different contributed to the implementation of this study's success.

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Implementing Northern Goshawk Habitat Management in Southwestern Forests: A Template for Restoring Fire-Adapted Forest Ecosystems

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ABSTRACT

Developing and displaying forest structural targets are crucial for sustaining the habitats of the northern goshawk, a sensitive species in Southwestern forests. These structural targets were described in Management Recommendations for the Northern Goshawk in the Southwestern United States (MRNG) (Reynolds, et al., 1992). The MRNG were developed in a unique food-web approach that identified desired forest conditions for the goshawk. These desired conditions were based on syntheses of the habitats of both goshawks and the diverse suite of plants and animals in their food web. Not surprisingly, implementing these structural targets results in forests restored to conditions similar to our understanding of pre-European settlement forests.

Silviculturists are responsible for:

- 1) developing a silvicultural system,
- 2) documenting the system in a silvicultural prescription, and
- 3) establishing protocols for monitoring the development and perpetuation of the desired forest conditions.

We present spreadsheet tools to aid silviculturists with diagnosis and development of silvicultural prescriptions and tree-marking guides that produce the desired uneven-aged, heterogeneous forest structures that comprised historic forests, as well as goshawk habitats, in Southwestern ponderosa pine landscapes. These tools incorporated fire behavior and fuel considerations, rendering them appropriate for developing prescriptions for other management objectives (e.g., restoration of fire-adapted forest ecosystems) and biophysical situations.

Key words: forest management, marking guides, northern goshawk, reference conditions, restoration, silvicultural prescriptions, structural targets

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HISTORY OF GOSHAWK HABITAT MANAGEMENT IN THE SOUTHWEST

The northern goshawk (*Accipiter gentilis*) is a large forest raptor that typically occurs in old forests and feeds on birds and mammals such as tree and ground squirrels, rabbits, jays, woodpeckers, and grouse. Many of these prey species are more abundant in older forests (Squires and Reynolds, 1997). Over the last 30 years, considerable controversy has developed regarding the conservation of the goshawk (Greenwald, et al., 2005). It has been suggested that tree harvests negatively affect goshawks by altering the composition and structure of their habitats (Reynolds, 1971; Reynolds, et al., 1982; Moore and Henny, 1983), and Crocker-Bedford (1990) showed that territory occupancy and reproduction of goshawks in northern Arizona were lowered by tree harvests. These findings initiated a flurry of appeals and litigation aimed at reducing tree harvests in the Southwest and provided the basis for petitions to list the goshawk as threatened (Boyce, et al., 2006). In response to these challenges, the Southwestern Region of the Forest Service (FS) convened a scientific committee of goshawk biologists and silviculturists to develop forest management recommendations to protect and enhance habitats of the goshawk in the southwestern United States (Boyce, et al., 2006). The committee's recommendations were published in the Management Recommendations for the Northern Goshawk in the Southwestern United States (hereafter MRNG) (Reynolds, et al., 1992). In 1996, management standards and guides (hereafter "guidelines") were developed by the Southwestern Region based on the MRNG, which appeared in a Final Environmental Statement for amending National Forest Management Plans in Arizona and New Mexico (Boyce, et al., 2006). In 1996, a Record of Decision (ROD) amended the Forest Management Plans to implement the MRNG in all National Forests in the Southwest (USDA Forest Service 1996). The Plan Amendment directed National Forests to apply the goshawk guidelines in almost all vegetation manipulation projects (timber sales and fuels management treatments). An initial strategy was developed in 1993 (Higgins, 2003) for implementing the guidelines on the Kaibab National Forest (KNF), based on the concepts and desired forest conditions described in the MRNG. Despite this, there continued to be uncertainty in forest management applied in the Southwest regarding the MRNG desired forest conditions (Boyce, et al., 2006).

Here we present tools to help managers with the process of prescription development, implementation layout, and monitoring. These tools facilitate understanding and application of the concepts outlined in the MRNG and meeting the guideline specifications outlined in the ROD.

GOSHAWK HABITAT MANAGEMENT: FOREST ECOLOGY

The MRNG described sets of desired goshawk breeding habitats for Southwestern forests based on syntheses of the life history and habitats of goshawks and 14 important Southwestern goshawk prey species and the ecology of dominant overstory and understory plants in each forest type (Reynolds, et al., 1992; Reynolds, et al., 2006). The MRNG described habitats at three spatial scales:

- (1) a 12-ha (30-acre) nest area with relatively large trees and high canopy cover relative to forest type,
- (2) a 168-ha (420-acre) post-fledging family area (PFA) surrounding the nest area providing a transition from forest structures similar to those in nest areas to structures suitable for goshawk foraging (see below), and
- (3) a 2,160-ha (5,400-acre) foraging area surrounding the PFA comprised of structures suited for goshawk foraging and mosaics of prey habitats (Reynolds, et al., 1992) (Figures 1-2).

Goshawk foraging habitat included subcanopy flight space (lifted crowns), abundant tree perches, and available prey. Prey habitats included highly interspersed groups of mid-aged (140+ years) and old trees (230+ years) with interlocking tree crowns (for tree squirrels), small openings (typically <0.5 ha, 1.25 acre) around tree groups (for ground squirrels, rabbits, birds), decadent reserve trees and snags (for woodpeckers, tree squirrels), logs (for ground squirrels, rabbits, woodpeckers), and wood debris (for ground squirrels, rabbits, birds) (Reynolds, et al., 1992). Mid-aged, mature, and old forests provided the best habitat for most prey species, but small openings were also important (Reynolds, et al., 1992). Old forests also provide subcanopy flight space suited for goshawk hunting. Predator and prey habitats were synthesized into desired landscapes so that the distribution and interspersed of habitats optimized their availability to territorial goshawks and their prey with an objective of maximizing goshawk occupancy, reproduction, and survival (Reynolds, et al., 2006b). To

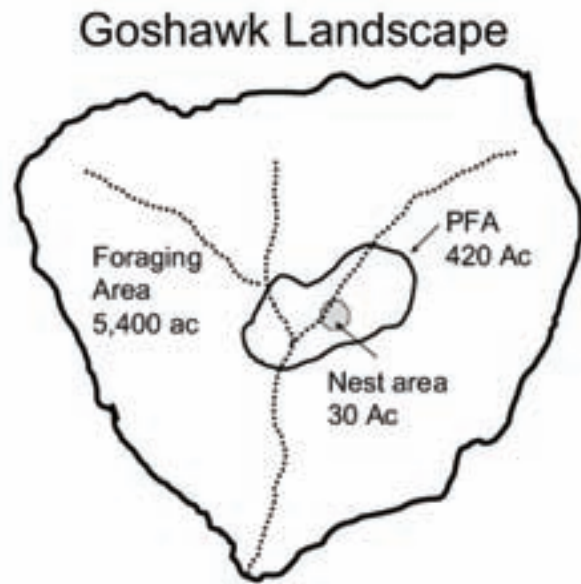


Figure 1. Conceptualized Goshawk Home Range

assure that the specific desired habitats were within the biophysical capabilities of a forest and could therefore be attained and sustained, the MRNG developed specific desired conditions for Southwestern ponderosa pine, mixed conifer, and spruce/fir forests by incorporating local and regional variations in vegetation composition and structure, tree development rates and longevity, natural disturbances and succession, the sizes, shapes, juxtapositions of plant aggregations, and site capabilities (Reynolds, et al., 1992, Reynolds, et al., 1996, Long and Smith 2000, Reynolds, et al., 2006).

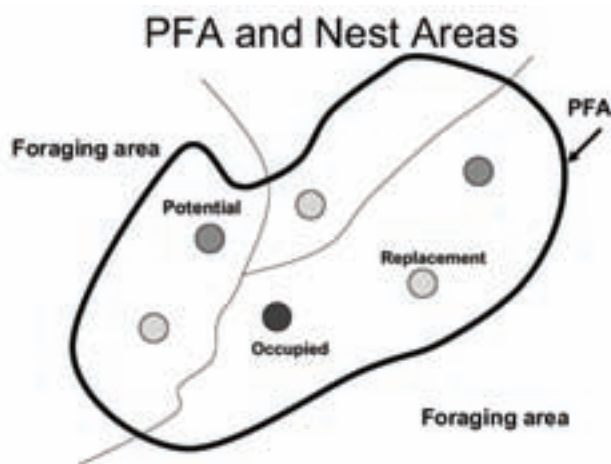


Figure 2. Conceptualized Goshawk PFA and Nest Areas

The intent of the MRNG was to maintain goshawk reproduction by sustaining predator and prey habitats on each home range and used vegetative structural stages (VSS) to describe the desired vegetation. VSS integrates the stages that vegetation complexes (e.g., composition, structure) go through beginning with regeneration through maturation and mortality (Oliver and Larsen 1990, Franklin, et al., 2002, Thomas et al. 1979). The MRNG defines 6 vegetation structural stages from forest initiation (VSS 1) to old forest (VSS 6) (Figure 3). Due to forest dynamics (e.g., resulting from vegetation growth, succession, natural disturbances) landscapes entirely of old forest (VSS 6) can not be sustained. Therefore the MRNG used maturation rates of Southwestern forests to estimate sustainable landscape proportions of old forest, and recommended that about 10% of a naturally forested landscape be in a grass/forb/shrub stage (VSS 1; to 20 yrs), 10% in the seedling/sapling stage (VSS 2, to 50 yrs), 20% in young forest (VSS 3; to 96 yrs), 20% in mid-aged forest (VSS 4; to 137 yrs), 20% in mature forest (VSS 5; to 183 yrs), and 20% in old forest (VSS 6; to 233+ yrs) (Reynolds, et al., 1992, Appendix 5, Table 1). Excluding grass/forb/shrub, each VSS comprised similarly-aged trees and elements such as live-tree decadence, snags, logs, and vertical and horizontal heterogeneity. Over time (≈ 250 years), the desired landscape consisted of a temporally shifting mosaic of highly interspersed VSS groups in the desired proportions of VSS (Reynolds, et al., 1992, Long and Smith 2000). Sizes of VSS groups approximated the natural (prior to tree harvests and fire suppression) conditions in these forests and contained 2 - 44 trees occupying 0.2-0.3 ha (0.5- 0.75 acre) (Cooper 1961, White 1985, Pearson 1950). At the coarse scale (landscape), ponderosa pine was all-aged, but trees within each group (fine scale) tended to be similar age (Pearson 1950). The desired within-group structure in the mid-aged to old classes (VSS 4-6) included open understories, interlocking tree crowns, abundant large limbs (goshawk hunting perches), and shade for mycorrhizal fungi (food of several prey species) (Reynolds, et al., 1992). Grass, herb, shrub habitat was interspersed in and around groups and provided habitat for rabbits, ground squirrels, and birds (e. g., grouse, doves) (Reynolds, et al., 1992). Additional desired conditions include retention of large live reserve trees within regeneration groups > 0.4 ha (1 acre) in size, snags, downed logs, and woody debris. An ideal MRNG landscape had home ranges spaced at about 4 km (2.5 mile) between centers (Reynolds, et al., 1992, Boal et al. 2001, Reynolds, et al., 2005, Reynolds and Joy 2006). Because

Structural Stages & Desired Proportions

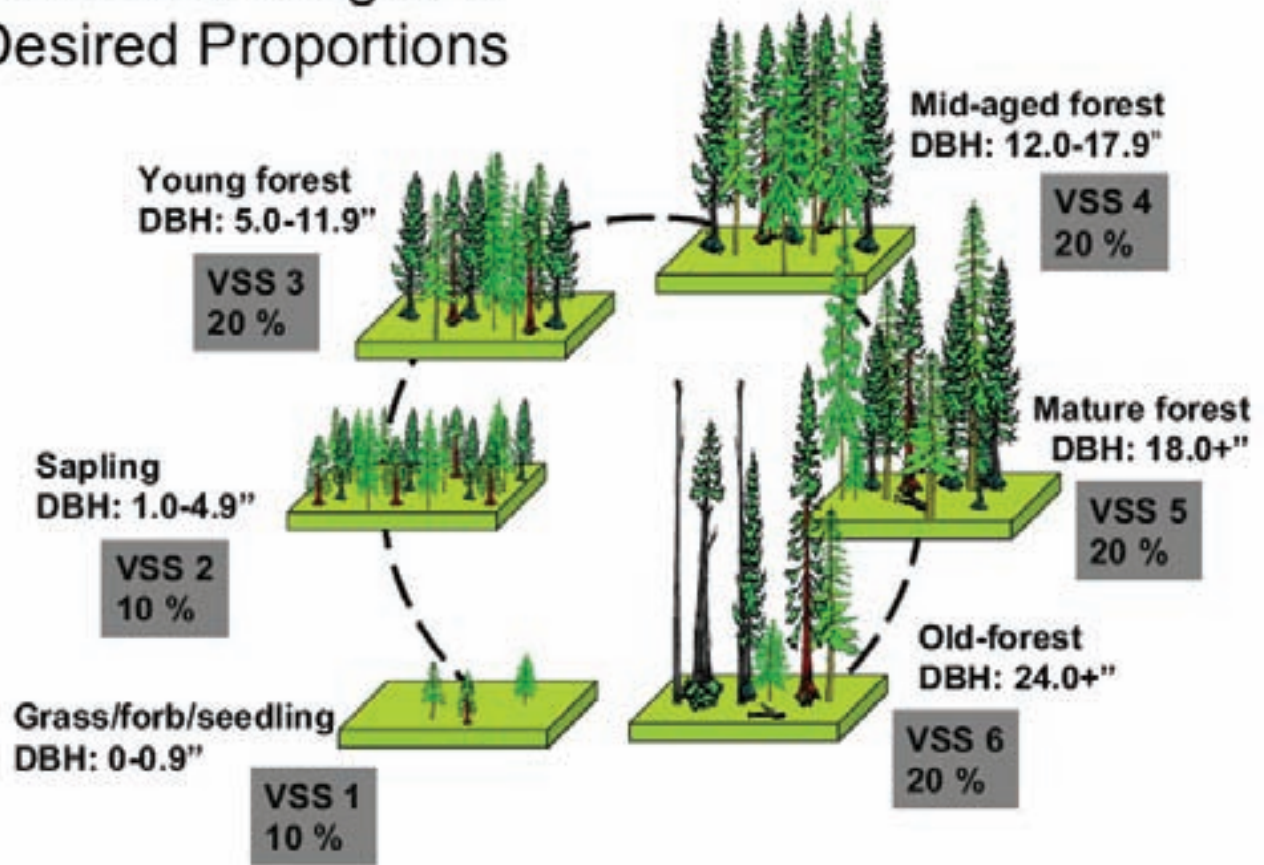


Figure 3. Forest Structural Stages

the desired MRNG condition for Southwestern ponderosa pine forests closely resembles the pre-European settlement (before grazing, fire control and major harvesting) conditions in this forest type (Cooper 1961, White 1985, Pearson 1950, Covington and Moore 1994, Fulé et al. 1997, Long and Smith 2000), implementation of the MRNG is a large step towards ecological restoration.

HISTORIC REFERENCE CONDITIONS: FOREST RESTORATION AND MRNG

Historic reference conditions (HRC) provides a framework for understanding forest conditions, ecological processes, and the historic range of variability prior to extensive European settlement (Moore and others 1999). Figure 4 shows historic forest patterns in the Fort Valley Experimental Forest near Flagstaff, Arizona (Covington, 1997).

Note the similarity of historic forest tree group patterns to the desired conditions described under the MRNG. Such patterns are relevant for developing management strategies in Southwestern ponderosa pine and dry mixed-conifer forest types.

Following are some concepts related to MRNG and ecological restoration of historic reference conditions:

- Implementation of MRNG and other ecological restoration approaches based upon historic conditions will lead to restoration of forest resiliency to disturbances within the historic range of natural variability.
- Strict-sense ecological restoration involves restoration of historic reference conditions relative to forest structure, patterns, species and ecological processes. It is the end-goal of a process that provides for maintenance of desired conditions by re-introduction of or mimicking historic ecosystem processes (fire, insects,

etc.). It will hereafter be referred to as restoring historic reference conditions (RHRC). Some groups and individuals believe most, if not all, management actions on southwestern forests should be designed to restore HRC.

- MRNG is a strategy that provides for development, maintenance and sustainability of the desired forest structural conditions. These desired conditions are based on the habitat requirements of the goshawk and their prey species and are similar to the historic range of variability of natural forest conditions. Implementation of MRNG will result in forest landscape restoration (structure, patterns and species composition within this historic range of variability of natural forest conditions). But MRNG is a management system designed to provide sustainability within a management framework that recognizes multiple resource objectives. Resultant desired future condition of the MRNG strategy will be similar but not necessarily identical to RHRC, and the maintenance strategy may differ between the MRNG and RHRC.

The following are some implementation similarities and differences between MRNG and RHRC:

- Both approaches may utilize tree harvest as a tool for obtaining desired conditions. But some strict-sense RHRC advocates have suggested that desired conditions should be maintained by either prescribed fire

or wildland fire use alone and tree cutting is not warranted. In contrast, MRNG utilizes tree cutting as a scheduled activity for maintaining desired forest conditions. Timber production isn't an objective, but rather is a by-product of maintaining structural composition. In addition, prescribed fire is the preferred method of treating surface fuels but mechanical and hand methods are not excluded. Because of this flexibility, implementing the MRNG appears to be more feasible than maintaining desired forest structures by prescribed fire alone.

- Grouped and single trees are interspaced within a grass/forb/shrub mosaic (see below) under both management approaches. This results in an irregular and discontinuous forest canopy with variable tree densities and discontinuous arrangement of fuels which minimizes the potential for crown fire and facilitates the use of low intensity prescribed fire. Therefore both MRNG and RHRC are applicable for decreasing fuels and crown fire hazard.
- Both RHRC and MRNG focus on development of clumps/groups of trees, surrounding root development zones, natural openings, and replacement trees arranged in fine scale mosaics [e.g., 0.04 to 0.8 ha (0.1 to 2.0 acre) in ponderosa pine forests].
- Depending on the setting, type and current forest condition, implementing the MRNG could tend to create and maintain even-aged groups of trees while

the RHRC would tend to create uneven-aged groups. Nevertheless, depending on the frequency and intensity of treatments utilized by both approaches, a variety of forest compositions and structures could be maintained.

- The RHRC approach focuses on historic locations of tree groups and clumps based upon remnant evidence (stumps and logs representing pre-settlement trees) for determination of the number and location of replacement trees. Such pre-settlement evidence can also be used to develop desired forest structures identified in the MRNG. A key to using both historic reference conditions and the MRNG for treating forests is to use this information along with intrinsic site information such

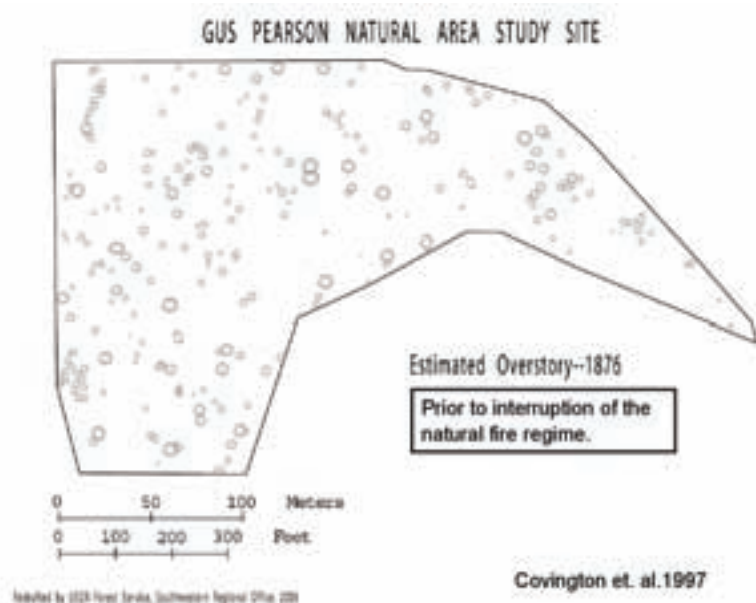


Figure 4. Historic Reference Conditions

Table 1. Forest Density and Structural Targets (Full Stocking Level, Ponderosa Pine PFA)

DESIRED FUTURE CONDITION									
Ponderosa Pine Forest Type - All Sites - Post-Fledging Family Area									
VSS Class	DBH Class	% of Area	Mean DBH	SDI	Group basis Mean Trees/ acre	Basal Area (ft ² /acre)	Canopy Cover (percent)	Per acre basis mean (average acre) Trees/ acre	Basal Area (ft ² /acre)
1	<1	10	0.1	0	203	0		20.3	0
2	1-4.9	10	3	28	193	9		19.3	1
3	5-11.9	20	8.5	105	136	54		27.3	11
4	12-17.9	7	15	137	72	88	60	4.8	6
4	12-17.9	13	15	130	68	83	50	9.0	11
5	18-23.9	20	21	127	39	93	50	7.7	19
6	24+	20	27	135	27	109	50	5.5	22
								93.9	69
===== Note on per acre basis =====								Dq* =	13.11
Reserve trees and grass matrix areas are included in these figures. Trees are closely grouped, allowing for open rooting zone interspaces between groups.								SDI* =	113.7
								*Quadratic mean diameter includes trees >= 1" dbh only.	

Modeling assumptions: Site Index = 78 (Minor), Stand Density Index (SDI) max = 450

Table 2. Forest Density and Structural Targets (Full Stocking Level, Ponderosa Pine FA)

DESIRED FUTURE CONDITION									
Ponderosa Pine Forest Type - All Sites - Foraging Area									
VSS Class	DBH Class	% of Area	Mean DBH	SDI	Group basis Mean Trees/ acre	Basal Area (ft ² /acre)	Canopy Cover (percent)	Per acre basis mean (average acre) Trees/ acre	Basal Area (ft ² /acre)
1	<1	10	0.1	0	203	0		20.3	0
2	1-4.9	10	3	28	193	9		19.3	1
3	5-11.9	20	8.5	105	136	54		27.3	11
4	12-17.9	20	15	89	46	57	40	9.3	11
5	18-23.9	20	21	100	30	73	40	6.1	15
6	24+	20	27	104	21	84	40	4.2	17
								86.5	54
===== Note on per acre basis =====								Dq* =	12.29
Reserve trees and grass matrix areas are included in these figures. Trees are closely grouped, allowing for open rooting zone interspaces between groups.								SDI* =	92.1
								*Quadratic mean diameter includes trees >= 1" dbh only.	

Modeling assumptions: Site Index = 78 (Minor), Stand Density Index (SDI) max = 450

Table 3. Forest Density and Structural Targets (Full Stocking Level, Mixed Conifer PFA)

DESIRED FUTURE CONDITION									
Dry Mixed Conifer Forest Type - All Sites - Post-Fledging Family Area									
VSS	DBH	% of	Mean	Group basis				Per acre basis mean	
Class	Class	Area	DBH	SDI	Mean Trees/ acre	Basal Area (ft ² /acre)	Canopy Cover (percent)	(average acre) Trees/ acre	Basal Area (ft ² /acre)
1	<1	10	0.1	0	203	0		20.3	0
2	1-4.9	10	3	29	200	10		20.0	1
3	5-11.9	20	8.5	140	182	72		36.3	14
4	12-17.9	20	15	145	76	93	60	15.2	19
5	18-23.9	20	21	153	47	112	60	9.3	22
6	24+	20	27	170	34	137	60	6.9	27
===== Note on per acre basis =====								108.0	84
Reserve trees and grass matrix areas are included in these figures. Trees are closely grouped, allowing for open interspaces between groups.								Dq* =	13.23
								SDI* =	137.4
								*Quadratic mean diameter includes trees >= 1" dbh only.	

Modeling assumptions: Site Index = 85 (Minor), Stand Density Index (SDI) max = 486, Species composition = 75% ponderosa pine, 25% Douglas-fir (actual species composition will be project-specific)

Table 4. Forest Density and Structural Targets (Full Stocking Level, Mixed Conifer FA)

DESIRED FUTURE CONDITION									
Dry Mixed Conifer Forest Type - All Sites - Foraging Area									
VSS	DBH	% of	Mean	Group basis				Per acre basis mean	
Class	Class	Area	DBH	SDI	Mean Trees/ acre	Basal Area (ft ² /acre)	Canopy Cover (percent)	(average acre) Trees/ acre	Basal Area (ft ² /acre)
1	<1	10	0.1	0	203	0		20.3	0
2	1-4.9	10	3	29	200	10		20.0	1
3	5-11.9	20	8.5	140	182	72		36.3	14
4	12-17.9	7	15	145	76	93	60	5.1	6
4	12-17.9	13	15	95	50	61	40	6.6	8
5	18-23.9	20	21	123	37	90	50	7.5	18
6	24+	20	27	149	30	120	60	6.0	23
===== Note on per acre basis =====								101.8	72
Reserve trees and grass matrix areas are included in these figures. Trees are closely grouped, allowing for open rooting zone interspaces between groups.								Dq* =	12.69
								SDI* =	119.6
								*Quadratic mean diameter includes trees >= 1" dbh only.	

Modeling assumptions: Site Index = 85 (Minor), Stand Density Index (SDI) max = 486, Species composition = 75% ponderosa pine, 25% Douglas-fir (actual species composition will be project-specific)

as but not limited to potential vegetation type, soil type, climate, successional stage, insect and disease conditions, and fire regime condition class. This information should be incorporated in all silvicultural applications utilizing fundamental knowledge of tree silvics, growth patterns etc. and displayed and quantified where appropriate using metrics such as stand density index, site index, etc.

SILVICULTURAL METHODS AND SYSTEMS

Desired forest conditions for the goshawk food web are described in the MRNG and goshawk guidelines. Attainment and maintenance of these forest conditions can be achieved by applying appropriate silvicultural systems documented in a silvicultural prescription (Graham and Jain 2004). In general the desired forest conditions and suggested forest dynamics inherent within them can be created and maintained using selection systems (Pearson 1950, Long and Smith 2000, Graham and others 2007). However, even though selection systems and silvicultural methods necessary to create and maintain goshawk habitat are slightly different than those aimed at producing commercial timber, the principles and concepts are relevant to selection systems for sustaining goshawk habitat, such as:

- Maintenance of high-forest cover on landscapes at all times (forests and stands have no origin or endpoint).
- Forest regeneration is established following each management entry (cutting cycle) on a scheduled basis.
- Structural regulation and sustainability is provided for at the local and landscape scale and smaller spatial scales (stand, group, clump) are integral to attaining desired landscape conditions (Long and Smith 2000).

The MRNG focuses on creating and sustaining a patchy forest of highly interspersed structural stages ranging from regeneration to old forest throughout a goshawk territory ($\approx$ 2,400 ha, 6,000 acre landscape). These variable density tree groups can be comprised of 2-40 trees occupying up to 1.6 ha (0.1 to 4 acres) but tree groups are generally less than 0.3 ha (0.75 acres). Sufficient growing space between tree groups is required for producing the desired large trees and the canopy densities within tree groups. Depending

on the biophysical setting and existing forest structure, occasional openings to 1.6 ha (4 acres) in size may be created, but each regeneration group larger than 0.4 ha (1 acre) must provide for retention of 3-5 mature reserve trees per 0.4 ha (1 acre). The discontinuous, irregular forest structural stage distribution is similar to that maintained by the mature selection system described by Pearson (1950) or the grade selection system described by Meyer (1934). Meyer (1961) perfected the use of “q” defined diameter distributions for sustaining the production of timber. In fact the structural stage distribution defined in the MRNG can be represented by a diameter distribution defined by a “q” of 1.16 (2 inch diameter class basis). Perhaps “individual/clump or clump selection” are appropriate terms that describe the silvicultural system used to sustain goshawk habitat. As Baker (1934) stated there are not four or five silvicultural systems but general classifications that contain a myriad of systems that can be developed to sustain forest conditions to meet management objectives.

The Forest Vegetation Simulator (FVS) is an excellent tool for planning and displaying either group selection or individual tree selection systems (Dixon 2002). It can be calibrated to local conditions and variants are available for most forests. FVS’s default individual tree selection and group selection options readily project diameter distributions and tree densities through time; however, FVS has many other options for designing and projecting stand treatments. For example, trees can be selected to leave or remove by species, diameter, height, canopy cover, stand density index, basal area, trees/unit area, crown class, and from inventory plots to name a few. Stand dynamics (i.e., regeneration, growth, death) projected by FVS are readily displayed by the Stand Visualization System (SVS). Because the automatic default in SVS randomly distributes trees, the tree patterns and their juxtaposition will most likely differ from the desired goshawk forest structures displayed in the MRNG. Nevertheless, mapped tree locations or estimated tree locations can be input and used by SVS to display the group/clump distribution inherent to the MRNG. In addition, individual trees can be selected by location within the stand to be left or removed or trees can be selected by characteristics (e.g., diameter, species height etc.) This information can be used to schedule treatments in FVS (see Graham and others 2007, Appendix A for an example of using FVS and SVS for projecting and displaying complex stand structures). In addition the fire and fuel extension

of FVS can readily be used to display fire behavior metrics such as torching and crowning index.

PROJECT PLANNING

Regional management priorities

“The restoration of the ecological functionality of Southwestern forests and grasslands, with primary emphasis on fire adapted systems, has been identified as the central priority for this Region... Nationally, the Forest Service recently decided that each Region would develop a 5-year, integrated regional strategy outlining how they plan to address designing land management programs that achieve resource-specific objectives and work to create a landscape pattern that effectively lessens the likelihood of large wildland fires.” (USDA Forest Service, Southwestern Region, 2004).

Management direction linking this central priority to implementation of the MRNG was stated in a transmittal letter from the Southwestern Regional Forester: “...we have come to understand there is a high level of compatibility between research findings for northern goshawk habitat, ecological restoration, sustainability, and the restoration of fire adapted ecosystems.” (Forsgren, 2006).

Forest plan standards: Incorporation of MRNG

In the Southwestern Region, Land and Resource Management Plans (LRMPs) provide the management direction and Standards and Guidelines for the Mexican Spotted Owl (MSO) and other Threatened & Endangered species as well as the northern goshawk (MRNG). Under current Forest LRMPs, most forest types that are not classed as restricted habitat under MSO or management-limited by other specified requirements will be managed under the MRNG strategy. Additionally, 75-80% (depending upon geographic area) of the MSO restricted forest types may be managed under the MRNG strategy, but this is not required under Forest LRMPs. Cutting trees greater than 60 cm $\approx$ 24 inch diameter at breast height (dbh) is prohibited in Mexican spotted owl restricted habitat (mixed conifer and pine/oak forest types). Consultation with U.S. Department of Interior, Fish and Wildlife Service must be initiated to resolve conflicts when activities conducted in conformance with the MRNG may adversely affect other threatened, endangered, or sensitive species or may conflict with other established recovery plans or conservation agreements.

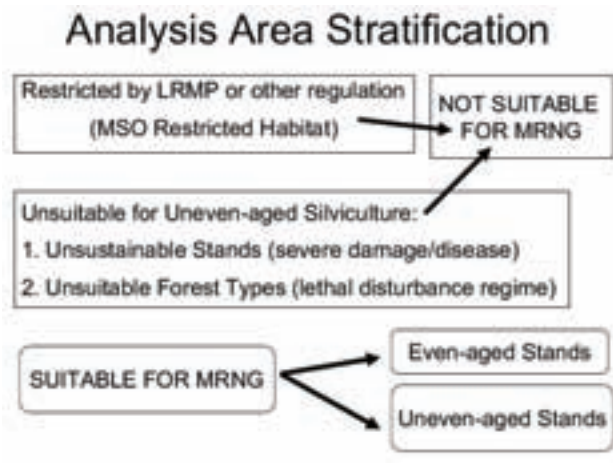


Figure 5. MRNG Forest Stratification Criteria

Project planning is initiated by stratification of an analysis area to classify both plan-level (MSO, other special areas), and bio-physical (forest types, structure, condition, etc.) management stratum. It is critical to identify existing forest conditions that are unsuitable for uneven-aged silvicultural treatments, and to recommend other management alternatives. Examples of unsuitable forest conditions include:

- (1) those affected by severe insect, disease, or other damage;
- (2) forest types that may experience windthrow damage if openings are created within the stand; and
- (3) inappropriate forest types (characteristic lethal fire regime with ecologically-adapted even-aged forest development).

Some forest conditions may be permanently unsuitable for uneven-aged silvicultural treatments (i.e. characteristic lethal fire regime), while other conditions (severe insect, disease, or other damage) may be more appropriately managed with even-aged silvicultural methods during the initial treatment, but in the long-term these areas are suitable for uneven-aged management strategies (planning analysis and stratification flowchart, Figure 5). There are also multiple other decision criteria for selection of appropriate silvicultural systems and methods. Some of the most important are listed below:

- Biophysical setting and/or potential vegetation
- Current condition relevant to both short- and long-term desired condition
- Wildfire hazard/fire regime condition class
- Dwarf mistletoe/other insect-disease conditions

Uneven Aged Foraging Area Summary

Total Stratum

	Number	Acres
Stands in Stratum	87	6853

Based upon Proportional Acres

	Acres
Percent of Stratum in Commercial Conifer	92 5971

Acres

Percent of Stratum in Other Species	8 582

Commercial Conifer in Stratum

5971 Acres

(Averages)

VSS	Canopy Cover	Merch Volume - Board Feet	Merch Volume - Cubic Feet	Trees Per Acre	Basal Area	SDI	Proportional Acres	% of Total Comm. Conifer Acres
1	2	193	51	313	4	9	212	4
2	23	1744	400	2496	37	126	174	3
3	36	3345	1040	928	82	195	1033	17
4	46	6861	1475	722	84	178	1293	22
5	51	11370	2255	655	93	189	1938	32
6	54	18532	3300	691	108	210	1321	22

Figure 6. Plot by Plot Spreadsheet Summary, Example Project Analysis Area, Current Condition

- Operability, logging systems, economics, and feasibility of treating
- Regeneration of desired species
- Snags and woody debris

A specific need to manage a stand outside goshawk guidelines must be identified and discussed by an interdisciplinary planning team during project development. Managing outside the guidelines requires a site specific Forest Plan amendment for Southwestern Region National Forests.

Inventory and analysis

The MRNG recommends the targeted vegetative structural stages (VSS, 6 stages) proportions be managed with juxtapositions such that there is a high degree of interspersions of VSS at the sub-stand level. This implies an uneven-aged forest, regulated to provide landscape (2,400 ha $\approx$ 6,000 acre goshawk territory home range) sustainability.

Sub-stand structural classification of uneven-aged forests poses a challenge relative to stand inventory and analysis for prescription development and planning. Generally, structural stages are determined by which tree component represents the highest basal area. However, even though a big tree may dominate a group's basal area, a predominance of grass/forb/shrub, seedlings, or young forest may determine the VSS (i.e., VSS 1-grass/forb/shrub, VSS 2-

seedling/sapling). "The area" in which VSS is determined can be as small as a group of 2 to 3 trees up to stand-sized areas if little diversity is present. As such, depending on the locale, inventory methods, and data needs, it may be more appropriate to classify the vegetation to VSS before quantifying the metrics (see Graham and others, (2007) for methods of classifying VSS). Typical stand exam data is collected at the plot scale and subsequently summarized to the stand. This homogenizes sub-stand structural variability such as VSS distributions and tree group characteristics, etc. A stand-level summary is adequate for one and two-storied stands that will be treated to initiate conversion to future uneven-aged structure, but is

inadequate for stands currently composed of three or more VSS classes. One method for analysis of uneven-aged forest sub-stand characteristics is to classify and summarize each sample plot independently. In this way, all uneven-aged stands in the project area can be combined into a stratum and individual plots within a stratum can then be compiled to generate stratum statistics. This method may not perfectly determine VSS distribution but it does provide a sub-stand summary of forest characteristics sufficient for planning and prescription development. The use of fixed-area rather than variable-area sample plots will also provide a better representation of local structural characteristics. In addition, stand maps could be obtained for a few of these fixed area plots and displayed using both FVS and SVS, an excellent communication tool. Other inventory methods such as transects can also provide an overall summary of VSS distribution [see Graham and others (2007) for methods for quantifying and displaying VSS distributions].

The Southwestern Region has developed a methodology for project planning analysis. An analysis area is stratified according to previously-described criteria: forest types by structure/condition (e.g. even-aged/two-aged, uneven-aged, unsustainable stands) are classified and grouped into separate stratum. Standard stand-level summaries are computed to determine current condition for the even-aged/two-aged and unsustainable stands and summarized by stratum. The following process is used to develop current

VSS 5		Uneven Aged Foraging Area		
	Average	Variation		
		Minimum	Maximum	Range
Canopy Cover	51	15	82	67
Mech Volume - Board Feet	11370	1777	50054	48287
Mech Volume - Cubic Feet	2255	365	9548	9183
Trees per Acre	655	20	9733	9713
Basal Area	93	15	334	319
SDI	189	40	600	559

Species Composition (percentage) by Basal Area	
PP	95.38
DF	1.06
WF	0.00
WP	0.00
BS	0.00
ES	0.00
CB	0.00
WS	0.00
LM	0.00
OA	0.56
OH	0.61
CO	0.00
PI	0.91
RM	0.00
BC	0.00
FE	0.00
JU	1.19
AS	0.29
	100.00

Figure 7. Plot by Plot Spreadsheet VSS Classification Statistics, Example

condition statistics for uneven-aged stratum (USDA Forest Service. 2007b):

- Stands are sorted and grouped by goshawk habitat management emphasis (PFA, FA or nest), and each grouping of stands is then aggregated into separate stratum for analysis.
- The exam plot data for each stand are assigned to their respective stratum, and disassociated with individual stands.
- Data for each plot is expanded to a unit area basis, and grown to a common analysis year through the FVS model. Plots comprising a specific stratum are then batch run through FVS. FVS will summarize VSS classification, normal standard summary statistics and computed values for each plot independently.
- FVS individual plot summaries are exported to the PLOT_BY_PLOT excel workbook. Macros in this workbook are used to combine plots by VSS class and display summary statistics for each VSS class. Examples of these summaries are shown in Figures 6 and 7.

Target Stand Conditions

Figure 3 shows the six structural stages and their target distribution for sustaining goshawk habitat, or for that matter sustaining any vegetative community (Odum 1971).

Understanding canopy cover and how it is measured is required for developing the desired goshawk habitat. Moreover, canopy cover recommendations vary depending on goshawk home range components (i.e., nest area, PFA, FA). Canopy cover is defined in MRNG as: “the percentage of a fixed area covered by the crowns of plants delimited by a vertical projection of the outermost perimeter of the spread of the foliage” and is determined at the group or clump level for VSS 4 thru 6 (mid-aged through old forest).

The group and clump nature, especially of native ponderosa pine forests in the Southwest, not only encompasses the area occupied by tree boles and canopies but it also includes areas extending beyond tree crowns that are often occupied by tree roots (Pearson 1950). These areas (primary rooting zones) between tree clumps are free of trees and are occupied by roots of trees within the clump/group. These areas not only provide a rooting zone, they also provide growing space for crown expansion, needle development and other factors that allow trees within groups to develop while the desired relatively high canopy cover within the group is maintained (Figure 8). Therefore, a highly heterogeneous stand containing the entire suite

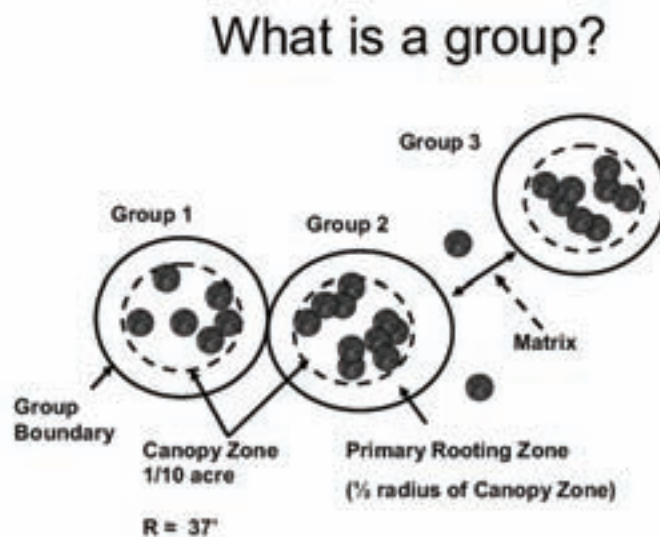


Figure 8. Groups and Group Spacing

of VSS (grass/forb/shrub through old forest) and the open areas occupied by tree roots would be at full site occupancy even though timber management metrics (e.g., stand density index, trees per unit area, basal area) may not indicate these conditions. In addition, by separating the crowns between tree groups and creating and maintaining highly variable forest structures with multiple tree juxtapositions, these conditions reflect the historic conditions and reduce crown fire potential as measured by crowning and torching index. Additional desired conditions include a scattering of large reserve trees (7.4-12.3 per ha $\approx$ 3- 5 per acre), large snags (4.9 per ha $\approx$ 2 per acre), large logs (7.4 per ha $\approx$ 3 per acre), and woody debris (2.2-3.1 Mg per ha $\approx$ 1965 to 2947 pounds/acre) through the landscape. The forest floor should retain a reasonable amount of down woody debris the amount of which should be commensurate with local site historic conditions and provide for the ability to restore frequent surface fires for ecosystem maintenance.

Prescription Targets

Various threshold target densities for structural groups are recommended in the MRNG for differing forest types and goshawk home range components. Based upon these recommendations, targets have been developed for goshawk PFA and FA for ponderosa pine and mixed conifer forest types. Determination of target canopy cover thresholds during field implementation has been problematic due to various methods used to measure canopy cover and natural variability of forest characteristics. There have also been misunderstandings of the recommended means for measuring canopy cover. Therefore target stocking levels for project implementation have been developed by using the FVS canopy cover model. With this model, stocking levels were developed based on canopy cover and translated to measurable variables such as trees per unit area and basal area (BA) and by VSS class (Tables 1-4). Growth modeling to develop target stocking levels was conducted using the following assumptions:

- The MRNG target canopy cover thresholds for VSS 4-5-6 would be achieved and maintained.
- Target stocking levels for VSS 1-2-3 were developed to provide necessary stocking levels of younger tree age classes to achieve future mid-aged to old tree target size classes/densities.

- Differing site indices were examined for ponderosa pine and dry mixed conifer forest types. No significant stand developmental differences were noted based upon an average site index range, so stocking targets were developed by forest type and goshawk home range component (PFA and FA). (Stocking levels may need to be adjusted for very dry or otherwise poor site locations.)
- Structure and density is maintained by mechanical treatments on a 20-year cycle, and prescribed fire on a 10-20 year cycle.

Projections based on the stand modeling process determined that threshold canopy cover densities could not be achieved nor maintained as a whole stand condition. Therefore, it is recommended that group size be limited to less than 0.2 ha (0.5 acre) on average to sustain the desired canopy cover levels. Primary rooting zones provided growing space necessary to develop and sustain the desired canopy cover within tree groups. Full-stocking level targets were developed with these concepts in mind and are a starting basis for project planning. In the context of MRNG, full-stocking levels are defined as the condition where the forest is patterned as shown in Figure 9. Note that full-stocking in this context is not equivalent to full-stocking as defined by traditional density measurements such as stand density index (SDI).

Many current Southwestern Region projects are focused on RHRC, since there has been a tremendous increase in the number of trees per unit area since the late 1800's. Historic reference conditions are generally less than the full-stocking level MRNG targets for average growing site conditions. On forest sites of above-average growth potential, the historic condition may be greater than the full-stocking level MRNG targets. Sometimes hazardous fuels reduction project objectives cannot be achieved at the full-stocking level. However, the broad objectives of the MRNG can be met at a wide range of forest densities; overall stand density can be reduced or slightly increased from the full-stocking level, while adhering to MRNG sustainability principles. During project implementation, canopy zone separation of tree groups may be adjusted from full-stocking, as long as forest structural sustainability and group canopy cover guidelines are met. If the desired conditions can not be met, a site specific Forest Plan amendment must be prepared. However in most cases, conditions suitable for

both MRNG and project objectives can be met by creation of additional inter-group space (matrix) between rooting zones to achieve a desired average distance between tree canopy zones in adjacent groups (Figure 10). This inter-group space can be described as an additional open area composed of few or no trees. It may be desirable to retain a low density scattering of individual trees throughout this area, both to provide for natural forest patterns, and to provide for long-term development of other structural components (i.e., large/old reserve trees). To facilitate prescription development, a spreadsheet tool has been developed to calculate stand metrics at variable spacing

distances between tree group canopy zones (USDA Forest Service, Southwestern Region, 2006). Figure 11 displays the spreadsheet summary data for a ponderosa pine forest type PFA at the full-stocking (100%) level. In this example, the spacing between canopy zones is 37.1 feet (11.3 meters) and the average acre mean target density is 69 square feet basal area (2.6 square meters/hectare). Figure 12 shows the summary data for an adjusted target condition with spacing between canopy zones of 55.5 feet (16.9 meters). In this example, the adjusted average acre mean density target is 51 square feet basal area (1.9 square meters/hectare), representing a resulting stocking density of 74% of

MRNG "full-stocking" Concepts*



- * ponderosa pine PFA full-stocking target = 25% SDI max
- * ponderosa pine FA full-stocking target = 20% SDI max

Figure 9. Full-Stocking (Illustration)

Adjustments to MRNG "full-stocking" targets

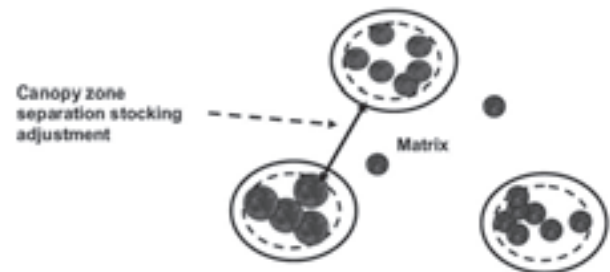


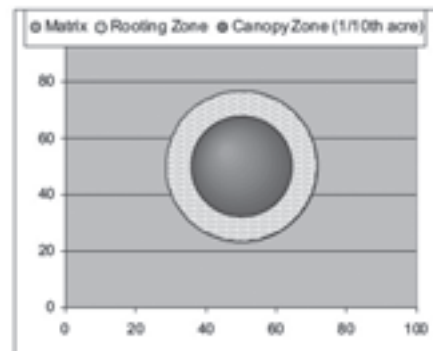
Figure 10. Groups and Stand Density Adjustments from Full-Stocking (Illustration)

MRNG "full-stocking" (PP PFA)

VSS Class	Per acre basis mean (average acre)	
	tpa	BA/Ac
1	20.3	0
2	19.3	1
3	27.3	11
4	4.8	6
4	9.0	11
5	7.7	19
6	5.5	22
	93.9	69

Dq* = 13.11
SDI* = 113.7

*includes trees >= 1" dbh only.



Functional Area	Radial Spacing	Radial Factor
Canopy Zone (1/10th acre)	37.2	1.00
Canopy + Rooting Zone	55.8	1.50
Matrix	0	0
Canopy Zone Separation Distance		37.1
Stocking Multiplier		100%

Figure 11. Full-Stocking—ponderosa pine PFA (Spreadsheet Summary)

MRNG Adjusted Stocking (PP PFA)

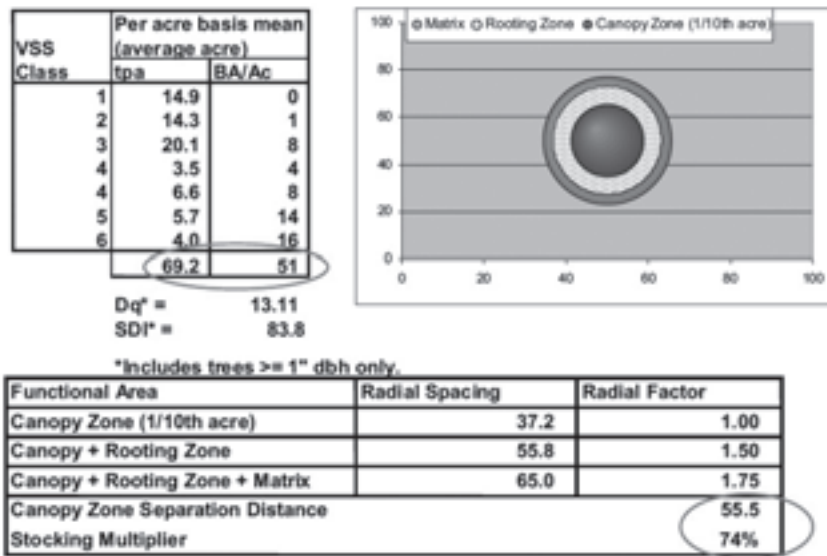


Figure 12. Groups and Stand Density Adjustments from Full-Stocking (Spreadsheet Summary)

full stocking. Use of this spreadsheet tool facilitates rapid analysis of target stand conditions for project planning. The leave stand characteristics displayed by the spreadsheet can then accompany the marking guide templates to provide implementation instructions to field layout crews.

IMPLEMENTATION

Project Layout

Surveys for goshawks are made within the management analysis area prior to management activities, including an area 0.8 km (0.5 mile) area beyond the boundary. Survey requirements are spelled out in the MRNG. These surveys will help identify nest areas, PFAs, and foraging areas and a requisite spacing of territories across the landscape (3.2 to 4 km $\approx$ 2 to 2.5 mile). Goshawk Territories are made up of the following components (Figures 1-2):

- 1) PFA (168 ha $\approx$ 420 acres total), including 6 nest areas, each 12 ha ($\approx$ 30 acres) in size; 72 ha ($\approx$ 180 acres) minimum of nest areas should be identified within each PFA (six nest sites - three nests are suitable and three are replacements).
- 2) One Foraging Area = 2,160 ha ($\approx$ 5,400 acres) surrounding the PFA.
- 3) Total Home Range Size = about 2,400 ha ($\approx$ 6,000 acres).

In summary, planning and management of goshawk territories entails the map delineation of 6 nesting areas (3 potential, 3 replacements), a post-fledgling family areas that encompasses the six nesting areas, all surrounded by the foraging area. A territory with these components is established where there are known nest sites, old nest sites, areas where historical data indicates goshawks have nested in the past, and where goshawks have been repeatedly sighted over a 2-year period. When possible, all historical nest areas should be maintained. Human activity should be limited in nesting areas and PFAs during the breeding season (March 1 through September 30). The remainder of the 2,400 ha (6000 acre) management territory consists shall be managed as

foraging areas, according to applicable guidelines

Tree Designation

This management strategy focuses on attainment and monitoring of residual forest characteristics. For these reasons, implementation of project prescriptions would be difficult or impossible without designation by leave tree marking.

General Guides for Marking Even-aged and Two-aged Stands

Stands with these structures cannot be regulated with uneven-aged management systems during initial treatment entries. The following guidelines outline a process for converting these types of stands to uneven-aged structures within the context of MRNG desired conditions.

Initial steps toward conversion to an uneven-aged forest structure:

1. Where some age class diversity is present, leave as many under represented VSS trees, in small groups, as possible.
2. Create openings for VSS 1 age class recruitment (including the primary rooting zones) on approximately 15-20% of project/stand area. Do this by removing entire groups of trees from the predominant age class, but only in those areas with trees of sufficient maturity and vigor to provide adequate seed for

natural regeneration. Diseased (e.g., mistletoe) and damaged trees may be targeted for regeneration, but because these trees are often important elements in forest ecosystems, some may be retained. It is recommended that diseased trees not be retained as seed trees for new VSS 1 groups. Diseased trees can be retained within larger VSS tree groups with other trees of similar height. Diseased trees can also be isolated from younger trees when the desired distances between tree groups is attained. Follow MRNG regarding tree group sizes, opening sizes, reserve trees, downed logs, woody debris, etc.

3. Thin trees to initiate development of greater forest diversity, to create desired forest structural characteristics, and as necessary to meet project-level resource objectives.

- a) Begin to create group structure, including both the canopy and primary rooting zone portions of the group. Initial creation of groups could be referred to as creating doughnuts. The doughnut is the primary rooting zone and the doughnut hole is the canopy zone that is left. These groups should generally be sized from 0.04 to 0.13 ha ($\approx$ 0.1 to 0.3 acre) of various shapes. Once trees within identified primary rooting zones (and the inter-group matrix, if necessary) are removed, thinning of retained groups may be required to grow young trees (VSS 2 and 3) into large trees more rapidly.

- b) Thin VSS 2 and 3 groups to variable densities to accelerate their growth. However, thin VSS 2 and 3 commensurate with attaining the desired interlocking crowns when these tree groups grow into VSS 4-6.

- c) For high-density mid-aged groups that have not been previously thinned, only thin groups to the extent necessary to sustain group maintenance and development, taking care to maintain required canopy cover density. Do not thin VSS 6 groups, except to remove smaller young trees that pose a threat to sustainability of the mature tree group. Do not thin groups such that the structural attributes are altered. Suppressed and damaged trees that have developed with the group are important habitat elements for wildlife species. Squirrels, for example use overtopped trees for nesting. Only thin groups to the extent necessary to maintain desired current and future species composition, sustainability and development.

- d) Areas lacking adequate seed bearing trees will be difficult or impossible to regenerate naturally during an initial management entry. In this situation, accelerating tree growth to larger VSS classes is the primary management objective. Thinning should be prescribed to develop trees within groups that have full live crowns for viable cone production.

4. Create a matrix between tree groups (groups = canopy and primary rooting zones) if less than full stocking is desired, based upon project-level objectives. Maintain matrix (beyond rooting zones) with few or no trees. A matrix is designed to provide for additional spacing between high density canopy zones, beyond that provided by the primary rooting zones.

General Guides for Marking Uneven-aged Stands

Some project areas and ecosystem management areas presently have a mix of VSS classes. These areas can be managed to further develop and maintain desired VSS class diversity. Inventory data at the landscape level is desired to ensure the VSS distribution proportions and juxtapositions are known so the desired structural proportions on the landscape can be developed. By having a landscape view, prescriptions can be developed to ensure the landscape desired condition is planned for and being developed. In order to maintain desired structural proportions on the landscape, it's important to balance structural distribution at the local (stand scale) within larger landscapes. Some local areas could be managed to temporarily provide a disproportionately large percentage of a structural stage that is otherwise limited or lacking in the larger landscape. Any VSS in surplus of the desired percentage may be regenerated to create a future balance of VSS classes. Marking guide templates have been prepared for use as samples for different forest cover types and goshawk habitats in uneven-aged forest conditions, based upon full stocking density targets (USDA Forest Service, 2006b). As previously discussed, stocking levels can be adjusted to meet various management objectives and provide prescription parameters for residual stand desired conditions. A marker's tally form spreadsheet has been developed as a guide to measure attainment of desired structures during marking. Each marker tallies leave tree groups by VSS and group size (1/10 acre basis) during the marking process. This tool summarizes post-marking conditions to provide information on how many acres of each VSS class are being retained as well as the residual percentage of each VSS class over the entire

VSS Class	Group Equivalent Acres	Individual Tree Equivalent Acres	Total Equivalent Acres	Percent of Area by VSS Class
VSS 1	5	0	5	4
VSS 2	9	0	9	7
VSS 3	35	3	38	27
VSS 4	58	0	58	41
VSS 5	24	2	26	18
VSS 6	4	2	6	4
Total Acres	134	7	141	100

Marker's Talley Form Spreadsheet Summary

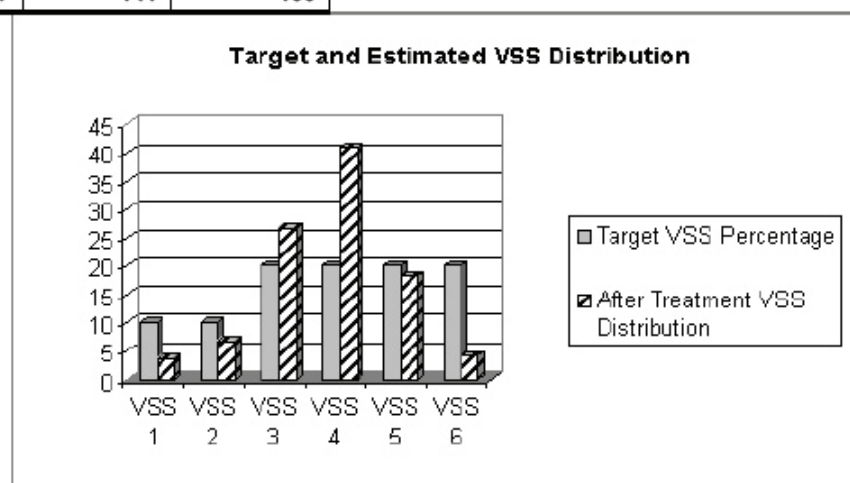


Figure 13. Marker's Tally Form Summary

project area (USDA Forest Service, 2006c) (Figure 13). As such, this tool can be used for both quality control and as monitoring documentation.

CONCLUSION

The goal of silviculture is to develop vegetation management strategies to meet desired conditions (silvicultural systems- a planned series of treatments through the life of a forest) and to document them in silvicultural prescriptions to meet management objectives. The Southwestern Region is committed to restore the historic resilience and function of fire adapted (ponderosa pine and dry mixed conifer) forests and is committed to develop and maintain habitat for both the Mexican spotted owl and the northern goshawk through a Regional Amendment of Forest Plans. Management recommendations for the northern goshawk,

ecological restoration research, ponderosa pine management information, and fire hazard and fuel treatment information are only a partial list of science that can be used to inform the management actions (silvicultural treatments) in the Southwestern Region (Reynolds and others 1992, Covington and Moore 1994, Pearson 1950, White 1985). Through this effort we have been able to establish a strong link between restoring historic conditions and implementing the MRNG. However, this information in most circumstances is inadequate for planning and executing on-the-ground activities. Therefore the Southwestern Region developed tools that could help planners, silviculturists, fuels specialists, and others involved with designing and implementing treatments directed at restoring fire adapted forests of the Southwest. These tools and the MRNG both provide a template, a process, and approach that are adaptable to a wide variety of forests and a wide variety of management

objectives. We do not postulate that the exact metrics in the MRNG or those presented in spreadsheet tools are precisely applicable to other locales. However, we argue the tools and procedures have sufficient flexibility to allow project planners to incorporate other objectives.

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**Assessing the Ecological Benefits and Opportunity Costs
of Alternative Stream Management Zone Widths
for Eastern Hardwoods**

Chris B. LeDoux and Ethel Wilkerson

ABSTRACT

Leaving buffer zones adjacent to waterways can effectively reduce the water quality concerns associated with timber harvesting. However, riparian areas are also some of the most productive sites and can yield high quality wood. The amount of unharvested timber left in SMZs (Streamside Management Zones) can represent a substantial opportunity cost to landowners. In this study we used computer simulation to integrate contemporary scientific data among disciplines to develop opportunity cost and ecological function protection tradeoffs that result from the implementation of alternative SMZ widths. We quantified the opportunity costs and ecological benefits of using different buffer zone widths. We used the principles of benefit/cost analysis to

compare the results. Results suggest that benefit/cost ratios range from 5.89 to 1.49 depending on the buffer zone width, the species composition of the stand, and the logging technology used to harvest the timber. A literature review was used to score the ability of different buffers to protect riparian functions. Results show that to fully protect the riparian functions modeled, 45 meter buffers are needed. The study results should be of high interest to landowners, managers, loggers, land use planners, and other decision and policy makers who need to understand the opportunity costs and ecological benefits associated with implementing different widths of streamside management zones.

Keywords: ecological functions; capital recovery costs; simulation; optimization; riparian zones; benefit/cost ratio

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INTRODUCTION

Streams, wetlands, and riparian areas are among our most valuable natural assets. From an ecological perspective, riparian zones are among the most productive wildlife habitats on the continent (Bisson and others 1987, Kentucky Department of Fish and Wildlife Resources 1990). In addition, riparian areas protect water quality and aquatic communities by reducing the amount of sediment entering the stream channel (Castelle and Johnson 2000), shading the stream channel from solar radiation (Brown and Krygier 1967), supplying organic material for food (Allan 1995), and contributing woody material that increases the hydraulic and structural complexity of the stream channel (Bisson and others 1987, Hilderbrand and others 1997). Removal of streamside vegetation during logging operations has been shown to increase the sediment load in the stream (Davies and Nelson 1994), increase water temperature (Brown and Krygier 1967), and change the food supply and condition of the habitat, altering the aquatic and riparian communities (Hawkins and others 1982, Hanowski and others 2002). Leaving buffer strips adjacent to waterways can effectively reduce the water quality concerns associated with timber harvesting.

Because of their ecological importance, the protection of riparian areas is a top priority for most state and federal conservation agencies (Blinn and others 2001). This goal is usually achieved by establishing streamside management zones (SMZs) adjacent to waterways and by adopting best management practices (BMPs), which are guidelines for locating haul roads, skid trails, log landings, and stream crossings. Recommendations for SMZs and BMPs vary among states (Huyler and LeDoux 1995, Shaffer and others 1998, Vasievich and Edgar 1998, Blinn and Kilgore 2001, Williams and others 2004) and are often voluntary. For example, a commonly suggested BMP includes no harvesting activities in 15-45m buffer strips adjacent to the waterway, sometimes with allowances for up to 50 percent removal of the volume of standing trees to leave an evenly spaced stand to protect the riparian function (LeDoux and others 1990, Phillips and others 2000).

Riparian areas also are some of the best sites for producing high quality wood products. The unharvested timber left in SMZs can represent a substantial opportunity cost to landowners (Shaffer and Aust 1993, Kilgore and Blinn

2003, LeDoux 2006). The opportunity costs are influenced by the species mix in the stand, by the logging technology used, the level of riparian protection desired (Peters and LeDoux 1984, LeDoux 2006), the stream network to be protected (Ice and others 2006), and the increasing proportion of isolated SMZ units within a watershed (Olsen and others 1987, University of Washington 1999). Simultaneous economic and environmental assessments have been reported addressing the consequences of alternative fuel management strategies (Mason and others 2003) and the layout and administration of fuel removal projects (Hauck and others 2005). Companion papers address the opportunity costs/capital recovery cost of managing for old growth forest conditions (LeDoux 2004), of alternative patch retention treatments (LeDoux and Whitman 2006), and of implementing streamside management guidelines in Eastern hardwoods (LeDoux 2006, Li et al 2006). In this study, we had two objectives:

- 1) to evaluate the opportunity costs of different SMZ protection options for two different stand types using four different logging technologies; and
- 2) to compare the opportunity costs with the ecological benefit of different SMZ widths using the principles of benefit/cost analysis.

The data and results summarized in this paper are borrowed heavily from LeDoux (2006) and LeDoux and Wilkerson (2006).

METHODS

Stand Data

The two 27.5-ha stands selected for this study were similar in age (120 years old), density, average diameter at breast height (d.b.h.), and volume. One stand represents a medium- to low- value species mix comprised predominately of yellow-poplar (*Liriodendron tulipifera* L.) with some red maple (*Acer rubrum* L.), black cherry (*Prunus serotina* Ehrh.), and sycamore (*Plantanus occidentalis* L.). This stand has 232 trees/ha, an average d.b.h. of 45.6cm, and a merchantable volume of 329m³/ha. We refer to this stand as yellow-poplar or “YP.”

The second stand represents medium- to high-value mixed hardwood species comprised of yellow-poplar, American beech (*Fagus grandifolia* Ehrh.), shagbark hickory (*Carya ovata* (Mill.) K. Koch), black cherry, red maple,

Table 1. Logging system configurations and costs used to simulate the harvest of the 27.5 ha tracts.

Logging Technology	Description	Cost/unit	
		Yellow-poplar stand	Mixed hardwood stand
		-----\$/m ³ -----	
A	Chainsaw felling with Ecologger I cable yarder	20.83	20.47
B	Timbco 445 Cut-to-length harvester with Valmet forwarder	17.65	17.30
C	Chainsaw felling with John Deere 640 cable skidder	16.24	16.24
D	Timbco 425 feller buncher with Valmet forwarder	15.88	15.88

cucumber tree (*Magnolia acuminata* L.), sugar maple (*Acer saccharum* Marsh.), red oak (*Quercus rubra* L.), white oak (*Quercus alba* L.), and eastern hemlock (*Tsuga canadensis* (L.) Carr.). This stand has 224 trees/ha, an average d.b.h. of 46.4cm, and a merchantable volume of 341m³/ha. We refer to this stand as mixed hardwood or “MH.”

These stands were selected because of their similarities, availability of detailed tree measurements, and a relatively low and high value species mix level. These stands are typical of the eastern hardwood region of the United States. Both stands were subjected via computer simulation to the same even-aged silvicultural treatment, all merchantable timber was harvested.

Logging Systems Evaluated

Computer simulations of four logging systems were used in this study (Table 1). These logging systems were selected because we have robust time and motion study data for each and they represent contemporary methods being used by loggers to harvest eastern hardwood stands. Machine capacities were matched to the size of logs to be removed. Machine configurations are ranked by their per-unit operating cost, with the Ecologger I1 cable yarder being the most expensive and the Timbco 425 feller buncher with the Valmet forwarder being the least expensive. The per-unit operating cost for logging system combinations C and D are very similar, but reflect different on-the-ground operating conditions since logging technology D is mechanized.

Models Used

Two computer software models were used. The first model, ECOST (LeDoux 1985), estimated the stump-to-mill logging costs for the logging technology configurations (Table 1). ECOST is a computer program that can estimate the stump-to-mill costs of cable logging, conventional ground-based skidding, cut-to-length, feller-buncher applications, forwarding, and several small farm tractors for logging eastern hardwoods. Stand data were input into ECOST to develop simulated estimates of the stump-to-mill costs. The cost information within ECOST comes from time studies and simulations conducted over the years. The cost information is part of the model and is updated yearly. All costs are in 2005 U.S. dollars and reflect new equipment.

The second model, MANAGE-PC (LeDoux 1986) computer program, provides the volume yield and volume/product estimates. MANAGE-PC integrates harvesting technology, silvicultural treatments, market prices, and economics in a continuous manner over the life of the stand. The simulation is a combination of discrete and stochastic subroutines. Individual subroutines model harvesting activities, silvicultural treatments, growth and yield projections, market prices, and discounted present net worth (PNW) economic analysis. The model can be used to develop optimal economic management guidelines for eastern hardwoods. Stand data were entered into MANAGE-PC to provide volume/production yield estimates. The

Table 2. Delivered prices for sawlogs and fuelwood/pulpwood by species

Species	Product			
	Large ^a sawlogs	Medium ^b size sawlogs	Small ^c sawlogs	Fuelwood ^d / Pulpwood
	----- \$/m ³ -----			--- \$/cunit ---
Sugar maple	135	103	58	45
Red maple	106	81	56	45
Sycamore	119	75	53	45
American beech	119	75	53	45
White oak	173	118	58	45
Yellow-poplar	119	75	53	45
Red oak	238	168	95	45
Cucumber tree	119	75	53	45
Shagbark hickory	71	61	56	45
Hemlock	64	64	64	45
Black cherry	119	75	53	45

^aMinimum small-end diameter ≥ 33.02 cm, length ≥ 3.05 m.

^bMinimum small-end diameter ≥ 27.44 cm, length ≥ 2.44 m.

^cMinimum small end diameter ≥ 25.40 cm, length ≥ 2.44 m.

^dMinimum small-end diameter ≥ 10.16 cm that will not make large, medium, or small sawlogs.

average delivered prices for sawlogs and pulpwood (Table 2) were obtained from forest products price bulletins (Ohio Agriculture Statistics Service 2007, Pennsylvania State University 2007, Tennessee Division of Forestry 2007, West Virginia University Division of Forestry 2007).

SMZ Protection Options

The stands were modeled identically and it was assumed they were bisected by a perennial stream (Fig. 1). Although riparian area cross-sections adjacent to streams can be quite variable, we assumed homogeneity of stand composition and consistent 20-25 percent sideslopes to simplify the simulations. The simulated harvesting plan removed timber from both sides of the stream to landings on truck haul roads located on both sides of the stream under the five SMZ treatment levels. SMZ protection options evaluated include:

- 1) no protection, harvest all 27.5ha without buffers;
- 2) unharvested 15m SMZ on both sides of the stream;
- 3) unharvested 30m SMZ on both sides of the stream;
- 4) unharvested 45m SMZ on both sides of the stream; and
- 5) a partially harvested 30m SMZ on both sides of the stream with approximately 50 percent of the timber volume removed from the SMZ.

Although commonly recommended riparian management zone guidelines call for partial volume removal (Blinn and Kilgore 2004), we wanted to evaluate the opportunity costs and ecological benefits for more restrictive treatments, such as options 3 and 4. For the no-protection option, we assumed that the operator could select where haul roads and skid trails would occur with no restrictions on soil distur-

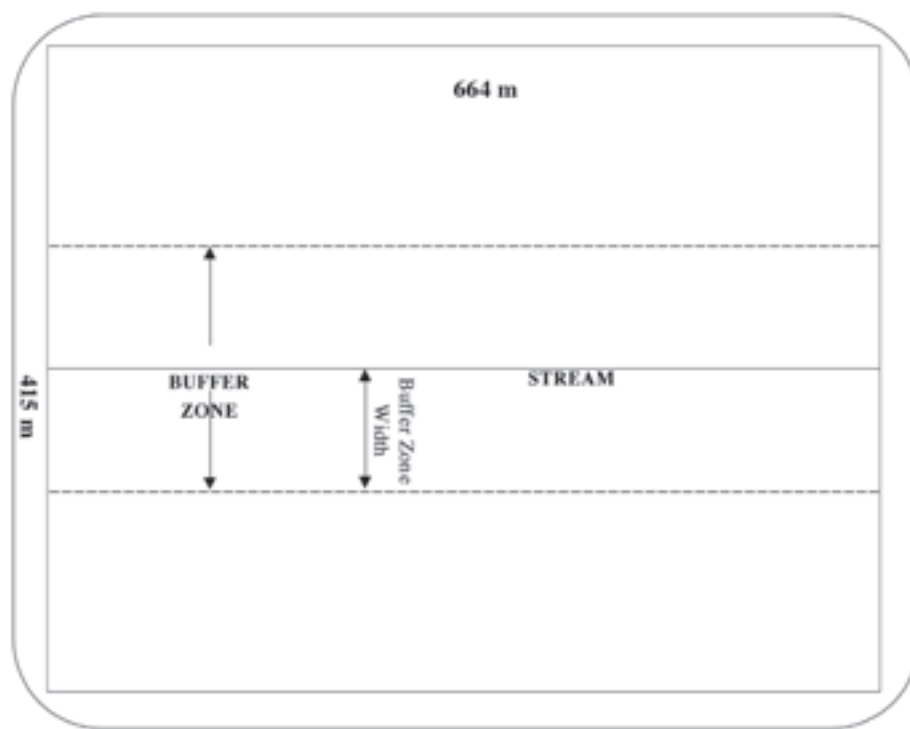


Figure 1— Diagram of modeled harvest area, stream, buffer zone, and buffer zone width.

bance or exposure. For all of the options, we assumed the volume and species mix removed would remain constant as we moved further away from the stream to simplify the simulations.

Riparian Protection Score

The ecological functions of riparian zones are numerous and range from stabilizing near-stream soil (Castelle and Johnson 2000) to providing travel corridors for large terrestrial mammals (Klapproth and Johnson 2000). Quantifying the full range of physical and biological functions that occur within riparian areas would be a daunting task. In this study, we focused on the processes and biota that are easily measurable and strictly dependent on and/or unique to riparian zones. We limited the various functions of the modeled riparian forests to the following five categories:

- 1) coarse woody debris supply;
- 2) shade/temperature maintenance;
- 3) sediment filtering;
- 4) maintaining aquatic communities (macroinvertebrates and periphyton); and
- 5) maintaining habitat for riparian-associated passerine birds.

We conducted a literature review to identify studies examining at least one of our five categories of riparian function. Studies with SMZ widths that did not correspond exactly to those used in our economic models were placed in the most logical category, while studies with large discrepancies in SMZ width or experimental design were excluded from this study. We found that few studies examined partial timber removal in SMZs (option 5 in this study) so this treatment was not evaluated for riparian protection. Research on the ecological assessment of SMZs does not exist in adequate quantities from a single region of the United States. To complete the analysis, we tried to focus on literature from the eastern United States, but as data was limited

we included studies from other regions. The evaluation of SMZ protection was limited to no SMZ (option 1), and unharvested SMZs with widths of 15m, 30m, and 45m (options 2, 3, 4, respectively).

For each SMZ width (excluding the partial harvest treatment) we assessed the capacity of the SMZ to protect against post-harvest changes for each of the five categories of riparian function based on the following criterion: the SMZ does not protect the component resulting in large post-harvest changes (score = 0); SMZ results in moderate post-harvest changes (score = 1); SMZ results in small post-harvest changes (score = 2); or SMZ protects against measurable changes in the component (score = 3). Scores were determined by comparing the magnitude of change to other studies or other SMZ widths and the statistical significance/non-significance of post-harvest changes. Each SMZ width was given a numerical score (0-3) for each of the five categories of riparian function. An overall score for each SMZ width was calculated by summing the score of each category of riparian function. The overall scores had a minimum value of 0 and a maximum value of 15. The overall score was then converted into a percentage with 0 percent representing no protection of riparian functions

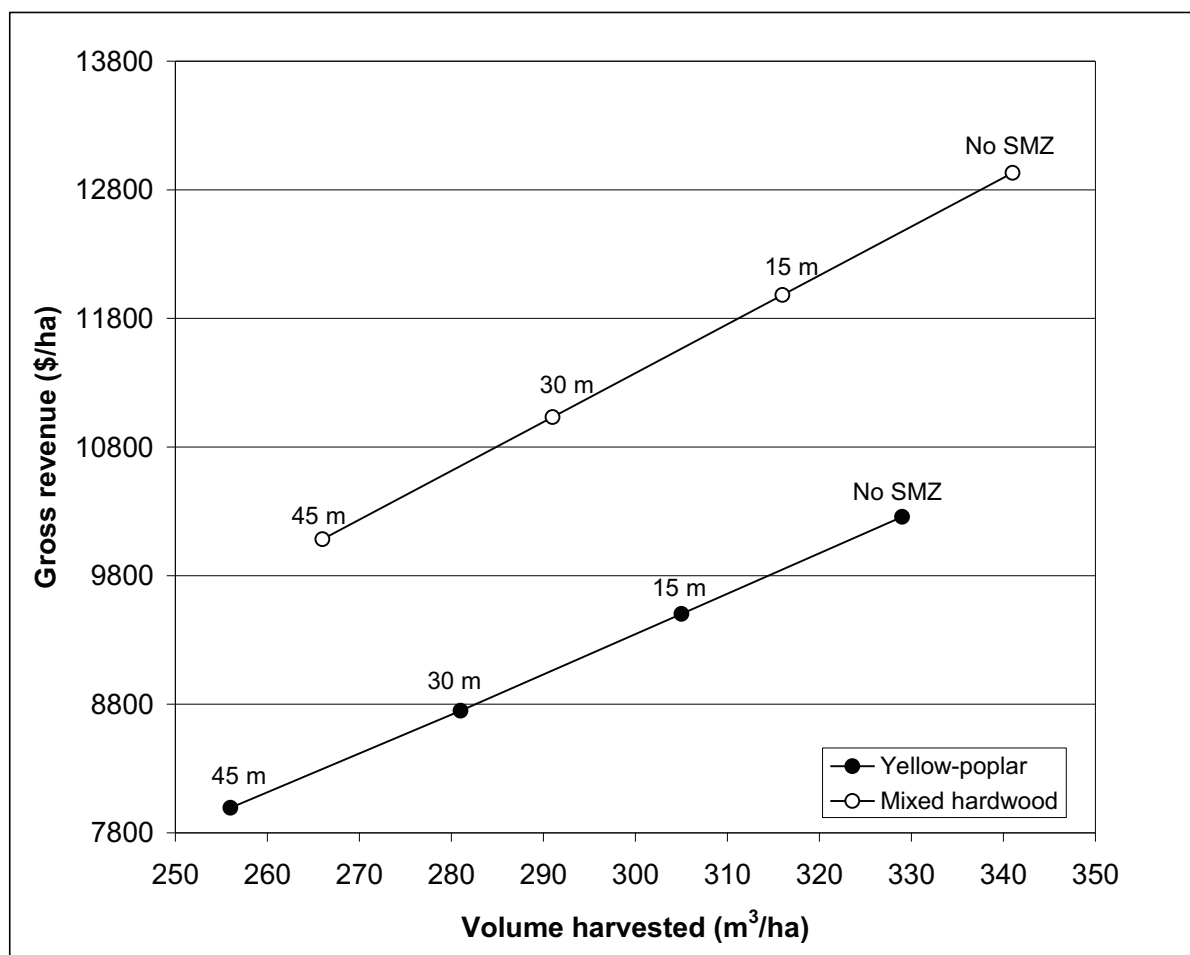


Figure 2— Gross revenue by timber volume harvested from yellow-poplar and mixed hardwood stands.

(value of 0) and 100 percent representing complete protection against measurable changes in riparian functions creating conditions similar to undisturbed riparian areas (value of 15). Each SMZ protection option has a score ranging from 0 to 100 percent and represented the effectiveness of the SMZ in protecting riparian functions. It is hereafter referred to as the SMZ protection score. Although the structure within the SMZ changes over the 120-year rotation, our SMZ protection score described above is based on the immediate condition of the riparian area. The canopy cover may recover quickly even for the completely harvested unit however, large wood recruitment may take much longer where the unit is completely harvested but it may be shortened for the partial harvest options (Zobrist and others 2005). We did not consider changing SMZ protection score over the 120-year rotation because we simply do not have the necessary data.

RESULTS AND DISCUSSION

Gross revenues from timber cutting depended on stand composition and the volume of timber volume harvested. The gross income from the yellow-poplar stand ranged from \$7,995/ha to \$10,257/ha, while the gross revenue of timber from the mixed hardwood stand ranged from \$10,084 to \$12,931/ha (Fig. 2).

Logging costs, which varied with the equipment used, are deducted from the gross revenue of the timber harvest to determine net income. Harvesting costs for yellow-poplar range from \$15.88 to \$20.83/m³ and from \$15.88 to \$20.47/m³ for a mixed hardwood stand (Table 1). While logging costs are comparable between the two stands (Fig. 3), they represent a larger percentage of the gross revenue in the yellow-poplar stand because of a greater profit margin for the mixed hardwood stand.

Only the cost of the logging system was considered as a treatment in this study. Cable logging systems may reduce the roads and landings needed to harvest a tract thus reducing the potential for erosion and sediment production. Mechanized track mounted systems, such as the cut-to-length and the feller buncher with forwarder, may result in less soil disturbance and compaction, and thus reduce roading and landing area. In this study we did not address the physical or ecological impacts of alternative systems because we lack the necessary data. Undoubtedly, managers must consider logging system options when making decisions on SMZs.

SMZ Protection Options

Leaving no SMZ generated the most revenue (gross and net) to the landowner (Fig. 4a and 4b) by providing the largest volume of wood (Table 3) and gross revenues of \$10,257 and \$12,931/ha for the low value yellow-poplar

stand (Fig. 4a) and high value mixed hardwood stand (Fig. 4b), respectively. The net revenue ranges from \$3,415 to \$5,048/ha for the yellow-poplar stand (Fig. 4a) and \$5,903 to \$7,546/ha for the mixed hardwood stand (Fig. 4b) depending on the logging technology.

Compared to leaving no SMZ, a 15m unharvested buffer on both sides of the stream removed 24m³/ha less wood from the yellow-poplar stand and 25m³/ha from the mixed hardwood stand (Table 3). This scenario grossed \$754/ha less than leaving no SMZ for the yellow-poplar stand (Fig. 4a) and \$949/ha less for the mixed hardwood stand (Fig. 4b). The difference in net revenue can be viewed as the opportunity cost for retaining that width of SMZ. The cost of maintaining 15m SMZ ranges from \$252 to \$370/ha (yellow-poplar stand, Fig. 5a) and \$432 to \$553/ha (mixed hardwood, Fig. 5b), depending on the logging technology.

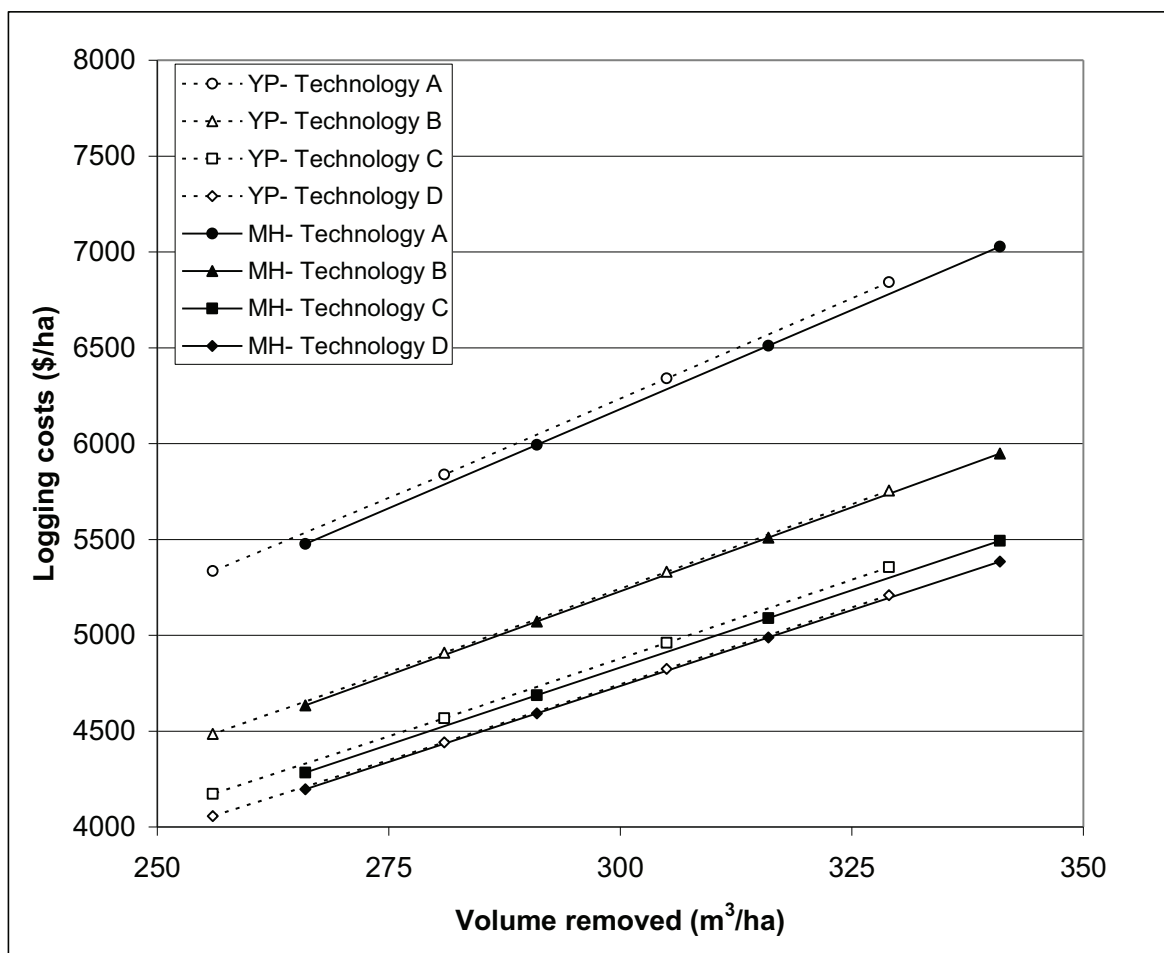


Figure 3— Logging costs by volume removed for mixed hardwood and yellow-poplar stands for four different logging systems (see Table 1 for description of technologies used).

Table 3. Volume of timber harvested and retained for each protection option in the yellow-poplar and mixed hardwood stands.

Protection Option	Yellow-poplar		Mixed hardwood	
	Volume Harvested	Volume Retained	Volume Harvested	Volume Retained
	<i>m³/ha</i>			
No SMZ	328	0	341	0
15 m	304	24	316	25
30 m	280	48	291	50
45 m	256	72	266	75
30 m with partial harvest	304	24	316	25

Leaving a 30m SMZ on both sides of the stream removes 48m³/ha and 50m³/ha less merchantable wood than harvesting without an SMZ (Table 3) for the yellow-poplar and mixed hardwood stands, respectively. This level of SMZ protection decreases gross revenue from harvesting with no SMZ by \$1,508/ha and \$1,898/ha for the yellow-poplar (Fig. 4a) and mixed hardwood (Fig. 4b) stands, respectively. Leaving 30m buffers on both sides of the stream has a protection cost of \$504 to \$740/ha for yellow-poplar (Fig. 5a) and \$864 to \$1,106/ha for the mixed hardwood stand (Fig. 5b), depending on the logging technology.

A 45m SMZ on both sides of the stream removes 72 m³/ha and 75 m³/ha less merchantable timber for the yellow-poplar and mixed hardwood stand, respectively, as compared to harvesting with no SMZ (Table 3). Gross revenues decrease by \$2,262/ha for yellow-poplar (Fig. 4a) and \$2,847/ha for mixed hardwood (Fig. 4b) and protection costs range from \$756 to \$1,110/ha for yellow-poplar (Fig. 5a) and \$1,296 to \$1,659/ha for mixed hardwood (Fig. 5b) when compared to leaving no streamside buffer.

Removing 50 percent of the timber volume from a 30m SMZ results in removal of the same volume of timber as unharvested 15m buffers on both sides of the stream (Table 3). Compared to unharvested 30m buffers, harvesting 50 percent of the timber volume from the 30m SMZs can increase the gross revenue by \$754/ha for yellow-poplar (Fig. 4a) and \$949/ha

for mixed hardwood (Fig. 4b) as well as decreasing the protection costs between \$252 and \$370/ha for yellow-poplar (Fig. 5a) and \$432 and \$553/ha for mixed hardwood (Fig. 5b).

Ecological Benefit

While maintaining SMZs represents a sizeable opportunity cost to the landowner, buffers provide a wide range of ecological benefits to riparian areas. SMZs that are too narrow cannot adequately protect all riparian functions, but

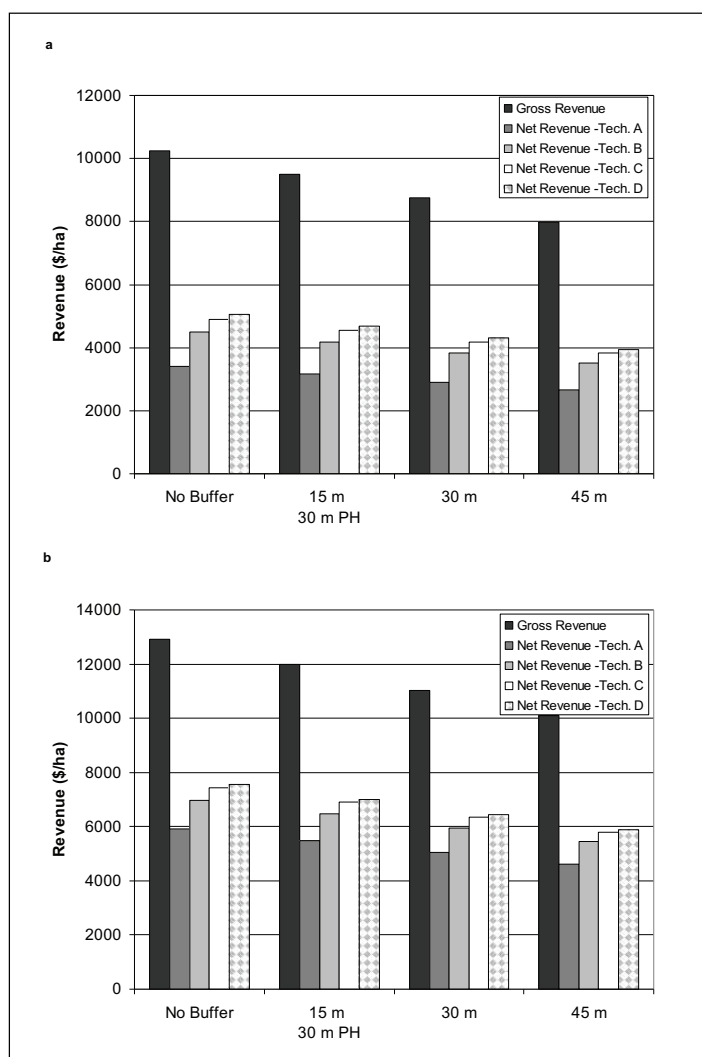


Figure 4—Gross and net revenues for different levels of SMZ protection for the (a) yellow-poplar and the (b) mixed hardwood stands under the four harvesting technologies (PH=partial harvest, see Table 1 for description of technologies used).

unnecessarily wide buffers produce an avoidable economic loss to the landowner (Castelle and Johnson 2000). The SMZ protection options (no SMZ, and unharvested 15m, 30m, 45m SMZs) resulted in varying levels of post-harvest change for the individual riparian functions (Table 4). The SMZ protection score increases with buffer width (Table 4). No SMZ results in a protection score of 0 percent; it did not protect any of the five categories of riparian function resulting in large changes following the harvest. A 15m SMZ has an SMZ protection score of 60 percent and a 30m SMZ has a protection score of 87 percent (Table 4). A 45m SMZ has a protection score of 100 percent; it protected against measurable changes in all five of the categories of riparian function (Table 4).

Comparing Financial Costs with Ecological Benefits

Forest landowners are responsible for protecting water quality and maintaining riparian habitat for the public good, but they also have other objectives that may include making a return on their investments. The challenge for landowners is to find a balance between financial sacrifice and ecologic protection. To find this balance, we must consider that the revenue reductions attributed to SMZ protection occur only once—at the time of timber harvest—but the ecological benefits of SMZ protection accrue after the harvest and continue through the next rotation. To compare the current costs with future ecological benefits, a capital recovery factor can be calculated to convert revenue reductions to a series of uniform annual costs that begin at the time of harvest and extend through the next rotation.

The capital recovery cost takes the protection costs of retaining an SMZ and, using a real interest rate of 4 percent, divides that cost into annual allotments. These calculations are the per-hectare cost to leave an SMZ for each year of a 120-year rotation. In an ecological context, capital recovery costs can be viewed as the annual monetary cost required to maintain a particular level of riparian function.

Benefit/Cost Analysis

We used the principle of benefit/cost ratios (Gregory 1972) to compare the ecological benefits with the opportunity costs. By comparing the capital recovery costs with the SMZ protection score, we can determine the benefit-cost ratio between ecological protection and SMZ width (Fig. 6a and 6b and Table 5). In summary:

- All B/C ratios were greater than or equal to 1. Thus, it is desirable to use SMZs.
- In all cases, B/C ratios are more robust for 15m SMZs.
- In all cases, B/C ratios are still very desirable for 30m SMZs
- B/C ratios, although still desirable for 45m SMZs, are not as robust as those from 15m and 30m SMZs, suggesting that the benefit produced is decreasing while costs are increasing.

SMZ protection scores increase with capital recovery costs for both stand types and all logging technologies. Harvesting without an SMZ leaves no

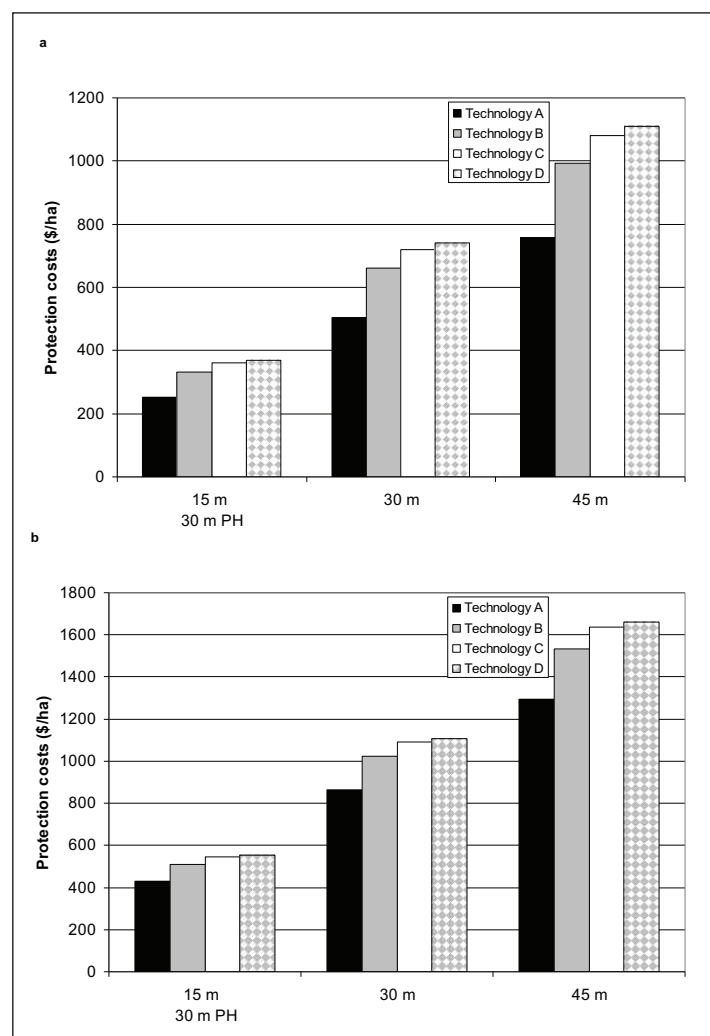


Figure 5— Costs for different SMZ protection options for the (a) yellow-poplar and (b) mixed hardwood stands (PH=partial harvest, See Table 1 for description of technologies used).

Table 4--SMZ protection scores for different SMZ widths for protecting against post-harvest changes in riparian functions for 2 to 4th order streams.

Riparian Function	Width				References
	No SMZ	15 m	30 m	45 m	
Coarse woody debris	0 ^a	1	2	3	Murphy and Koski 1989, Harmon and others 1986, McDade and others 1990, Robinson and Beschta 1990, Van Sickle 2000, May and Gresswell 2003.
Shade/temperature maintenance	0	2	3	3	Burton and Likens 1973, Moring 1975, Brown and Krygier 1967, Rishel and others 1982, Lynch and others 1984, Lynch and others 1985, Noel and others 1986, Beschta and others 1987, Budd and others 1987, Caldwell and others 1991, Kochenderfer and Edwards 1991, Davies and Nelson 1994, Jackson and others 2001, Kiffney and others 2003, Wilkerson and others 2006
Sediment filtering	0	2	2	3	Karr and Schlosser 1977, Moring 1982, Lynch and others 1985, Davies and Nelson 1994, Jackson and others 2001
Aquatic communities (macroinvertebrates and periphyton)	0	2	3	3	Newbold and others 1980, Noel and others 1986, Davies and Nelson 1994, Hetrick and others 1998, Kiffney and others 2003, Wilkerson and others in review ^b
Riparian bird communities (riparian associated passerines)	0	2	3	3	Triquet and others 1990, Whitaker and Montevecchi 1999, Pearson and Manuwal 2001
Total Score	0	9	13	15	
Percent SMZ effectiveness	0%	60%	87%	100%	

^aScoring: 0) Does not protect riparian function; 1) Results in moderate post-harvest changes in riparian function; 2) Results in small post-harvest changes in riparian function; 3) Completely protects against measurable changes in riparian function

^bWilkerson, Ethel; Hagan, John M.; Whitman, Andrew A. In review. The effectiveness of different buffer widths for protecting water quality and biotic communities of headwater streams in Maine. *Freshwater Biology*.

Table 5--SMZ protection scores, capital recovery costs, and B/C ratios by logging technology and stand type.

SMZ Protection Score	Yellow-poplar		Mixed Hardwood	
	Capital		Capital	
	Recovery Cost	B/C Ratio	Recovery Cost	B/C Ratio
-----%	---\$/ha/yr---		---\$/ha/yr---	
		Technology A		
0	0.00	0.00	0.00	0.00
60	10.18	5.89	17.44	3.44
87	20.36	4.27	34.88	2.49
100	30.54	3.27	52.32	1.91
		Technology B		
0	0.00	0.00	0.00	0.00
60	13.37	4.49	20.66	2.90
87	26.74	3.25	41.32	2.11
100	40.11	2.49	61.98	1.61
		Technology C		
0	0.00	0.00	0.00	0.00
60	14.55	4.12	22.04	2.72
87	29.10	2.99	44.08	1.97
100	43.65	2.29	66.12	1.51
		Technology D		
0	0.00	0.00	0.00	0.00
60	14.95	4.01	22.34	2.69
87	29.90	2.91	44.68	1.95
100	44.85	2.23	67.02	1.49

timber adjacent to the stream resulting in an SMZ protection score of 0 percent and no capital recovery costs. Retaining a 15m SMZ results in an SMZ protection score of 60 percent and costs between \$10.18 and \$14.95/ha/year for a yellow-poplar stand (Fig. 6a) and between \$17.44 and \$22.34/ha/year for a mixed hardwood stand (Fig. 6b), depending on the logging technology. A 30m SMZ results in an 87 percent SMZ protection score and cost between \$20.36 and \$29.90/ha/year (yellow-poplar, Fig. 6a) and between \$34.88 and \$44.68/ha/year (mixed hardwood, Fig. 6b). A 45m SMZ achieves a 100 percent riparian protection score but costs between \$30.54 and \$44.85/ha/year (yel-

low-poplar, Fig. 6a) and between \$52.32 and \$67.02/ha/year (mixed hardwood, Fig. 6b).

The relationship between increasing capital recovery costs and increasing SMZ protection score is not linear. This analysis shows that for SMZs wider than 15m, the rate of increasing SMZ protection begins to diminish while capital recovery costs continue to increase (Fig. 6a and 6b). Therefore, increasing streamside protection from a 15m SMZ to a 30m SMZ results in an increase in economic cost that is disproportional to the increase in ecological protection gained. However, if the goal is to completely protect riparian functions against measurable post-harvest changes (a 100 percent SMZ protection score), a 45m SMZ is required and landowners will pay an economic premium to achieve this level of protection. Although we could not calculate an SMZ protection score for the 30m partially harvested SMZ, the capital recovery costs are 50 percent less than the 30 m SMZ without timber removal. Landowners may chose partial removal of timber within the SMZ to reduce capital recovery costs

while still maintaining a portion of riparian structure that can contribute to riparian function.

Considerations for Managers

Ultimately, landowners and managers and concerned public/outside interests must determine the appropriate balance between opportunity/capital recovery costs and SMZ protection. The level of riparian protection will vary between ownerships and even within different landscapes on a single ownership. On an ownership level, managers should consider state and local laws and BMPs, certification requirements, and their long-term strategies for

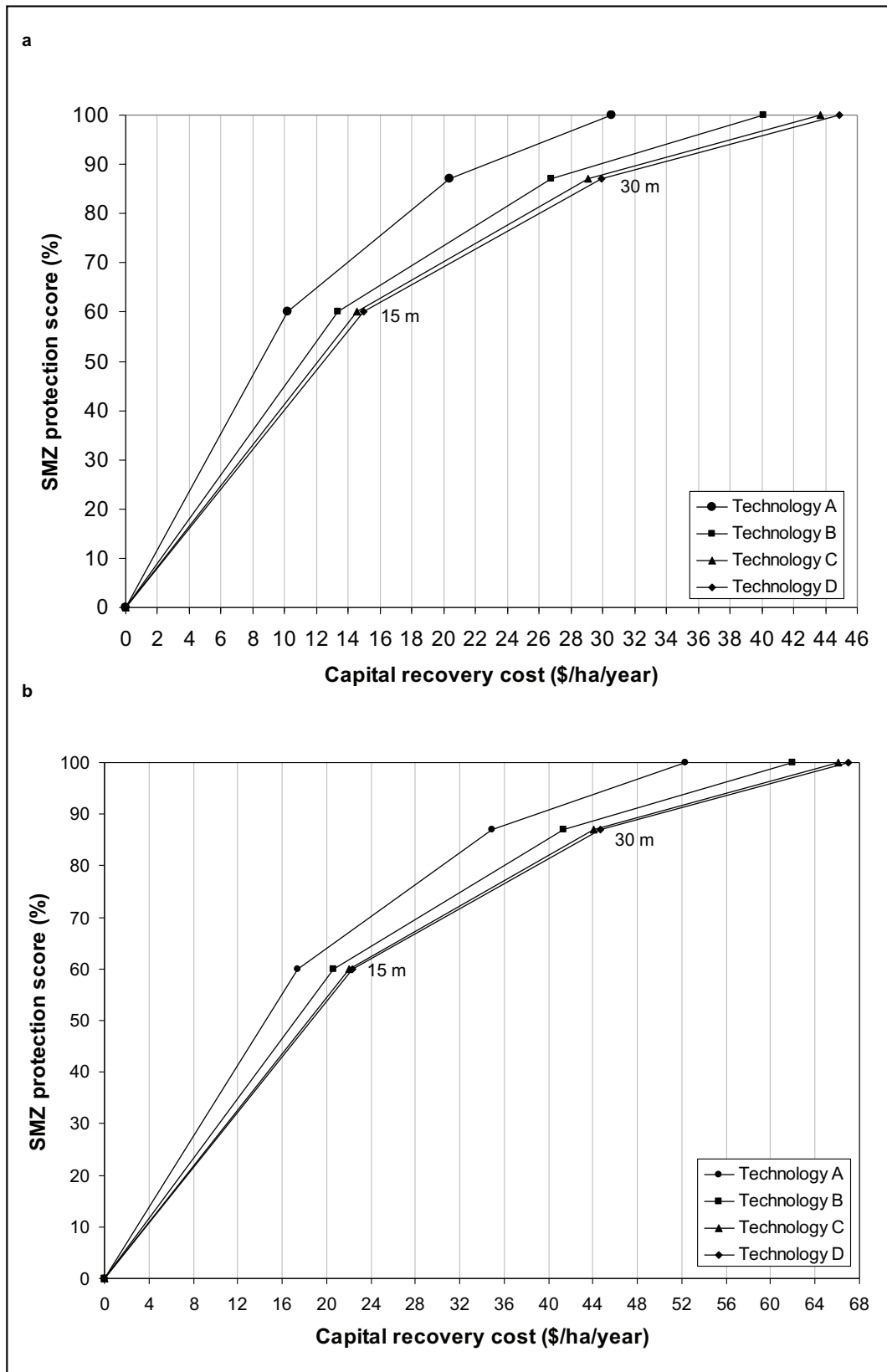


Figure 6--SMZ protection scores compared with capital recovery costs for the (a) yellow-poplar and (b) mixed hardwood stands under the four harvesting technologies. Symbols and lines represent different logging systems. SMZ protection scores are labeled on corresponding SMZ width (See Table 1 for description of technologies used).

maintaining and protecting fisheries and wildlife on their land base. At a smaller scale, managers should consider the slope and topography of the stand, the age class and disturbance history of the surrounding forests, and determine if there are species of special management concern within the watershed.

The decision on which logging system to use to harvest wood adjacent to and from within buffers requires careful consideration of the ecological functions that one wishes to protect. For example, cable logging systems usually require ridge top roads and landings, which may result in less erosion and sediment production. This could justify narrower buffer widths with partial volume removal if one was concerned with erosion and sediment production only. However, if the objective is to also provide habitat for breeding birds (Hanowski and others 2005), amphibians (Perkins and Hunter 2006) and use by some mammals, such as martens (Fuller and Harrison 2005), then wide (45m) buffers with no volume removed may be required.

Using computer simulations and the principles of benefit/cost analysis, we evaluated two stands, four logging technologies, five SMZ protection options, five riparian/ecological functions, a fixed real interest rate of 4 percent, and fixed market prices. The results reported here are specific to the conditions simulated and to the models and assumptions used and should not be generally inferred. However, the results provide an understanding of the costs and ecological benefits associated with alternative levels of SMZ protection.

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Restoring Old-Growth Southern Pine Ecosystems: Strategic Lessons From Long-Term Silvicultural Research

Don C. Bragg, Michael G. Shelton, and James M. Guldin

ABSTRACT

The successful restoration of old-growth-like loblolly (*Pinus taeda*) and shortleaf (*Pinus echinata*) pine-dominated forests requires the integration of ecological information with long-term silvicultural research from places such as the Crossett Experimental Forest (CEF). Conventional management practices such as timber harvesting or competition control have supplied us with the tools for restoration efforts. For example, the CEF's Good and Poor Farm Forestry Forties have been under uneven-aged silvicultural prescriptions for 70 years. Monitoring these demonstration areas has provided insights on pine regeneration, structural

and compositional stability, endangered species management, and sustainability capable of guiding prescriptions for old-growth-like pine forests. Other studies on the CEF's Reynolds Research Natural Area have provided lessons on the long-term impacts of fire suppression, woody debris and duff accumulation, hardwood competition, and pine regeneration failures. This experience leads us to believe the productivity and resilience of these forests can be adapted to create functionally sustainable old-growth-like stands by integrating silviculture and restoration.

Keywords: coarse woody debris, Crossett Experimental Forest, loblolly pine, red-cockaded woodpeckers, shortleaf pine, Upper West Gulf Coastal Plain.

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Figure 1—Examples (a) of the presettlement pine forests that once dominated the Upper West Gulf Coastal Plain of southern Arkansas and (b) of the cutover landscapes found by the early 20th Century.

INTRODUCTION

Restoration efforts are now one of the primary driving forces in national forest management and the research programs designed to support this policy (Bosworth and Brown 2007). For many, this is a radical departure from the traditions of forestry and silviculture, in large part because of the shift in emphasis from timber harvesting to management for a variety of benefits. In reality, the emphasis on restoration shares many of the original aims of forestry. As an example, silviculture was in part designed to rehabilitate degraded forest ecosystems and renew their commercial productivity and natural resilience (Pinchot 1947). Even though the ultimate objective (timber only versus multiple use) has changed considerably, our silvicultural toolkit can still serve us well in the contemporary management of public lands (Guldin and Graham 2007). Indeed, it is the many lessons we have learned on how to manipulate the forest to yield predictable and desirable outcome(s) that permits us to attempt to restore these systems.

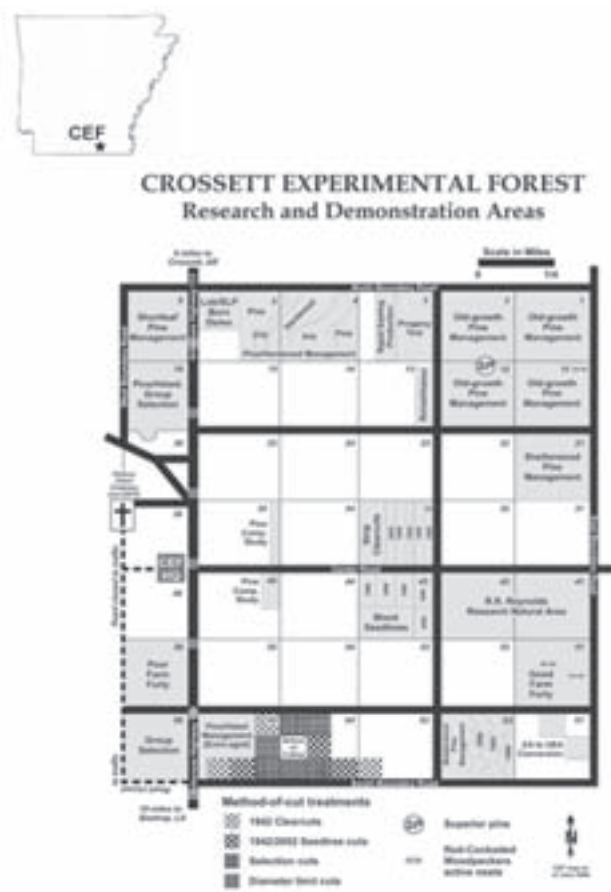


Figure 2—Location of the Crossett Experimental Forest (CEF) and map of the research and demonstration areas.

The establishment and operation of the Crossett Experimental Forest (CEF) in southern Arkansas is a classic example of how the implementation of science-based forestry and silvicultural research was and can still be used to restore landscapes. By the early 20th Century, decades of exploitive logging and land clearing had devastated the virgin pine forests of the region (fig. 1). As the 1920s closed, some in the local timber industry recognized the potential of their lands to sustain productive second-growth pine forests, if they only knew what to do (Reynolds 1980). Even though operations such as the Crossett Lumber Company supported the research and extension efforts of distant university academics, their work was too limited to be of much practical use. Starting in the early 1930s, the Southern Forest Experiment Station of the U.S. Forest Service offered another possibility—the establishment of permanent experimental stations staffed by federal researchers to conduct long-term research and demonstration projects and help landowners manage their properties (Bragg 2005, Demmon 1942, Reynolds 1980).

It was into this environment that Russ Reynolds, a recent graduate of the University of Michigan, came to work with Ozark-Badger, Crossett, and other lumber companies. Reynolds helped carve the CEF out of the cutover landscapes of southern Arkansas (fig. 2). When established in 1934, the CEF offered a means to design, demonstrate, and monitor the long-term response of silvicultural treatments in loblolly (*Pinus taeda*) and shortleaf (*Pinus echinata*) pine-dominated forests (Reynolds 1980) on productive sites (SI50 = 85 to 90 feet). The CEF opened for “business” on January 1, 1934, and over the next few years, Reynolds installed a number of key long-term demonstration and research areas (Reynolds 1980). During the following decades, other research was established that either continued existing projects or addressed new questions based on changing markets, utilization, silvicultural strategies, and resource interests.

As the CEF enters its eighth decade of service, a number of these long-term studies (e.g., the Good and Poor Farm Forestry demonstration areas, Reynolds Research Natural Area (RRNA), Methods-of-Cut) have had many years of detailed observations, and the silvicultural lessons learned in their implementation, maintenance, and analysis continue to develop. In this paper, we describe how information gleaned from these long-term experiments and demonstrations can be adapted to questions arising from a new silvicultural direction—the restoration of stands with old-growth-like characteristics. Unlike some existing large-scale projects in the South (e.g., Stanturf et al. 2004), these lessons are more strategic than tactical because

we are still in the process of adapting our knowledge of traditionally managed southern pine to produce both old-growth attributes and commodities.

SILVICULTURAL CONTEXT

Repeated logging and conversion to other land uses over the last 150+ years have virtually eliminated old-growth loblolly and shortleaf pine-dominated forests that once covered much of the coastal plain of the southern United

Table 1—A sample of the desired standards and guidelines for successful restoration of old-growth-like conditions in loblolly and shortleaf pine stands of the UWGCP in southern Arkansas, adapted from Bragg (2004a)

Attribute	Reference target
Species composition	50 to 60 percent loblolly pine 35 to 45 percent shortleaf pine up to 10 percent hardwoods
Basal area	50 to 70 square feet per acre
Maximum tree DBH/age	unlimited
Number of big trees	5 to 15 pines > 30 inches d.b.h. per acre
Reserved timber volume	5,000 to 10,000 board feet (Doyle) per acre
Spatial pattern	patchy
Under/mid-story	open
Red heart	10 to 50 percent cull in retained trees
Large woody debris	5 to 10 snags (285 to 715 cubic feet) per acre

States. Second-growth natural stands of mixed loblolly and shortleaf pine are still common, although none of them possess all of the attributes of their presettlement versions. Furthermore, naturally regenerated pine stands are under considerable pressure from commercial timber interests, land developers, and other threats to forest health (e.g., southern pine beetle (*Dendroctonus frontalis*)). For instance, although the generic “loblolly-shortleaf pine” cover type has held steady since the early 1950s at about 25 percent of the timberland in the southeastern United States, natural pine stands have declined from almost 72 million acres in 1952 to just under 33 million acres in 1999, with concurrent increases from 2 to 30 million acres of planted pine (Conner and Hartsell 2002).

There is growing interest in using silviculture to restore existing loblolly/shortleaf stands into old-growth-like conditions, especially from some national forests. A classical example of this interest is the pine-bluestem restoration project initiated in the 1990s on the Ouachita National Forest (Stanturf et al. 2004, U.S. Forest Service 1996). This large-scale project has been successful, spawning interest from other national forests desiring the same multiple-use results out of an integrated silviculture and restoration program. Success in one location, however, does not necessarily translate into acceptable results in other regions—there are enough unique attributes in any given landscape to require that every project be at least somewhat custom-designed. As an example, the management prescription for old-growth-like conditions in pine-dominated upland forests of the Upper West Gulf Coastal Plain (UWGCP) of southern Arkansas, northern Louisiana, southeastern Oklahoma, and northeastern Texas (Bragg 2004a) may not work in the shortleaf pine forests of the Ouachita or Ozark mountains, or even the hardwood-dominated forests of the UWGCP.

The successful restoration of old-growth-like forests requires the integration of historical information with lessons learned in places such as the CEF. Long-term silvicultural research and demonstration projects, some of which date back to the mid-1930s, can be modified into new strategies for multi-resource objectives. This paper provides examples taken from these projects on the CEF and related studies, and translates the lessons from conventional silvicultural treatments to the emulation of old-growth-like conditions. Ultimately, our prescriptive goal is not to reproduce an exact

replica of a presettlement old-growth, but to incorporate as many of these key attributes as possible in a stand that is functionally self-supporting and actively managed.

WHAT ARE OLD-GROWTH-LIKE CONDITIONS?

Managing for old-growth-like conditions in the pine-dominated forests of the UWGCP refers to the encouragement of forest conditions dominated by features and processes akin to old forests of the presettlement (circa 19th Century) era. In other words, we aim to duplicate many of the same structural, compositional, and functional attributes of the virgin pine forests of the region (see fig. 1a) in contemporary stands still managed for some degree of commodity production. Rather than using a fixed set of narrowly defined criteria to judge success of this restoration effort, our standards and guidelines (table 1, see also Bragg 2004a) call for a range of conditions to be emulated using conventional silvicultural tools such as thinning, reproduction cutting, competition control, and even supplemental planting.

Unfortunately, we have only spotty information on what presettlement pine-dominated forests were like in the UWGCP. According to anecdotal reports, pines in excess of 40 inches d.b.h. (diameter breast height) or greater were common, and some individuals apparently exceeded 70 inches d.b.h. and over 5,000 board feet (unless otherwise specified, all board measures are Doyle scale) of lumber (Bragg 2002, 2003). These large trees, while very noticeable, were not so numerous that stand densities or sawtimber yields were particularly high. In a review of available information, Bragg (2002) noted that stand sawtimber volumes often ran between 5,000 and 20,000 board feet per acre, with most ranging between 10,000 and 15,000 board feet per acre. Annual growth data was even spottier, although some mature- to old-growth pine stands from southern Ashley County added between 50 and 120 board feet per acre per year (Chapman 1912, 1913). Size class patterns varied, but most exhibited at least a few pines across a broad range of diameters.

Hence, when compared to modern, managed examples of uneven-aged loblolly and shortleaf pine in the UWGCP, these virgin forests grew appreciably slower, contained greater volumes of sawtimber in (typically) larger trees,

and probably had considerably fewer small stems. While intensively-managed uneven-aged pine stands contain dense pockets of regeneration and thus are not relatively open, it is likely that modifications to an uneven-aged prescription can be made to develop regeneration cohorts in a more episodic manner. It is possible that they could even be treated with prescribed fire. This would eventually produce a structural effect similar to that which is reported for virgin pine forests. Both of these would have relatively open canopies with multiple size classes distributed throughout the stand. Such a functional similarity would make it possible to emulate at least some aspects of old-growth-like forests while simultaneously supporting timber production (Bragg 2004a).

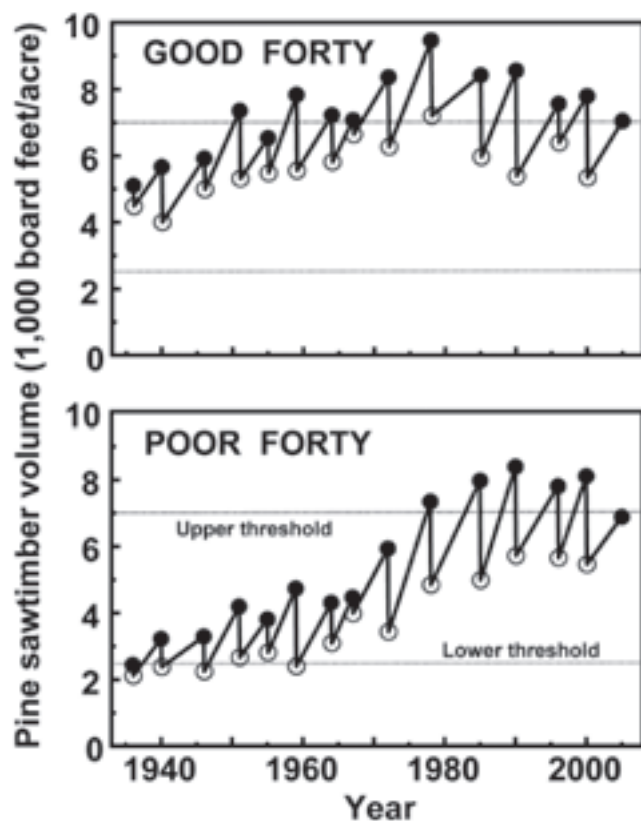


Figure 3—Pine sawtimber volume (Doyle scale) in the Good and Poor Farm Forestry Forties on the Crossett Experimental Forest (CEF) over a 69-year period. The filled symbols are before harvest and open symbols are after harvest. The dotted lines are the upper and lower stocking thresholds recommended by Baker et al. (1996). From Guldin (2002) and unpublished data on file at CEF.

UNEVEN-AGED MANAGEMENT AND RESTORATION SILVICULTURE

Loblolly and shortleaf pine forests can be managed under a number of different uneven-aged silvicultural methods, using either single-tree selection or group selection (Baker et al. 1996). The silvicultural key to making these methods successful involves maintaining a critical balance in stand stocking—a sufficient overstory is retained so that growth and yield continues at high rates, but this overstory is also periodically reduced to permit pine regeneration and canopy recruitment. In addition to the generation of quality pine sawtimber, uneven-aged silviculture supports many other non-timber resources. The structure and composition of mature, uneven-aged southern pine stands are considerably better for co-managing certain types of wildlife species. The Good Forty, for example, contains two active red-cockaded woodpecker (RCW) (*Picoides borealis*) clusters. The RCW has become an endangered species in large part because the older pines suitable for nest cavities have been largely eliminated in the younger, more intensively managed forests of the South (Conner et al. 2004a, 2004b). The loss of mature, open pine woods has greatly contributed to the decline of the RCW (Saenz et al. 2001).

The Good and Poor Farm Forestry Forties on the CEF have been successfully managed using uneven-aged silviculture with single-tree selection for 70 years. Some may find the success of uneven-aged silviculture in loblolly and shortleaf pine-dominated stands surprising, due to the relative shade-intolerance of these species. However, decades of demonstration work in the Good and Poor Farm Forestry Forties have shown the efficacy of this technique, given some adjustments for species autecology and local site conditions (Guldin 2002). During these years, our monitoring of these parcels has provided critical information about pine regeneration, structural and compositional stability, and resource sustainability capable of guiding prescriptions for old-growth-like pine forests.

The initially poorly stocked Poor Forty was established to show how stands could be rehabilitated using uneven-aged silviculture, while the better-stocked Good Forty was established to document rates of growth and yield once rehabilitation had been achieved. And grow they did! Long-term records (from 1936 to 2005) have shown annual growth averages about 400 board feet per acre



Figure 4—Contrasts in the structure and composition between the Good Forty (a) and the RRNA (b) are obvious when these stands on the Crossett Experimental Forest are compared.

in the Good and Poor Forties. Stocking was rapidly improved in the Poor Forty by harvesting less than growth, while the well-stocked condition in the Good Forty was essentially maintained by harvesting the periodic growth (fig. 3). These cuts, now done about once every 5 years, remove mostly mature, high-quality sawtimber. Harvests over this 69-year period totaled 24,000 board feet per acre in both the Good and Poor Forties. In addition, standing pine volume in 2005 was 50 to 200 percent greater than in 1936 (fig. 3). Throughout this period, the Good and Poor Forties retained a component of big trees, averaging 5 trees per acre with d.b.h. of 20 inches and larger before the periodic harvests.

THE HISTORICAL AGE STRUCTURE OF UNEVEN-AGED PINE STANDS

There is no reliable information on the age structure of presettlement forests of loblolly and shortleaf pine in the UWGCP. Few stands of old-growth pine or pine-hardwood timber remain, making it difficult to generalize using these remnants as models. However, if appropriately applied, some inferences can be made that have important silvicultural consequences. After examining recent stumps following salvage and limited increment coring, Bragg (2006) reported that the pine overstory in the Levi Wilcoxon Dem-

onstration Forest (LWDF) in Ashley County, Arkansas, was uneven-aged. Overstory pines ranged from about 80 years old to as much as 300 years old, and were primarily recruited between 1800 and 1920. In a bottomland hardwood-loblolly pine stand in nearby Calhoun County, Heitzman et al. (2004) placed pine establishment during three primary periods: 1850-60; 1861-90; and 1981-90.

Hence, it appears that many (if not most) of the presettlement pine forests of the UW-GCP were at least broadly uneven-aged, with periodic small- and large-scale distur-

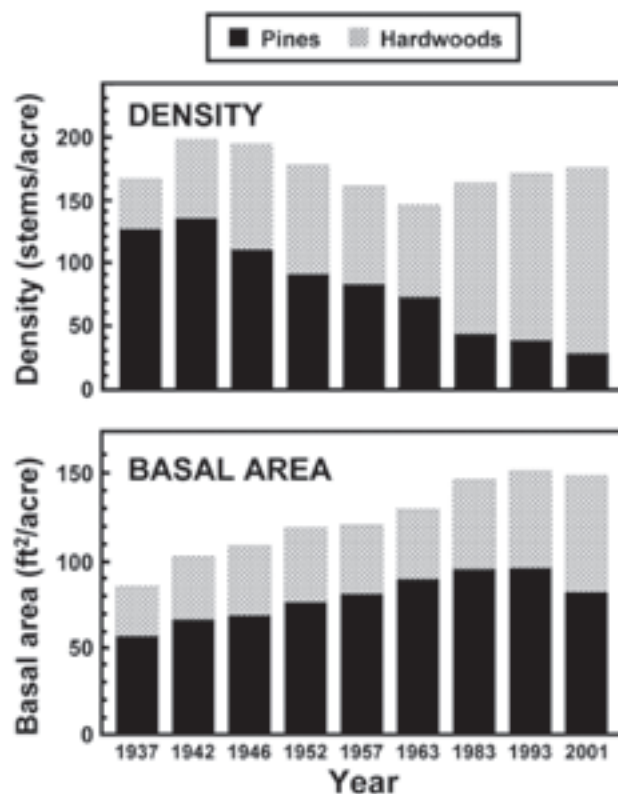


Figure 5—Successional changes in pines and hardwoods 4 inches d.b.h. and larger over a 64-year period in the Reynolds Research Natural Area on the Crossett Experimental Forest (CEF). Adapted from Cain and Shelton (1996) and updated with unpublished data on file at CEF.

bances allowing for the establishment of new cohorts of regeneration (Bragg 2006, Heitzman et al. 2004). This produced scattered individuals or patches of older “veteran” trees intermingled with more extensive areas of maturing timber (Bragg 2002). This pattern is consistent with stand maps made in southern Ashley County from Chapman (1912) that distributed large, decadent old pines in a matrix of maturing trees. Areas of even-aged stands arising from natural disturbances were also noted elsewhere in Arkansas (Turner 1935), and probably comprised a significant portion of the presettlement landscape. From a restoration perspective, this distribution implies that at least some degree of age variation should be present for old-growth-like conditions. This, in turn, suggests that the stands be treated in a manner capable of producing multiple age classes.

Even though fire was an important dynamic in the presettlement pine forests of the UWGCP, conventional wisdom holds that intensively managed uneven-aged pine stands are best managed without fire. However, work on the CEF has shown that fire can be compatible with uneven-aged mixed pine forests. A 19-year study of different burning regimes showed that, though not very efficient at producing large numbers of saplings, low- to moderate-frequency dormant season fires did not completely eliminate them from the treatment areas (Cain and Shelton 2002). Regularly applied dormant season fires were found to only temporarily suppress competing hardwoods, and the fires also killed much of the pine regeneration. Thus, the authors suggested that fires be applied in conjunction with herbicide use, which would allow pine regeneration to more quickly reach fire-tolerant size (Cain and Shelton 2002). Another adaptation using fires to control species composition in uneven-aged pine stands would involve an interrupted burning cycle. Repeated dormant season prescribed fires would be used to control competing hardwoods, but this burning cycle would be interrupted long enough for pine seedlings to become established and to grow to fire-tolerant sizes. After a brief interval, the burning cycle would then be reestablished (Cain and Shelton 2002).

Residual old trees are also important for the restoration of old-growth-like conditions. Unlike most pine plantations in the UWGCP, which are generally harvested after 30 to 35 years when the pines have just reached reasonable sawlog size, uneven-aged stands will frequently have trees greater than 70-years old, and often these trees are of considerable

diameter and height (Guldin and Fitzpatrick 1991). The older component of uneven-aged pine stands often possesses a greater incidence of red heart fungus (*Phellinus pini*), an attribute critical to RCW nest cavity excavation (Conner 2004a, 2004b). Since the current managing for old-growth-like condition strategy calls for the retention of some of the oldest and largest pines regardless of how old they may be (Bragg 2004a), these trees provide attributes such as RCW cavities and aesthetics without excessively compromising fiber production.

LESSONS FROM UNMANAGED STANDS OF LOBLOLLY AND SHORTLEAF PINE

The dynamics and long-term composition of the RRNA have also taught us much about the behavior of unmanaged forests in the UWGCP. This research natural area, an 80-acre no-harvest reserve on the CEF, has been sheltered from fire and logging for over 70 years. When established, the RRNA was intended to showcase stand development in an unmanaged parcel compared to the adjacent managed Good Forty (fig. 4). The long-term dynamics of the RRNA have vastly differed from those of the Farm Forestry Forties. Even though some structural elements of the unmanaged RRNA are similar to the virgin forests of the region (for example, older pines, large quantities of dead wood), others are more comparable to managed stands.

For instance, after decades of stable stocking, net growth, and increasing yields, pine mortality rates in the RRNA have recently increased considerably, while growth has slowed and pine recruitment dropped to virtually nothing (fig. 5). Pine no longer dominates the overstory or understory composition of the RRNA, while the Farm Forestry Forties are heavily pine dominated in all age and size classes (figs. 4 and 5). This disparity is largely the result of lower over- and midstory density and the use of herbicides to control competing vegetation in the Farm Forestry Forties. Presettlement UWGCP forests typically had a variable and potentially significant component of hardwoods (Bragg 2002), but pine would have been very prominent, and its gradual disappearance in the RRNA does not bode well for the sustainability of conifers in this formerly pine-dominated tract.

Over the last half-decade, pine basal area in the RRNA and the nearby LWDF has continued to decline precipitously (Bragg 2006). Mortality is a natural component of any stand of timber, managed or otherwise, but it must be carefully regulated when restoring old-growth-like conditions so as not to lose the key overstory component that defines the type—pine. At times, preemptive harvests of pine beetle spots, storm damaged timber, or even large-scale salvage logging may be necessary to ensure that the overall health of the pine overstory is maintained. However, it is also important to ensure that at least some accumulation of dead wood is allowed, as this is a critical ecological attribute of old-growth (see later section).

Net pine growth in the first half of the 1990s became negative for the RRNA, with mortality losses double that of survivor growth and zero contribution from recruitment (Shelton and Cain 1999). While growth does not need to be maximized or optimized in a silvicultural strategy focused on restoring old-growth-like stand conditions, there is need for positive increment in at least some of the merchantable size classes. Presumably, the relatively open stand conditions and continual recruitment of young pines to the overstory will ensure that there is enough fiber pro-

duction to support periodic harvests to regulate overstory density. The long-term regenerative success of the Farm Forestry Forties, coupled with their high-value sawtimber yield, suggest that it should be possible to maintain desired structures within these old-growth-like stands by encouraging vigorous smaller classes, even if mortality in the largest classes (retained for their contributions to stand structural complexity) is high at times.

RETAINING A PINE OVERSTORY VIA NATURAL REGENERATION

The failure of pines to replace themselves in the RRNA and the LWDF is a clear lesson on how not to perpetuate a pine-dominated canopy. Pine regeneration is a complicated process, involving the impacts of stand conditions, fire suppression, hardwood competition, and woody debris and duff accumulation. The presettlement pine-dominated forests were typically self-replacing, propagated by frequent fire and other large-scale, intense perturbations that produced adequate pine regeneration conditions while simultaneously restricting the success of competing hardwoods. For our old-growth restoration efforts to be successful, we must be

Table 2—Quantitative expression of some factors affecting the success of establishing pine regeneration and providing for its development in the Reynolds Research Natural Area (RRNA) and the Good Farm Forestry Forty on the Crossett Experimental Forest (CEF)

Factor ^a	RRNA	Good Forty
Mean seed production per acre	503,000	418,000
Median seed production per acre	60,000	195,000
Litter depth: inches	1.0	1.1
Light intensity near ground: percent of full sunlight	4	66
Pine basal area, trees ≥ 4 inches d.b.h.: ft ² /acre	82	63
Hardwood basal area, trees ≥ 4 inches d.b.h.: ft ² /acre	68	1

^a Compiled from several published sources providing data and/or background information on methods (Cain and Shelton 1995, 2001; Guo and Shelton 1998; Shelton and Cain 1999) and unpublished data on file at CEF. Seed production was for 16 years (1991 through 2006); tree inventories were from 2000; litter depth was determined in 1994, about 4 years after the Good Forty was last harvested.

able to emulate this result without investing heavily in artificial regeneration or competition control. After all, low-cost, low-impact, self-replacing pine stands can produce both environmental complexity characteristic of presettlement old-growth and make it much easier to convince landowners to sustain their efforts over the long run.

First, to sustain the pine overstory for the foreseeable future, conditions must permit the recruitment to the canopy. For naturally regenerated pine stands, the key factors to ensure adequate regeneration include an ample seed supply, an acceptable seedbed, and sufficient amounts of limited resources such as light, water, and nutrients (Shelton and Cain 2000). Seed production and seedbed conditions generally affect the initial establishment of regeneration, while its subsequent development is largely determined by the amount of limited resources that are available to established seedlings. These regeneration factors are also under a varying level of silvicultural control. For example, both loblolly and shortleaf pine tend to experience a bumper seed crop, a seed crop failure, and three average seed crops during a 5-year period (Cain and Shelton 2001). Al-

though silviculturists can do little to modify this natural variation, they can determine the number, size, vigor, and quality of the seed-producing trees that occupy the site. In contrast, seedbed conditions and the resources available to the species targeted in regeneration are more easily modified by tools such as site preparation, competition control, and harvesting.

Some of the factors that our silvicultural experiences have found to affect pine regeneration can be contrasted for the RRNA, where the pine component is not being sustained (Cain and Shelton 1996), and the adjacent Good Farm Forestry Forty, with its stable pine dominance. These differences did not arise from seed limitations. Pine seed production was ample in both areas (table 2). Historically, seed production in the unmanaged RRNA has exceeded that of the Good Forty by an average of about 25 percent. This difference was mainly due to exceptionally high production in the RRNA during bumper years. The effects of stand management are more apparent by considering the median value for seed production, where the Good Forty exceeded that of the RRNA by three times.

Table 3—Density of pine regeneration in the Reynolds Research Natural Area (RRNA) and Good Farm Forestry Forty on the Crossett Experimental Forest

Year ^a	Seedlings ^b		Saplings	
	-----		-----	
	RRNA	Good Forty	RRNA	Good Forty
	<i>per acre</i>		<i>per acre</i>	
1982	8 ^c	1,210	0	80
1993	733	6,920	0	480
2001	110	825	0	239

^a Sampling methods vary from sixty 0.002-acre subplots to one hundred 0.001-acre subplots.

^b Seedlings have d.b.h. <1 inch; saplings have d.b.h. of 1 to 3 inches.

^c Does not include seedlings <0.5 feet tall.

Likewise, seedbed conditions in the two areas also were found to be similar. Both areas consisted of a litter seedbed with a depth of about 1 inch (table 2). However, the litter depth reported for the Good Forty in table 2 was measured 4 years after the most recent harvest. Logging has both positive and negative effects on the germination of pine seeds. Harvesting creates favorable conditions by displacing forest floor litter and exposing mineral soil, but it also creates an unfavorable seedbed in certain areas covered by logging debris. Shelton and Cain (2000) reported that typical seedbed conditions following harvesting of the Good Forty was 40 percent undisturbed litter, 30 percent disturbed litter, 25 percent logging debris, and 5 percent exposed mineral soil.

The biggest difference in the conditions for regeneration between the RRNA and Good Forty was expressed in light levels near the ground, which averaged 4 percent of full sunlight in the RRNA compared to 66 percent in the Good Forty (table 2). Even though loblolly and shortleaf pine seedlings are moderately shade-tolerant during their first few years, seedlings become more intolerant to shade as they develop. Thus, the low light environment under the relatively continuous, dense canopy of the RRNA strongly limits pine regeneration. Pulses of pine seedlings can establish in the RRNA during years with high seedfall and ample summer moisture, but these seedlings die long before reaching sapling size (table 3). In contrast, pine canopy recruitment is continuous in the Good Forty, where enough seedlings grow into saplings and saplings grow into merchantable trees to sustain the pine overstory.

The favorable light environment for the overstory recruitment of pine was maintained in the Good Forty by periodic harvests and repeated control of non-pine competing vegetation, principally using selective herbicides. These silvicultural activities have virtually eliminated merchantable hardwoods in the Good Forty, while the RRNA in comparison had 68 square feet per acre of hardwood basal area (table 3). Due to their broad leaves and large crowns, hardwoods generate about twice the level of shade as do pines per unit of basal area (Guo and Shelton 1998), so the acceptable level of hardwoods dispersed amongst the pines is inherently low. A basic tenet regarding competition control is that as site quality increases, the aggressiveness of competition control must also increase (Shelton and Cain 2000).

In addition to its impact on pine regeneration, the dense hardwood under- and midstory of the RRNA has affected other ecosystem properties. Traditionally, uneven-aged stands have a much more open canopy structure than plantations because uneven-aged stands are managed at lower stocking levels. This openness, once prevalent in the presettlement landscapes, permits certain ecosystem attributes not possible in intensively managed stands. For instance, with some minor modifications to traditional uneven-aged practices (for example, retention of large red heart infected pines, reducing the density of the midstory near cavity trees), sustainable RCW clusters can be incorporated. While the RRNA contains some of these attributes (for example, widespread occurrence of red heart in large pines), there are no active nest clusters in this stand. Even though it can be argued that the perpetuation of the RCW clusters in the Good Forty has been made possible, in part, by the use of nest box inserts, without the open nature of the uneven-aged stands of the CEF, these clusters would not have been able to persist.

Because management for old-growth-like conditions in pine-dominated forests implies the direct manipulation of the physical environment, there is considerable flexibility in how regeneration is achieved. As an example, the successful regeneration of pine in the Good Forty involves regulation of the overstory pine through volume-guided harvests typically conducted about every 5 years. Ideally, harvests are scheduled so that the volume in merchantable trees does not exceed 7,000 board feet per acre, which is the observed threshold level at which the overstory competition begins to unacceptably suppress recruitment (Baker et al. 1996). Without the ability to recruit new pines to the canopy, any uneven-aged silvicultural system intended to perpetuate a pine overstory would quickly fail. Harvesting, seedbed disturbance, and chemical and/or fire-based competition control are necessary to ensure viable pine recruitment. Even supplemental planting may prove to be the best strategy under certain conditions.

DEAD WOOD MANAGEMENT

Overstory attrition, whether in intensively managed seed-tree or uneven-aged stands or unmanaged old-growth, is a continual process. Under most circumstances, large crop trees are very exposed, and hence particularly susceptible to mortality agents such as ice storms, lightning,

or windthrow. Losses to bark beetles and other insect pests, disease, wildfire, and logging damage further reduce the number of canopy pines that must eventually be replaced. While it is possible to mitigate these losses with well-planned harvest entries, salvage of dead or declining trees, and other protective techniques, there is no way to eliminate tree mortality.

To some degree, mortality losses are more than just a cost of doing business, they are an ecological necessity. For instance, without dead trees (or the loss of large parts of live trees), it is impossible to accumulate coarse woody debris (CWD), a critical habitat element that serves many vital ecosystem functions (Harmon et al. 1986, Spetich et al. 1999, Van Lear 1996). CWD volumes tend to be higher in unmanaged forests. When last measured, the RRNA exceeded 1,700 cubic feet per acre of CWD, compared to 470 cubic feet per acre in the occasionally salvaged LWDF and only 214 cubic feet per acre in managed stands on the CEF (Bragg 2004b, Zhang 2000). Conceivably, an intensively cultured stand with continual salvage and high rates of product utilization may have almost no CWD. This is a desirable outcome when fiber production is the primary goal, but an unwanted simplification when managing for old-growth-like conditions (Harmon et al. 1986, Spetich et al. 1999).

In the end, a stand managed for old-growth-like conditions needs to retain a large quantity of CWD—perhaps not to the extent seen in the RRNA, but more than that found in conventionally managed stands of mature pine timber. Salvage operations to remove dead and dying trees should be limited to circumstances where either an insect or disease outbreak threatens management objectives, or where safety factors override environmental goals (for example, along roads, trails, campsites, near buildings). Otherwise, moribund trees should be left to expire on their own accord. Snags should be allowed to accumulate and fall. The less than complete utilization of all the trees on these sites is not necessarily a bad thing.

ANTICIPATED CHALLENGES

Even though we expect this silvicultural system to work, a number of challenges in its implementation and long-term success are expected. Certainly, total fiber and sawlog volume production will be lower under this strategy

when compared to more conventional treatments such as uneven-aged silviculture. Logistical issues will also arise. For instance, marking the appropriate pines for removal in harvests will take additional training to ensure that the desired ecosystem attributes are retained. We also expect there to be some difficulty in getting loggers to harvest the stands without excessively damaging some of the residual trees and the pine regeneration, as most are no longer accustomed to cutting uneven-aged stands.

While these challenges can be addressed through additional training and discussions (coupled with close monitoring), other factors are harder to control and may require modifications to management strategies over time. As an example, shortleaf pine is noticeably harder to regenerate than loblolly on the UWGCP. This tendency is likely due to long-term decreases in shortleaf pine overstory density from discriminatory harvesting and shifts in disturbance regimes from one driven by fire (favoring shortleaf) to one influenced primarily by logging and land clearing, which favors the reproductive proclivity of loblolly. Hence, it may be necessary to facilitate shortleaf pine during the canopy recruitment stages of stand development by discriminating against loblolly. Even this may still not produce the desired outcome of significantly greater shortleaf canopy representation.

CONCLUSIONS

Using these lessons, we believe the productive potential and resilience of mixed loblolly/shortleaf pine forests can be adapted to create functional examples of old-growth-like stands capable of sustainably producing both timber and non-timber outputs. Although considered shade-intolerant, both loblolly and shortleaf pine possess characteristics that make them conducive to the creation and maintenance of old-growth-like conditions. These include their moderately long-lived nature, adaptations for fire, rapid growth on a wide range of soils and sites, favorable timber characteristics, an ability to recover well from logging and weather damage, response to release following suppression, and the ease in which loblolly and shortleaf pine regeneration is secured. Fortunately, our conventional silvicultural tools are exactly the instruments of change required to take advantage of these characteristics.

In addition to the conduciveness of the primary overstory species involved, there are less direct lessons from our long-term silvicultural research that can be adapted for more successful restoration projects. For instance, we have found that managing for a particular ecosystem element, such as the stand openness required to regenerate a pine overstory in uneven-aged stands, also encourages structural and compositional characteristics that can contribute desirable habitat for wildlife species (Conner et al. 2002, Thill et al. 2004). Translated to old-growth-like stand management, this suggests that more of the physical and biotic environments will be restored, furthering the value of the ecological rehabilitation of the landscape.

A tentative blueprint for the implementation of restoration silviculture in pine-dominated forests in the UWGCP proposes to use a combination of timber harvesting, competition control, and adaptive management to restore existing stands into a similar composition, structure, and dynamics as the old-growth stands that once characterized the region (Bragg 2004a). Any such design would not have been possible without the decades of long-term experiments and demonstrations to illustrate how these forests behave under certain controlled conditions. This is a powerful argument for the continuation of the use of experimental forests and long-term projects in the U.S. Forest Service's research and development program.

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Adaptive Management of Young Stands on the Tongass National Forest

Michael H. McClellan

ABSTRACT

The idea of adaptive management as a means of improving natural resource management has been around since the 1970s, but to date there have been relatively few examples of fully successful adaptive management efforts in forest management. In 2001, the Tongass National Forest and the Pacific Northwest Research Station began a series of adaptive management studies aimed at improving the management of young, even-aged conifer stands for multiple values: the Tongass-Wide Young-Growth Studies, or TWYGS. This effort is notable both for its scale and for its successes to

date—the Tongass devoted its entire timber stand improvement program for two years to the establishment of the first three operational scale experiments, each replicated widely across southeast Alaska, and a fourth experiment has subsequently been implemented. The experiments are continuing, but the successes to date in assessing the problem, and designing and implementing the studies are largely due to capitalizing on the respective strengths of management and research, and could serve as a model for answering important management questions elsewhere.

Keywords: adaptive management, young growth, silvicultural treatments, wildlife habitat

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INTRODUCTION

The 17-million acre Tongass National Forest is our nation's largest national forest and it includes over 85% of the forest land in southeast Alaska. In the past five years, the Tongass has embarked on an ambitious adaptive management program to improve its management of young stands for wood production and wildlife habitat. It is too early to judge the full success of this program, but the early steps in the adaptive management cycle have been successfully completed and they provide useful examples of adaptive management at work at a regional scale.

Large-scale timber harvesting on the Tongass began in the 1950s and created 430,000 acres of young, even-aged stands. This is a relatively small part of the total area of the Tongass, but harvesting tended to be focused on the more productive and accessible stands along shorelines, valley bottoms, and gentle, lower elevation slopes. Old-growth stands in these areas have high value for wildlife species such as the Sitka black-tailed deer (*Odocoileus hemionus sitkensis*) because they have an abundance of understory forage plants and in winter they have less snow cover than higher elevation or more open sites. The dense, even-aged conifer stands that typically develop after clearcut logging, however, are recognized as having negative consequences for wildlife and fish (Wallmo and Schoen 1980; Schoen et al. 1988; Thedinga et al. 1989; Hanley 1993; Dellasala et al. 1996), primarily due to the loss of understory plants. The moderate temperatures and abundant rainfall of southeast Alaska foster abundant natural regeneration of western hemlock (*Tsuga heterophylla* (Raf.) Sarg.), Sitka spruce (*Picea sitchensis* (Bong.) Carr.), and other conifers following clearcutting. Within ten years of clearcutting, newly established conifers begin to overtop the shrubs and crown closure may be complete by 25 years. Few understory plants can survive in the low light levels of this stem-exclusion stage, and this condition may persist until the stand reaches 120-150 years of age (Alaback 1982, 1984; Hanley 2005).

Throughout the 1990s, land managers on the Tongass implemented a variety of silvicultural treatments in an attempt to promote the growth of understory plants. Their motivation was to improve habitat conditions for the Sitka black-tailed deer, a species valued as a game animal, subsistence food source, and prey for large carnivores such as the

Alexander Archipelago wolf (*Canis lupus ligoni*) and bears (*Ursus americanus* and *U. arctos*). Most of the treatments were variations of precommercial thinning and they often included wide or variable spacing, unthinned patches or corridors, and sometimes gaps where all trees were cut. These operational field trials rarely followed the principles of experimental design and lacked appropriate controls, replication, and random assignment of treatments. Each ranger district worked independently and there was no forest-wide coordination of testing. Follow-up monitoring, evaluation, or reporting of these trials was done in only a few cases. By the end of the 1990s, it was apparent that this approach was not working and that reliable information would come only from a more organized and coordinated effort. The Tongass consulted with researchers from the Pacific Northwest Research Station and representatives from other federal and state agencies, and they decided to use forest-wide adaptive management studies to evaluate their young-growth silvicultural options. This action was aided by two developments in the late 1990s. First, the Tongass transformed its organizational structure from three areas—Chatham, Stikine, and Ketchikan, each having a Forest Supervisor—to a single administrative unit with one Forest Supervisor. This made it much easier to apply a uniform approach forest wide. Second, there was significant scientist involvement in the revision of the Forest plan from 1995 to 1997 (Everest et al. 1997).

Adaptive management is more than simply learning from experience or trial and error. It is a formalized, systematic, and deliberate approach to learning from management actions (Nyberg 1999). The concept was developed during the 1970s (Holling 1978) and 1980s (Walters 1986), and there have been attempts to develop these concepts and apply them to forest management questions in the Pacific

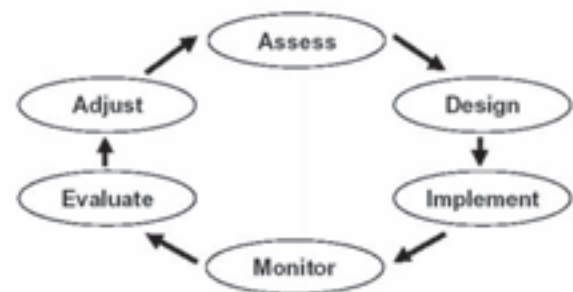


Figure 1— The adaptive management cycle of activities (from Nyberg 1999).

Northwest and elsewhere (Bormann et al. 1994, Stankey et al. 2005). The system of adaptive management can be viewed as a continuous evaluation of management alternatives based on a repeating cycle of activities (Figure 1): assess the problem, design a management experiment, implement the experimental treatments, monitor the response to the treatments, evaluate the monitoring data, and, finally, adjust management policy and actions in line with the adaptive management findings (Nyberg 1999). There have been many attempts to apply adaptive management approaches to forest management issues, not always successfully (Nyberg 1999, Stokstad 2005, Marmorek et al. 2006).

The process of adapting management activities in light of new knowledge may proceed in three ways (Walters and Holling 1990): (1) an evolutionary or trial-and-error pathway; (2) passive adaptive management, where existing knowledge is used to select a single “best” treatment to test; or (3) active adaptive management, where uncertainty regarding outcomes is recognized and a range of alternative treatments are tested concurrently. The second, passive, approach is institutionalized in the concept of “best management practices.” The Tongass chose to use the active approach, even though it would require more resources to implement multiple treatments.

There are other means of learning new ways to manage. Historically, most silvicultural studies have employed relatively small plots (0.1 to 1 acre) and focused on a small group of resource values—an example of this on the Tongass would be the Cooperative Stand-Density Study (DeMars 2000). More recently, silvicultural studies worldwide have addressed a broader range of resource management objectives, and in doing so, have typically employed operational scale treatment units (Peterson and Monserud 2002, Szaro et al. 2006). An example of this approach on the Tongass would be the Alternatives to Clearcutting study (McClellan et al. 2000). Both of these approaches contain elements of the adaptive management approach, but they differ in that they are typically researcher-driven and are implemented generally outside of the day-to-day operations of the land management agency. The present study, and adaptive management studies in general, can be distinguished by the fact that learning as an objective is incorporated into the normal program of work.

FIRST STEP: ASSESSMENT OF YOUNG-GROWTH MANAGEMENT OPTIONS

A significant body of scientific research has documented the negative effects of the stem-exclusion stage on understory plant communities (Alaback 1982, 1984; Hanley 2005). What has not been clear is the best method of mitigating ill effects, either by shortening its duration or lessening its intensity. Managers and researchers discussed various options in a series of workshops, white papers, and field trips during 2000-2001, culminating in an interagency meeting in November 2001 attended by representatives from the Tongass, the Alaska Region of the Forest Service, the PNW Research Station, the U.S. Fish and Wildlife Service, and the Alaska State Department of Fish and Game. The group was asked to:

- 1) prioritize young-growth management information needs
- 2) develop standardized young-growth silvicultural prescriptions for evaluation
- 3) develop implementation protocols for each prescription that were practical from a management viewpoint as well as being consistent with good experimental study design (i.e., including controls, randomization, and replication)
- 4) develop a plan for peer review of prescriptions and protocols

As part of this process, the group reviewed the existing silvicultural information, including new results regarding precommercial and commercial thinning, pruning, and the development of mixed stands of red alder (*Alnus rubra* Bong.) and conifers. Since 1950, precommercial thinning has been performed on approximately 204,000 acres of young-growth stands on the Tongass, roughly one-third of the available acres (McClellan 2005). Typically, thinning was used as a method to increase merchantable wood outputs, but other possible benefits include delaying the onset of the stem exclusion stage, increasing understory plant diversity, and improving wildlife habitat. Attempts to reestablish understory herbs and shrubs through thinning young-growth conifer stands have met with mixed success, however. For example, Deal and Farr (1994) found that thinning of young even-aged stands promoted tree growth but not herbaceous colonization, and that wide spacing resulted in a second episode of western hemlock regeneration.

As part of the Alaska Region's Second-Growth Management Program (SGMP), five demonstration sites in southeast Alaska were commercially thinned in 1984-85. The purpose of the study was to evaluate the ability of commercial thinning to enhance wood production, the development of understory vegetation, and the availability and quality of Sitka black-tailed deer forage. When the SGMP sites were examined in 1998, researchers found that the strip and strip + individual tree selection treatments had the highest total understory biomass, but most of the biomass consisted of conifer regeneration (Zaborske et al. 2002). The individual tree selection treatment had less understory biomass per acre but over half the biomass was shrubs, ferns, and forbs which had greater nutritional value for deer. Estimates of deer-forage availability showed that this treatment created better forage resources for deer than did the other treatments and that summer forage availability was similar to the values estimated for old-growth forest (Zaborske et al. 2002).

In the early 1990s, five field trials were established in southeast Alaska to monitor the response of western hemlock and Sitka spruce to thinning and pruning (Petruncio 1994). Follow-up monitoring showed that developing clear wood in Sitka spruce was doubtful due to epicormic branching (Deal et al. 2003). Although pruning may not fully achieve wood-quality objectives, it may have added value for habitat objectives. Recent observations of Petruncio's field trials suggest that pruning was more effective than thinning alone in promoting understory diversity and abundance (unpublished data on file at the Juneau Forestry Sciences Laboratory). Understory response was not an objective of this study, so we can draw only limited inferences from it, but it is reasonable to conclude that the pruning increased understory vigor by admitting increasing sidelight from low sun angles.

Recent studies of mixed red alder-conifer stands have indicated that successional pathways alternate to the stem exclusion stage (i.e., loss of understory vegetation) are possible following clearcutting in southeast Alaska. Logging practices used after 1970 consisted of high-lead log yarding in which trees are carried through the air and soil is not disturbed. After this type of logging, a dense, uniform conifer stand lacking understory plants develops. Earlier methods of logging (prior to 1970) resulted in considerable soil disturbance which made excellent seed beds for red

alder to colonize (Ruth and Harris 1979). In contrast to the dense, uniform conifer stands created following high-lead logging, alder-conifer mixed young-growth stands have a species-rich and highly productive understory with a biomass similar to that found in old-growth stands. This species-rich understory has been found to persist for as long as 45 years after logging. Understory richness has been shown to be highest in stands with 18-51% alder and lowest in pure conifer or pure alder stands (Hanley 1996; Hanley and Hoel 1996; Deal 1997; Hanley and Barnard 1998).

SECOND STEP: DESIGN OF THE TONGASS-WIDE YOUNG-GROWTH STUDIES (TWYGS)

The interagency group of scientists and managers that met in 2001 designed three experiments to test the treatments suggested by their assessment. The primary research objectives set by the group are to assess the response of understory plants (herbs and shrubs), overstory trees, and slash loading to several silvicultural treatments, including altering stand composition by artificially regenerating red alder, pruning, and precommercial thinning. A fourth experiment was added later to compare thinning by conventional methods and by girdling. The fourth experiment was designed more informally than the first three, largely through discussions between PNW researchers and Tongass managers.

All the experiments share a common set of guidelines. First, the group decided that the treatments would lend themselves to practical application within the Tongass stand improvement program. Second, the treatments within an experiment should differ sufficiently to yield widely differing treatment effects—the designers frequently stated they were looking for “bushel basket” differences among treatments. Third, the treatments would be applied at an operational scale, generally ten acres or more per experimental unit. Fourth, the experiments would be widely replicated across the Tongass, twenty being the target number of replicates. This study is intended to last a minimum of 20 to 30 years in order to adequately assess long-term responses to silvicultural treatments and responses will be assessed at five-year intervals after treatment.

At present TWYGS includes four experiments:

1. A test of mixed hardwood/conifer stands, created by planting red alder at low (20 TPA) and high (80 TPA) densities in 0 to 5 year-old stands.
2. A test of moderate (222 TPA) and heavy (135 TPA) precommercial thinning of 15 to 25 year-old stands;
3. A test of moderate (170 TPA) precommercial thinning combined with two pruning treatments (25 or 50% of the trees were pruned), in 25 to 35 year-old stands;
4. A test comparing girdling and conventional thinning, with and without slash treatment, in stands over 35 years-old.

The 0-5 year age class was chosen for experiment 1 because conifer regeneration was assumed to be small enough to allow for the successful planting and survival of red alder seedlings. The 15-25 year age class was chosen for experiment 2 because this age is the normal precommercial thinning period. The 25-35 year age class was chosen for experiment 3 because this is the typical age for pruning—with or without additional precommercial thinning. Unthinned stands over 35 years-old were chosen for experiment 4 because they frequently occur in areas where commercial wood-production is not allowed, such as in beach-fringe stands. These stands have developed past the normal precommercial thinning stage and the large tree size and dense, interlocking crowns make it difficult

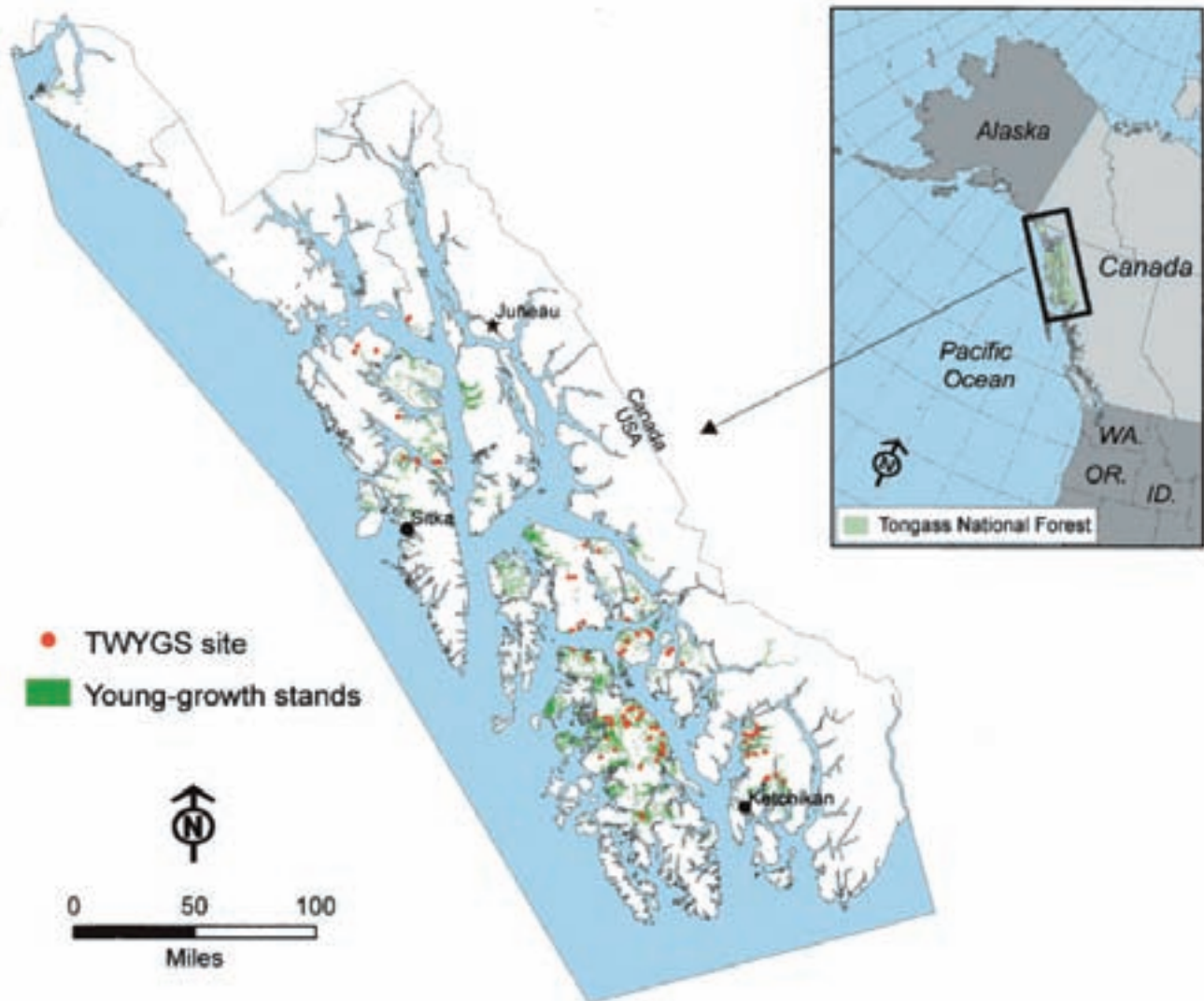


Figure 2— Map of southeast Alaska showing distribution of young-growth stands and TWYGS experimental sites.

to fell the cut trees and the thinning generates enormous slash loads. Girdling may avoid the operational and safety issues associated with cutting and may deliver slash to the forest floor over an extended period, thus reducing peak slash loads.

Each of the four existing experiments uses a randomized complete block design, with a target of 20 blocks (replicates) per experiment, distributed across the Tongass (figure 2). An untreated experimental unit is included in each block.

THIRD STEP: IMPLEMENTATION OF TWYGS

Tongass personnel screened and selected sites based on criteria developed by the design group. Candidate sites needed to be in the target age range, of moderate to high productivity, and at elevations below 1200 ft. Stands selected for treatment were to have relatively uniform productivity, stand density, and stand composition within the stand, but this was not always the case. Ranger district crews laid out the experimental units, typically dividing a single harvest unit into three to five experimental units. The scale of each experiment is impressive: because each experimental unit has a minimum of five to ten acres. The four experiments have from 17 to 23 replicates (blocks) and each of the four experiments covers 520 to 1773 acres in total. Once the units were laid out and the treatments were assigned randomly, Tongass personnel prepared and administered the treatment contracts necessary for the treatments to be applied by private silvicultural contractors. Treatments began in 2002 and all were completed by 2006. The silvicultural treatments were applied within the forest's normal program of work and it has been estimated that the adaptive management approach increased administrative costs by less than five percent¹.

¹ Russell, J.M. 2007. Personal communication. Silviculture Program Manager, Tongass N.F., Sitka, AK.

FUTURE STEPS: MONITOR, EVALUATE, AND ADJUST

The Tongass young-growth studies have passed through the first three of Nyberg's (1999) stages—assessment, design, and implementation—and are now entering the monitoring stage. PNW researchers have the primary responsibility for monitoring and evaluating treatment responses, including the design of measurement protocols, hiring and managing field data collection crews, managing data, and analyzing and reporting results. Most of the funding for these efforts, however, is provided by the Tongass. Just as the implementation phase took advantage of the experience and skills of NFS managers in working with private contractors to accomplish targets, the monitoring and analysis of responses relies on the traditional strengths of researchers to produce scientifically credible results. Once results are available and evaluated, the Tongass will have an opportunity to adjust its management at both the project and forest-plan levels. Based on our past experience, we expect that future adjustments will be made through a collaborative process involving NFS managers, PNW researchers, and interested parties from other state and federal agencies.

During the 2007 field season, we began monitoring with the measurement of responses in experiment 2, the precommercial thinning study. This is a critical point in this study, because effectiveness monitoring is often a low priority for land managers—there is an institutional bias to treat acres and meet targets, but when funds are limited, monitoring may be seen as expendable. By including researchers in the TWYGS program, the Tongass has gained a partner strongly motivated to monitor, evaluate, and report results, and we expect that researchers will be strong advocates for completing the adaptive management cycle.

The TWYGS treatments were designed to produce large differences in response between treatments—differences that should be readily perceived by an observer in the field—so it is possible that the success or failure of the treatments could be judged by a qualitative, anecdotal approach. A quantitative, statistically defensible assessment of the treatment responses, however, will better provide a sound, scientific basis for young-growth management in southeast Alaska, and will provide land managers with critically needed information on the relative costs and benefits of these treatments.

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Yellow-Cedar Decline: Conserving a Climate-Sensitive Tree Species as Alaska Warms

Paul E. Hennon, David V. D'Amore, Dustin T. Wittwer, and John P. Caouette

ABSTRACT

Yellow-cedar is a valuable, long-lived tree species that has been dying in concentrations on 500,000 acres of forest land for about 100 years in southeast Alaska. Recent research implicates climatic warming, specifically warmer springs and reduced insulating snow pack, which initiates premature dehardening and predisposes trees to spring freezing injury and death. Knowledge of the likely mechanism and spatial occurrence of the decline informs decisions about where on the landscape to favor active cedar conservation and management. Scientists and managers are devising a conservation strategy for yellow-cedar in the context of this decline problem. The strategy involves shifting more timber harvesting to the dead yellow-cedar forests, where

most wood properties are maintained even 80 years after tree death, and then favoring other tree species on those sites. The strategy also includes restoration and facilitated migration of yellow-cedar to cooler sites where decline is not predicted to occur as the climate warms. These cooler areas of favorable habitat are where spring snow is consistently present or in well-drained soils where deeper roots escape freezing injury. Because of yellow-cedar's low reproductive capacity, silvicultural practices such as site preparation, planting, and thinning are being used on favorable sites to maintain populations of this valuable tree species.

Keywords: *Chamaecyparis*, yellow-cedar, forest decline, snow, climate change, conservation.

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INTRODUCTION

Yellow-cedar (*Chamaecyparis nootkatensis* (D. Don) *Spach*)¹, is a commercially, ecologically, and culturally important tree species in Alaska and British Columbia. The species range extends from the California-Oregon border in forested montane areas to Prince William Sound in Alaska. It is limited to high elevation throughout most of its range, except in Alaska where yellow-cedar grows from near timberline down to sea level (Harris 1990). It is these lower elevation forests in the northern portions of its range where extensive mortality exists (Fig. 1).

Yellow-cedar is a defensive, slow-growing tree with few natural enemies and is capable of achieving great longevity (Jozsa 1992). The chemical deterrents to pathogens and insects in the foliage and heartwood are examples of this defensive nature. Reproduction capacity is low, leading to poor natural regeneration in some areas. The tree's resources are routed to chemical defenses rather than rapid growth or prolific reproduction. The extensive mortality problem in Alaska poses challenge of discovering some unique vulnerability of this tree species.

The landscape of southeast Alaska has complex geologic origins (Conner and O'Haire 1988) where accreted terrain and faults created many islands and deep fjords that bisect the mountainous mainland. The current climate of southeast Alaska is hyper-maritime, with abundant year-round precipitation, no prolonged dry periods, and high summer temperatures mediated by abundant rain and cloud cover. Winter temperatures average near freezing for the winter months at many weather stations, creating widely variable amounts of winter snow.

¹ The taxonomic status of yellow-cedar is in question because of the discovery of a tree species with close phylogenetic affinity in northern Vietnam, *Xanthocyparis vietnamensis* Farjon & Hiep. (Farjon et al. 2002). Yellow-cedar joins the Vietnamese tree in this newly erected genus as *Xanthocyparis nootkatensis* Farjon & Hiep. Whether that name, or the older name *Callitropsis nootkatensis* (D. Don) Orest. (Little et al. 2004), is adopted will be determined at the next International Botanical Congress in 2011 (Mill and Farjon 2006).

This near-freezing threshold winter temperature regime suggests that modest changes in climate could dramatically influence snow deposition and accumulation.

Without fire as a disturbance factor, the region supports the largest temperate rainforest in the world, which extends south through British Columbia. Cool temperatures, short growing seasons, and saturated soils slow decomposition of plant material, resulting in peat formation. Slope and soil properties, including peat accumulations, produce gradients of soil drainage that are largely responsible for driving forest productivity from large-stature, closed canopy forests on well drained soils to stunted, open canopy forests on saturated organic soils (Neiland 1971). Yellow-cedar has



Figure 1— Intensive yellow-cedar decline on Chichagof Island near sea level in southeast Alaska.

been competitive on these latter wet soils, typically reaching its greatest abundance here relative to other trees.

This paper represents a continuing effort to update and synthesize knowledge on yellow-cedar decline relevant to forest management by building from ongoing studies, published research, and previous summaries (Hennon and Shaw 1994, Hennon and Shaw 1997, Hennon et al. 2006). In this paper, we illustrate the probable mechanism leading to tree death, supply evidence at different scales supporting the rationale, and provide conservation suggestions to maintain the species in southeast Alaska.

YELLOW-CEDAR DECLINE

Yellow-cedar decline occurs at several thousand locations, that total approximately 200,000 hectares (1/2 million acres), in southeast Alaska (Wittwer et al. 2004) and a smaller amount in nearby British Columbia (Hennon et al. 2005). Yellow-cedar mortality far exceeds that of other tree species. In these forests, approximately 70 percent of yellow-cedar mature trees are dead, but some areas (e.g., Fig. 1) have even more intensive tree death (Hennon et al. 1990b, D'Amore and Hennon 2006). Most of the forest decline is on wet soils (Johnson and Wilcock 2002) where yellow-cedar was previously well adapted and competitive (Neiland 1971, Hennon et al. 1990b).

We examined trees in varying stages of dying by evaluating tissue death in their roots, bole, and crown to develop a general sequence of these symptoms (Hennon et al. 1990d). Initially, fine roots died, then small diameter roots died, followed by formation of necrotic lesions on coarse roots, and finally necrotic lesions spread from dead roots vertically from the root collar up the side of the bole. Crown symptoms occur after the early root symptoms. Crowns typically died as a unit with proximal foliage dying first, and then as trees finally died, distal foliage died. Note that this sequence of foliar symptoms differs from acute freezing injury to seedling and sapling foliage where newer, distal foliage is killed first. Generally, the study of symptoms suggested a below-ground problem as the cause of tree death. A number of types of organisms

were evaluated as potential pathogens, but each was ruled out by inoculation studies or by the lack of association with symptomatic tissue or dying areas of the forest: higher fungi (Hennon, 1990, Hennon et al. 1990d), Oomycetes (Hansen et al. 1988, Hamm et al. 1988), insects (Shaw et al. 1985), nematodes (Hennon et al. 1986), viruses and mycoplasmas (Hennon and McWilliams 1999), and bears (Hennon et al. 1990a). Thus, the mechanism leading to tree death appeared

to be underground, but not directly related to any biological agent.

INFLUENCE OF CLIMATE

Historical climate and cedar occurrence

An examination of the past climate of southeast Alaska and the historic abundance of yellow-cedar should offer clues about the climate preferences of the species, and could perhaps even reveal past episodes of decline. The last glacial maximum in southeast Alaska extended until between 16,000 and 12,000 years BP, before which southeast Alaska was thought to have been covered by ice (Hamilton 1994). Recent discovery of human remains and bones of large predators in caves on Prince of Wales Island

in Alaska (Dixon et al., 1998), as well the current distribution of several plants and animals, indicate the existence of sizable low elevation refugia in the southwestern portion of Alaska's panhandle (Fig. 2) (Carrarra et al. 2003) during that glacial maximum. Here, trees and other sub-alpine vegetation existed during the late Pleistocene and provided seed sources for subsequent re-colonization as glaciers receded.

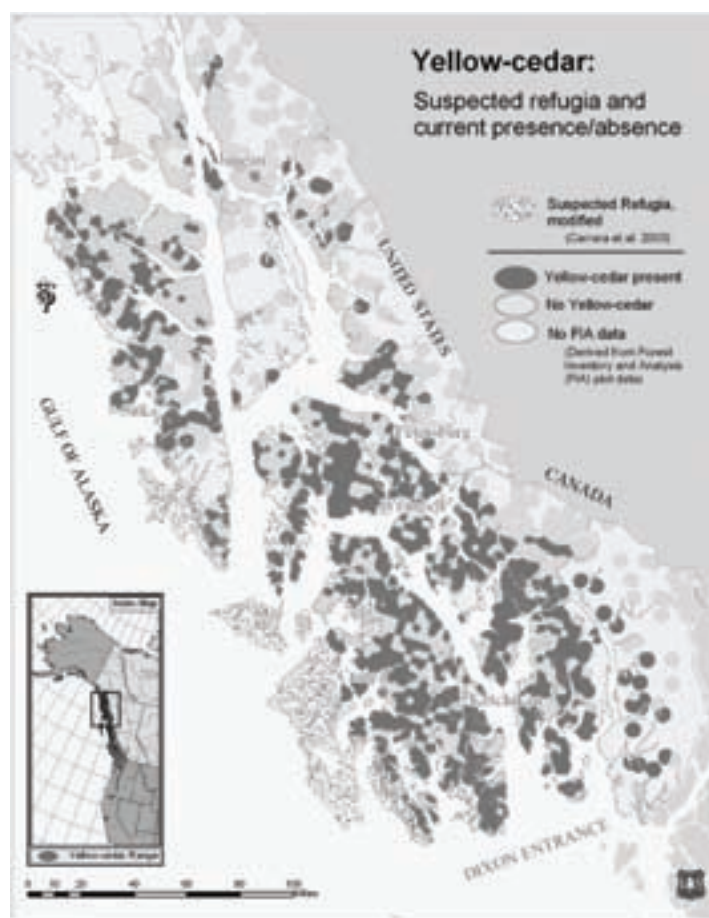


Figure 2— The occurrence of yellow-cedar (dark polygons) in southeast Alaska based on Forest Inventory and Analysis plot data. Areas where yellow-cedar was absent are depicted with lighter polygons; unsampled areas shown as very light grey. Also represented are areas of suspected refugia (stippled) (Carrarra et al. 2003), which may represent seed sources for post-glacial migration and colonization.

Climate during the Holocene Epoch can be inferred by examining the composition of trees and other plants using pollen profiles taken from lake and peat sediments, including 17 sites investigated by Heusser (1952, 1960). Unfortunately, yellow-cedar was not included in the early pollen profile studies because, as Heusser (1960, Page 78) stated, the pollen of *Chamaecyparis* and some other species had, “fragility and non resistance to decay....it was decided they be omitted [from analysis].” Recent investigations that included cedar pollen indicate that *Cupressaceae* became abundant about 7,000 years ago (Banner et al. 1983, Hebda and Mathewes 1984). In southeast Alaska, cedars may have become prevalent about 5,000 years ago (Tom Ager, USGS, Pers. Comm.). Our restricted understanding of the current distribution of yellow-cedar suggests that it originated from refugia in the southwest portions of Alaska’s panhandle (Fig. 2). Preliminary genetic analysis supports this contention (Ritland et al. 2001). Because of its limited reproductive capacity (Harris 1990, Pawuk 1993), the post-glacial spread of the tree has been very slow, but it is migrating to suitable habitat towards the northwest (Fig. 2) (Hennon et al. 2006) where colder winters appear to be more favorable.

The late Holocene (4500 years BP to 200 years BP) was moist and cool, which promoted rapid organic matter accumulation and provided favorable conditions for the expansion of yellow-cedar populations. A cooler shift within this period, known as the “Little Ice Age”, occurred approximately 500 years ago. Although the influence of the Little Ice Age on climate in southeast Alaska is not clearly understood, advances and retreats of glaciers are consistent with a change in climate (Viens 2001). The end of the Little Ice Age in the mid to late 1800s was associated with warming temperatures and marked the onset of yellow-cedar decline (about 1880 to 1900, discussed below). Information on the ages of canopy-level yellow-cedar trees (i.e., nearly all > 100 years old, (Hennon and Shaw 1994)), suggests that the trees that died throughout the 1900s, and those that continue to die today, regenerated and grew into their dominant positions during the Little Ice Age. We speculate that yellow-cedar colonized low elevation sites during this period, flourishing with deeper winter snow packs and late spring snow melt.

Onset and epidemiology of yellow-cedar decline

The earliest report of yellow-cedar decline was by the hunter Charles Sheldon (1912) who in 1909 noted, “vast

areas of rolling swamp, with yellow cedars, mostly dead.” Also, yellow-cedar decline can be observed on aerial photographs taken by the US Navy in the late 1920s (Sargent and Moffit 1929). A snag (standing dead tree) classification (Fig. 3) system was developed, with associated time-since-death estimates (Hennon et al. 1990c), and used to reconstruct coarse changes in cedar populations through the 1900s as expressed by annual mortality rates. The remarkable decay resistant heartwood of dead yellow-cedar trees (Kelsey et al. 2005) allows them to remain standing for 80 to 100 years after death, making this reconstruction possible. Results suggest that onset of yellow-cedar decline occurred in about 1880 to 1900 on most sites where trees are still dying (Hennon et al. 1990b). The higher proportion of class 3 snags (primary and secondary branches retained, but twigs missing--see Fig. 3) indicates yellow-cedar mortality accelerated to even higher rates in the later half of the 1900s (Fig. 3). Thus, mortality is progressive in declining forests, which now contain long-dead trees, more recently-killed trees, dying trees, and some survivors which are mainly other tree species (Hennon and Shaw 1997). The older mortality is typically on the wettest soils and recently-killed and dying trees are frequently found on better-drained soils and on the perimeters of the dying forests. This slow spreading pattern of tree death occurs along a hydrologic gradient (Hennon et al. 1990b, D’Amore and Hennon 2006). An annual mortality rate slower than 0.4 or 0.5 percent, which occurred in the first half of the 1900s, would be expected in a slow growing, long-lived tree species such as yellow-cedar. Such a sustainable mortality, more or less in balance with regeneration and growth to canopy status, has not been determined for mature yellow-cedar, but presumably would be very low (Parish and Antos 2006). Another tree species with similar very slow forest dynamics, *Sequoia sempervirens*, has annual mortality rates of approximately 0.1% (Barnett 2005) or 0.2% (Busing and Fujimori 2002).

A current study on the dendrochronology (i.e., tree ring research) of live yellow-cedar trees in southeast Alaska reveals that they were growing well during the Little Ice Age, but showed a synchronous reduction of radial growth rate in the later portion of the 1880s and into the 1900s (Beier 2007). More results on long-term cedar dendrochronology and correlations of cedar growth with weather station data will be available soon from Beier and his colleagues at the University of Alaska, Fairbanks. A challenge in this

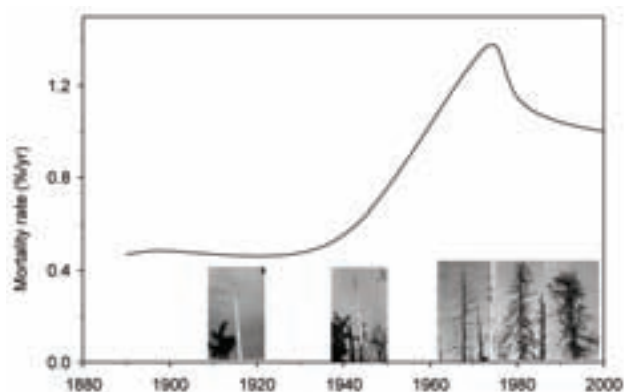


Figure 3—Estimated annual mortality rate of yellow-cedar in declining forests. This reconstruction combines time-since-death results of the five snag classes shown (Hennon et al. 1990c) with ground plot data (e.g., snag class frequencies) to create a splined-curve response for mortality rates through the 1900s.

research is to detect weather-induced episodes of tree injury, presumably before the growing season, in the context of weather patterns that influence annual radial growth during the growing season.

THE LEADING HYPOTHESIS FOR THE CAUSE OF YELLOW-CEDAR DECLINE

The culmination of research on yellow-cedar decline led to a working hypothesis to explain tree death (Fig. 4). This scenario is too complex to be evaluated by a single study; thus, it has become the framework for an ongoing research program. Each of these interactions is evaluated with one or more studies on hydrology, canopy cover, air and soil temperature, snow, yellow-cedar phenology, and freezing injury to seedlings and mature trees. These topics are discussed in more detail elsewhere (Schaberg et al. 2005, D'Amore and Hennon 2006, Hennon et al. 2006).

The association of yellow-cedar decline with wet soils now has a reasonable explanation. Yellow-cedar trees growing on poorly drained soils have shallow roots. Exposure on these wet sites is created from open canopy conditions that allow for solar radiation to warm soil and shallow roots. Canopy exposure also promotes rapid temperature fluctuation and more extreme cold temperatures. These factors appear to work together resulting in root freezing as the primary injury mechanism to explain the cause of yellow-cedar decline.

An evaluation of seasonal cold tolerance of foliage on mature yellow-cedars and co-existing western hemlocks in open- and closed-canopy forests at several elevations (Schaberg et al. 2005) revealed strong seasonal tendencies for both species. In fall, yellow-cedars in open canopy settings were more cold tolerant than in closed-canopy settings, whereas western hemlocks appeared unresponsive to canopy conditions. In winter, yellow-cedar had cold tolerance to about -40°C , more cold tolerant than hemlock, and tolerant below any recorded temperature for the region. Susceptibility of yellow-cedar to cold temperatures develops in late winter and spring. Yellow-cedar foliage dehardened almost 13°C more than western hemlock between winter and spring, so that yellow-cedar trees were more vulnerable to freezing injury in spring than western hemlock (Schaberg et al., 2005). Also, trees above 130m elevation were more cold hardy than those growing below 130m. These results indicated that if freezing injury is an important factor in yellow-cedar decline, then damage to trees most likely occurs in late winter or spring.

The susceptibility of yellow-cedar to spring freezing injury has been the subject of study in British Columbia, with a focus on seedlings and rooted cuttings (Hawkins et al. 1994, 2001; Davradou and Hawkins 1998; Puttonen and Arnott 1994). Severe freezing injury to yellow-cedar seedlings growing in Juneau has been observed in recent

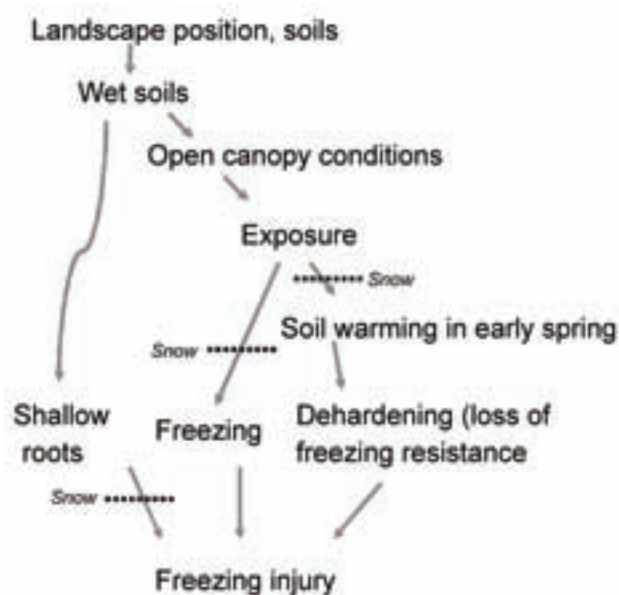


Figure 4—Conceptual diagram showing the cascading factors which form the leading hypothesis for the cause of yellow-cedar decline. The manner in which snow disrupts this process, thereby protecting yellow-cedar, is illustrated (dotted lines).

years, each time injury symptoms developed at the end of March or early April. The next step in this research was to study seedlings and evaluate late winter and early spring dehardening and cold tolerance of root and foliage tissue. Results (Schaberg et al., in press) demonstrate that initial injury is to roots, which were fully dehardened to a tolerance of about -5°C in February and March, earlier than expected. Foliar symptoms were delayed for about two months after root injury and only appeared when warm weather put transpiration demands on the seedlings. Seedlings whose roots were covered with perlite, used to mimic insulating snow cover, had complete protection and roots were not injured. All seedlings without this protection had severe root injury and died. Thus, this experiment on seedlings replicated the phenomenon of yellow-cedar decline, including root mortality leading to whole-plant mortality, as well as protection from snow.

SPATIAL EVALUATION OF YELLOW-CEDAR DECLINE

An evaluation of yellow-cedar decline at each of three spatial scales offers unique clues about the cause of yellow-cedar decline. Each scale shows close association of the absence of snow with decline, providing ideas for proactively managing the species. The three spatial scales include broad scale ($\sim 7 \times 10^6 \text{ km}^2$, regional--southeast Alaska), meso-scale ($\sim 800 \text{ km}^2$, medium-sized island), and fine scale ($\sim 1 \text{ km}^2$; small watershed).

Regional (broad) scale

A complete distribution map of yellow-cedar decline for southeast Alaska was developed. It depicts more than 2,500 locations totaling over 200,000 hectares of dead and dying yellow-cedar forests (Wittwer, 2004) (Fig 5). This map was derived from sketch mapping from small aircraft, an approach that yields inexact locations and polygon boundaries. However, it is instructive to examine broad areas where decline is present or absent and relate any pattern to regional variation in climate. A previous use of the map illustrated that the forest decline aligns with warmer average winter temperature isotherms (Hennon and Shaw 1994), an early suggestion that climate was involved in the problem. Here, distribution of yellow-cedar decline is contrasted with the first detailed model of snow accumulation zones in southeast Alaska (Fig. 5). The snow accumulation model, developed by Dave Albert of The Na-

ture Conservancy, is derived from PRISM data estimates of monthly temperature and precipitation (i.e., precipitation during months when mean temperature $< +2^{\circ}\text{C}$). There is a close association between the occurrence of yellow-cedar decline and the lowest snow accumulation zone (Fig. 5); the three other zones of higher snow accumulation could not be visibly depicted on this grey scale map but appear in color elsewhere (Hennon et al. 2006).

Our yellow-cedar decline distribution map documents the occurrence of mortality in Alaska, but not in adjacent British Columbia. Recently, intensive areas of yellow-cedar decline were detected about 150 km south into British Columbia where it frequently occurred in bands at approximately 300 to 400 m elevation (Hennon et al. 2005). The British Columbia Forest Service continues to map the southern extent of the mortality. Generally, yellow-cedar decline in Alaska and British Columbia reaches higher elevations with decreasing latitude.

Island (meso) scale

Higher resolution meso scale maps of Peril Strait (adjacent areas of Baranof and Chichagof Islands) and southern Kruzof Island delineate polygons of yellow-cedar decline on color infrared photographs. These maps are useful in associating yellow-cedar decline with landscape position features including slope, aspect, and elevation. Mapped polygons of decline are concentrated at lower elevations: greater amounts below 150m, lesser amounts between 150 and 300m, and very little above 300m. Yellow-cedar decline occurs on all aspects within these zones, but more decline was evident on warm (south and southwest) aspects. The Mount Edgecumbe study area on Kruzof Island near Sitka is a dormant volcano with radial symmetry and fairly even slope gradients. The open canopy forests with abundant yellow-cedar extend from sea level to close to timberline. These features help control confounding factors and allow us to detect the influence of elevation and aspect on the decline problem. The elevational limits of yellow-cedar decline and interaction of aspect (i.e., decline occurs higher on the warmer aspects) support the contention that the lack of spring snow is an important factor for yellow-cedar decline.

Watershed (fine) scale

Research at the small watershed scale is directed at understanding how forest conditions vary over local areas of a landscape. Vegetation plots on 100m grids at two small watersheds, Goose Cove on Baranof Island and Poison

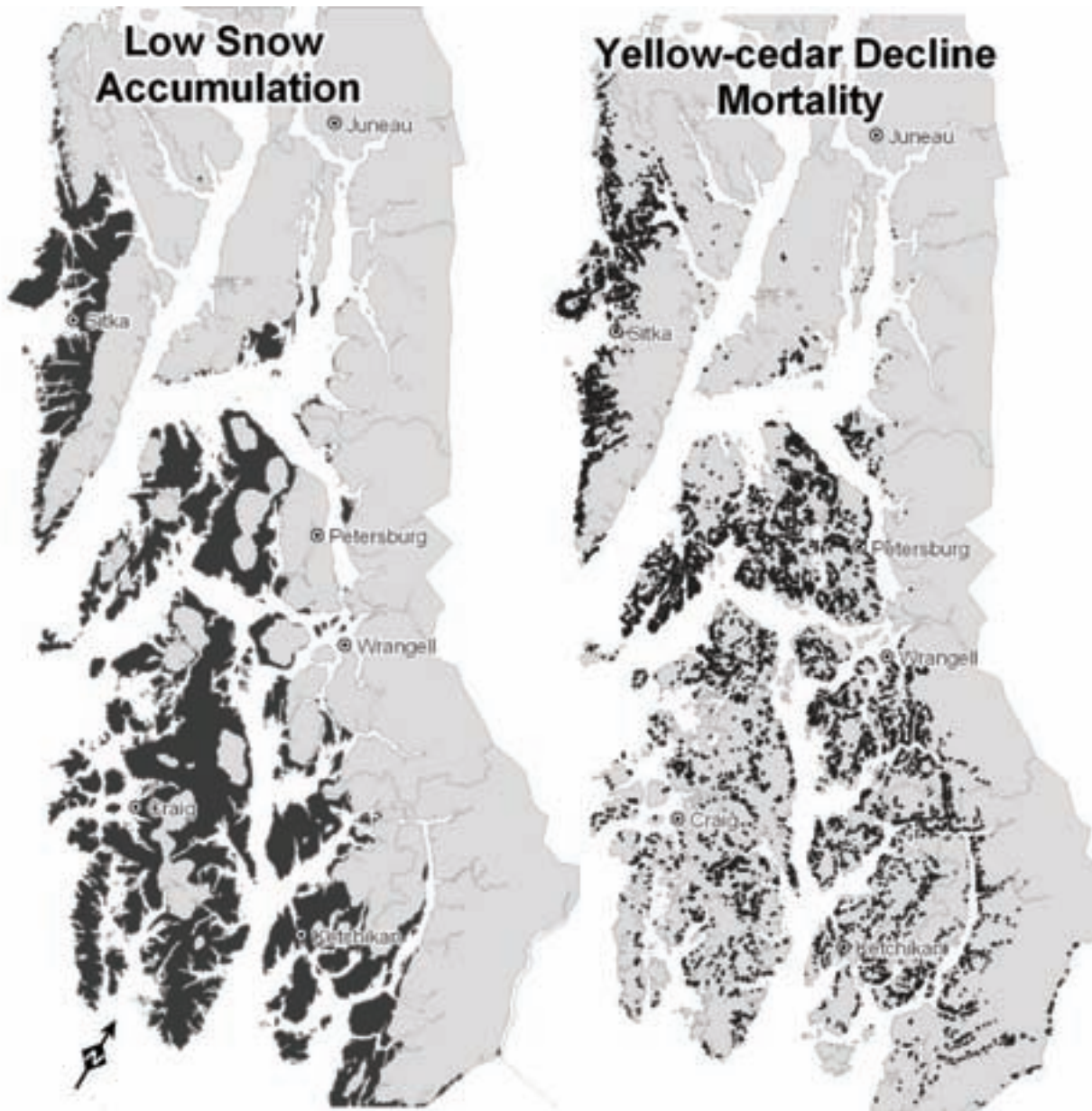


Figure 5— Association of yellow-cedar decline (right) with low snow accumulation (left). Yellow-cedar decline map was derived from aerial reconnaissance surveys. Map of lowest of four snow accumulation levels is from a regional snow model based on PRISM data estimates. The close association of yellow-cedar decline with low snow accumulations suggests that yellow-cedar could be favored in areas where late winter and spring snow is more abundant.

Cove on Chichagof Island, serve to measure live and dead trees and environmental variables, including hydrology, soil chemistry, canopy cover, air and soil temperature (D'Amore and Hennon 2006) and snow. Automated snow cameras were developed for daily snow measurements (Fig. 6). Digital cameras were housed in a plastic case with a Plexiglas window and contained a large battery pack and a circuit board with an intervalometer that directed the camera to turn on and record pictures daily. These snow

cameras were mounted to the sides of trees and pointed toward scenes to photograph graduated meter boards so that daily snow depths could be recorded. Soil temperature loggers were located in some of the scenes to associate the presence of snow with patterns of soil temperature.

Snow appears to protect yellow-cedar from this presumed freezing injury. Measurements of snow pack at the Poison Cove study site indicate that yellow-cedar growing around an open-canopy bog at 240m, a setting without the



Figure 6— Left, automated snow camera used to record daily snow depths. Right, healthy cedar forest surrounding a bog at 240m elevation with snow covering the ground in April. Snow typically occurs at this site until April or May, often several months after snow melt in the lower elevation dead yellow-cedar forests in the same watershed.

decline problem, has snow covering the ground through April and through May during some years (Fig. 6). Snow appears to offer protection for yellow-cedar by: 1) delaying the dehardening process; and/or 2) protecting fine shallow roots from freezing. The depth of snow required to buffer soil temperature may be as little as several centimeters. Thus, the presence of snow from February through March or April allows yellow-cedar to pass a period of potential vulnerability (during spring freezing episodes) that kills trees growing without snow.

CONSERVATION AND MANAGEMENT

Yellow-cedar is closely associated with snow zones, suggesting that snow plays an important role in protecting yellow-cedar. At our meso-scale analysis, the lack of spring snow may explain why yellow-cedar decline is limited to lower elevations and why it reaches higher elevations on warm aspects compared to cold aspects. At the broad scale, the distribution of yellow-cedar decline aligns closely with the lowest snow zone (Fig. 5). Some modification in the environment must have initiated yellow-cedar decline. It appears likely that reduced late winter and spring snow pack, which occurred as the region emerged from the Little Ice Age, represents that environmental change.

A strategy to manage yellow-cedar in the presence of climate-induced change is proposed (Fig. 7). One stage in this endeavor is to partition the landscape into areas that have yellow-cedar decline and areas that have healthy yellow-cedar forests (Fig. 7). Dead and dying forests have already been mapped (i.e., Fig. 5). These represent areas where yellow-cedar was once well adapted and is now maladapted due to climate change. In the dead zones, there is an opportunity of capturing economic value from the dead trees through salvage harvesting; this could help meet the timber demand for yellow-cedar. The various wood properties are preserved by the unique heartwood chemistry for decades, only diminishing slightly in the oldest snag classes some 50 and 80 years after tree death (Green et al. 2002, Hennon et al. 2000, Hennon et al. 2007, Kelsey et al. 2005). Evaluating the habitat potential of dead standing yellow-cedar trees for birds and small mammals is still a research need. Information on tissue deterioration through time, and the persistence of hard wood in snags (Green et al. 2002, Hennon et al. 2002), suggest that cavity excavating animals would not frequently use dead yellow-cedar. Insectivorous birds feeding on insects that colonize recently dead cedars would represent a more likely use. Knowledge on the successional trajectory in the declining yellow-cedar forests is also needed, to document the future composition of these forests. Other conifer species, already present as understory trees, appear to be favored where the yellow-cedar overstory has died. Observations suggest that the

successional trajectory will vary by soil drainage and overall vegetation productivity. Successional processes will occur whether or not declining forests are salvaged, especially if snags can be yarded selectively by helicopter.

To help compensate for losses due to yellow-cedar decline and commercial logging on other sites, an active yellow-cedar forest regeneration program could be expanded. Yellow-cedar does not regenerate as prolifically as other species in the region. The success of natural regeneration (e.g., seed tree harvests) should be evaluated. Yellow-cedar can be successfully regenerated by planting either seedlings (Hennon, 1992) or rooted cuttings (Russell, 1993), but the barriers to seedling performance (competing vegetation, deer browsing, and spring freezing) need to be considered. Favoring yellow-cedar during thinning operations will increase the yellow-cedar component in managed forests; however, planting may be necessary to establish a viable population to be manipulated. A schedule for timing thinning operations based on site productivity and the severity of competing vegetation is currently underway at several USFS ranger districts led by Chris Dowling and Sheila Spores. More knowledge on yellow-cedar silvics and experience with young-growth yellow-cedar management are needed in southeast Alaska.

Our present information suggests that yellow-cedar should be favored in:

- 1) northern and eastern regions of southeast Alaska that have cold winters,
- 2) higher elevations within the general distribution of yellow-cedar decline, and
- 3) better drained soils supporting greater forest productivity where roots penetrate more deeply and canopy shading cools soils during early spring.

Note that the first two of these three factors are highly related to late winter and spring snow pack.

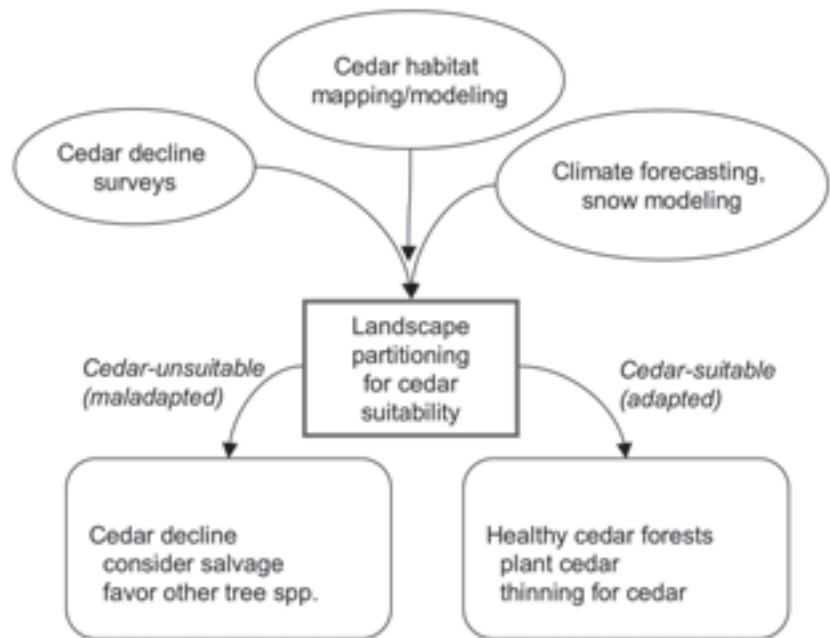


Figure 7— Management strategy for yellow-cedar and its decline problem involves (1) partitioning the landscape into areas that are favorable or unfavorable for yellow-cedar, (2) encouraging yellow-cedar in areas where it is currently healthy (i.e., typically with spring snow) or areas where yellow-cedar has not been competitive but can be planted and managed (i.e., well drained soils) and (3) encouraging other tree species where yellow-cedar is no longer well adapted (i.e., declining forests where dead trees could be salvaged).

It will not be sufficient to manage yellow-cedar where it is currently healthy because this approach would not account for climate warming. Managing this long-lived tree species requires predictive models that reveal where the decline problem is expected to occur in the next few centuries. Scenarios indicate that the climate may shift faster than vegetation is able to respond (Hamann and Wang 2006). Climatic models help focus management on areas where long-term persistence probability for yellow-cedar is high. Although we have a detailed map of dead cedar forests, our knowledge of the distribution of healthy yellow-cedar forests is surprisingly limited. Current USFS GIS layers, TIMTYPE and CLU (Common Land Unit), are based on interpretation of aerial photographs and are inadequate for managing Alaska's cedar species. Determining forest composition among plant communities that dominate wet soils is challenging because several tree species cannot reliably be distinguished on aerial photographs. We propose a different approach: use data from the many forest vegetation plots that have been collected over the last five decades in southeast Alaska.

A new project initiated with the Nature Conservancy uses a large number of permanent vegetation plots (approximately 50,000) to map and model the occurrence of healthy yellow-cedar in southeast Alaska. Assembling the various plot systems will reveal the distribution of western red cedar and yellow-cedar in southeast Alaska. The older plot systems are particularly valuable because they fill voids created by the lack of sampling in wilderness areas that were established in the 1980s. We intend to use all of these plots and “nearest ecological neighbor” methods (Ohmann and Gregory 2002) to model the distribution and habitat preferences of each cedar species. This approach should yield a high resolution map of cedar occurrence and also indicate specific habitat preferences for cedars.

Of course, yellow-cedar is no longer well adapted in all of these areas given the extensive forest decline. The yellow-cedar decline map as well as the low snow pack map (Fig. 5) can be used to determine where yellow-cedar is no longer suited and is maladapted. Another stage in partitioning the landscape for yellow-cedar suitability is to model expected snow accumulation zones into the future using several climate projections. This would appear to represent the best method for identifying areas that will be suitable for yellow-cedar over a long period of time. The final product will be a map that provides guidance to managers on areas that will and will not be suitable for favoring yellow-cedar over the next century or so.

SUMMARY

A plausible explanation for yellow-cedar decline must account for some particular vulnerability of this defensive tree species and some change in the environment. Yellow-cedar appears to be susceptible to premature dehardening and spring freezing injury, a vulnerability probably acquired by living at high elevations in heavy spring snow. Minor climate warming at the end of the Little Ice Age may have reduced late winter and spring snow pack at lower elevations, eliminating the protective snow that insulated yellow cedar roots during freezing events. This could be the environmental change that initiated yellow-cedar decline. If our explanation for yellow-cedar decline is correct, then this phenomenon represents an excellent example of how a shifting climate can cause dramatic change in a tree species and its associated ecosystem. The elusiveness of determining the cause of tree death and the complex-

ity of our hypothetical scenario illustrate the difficulty in predicting forest ecosystem effects of climate change. Perhaps, however, several effects of a warming climate are predictable, such as the phenology of plants no longer in tune with seasonal weather events. Also, as yellow-cedar decline demonstrates, some species may develop problems related to altered snow accumulation and melt in regions such as southeast Alaska with winter climate at the snow-rain threshold. A clear understanding of the mechanism of decline, future climate projections, and landscape modeling will be needed to solve the problem of where to favor this long-lived, valuable tree species in the future.

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The Role of Silvicultural Thinning in Eastern Forests Threatened by Hemlock Woolly Adelgid (*Adelges tsugae*)

Mary Ann Fajvan

ABSTRACT

In order to increase hemlock survivability in stands threatened by hemlock woolly adelgid (HWA), a new study is developing silvicultural thinning guidelines to reduce stand densities, reallocate resources, and increase hemlock vigor across a range of stand types and structures before HWA invasion. The 7 study areas are all geographically similar in that they regularly experience winter temperatures that can be lethal to hemlock woolly adelgid. The combined effects of climate and silvicultural treatment may serve

to moderate adelgid populations and increase hemlock survival. Four Pennsylvania stands received thinnings in 2006 to reduce relative stand density by 30-40%. Half of each stand was not harvested and serve as controls. In New England, four additional sites will be thinned in 2007-08. Pre- and post-harvest sampling includes measurements of stand density and structure, residual “crop” tree stems and crowns; understory vegetation; soil moisture, temperature and nutrients; foliar nutrients; and insect populations.

Keywords: hemlock woolly adelgid, silvicultural thinning, stocking, foliar nutrients

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INTRODUCTION

The hemlock woolly adelgid, (HWA, *Adelges tsugae* Anand) is a major threat to the health and sustainability of Eastern North American hemlock species – eastern hemlock (*Tsuga canadensis* (L.) Carr.) and Carolina hemlock (*Tsuga caroliniana* Engelm.) (Knauer et al. 2002). This introduced insect is currently found in 17 states, covers approximately 50% of eastern hemlock's range (http://www.na.fs.fed.us/fhp/hwa/maps/hwa_2006.pdf), and is causing tree decline and wide-ranging mortality effects. Hemlock mortality associated with HWA establishment can range from almost none to 95% (Bair 2002, Mayer et al. 2002, Orwig and Foster 1998). Mortality can occur quickly and uniformly throughout a stand, especially in the presence of other stressors, or can occur slowly and in patches for more than a decade. The resulting effects on stand dynamics may cause changes in species dominance, age and canopy structures, (Orwig and Foster 1998, Stadler et al. 2006) and hence management direction.

HWA is rapidly spreading to new areas at a rate of 15.6 km/year south of Pennsylvania and 8.13 km/year (or less) in the northern portion of hemlock's range (Evans and Gregoire 2007). Colder winter temperatures are believed to slow spread rates (Evans and Gregoire 2007, Parker et al. 1999). HWA attacks all sizes and ages of hemlock from seedlings to mature canopy trees. Adelgid feeding depletes hemlock's stored nutrient reserves causing a reduction in new shoot production in infested parts of the crown, followed by needle drop, branch tip dieback, foliage thinning, lower branch dieback and mortality in 2 to 15+ years (McClure 2001, Mayer et al. 2002, Orwig 2002). Biological and chemical methods to control the pest are costly and impractical for large forest tracts.

The wide range in the rate and extent of eastern hemlock decline and mortality suggest that tree and site variables influence the response of hemlock to HWA infestation. A new study is being implemented to determine if stand structure manipulations can be used ahead of the leading edge of HWA invasion to increase individual host tree and stand resistance. Specifically, silvicultural thinning treatments are being used to remove low vigor trees, which would reallocate site resources to enhance the health of the remaining hemlocks. However, nothing is known about whether such treatments are effective in HWA control. Because thinning

also alters forest climate, and soil and foliar nutrient cycles (Van Cleve and Zasada 1979), the attractiveness of thinned stands to HWA feeding must also be evaluated. Therefore, research is needed to determine whether thinning mixed hemlock-hardwood stands to increase the vigor of hemlock trees ahead of the leading edge of HWA will also increase hemlock survivability when HWA attacks a stand. The project involves the collaborative efforts of many individuals from federal and state agencies located in the mid-Atlantic and New England states. Working across organizations and ownerships will develop linkages necessary to translate the research findings quickly and directly to natural resource managers for application.

The study has two objectives. The first is to determine whether thinning mixed hemlock-hardwood stands to increase the vigor of hemlock trees prior to the arrival of HWA will increase the survivability of hemlock when HWA attacks a stand. Scientific and empirical evidence indicates that while healthy hemlocks are not less susceptible to attack, they do have a better chance of surviving longer than stressed trees (e.g., crowded conditions). Increased survivorship may improve hemlock's chances for recovery (Orwig and Foster 1998), especially if HWA populations crash and/or effective biological controls are developed. To monitor the effects of thinning, it is critical in this study that the approximate time of arrival and subsequent spread of HWA in these stands be carefully monitored. The results from the study will provide management options to forest landowners and land managers faced with pending HWA invasion and potentially prevent pre-emptive harvesting of hemlock, which can contribute to future regeneration failures of this species (Brooks 2004).

A second objective is to monitor ecological changes in forest structure as an HWA infestation progresses in thinned and unthinned stands. Treatments are established in a variety of stand structures located on a range of site conditions in the northern portion of hemlock's range. A number of tree health variables (e.g. mortality, foliage density, foliar nutrients, crown transparency, percent dieback, direct HWA counts, and amount of new crown and stem growth) are used to measure the effects of thinning, and subsequent HWA infestation, on hemlock growth and survival (Colbert et al. 2002, Kimple and Schuster 2002, Scudder et al. 2002). Because both thinning and insect infestation can cause increases in soil N mineralization

and nitrification rates, in some of the study areas, soil and foliar sampling are also included.

BACKGROUND

Use of Silviculture in Invasive Species Management

Loss of a tree species due to an introduced pest can initiate ecological change and have potential economic effects (Liebhold et al. 1995). The appropriate silvicultural treatment for managing introduced forest pests requires an understanding of the ecological interactions of both the pest and the host species. Integration of silvicultural techniques at a variety of spatial and temporal scales may be necessary to prevent or mitigate the effects of a pest invasion. Because introduced pests have no natural enemies in native forests, traditional silvicultural approaches designed for managing native pests, such as sanitation or salvage operations (Smith et al. 1997) may need to be modified (Waring and O'Hara 2005).

Hemlock's ecological role in eastern forests far outweighs its economic importance. Hemlock's wood characteristics limit its commercial use to relatively low-grade products (Howard et al. 2000) for which market substitutions are easily made. Conversely, hemlocks occupy an extraordinarily important ecological niche in both pure stands and growing in mixture with hardwoods. The species are very tolerant of shade and typically are found in understory positions as well as in sub-dominant overstory canopy positions. Dense hemlock understories utilize growing space that would be lost to less shade-tolerant species (Kelty 1989), provide winter cover and vertical structure for many wildlife species (DeGraaf et al. 1992), and maintain aquatic habitat integrity by shading streams (Evans 2002). Hemlocks grow slowly even when young and at high densities can survive in suppressed conditions for hundreds of years (Godman and Lancaster 1990). Self-thinning is slow and canopy disturbance is typically required for accelerated growth to occur.

The extent to which the presence of other insects, site conditions, and stand structure affect hemlock's susceptibility to HWA or vice-versa is still unclear. Hemlocks growing in areas with adequate, consistent moisture are generally less vulnerable to attack and can survive an HWA infestation better than trees growing on droughty or waterlogged

sites (Mayer et al. 2002, Orwig 2002, Sivaramakrishnan and Berlyn 2000, Ward et al. 2004). HWA impacts to stem and branch growth typically progress upward from the lower crown, and reduced growth and possible mortality occur depending on infestation intensity and duration. Hemlocks in dominant and codominant overstory crown class positions had lower mortality during moderate infestations than overtopped and intermediate class trees (Orwig and Foster 1998). However, new evidence suggests that live-crown ratio may have a stronger relationship with vulnerability regardless of crown class (Rentch 2007).

HWA invasions typically follow three phases: arrival, establishment and spread (National Research Council 2002). Combined with local quarantines, preventive silvicultural techniques can be used along the leading edge of a HWA outbreak prior to arrival. Seasonal alterations in management activities (e.g. road closures, harvesting) near HWA infested areas can also be preventive (Waring and O'Hara 2005). Curbing management activities during HWA crawler emergence in both early spring and mid-summer may help curtail spread.

Reduction in stand density through thinning and removal of the most vulnerable individuals of targeted host species can be used alone or in combination with eradication tools, to reduce potential infestation targets and minimize mortality. For example, the goal of thinnings used to lower the density of gypsy moth-preferred host species is to select crop trees with the most vigorous crowns because healthier trees can survive years of defoliation (Gottschalk 1993). The best results from this type of density manipulation occur when the treatments are applied before arrival of the pest.

Specific chemical characteristics of the foliage may affect a tree's resistance to HWA or may act as a physiological defense strategy (Clancy et al. 2004). Healthy hemlock forests typically have slow decomposition and N-cycling rates because of their low foliar N content and cool microclimate. Trees infested with HWA were found to have significantly higher needle concentrations of N, Ca, K, Mn, Al, lignin and cellulose compared to those not infested (Pontius et al. 2002); although, total soil nutrient content did not change in infested stands. HWA feeding and needle mortality effects nutrient cycling in the canopy, the chemical composition of the litter, and hence nutrient availability in biogeochemical cycles (Stadler et al. 2006). In HWA-infested

forests, nitrogen mineralization and nitrification rates are higher than uninfested forests (Jenkins et al., 1999, Yorks et al. 2000). Yet, there is a decline in nitrogen fluxes over time as infestation increases and needle biomass decreases (Stadler et al. 2006).

Thinning also effects biogeochemical cycles because of increases in light, soil temperature, soil moisture, and litter and root system decomposition, at least until canopy closure (Van Cleve and Zasada 1976). Thinning hemlock stands will probably cause temporary increases in soil nitrification rates, which may or may not cause increases in foliar N and other elements. Depending on individual site conditions for a given stand, microbial soil populations can increase enough to tie up nitrogen so that foliar amounts remain unchanged (Van Cleve and Zasada 1976).

After arrival, the next invasion phase occurs when the pest begins the process of establishing a permanent population. Early in the establishment phase when pest populations are typically low, biological and chemical eradication tools can be successful for removal of all or most individuals, especially if treatments are repeated. There are several insecticides used for controlling HWA (McClure 1991b) but these are only effective as individual tree applications, restricting their use in treating entire stands.

Once an established population begins to rapidly increase, silvicultural efforts should focus on containment of the pest. HWA has been spreading through the Eastern United States for over 50 years (Cheah et al. 2004). Spread rates were initially slow but have increased substantially in the last decade (Ward et al. 2004). In the spread phase, silvicultural efforts target reducing host damage/mortality and slowing the spread.

Detection and assessment of HWA

Understanding HWA's complex reproduction cycle is important for monitoring population spread and for planning silvicultural activities. HWA produces abundant progeny in two parthenogenetic (all-female) wingless generations per year. The winter generation develops over 9-10 months (early summer to mid-spring of the following year) and the spring generation occurs from spring to early summer of the same year. These generations overlap in middle to late spring and life cycle timing is influenced by local climate. The winter generation lays up to 300 eggs per female in

late winter/very early spring (McClure et al. 2001). The eggs hatch in April and May and mobile brownish-red nymphs ("crawlers") crawl along the branches but can only be dispersed between trees by wind, animals and humans. Crawlers settle on twigs and insert their stylets into the base of hemlock needles and obtain nutrients from xylem ray tissue (Young et al. 1995). They remain at the same feeding site for 4 nymphal stages and develop into adults in early June. These spring generation adults lay eggs between mid-June to July, but the number of eggs produced is only 20-75 per adult (McClure 1989). Eggs hatch in early July, enter a summer dormancy period and begin feeding around October. The prolonged winter feeding of this generation allows it to tap into a tree's stored food reserves in the absence of any potential predators.

Several studies examined the spatial distribution and dispersion patterns of HWA throughout the forest (McClure 1990, Gray et al. 1998, Kimple and Schuster 2002). Other studies on egg, crawler and adult distributions (McClure 1990, 1991a, Evans 2004) suggest that HWA is evenly distributed throughout the vertical layers of forest and tree canopies. More recently, however, entire stem and branch dissections reveal that in trees with high HWA populations a larger percentage of nymphs are found in the lower crown while in trees with low HWA populations a larger percentage of nymphs are found in the upper crown (Evans and Gregoire 2007). These findings suggest that early detection of HWA invasion would be more difficult because the population would at first be more numerous in the inaccessible upper crown. Costa (2005) developed a sampling plan for the detection of HWA but it is based on sampling only lower branches within reach. A modified version of his sampling system, as well as visual observation with binoculars will serve as the foundation for the detection surveys in this study. Impact survey assessments similar to those used by Colbert et al. (2002) will be used to measure the effects of HWA on hemlock tree health variables.

Management Guidelines for Hemlock-Hardwood Stands Threatened by HWA

Current management guidelines target stands with an identified HWA infestation and anticipated hemlock decline and mortality within several years. Management options include harvesting stands before notable decline, salvage harvesting, felling trees without removal, protecting individual trees using biological controls or insecticides, and

taking no action (Ward et. al 2004). Stands are prioritized for type and intensity of management activities based on economic, ecological, esthetic criteria and site characteristics. Regardless of management objective, stands are selected for treatment with the goal of maintaining their health and vigor until effective biological controls are developed.

There are few thinning studies in hemlock-hardwood stands that focus on reducing stand density to favor hemlock growth. Hemlock height and diameter growth after crown release is related to how dense the preharvest stand was and a tree's canopy status (overstory vs. understory) (Merrill and Hawley 1924, Marshall 1927). Trees growing in dense stands showed 60% greater growth after thinning (from above and below) than stands that were understocked. Stands with higher hemlock stocking exclude more rapidly growing shade intolerant competitors and growth is concentrated on the main stem instead of branches (Tubbs 1977). Hemlocks previously growing in suppressed canopy positions tend to have better stem form and faster growth after release than open-grown associates of similar size. Age appears to be a minor factor in tree growth response; older trees typically grow better than younger ones unless a tree has been suppressed for so many decades (>60 years) that it has lost too much crown area to increase growth (Marshall 1927). Generally, hemlock can be thought of as a species that shows a growth response to environmental change based on its size (e.g. smaller trees grow faster than larger trees) and not its age. Evidence from thinned hemlock stands indicates that hemlock can continue favorable volume increases through at least age 200 (Ward and Smith 2000). Current management guidelines recommend that thinning operations should remove at least 6 to 7 m²/ha of basal area; however, if stands are very dense (> 46 m²/ha), basal area removal should not exceed more than 1/3 of the total in any given operation (Lancaster 1985).

In this study, thinnings are designed to reduce stand densities in fully-stocked and overstocked hemlock-hardwood stands and hemlock-hardwood-white pine (*Pinus strobus* L.) stands. When a stand is at full-stocking it contains the maximum sustainable amount of foliage given fixed site resources. Fixed site resources are those resources that cannot be manipulated silviculturally (Vose and Allen 1988). Thinnings reallocate fixed resources among fewer stems increasing the amount of light, water and nutrients per tree. Foliar quantity is affected because of modifica-

tions to the branch size – foliage weight relationship and the number and (or) size of branches within the crown (Gillespie et al. 1994). Healthier hemlocks can tolerate high densities of HWA better than can trees of low vigor (McClure 1995, Sivaramakrishnan and Berlyn 2000) and hence may survive or persist longer during an infestation. However, because thinning affects foliar nutrient content, measuring the changes in foliar nutrients in thinned and unthinned stands prior to and during HWA infestation, may indicate whether silvicultural manipulations make hemlocks more or less attractive to HWA attack.

In this study, the silvicultural approach to HWA management is modeled after the gypsy moth management guidelines developed by Gottschalk (1993) from research conducted since the early 1980s. Currently, all standard silvicultural thinning guidelines for hemlock-hardwood stands are based on data from stands in New England and the Lake States (Lancaster 1985, Tubbs 1977), and none of them addresses mitigating the impacts of HWA. Because the study sites are located in areas either on the leading edge of the invasion or up to 10 years away from invasion, pre-arrival stand conditions can be measured, silvicultural treatments applied, and ecosystem responses after arrival and establishment can be monitored.

METHODS

Site Selection

Study sites, on federal and nonfederal land, are being used in the study. Selection criteria include stand size, soil/site limitations to harvest, and stand structure (density, species composition, hemlock component). Stands need to be at least 24 ha and have a minimum of 7 m²/ha of hemlock basal area distributed uniformly throughout the stand. As of May 2007, four stands received treatments on the Allegheny National Forest in northwestern Pennsylvania. An additional Pennsylvania stand, managed by the Pennsylvania Game Commission, is currently being marked for thinning. All those areas were initially overstocked with total basal areas averaging around 46 m²/ha. Three study areas in New England will be thinned in summer/fall 2007. Two stands are on Massachusetts state forest land and the other stand is located at the U.S. Department of the Air Force, New Boston Air Force Station in New Hampshire.

Treatment

Prior to timber marking, six, 4-ha treatment blocks are established in each stand. Three of the treatment blocks are then randomly selected to be marked for a thinning to reduce relative stand density 30–40 percent. Because stand structures are typically spatially variable throughout the treatment areas, a combination of low and crown thinning (Smith et al. 1997) is sometimes used to remove hardwood trees overtopping hemlocks with good stem form and live crown ratios > 30%, and to reduce density in hemlock clumps. Hemlocks with <30% live crowns are less likely to respond to the treatment (Smith et al. 1997). Otherwise the treatment is primarily a crown thinning to reduce density among competing hemlocks. Both commercial and noncommercial harvests are sometimes necessary. The other three treatment blocks in each area are not thinned and serve as reference stands for sampling vegetation dynamics pre- and post-HWA invasion.

Trees are marked for thinning according to guidelines developed specific to each stand's characteristics based on recent inventory data summaries. After marking, ten, 0.04 ha circular vegetation sample plots are located in each treatment block according to a systematic random procedure wherein each block is roughly divided into ten, 0.4 ha sections and one sample plot randomly located within each section. In the blocks designated for thinning, a residual hemlock tree that was targeted for canopy release (e.g. has 3–4 surrounding stems marked for removal), serves as the plot center. This center tree is referred to as the "subject" tree. Similar selection criteria are used to identify 10 hemlocks to serve as subject trees in the untreated blocks. In addition to the subject trees, each spring 120 randomly selected hemlock "HWA monitoring" trees at each site (20 trees/block) will be examined for presence of HWA according to a procedure described by Costa (2005) as modified by Turcotte (described in Fajvan et al. 2005).

Vegetation Sampling

Plot-level vegetation measurements of overstory and understory are conducted prior to thinning and repeated the first growing season after thinning (Fajvan et al. 2005). Plots are revisited at 1, 3, 5, 7 and 9 years post harvest to re-measure the vegetation. Future sampling will be determined by timing of HWA invasion.

Measurements collected on the subject trees include: stem diameter, total height and height to base of live crown, and crown width. Crown health variables (foliage density, transparency, percent dieback) and branch tip assessments of new growth, are measured according to criteria outlined in Colbert et al (2002). On the Pennsylvania sites, soil and foliar sampling for nutrient elemental content were conducted prior to harvest and continue annually for 5 years starting the first growing season after harvest (Piatek 2006).

DISCUSSION

Because the timing, extent, and duration of HWA invasion will differ for individual study areas, each 60 + acre stand is considered an experimental unit, with 3 replicated treatment blocks each of thinned and control. Valuable information about the effects of thinning on hemlock growth, vigor, and vulnerability to HWA will be gained by monitoring the subject trees. The value of having pre-treatment and pre-HWA vegetation sampling plots will also serve to monitor changes in vegetation structure cause by HWA-induced mortality. In addition, in stands with foliar and soil elemental chemistry monitoring, data will be collected on the effects of thinning on biogeochemical cycles in hemlock-hardwood mixtures. The relationship of foliar nutrients and hemlock's attractiveness and palatability to HWA can be determined as well as the interaction of thinning and HWA infestation on elemental chemistry.

This new study faces many challenges in its implementation and subsequent monitoring. Study areas are geographically distant requiring the coordination of different field personnel to sample vegetation and monitor HWA at each location. HWA is difficult to detect during early stand invasion because initial infestations are typically in upper crown locations, so sites need to be carefully monitored.

At the northern range of HWA spread, cold winters are believed to cause slower spread rates and help explain the longer survival of infested hemlocks (Paradis et al. 2007). The study areas are all located in this northern spread range and it is hoped that the combination of silvicultural thinning to improve hemlock vigor, combined with colder temperatures, will increase survival rates. Entomologists believe that a complex of introduced predatory insects and diseases will be needed to maintain HWA below lethal levels (Ward 2004). Forest managers in the hemlock region need

to test various treatments in order to maintain a hemlock component in stands while biological controls are tested and released in the field. Silvicultural thinning is one technique that can be tested in an attempt to avoid the loss of an ecologically important tree species found across most of the eastern United States.

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Hayes Creek Fuel Reduction Project: A Success Story

Kim M. Johnson

ABSTRACT

The Hayes Creek fuel reduction project was small in terms of the number of acres treated but very complex. This complexity included figuring out how to go about meeting a variety of multiple resource objectives given the values associated with a unique area. Doing fuel reduction treatments would reduce the risk of losing the vegetation on site in the

event of a wildfire however there was still a need to provide habitat for big game animals and goshawks, rejuvenate the aspen stands, and protect historic apple trees. This paper describes our project from planning through implementation and the lessons we learned along the way.

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INTRODUCTION

There are many challenges facing land managers when applying fuel reduction treatments. These challenges range from public support of fuel reduction treatments to our own internal agency planning and implementation challenges. Can proposed treatments meet multiple objectives? Are there issues with minimal to no resolution options that would make the project a “no go”? Will private land owners be supportive to treatments done on national forest land adjacent to their property? Will there be resistance from groups that traditionally litigate projects? Does the agency have the right tools in the tool bag to get the work done on the ground? Will the project pay for itself? Will the prospective contractors be receptive to a non-traditional contract? Then there are always the unknown questions and issues that arise when you least expect them. These are just some of the challenges land managers face on a day to day basis. The goal of this paper is to tell the tale of a complex fuel reduction project located within the wildland urban interface (WUI) on the Bitterroot National Forest.

PROJECT OBJECTIVES

The main objective of the Hayes Creek fuels reduction project was to “...reduce the potential for uncharacteristically intense, large-scale wildfire in the wildland urban interface by reducing the fuel loadings, creating fuel breaks, and diversifying stand structure” (USDA Forest Service, 2003). The treatments were intended to open up the overstory tree canopy through thinning and remove most of the ladder fuels from the understory. Prescribed fire would follow thinning.

While vegetation management to reduce the fire risk and change fire behavior was the primary objective, there were also other vegetation benefits that would result from proposed treatments. Thinning would reduce stocking levels and promote ponderosa pine leaving the residual vegetation in a healthier condition and lessen the risk of losing this vegetation to insects and diseases. Aspen vegetation needed rejuvenation and thinning and prescribed fire would act as the disturbance process for rejuvenation. Historic apple trees were a major component of the understory and while no formal protection was required, we wanted to retain as many of these trees on site from the historic and scenic perspective and because it was the right thing to do.

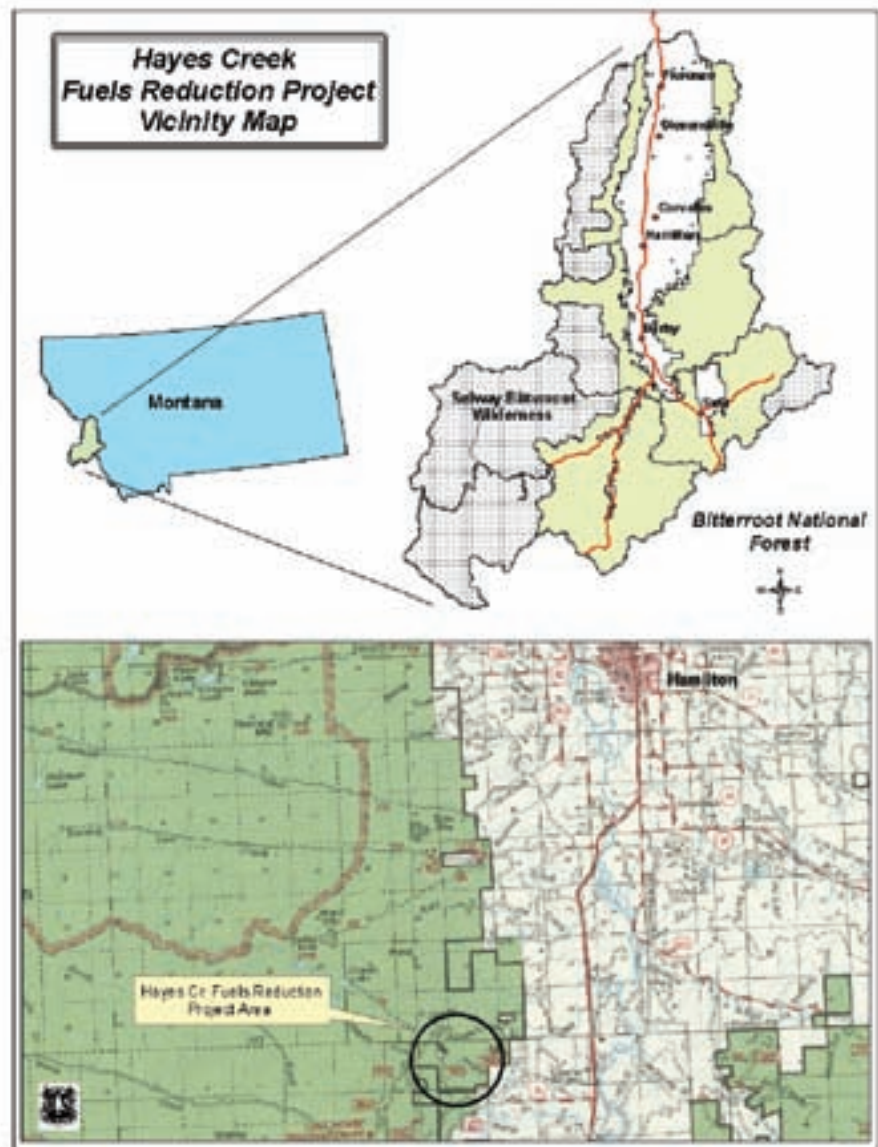


Figure 1—Hayes Creek fuel reduction project, vicinity map.

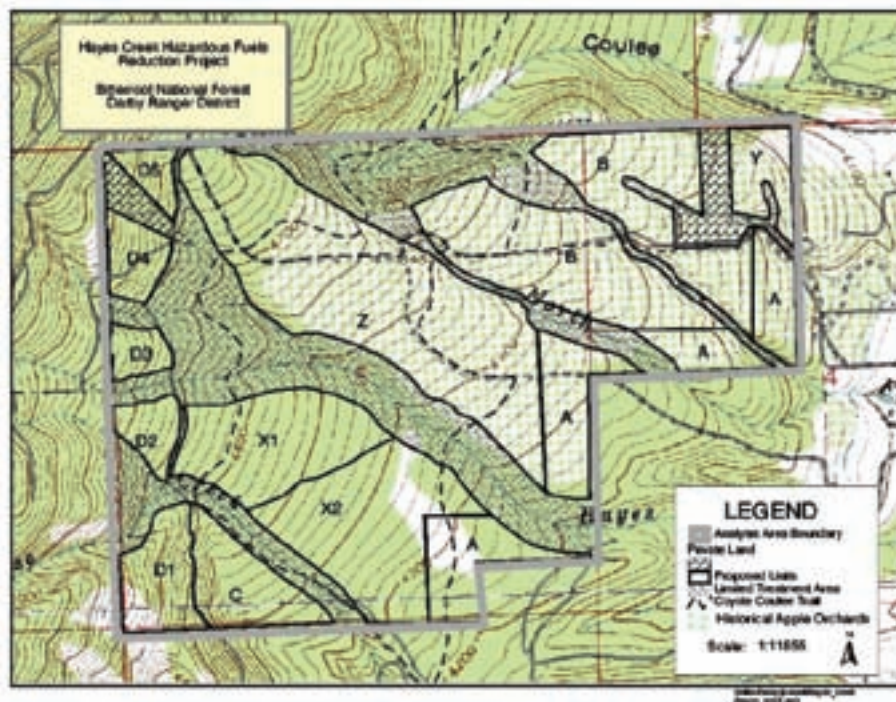


Figure 2—Hayes Creek fuel reduction project, treatment unit map.

Wildlife habitat would benefit from treatments by opening up the canopy and increasing the browse component of the understory. This worked towards meeting Forest Plan standards and guidelines of optimizing big game winter range habitat. Goshawk habitat will also be enhanced by proposed treatments. Treatments will set up the trajectory for future flammulated owl habitat.

The project was developed with coordination between wildlife, fisheries, timber sale, hydrology, soil, silviculture and heritage program resource specialists. The integrated approach ensured that the project design and prescriptions for fuel reduction treatment incorporated the appropriate forest plan standards and guidelines and met project objectives. Mitigations were developed to address other issues and concerns.

BACKGROUND

Project Area Setting

The Hayes Creek Hazardous Fuel Reduction project area is located adjacent to private land in the vicinity of Moose and Hayes Creeks and Coyote Coulee on the Darby Ranger District, Bitterroot National Forest (refer to figure 1—vicinity map). Terrain is relatively flat and rolling with

drainages running west to east. Elevation ranges between 4200 to 4800 feet. The project area is approximately 680 acres in size with treatment proposed on about 440 acres (refer to figure 2—treatment unit map). The letters on the project map designate individual treatment areas.

Only one drivable road exists in the project area. This road runs within the western portion of the area (refer to figure 2), at higher elevations along the east edge of treatment units D-1 through D-5. Figure 2 shows roads through the middle of the project in treatment unit Z and B however most of these roads are actually designated as ATV trails and not drivable with cars or trucks.

The forest plan management area designation for a majority of the area is big game winter range. Standards and guidelines specify to optimize elk winter range habitat using timber and other vegetation management practices. The higher elevations of the project area are designated as visually sensitive foreground and middle ground. Standards and guidelines for this visually sensitive area are to maintain partial retention visual quality objectives and manage timber (USDA Forest Service, 1987).

PROJECT AREA HISTORY

Human Use

The project area land was acquired by the Forest in 1979. Since acquisition, minimal treatments on the land have occurred.

Historic Logging Practices

In the mid 1800s subsistence logging mainly occurred by individual land owners as the land was cleared for farming and building cabins. Subsistence logging transitioned into full blown extensive timber harvesting as settlement of the valley increased. This large scale logging effort occurred in the late 1800s to early 1900s in the Hayes Creek area

by the Anaconda Copper Mining Company. The company bought up land from individual land owners and basically logged the land clean. The timber was used for railroad ties and to fuel the company copper mill in Butte, Montana. All trees over six inches in diameter were removed and the logging slash left on the ground. Logging of the time was done by horses. Logs were dragged to creeks and floated down stream during high water or dragged to the railroad track for haul to the mill.

Apple Orchards

After the large scale logging was complete in the early 1900s, apple orchards were planted on the land where ponderosa pine trees once dominated. The land was then sold to land speculators back east and these “new” land owners would hire someone to manage the orchards for them. The apple boom only lasted until about the mid 1920s. Remnants of these orchards exist on the landscape today within and outside of the project area.

Recreation Use

This area receives a high amount of recreation use. The first loop of the Coyote Coulee recreation trail system is located within the project area boundary. The low elevation of the trail system allows the trail to be enjoyed year round by horseback riders, endurance riders, cross country skiers, mountain bikers and people walking their dogs. The trail system and trailhead were greatly improved when the forest entered into a partnership with a local stake holder group.

Wildlife Use

Habitat exists for several big game species. Moose (*Alces alces*) are normally seen in riparian areas and elk (*Cervus canadensis*) can be found throughout the project area. An active goshawk nest is located just outside the project area adjacent to the recreation trail. During nesting season the bird does shown aggressive behavior to the recreating public.

CURRENT CONDITION

Vegetation

Native

Forested vegetation has slowly recovered since logging occurred in the early 1900s. Vegetation today is composed of predominantly ponderosa pine (*Pinus ponderosa* Dougl.) with Douglas-fir (*Pseudotsuga menziesii* (Mirabel) Franco) in

varying amounts. As elevation increases, the land supports more of an even mixture of these two tree species. Riparian areas are composed of a variety of tree species including grand fir (*Abies grandis* (Dougl.) Lindl.), Engelmann spruce (*Picea engelmannii* Parry), quaking aspen (*Populus tremuloides* Michx.), black cottonwood (*Populus trichocarpa* T. & G.), as well as ponderosa pine and Douglas-fir.

Tree vegetation ranges in size from seedlings and saplings up to about 20 inches in diameter, with a few larger trees present in more moist areas. The majority of the conifer trees are three to 15 inches in diameter, at the most 70 feet tall and no more than 70 to 80 years old. Multiple canopy layers exist and trees are clumpy and densely stocked in appearance. Overall, tree form and crowns are good with very minor damage or forks.

Aspen

Aspen vegetation is found mainly within or immediately adjacent to riparian areas. A majority of the trees are older, small in height, and have small crown ratios. Some regeneration is occurring. Dead aspen trees are starting to fall. The overall aspen condition could be categorized as bordering on decadence in the majority of the project area.

Apple Trees

Remnants of the old apple orchards remain as an unusual but interesting component of the understory vegetation. The apple orchards are found within the project area on the north side of Hayes creek. Approximately one-third of the apple trees remain alive today and most still produce apples. The trees are very branchy and bushy (not pruned, as compared to managed apple orchards) and no more than twelve feet in height.

Fire Risk

Historic fire regimes for the project area in general would be classified as frequent, low intensity fires of low to mixed severity, typical of low elevation ponderosa pine/dry Douglas-fir sites. Fuels mainly consist of ladder fuels in the form of small size, densely stocked trees. These ladder fuels are found throughout the entire project area, situated under the crowns of the larger ponderosa pine trees as well as in dense pockets in-between the more dominant trees. Ladder fuels are also heavy in the riparian areas where a variety of conifer tree species are becoming established under the aspen. Ground fuels within the riparian area are beginning to be created as aspen trees die and fall.

Insects and Diseases

Pine engraver beetles (*Ips* Spp.) are present at endemic levels and have the potential to become a larger problem. Minor amounts of Armillaria root disease (*Armillaria ostoyae* (Romagnesi) Herink) are present and sparsely scattered in the higher elevations of the project area. Large size snags are basically non-existent and only a minor amount of smaller size snags are present and usually result from *Ips* beetle attacks or root disease.

WHY HERE WHY NOW

This project was initiated in the summer of 2003 for a variety of reasons. Fuel reduction had become a high priority for management on federal lands, especially within the wildland urban interface. The project area landscape had not experienced any commercial vegetative activity for nearly 100 years. Fuel reduction treatments had been implemented very recently on national forest land surrounding Hayes Creek. Some adjacent land owners had or were in the process of treating their land to reduce fuels and provide more protection for their homes. Treatments in Hayes Creek would nicely tie into these completed treatments. Ravalli County had completed a county wide community fire protection plan which identified the Hayes creek area as a high priority for treatment. The Healthy Forests Initiative (HFI) provided the agency additional tools that would expedite fuel reduction projects.

Healthy Forests Initiative

The Healthy Forests Initiative was proposed by President Bush in August 2002 and agencies were directed to develop administrative and legislative tools to improve regulatory processes. These tools were intended to ensure more timely decisions, greater efficiency, and better results in reducing the risk to forest and rangeland ecosystems from catastrophic wildland fires (USDA Forest Service, USDI Bureau of Land Management, 2004).

Expanded Stewardship Contracting Authority

One key component of HFI was the legislation enacted by Congress in December 2002 which expanded the stewardship contracting authority. The expanded authority allowed the Forest Service and the Bureau of Land Management to enter into long term contracts, up to 10 years in length. The contractor would keep wood products in exchange for service work of thinning trees, brush disposal and removal

of dead material. Achievement of desired results on the ground would improve forest and rangeland health and provide benefits to communities (USDA Forest Service, USDI Bureau of Land Management, 2004).

Categorical Exclusion 10

Another key component was the development of two new categorical exclusions (CE) that were adopted by the Forest Service and Bureau of Land Management in June 2003. This project fit the requirements of category 10 which allowed for up to 4500 acres of prescribed fire and up to 1000 acres of mechanical activities. Mechanical activities could include crushing, piling, thinning, pruning, cutting, shipping, mulching and mowing. This categorical exclusion allowed for the sale of treated vegetative material as long as fuel reduction was the primary objective. Other criteria to meet when utilizing this CE included:

- No new roads would be built.
- There would be no sale of vegetative material that did not have hazardous fuel reduction as its primary purpose.
- No pesticides or herbicides would be used.
- Activities proposed would only be allowed in the wildland urban interface or in condition class two or three in Fire Regime I, II or III outside of WUI.
- Treatment activities would not be proposed in wilderness areas or in wilderness study areas if the activity would impair the suitability of the study area.

Forest plan standards and guidelines, other resource management plans, and any other legally applicable direction still had to be met if the new CE's were used. Projects developed with the new categorical exclusions also needed to be identified through public processes in collaboration with; the State, local governments, Tribes, community based groups, and private landowners as well as other interested parties.

Bitterroot Community Based Wildland Fire Risk Mitigation Plan

During the winter of 2002 - 2003 the Bitterroot Community Based Wildland Fire Risk Mitigation Plan for Ravalli County was developed. The plan development followed the collaborative framework outline in the August 2001 document "A Collaborative Approach for Reducing Wildland Fire Risks to Communities and the Environment, a 10 Year

Comprehensive Strategy and Implementation Plan” (USDA Forest Service, USDI Bureau of Land Management, 2001). The Bitterroot National Forest actively collaborated with six working groups in Ravalli County. The fuel reduction group identified the Hayes Creek area as one of the high priority areas within the Hamilton Rural Fire District needing hazardous fuel reduction on both private and national forest land.

DESIRED CONDITION

Overall, the desired condition within the project area was to create an open canopy of residual trees where fire behavior would be altered. The threat of fire was from private land onto national forest land and from national forest land onto private land. Adjacent to private land the canopy openings would be the greatest, fire risk would be significantly reduced and this area would provide a safe place from which suppression activities would occur. Further away from private land, canopy openings would be more moderate and allow for changes in fire behavior, while still optimizing big game winter range habitat. Fire killed snags and down woody debris would be maintained across the landscape approximating historic conditions.

Healthy ponderosa pine would be the dominant species at lower elevations with a transition to a mix of species as elevation increased. Average size of trees would be larger but there would be fewer trees per acre. The landscape would be predominately lightly forested and single storied with scattered pockets of regeneration. Insect and disease infestation would occur at endemic levels. Aspen vegetation would begin to be rejuvenated through human created disturbance processes.

SILVICULTURE PRESCRIPTIONS

Thinning

Thinning from below to a designated spacing was the main treatment prescribed. Fire risk would be greatly reduced along the residential/wildland interface boundary for 500 feet in width in unit A (refer to figure 2), where treatment prescriptions created very open canopy conditions through a 40 x 40 foot spacing across all size classes. Further away from private land, units B1, B2, D, Z, X2 (refer to figure 2), canopy openings would be moderated yet still allow for changes in fire behavior with a 20 x 20 foot spacing across

of size classes. Losses from prescribed fire and from future wildfires would be within acceptable limits and would not result in large scale losses of existing vegetation.

Thinning would reduce stocking levels and increase the water and nutrients available for residual trees resulting in increased individual tree health and vigor. Conifer trees, aspen trees and apple trees would all benefit from an increase in available water and nutrients.

Ponderosa pine would be favored as much as possible, retaining on site a more resilient fire adapted species. In addition, since ponderosa pine was more resistant to Armillaria, retaining more pine on site would lessen the vegetation impacts resulting from the root disease. At higher elevations where a mix of species currently exists, a diversity of tree species would still be retained.

More open canopy conditions would allow for an increase in the amount of browse and foraging species favored by big game animals. Changes in canopy density would also allow for easier foraging (flying through the canopy) for goshawks (*Accipiter gentiles*) as well as an increase in its prey base (small animals) resulting from an increase in the browse component. Flammulated owl habitat does not currently exist, however thinning done now would allow for residual trees to grow larger and set the stage for creating the desired flammulated owl habitat.

Single Tree Selection

Single tree selection was prescribed for unit X1 (refer to figure 2). The vegetation composition within this treatment unit was different than the other treatment areas thus the need for a different prescription. The objective of using a single tree selection was to target for removal the majority of the Douglas-fir trees within a certain diameter range and remove all lodgepole pine to slowly change the vegetative composition back to more predominately ponderosa pine. No commercial size ponderosa pine would be removed in this unit.

Unit X2 (refer to figure 2) was a combination treatment unit including thinning across all size classes of ponderosa pine and single tree selection removing Douglas-fir within a limited diameter range and removal of all lodgepole pine. Treatments within both of these units would meet the fuel reduction and fire risk objectives as well as provide for a diversity of habitat for big game animals.

Slashing

The intent of slashing was to completely remove the ladder fuel component (non commercial size trees) to address the fire risk and fire behavior objectives. Slashing would occur within units X1 in conjunction with single tree selection, within unit C and unit Y, and riparian areas (refer to figure 2). Slashing was the only treatment prescribed within units C and Y which reflected the need based on the vegetative condition. Within riparian areas, the objective of slashing was to remove the smaller size trees that were competing with the aspen and cottonwood trees. All slashed material would be lopped and scattered, and/or piled and piles burned as needed.

Activity Fuel Treatments

Many options were available for treatment of activity fuels. One treatment prescribed that all ponderosa pine material greater than three inches in diameter had to be removed off site, chipped, or piled. The objective was to deal with the three inch and greater material that had the potential to be the catalyst for Ips insect problems.

Lop and scatter of ponderosa pine less than three inches in diameter and lop and scatter of all other tree species of non-commercial size was also utilized. This treatment allowed us to treat the non-commercial ladder fuel component found throughout the area.

Hand piling and burning piles of heavy fuel concentrations was another treatment prescribed. This treatment was specifically prescribed along the recreation trail and along the only access road to reduce risk of a fire start adjacent to these heavily used areas. This treatment could also be initiated on an as need basis within other parts of the project area.

Prescribed Fire

Given that this area historically had a fire regime of frequent fires of low to mixed severity, we wanted to begin the process of returning fire to the site. We also recognized that this meant repeated entries of fire at fairly regular intervals. The commercial treatments, understory ladder fuel treatments and activity fuel treatments would help us be successful with our first entry of prescribed fire. Prescribed fire would also serve as the second disturbance element to help rejuvenate the aspen trees. Within riparian areas prescribe fire would be the only disturbance. The vision for

prescribed fire treatments would be a mosaic of burned and unburned areas with minimal residual tree mortality.

METHODS: IMPLEMENTATION

A variety of tools were available for implementation and the one that best fit our need was the new stewardship contract. In addition, approval to utilize stewardship contracting had to be granted by the Regional Forester. The forest timber sale contracting officer, as well as regional timber administration personnel were supportive of the project and instrumental in obtaining the Regional Foresters approval.

STEWARDSHIP CONTRACTING

Integrated Resource Service Contract - Renewable Resources Handbook Direction

There were two types of stewardship contracts available for use. For this project, the value of the timber (receipts) would not be enough to pay for all the service work the project proposed, so our choice was to use the Integrated Resource Service Contract (IRSC). The IRSC allowed us to trade goods for services, obligate appropriated money to the contract as well as use receipts from other stewardship contracts to help pay for the project work. This type of contract gave us the flexibility we needed to find ways of accomplishing the work on the ground. The following is a brief list of options outlined in the stewardship contracting handbook (USDA Forest Service, 2005) that were applicable to our project:

- Stewardship contracts may be selected on a best value basis, contracts may use a performance-based format to meet end result objectives, and the of the service contract length may exceed five years but must not exceed 10 years.
- The Forest Service unit may apply the value of timber or other forest products removed as an offset against any services received.
- Designation and marking of trees, portions of trees, or special forest products may be done by persons not employed by the Secretary of Agriculture in accordance with the 16 U.S.C. 2104 Note.
- Monies received from the sale of forest products or vegetation removed under a stewardship contract may

be applied at the project site or at another stewardship contracting project site without further appropriation.

- Treatment units analyzed and approved in other NEPA documents could also be included within a single stewardship project.

FUNDING SOURCES

We needed to infuse additional appropriated money into the project to help pay for the all service work designated in the contract. This additional money supplemented the receipts from the product removal and allowed for the full range of work to be accomplished. Hazardous fuel money from forest funds contributed \$55,000. Forest Health Protection funds of \$113,250 were successfully competed for and contributed to the project. These two supplemental funding sources were specifically designated for our project and obligated in the service contract.

CONTRACT PROCEDURES

Contract Preparation

Preparation of the integrated resource service contract occurred early in the spring of 2004. Several district resource personnel were involved with writing the specifications for the service portion of the contract as well as reviewing the timber sale provisions relating to timber removal. The contract had a schedule of work that included base items which were mandatory and elective work items. The elective work items were an indefinite quantities delivery item and could be initiated at the governments discretion based on need. In addition, the contract included an equipment rental rate bid item which could be initiated by the government for work on an as need basis. Task orders would be issued identifying all work to be accomplished.

The contract utilized designation by description for the cutting of trees (USDA Forest Service, 2005). This meant that the contractor would determine, on the ground, the material that would be removed. The designation by description was documented in the silviculture prescriptions which became part of the official IRSC. The description for cutting of trees needed to be based on characteristics that could be verified after the material was cut and removed. This method was used since it was the most efficient means

of designating material for removal (minimizing tree marking by agency personnel) and because several individuals applying the description on the ground would arrive at the same end result. Inspection procedures outlined in the contract ensured that the material removed was verifiable and accountable.

We added four units, called the Waddell units, to the stewardship contract that were in need of treatment but analyzed under a different decision memo. The material to be removed in these four units was mainly saw log size with a lesser amount of service work as compared to the Hayes Creek units. The inclusion of the Waddell units gave us additional timber volume and thus value, that helped offset the cost of the service work over the entire contract area.

Mitigation items outlined in the Decision Memo were incorporated into the treatment prescriptions and the contract language. These added to the complexity of the work and included:

- Retention of visual leave islands; visual islands were marked on the ground by the wildlife biologist to address site distance concerns within the designated winter range management area. No treatment would be allowed within these leave islands.
- Thinning variance; a thinning spacing variance of 25 percent was allowed.
- Operational restrictions; operations would not be allowed on weekends and would only be permitted between dawn and 6:00 P.M daily. This allowed the recreation trail to be used in the evening and on weekends.
- Restricted season of work; season of work was designation from August 15 to November 15 to address Ips beetle concerns.

Contract Solicitation and Acceptance

Our contract solicitation was an indefinite quantities indefinite delivery, firm fixed price for contract services with provisions for timber removal. The solicitation was a request for proposal solicitation. Each prospective contractor had to submit a technical proposal telling the government how they planned to accomplish the work to meet the treatment prescription. Details of what should be addressed in the technical proposal included a list of equipment to be used, key personnel and subcontractors, a plan of operations,

and price to be paid for included timber.

During the contract solicitation period, the district held a “show me” field trip for prospective contractors. The trip was a chance to discuss the treatments, the stewardship contract and technical proposal process. We had a follow up office meeting to answer additional questions.

A panel of agency personnel assessed the submitted proposals to determine the ability of a prospective contractor to perform the work. When a proposal was selected, questions resolved and items negotiated, the contract award was made. The contractor’s proposal became part of the contract.

The selected proposal presented a different method for accomplishing the work than what we anticipated would occur. The proposal outlined using whole tree yarding and a feller buncher to cut and bunch material along designated skid trails. The proposal also included cutting the small size non commercial material and hauling this material to the landing. Cutting and hauling all material meant the landing size would be larger than anticipated and the piles at the landing would also be very large. We verified with other resource personnel that there were no additional issues associated with this change and that the intent of the project would still be met. Landing size and location was dealt with through contract administration. The landing pile size description in the contract was changed to address the potential Ips issue and the contract modified accordingly. The landing piles had to a minimum of 20 feet wide and 10 feet in height and would serve as beetle sinks, attracting local beetle populations. The intent was to keep the beetles within the pile, using the piled material as a breeding ground and not our residual standing trees.

Contract Inspections

Inspection of the treatment implementation would be done by the Contracting Officers Representative or harvest inspector. If problems surfaced, it would be dealt with through the contract process and immediate resolutions and adjustments made to the contract as needed.



3a – Feller buncher.



3b – Stroke delimber processing at landing

Figure 3a and 3b— Hayes Creek fuel reduction project, logging in progress. Photos by Kim Johnson

HARVEST ACTIVITIES

Work began on the project in August of 2004 within unit Z and the portion of unit A that was immediately adjacent to unit Z (refer to figure 2). Thinning was accomplished using a feller buncher (refer to figure 3a) and a rubber tired skidder hauled the cut and bunched material along designated skid trails to a designated landing. Approximately 160 acres were treated between these two units with 10 landings designated and utilized for processing. At the landing a stroke delimber processed each tree, removing the branches and cutting the tops off into the landing pile (figure 3b). The remaining piece was then cut and sorted into pulpwood or commercial size material. The contractor hauled the pulpwood and commercial size material to the mill and the residual tops and limbs became the large landing piles. To date, units X1, X2, and one portion of unit A still need to be treated through the contract. Units C and Y will be done with forest service crews since the treatment prescribed was slashing with no commercial component.

RESULTS

Fulfilling the Desired Objectives

Vegetation Objectives

Decreased Fire Risk

The main treatment prescribed to meet the fuel reduction objectives was thinning from below across all size classes. Adjacent to private land a very open canopy was desired with the intent of changing fire behavior and providing a safe place for firefighters. Crown fires would drop to the ground and become surface fires that were more safely and successfully attacked. Crown fire would also not be able to be initiated within this area. In addition, the reduction



4a – Before treatment.



4b – After treatment.

Figure 4a and 4b— Vegetative conditions before and after treatment in unit A. Thinning from below to a 40 by 40 foot spacing prescribed. Photos by Kim Johnson

of hazardous fuels and potential for large fire spread from national forest land to private land and from private land to national forest land would be greatly reduced. Figures 4a and 4b display the before and after treatment vegetative condition within unit A where a 40 by 40 foot spacing was prescribed.

Further away from private land, the intent of the treatments was to still change fire behavior however this change would not be as significant as the change adjacent to private land. These areas thinned to 20 by 20 foot spacing, would have a more moderated canopy opening and torching of single or groups of trees would still occur during a fire event. Figures 5a and 5b display the before and after treatment vegetative condition within unit Z. Overall within both treatment areas, ladder fuels were greatly reduced and canopy openings created at varying levels.

Field trips during and after treatments have included several fire personnel from the forest and district level. A comment from this group was that they prefer the very open thinning treatment of unit A, 40 and 40 foot spacing (figures 4a and 4b), over the moderated thinning treatment

this example for discussion and development of future fuel reduction projects.

Decreased Insect and Disease Risk

Another objective of thinning was the reduction of stand densities (figures 4a and 4b, figures 5a and 5b) and the resulting increase in the amount of nutrients and water available to the residual vegetation, thus increasing individual tree health and vigor. It is too soon to tell if tree health has increased based on width of tree rings and leader growth. Informal checks of the treated area indicate that a minor amount of mortality due to Ips infestations has occurred since thinning was accomplished.

Aspen Rejuvenation

Aspen pockets outside the riparian areas were allowed to be disturbed through the thinning and prescribed burning operations. Most of the thinning treatments have been completed however none of the prescribed fire treatments have occurred. It is too soon to tell if we have been successful in rejuvenating the aspen vegetation.

Apple Tree Protection

The prescriptions called for minimizing damage to the residual apple trees. Since it was difficult to quantify what minimal damage meant and design a specific contract provision, we had discussions with the contractor at the contract pre-work and in the field explaining that our intent was. We did not want the apple trees to be intentionally run over nor have skid trails run through them if possible. However, we recognized some



5a – Before treatment.



5b – After treatment.

Figure 5a and 5b— Vegetative conditions before and after treatment in unit Z. Thinning from below to a 20 by 20 foot spacing prescribed. Photos by Kim Johnson

damage would occur during treatment operations. Once the contactor saw the apple trees on the ground and listened to our concerns, they were very willing to work with the forest to ensure minimal damage occurred. It is too soon to tell if our thinning treatments have enhanced the overall health of the remaining apple trees however, at the time of this writing, some of the apple trees are producing fruit.

Wildlife Objectives

Current research regarding big game habitat needs in winter range focuses more on forage availability than thermal cover however, both issues were addressed (Lockman, 2007). Creating a less dense canopy through thinning treatments would increase the diversity of plant communities available as forage for big game animals. Riparian Habitat Conservation Areas, designated in the project design to address fisheries issues, also functioned as untreated thermal cover corridors for big game. While we feel the objective was met, it will be several years before we start to see a change in plant community diversity.

The visual leave islands addressed sight distance concerns during hunting season and also served to maintain small patches of habitat diversity as well as some structural diversity throughout the area.

The active goshawk nest location was considered during the project design and an area adjacent to the nest site was not treated. The district wildlife biologist (Lockman, 2007) commented that while the project area was not goshawk nest habitat, it did provide foraging habitat. Thinning treatments would reduce stem densities, specifically of the understory vegetation. Plant community diversity would increase with a reduction of tree stems which would in turn increase the diversity of wildlife species available for foraging goshawks. It will be several years before we actually begin to see this change.

The wildlife biologist also commented that thinning treatments in the low elevation ponderosa pine vegetation type will help set up the trajectory to provide for future flammulated owl habitat. Habitat for flammulated owls



6a – Trail before treatment.



6b – Trail after treatment resolution.

Figure 6a and 6b— Coyote Coulee recreation trail before and after vegetative treatment activities occurred. Photos by Kim Johnson

(*Otus flammeolus*) does not currently exist within the project area. By favoring ponderosa pine trees and reducing stocking levels, residual tree health and vigor would be increased. At some point in the future, a healthy mature to old growth ponderosa pine forest with snags, desired by flammulated owls will result.

Human Need Objectives

Recreation Trail

The original prescriptions were no different along the recreation trail than within any other portion of the treatment units. Figure 6a shows the vegetation along the trail prior to treatment. One stakeholder group expressed concern over the open forest appearance along the recreation trail after some of the treatment had occurred. The district and the contractor worked with the group to reach a favorable resolution. The resolution changed the treatment along the trail. Instead of thinning from below, a 50 foot semi-treated buffer along each side of the trail would occur. For 25 feet in width on each side of the trail no trees would be removed and within the next 25 feet only the commercial size trees would be removed. Figure 6b shows the vegetative condition trail after the treatment change. This left a more densely stocked area adjacent to the trail and a less stocked area further away. The stakeholders were told that if a fire event occurred, the trail corridor vegetation would most likely suffer extensive damage because the tree canopy was not opened up and ladder fuels still existed. They were willing to accept this risk. Addressing their concerns and reaching an agreeable resolution was important to the group.

External and Internal Relationships

During implementation, we worked with an adjacent landowner who had expressed concerns about the large landing piles (refer to figure 7). He understood the concept



Figure 7—Hayes Creek fuel reduction project, large landing piles.
Photo by Kim Johnson

of the large piles acting as beetle sinks, but was afraid the beetles would still come out of the piles and attack trees on his property. To address his concerns, we pursued additional protection at the three large landing piles closest to his property and installed funnel traps to catch the stray beetles. The funnel traps were installed prior to the 2005 spring emergence of the beetle and were monitored weekly during the spring and summer beetle flight season. What we accomplished with the funnel traps was difficult to assess with any degree of certainty. Responding to the adjacent landowners concerns and maintaining good relationships far outweighed any benefit the funnel trapping may have provided.

A main component of our implementation success was the upfront willingness of our contractor to do a good quality job. In that regard they asked questions for clarification and took time out while running harvesting equipment to talk with the public and agency personnel who visited the project site. They fostered a good working relationship with the agency when issues and concerns arose by helping to reach solutions.

Another part of our success was attributed to the support of the Region and Forest for assisting us as requested during the contract preparation and solicitation process. Since the IRSC was a new format and untested on our forest, it presented challenges to agency personnel as well as prospective contractors. Regional and Forest timber administration specialists worked with us and prospective contractors to answer questions and resolve concerns.

Interpretative Kiosk Benefits

Fund received from our Forest Health Protection proposal included not only thinning dollars, but also money to design and install a three panel interpretative kiosk at the Coyote Coulee trailhead, the recreation access to the trail within the project area. The intent was to provide basic information to the public about forest ecology and why the need for treatment in the Hayes Creek area. Figure 8 shows the three panel kiosk. One panel presented information on fires historic role in the forest and how the forest has changed over time. Another panel presented information and a map of the coyote coulee trail winding through the project area. The last panel displayed the proposed treatments for the project with photos of similar work done that would provide an example of how the treatment in Hayes Creek would look. Other trail information and notices could also be posted at the kiosk.

Economic Benefits

Product removal for the contract was measured in tons. The cruise volume for the Hayes Creek units was approximately 5115 total tons, for an average of 12 tons per acre.¹ This translated into a total of 1650 hundred cubic feet (CCF) and an average of 3.75 CCF per acre or a total of 825 thousand board feet (MBF) and an average of 1.85 MBF per acre. These figures only included commercial size material.

The stewardship contract allowed for the removal of additional material “subject to agreement”. At the contractor’s request, we made an agreement that allowed them to remove pulpwood material from the project area. This agreement was a win-win situation because more material was utilized as product and not left in landing piles. With the addition of the pulpwood material, the amount of material removed to date almost equals the amount of material offered in the service contract solicitation.

The Fuels for Schools program in Darby, Montana, also benefited from our project.² Receipts from other stewardship projects, \$12,485, were designated to pay for the chipping

¹ Timber cruise volume estimates (2004) for the Hayes Creek Fuel Reduction Project are on file at the Bitterroot National Forest, Darby Ranger District.

² The fuels for schools program is sponsored by the USDA Forest Service and is designed to convert unmarketable material from thinning and logging into low cost heat for schools.



Figure 8— *Interpretive kiosk at Coyote Coulee trailhead.* Photo by Kim Johnson

of residual slash from three large landing piles and hauling this chipped material to the Darby school. This work was done in the winter of 2005 - 2006. Approximately 383 tons or 12 log truck loads were hauled to the school. The amount of material chipped and hauled would provide heat for the school for approximately half a winter.

Lastly, \$7340 of receipts from other stewardship projects on the forest was transferred into the project. That money helped supplement the other appropriated funding so additional work could be accomplished.

DISCUSSION

Lessons Learned

This project was very challenging and we learned many lessons while working through the entire process. A mix of appropriate agency personnel need to be involved during various phases of the project. It is best to have them involved up front rather than after the fact.

Having an open mind and being responsive to different ways of accomplishing the work was critical to our success. This meant modifying and tweaking treatment prescriptions while still meeting the intent of the fuel reduction objectives.

We faced many bumps and hurdles along the way. Addressing and solving the problems was not difficult but time consuming and required us to come up with creative and innovative solutions.

We found that we improved and maintained our working relationships with stakeholders, individual land owners and the contractor by working in a collaborative effort that allowed us to deal with conflicts and find solutions that were acceptable to all interested parties.

Accomplishing our fuel reduction treatments was not an easy task. It took considerable time, energy and commitment from agency personnel. Having the proper attitude to work through problems and find creative solutions was the key ingredient. This led to a sense of accomplishment and made for a very rewarding experience.

CONCLUSIONS

Being able to share and showcase a successful fuel reduction project was the goal of this paper. The phrase “timing is everything” certainly sums up our success. We were able to take advantage of all the tools in the toolbox from utilizing a new categorical exclusion for planning to implementation through the use of a stewardship contract. The project required commitment, innovative thinking and creative solutions to problems. It required us to be receptive to different ways of doing business. Lastly we had to be willing to work with a variety of people, internal and external, to gather as much support as possible to be able to implement the treatments. Being successful with this project will set the stage for being successful with future fuel reduction projects.

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Proceedings of the 2007 National Silviculture Workshop

Long-Term Basal Area and Diameter Growth Responses of Western Hemlock-Sitka Spruce Stands in Southeast Alaska to a Range of Thinning Intensities

Nathan J. Poage

ABSTRACT

To better understand the long-term basal area and diameter growth response of young, well-stocked, even-aged, mixed-species stands of western hemlock (*Tsuga heterophylla* (Raf.) Sarg.) and Sitka spruce (*Picea sitchensis* (Bong.) Carr.) to a range of thinning intensities (heavy, moderate, light, and unthinned), 20 years of post-treatment (i.e., thinning) data were analyzed from 128 permanent study plots at 32 installations in southeast Alaska. Thinning significantly increased the basal area growth of the thinned plots relative to that of the unthinned plots. Basal area growth of the thinned plots decreased with increasing thinning intensity, likely a consequence of the reduction in

growing stock with increasing thinning intensity. Diameter growth increased with increasing thinning intensity. Younger stands (typically with smaller pre-treatment basal areas and diameters) responded more vigorously to thinning than did older stands, as did stands growing on more productive sites. The models developed in this analysis can be used by land managers to make general predictions of how western hemlock-Sitka spruce stands similar to those investigated in southeast Alaska are likely to respond to different thinning intensities in terms basal area and diameter growth during the first 20 years post-treatment.

Keywords: *Picea sitchensis*, southeast Alaska, thinning, Tongass, *Tsuga heterophylla*, young stand growth.

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INTRODUCTION

The majority of the forests along the North American Pacific Northwest coast are mixed-species stands containing a significant proportion of western hemlock (*Tsuga heterophylla* (Raf.) Sarg.) and, to a lesser degree, Sitka spruce (*Picea sitchensis* (Bong.) Carr.), western redcedar (*Thuja plicata* Donn), yellow cedar (*Chamaecyparis nootkatensis* [D. Don] Spach), and mountain hemlock (*Tsuga mertensiana* (Bong.) Carr.). Spanning over 22° of latitude, coastal hemlock–spruce forests occur from northern California to south-central Alaska, a distance of more than 1500 miles (Harris 1990). Low respiration demands associated with cool temperatures and abundant precipitation throughout the year in these coastal forests have led to some of the highest recorded accumulations of above-ground biomass in the world (Waring and Franklin 1979).

Mixed-species stands of hemlock and spruce regenerate naturally following harvesting and heavy thinning in southeast Alaska (Ruth and Harris 1979, Deal and Farr 1994, Zasada and Packee 1994, Deal and Tappeiner 2002, Deal et al. 2002). In southeast Alaska, harvesting of primary forest has led to the establishment of an estimated 649,000 ac of western hemlock–Sitka spruce young-growth stands since the 1950s (Barbour et al. 2005). Interest in actively managing these naturally regenerated young-growth stands led, in the early 1970s, to the initiation of the USDA Forest Service's Cooperative Stand-Density Study (CSDS; DeMars 2000).

A primary objective of the CSDS is addressed in this paper: to understand the long-term basal area and diameter growth responses of young, well-stocked, even-aged, mixed-species stands of western hemlock and Sitka spruce

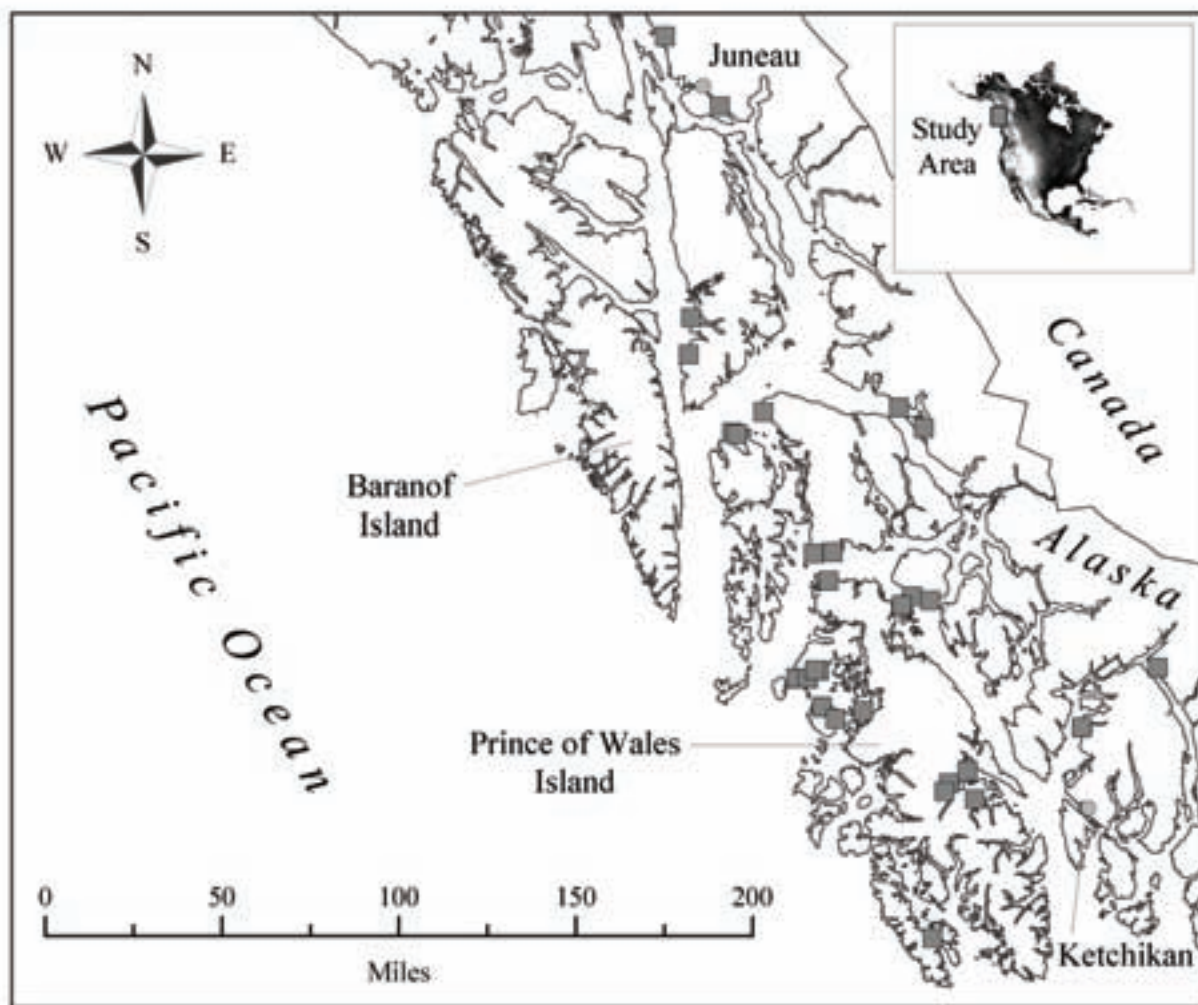


Figure 1— Locations of the 32 Cooperative Stand Density Study installations (red squares) discussed in detail in this paper.

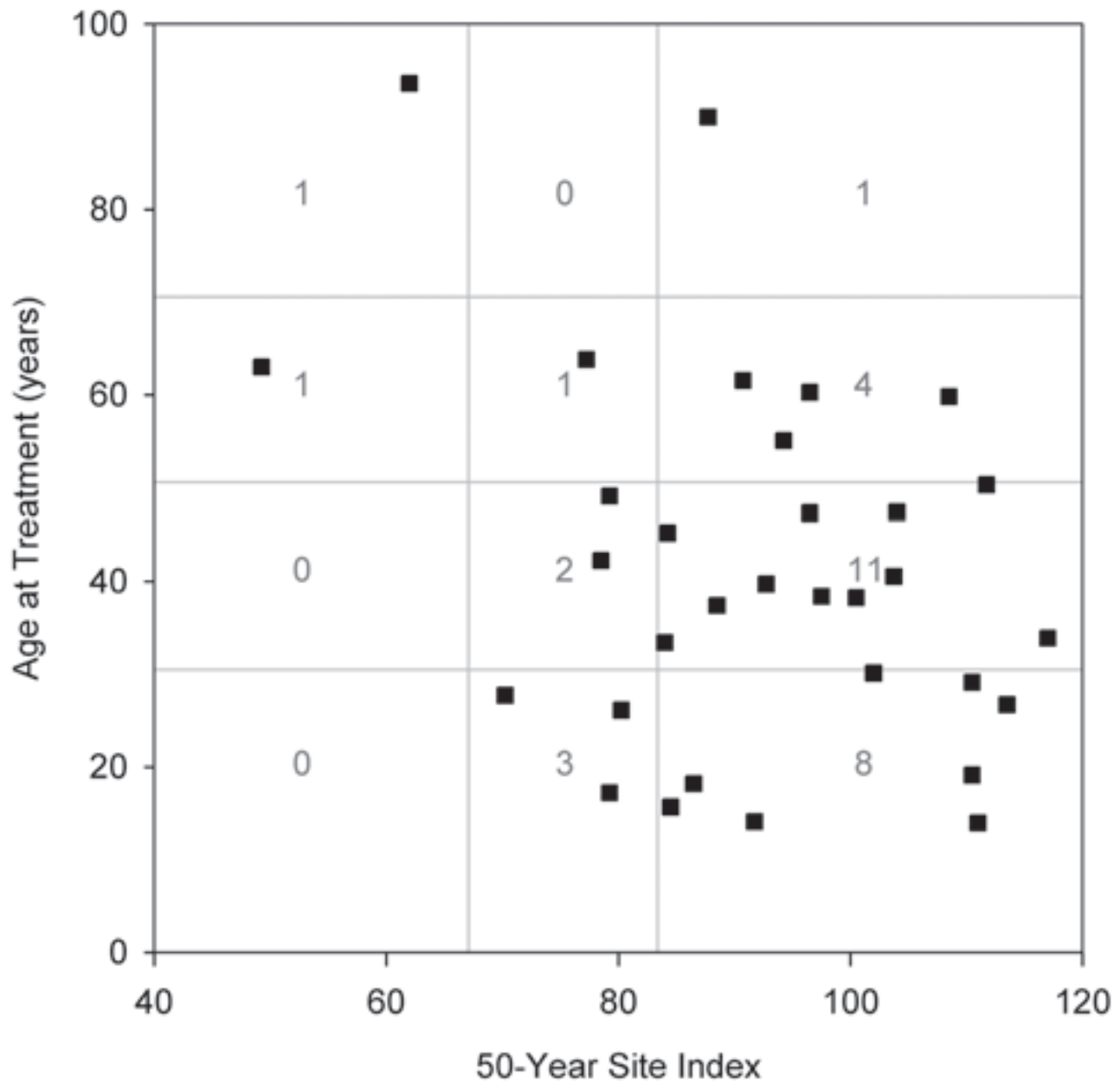


Figure 2— Mean age at treatment over 50-year site index (Farr 1984) for 32 installations in southeast Alaska. Gray lines indicate the breaks between the 4 age classes and 3 site index classes defined in the original study plan for the Cooperative Stand Density Study (Farr 1976, DeMars 2000). Gray numbers indicate the number of installations (out of 32) occurring in each of the original 12 combinations of age at treatment class $\times$ site index class defined in the original study plan.

in southeast Alaska to a range of thinning intensities (Farr 1976, DeMars 2000). Of particular interest is to better understand how the basal area and diameter growth responses are influenced by age and stand structure at the time of thinning and site productivity (site index).

METHODS

Twenty years of post-treatment (i.e., thinning) data were analyzed from 128 permanent study plots (96 thinned plots and 32 unthinned plots) maintained by the USDA Forest Service at 32 sites or installations in southeast Alaska (Figure 1). The north-to-south distribution of the plots

was approximately 250 mi. Most plots were located close to the coastline at elevations of < 500 ft. All plots were located in the *Picea sitchensis*-*Tsuga heterophylla* forest type (Franklin and Halpern 2000). More than 95% of total basal area at each plot was made up by hemlock and spruce. The stands containing the plots were between 13 and 94 years old at the time of plot establishment; 50-year site index (Farr 1984) ranged between 49 and 118 (Figure 2). Pre-treatment basal areas (BA) were between 34 and 407 ft²/ac; pre-treatment quadratic mean diameters (QMD, the diameter of tree of average basal area) were between 1.2 and 16.3 in. DeMars (2000) provides a more detailed description of the study area.

All installations meeting 2 criteria were selected for analysis. First, the plots at each installation had measurements made on live trees for least 20 years of post-treatment measurements (generally made every 2–4 years). Second, an installation comprised 1 unthinned “control” (C) plot, 1 lightly (L) thinned plot, 1 moderately (M) thinned plot, and 1 heavily (H) thinned plot. Thirty-two installations

(96 thinned and 32 unthinned plots) met these selection criteria.

All plots were square, 0.2 ac in size, and centered in the middle of a 1.0 ac treated area (Farr 1976, DeMars 2000). Each thinned plot was thinned once from below at plot establishment, generally to a uniform spacing. Pre-treatment QMD and thinning guidelines included in the CSDS study plan (Farr 1976, DeMars 2000) were used to determine post-treatment BA and QMD, average spacing, and number of trees per acre (TPA) for the L, M, and H thinning treatments. Preliminary analysis of the thinning guidelines indicated that the L, M, and H thinning treatments reduced stand density—expressed in terms of either relative density (Curtis 1982) or maximum stand density index (Poage et al. 2007)—to approximately 50% (L), 35% (M), and 25% (H) of full stocking (Figure 3).

Species and diameter at breast height (dbh, the diameter at 4.5 ft above the ground) were recorded for each live tree with a dbh > 0.5 in within each plot immediately pre- and

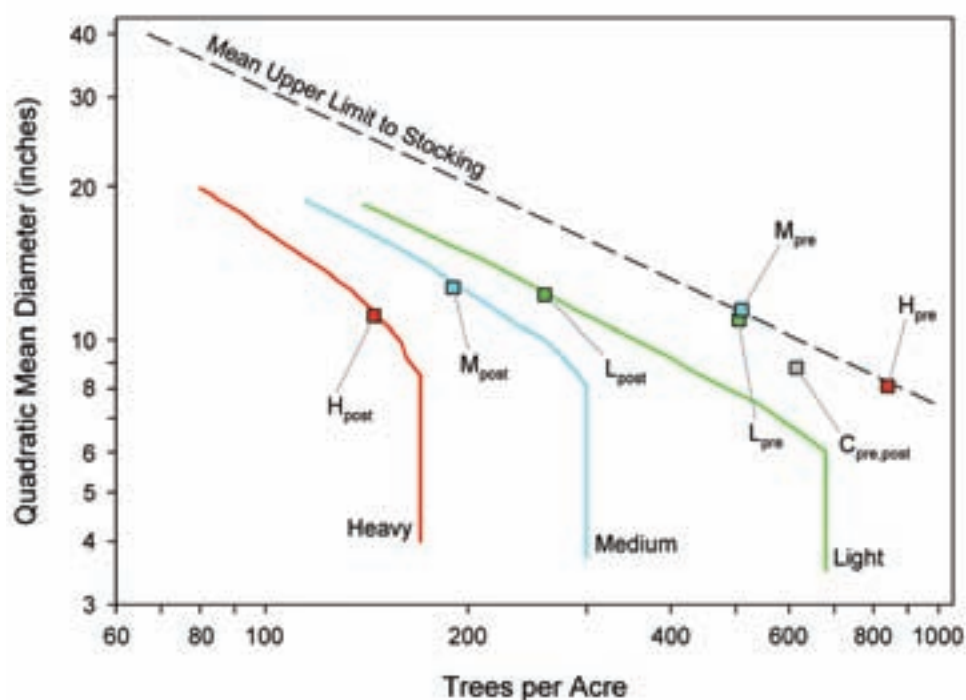


Figure 3—Thinning guidelines defined in the original study plan for the Cooperative Stand Density Study (Farr 1976, DeMars 2000). Post-treatment quadratic mean diameters and trees per acre are shown for lightly (L) thinned plots in green, moderately (M) thinned plots in blue, and heavily (H) thinned plots in red. To illustrate, the squares indicate the pre- and post-treatment quadratic mean diameters and trees per acre for the 4 plots at one installation. The dashed black line labeled Mean Upper Limit to Stocking represents the mean maximum stand density index (SDImax = 619) recently identified for western hemlock–Sitka spruce stands in southeast Alaska (Poage et al. 2007).

post-treatment (i.e., at 0 years since treatment). Exceptionally dense young stands (generally greater than 1500 trees/acre) were sub-sampled using nine 0.004 ac circular subplots spaced at 23.3 ft intervals on a 3 x 3 grid. All trees measured at plot establishment were tagged. For each plot, total and species-specific BA, TPA, and QMD were calculated pre- and post-treatment (i.e., at 0 years since treatment) and for the measurements closest to 5, 10, 15, and 20 years since treatment. Stand age and site index were determined at the time of treatment (DeMars 2000). DeMars (2000) provides additional details about the plot design and sampling scheme.

How **BA** and diameter (**QMD**) growth responses during the 20 years post-treatment were influenced by thinning intensity, age when thinned, pre-treatment **BA**, pre-treatment **QMD**, site index, and years since treatment was modeled using a repeated measures approach and the **PROC MIXED** procedure of SAS (2003). Restricted/residual maximum likelihood (**REML**) was selected as the estimation method.

Backwards, stepwise selection of explanatory variables resulted in the final mixed models for **BA** and **QMD** shown in Equation 1:

$$y_{jkt} = (\beta_0 + \beta_1 BA_{pre,jk} + \beta_2 QMD_{pre,jk} + \beta_3 SI_{jk} + \beta_4 L_{jk} + \beta_5 M_{jk} + \beta_6 H_{jk}) + \\ time \times (\beta_7 + \beta_8 QMD_{pre,jk} + \beta_9 SI_{jk} + \beta_{10} L_{jk} + \beta_{11} M_{jk} + \beta_{12} H_{jk}) + \\ b_j + e_{jkt}$$

where y_{jkt} is the **BA** or **QMD** of plot k of installation j in year t ,

$BA_{pre,jk}$ is the pre-treatment **BA** of plot k on installation j ,

$QMD_{pre,jk}$ is the pre-treatment **QMD** of plot k on installation j ,

SI_{jk} is the 50-year site index of plot k on installation j ,

L_{jk} , M_{jk} , and H_{jk} are indicator variables that take the value of 1 if plot k of installation j received the light, medium, or heavy thinning treatment, respectively, and 0 otherwise,

$time$ is year t since the thinning treatment,

b_j is the random effect of installation j , and

e_{jkt} is the random error.

It is assumed that b_j is normally distributed with mean 0 and variance $\text{Var}(b_j) = \sigma_b^2$.

The vector of errors of the t observations from plot k of installation j , $\mathbf{e}_{jk}$, is assumed to be normally distributed with mean $\mathbf{0}$ and covariance matrix $\text{Var}(\mathbf{e}_{jk}) = \Sigma$. It is further assumed that b_j and $\mathbf{e}_{jk}$ are independent and that observations from different installations are also independent

Note that Equation 1 is structured to highlight the variables associated with the immediate post-treatment **BA** or **QMD** (i.e., the intercept term) in the first row, the variables associated with the change in **BA** or **QMD** over time (i.e., the slope term) in the second row, and the error terms in the third row.

RESULTS AND DISCUSSION

The plots responded to thinning as one might have predicted. Thinning significantly increased the basal area (**BA**) growth of the thinned plots relative to that of the unthinned plots, but **BA** growth of the thinned plots decreased with increasing thinning intensity (Figure 4, Table 1). The highest **BA** growth was observed in the lightly (**L**) thinned plots. **BA** growth of the **L** plots was, on average, 2.18 ft²/ac/yr greater than that of the unthinned, control (**C**) plots

(see β_{10} , the time $\times$ **L** parameter estimate for the **BA** mixed model in Table 1). Average **BA** growth of the moderately (**M**) and heavily (**H**) thinned plots was 1.68 ft²/ac/yr and 1.16 ft²/ac/yr greater, respectively, than that of the **C** plots (see β_{11} and β_{12} for the **BA** mixed model in Table 1). The observed decrease in **BA** growth with increasing thinning intensity was likely a consequence of the corresponding reduction in growing stock with increasing thinning intensity. Although the **BA** growth of individual trees was greater in heavier thinnings than in lighter thinnings, too few trees were retained in heavier thinnings to generate the same plot-level **BA** growth observed in lighter thinnings.

In contrast to the **BA** growth of plots, diameter (**QMD**) growth increased with increasing thinning intensity (Figure 5, Table 1). Thinning significantly increased the **QMD** growth of the thinned **M** and **H** plots compared to that of

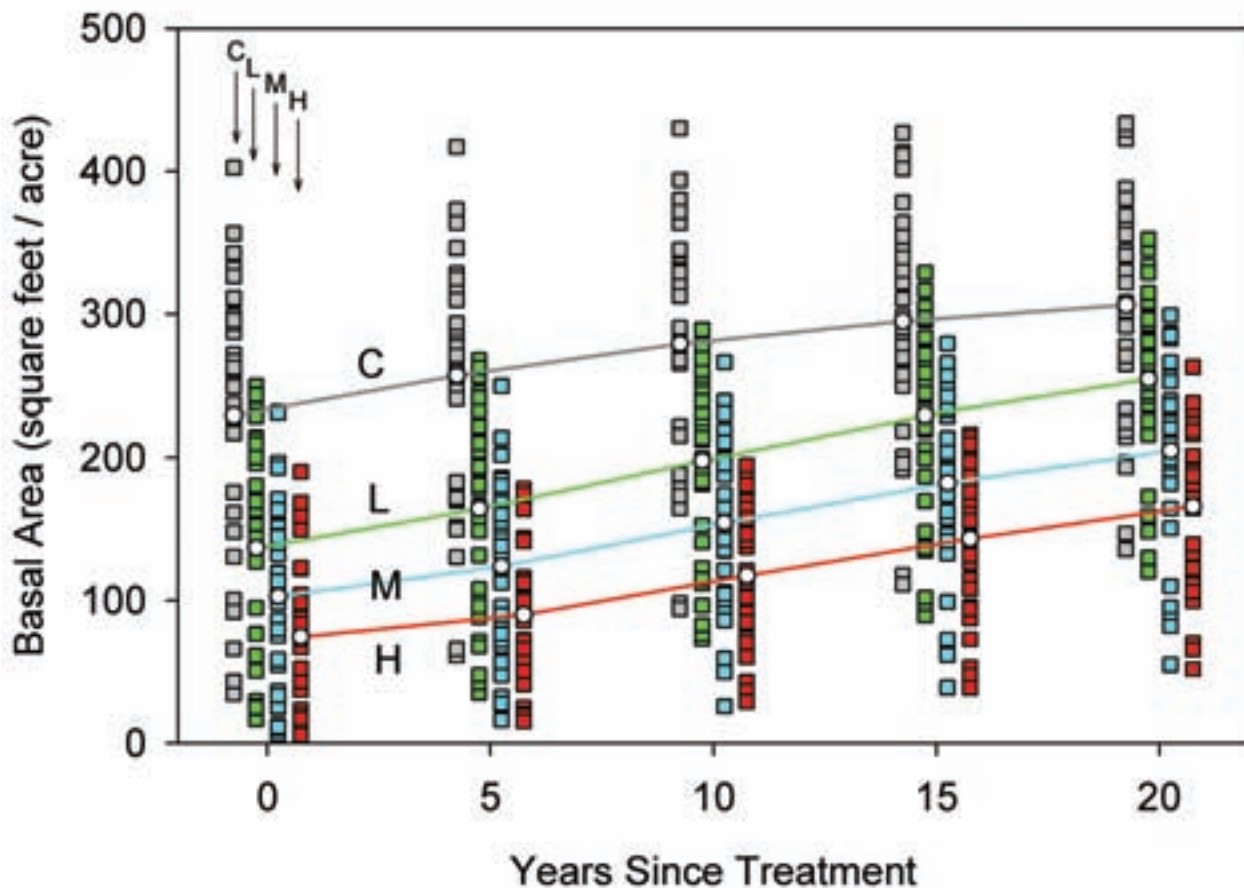


Figure 4— Plot-level basal area data shown by treatment at 0, 5, 10, 15, and 20 years since treatment. Data for unthinned “control” (**C**) plots are shown in gray, lightly (**L**) thinned plots in green, moderately (**M**) thinned plots in blue, and heavily (**H**) thinned plots in red. White circles indicate the average basal areas for each treatment at 0, 5, 10, 15, and 20 years since treatment.

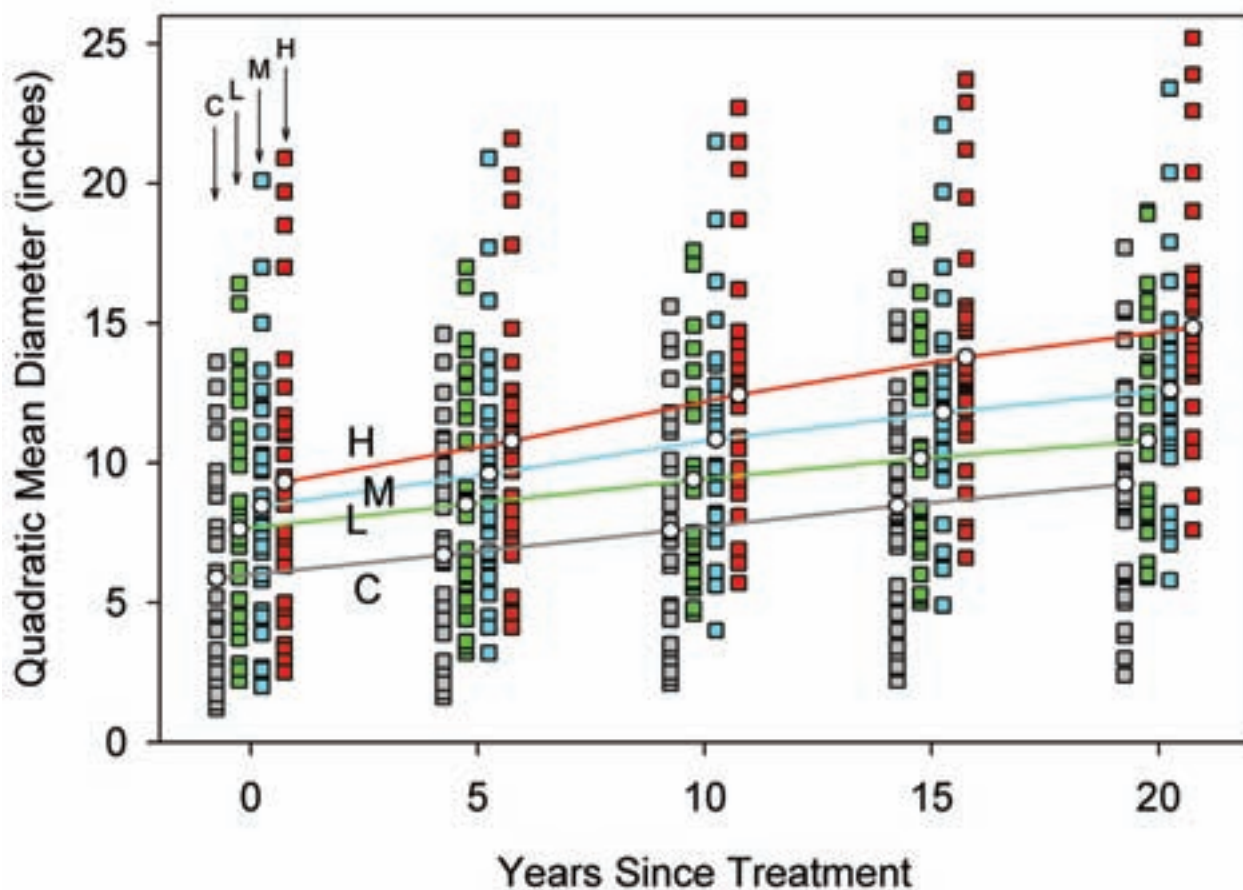


Figure 5. Plot-level quadratic mean diameter data shown by treatment at 0, 5, 10, 15, and 20 years since treatment. Data for unthinned “control” (C) plots are shown in gray, lightly (L) thinned plots in green, moderately (M) thinned plots in blue, and heavily (H) thinned plots in red. White circles indicate the average quadratic mean diameters for each treatment at 0, 5, 10, 15, and 20 years since treatment.

the unthinned C plots. Average QMD growth of the M and H plots was 0.03 in/yr and 0.09 in/yr greater, respectively, than that of the C plots (see β_{11} and β_{12} for the QMD mixed model in Table 1). Average QMD growth of the L plots was not significantly different than that of the C plots.

Younger stands responded more vigorously to thinning than did older stands, as did stands growing on more productive sites. Both BA and QMD growth decreased with increasing pre-treatment QMD, pre-treatment BA, and age when thinned. Age when thinned, pre-treatment QMD, and pre-treatment BA were all positively correlated (i.e., an older plot typically had a larger QMD and more BA before thinning than a younger plot). The positive correlations between these three explanatory variables resulted in age when thinned being dropped as an explanatory variable

from the final models for both BA and QMD (Equation 1). Both BA and QMD growth increased with increasing site index (see β_9 for the BA and QMD mixed models in Table 1).

The models developed for BA and QMD can be used by land managers to make general predictions of how western hemlock-Sitka spruce stands similar to those investigated in southeast Alaska are likely to respond to different thinning intensities in terms BA and QMD growth during the first 20 years post-treatment. To illustrate, consider a 40-year-old stand with a pre-treatment BA of 275 ft²/ac, a pre-treatment QMD of 6.5 in, and an estimated 50-year site index of 85. Equation 1 can be used to compare how, for example, the BA of this stand might develop over 20 years if it were a) lightly thinned and b) heavily thinned.

Table 1. Parameter estimates for the final basal area and diameter mixed models (Equation 1). With the exception of those parameter estimates indicated as not significant by the “n.s.” superscript, all parameter estimates were significantly different than zero ($p < 0.05$). L, M, and H are indicator variables for the light, medium, and heavy thinning treatments, respectively.

Variable	Basal Area (<i>BA</i>)		Diameter (<i>QMD</i>)	
	Estimate	(s.e.) [*]	Estimate	(s.e.) [*]
(β_0) Intercept	13.3303 ^{n.s.}	(17.5978)	-2.4115	(0.4505)
(β_1) Pre-treatment Basal Area (<i>BA_{pre}</i>)	0.4803	(0.0430)	0.0039	(0.0011)
(β_2) Pre-treatment Quadratic Mean Diameter (<i>QMD_{pre}</i>)	5.3355	(1.1599)	0.9769	(0.0299)
(β_3) Site Index (<i>SI</i>) [†]	0.8704	(0.1763)	0.0167	(0.0045)
(β_4) Light thinning (<i>L</i>)	-92.1263	(6.2740)	2.1071	(0.1650)
(β_5) Medium thinning (<i>M</i>)	-125.3000	(6.2544)	2.6730	(0.1642)
(β_6) Heavy thinning (<i>H</i>)	-150.0500	(6.2526)	3.4786	(0.1646)
(β_7) <i>time</i> [‡]	-1.2338 ^{n.s.}	(0.6909)	-0.0104 ^{n.s.}	(0.0226)
(β_8) <i>time</i> × <i>QMD_{pre}</i>	-0.3710	(0.0299)	-0.0095	(0.0010)
(β_9) <i>time</i> × <i>SI</i>	0.0748	(0.0072)	0.0026	(0.0002)
(β_{10}) <i>time</i> × <i>L</i>	2.1824	(0.3277)	-0.0170 ^{n.s.}	(0.0107)
(β_{11}) <i>time</i> × <i>M</i>	1.6815	(0.3282)	0.0305	(0.0107)
(β_{12}) <i>time</i> × <i>H</i>	1.1622	(0.3277)	0.0940	(0.0107)

^{*} standard error of parameter estimate

[†] 50-year site index (Farr 1984)

[‡] years since treatment (i.e., thinning)

(The following steps are perhaps most easily done using an electronic spreadsheet.) Begin by inserting the parameter estimates for **BA** in Table 1 into Equation 1 (e.g., $\beta_1 = 0.4803$). Add the pre-treatment **BA** (275 ft²/ac), pre-treatment **QMD**

(6.5 in), and site index (85) where indicated in Equation 1. When **L** = 1, **M** = 0, **H** = 0, and time = 0, Equation 1 predicts a **BA** of 162 ft²/ac immediately post-treatment for the lightly thinned stand; 20 years after thinning (i.e., time

= 20) the lightly thinned stand is predicted to be 260 ft²/ac. Similarly, if **L** = 0, **M** = 0, **H** = 1, and time = 0, Equation 1 predicts that the heavily thinned stand will have a **BA** of 104 ft²/ac immediately post-treatment; 20 years after thinning the heavily thinned stand is predicted to be 182 ft²/ac. In this scenario, average annual **BA** growth during the 20 years post-treatment is predicted to be 25% greater in the lightly thinned stand (4.9 ft²/ac/yr) than in the heavily thinned stand (3.9 ft²/ac/yr).

This paper is the first to address the primary objective of the Cooperative Stand-Density Study (CSDS): to understand how young hemlock-spruce stands of different ages and site productivities respond to a range of thinning intensities. The primary reason for this is simply a matter of time. Over a decade was required to establish the study plots (1974-1987). Additionally, if long-term responses to thinning are to be analyzed, decades must pass post-treatment; to analyze 20 years of post-thinning growth response requires that a sufficient number of plots have had 20 or more years pass following treatment. It is sincerely hoped that the CSDS plots—which are now beginning to pay long-term research dividends—continue to be maintained and measured into the future.

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Developing a Unified Monitoring and Reporting System: A Key to Successful Restoration of Mixed-Oak Forests Throughout the Central Hardwood Region

Daniel A. Yaussy, Gregory J. Nowacki, Thomas M. Schuler, Daniel C. Dey, and Eugene J. DeGayner

ABSTRACT

Many national forests and grasslands in the Central Hardwoods region of the United States recently have undergone Land Management Plan revision, which include management areas that promote restoration through a variety of management activities. Monitoring is a vital component of adaptive management whereby the effects from a variety of treatments (including controls) can be analyzed and compared all within a landscape context. A statistically sound and cost-effective opportunity is presented through a unified monitoring effort for national forests spanning the Central Hardwoods region. Statistical power will be gained by increased replications across the landscape and cumulative effects will be addressed more comprehensively. Sharing a common protocol for monitoring activities and a reporting system will enable collective analysis and inference. Challenges will undoubtedly arise in forming a unified

monitoring system across multiple forests. The objectives of each forest's management areas and the measures of restoration success need to be similar and reconcilable among the forests. The process of developing and implementing a unified monitoring system must be mutually accepted and financially supported by participating Districts, Forests, Regions, and Research Stations.

An opportunity to efficiently and effectively monitor the ecological restoration of Central Hardwoods region is taking shape. It may be economically beneficial to the Regions and the forests if quick action to take advantage of the "economies of scale" available in designing and collectively implementing a unified monitoring system across forests occurs before each forest commits time, effort and funding to develop their own monitoring system.

Keywords: central Hardwoods, ecosystem restoration, monitoring, surface fires, adaptive management

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INTRODUCTION

Much of the eastern United States was covered by oak-dominated forests (*Quercus sp.*), woodlands, and savannahs prior to Euro-American settlement. Charcoal and pollen records (Delcourt 2002), fire-scars (Guyette and others 2006), and anecdotal accounts from early Euro-American settlers (Whitney 1994) indicate the widespread use of fire by Native Americans to manage habitats. Euro-Americans continued, and possibly increased, burning practices on many of these landscapes (Delcourt 2002, Schuler and McClain 2003) until fire suppression policies were implemented in the 1920s.

Oak-hickory remains the most abundant forest type in the United States, with oak dominated stands that regenerated prior to fire suppression. Because fire was removed as a natural process from these ecosystems, understories readily converted to shade-tolerant, fire-sensitive species such as red and sugar maples (*Acer rubrum*, *A. saccharum*), beech (*Fagus grandifolia*), and blackgum (*Nyssa sylvatica*) (Abrams 1992, 1998; Schuler 2004). Under these conditions, subsequent timber management that removes overstory canopies, releases these species to dominate the future forest (Abrams and Nowacki 1992). In heavily disturbed areas that are exposed to full sunlight by clearcutting, germinants of shade intolerant species such as yellow-poplar (*Liriodendron tulipifera*) and black cherry (*Prunus serotina*) grow rapidly in height to dominate the regeneration (Brashears and others 2004, Loftis 1990).

In the recent round of Land and Resource Management Plan (forest plan) revisions conducted by the national forests and grasslands in the Eastern and Southern Regions, there has been an increased emphasis on restoring and sustaining oak-dominated ecosystems (Forest Service 2004a-d, 2005a, 2005c, 2006a-e, 2007a-b). Multiple benefits are achieved by maintaining oak-dominated ecosystems through active management, including timber products, wildlife food (mast) and habitat, recreation, and rare plant species (Hutchinson and others 2005, McShea and Healy 2000). A framework for ecosystem restoration established by the National Forest System (Day and others 2006) provides a definition of ecosystem restoration and guiding principles such as a National Strategy and the need for collaboration in attaining these goals. Integration and collaboration within the agency are necessary to maximize limited resources be-

cause the magnitude of ecosystem restoration needs greatly exceeds the financial capacity of disparate units of the Forest Service or other resource management organizations. Equally important are external partnerships as ecosystems do not stop at administrative boundaries.

As stated in their forest plans, each national forest (NF) within the Central Hardwoods region will monitor their activities to determine if management actions are achieving the desired future conditions (DFCs). The movement toward these DFCs will be most easily evaluated if they are written with reasonable quantitative metrics as defined in forestry, ecology, and wildlife sciences. Current management activities include: harvesting, prescribed fire, herbicide treatments, and deer abatement techniques. Most monitoring schemes require collection and summarization of data to determine pre- to post-treatment changes and whether advances toward DFCs are truly taking place. Data collected from a single site can be used to explain specific management effects on that specific area. However, credibility is reduced when those results are extrapolated to other sites. For instance, if the Shawnee NF in Illinois finds that frequent, low-intensity fire improves plant biodiversity, it may be sensible to conclude that the same will hold true on the Mark Twain NF (Missouri) or Hoosier NF (Indiana). However, without data from neighboring forests, evidence supporting this claim could be called into question for the Uwharrie NF (North Carolina). As such, the networking of sites allows data comparisons of local vs. regional trends with more confident extrapolation of findings. The National Ecological Classification System (ECS) can be used to stratify the landscape and help guide placement of sites for monitoring. As data accumulate, we are more confident about making claims of positive movement toward DFCs as well as our understanding of what might cause differences in effects. The level of data collection needed to recognize a trend may be more than a single NF would be willing to fund. However, if the intensity of data collection was distributed across several NFs with similar management areas, in similar ECS units, and DFCs, the cost burden would be reduced substantially while maintaining the statistical power of the monitoring. Understanding the differences between sites can be quantified and interpolation of results to new sites is more defensible using an ECS framework.

The need and opportunity exists to establish a coordinated, unified monitoring system for managed oak-domi-

nated stands in eastern forest. However, there are many impediments that must be overcome in the adoption of such a monitoring system. This paper identifies some of these issues.

OBJECTIVES

Several NFs in the Central Hardwoods region have management areas on which they intend to utilize partial harvests and prescribed fire for restoration purposes. In this context, partial harvests are defined as shelterwood harvests and thinning from below, removing the mid-story and leaving the most dominant trees. However, restoration goals and the silvicultural practices used are not necessarily identical among management units or NFs. Typically, forest structure, oak regeneration, and habitat development are listed as objectives of the prescriptions, but these may be given different priorities among the several NFs. For instance, the Wayne NF (Ohio) intends its "Historical Forest" management area to mimic pre-Euro American settlement forests: open woodlands maintained with frequent, low-intensity fires. This forest structure should provide critical habitat for native wildlife species that co-evolved with this vegetation type. Timber production will be a by-product of these management efforts. On the Cherokee NF (Tennessee), restoring historic plant assemblages is the goal of the 9.H management prescription. The intent is to reduce stand density by partial harvesting and manipulate or maintain understory structure. The maintenance will be accomplished using low-intensity fires to create the proper conditions for encouraging oak reproduction and its recruitment into the overstory to provide sustainable woodland, barrens, or savanna ecosystems. The Daniel Boone NF emphasizes animal habitat diversity for prescription area 1.K. It is probable that a single monitoring scheme could be developed that would adequately evaluate the effectiveness of the treatments in attaining the various objectives on the different forests.

The proposed unified monitoring system requires measuring several relevant factors that help detect changes due to management or non-anthropogenic causes including: forest structure and composition, fuel loads, wildlife populations, plant diversity, water quality, and tree regeneration. Knowledge of the stand history, initial vegetation conditions, structure, composition, size distributions, densities, and other overstory and understory features are

essential to interpreting the effectiveness of management practices toward achieving the DFC. Standards of factors to be measured will be developed to characterize initial starting conditions. There may be resistance from those in the field that are most knowledgeable about their specific forest conditions. They may feel that their ecosystems differ from other oak-dominated systems enough that a one-size-fits-all approach will not provide them with the information necessary to evaluate the changes specific to their forest. The program design will need flexibility to allow each NF to add local variables if needed without sacrificing the integrity of the regional monitoring scheme. For example: timber rattlesnakes are a species of interest in the state of Ohio. Therefore, if there is a rattlesnake population within their Historical Forest management area, the Wayne NF may wish to add a monitoring program to evaluate the effects of the management practices on the rattlesnake population. However, the addition of local variables requires more intensive sampling to detect changes at that specific site.

PROCESS

Networking and collection

Successful implementation of a unified monitoring program for the NFs in the Central Hardwoods region requires the cooperation and support of District, Forest, and Regional levels of the Forest Service. This endeavor would require a strategic monitoring plan and implementation standards and guidelines developed through collaboration among the Eastern and Southern Regions and the Northern and Southern Research Stations. The Eastern Region's "Courageous Conservation: A sustainable future, a legacy of restoration" strategic framework, promotes this type of collaboration through its mission as a "Partnership Agency" (Forest Service 2005d) and goal of "Protecting ecosystems across boundaries" (Forest Service 2005b).

Responsibilities and funding sources would need to be negotiated and formalized to ensure continued implementation of the program. Key participants will enter and exit positions whose duties include management, data collection, analysis, and reporting. The importance of maintaining the program on each forest will need to be communicated to these participants by their supervisors. Establishment of field data points must be timely, monumented, and fully recorded. Initial training of field data collection crews and continued quality control must be assured. The Forest

Service has an Inventory and Monitoring Institute and the Northern Station has the Northern Monitoring Program, both of which would be able to assist in organizing this unified monitoring program. Accessing the expertise of these groups should prevent “reinventing the wheel” regarding monitoring protocols. Tying into existing multi-organizational networks, such as The Natural Conservancy’s Fire Learning Network, also would be advantageous.

Once a baseline of common variables and protocols is implemented, a standard database system would allow for pooling and sharing data and results for local and regional analyses. No network exists within the National Forest System for coordinating the data from different forests. Creating this network would be an ongoing process throughout the span of the restoration projects. Once the data are assembled, researchers will examine the response to treatments across a much broader environmental template. Personnel responsible for regional analysis will be identified in advance and can help to ensure monitoring protocols are followed. Analysis of monitoring data following oak restoration activities increases the feasibility for collaboration among NF and Research Station personnel.

The magnitude of the coordination and cooperation necessary to ensure the successful and continued implementation of a unified monitoring program will probably be the most difficult obstacle to overcome. It could start with a commitment by the Eastern and Southern Regional Foresters and the Station Directors of the Northern and Southern Research Stations and formalized in a memorandum of understanding signed by all parties.

The program would have to provide career and cost reduction benefits to be accepted widely. The monitoring would need to provide evaluations of management activities that are a higher quality and a lower cost than if each NF implemented individual monitoring programs. Scientists would buy into the program if the available data was of the quality and intensity necessary for rigorous statistical analysis. FIA data is used by many scientists for regional analyses. The exact locations of the data collection points are not released to protect the privacy of the cooperating land owners, limiting the ability of researchers to incorporate other georeferenced data into their analyses. Scientists would not have to worry about the limitation on data point locations for the monitoring data collected on NFs.

Adaptive Management

Once a unified monitoring program for the Central Hardwoods region is in place and generating data, individual NFs may have the option of adjusting management prescriptions for adherence to DFCs. DFCs may also need to be altered to fit redefined management objectives especially if the management changes can be quantified and analyzed as extensions to the existing practices. In the process of restoration, there may be one set of management activities used to move from the current state to the DFC, then a shift to a different set of management activities to maintain the DFC. Sustaining the desired state may require periods of variation in activity to allow for regeneration and recruitment of the regeneration into the overstory of the savannah or woodland. There is a possibility that the practices will change drastically enough that the management areas being monitored no longer fit into the monitoring program (e.g. no treatment). The overall monitoring program should not suffer greatly if a few NFs withdrew. The withdrawing NFs would still have a functioning monitoring program, albeit with less statistical power.

CONCLUSION

Many NFs in the Central Hardwoods region have revised forest plans that include management to restore and sustain oak-dominated ecosystems using thinning, regeneration harvests, and frequent, low-intensity prescribed fire. Even though the management practices and DFCs may differ in some aspects, it should be possible to design a unified monitoring program for use by those NFs involved in restoration. Most forests will collect more than this baseline data, but establishing a core set of variables and protocols will provide the ability to pool the data for improved evaluations. It would be a great benefit to all involved if this unified monitoring system could be developed before each forest spent the time and money developing its own stand-alone monitoring systems.

A new network or framework for pooling the monitoring data once it has been collected will increase the accessibility and usefulness to scientists. An integral part of this framework will be the personnel assigned to perform the analysis. The analysis will cover many interrelated components of oak-dominated ecosystems and will require several integrated teams to complete.

The greatest obstacle to implementation of such a program is obtaining the cooperation and support of all the various Forest Service staffs involved. Leadership from the Regional Offices and the Research Stations will be critical. Individual NFs and numerous laboratories within the Stations would have vested interests for participating. Essentially, a coordinated monitoring system applied to several forests in a forest type is a new way of doing business that leverages the assets of each entity. It will make monitoring more affordable and meaningful, research more applicable, management more effective, and increase the Forest Service's public accountability. This coordinated effort can begin with restoration of mixed-oak forests, but the framework and relationships could be used to address many management problems too big for any one entity to handle alone.

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Poster Abstracts

Poster Abstract from the 2007 National Silviculture Workshop

Restoring the Ecosystem With Silviculture and Stewardship Contracting: The Nordic Project

Christopher E. Casey and Joseph Torres

ABSTRACT

As a first step in implementing the Green Mountain National Forest's new Land and Resource Management Plan, the Nordic Project is located on the Manchester Ranger District in the Towns of Peru, Winhall, and Landgrove in Bennington County, and in the Town of Londonderry in Windham County. It consists of 2,236 acres featuring silvicultural treatments in support of ecosystem restoration involving regeneration of uncommon species such as aspen and oak, creation of permanent upland openings, conversion of non-native softwood plantations to restore native species, improvements to deer wintering habitat, restoration of aquatic habitat and riparian zones, production of high quality sawtimber and other forest products. Also included is designation of cross country ski trails, scenic vistas, trail relocation and identification and preservation of historic archeological sites. Fourteen timber sales with multi-resource treatments total about 9.5 Million board feet or 15,000 CCF of hardwood and softwood sawtimber and pulpwood. Twelve will be implemented using Stewardship Contracts. It is significant because up front planning and collaboration with citizens, partners and town governments created broad support resulting in no appeals or litigation when regarding it is the largest and most complex project on the forest in memory.

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Variation in Microclimate Associated With Structural Retention Harvests in the Pacific Northwest

Troy D. Heithecker and Charles B. Halpern

ABSTRACT

Green-tree or structural retention is becoming increasingly common as a method of regeneration harvest in the Pacific Northwest. Amelioration of microclimatic stress is assumed to be one mechanism by which overstory retention enhances the survival of forest organisms and the potential for ecosystem recovery following timber harvest. We examined patterns of transmitted light (photosynthetic photon flux density, PPFD), air and soil temperature, and soil moisture in a large-scale green-tree retention experiment in mature, coniferous forests at three locations in western Washington. To quantify the effect of varying levels of retention, we compared daily growing season microclimate and residual forest structure in dispersed retention harvest units representing 0, 15, 40, and 100% of original basal area. Additionally, we quantified spatial gradients in microclimate inside 1-ha forest aggregates, effects of aspect on these gradients, and how microclimatic conditions compared to adjacent harvest areas and larger tracts of undisturbed forest (controls).

Level of retention had a significant and predictable effect on microclimate: PPFD and mean and maximum air and soil temperatures decreased with increasing level of retention. PPFD showed the strongest response, but did not differ between 40 and 100% retention. Mean and maximum air temperatures were significantly greater at 0 and 15% retention than at 100%. In combination, topography, residual forest structure, and understory variables

were good predictors of PPFD and mean and maximum temperatures (R^2 of 0.55-0.85 in multiple regression models), but were poorer predictors of minimum temperatures and soil moisture (R^2 of 0.10-0.51).

Inside 1-ha aggregates, trends in light and temperature were generally similar; values were greatest at the edge, but declined sharply inside the aggregate, with most change occurring within 15 m of the edge. Beyond this distance, light generally declined to levels observed in the controls. Soil temperatures exhibited greater spatial variation and stabilized further from the edge (10-30 m), but air temperatures were generally higher than those in controls. Aspect exerted strong effects on light and temperature, particularly within 15 m of the edge, as did forest structure. Where tree density was low, elevated temperatures penetrated further into the aggregates and aspect-related differences were small.

Our results suggest that when applied in a dispersed pattern, 15% retention, the current minimum standard on federal forests within the range of the northern spotted owl, does little to ameliorate microclimatic conditions relative to traditional clearcut logging. Conversely, our results suggest that 1-ha aggregates are sufficiently large to support areas in which microclimate is comparable to undisturbed forest and suitable, in the short-term, for persistence of forest-dependent species.

Poster Abstract from the 2007 National Silviculture Workshop

Pruning Western White Pine to Reduce White Pine Blister Rust

Holly Kearns, John Schwandt, Brennan Ferguson, and Chris Schnepf

ABSTRACT

White pine blister rust (WPBR) is an exotic rust fungus that has caused dramatic changes in forested ecosystems throughout the West. *Cronartium ribicola*, the causal agent, was imported in 1910 to British Columbia on infected nursery stock from France. All nine native North American species of white pines (five-needled pines) are highly susceptible. WPBR has spread throughout the range of most of these species, which includes some of Western North America's most ecologically and economically important forest ecosystems.

Infection of white pines can only occur by spores produced on an alternate host (usually gooseberries or currants – *Ribes* spp.). Under very moist conditions, these spores land on the needles, germinate, and enter the host through the stomata. The fungus then grows within the vascular tissues into the bark, growing down the branch toward the main stem. This creates an expanding canker that girdles the branch. As the branch beyond the canker is killed, a distinctive red "flag" is created. When the fungus reaches the stem, it girdles the stem resulting in top kill or tree mortality.

WPBR infects white pines only through green needles during prolonged periods of high humidity. Green branches close to the ground are at highest risk of infection because environmental conditions are favorable for the rust (i.e.

shady, cool, and moist). Young trees are especially vulnerable because blister rust infections can reach the stem and girdle small diameter trees very quickly.

Pruning takes advantage of the fact that WPBR is an obligate parasite; the fungus requires a living host and cannot survive on dead plant material. Pruning lower branches not only removes infections before they reach the stem, but also removes the lower needles as infection sites. In studies by the U.S. Forest Service in northern Idaho, pruning the lower 8 – 10 feet of 20-foot tall, 15-year-old, naturally regenerated white pine decreased mortality by nearly 50 percent over the next 35 years compared with unpruned white pines.

Breeding programs to improve the level of WPBR resistance were initiated in the 1950s after individual uninfected white pines were observed in severely infected stands. This improved stock is outperforming natural white pine regeneration but is not immune to WPBR. While pruning does not change the level of genetic resistance in a tree, it can help maintain white pine as a functioning component in mixed forests. This is especially important in mixed conifer stands due to white pine's high tolerance to native root diseases, which are seriously impacting the Douglas-fir and true firs now dominating lands historically dominated by western white pine.

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Pruning can provide many benefits including: improved survival, improved numbers of infection-free trees, preventing stand transition to less desirable species, protecting investment in western white pine stock with genetically improved resistance, and production of higher quality, clear wood. Pruning should be performed when WPBR infection levels threaten management objectives, and typically, only those trees desired as crop trees are pruned. We recommend that pruning be delayed until white pines are old enough (greater than 10 years) to have been exposed to WPBR and be exhibiting symptoms of infections, and the average height of white pines is at least 16 feet to allow the lower 8 feet to be pruned while maintaining at least 50% live crown. All branches (live and dead, infected or not) should be pruned from the bottom 8 feet of the stem, but never prune more than 50% of the live crown. White pines with stem cankers should also not be pruned; these trees cannot be “helped” by pruning. In addition, only trees with branch cankers at least 4 inches from the main stem should be pruned. WPBR cankers less than 4 inches from the stem may already have fungal tissue in the stem and may be considered “lethal.”

Poster Abstract from the 2007 National Silviculture Workshop

Precommercial Thinning in Southeast Alaska and Its Impact on Wood Quality

Eini C. Lowell, Robert A. Monserud, and Alexander Clark III

ABSTRACT

Since the 1900's, about 400,000 acres of timber on Forest Service land and an equal amount on other ownerships in southeast Alaska have been harvested and successfully naturally regenerated. Among several management themes that have been developed for young-growth stands are those that are designed to create stand structures that are beneficial to wildlife, enhance riparian zones, and produce sustainable timber resources. These themes often include the need for pre-commercial thinning (PCT) to accomplish the desired stand features. It is important to understand the effect of such management activities on the quality of wood fiber that may later be removed from these stands and how that quality impacts product opportunities and associated economic potential. The general response of trees to thinning is slower crown recession, increased branch diameter, increased taper and reduced wood specific gravity. This project establishes baseline product recovery information about the volume and quality of lumber products manufactured from young-growth western hemlock (*Tsuga heterophylla* (Raf.) Sarg) and Sitka spruce (*Picea sitchensis* (Bong.) Carr.) from even-aged stands in southeast Alaska. Wood quality is addressed from several perspectives: visual lumber grades, mechanical properties, and anatomical features. The study compares the quality differences between stands that have been pre-commercially thinned and those not thinned, and determines if spacing at time of pre-commercial thinning influences product volume, quality, and value.

Nine young-growth stands that were thinned approximately 20 years ago were identified by Forest Service personnel at the Craig, Thorne Bay, and Petersburg Ranger Districts on the Tongass National Forest. Eight of the sites were located on Prince of Wales Island and the ninth on Mitkof Island. Two sites were on State of Alaska Department of Natural Resources (DNR) land. All stands contained a known precommercial thinning treatment and an adjacent no-treatment control stand. To facilitate the removal of a relatively small number of trees, sites were accessible from existing roads and on terrain where locally available harvesting systems could operate efficiently. Sites were selected to represent as wide a range as possible of stand ages and site classes for both thinned and unthinned stands. Thinning spacings ranged from 10 x 10 ft to 25 x 25 ft.

The study is designed using paired-plots (treatment vs. control) on a series of stands with different levels of treatment (thinning to a specified regular spacing). The sample was stratified by 4-in breast height diameter (dbh) classes with three trees in each class for a total of fifteen trees/species/treatment/site. Because not all dbh classes were present, a total of 461 trees were selected. The range and average dbh of harvested trees were:

	Hemlock		Spruce	
	Control	Thinned	Control	Thinned
no. trees	97	116	127	127
dbh range	9.5" - 24.6"	10.0" - 31.2"	9.5" - 28.2"	10.0" - 32.3"
avg. dbh	14.0"	14.5"	14.2"	16.3"

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During tree selection, it was observed that target spacing and actual spacing at the time of conducting this project appeared to differ. Stand exams were conducted to determine actual spacing at time of sampling.

Trees were harvested and long logs transported to the Ketchikan Wood Technology Center in Ketchikan, Alaska. Woods length logs were graded and scaled (both cubic and Scribner) by the Pacific Rim Scaling Bureau, bucked into mill length logs, and re-scaled. A disk was cut from the large end of the butt log and from the end of every subsequent short log. Radial strips $\_ \times \_$ inches square were cut from bark to bark through the pith of each disk, dried, glued to core holders and sawn into 2 mm thick strips. Annual ring growth, ring specific gravity and proportion of latewood were determined for each radial strip using a scanning x-ray densitometer. Logs were sawn into dimension lumber (2 x 4 in and 2 x 6 in) using a portable sawmill. The lumber was kiln dried, planed, and graded according to National Lumber Standard Grading Rules. Grade controlling defect for each piece of lumber was also recorded. Every piece of lumber was non-destructively tested (transverse vibration according to ASTM standard D6874, 2003) for Modulus of Elasticity (MOE). A 20 pct stratified random sub-sample of trees was selected for further testing of mechanical properties. All lumber from these trees was destructively tested (static bending of full size lumber according to ASTM standard D6874, 2003) to determine modulus of elasticity (MOE) or stiffness and modulus of rupture (MOR) or strength. Complete tracking of each piece of lumber allows for reconstruction of recoverable volume and value from individual logs and trees.

The hemlock and spruce trees growing on the thinned plots responded positively to the thinning treatments. Thinning reduced the over crowding and there was sufficient moisture and nutrients to maintain wood quality. Thus, spruce and hemlock stands on the Tongass National Forest could be pre-commercial thinned to improve wildlife habitat, and enhance riparian zones with no negative impacts on sustainable timber resources.

Poster Abstract from the 2007 National Silviculture Workshop

3P Remote Sensing (3PRS): An Efficient Way to Estimate Stand Volume in Low Value/Biomass Timber

Matt Oberle, Dave Johns, Ken Cormier, Gary Boyack, and Andrew Sánchez Meador

ABSTRACT

3P Remote Sensing (3PRS), a two-stage sampling method using large-scale aerial photography and traditional 3P (probability proportional to prediction) sampling techniques may be an efficient and cost-effective way to estimate the quantity of low value timber products. The first stage estimates relative volume per acre for each sample plot based on visual clues from aerial photos. Estimates are on a normalized scale from 0 to 100, called the RVI (Relative Volume Index), where 0 is no volume and 100 is the highest volume in the population. In the second stage, a subset of the first stage sample plots is chosen for field measurement using 3P selection. Plot centers for each measured plot are located on the ground, a fixed area plot is established and every tree to be harvested on the fixed plot is measured. Predicted to measured volume ratios are then calculated and used to adjust the total estimates.

Tools and technology used for this process, such as stereo photos, large-scale ortho-photography and computer software are normally available to forestry units. While

the 3PRS method can provide an estimate of total biomass volume and a measure of sampling error, it is not able to accurately furnish other information such as individual tree characteristics and thus is not suited for all cases. The 3PRS method requires a uniform silvicultural prescription, a single value for the timber product, a familiarity with the timber's volume variability via a walkthrough or guidance from a local forester, consistent volume (RVI) estimates and good ground truthing during the second stage for the measured / predicted ratio.

Further testing of this method will be conducted in the Estes Valley Stewardship / Service Contract Pilot Project on the Arapaho-Roosevelt National Forest starting in 2007.

Overall, 3PRS may serve as a cost effective tool for estimating the quantity of low-value biomass timber.

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Poster Abstract from the 2007 National Silviculture Workshop

Fiber, Feathers and Friends—An Integrated Approach to Managing Jack Pine to Benefit Kirtland's Warbler, Maintain Yields and Involve Partners

Kirk Piehler and Jean Perkins

ABSTRACT

Jack pine (*Pinus banksiana*) is a common fiber producing tree in the Great Lakes states. Dense stands of young jack pine provide optimal nesting habitat for Kirtland's warbler (*Dendroica kirtlandii*) or "KW", a federally listed species. The species breeding range is primarily Michigan's northern Lower and eastern Upper Peninsulas. The Hiawatha National Forest's 2006 Forest Plan contains management direction, including goals for average annual acres of jack pine regeneration, designed to promote recovery of KW. On the Hiawatha National Forest (HNF), located in the eastern Upper Peninsula, young jack pine stands about 6-16 years of age (about 5-15 feet tall) are considered suitable for nesting. Jack pine is adapted to well drained and sandy soils (i.e. "barrens"), conditions that have been present since the retreat of the Wisconsin ice sheet about 14,000 years ago. Historically, vegetation on the jack pine barrens was maintained by natural wildfires that occurred frequently throughout the region. With the arrival of wildfire suppression, an emphasis on forest management practices to regenerate jack pine is required for the continued recovery and viability of the Kirtland's warbler.

Habitat standards suggest that jack pine stands less than 80 acres in size are seldom occupied. The highest potential for nesting will occur in stands or complexes greater than 1000 acres with an average stocking density of about 1,100 trees per acre or more, including small non-forest inclusions (approximately 25 percent open area per acre),

or approximately 5'x 6' spacing or less. Due to the open-land requirement, the actual tree density would be about 1,450 trees per acre in habitat outside of the open areas. Natural regeneration or planting can be used to regenerate jack pine for KW on the HNF.

Reforestation of jack pine for KW on the HNF emphasizes natural regeneration, which is less expensive than full planting. Site preparation consists of crisscross passes with a rollerchopper to crush jack pine slash remaining after a mature jack pine stand has been clearcut. Spiked anchor chains dragged behind the rollerchopper scarify the soil at the same time. Jack pine is extremely intolerant of shade, and cones are serotinous. Rollerchopping brings the cones on the slash close enough to the ground that radiant heat from the sun will open them. Site preparation takes place as soon as possible after harvest so seed viability will be at maximum, and so the soil will be scarified before the jack pine cones open (generally in July following harvest), so that seeds falling from cones in the logging slash germinate on mineral soil. Natural regeneration must sometimes be supplemented with additional seeding or planting to reach the stocking density needed to meet KW nesting requirements. Stocking density must also be kept below densities that might reduce later marketability. After stands grow out of the size that provides KW nesting habitat, they are managed conventionally for fiber production.

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Partnerships appear to be a promising means for funding KW habitat work if either full or supplemental planting of jack pine for KW is prescribed. The Hiawatha National Forest was the only Forest in Region 9 to receive a grant from the National Arbor Day Foundation in 2006. This grant is being used in the spring of 2007 to create and/or restore habitat for Kirtland's warbler through supplemental planting in areas where natural regeneration of jack pine alone did not result in sufficiently dense stocking to meet KW habitat needs. Stewardship contracting is another emerging tool that may be used to facilitate KW habitat creation or restoration.

Throughfall Monitoring of Old Growth, Second Growth, and Cleared Vegetation Plots on Prince of Wales Island, Alaska

Katherine M. Prussian

ABSTRACT

The density of forest canopy affects the amount of rain reaching the forest floor in forested environments of Southeast Alaska. Automated tipping bucket rain gages were used to monitor throughfall in Old Growth (OG), Second Growth (SG), and Cleared (CC) vegetation sites on Prince of Wales Island, Alaska, as an indicator of varying forest conditions. Data collected during 2004 and 2005 included 24 storms ranging from 0.4 to 7 inches of gross rainfall. Data following 2006 included 11 storms ranging from .3 to 2.2 inches of gross rainfall. This monitoring is an effort to determine the affect, if any, that forest management could have on throughfall, and furthermore, lend information to forest management effects on the water balance within a watershed. These results suggest that gross rainfall in cleared vegetation sites is greater than throughfall occurring in both old growth and second growth stands. Further, interception by a dense second growth canopy accounted for an average of 35% at Site A, and 65% at Site B, of the gross precipitation. Interception of an old growth canopy accounted for an average 51% at Site A, and 42% at Site B of the gross rainfall. Similar results were found in an earlier study by J.H. Patric where interception accounted for 25% of the gross annual precipitation in a forested areas near Juneau, Alaska (Patric, J.H., 1966). Further rainfall monitoring, and additional site sampling, will provide greater detail relative to canopy density, elevation, aspect, antecedent conditions, and storm intensity.

Direct Seeding of Jack Pine on the Hiawatha National Forest

Al Saberniak and Paul Berrang

ABSTRACT

The recent discovery of the federally listed endangered Kirtland's warbler (*Dendroica kirtlandii*) on the Hiawatha National Forest (NF) in Michigan's Upper Peninsula has increased interest in regenerating dense stands of jack pine (*Pinus banksiana* Lamb.) on outwash sands. The Hiawatha NF has frequently used direct seeding of jack pine to augment natural regeneration on these sites, but we don't know if this is effective or even necessary.

Direct seeding of jack pine has been very effective when the seed is applied directly to bare mineral soil after the site had been burned and deeply scarified with an Athens disk. The Hiawatha NF uses a less intensive site preparation and applies the seed on top of the snow in late spring. We wanted to evaluate the effectiveness of direct seeding under these conditions.

Six stands were selected that had been cut in either 2001 or 2002 with the intention of regenerating jack pine. The number of trees left uncut varied considerably among stands. Each stand was prepared for seeding by double rollerchopping and chaining in 2003. Either 20 or 30 plot centers were established along transects in each stand at places where site preparation was judged to be adequate. Each plot was randomly assigned to either the seeded or non-seeded treatment with a coin toss. The number of cones on the ground at each plot was qualitatively estimated. The residual overstory around each plot was measured as well. Seed was applied at the rate of 4 oz/ac to the ap-

propriate plots in spring of 2004 with a seeder attached to the back of a snowmobile. A 250th ac circular plot was established around each plot center and with the help of numerous employees of the Hiawatha NF, the number of pine seedlings was counted in 2004, 2005, and 2006. Seedlings were assumed to be jack pine unless they could be otherwise identified.

Full stocking would be the equivalent of either 3 or 4 seedlings/ac of acceptable growing stock depending on whether the site was being regenerated for Kirtland's warbler habitat or for other objectives. Across all six stands in 2006 there was an average of 10 seedlings per unseeded plot and 17 seedlings per seeded plot, a statistically significant difference. Increasing the number of seedlings on plots that are already overstocked only makes sense if large numbers of seedlings are expected to die. We do expect many of the smaller seedlings to die, but not enough to decrease stocking below target levels.

We compared seeded and unseeded plots at each of the six sites (Table 1). In 2006 seeding increased the number of seedlings at two of the six stands. The difference in the effectiveness of direct seeding among stands appeared to be related to the cone crop. Plots that had a low number of cones on the ground at the time of seeding had only a third as many seedlings as plots that had a high number of cones. Foresters may be able to predict the need for seeding by evaluating the cone crop.

Some plots had regeneration of red pine (*Pinus resinosa* Ait.) as well as jack pine. This was related to the residual red pine overstory, plots with red pine basal area had four times as many red pine seedlings as plots without red pine basal area. While a mixture of red and jack pine is compatible with some management objectives it is not desirable for Kirtland's warbler habitat.

Table 1 – Average number of live jack pine seedlings per 250th acre plot three growing seasons after half the plots were seeded. Pairs of numbers in the same row followed by the same letter are not significantly different ($P < 0.05$).

District name	Stand/ compartment	Number Live Jack Pine Seedlings per Plot	
		Seeded	Not seeded
Munising	81/7	23 a	25 a
Munising	81/19	14 a	11 a
Sault Ste. Marie	23/2	16 a	7 b
Sault Ste. Marie	23/16	10 a	4 a
Sault Ste. Marie	24/32	10 a	7 a
Sault Ste. Marie	24/34	25 a	12 b

Poster Abstract from the 2007 National Silviculture Workshop

Spruce Aphid Infestation of Sitka Spruce in Southeast Alaska

Mark Schultz, Dustin Wittwer, Paul Hennon, Melinda Lamb, Andris Eglitis, Nellie Olsen, Ann Lynch, Chris Fettig, Robert Borys, Chris Dabney, Roger Burnside, and Neil Kidd

ABSTRACT

The Problem

(Andris Eglitis)

Sitka spruce (*Picea sitchensis* (Bong.) Carr.), historically has been subjected to attack by the spruce aphid, *Elatobium abietinum* (Walker) (Homoptera: Aphididae). Spruce aphid can cause severe needle drop and death of the tree after several defoliation events. Spruce aphid outbreaks are usually preceded by mild winters, and normally last for a short time, perhaps two or three years. Since 1975 the outbreak has been more or less continuous. Aphid damage may become apparent in March or April before the new spruce growth begins. Populations continue to increase until early summer, when sap nutrition decreases. By midsummer the populations may reach a low point because reproduction rate drops with decreased nutrition, migration by winged females, and increased parasitism. In the fall, the aphids may increase in number and infest the current year needles. This second peak in September and October is usually not as damaging as the spring population, but significant defoliation may still occur if the autumn is a mild one.

Distribution

(Dustin Wittwer, Paul Hennon, and Nellie Olsen)

Aspect and maritime influence at the beach fringe have a great deal to do with spruce aphid survival. Southern aspects that are sheltered and near the western edge of the

archipelago of southeast Alaska islands are many degrees warmer than elsewhere. Warmer temperatures, by 5-10 °C, occurred as much or more than two weeks longer in the fall and two weeks sooner in the spring in Sitka and Craig than in Juneau, AK.

Climate Change

(Ann Lynch)

Since 1940, mean annual midpoint temperatures (the midpoint is halfway between the minimum and maximum temperatures) at Sitka, AK increased 2.0°C. The minimum winter temperature has increased by 6°C. The number of winter days with below-freezing temperatures has decreased by 20 days. The date of the last spring frost has retreated 15 to 20 days in Sitka.

Chemical Control

(Chris Fettig, Roger Burnside, Robert Borys, Chris Dabney, Melinda Lamb)

The application of Acecaps significantly reduced the density of aphids on Sitka spruce by 92.4% (June 2005) and 100% (September 2006). No other significant differences were observed among treatment (Kioritz or Arborjet) means during these two sample periods. Acecaps were the most effective means of control but during most evaluations relatively few aphids were encountered overall.

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Vertical and horizontal digital photographs were taken in April and September 2005 and 2006 to compare the changes in needle density on treated and untreated trees. Using the HemiView software, developed by Delta-T Devices, control trees lost on the average one and one-half percent of their needle cover each in 2005 and 2006 and, this includes the gain in needle cover by new growth after leaf-out in June. The Acecap treated trees had an average gain in needle cover of two percent.

Future Monitoring

(R. Burnside)

Little is known about where spruce aphid resides during the summer hiatus and during cold periods or much about how populations recover in the spring. Using fluorescent dyes and netted branches and trees we hope to determine some site specific information about spruce aphid biology.

Climate Modeling

(Neil Kidd and A. Lynch)

Winter snowfall and low winter temperature account for over half of the variability in the spruce aphid outbreak frequency.

English equivalents

When you know:	Multiply by:	To find:
Millimeters (mm)	0.0394	Inches
Centimeters (cm)	0.394	Inches
Meters (m)	3.28	Feet
Kilometers per hour (km/hr)	0.621	Miles per hour
Hectares (ha)	2.47	Acres
Kilograms (kg)	0.0011	Tons
Megagrams per hectare (Mg/ha)	0.446	Tons per acre
Gram-force per square centimeter (g/cm ²)	0.014	Pounds per square inch
Kilograms per cubic meter (kg/m ³)	0.0624	Pounds per cubic foot
Square meters per hectare (m ² /ha)	4.37	Square feet per acre

