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Integrated Management of Carbon Sequestration and Biomass Utilization Opportunities in a Changing Climate:

Proceedings of the 2009 National Silviculture Workshop; 2009 June 15-18; Boise, ID

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Abstract

Forests are important for carbon sequestration and how they are manipulated either through natural or human induced disturbances can have an effect on CO_2 emissions and carbon sequestration. The 2009 National Silviculture Workshop presented scientific information and management strategies to meet a variety of objectives while simultaneously addressing carbon sequestration and biomass utilization. The focus areas were: the role of climate change in science and management; silvicultural methods to address carbon sequestration and biomass utilization; alternative silvicultural strategies to address the growth and development of forests; and current applications of computer simulation models or modeling techniques designed to provide decision support.

Keywords: silviculture, carbon sequestration, climate change, forest management

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Technical Editors

Theresa B. Jain
Russell T. Graham
Jonathan Sandquist

Contents

Introduction 1
Theresa B. Jain

Part 1: Climate Change in Science and Management 3

Remarks 5
Cecil D. Andrus

Ecosystem Services and Climate Change: Understanding the Differences and Identifying Opportunities for Forest Carbon 9
Robert L. Deal, Crystal Raymond, David L. Peterson, and Cindy Glick

Integrating Climate Change Considerations into Forest Management Tools and Training 27
Linda M. Nagel, Christopher W. Swanston, and Maria K. Janowiak

Part 2: Carbon Sequestration and Biomass Utilization 37

Carbon Concentrations and Carbon Pool Distributions in Dry, Moist, and Cold Mid-Aged Forests of the Rocky Mountains 39
Theresa B. Jain, Russell T. Graham, and David Adams

Carbon Benefits from Fuel Treatments 61
Jim Cathcart, Alan A. Ager, Andrew McMahan, Mark Finney, and Brian Watt

Forest Biomass and Tree Planting for Fossil Fuel Offsets in the Colorado Front Range 81
Mike A. Battaglia, Kellen Nelson, Dan Kashian, and Michael G. Ryan

To Manage or not to Manage: The Role of Silviculture in Sequestering Carbon in the Specter of Climate Change 95
Jianwei Zhang, Robert F. Powers, and Carl N. Skinner

Estimating Long-Term Carbon Sequestration Patterns in Even- and Uneven-Aged Southern Pine Stands 111
Don C. Bragg and James M. Guldin

The Role of Forests in Energy and Climate Change - Integrating Objectives. 125
Dave Atkins

The Forest—Bioenergy—Carbon Connection. 129
Jay O'Laughlin

Biomass Utilization Opportunities to Achieve Diverse Silvicultural Goals 135
Barry Wynsma and Christopher R. Keyes

Emissions, Energy Return and Economics from Utilizing Forest Residues for Thermal Energy Compared to Onsite Pile Burning 145
Greg Jones, Dan Loeffler, Edward Butler, Woodam Chung, and Susan Hummel

Can Portable Pyrolysis Units Make Biomass Utilization Affordable While Using Bio-Char to Enhance Soil Productivity and Sequester Carbon? 159
Mark Coleman, Deborah Page-Dumroese, Jim Archuleta, Phil Badger, Woodum Chung, Tyron Venn, Dan Loeffler, Greg Jones, and Kristin McElligott

Ethanol Production from Woody Biomass: Silvicultural Opportunities for Suppressed Western Conifers 169
Andrew Youngblood, Junyong Zhu, and C. Tim Scott

Part 3: Alternative Silvicultural Strategies 185

Estimating Site Index from Tree Species Composition in Mixed Stands of Upland Eastern Hardwoods: Should Shrubs be Included? 187
W. Henry McNab

Consistent Definition and Application of Reineke's Stand Density Index in Silviculture and Stand Projection 199
John D. Shaw and James N. Long

The role of strategic forest inventories in aiding land management decision-making: Examples from the U.S. Forest Inventory and Analysis Program 211
W. Keith Moser, Renate Bush, John D. Shaw, Mark H. Hansen, and Mark D. Nelson

Variable-Density Thinning for Parks and Reserves: An Experimental Case Study at Humboldt Redwoods State Park, California 227
Christopher R. Keyes, Thomas E. Perry, and Jesse F. Plummer

Site Quality Changes in Response to Slash Retention and Prescribed Fire in Thinned Ponderosa Pine Forests 239
Matt Busse

Stand Density Guides for Predicting Growth of Forest Tress of Southwest Idaho 251
Douglas D. Basford, John Sloan, and Joy Roberts

Built for the Future: New Directions in Silviculture Research and Demonstration at Montana's Lubrecht Experimental Forest 267
Christopher R. Keyes and Thomas E. Perry

Part 4: Decision Support Tools 271

Addressing Climate Change in the Forest Vegetation Simulator to Assess Impacts on Landscape Forest Dynamics 273
Nicholas L. Crookston, Gerald E. Rehfeldt, Gary E. Dixon, and Aaron R. Weiskittel

Calibration of State and Transition Models with FVS 275
Melinda Moeur and Don Vandendriesche

FVS Out of the Box – Assembly Required 289
Don Vandendriesche

An Empirical Approach for Estimating Natural Regeneration for the Forest Vegetation Simulator 307
Don Vandendriesche

Evaluating the Ecological Sustainability of a Pinyon-Juniper Grassland Ecosystem in Northern Arizona 321
Reuben Weisz, Jack Triepke, Don Vandendriesche, Mike Manthei, Jim Youtz, Jerry Simon, and Wayne Robbie

Part 5: Summary 337

Summary of the 2009 National Silviculture Workshop 339
James M. Guldin

Introduction

Theresa B. Jain[1]

Forests can play a role in carbon sequestration and mitigating CO_2 emissions. However, what course of action needed to meet issues concerning carbon management and other ecosystem services for specific situations is not always clear. The National Silviculture Workshop, held in Boise, Idaho on June 15-18, 2009, focused on scientific information and management opportunities and strategies applicable for meeting a variety of objectives, while simultaneously addressing carbon sequestration and biomass utilization. The symposium and subsequent proceedings covered four general areas of interest: the role of climate change in science and management; silvicultural methods to address carbon sequestration and biomass utilization; alternative silvicultural strategies to address the growth and development of forests; and current applications of computer simulation models or modeling techniques designed to provide decision support.

Climate change in science and management was introduced by emeritus Governor of Idaho and current director of the Andrus Center for Public Policy, Cecil Andrus. He emphasized that society depends on forest scientists and managers in silviculture to provide information, including the trade-offs, benefits, and alternatives, involved in making sound public policy. The challenge facing managers is finding the ways for forests to meet energy, economic development, and environmental quality in a changing climate. Bob Deal and others define and provide alternatives that integrate ecosystem services and climate change. Linda Nagel and others discuss methods and techniques that provide state-of-the-art knowledge about climate change and potential impacts to facilitate development of silvicultural objectives and prescriptions that are flexible and enhance ecosystem resistance and resilience.

Forest management is linked to the manipulation of carbon through either carbon sequestration or the use of biomass for other resource needs. For example, there is a trade-off between management activities that decrease the consumption of biomass from a wildfire and meeting other objectives (for example, timber production) and the amount of biomass a site can hold for carbon sequestration. Theresa Jain and others provide estimates of carbon amounts in different forest components. Jim Cathart and others, Mike Battaglia and others, and Jianwei Zhang and others discuss the effectiveness of fuel treatments in avoiding carbon emissions from wildfires. Don Bragg and James Guldin discuss the tradeoffs of thinning in southern forests of the United States in relation to carbon sequestration. A variety of alternatives and techniques are presented, including the application of logging systems (Barry Wynsma and Christopher Keyes) and innovative ideas for biomass utilization (Dave Atkins, Jay O'Laughlin). Greg Jones and others compare total emissions from delivering and burning forest residue biomass for thermal energy to onsite disposal by pile-burning and using fossil fuels to produce the equivalent amount of useable energy. Two other alternatives for using biomass were presented: Mark Coleman and others discuss the economic feasibility of producing biochar and its role in soil productivity and Andrew Youngblood and others present the use of biomass for ethanol production.

Ultimately, any type of silvicultural activity has an effect on carbon sequestration and the production of biomass. Therefore, we encouraged discussions that provide alternative silvicultural strategies for insuring the growth and

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[1] USDA Forest Service, Rocky Mountain Research Station

development of forests that meet a variety of objectives. Because quantification of forest metrics continues to be a challenge, alternative approaches were presented. Henry McNab evaluates the inclusion of shrubs for estimating site index in oak dominated forests. John Shaw and James Long discuss the limitations and confusion concerning the application of Reineke's Stand Density Index and provide guidelines to avoid the misapplication of this metric. Moser and others discuss how FIA data can be used for Forest Plan revision, monitoring conditions and trends at mid- to broad spatial scales over time, and to set the context for proposed projects. Two manuscripts focus on implementation of silviculture treatments: Christopher Keyes and others evaluate the effectiveness and challenges involved with accomplishing variable density prescriptions and Matt Busse discusses the tradeoff between maintaining long-term soil productivity and removing biomass, an issue that creates much discussion among managers. Although silviculture can influence the role of disturbances at multiple spatial scales, it is a discipline that focuses on site specific management activities. Douglas Basford and others present a method, adapted from a model developed for tree species on the Salmon National Forest, for estimating stand growth from stand density and average diameter in stands of pure and mixed species in southwest Idaho. Christopher Keys and Thomas Perry highlight the role of experimental forests, using Lubrecht as an example, to implement and demonstrate alternative silvicultural strategies.

Associated with climate change is an element of uncertainty. Typically uncertainty is addressed through the development and application of models which are used to evaluate a variety of scenarios that are applicable in decision support. Nicholas Crookston and others introduce a modification of the Forest Vegetation Simulator (FVS) to account for climate change. Melinda Moeur and Don Vandendriesche describe how FVS is being used with regional inventory data to empirically derive state transition models such as residence times in states, pathways between states, and transition probabilities between states, and to link these outputs to vegetation states. Don Vandendriesche provides a set of guidelines for using FVS and also discusses the process for estimating natural regeneration. Reuben Weisz and others present a methodology for developing a state-and-transition model with the Vegetation Dynamics Development Tool (VDDT), using outputs from FVS.

James Guldin closes this proceedings by summarizing the workshop, including the field tour and discussions. He concludes that there was a consensus in the meeting that climate change will be the defining issue for this generation of resource managers and that silviculture will play a key role.

The content of this paper reflects the views of the authors, who are responsible for the facts and accuracy of the information presented herein.

Part 1: Climate Change in Science and Management

Remarks

Cecil D. Andrus[1]

I want to thank Harv Forsgren, the Intermountain Regional Forester, for his invitation to me to offer a few remarks to this very impressive gathering. I was raised in western Oregon, which provided the genesis for my love of forests. My beginnings of government service started over 50 years ago by serving in the Korean War and I subsequently became a political accident. As many of you know, I was Idaho's only 4-term governor and served as Secretary of the Interior in the Carter Administration, which was only a portion of my long career in public service. I am impressed with the agenda and really impressed that you would invite a former lumberjack to keynote a conference on science. For those of you from out of town—welcome. Spend your per-diem. We need the revenue.

When you have been around politics and government as long as I have—and in my case it has been a pretty steady preoccupation for close to 50 years—you sometimes think that no big problem ever gets fully resolved. If we are lucky, we make a little progress at the edges and maybe things get a little better, but it is pretty rare that we ever say: "Well, we got that problem licked."

It reminds me of the old story about the inmates at a prison who had all been locked up for many years. They had been locked up for so long that they had heard every joke from everyone on the inside a thousand times. So, to streamline the joke telling, they just started numbering the jokes. One of the inmates would say, for example, I feel like telling old number 12 and everyone would laugh. Finally, one day one of the newer inmates who had only been in the joint for a few years decided he would try a joke and he said "how about number 7." All the other inmates just stared at him and no one was laughing. Finally, the joke teller said, "What's wrong—no one is laughing." One of the real old timers spoke up and said: "You didn't tell it right."

I do sometimes feel we could just assign a number to forest health, another number to climate change, a different number to multiple use, and so forth, and I could just recite the numbers and sit down. The policy makers make endless speeches on these kinds of issues, but it frequently seems we make little real progress in solving problems. I am hoping that with a new administration now in place we will start solving some problems and I do have some optimism. I am encouraged by the stated determination of the new Obama Administration to sweep aside what I can only call an anti-science bias that existed in the previous administration and begin once again to rely on the kind of science and analysis that all of you produce. We simply must use the best science, regardless of how it may test the popular will or the politically easy position, if we are to make progress on terribly important policies related to climate change and utilization of biomass, among many others.

So, part of my message today is to encourage you to do all you can to recapture a central role for science—in your case silviculture—in the making of public policy. You are the experts and the rest of us depend on you to give us the data and the facts about trade-offs and options in order to make sound, smart public policy. Unfortunately, for too many years, the voices of the experts were drowned out by the political operators who think every problem is a partisan problem that can be solved by making the loudest argument. I truly hope the new administration values science. It is only common sense to do so.

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[1] *Directed the Andrus Center for Public Policy of Boise State University, Boise, Idaho since 1995.*

As some of you know, when I left the Idaho Governor's office in 1995, we established the Andrus Center for Public Policy at Boise State University. We have convened a number of conferences over the last 10 years or so that often deal with western natural resource issues. Back in 2004, we did a conference to commemorate the 100th anniversary of the Forest Service. We talked a great deal about the management challenges the Forest Service has always faced and we reflected on the job going forward. As I was thinking about this gathering, I went back and looked at the report from that conference and I was struck by one section that really jumped out. It was a remark about climate change, and I quote the prominent climatologist, Dr. Tim Brown of the Desert Research Institute when he noted "there is scientific consensus that we are in the midst of climate change. For the West in the 21st Century, this means a warmer winter, less snow pack but more precipitation, and warmer summers. Drought will continue, comparable to the period of the 1930s or 1950s." In Dr. Brown's understated summary, he said, "This will be the challenge for management."

I think that is about right and that makes your work all the more important. Harv Forsgren's letter inviting me to offer some remarks today asked me to address how the Forest Service—and each of you—meets the challenge of managing the national forests for energy, economic development and environmental quality "in a changing climate." That will be a tall order and as Dr. Brown suggested in 2004, "a challenge for Forest Service management." I don't need to tell you that all three of these purposes (energy, economic development, and environmental quality) are very important. They can also be contradictory and competing. Some groups will push for one outcome at the expense of the other two. Finding the right balance is the real challenge because I doubt whether you can maximize all three objectives at the same time.

The facts are that the American public wants—and the politicians want to give—everything all the time. We do not like making tradeoffs and we do not like having to choose. For years—maybe forever—the Forest Service has been caught in this struggle. As we continue the debate over just what exactly the purposes of the national forests are, how do we find something approaching agreement around that question? One Idahoan would tell you the national forests exist to produce wood fiber. Another would tell you they exist to provide hunting and fishing opportunities. Another would tell you the forests help drive the economy of the state and particularly rural communities. This Idahoan would tell you that there is a measure of large truth in each of those answers. So what you do, in my opinion, and what policy makers must do, is find the delicate balance that creates an equilibrium that gives the American public the opportunity to have it all. That is, increased energy from biomass, a stronger economy, and the hunting, fishing and outdoor recreation that we so enjoy in Idaho and the remainder of the West.

Let me make a couple of comments about the debate over climate change, and in addition, offer some suggestions for you and the leadership of the Forest Service to help the policy makers sort through this debate. I suspect we have some consensus around this issue that climate is changing. If some of the change is the result of a "natural cycle" then the predictable human reaction will want to have us focus on how we can keep forests the way they are because we value them this way, even if we must go up against "Mother Nature." On the other hand, if it becomes more certain—as I believe it will—that human activity is at the center of climate change, it may become easier to attempt to manage forests to "keep them the way we want them." The real point is that climate change is likely to make it more difficult for you to manage for the three purposes—energy, economic development, and the environment. If climate change really has become your new overarching management issue then, I believe, the Forest Service should

be extremely transparent about how managing for climate change will affect the three big objectives.

A couple of thoughts on the biomass to energy issue. According to the Department of Energy, biomass is right now providing about 3 percent of all energy consumed in the United States and nearly 50 percent of all the energy from renewable sources. Biomass supplies more BTU's than hydroelectric energy. The latest number in Idaho, I'm told, is that wood bioenergy accounts for about 4 to 5 percent of all the energy consumed. Electric generation from biomass (not counting municipal solidwaste) represents about 11 percent of all generation from renewable sources in the United States. Biomass supplies almost six times the energy of geothermal, solar, and wind energy sources combined and worldwide biomass meets about 14 percent of the world's energy needs. But, no one as far as I know has a good handle on how much potential really exists in the West or the rest of the nation. It is not clear to me—maybe you'll get it figured out at this conference—how, once we figure out the potential, we will maximize the opportunity to generate more energy while still managing the national forests for wood products and recreation.

We need to think about the infrastructure needed to get the potential energy delivered. In addition, we probably should not be rushing to construct a lot of infrastructure until we know the amount of biomass realistically available from national forest lands and better understand what is involved with making it available. Because Forest Service silviculturists play a critical role in providing this kind of information, realistic plans can be pursued for utilizing forest biomass to produce energy. There appear to be two primary challenges. One is cost, and transportation is the largest component of cost. The second challenge is supply. Biofuels facilities must have a consistent supply of material, day in and day out, for the life of a project and this could mean 20 to 25 years. That steady supply issue presents a real challenge for you and the national forests. And the data about the amount of material available should drive decisions for how large an energy facility should be built, not the other way around. On this point, the developers of these big biomass plans need to hear from the technical experts about how much biomass is realistically available and how much it costs to get it out of the woods. I would encourage you to continue to play the role of honest, science-based reality checkers and resist the pressure that sometimes occurs to come up with research that justifies a political position. Don't get me wrong. I think we need to improve biomass utilization—and quickly. I do not think we will be able to move as quickly as we might like, because we have to do all this sorting out of priorities first and we have barely begun that effort. For example, we may need to reconsider a whole range of issues related to how we manage the national forests. We know that a lack of harvest can have serious adverse consequences. I've had a preview of some research that will soon be public that says, among other things, that tree mortality in Idaho forests due to overcrowding and drought is at the highest level since we started keeping records nearly 60 years ago. Mortality in the forests is now removing more timber on an annual basis than harvest and the accumulation of dead wood has now reached an all-time high. The overwhelming majority of the dead wood is in the national forests where it contributes the fuel to feed big fires that in turn have major cost and environmental impacts. Here is one other challenge for you in the Forest Service: I think it is very important that you be as clear as possible about the state of the science on carbon sequestration, specifically growing trees to hold carbon. As far as I can tell, we have been talking about this for 15 or more years and the state of the science—at least in terms of public understanding—has not advanced much. I would like to see more attention—quickly—on just what various sequestration

strategies really mean to a more effective policy aimed at controlling carbon. We have limped along in the United States for 25 years or more without a coherent national energy strategy. We continue to import way too much energy and clearly we can't drill our way to energy independence. We need to attack the issue in a comprehensive way and make some tough decisions. We haven't, until recently, been aggressive enough about fuel efficiency standards and we haven't placed nearly enough attention on better mass transit. We cannot summon the political will to deal with nuclear waste, so that technology remains largely on the shelf. Everyone agrees now, I think, that we must aggressively pursue alternatives, but doing so will not be fast or easy. Nevertheless, we must get on with it. The best policy, whether it's related to biomass, climate change, or carbon, will come about when we really utilize the science that you offer to give us the best, most honest information about which way we should move. I'm optimistic, as I've said, that we have some new leadership and new commitment to solving some of these old and perplexing problems. I hope if Harv were to invite me back in a couple of years, I would have the opportunity to thank you all for moving the ball on these issues. Good luck. We're depending on you.

The content of this paper reflects the views of the authors, who are responsible for the facts and accuracy of the information presented herein.

Ecosystem Services and Climate Change: Understanding the Differences and Identifying Opportunities for Forest Carbon

Robert L. Deal[1], Crystal Raymond[2], David L. Peterson[3], and Cindy Glick[4]

Abstract—There are a number of misunderstandings about "ecosystem services" and "climate change" and these terms are often used incorrectly to describe different concepts. These concepts address different issues and objectives but have some important integrating themes relating to carbon and carbon sequestration. In this paper, we provide definitions and distinctions between ecosystem services and climate change. We describe some of the emerging markets for ecosystem services including carbon, water, wetland mitigation and species conservation banking and some of the national initiatives to address climate change with carbon markets. We also discuss some of the potential effects of climate change on forest ecosystems in the USA. Finally, we develop the concept of using an ecosystem services marketplace and the potential for mitigating climate change specifically focusing on the emerging markets for carbon. This integration of ecosystem services and climate change may provide some new opportunities for forest landowners and managers to enhance forest stewardship in addition to reducing greenhouse gas emissions through forest carbon sequestration.

Keywords: ecosystem services, climate change, carbon and forestry, carbon markets.

Introduction

Ecosystem Services and Climate Change

Ecosystem services are, in the broadest sense, the contributions or benefits that come from the natural environment. The Millenium Ecosystem Assessment (MEA 2005) provided a simple definition of *ecosystem services* as "the benefits people obtain from ecosystems." There is some disagreement among ecologists and economists on whether these benefits should include only the direct benefits for people or also include some of the supporting functions of ecosystems. The ecologist viewpoint (Daily 1997) focuses on the function and process of the ecosystem, and these services may include water purification, climate regulation, and biodiversity (figs. 1 and 2). The economist viewpoint generally focuses on ecosystem services that are components of nature, directly enjoyed, consumed, or used to yield human well-being (Boyd 2004; Boyd and Banzhaf 2006; Kroeger and Casey 2007). For our paper, we use the broad and previously defined typology for ecosystem services (MEA 2005) that highlights the wide-ranging importance and value of these services. The MEA divided up these services into four categories including provisioning, regulating, supporting, and cultural services (table 1). Provisioning services are a familiar part of the economy that provides goods such as food, freshwater, timber and fiber for direct human use. Regulating services maintain a world in which it is possible for people to live, and provide benefits

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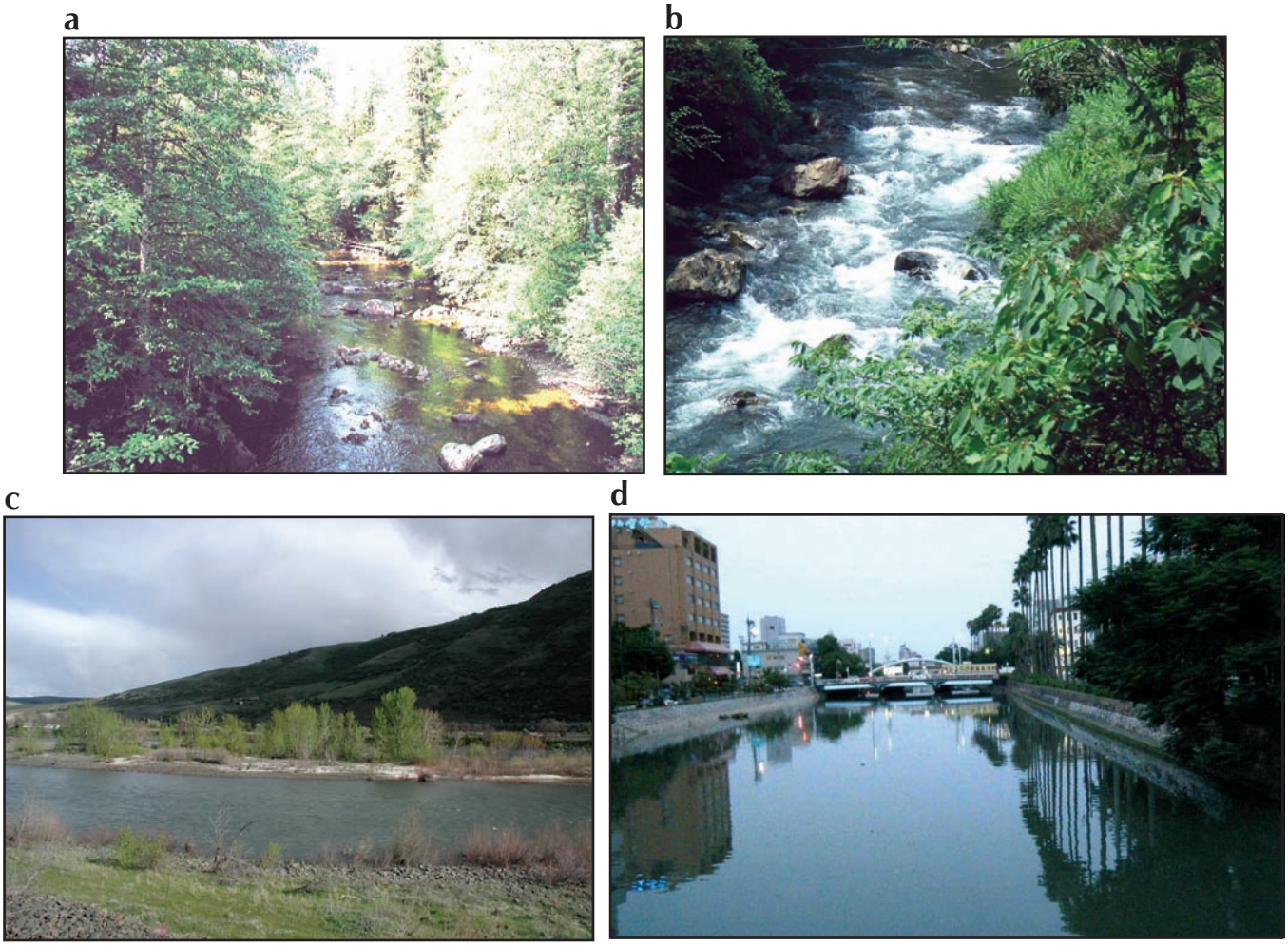

Figure 1—Examples of water as a global ecosystem service: a) Steelhead Creek, southeast Alaska, providing clear, cold water for high quality fish habitat and aquatic resources, b) headwater stream in Shikoku, Japan providing clear drinking water, c) Clearwater River in Idaho providing water for irrigation and flood control but with reduced habitat for migrating salmon, d) cement-walled river in Kochi, Japan, with highly degraded water quality and aquatic services.

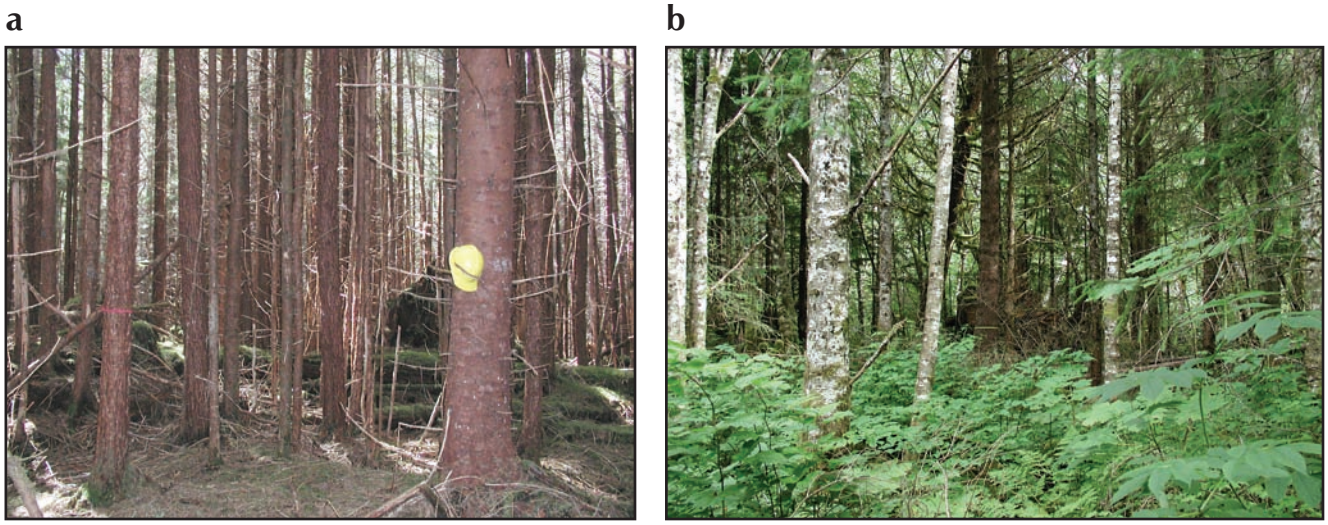

Figure 2—Examples of managed forest as an ecosystem service for biodiversity: a) pure 40-year-old conifer plantation in southeast Alaska with no plant understory and reduced wildlife habitat, b) mixed red alder-conifer 40-year-old forest in southeast Alaska with abundant plant understory and improved habitat for deer and small mammals.

Table 1—Categories of ecosystem services provided by nature. Modified from the Millenium Ecosystem Assessment (MEA 2005).

Ecosystem Services	
Supporting Services Nutrient cycling Soil formation Primary production	**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

such as flood and disease control, water purification, climate stabilization and crop pollination. Supporting services are the underlying processes that maintain the conditions for life on Earth and include nutrient cycling, soil formation and primary production from our ecosystems. Cultural services make the world a place where people want to live and include recreational, spiritual, aesthetic and cultural values.

Climate in a narrow sense is usually defined as the average weather over a period of time ranging from months to thousands or millions of years. The relevant quantities are most often surface variables such as temperature, precipitation and wind. Climate, in a wider sense, is the state of the climate system. Climate change refers to a change in the state of the climate that can be identified (e.g., by using statistical tests) by changes in the mean and/or the variability of its properties, and that persists for an extended period, typically decades or longer. Climate change may be due to natural internal processes or external forcings, or to persistent anthropogenic changes in the composition of the atmosphere or changes in land use. Note that the United Nations Framework Convention on Climate Change (UNFCCC 2007) defines *climate change* as: "a change of climate which is attributed directly or indirectly to human activity that alters the composition of the global atmosphere and which is in addition to natural climate variability observed over comparable time periods." The UNFCCC thus makes a distinction between climate change attributable to human activities altering the atmospheric composition, and climate variability attributable to natural causes.

Integration of Ecosystem Services and Climate Change

Ecosystem service providers are increasingly recognized as having an important role to play in ecosystem protection. The emerging regulated market for carbon could also provide incentives for reducing greenhouse gas emissions and sequestering more carbon; both important considerations for mitigating climate change. The concept of providing incentives through market-based programs for ecosystem services, and the recent emergence of markets for carbon, water, wetlands and biodiversity has stimulated interest from a broad suite of new stakeholders. These markets provide incentives for landowners to provide clean air and water, wildlife habitat, and other goods and services from their forests and wetlands. These new financial incentives expand opportunities for forest landowners to gain revenue from their lands while providing public goods to society. Ecosystem services when considered as "natural capital" leads land managers to regard landscapes as natural assets that requires measuring different ecosystem services and ensuring the people who use these services know their value and the cost of losing them (Collins and Larry 2008; Kline 2006).

The importance of healthy, functioning ecosystems is widely recognized. Forests play a major role in the global carbon cycle through the ability of trees to withdraw or sequester carbon, and forests serve as a terrestrial carbon sink during most stages of forest development. Forests also have high conservation value for a number of threatened and endangered species, for mitigating pollution, for flood control and for other ecosystem services. Forests can play a major role in reducing greenhouse gas emissions through maintaining current carbon stores and by increasing the rate of carbon sequestration. Forest carbon is a particularly important ecosystem service to monitor and manage because there is interest in both maintaining current forest carbon stocks and also increasing carbon sequestration as a mitigation strategy for reducing atmospheric CO_2.

Deforestation accounts for approximately 20 percent of total greenhouse gas emissions (FAO 2005) and one of the major forestry concerns is reducing the loss of forestland from development. Maintaining these carbon stores is an important component of global carbon management. Forests can sequester large amounts of carbon in several ways including as carbon sinks in the standing forest, in wood products, and in avoided emissions when wood is used as a substitute for more fossil fuel-consuming products such as steel, concrete and brick. Other considerations include forest management practices that increase carbon sequestration such as extended rotations or increased growth rates through intensive forest management. Forest management will be required to help forests adapt and maintain high levels of carbon sequestration as climate changes.

The integration of ecosystem services markets and the role of forest carbon to reduce greenhouse gas emissions may be an effective framework for mitigating some of the effects of climate change. These markets can be helpful for both increasing carbon sequestration as part of a regulated carbon market and as financial incentives for landowners to prevent forestland from being developed. We explore some of these concepts in our paper and describe some of the markets for ecosystem services and the potential effects of climate change on forestry and some of the management practices relating to forests and carbon sequestration. The specific objectives of this paper include: 1) describe the policy and regulatory frameworks of emerging markets for ecosystem services in the USA, 2) describe the relationship between climate change, forest ecosystems and carbon, and some of the opportunities to mitigate climate change, and 3) develop a framework for integrating ecosystem services markets and climate change using forest carbon.

Markets for Ecosystem Services

Policy and regulations have an essential role for establishing markets for ecosystem services and market-based programs have developed in response to regulations for water, wetlands and endangered species. Examples of regulation driven markets include the Clean Water Act (33 U.S.C. 1344) that helped establish wetland mitigation banking and water quality trading (Brauman and others 2007; Gaddie and Regens 2000), and the Endangered Species Act (USFWS 1988) that led to the emergence of species conservation banking (Carroll and others 2007). However, these different ecosystem services are regulated and controlled by several different federal and state agencies with their own sets of policies and regulatory frameworks. For instance, at the national level, air and water quality is regulated by the Environmental Protection Agency (EPA), wetlands are regulated by the Army Corps of Engineers (USACE), and species conservation is controlled by the U.S. Fish and Wildlife Service (USFWS). Several markets for ecosystem services are emerging in the U.S. with potentially new revenue streams for forest landowners. These new markets offer potential financial incentives to landowners to maintain and manage forestlands rather than converting these forests to other uses. Overviews of U.S. carbon markets, water quality trading and wetland and species mitigation banking are outlined here.

Emerging Carbon Markets in the USA

The United States is not a signatory of the Kyoto Protocol (UNFCCC 2007) and the U.S. does not have a comprehensive national policy mandating limits in CO_2 emissions. Instead, the U.S. has voluntary, or state and region-based programs to reduce greenhouse gas emissions. Project-based transactions can generate offset credits by an approved activity that compensates for emissions by a business in a regulated sector. Examples of offset credits include forest carbon sequestration, methane recapture, and alternative energy use. Since about 20 percent of human-induced carbon dioxide emissions are due to land-use change and deforestation (FAO 2005), sustainable forest management can play an important role in climate change mitigation. Forestry offsets also provide a range of environmental benefits, such as wildlife habitat and water quality improvement.

Due to the absence of a comprehensive GHG regulatory emissions reduction standards (e.g. national cap-and-trade legislation), voluntary carbon markets have dominated in the USA and state and region-based programs are being developed to reduce greenhouse gas emissions. Regional and state programs include the Regional Greenhouse Gas Initiative (RGGI) in the Northeast USA (RGGI 2007), the Western Climate Initiative (WCI) in the Western USA (Capoor and Ambrosi 2008) and the Climate Action Registry in California (CCAR 2005). However, due to different regulatory frameworks being developed in each region and state, there is a need for developing national standards to help develop the registration and trading of carbon offset projects (Sampson 2004). Ruddell and others (2007) further contends that in the absence of such national standards, forestry offset projects will continue to be limited and inconsistent.

Although the voluntary U.S. carbon market is small compared with the global carbon market estimated at about $130 billion ($US) in 2009, the U.S. voluntary carbon market increased by 200 percent in 2007 with 13 percent of the carbon trading including carbon sequestration or forestry credits (Forestry Source 2007). By comparison, no forestry credits are accepted under the European Union Emission trading scheme and less than 1 percent of total transactions of 475 million tons made under the Kyoto protocol's Clean Development Mechanism involved

forestry-based credits (UNFCCC 2007). With a regulated cap-and-trade mechanism that provides higher prices than current carbon values and the allowance of forest carbon offsets, the carbon market could provide a huge incentive for forestry. However, it is important that these forestry offsets provide high-quality carbon sequestration credits in order to assure early investors in the carbon market that these carbon offsets are credible and provide true reductions in GHG emissions.

To address GHG policy, the forestry community has a significant opportunity to shape what kinds of forest projects are included. Lawmakers in the U.S. have a variety of pending legislation with significant implications for carbon and forestry including the 2008 Farm Bill, 2009 American Clean Energy and Security (ACES) Waxman-Markey bill, and other federal and state legislation. Two key components for any forestry offset project include keeping forestland in forests, and increasing carbon sequestration through forest management. There are also a number of important policy issues to incorporate in forestry offsets including clear definitions for carbon baselines and additionality, permanence and leakage, possible inclusion of wood products for the long-term storage of carbon, and projects that promote additional carbon sequestration and discourage conversion of forests to other land uses (Cathcart 2000; Ruddell and others 2007).

Water Quality Trading

Ecosystem services for water include water supply, water damage mitigation, and water-related cultural services (Brauman and others 2007; fig. 1). Unlike global carbon markets, market-based schemes for improving water quality are generally limited to local or regional programs within a specific watershed. Forest landowners and farmers can be included as sellers of water quality credits in many programs. Other participants include water quality permitting authorities, third-party brokers, conservation organizations, watershed councils, and private industry groups. Local examples of water quality trading include the EPA watershed-based permit for the Tualatin River in Oregon that allows trading to achieve the permit requirement for temperature (Cochran 2007). Here, instead of installing refrigeration systems at two Tualatin River treatment plants (at a cost of $60 million), the wastewater utility paid upstream farmers to plant shade trees in the riparian areas (at a cost of $6 million).

Wetland and Species Mitigation Banking

Another market for ecosystem services is wetland mitigation and species conservation banking. These markets are based on regulations that require developers to obtain a permit to offset any loss of wetland or habitat before they are allowed to harm a wetland or an endangered species. Wetland mitigation banking has developed into a well-established, market-based system where buyers and sellers of credits conduct transactions through wetland banks. Wetland ecosystems provide a broad range of ecological services for people, and studies have shown the importance of services provided by wetlands including water quality and quantity, recreation, wildlife habitat, flood control and pollution interception (Azevedo and others 2000; Hoehn and others 2001). On-site wetland mitigation has been largely unsuccessful for restoring original wetland functions but larger offsite-wetland banks are now recognized for their broader functionality and production of multiple ecosystem services (Gaddie and Regens 2000; Willamette Partnership 2008).

Conservation banking, the creation and trading of credits that represent wildlife conservation values on private lands is more than a decade old, and the State of California has developed most of the conservation banking agreements in the U.S. (Fox and Nino-Murcia 2005). A conservation bank is a parcel of protected

natural land that is authorized to sell a set number of credits, usually in the form of land area of habitat, to the customer that is required by law to mitigate their impact to the same species and habitat on nearby land. Private landowners reported that financial motives were behind most of their interest in conservation banking but bureaucracy was the biggest challenge with the average time for establishing banks more than 2 years and varied from 8 months to over 6 years (Fox and Nino-Murcia 2005). However, as Fox and Nino-Murcia (2005) contend, the fact that banks are profitable in most cases is an indication that conservation banking offers viable incentives to protect species on private land.

Climate Change and Forestry

Climate Change and Forest Ecosystems

Some effects of climate change on forest ecosystems and natural resources in North America are already detectable (IPCC 2007), and no historical analog exists for the combination of future climate conditions, disturbance regimes, and land-use patterns expected in the future. Climate provides an overarching control on the distribution of tree species (Woodward 1987). Climate-induced stress occurs in areas where species may be marginally suited, such as the edge of their geographic distribution. As a result, a warmer climate will lead to potential changes in species distribution and abundance at various spatial scales. Changes in composition may be slow even in a rapidly warming climate, because mature individuals are typically resistant to climatic variation. Therefore, disturbance will be a major agent of change and will promote change through forest regeneration at shorter time scales than the direct influence of climate (McKenzie and others 2004; fig. 3).

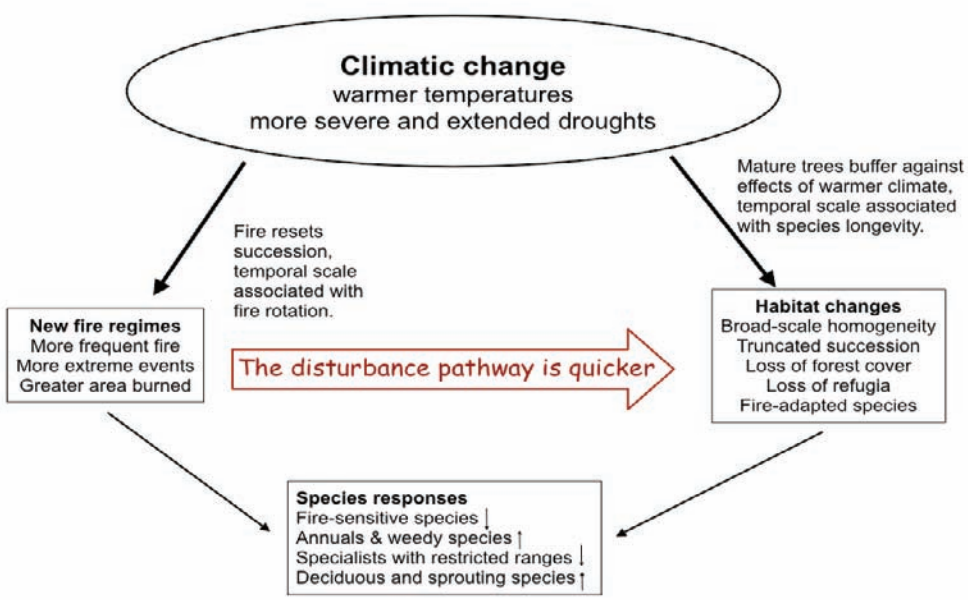

Figure 3—Conceptual model of the effects of climatic change and disturbance on forest ecosystems. Times are approximate. Adapted from McKenzie and others (2004).

Limiting factors act at the interface between organisms and their environment, and plant performance is affected when one or more resources (e.g., energy, water, nutrients) limits physiological function. Forests of western North America can be considered as energy limited, water limited, or some combination thereof (Littell and others 2008). Energy limitations are primarily light (e.g., productive forests where high leaf area reduces light exposure in the canopy) and temperature (e.g., subalpine and boreal forests). Some energy-limited forest systems appear to be responding positively to warming temperatures over the past 100 years (Peterson 1998).

Productivity in water-limited forests will decrease in a warmer climate, because negative water balances will reduce photosynthesis (Hicke and others 2002), although this may be partially offset if CO_2 fertilization increases water-use efficiency (Neilson and others 2005). For example, most montane Douglas-fir (*Pseudotsuga menziesii*) forests across the northwestern United States are water limited (Littell and others 2008), and the area and magnitude of this limitation will increase as the climate continues to warm. Limiting factors typically vary within species (Peterson and Peterson 2001), between seasons, and with respect to the balance between water and energy needs (Stephenson 1998).

The conceptual model of a "disease spiral" (*sensu* Manion 1991) in which tree death is caused by the accretion of multiple stresses can be scaled up to the concept of a "stress complex" for populations of tree species and for multiple populations at the ecosystem level (McKenzie and others 2009). Temperature increase is a predisposing factor causing stress in forest ecosystems of western North America by exacerbating negative water balance (Littell and others 2008; Stephenson 1998) and through increased frequency, severity, and extent of disturbances. Climate change and the combination of warmer temperatures, drought, and more severe disturbance regimes can create stressful conditions for forest ecosystems over large geographic areas.

The principal disturbance regimes of western North America, wildfire, and insect outbreaks, respond to short-term weather and annual-to-decadal cycles in climate. For example, synchronous fire years are associated with the El Niño Southern Oscillation cycle in the American Southwest and southern Rocky Mountains (Swetnam and Betancourt 1998; Veblen and others 2000) and to some extent in the Pacific Northwest (Hessl and others 2004). Short-term weather anomalies associated with atmospheric blocking ridges of high pressure promote extreme wildfire years in some areas of the West (Gedalof and others 2005). Insect defoliators are favored in years during which vegetation productivity is high (Weber and Schweingruber 1995), but overall forest vigor is low (Swetnam and Betancourt 1998).

Higher temperatures are expected to alter the frequency, severity, and extent of natural disturbances, and wildfire (McKenzie and others 2004; Westerling and others 2006) and mountain pine beetle outbreaks (Logan and Powell 2001) may become a more dominant feature of western landscapes. Where fire and insect disturbances interact, changes in forest ecosystem structure and function may be accelerated (Veblen and others 1994), resulting in altered combinations of species, productivity, and disturbance regimes (table 2).

Forest Management for Carbon Sequestration

Carbon sequestration in forests is one ecosystem service that will be sensitive to climate change, and forest management will be necessary to facilitate forest adaptation as the climate changes. Sustainable forest management can not only maintain carbon sequestration at current levels, but can also increase carbon sequestration to mitigate atmospheric CO_2 concentrations. Sustainable forest

Table 2—Examples of stress complexes in western North American forests that could be affected by a warmer climate.

Pinyon-juniper woodland (American Southwest)

Pinyon pine (*Pinus edulis* Engelm.) and various juniper species (*Juniperus* spp.) are among the most drought-tolerant trees in western North America and clearly occur in water-limited systems. Multi-year droughts have caused historical diebacks of pinyon pines over large geographic areas in the American Southwest, but the current dieback is unprecedented in terms of the scale of response to a period of low precipitation and high temperature (Breshears and others 2005). A warmer climate has been a predisposing factor, and wood-boring insects have contributed to weakening and ultimately killing trees.

Mixed conifer forest (Sierra Nevada, southern California)

Forests in central and southern California have a Mediterranean climate with long dry summers, and mild winters during which most of the annual precipitation occurs. Fire exclusion has increased fuel loadings (McKelvey and others 1996) and competitive stress as stand densities have increased (van Mantgem and others 2004). Elevated levels of ambient ozone have reduced net photosynthesis, growth, and interannual accumulation of biomass in ponderosa pine (*Pinus ponderosa* C. Lawson var. *ponderosa*) and Jeffrey pine (*P. jeffreyi* Balf.) in the Sierra Nevada and southern California mountains (Byternowicz and Grulke 1992; Miller 1992; Peterson and Arbaugh 1988; Peterson and others 1991). Bark beetle outbreaks in these regions have caused extensive mortality in recent years following protracted droughts.

Lodgepole pine forest (western North America)

Lodgepole pine (*Pinus contorta* Douglas ex Louden var. *latifolia* Engelm. ex S. Watson) is the principal host of the mountain pine beetle (*Dendroctonus ponderosae*), and dense stands that are stressed from low soil moisture are particularly vulnerable to mortality during beetle outbreaks (Hicke and others 2006). Recent beetle outbreaks have caused extensive mortality across millions of hectares in western North America (Logan and Powell 2001), with large mature cohorts (age 70-80 yr) contributing to widespread vulnerability. Tree mortality caused by beetles produces rapid necromass (fuel) accumulation, and the potential for species conversion following stand-replacing fires, including a favorable environment in some locations for establishment of drought-tolerant species such as interior Douglas-fir and ponderosa pine.

Boreal forest (central and southern Alaska)

Alaska has experienced historically unprecedented areas burned by wildfire in the last decade (NIFC 2006). Concurrently, large outbreaks of the spruce bark beetle (*Dendroctonus rufipennis*) occurred in white spruce (*Picea glauca* [Moench] Voss) forests on and near the Kenai Peninsula in southern Alaska (Berg and others 2006). Fire and beetle outbreaks are likely associated with warmer temperatures in recent decades (Duffy and others 2005, Werner and others 2006). In interior Alaska, white spruce and black spruce (*Picea mariana* [Mill.] Britton, Sterns & Poggenb.) are more flammable than co-occurring deciduous species such as paper birch (*Betula papyrifera* Marsh.). Conifers are the target of bark beetles, so in southern Alaska they are disadvantaged compared to deciduous species. As a result, this system may transition to deciduous trees via more frequent and extensive disturbance associated with a warmer climate.

management practices aimed at mitigating atmospheric CO_2 are more likely to be successful if they are specific to different forest types and disturbance regimes within western North America. Furthermore, these mitigation strategies will be more effective if they are implemented with consideration of the expected effects of climate change on forest ecosystems given that some degree of climate change is inevitable despite current mitigation actions.

Afforestation and reforestation of previously forested lands are two forest management practices with the greatest potential to increase carbon sequestration. These management practices, if properly implemented, can remove additional carbon from the atmosphere and sequester it for decades to centuries. These projects will be more successful if they are implemented in combination with

management practices that also facilitate forest adaptation to climate change. Adaptation strategies include selecting for planting species or varieties that are adapted to a warmer climate, planting a greater diversity of species, and planting at lower initial densities to reduce moisture stress in water-limited forests. These adaptation strategies will help maintain carbon storage by increasing forest productivity and resilience to warmer temperatures and more frequent disturbances.

Climate-driven increases in wildfire frequency, extent, and severity are expected to affect the potential of forest ecosystems to sequester carbon. In water-limited forests, climate change may also reduce regeneration success after severe wildfires due to greater climate-induced stress in seedlings. A vegetation type conversion (e.g. from forest to shrubland) or a reduction in forest density can reduce carbon sequestration more than the wildfire itself (Kashian and others 2006). Therefore, forest management practices that ensure adequate post-fire regeneration with appropriate species, genotypes, and densities are important for enhancing forest resilience to climate change and maintaining the carbon sequestration functionality of the forest ecosystem.

Thinning forests to reduce disturbance severity and extent (fuel treatments) is another forest management practice that can enhance resilience to disturbances, as well as maintain and enhance carbon sequestration. Individual wildfires are a large, one-time source of carbon emissions that can be significant in the short-term (Turner and others 2007; fig. 4). However, the carbon sequestration benefits of fuel treatments may be less than expected because of four common misconceptions regarding carbon and wildfires. First, wildfires, even those burning with high severity, typically consume less than 20 percent of total live and dead forest biomass (Campbell and others 2007). Although more than 80 percent of trees can be killed in high severity fires, the carbon is generally released slowly over decades as the biomass decomposes. Second, the difference between biomass consumption in high and low severity fires is small, about 10 percent (Campbell and others 2007). Third, as fire-killed material decomposes and releases carbon, carbon is returned to the system as post-fire regeneration and the productivity of these young regenerating forests is higher than that of the older forests they

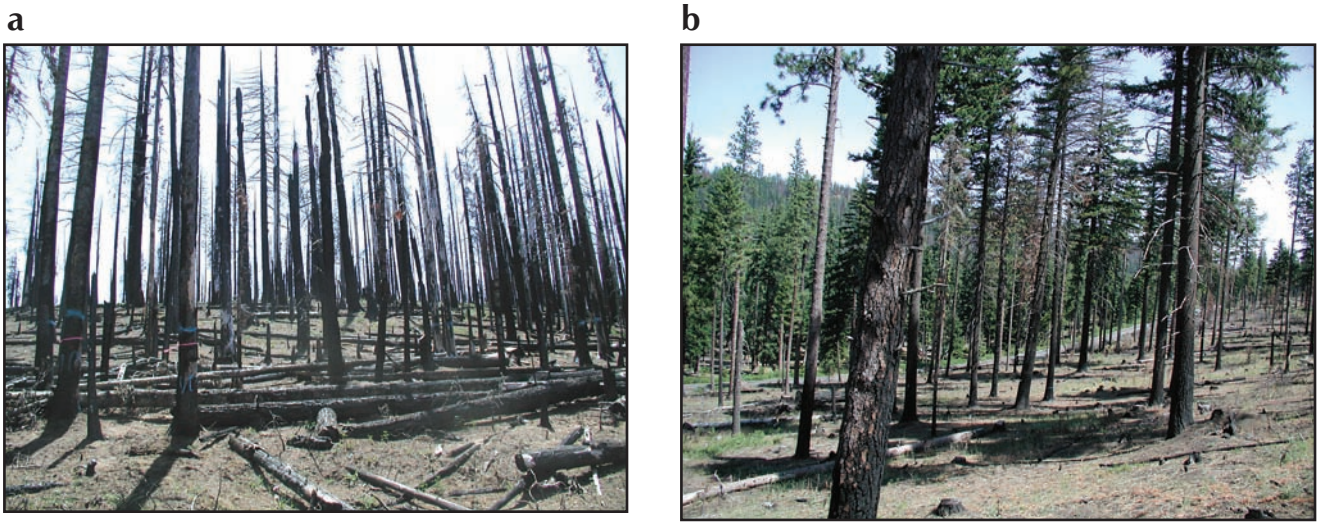

Figure 4—Examples of fire as an ecosystem service: a)severe fire intensity after B&B fire in central Oregon with loss of organic layers, exposed mineral soil and reduced forest productivity, b) moderate fire intensity after B& B fire with pre-fire thinning treatment to reduce fire severity resulting in relatively healthy forest ecosystem after fire.

replace. Fourth, at long temporal scales (the scale being relative to the ecosystem-specific fire return interval) the net release of carbon from any fire-disturbed ecosystem may be zero as long as the forest regenerates and reaches the pre-fire age and density (Kashian and others 2006).

The carbon benefits of fuel treatments in forest ecosystems depend on the fire regime characteristic of the ecosystem (fig. 4). Fuel treatments will not incur carbon storage benefits in high severity, low frequency fire regimes (fire return intervals on the order of centuries) (Mitchell and others 2009). Treatments may need to be repeated to maintain low fire hazard and the total carbon removed in successive treatments over centuries can exceed the carbon emitted in a single, high severity fire (Hurteau and North 2009). However, fuel treatments are unlikely to reduce fire severity in these forest types because fire severity is more a function of weather than fuel availability (Brown and others 2004).

Conversely, fuel treatments can enhance carbon storage in forests with low severity, high frequency fire regimes (fire return intervals on the order of years to decades) (Hurteau and North 2009; Mitchell and others 2009), especially forests that have experienced biomass accumulation due to fire suppression (Brown and others 2004). In these ecosystems, fuel treatments can effectively reduce subsequent wildfire severity and carbon emissions. Fuel treatments reduce forest productivity in the short term (1-3 years), but ecosystem productivity often returns to or exceeds pre-treatment levels within only a few years (Boerner and others 2008). Furthermore, reduced productivity in the tree component (in proportion to tree removal) is compensated by increased productivity in roots and understory vegetation, which respond positively in more open stands (Campbell and others 2009). However, the carbon benefits of fuel treatments are marginal even in low severity fire regimes. Fuel treatments remove substantial carbon from the site and a subsequent wildfire, even with effective fire severity reduction, will release additional carbon (Mitchell and others 2009). The total carbon removed in only a few treatments may exceed the carbon gains from fire severity reduction because of the small difference in biomass consumption between high and low severity wildfires (Hurteau and North 2009; Mitchell and others 2009). However, fuel treatments are a useful management tool for maintaining other ecosystem services, including air quality, water quantity, and wildlife habitat, and should be considered based on their benefits to multiple ecosystem services, not just carbon sequestration (fig. 2).

Certain forest management practices may increase the carbon sequestration potential of fuel treatments. Fuel treatments will have greater carbon storage benefits if a small area can be treated to reduce fire severity over a larger area through the strategic placement of treatments on the landscape (Finney 2001). Carbon sequestration can also be enhanced with specific uses of the biomass that is removed in treatments. The carbon may be stored for up to 100 years or longer if the material is used in long-lived forest products. Carbon benefits also increase if the biomass is used as an energy source that is substituted for energy that would otherwise be derived from fossil fuels (Mitchell and others 2009). Increasing the production of biofuels using biomass removed from thinning forests can increase the carbon benefits of fuel treatments (fig. 5).

Discussion

The Role of Ecosystem Services for Ecosystem Protection

Ecosystem services provide provisioning, supporting, and regulating services that are critical for the functioning of life on Earth and provide natural assets that are intrinsic components of our economy. However, recent evaluation of the state

Figure 5—Examples of biomass utilization and use of forest carbon: a) small-to-medium size diameter logs following forest thinning, b) grinding of branches and small diameter trees for wood chips and hogg fuel for biomass energy, c) bundling of biomass for hogg fuel for co-generation energy source.

of the world's ecosystems shows that about 60 percent of all ecosystems are rapidly degrading or are being used unsustainably (MEA 2005). Emerging markets for ecosystem services are increasingly recognized as having an important role to play in ecosystem protection. Market mechanisms can be used to provide incentives to private forest landowners to enhance provision of ecosystem services, often with the associated objective of providing a counterbalance to financial incentives to convert forests to other land uses (Kline 2006). These new financial incentives expand opportunities for forest landowners to gain revenue from their lands while providing public goods to society.

Collaborative efforts are being developed at local, regional, national and international levels to better conserve our natural resources (Boyd 2004; Daily 1997; Heal and others 2005; Oliver and Deal 2007). There are several organizations in the United States that are interested in developing an ecosystem marketplace that could buy and trade different ecosystem services (Bay Bank 2008; Katoomba 2007; Willamette Partnership 2008). This marketplace could help a single large

landowner or a group of landowners sell wetland, endangered species, water quality and carbon credits from the same piece of land. For example, in Oregon, the Willamette Partnership recently received a NRCS Conservation Innovation Grant to develop a multi-credit market system to measure and account for multiple types of ecosystem service credits for use within the Willamette Ecosystem marketplace (Willamette Partnership 2008). This multi-credit marketplace would be able to take advantage of efforts to combine or bundle different ecosystem services. Other examples include the 2008 USDA Farm Bill, section 2709 (USDA 2008) that shifted an emphasis from commodity-oriented programs to more market-based payment programs, achieving movement toward this ecosystem services goal. Pending carbon cap and trade legislation such as the 2009 ACES Waxman-Markey bill could further reduce greenhouse gas emissions using market mechanisms and forestry and agricultural offset programs. Market-based incentives for ecosystem services has provided a new framework for a diverse coalition of conservationists, forest landowners and other stakeholders to work together to develop market based strategies for conserving ecosystem services. This has led to a shift from thinking about conservation as a burden or endangered species as a liability, to the concept of restoration and stewardship of ecosystem services as a profit making enterprise (Collins and Larry 2008; Heal and others 2005).

The Role of Forests, Forestry, and Wood Products for Sequestering Carbon

Forests and forestry have an important role for sequestering carbon and reducing greenhouse gas emissions. Forests can sequester large amounts of carbon in several ways including as carbon sinks in the standing forest, in wood products, and in avoided emissions when wood is used as a substitute for more fossil fuel-consuming products such as steel, concrete and brick. One of the obvious and most important roles for reducing CO_2 emissions is avoiding deforestation and keeping forestlands in forests. Globally, about 20 percent of human-induced carbon dioxide emissions are due to land-use change and deforestation (FAO 2005). This is important at the global scale and here in the United States where land conversion and development has led to more than 2,500 acres of forest loss each day, with more area being impacted by forest fragmentation (Alig 2007). Afforestation and reforestation of previously forested lands is an important and widely accepted forest management practice to increase carbon sequestration. Storage of carbon in wood products can also have a significant impact in storing carbon and avoiding use of more fossil fuel-intensive products. Preliminary calculations suggest a 20 percent to 50 percent decrease in fossil fuel use if forests and wood products are used to sequester carbon in place of more fossil fuel-consuming products such as steel, concrete and brick (Lippke and others 2004). An example of local biomass utilization is highlighted on the Deschutes National Forest in central Oregon. In fiscal year 2008, biomass utilization included 150,000 green tons of small diameter wood converted into mulch, pulp chips, animal bedding, lumber and poles with an additional 69,000 green tons converted into hogg fuel or firewood that was used for energy or heat as an offset to fossil fuel consumption (fig. 5). Wood can play an important role as a substitute for fossil fuels; however, it is important to note that wood used for energy is much less efficient than wood used for construction. Currently, approximately 50 percent of the world's wood harvest is used for fuel, primarily in areas of low economy such as Africa where approximately 90 percent of the wood harvest is used for fuel for cooking (Oliver and Deal 2007). Another key consideration is how carbon markets, forestry management, wood products and carbon credit programs are administered. Actual carbon market trading will involve a number of complex variables relating to establishment of

existing carbon baselines and additionality, leakage and permanence, inclusion of wood products for the long-term storage of carbon, and programs that promote additional carbon storage through forest management practices that discourages forest land conversion (Cathcart 2000; Ruddell and others 2007).

Conclusion

Ecosystem services when considered as "natural capital" leads land managers to regard landscapes as natural assets. Furthermore, the integration of ecosystem services markets and the use of forests to sequester carbon may be an effective framework for mitigating some of the effects of climate change. Several markets for ecosystem services are emerging in the USA with potentially new revenue streams for forest landowners. These new markets offer potential financial incentives to landowners to maintain and manage forestlands rather than converting these forests to other uses. There is increasing interest in the use of market-based approaches to add value for these services and assist conservation of natural resources. This integration of ecosystem services and climate change may provide some new opportunities for forest landowners and managers to enhance forest stewardship in addition to reducing greenhouse gas emissions through forest carbon sequestration. There is a also a need for a more integrated approach that combines different ecosystem services and provides financial incentives for forest landowners to achieve broad conservation goals.

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Integrating Climate Change Considerations into Forest Management Tools and Training

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Abstract—Silviculturists are currently facing the challenge of developing management strategies that meet broad ecological and social considerations in spite of a high degree of uncertainty in future climatic conditions. Forest managers need state-of-the-art knowledge about climate change and potential impacts to facilitate development of silvicultural objectives and prescriptions that are flexible and enhance ecosystem resistance and resilience. Existing approaches also must be infused with adaptive techniques and strategies. We are working on a project that will help address these needs for forest management. Specifically, our objectives are to: (1) provide training on the ecological impacts of climate change to forestry professionals; (2) incorporate current understanding of species and forest responses to climate change into silvicultural strategies that meet management objectives while encouraging adaptation to changing climate conditions; (3) provide decision support tools to guide forest management planning under climate change; and (4) integrate climate change adaptation strategies into Forest Service silvicultural education and implementation. We describe the incorporation of training and tools from the larger project into the USDA Forest Service silviculture certification program, and report on lessons we have learned about the process attendant to the design and implementation of climate change training for land managers.

Keywords: climate change, forest management, NASP, silviculture certification, uncertainty

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Introduction

Evidence for global climate change is unequivocal, as is the implication that human activity has caused substantial rises in greenhouse gases that are contributing to the earth's changing climate (IPCC 2007). Temperature, precipitation, and other climate variables are expected to continue changing worldwide, with substantial implications for the composition and function of terrestrial ecosystems (Kirilenko and Sedjo 2007). In the United States, forestland covers 33 percent of the land surface (Smith and others 2004) and provides extensive ecosystem services. Managers of these lands are seeking information and direction to create flexible and adaptive management approaches that balance the need to manage forests today for multiple objectives with the reality that future climatic conditions are uncertain.

An abundance of information is available about global climate change and how to promote forests that are able to adapt, but the mechanisms to combine and apply this information to specific situations are less common. This leads to a great need for educating land managers about climate change, its uncertainties, and the importance of incorporating both into silvicultural prescriptions.

Our approach to this complex issue centers on a combination of education and problem-solving exercises. We first educate participants about state-of-the-art climate change science and the potential ecophysiological responses of forests. We then facilitate problem-solving by engaging critical thinking skills so that approaches and strategies are developed from the ground up, building on local knowledge and experience of participants.

This overall approach provides participants with fundamental information on likely climate change impacts on forests, and arms them with a structure for incorporating these considerations into future management strategies. A common criticism of continuing education training is that the information gained often lacks practical application and does not strengthen attendee's ability to solve complex, real-world problems largely due to a one-way flow of knowledge (Salwasser and others 1990). The approach we outline here does not instill a one-size-fits-all solution, but rather builds in mechanisms and tools for further creative problem-solving to generate approaches to complex problems relevant to local forest types. This acknowledges that every situation confronting land managers is unique (i.e., forest plans, vegetation types, stand conditions, conflicting objectives) and requires management strategies particular to those conditions.

As part of a larger project funded by the USDA Forest Service Global Change Program aimed at providing advanced training in climate change science to land managers within the USDA Forest Service, we extended a training module geared toward forest management in the context of an uncertain climate future to the contingent of agency silviculturists in training for certification. Silviculture is an interdisciplinary practice used to manage for multiple objectives at a variety of scales with the local-level (stand), the scale at which on-the-ground actions (prescriptions) are implemented. Silviculture is recognized as a leading discipline within the forestry profession for implementing science-based practices, and leads the forestry profession by setting standards in continuing education and training (Graham and Jain 2004). The National Advanced Silviculture Program (NASP) is the current silviculture certification program within the USDA Forest Service. The program contains four core training courses led by academic institutions, and a fifth local area module developed specifically for expertise in a given forest type. Each course consists of a variety of topics taught by experts in their respective fields with the goal of preparing silviculturists to design prescriptions that meet the Minimum National Standards for Silviculturist Certification (Forest Service Handbook 2005). We incorporated a climate change module into the most recent NASP class, with these overall goals: 1) provide the latest scientific information about climate change and forest ecosystem responses; 2) develop a mindset for silviculturists to think about climate change considerations as part of decision making in forest management; and 3) develop a list of general forest management strategies that are applicable across national forests in the context of an uncertain climate future. The process implemented and outcomes achieved from this module are described below.

Educational Process

Context

The two-week Ecological Systems Course of NASP is the first course of the certification program, and is conducted at Michigan Technological University, Houghton, MI. A pilot four-hour climate change module was integrated into the third cohort of NASP (May 2009) as a new and essential component of the curriculum, and represents a logical and appropriate place to begin incorporating the complex issue of climate change into formal silviculture training. The module was

presented at the beginning of the second week, with the students already having several hours of basic forest ecology, geology, landforms, soils, hydrology, tree physiology, and disturbance ecology.

Approach

The process implemented during the climate change module consisted of a modified version of approaches utilized during an initial Silviculture Workshop geared toward Region 9 silviculturists held in fall, 2008 (Janowiak and others, unpublished paper). The pedagogical approach utilized components of the learning cycle (Allard and Barman 1994) through participatory engagement on the topic (fig. 1). To begin, the class was presented with a brief overall introduction of the topic at hand, and given a description of the goals for the climate change module in the context of NASP.

Inform and educate—To provide a sound and consistent scientific basis for building discussion, the first hour of the module consisted of a presentation of climate change science and forest response followed by an extensive question and answer session (fig. 2). The presentation covered several topics central to climate change issues in forest management. It included an overview of climate change science (observations, mechanisms, and predictions), the global carbon cycle, distribution and density of forest carbon, and how climate change affects forest ecosystems including benefits and deleterious effects on forest productivity. Uncertainty surrounding climate change and forest response was emphasized as material was presented, encouraging students to synthesize and evaluate information at higher levels of learning (Bloom 1956). To help students prepare for this topic and to provide context for climate change as it affects silviculture prescription development, three papers were assigned as pre-work: Birdsey and others 2006, Millar and others 2007, and the USDA Forest Service Strategic Framework for Responding to Climate Change (2008). The goals here were twofold: 1) to inform and educate participants about climate change providing a common level of understanding; and 2) to address misconceptions and skepticism that may be present among the group about climate change as a pervasive and real issue facing forest managers.

Engage—The goal of the first learning activity (fig. 1) was to generate interest, enable active participation, and begin the flow of problem-solving ideas by posing this question to the group: *What new or altered considerations does climate*

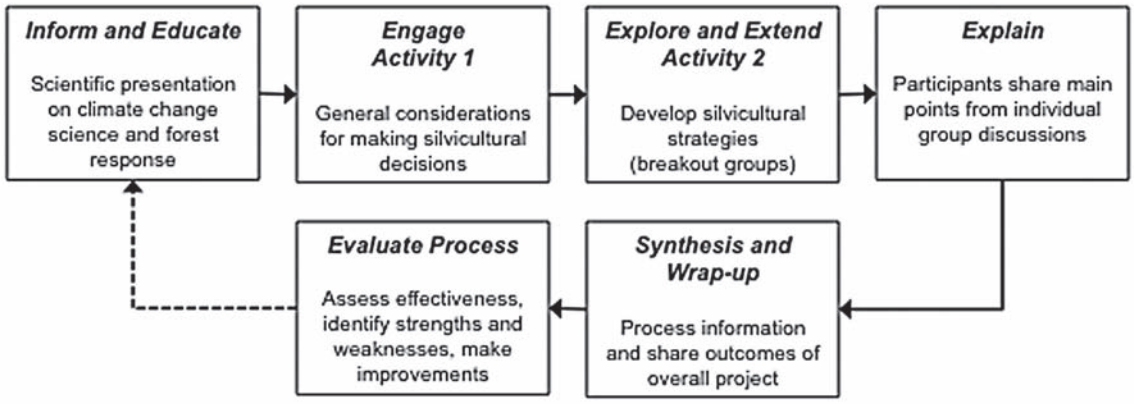

Figure 1—Modified learning cycle used in the climate change module of the Ecological Systems Course of the National Advanced Silviculture Program (NASP).

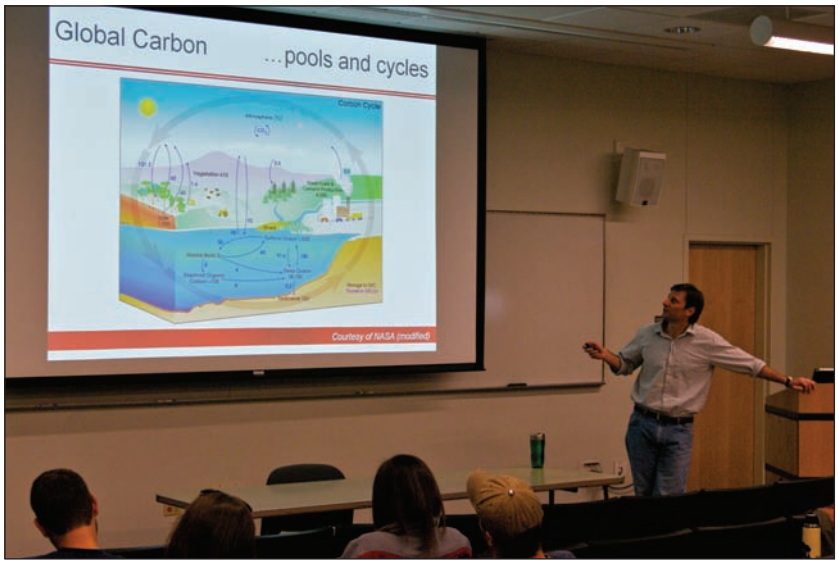

Figure 2—Presentation of climate change science and forest response. Photo credit: Jill C. Witt.

change bring to the process of making silvicultural decisions and devising strategies? Each person was given large sticky notes to write down individual thoughts, and then asked to share these ideas with the group by bringing them forward and posting them onto the whiteboard in the front of the classroom. As facilitators, we then guided the participants through a re-organization of ideas under broad themes that emerged from the group (fig. 3). This exercise actively engaged each participant and facilitated sharing of many individual ideas. The goal was not to seek full agreement or consensus among the group, but rather to generate a list of considerations that would help trigger discussion for the next exercise.

Explore and extend—The second activity involved brainstorming silvicultural strategies that could be used to sustain forests and reach management objectives. Each person was given a set of maps (fig. 4) representing projected change in

Figure 3—Responses to the question for Activity 1: *What new or altered considerations does climate change bring to the process of making silvicultural decisions and devising strategies?* Photo credit: Jill C. Witt.

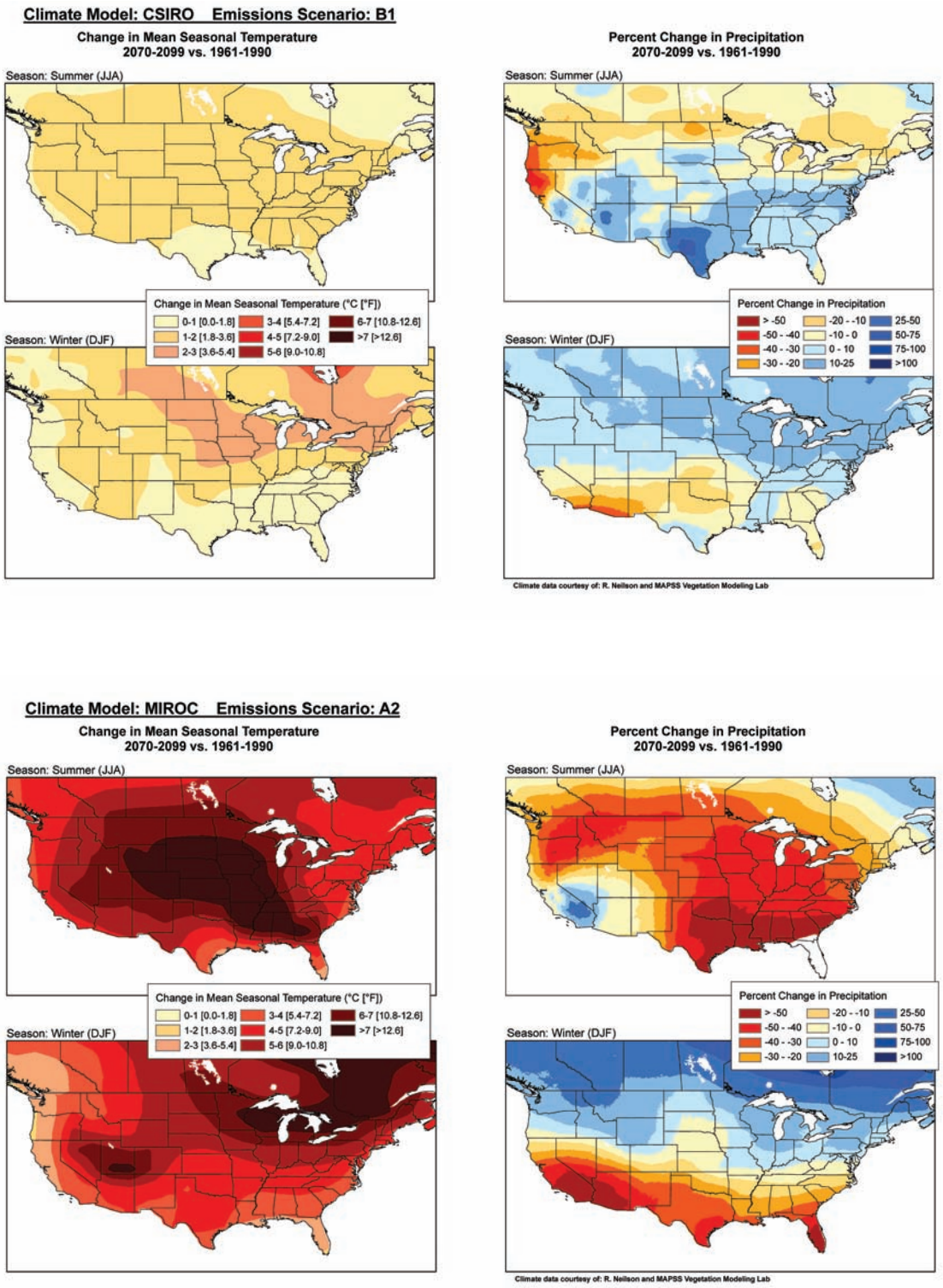

Figure 4—Climate change scenarios used for the activity to develop silvicultural strategies for forest management in the context of an uncertain climate future. The maps represent projected change in temperature and precipitation for the US using two climate models and emissions scenarios. Top: Projections using CSIRO global climate model and low (B1) emissions scenario represent a lesser degree of change in temperature and precipitation at the end of the century. Bottom: Projections using MIROC global climate model and high (A2) emissions scenario represent a greater degree of change in temperature and precipitation at the end of the century. Map data courtesy of R. Neilson and the MAPSS Vegetation Modeling Lab.

precipitation and temperature for summer and winter months at the end of the 21st century across the United States based on different emissions scenarios developed by the IPCC (2007). For simplicity and due to time constraints, two extremes (low model sensitivity and low emissions, high model sensitivity and high emissions) were used to frame the discussion. The class (32 participants) was then divided into five breakout groups by region or location consisting of five to eight people focused on common forest types in order to bring together similar experiences and to facilitate regionally oriented dialog specific to these two extremes.

The groups were instructed to address the following question given the range of uncertainty depicted by the different climate projections: *What silvicultural strategies may be helpful or necessary to sustain our regional forests in the face of climate change?* Each group was given a flipchart, an easel and markers, and the groups were relocated to a large open area conducive to group discussions outside the classroom. Each group chose a note-taker who recorded main points of the discussion on the flipchart. The groups were instructed to summarize five or six key points onto one flipchart page that would then be shared with the class by a volunteer spokesperson.

We acted as facilitators during this process, observing groups during their discussion to work through the overarching question. We deliberately avoided providing feedback or giving direction during these small group sessions. This enabled the participants to independently identify and grapple with complex issues, it encouraged open sharing of ideas, and resulted in the group framing their own problems and solutions. This exercise helped make climate change a real issue by forcing participants to extend and apply what they know conceptually about climate change to localized regions, and further required them to invoke local expertise and experience at the forest- and stand-level in order to develop practical strategies to address the issue.

Explain—The groups reconvened in the main classroom, and volunteer spokespersons shared a summary of the main ideas generated through the small group discussions with the full group. Their summary flipchart pages were posted on the board in the front of the room, and commonalities and unique ideas were identified. Participation from the entire group was encouraged during this process. This exercise allowed for a distillation of many different ideas that were discussed, provided an opportunity for individuals to practice organizing and presenting information, and exposed participants to general management issues from across different regions of the country.

Synthesis and wrap-up—The penultimate stage of the module was a synthesis of the day's activities and a brief description of how this module fit into the larger climate change project. Results from the initial Silviculture Workshop (Janowiak and others, unpublished paper) were shared, highlighting the many similarities to the ideas generated by the NASP participants during the current module. Final wrap-up comments were made emphasizing these broad concepts: 1) we know climate is changing, therefore we must be proactive in our approach to forest management; 2) there is no *single* shiny new silvicultural tool to use against climate change; instead we must be creative in developing silvicultural prescriptions and management approaches that are flexible and adaptive; and 3) continuing to practice sustainable forest management is our best approach toward addressing the uncertainties of climate change and other complex issues.

Evaluate process—We were interested in how effective our approach toward educating and training were for this group, so we implemented a survey at the end of the module to obtain feedback from the participants. We asked for comments on the information presented and activities conducted during the module. Participants were also asked to identify to what extent they had thought about this topic before, to what extent it has been discussed on their district or forest, if they thought they would incorporate ideas related to climate change into their management activities, and if their perceptions have changed during the module. We also asked them to identify the most and least useful parts of the module.

Outcomes

One of our main questions was whether this approach would be effective in helping participants develop silvicultural problem-solving skills to address the issue of climate change. We believe the process used in this module was highly effective at achieving our goals. The high response rate among the group, and the content of those responses on the evaluations, gave us positive evidence that our approach was effective at communicating climate change science and arming participants with a model structure for silvicultural strategy development. Some common themes are described below.

Feedback

Overall, the level of material presented was deemed appropriate for the audience, though many respondents desired more time, more in-depth information, and more activities for this topic within the overall curriculum of the two-week course. The detailed up-to-date information presented at the beginning of the module was often mentioned as a strength because it was clearly articulated with very useful graphics. The hands-on activities were highly rated and thought to be very effective at getting people to really think about the topic. The activities were consistently identified as a major strength of the module.

This reaction is in line with our overall experience with conducting training workshops and seminars: providing a detailed yet concise summary of climate change science is an essential component of educating land managers. But ultimately the most beneficial part of the process is the discussion generated and ideas shared within small groups. Our experience in this module was that the individual groups were reluctant to end their discussions and move to the final synthesis stage back in the classroom. It is clear that both components are necessary, but this finding reinforces the value and effectiveness of interactive teaching strategies that ultimately result in better learning and application of knowledge and skills (McNeal and D'Avanzo 1997).

Answers to the question *"To what extent have you thought about this topic before?"* ranged from "a lot" to "not very much. " Most people indicated they had thought about climate change, and active measures were being taken in some locations. Many faced limitations to incorporating climate change-oriented concepts and approaches into forest management plans from colleagues, and some perceived resistance to the idea across interdisciplinary boundaries as well. Time constraints and pressure to address other issues (i.e., restoration, invasive species) were also cited as major limitations to directly dealing with climate change in everyday activities. Several also mentioned that current forest plans were not developed in ways that allowed management flexibility in the face of climate change.

Conversely, carbon footprint reduction was identified as a positive step many local offices are taking. It was also pointed out that many of the silvicultural strategies currently implemented are geared toward increasing forest health and sustainability, which indirectly addresses many concerns regarding climate change. An overwhelming majority indicated that they would be incorporating ideas related to climate change into future management activities.

There was a mix of "yes" and "no" answers to the question of whether the module changed the respondent's perceptions about climate change. Many people responded that their understanding was enhanced and that they now feel more prepared for dealing with the issue on their forests. Some identified that climate change felt less overwhelming following the module. Though diminishing throughout the day, some skepticism remained within the group, largely surrounding the use of climate change models. There was appreciation for the overall approach in that silvicultural possibilities were elucidated and solutions were not dictated, recognizing that every situation is different. There was also a call for presenting this information to broader audiences.

Summary

As evidenced by the theme of this workshop, climate change is a real issue that must be confronted by land managers, and the USDA Forest Service is in a strategic position of opportunity to lead this charge. Education and training are necessary and pertinent first steps to move forward within the agency and beyond. The training module we described was effective at communicating state-of-the-art climate change science, and the hands-on activities successfully engaged participants into developing silvicultural strategies that will contribute to sustainable forest management in the context of an uncertain climate future. Feedback we received from participants is informative for improving content for future NASP sessions, including building in additional time for the topic of climate change. This ground-up approach to education and problem-solving, and the interactive flow of knowledge between academic, research, and management sectors, may serve as a model for future climate change training programs as well as for programs dealing with other contemporary issues of great importance facing natural resource managers.

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Part 2: Carbon Sequestration and Biomass Utilization

Carbon Concentrations and Carbon Pool Distributions in Dry, Moist, and Cold Mid-Aged Forests of the Rocky Mountains

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Abstract—Although "carbon" management may not be a primary objective in forest management, influencing the distribution, composition, growth, and development of biomass to fulfill multiple objectives is; therefore, given a changing climate, managing carbon could influence future management decisions. Also, typically, the conversion from total biomass to total carbon is 50 percent; however, we believe this value is not consistent across all forest components. Therefore, the objectives of this study are to: acknowledge the appropriate carbon concentrations and distribution of carbon pools and provide improved estimates of carbon content in four habitat types with different climatic regimes—(dry (Arizona), cold (Montana), and moist (Idaho)—of the Rocky Mountains, USA. We quantified biomass, carbon concentrations, and carbon amounts for trees, soils, woody debris, and coarse and fine roots. We found that in most cases our carbon concentrations were less than the typical conversion of 50 percent. Thus we recommend the following conversions from biomass to carbon: trees should be 49 percent for overstory crown, 48 percent for boles, 48 percent for understory trees, and 47 percent for coarse roots; for understory plants concentrations should be 47 percent for shrubs and 41 percent for forbs and grasses; woody residue should be 48 percent for solid logs, 49 percent for rotten logs, 48 percent for brown cubical rotten wood, and 44 percent for buried wood; cones should be 48 percent in ponderosa pine forests and 46 percent in cold and moist forests; sticks in ponderosa pine forests should be 49 percent and in the moist and cold climate regimes sticks should be 47 percent. Unique carbon pools often overlooked include cones, woody debris, and buried wood. Given these results, additional research questions could be pursued, such as the effect of successional stage on carbon pool distributions, or as forests grow and develop, if carbon concentrations change or if only biomass distribution changes over time.

Introduction

Forest plans and prescriptions on public lands emphasize a variety of values, such as biological diversity, scenery, wildlife, water quality, sustainable ecosystems, and other values, in addition to commodity production. Past forest practices consisted of managing individual stands of trees (Graham 1990) as separate entities; today managers need to consider overall ecosystem processes and functions before developing management prescriptions of large landscapes (Jain and Graham 2005), particularly with the uncertainty associated with climate change (Joyce and others 2008). In addition, because management actions have the potential to manipulate carbon, acknowledging changes in carbon pools may be a critical element that will need documentation in the future (Waring and Schlesinger 1985).

In forest ecosystems, organic carbon is stored in different locations, including live and dead standing biomass, down woody debris, litter, and soils. Thus

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the manipulation of these organic substances not only affects carbon storage but also other essential nutrients such as nitrogen, calcium, potassium, sulfur, and phosphorous (Binkley and Richter 1987; Jorgensen and Wells 1986). Therefore, recognizing the role of carbon and organic matter in the structure and function of forest ecosystems is essential for sustaining long- and short-term forest productivity.

Although a large portion of carbon is in live biomass, a significant amount of carbon is also stored in coarse woody debris (CWD), the forest floor, and soils. The forest floor and soils contain five organic components that contribute to carbon storage (fig. 1): 1) litter, which encompasses recognizable plant and animal materials such as conifer needles, insect frass, and deciduous leaves; 2) humus, which is unrecognizable, decomposed plant and animal material having a high content of complex hydrocarbons located above the mineral soil; 3) brown cubical rotten wood (BCR), which consists of woody debris in an advanced state of decay located on the surface; 4) soil wood, which is decaying wood incorporated within the mineral layers; and 5) mineral soil organic matter, which is organic matter incorporated in the mineral soil (Aber and Melillo 1991; Harmon and others 1986; Harvey and others 1987; Waring and Schlesinger 1985). The dead organic matter components of forests represent different substrate qualities, including sizes and state of decomposition; thus each has its own unique carbon pools.

Because the type of vegetation influences the kinds of carbon compounds present, carbon pools vary depending on forest type. This, combined with the

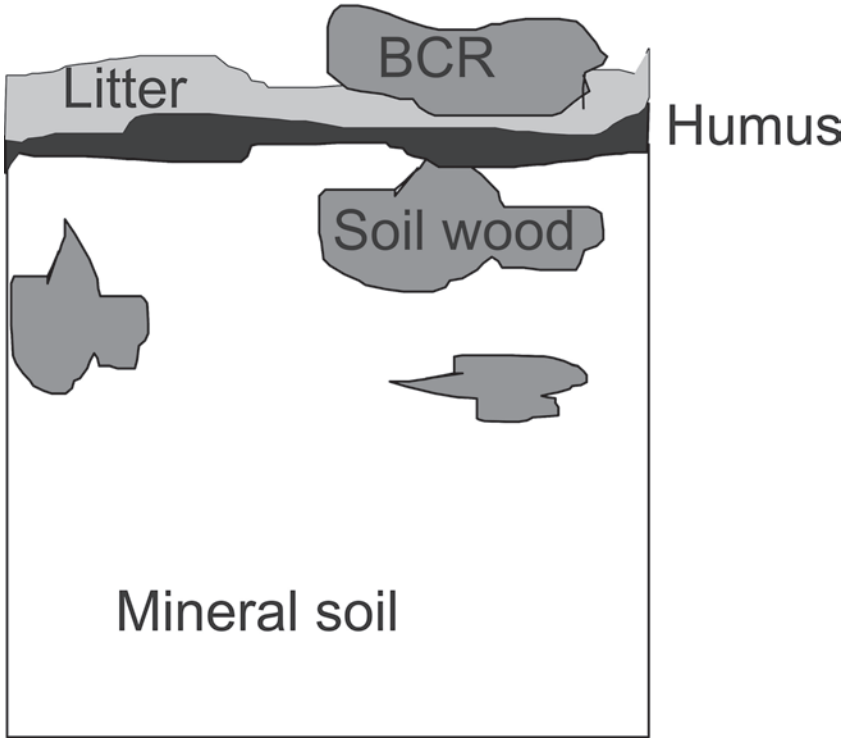

Figure 1—The forest floor and mineral soils contain five organic elements: litter, humus, brown cubical rotten wood (BCR), soil wood, and organic matter in the mineral soil. All these elements contribute to storing carbon. The difference between BCR and soil wood is the location of the material; soil wood is buried, often below the humus and litter, while BCR is on the surface. Soil organic matter typically decreases with depth (Woods 1989).

local climate, subsequently affects decomposition rates. For example, Douglas-fir (*Pseudotsuga menziesii* (Mirb.) Franco) decays more slowly than most conifers because the heartwood contains fungi-toxic compounds and high amounts of lignin (Scheffer and Cowling 1966). Therefore, if all other factors controlling decomposition were similar, a Douglas-fir forest may store more carbon in CWD, BCR, and soil wood than a true fir (*Abies* spp.) forest. In turn, the amount of CWD created within a forest type also affects soil wood amounts, which is incorporated into soil mineral layers through freeze-thaw action, soil mixing, and erosion (Harvey and others 1987). For example, on moist forests the accumulation of CWD and soils wood is much greater than dry forests in the southwestern United States (Graham and others 1994).

Carbon Estimates

There is wide variation in carbon storage among and within forest ecosystems. In forests of the Lake States, (Minnesota, Wisconsin, and Michigan), Grigal and Ohmann (1992) concluded that both stand age or successional stage and forest type influence the amount of carbon stored in the forest floor. They found that carbon continued to accumulate over time because in these ecosystems biomass was produced more rapidly than it decomposed. Other research has also indicated that forest type may affect carbon storage but only if ecosystems were significantly different (Post and others 1982). However, Grigal and Ohmann (1992) determined that wide variations in forest type were not necessary to notice subtle differences in carbon storage.

The role of CWD, BCR, and soil wood in storing carbon is often overlooked because most estimates consider only living biomass, forest floor (litter and surface humus), and mineral soil (Buringh 1984; Eswaran and others 1993; Franzmeier and others 1985; Huntington and others 1988; Post and others 1990; Schlesinger 1977). Studies have compared carbon storage in CWD between different forests (Harmon and Hua 1991; Keenan and others 1993). The results of these studies indicate that a large fraction of the terrestrial sink could potentially be located in woody debris. For example, Keenan and others (1993) reported that 60 percent of the forest floor in northern Vancouver Island was composed of woody material. In the Northern Rocky Mountains, up to 58 percent of the organic components can consist of CWD and soil wood (Harvey and others 1987).

To estimate carbon storage in vegetation, the amount of carbon is estimated to be 50 percent of the biomass (Grigal and Ohmann 1992; Hendrickson 1990; Lamlom and Savidge 2003; Linder and Axelsson 1982). Using this ratio assumes that all organic biomass has the same carbon concentration across different vegetation types and species. Although this is the best and most popular information currently available for estimating carbon, we hypothesize that ratios should differ among and between vegetation types.

Because estimating carbon storage is a key element in predicting the effects of climate change and determining carbon pools, it is important that valid conversion factors be used to minimize the amount of error these estimates may provide. Moreover, knowing where carbon is stored is important across vegetation types within the Rocky Mountains. Therefore, the objectives of this study are to acknowledge the appropriate carbon concentrations and distribution of carbon pools and provide improved estimates of carbon content in three forests types with different climatic regimes (dry, cold, and moist) of the Rocky Mountains. Although carbon management may not be a primary objective in forest management, knowing the changes and distribution of carbon pools may potentially influence management decisions in a future with climate change.

Methods

Site Selection

The sites selected for the study (fig. 2) include three climatic regimes: cool-wet, cold-dry, and warm-dry. The habitat types chosen to represent each of these regimes were selected after consultation with soil scientists, silviculturists, and forest managers. The wettest and most productive site was a western hemlock/queen cup beadlily (*Tsuga heterophylla* (Raf.) Sarg.)/ (*Clintonia uniflora* (Schult.) Kunth) (WH/CLUN) habitat type (Cooper and others 1991) on the Priest River Experimental Forest in northern Idaho (sites 1-3). A cold-dry subalpine fir/dwarf huckleberry (*Abies lasiocarpa* (Hook.)Nutt.)/(*Vaccinium scoparium* Leib.) (SAF/VASC) habitat type (Pfister and others 1977) was located on the Deerlodge National Forest near Butte, Montana (sites 4-6). Two warm-dry sites were selected in northern Arizona: a ponderosa pine (*Pinus ponderosa* C. Lawson)/gambel

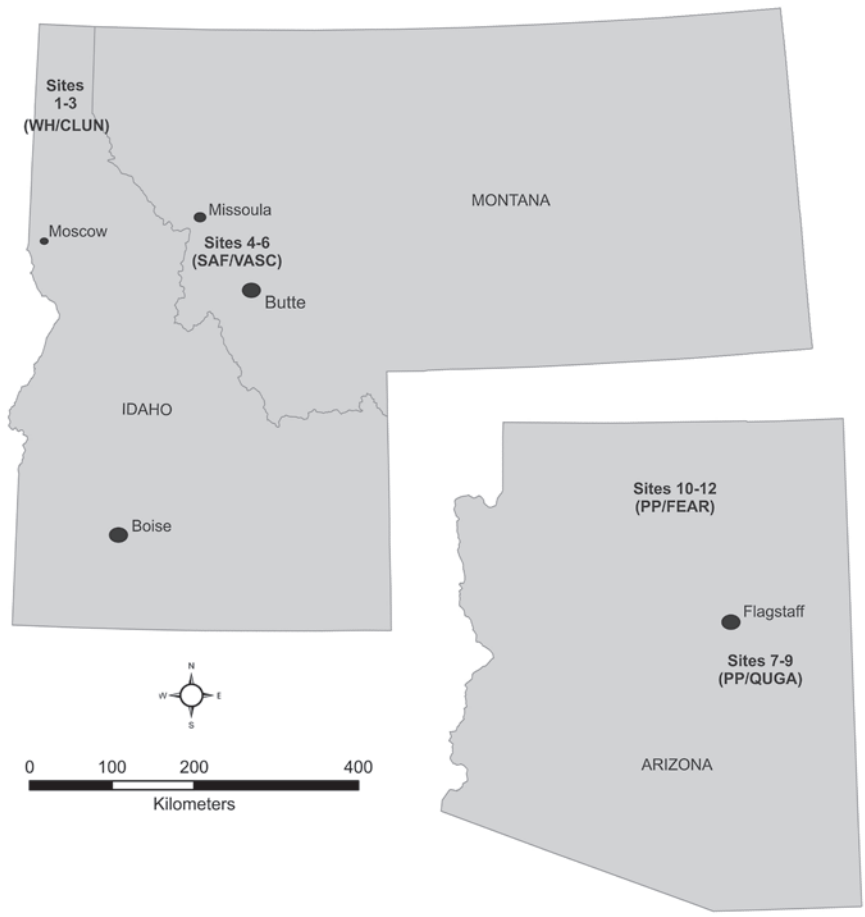

Figure 2—The general locations of study areas. Study sites 1-3 are located in northern Idaho within the western hemlock (*Tsuga heterophylla* (Raf.) Sarg./queencup beadlily (*Clintonia uniflora* (Schult.) Kunth) (WH/CLUN) habitat type (Cooper and others 1991). Study sites 4-6 are located in western Montana within the subalpine fir (*Abies lasiocarpa* (Hook.) Nutt.)/dwarf huckleberry (*Vaccinium scoparium* Leib.) (SAF/VASC) habitat type (Pfister and others 1977). Study sites 7-12 are located in northern Arizona: 7-9 are located within the ponderosa pine (*Pinus ponderosa* C. Larson)/gambel oak (*Quercus gambelii* Nutt.) (PP/QUGA) (Larson and Moir 1986) and 10-12 are located within the ponderosa pine (*Pinus ponderosa* Dougi. ex Lawsi/Arizona fescue (*Festuca arizonica* Vasey) (PP/FEAR). Please refer to table 1 for specific characteristics of each site.

oak (*Quercus gambelii* Nutt.) (PP/QUGA) habitat type on the Coconino National Forest (Larson and Moir 1986) (sites 7-9) and a ponderosa pine/Arizona fescue (*Festuca arizonica* Vasey) (PP/FEAR) habitat type on the Kaibab National Forest (sites 10-12).

The WH/CLUN habitat type (Cooper and others 1991) occurs at elevations from 760 to 1,580 m (2,500 to 5,200 ft). Parent material is an ash cap over belt metasedimentary rocks (Alt and Hyndman 1989). Tree species include Douglas-fir, western larch (*Larix occidentalis* Nutt), western white pine (*Pinus monticola* Dougl. ex D. Don.), lodgepole pine (*Pinus contorta* Dougl. ax Loud.), grand fir (*Abies grandis* Dougl. ex D. Don) Lindl.), subalpine fir, Engelmann spruce (*Picea engelmannii* Parry ex Engelm.), western redcedar (*Thuja plicata* Donn ex D. Don), and western hemlock. The overstory canopy of late seral stands is usually dense with a sparse herbaceous layer. WH/CLUN climate is characterized by dry summers with the majority of precipitation occurring during the fall and winter. Total precipitation averages between 710 to 1,520 mm (28 to 60 inches); snowfall averages 262 cm (103 inches). Average annual air temperature ranges from 4 to 10 °C (40 to 50 °F) (Graham 1990).

SAF/VASC is one of the most abundant habitat types east of the Continental Divide in Montana. Elevations range from 2,130 to 2,590 m (7,000 to 8,500 ft). The parent material of the study site is volcanic (Hunt 1972). The overstory in the sites for this study are dominated by lodgepole pine, with a minor component of Engelmann spruce, subalpine fir, and Douglas-fir. The understory is carpeted with dwarf huckleberry, scattered common juniper (*Juniperus communis* Pall.) and a minor component of pine grass (*Calamagrostis rubescens* Buckl.). Precipitation ranges from 280 to 740 mm (11 to 29 inches), with snowfall averaging 686 cm (270 inches). Average annual air temperature ranges from –4 to 2 °C (25 to 35 °F) (Alexander and others 1990; Pfister and others 1977).

The PP/QUGA habitat type occurs at elevations from 1,860 to 2,590 m (6,100 to 8,500 ft) with basalt parent material. The overstory consists of ponderosa pine with a minor component of gambel oak. Understory vegetation includes rose (*Rosa* spp.), skunk bush (*Rhus trilobata* Nutt.), New Mexico locust (*Robinia neomexicana* A. Gray), muttongrass (*Pea fendleriana* (Steud.) Vasey), and mountain muhly (*Muhlenbergia montana* Nutt.). PP/QUGA climate is similar to PP/FEAR (described below) but unlike the Kaibab Plateau, the majority of the precipitation falls during July through October (Brewer and others 1991; Larson and Moir 1986). Sites 7 through 9 were located on the Coconino National Forest in Arizona (table 1, fig. 2).

The PP/FEAR habitat type occurs at elevations from 2,300 to 2,500 m (7,540 to 8,200 ft) in northern Arizona. The parent material of the study site is limestone (Hunt 1972). The overstory consists of ponderosa pine with a small amount of quaking aspen (*Populus tremuloides* Michx.). Understory vegetation includes Arizona fescue, Oregon grape (*Berberis repens* Lindl.), Fendler's ceanothus (*Ceanothus fendleri* A. Gray), wax gooseberry (*Ribes cereum* Lindl.), mountain muhly, and muttongrass. Precipitation has a bimodal distribution with one wet season occurring July through October and another December through March. However, on the Kaibab plateau, greater than 50 percent of the precipitation falls between December and March. Total precipitation ranges from 520 to 600 mm (20 to 24 inches), with snowfall averaging 1,120 mm (47 inches). Mean annual air temperature ranges from 4 to 6 °C (39 to 43 °F) (Brewer and others 1991; Larson and Moir 1986).

Although we recognize that successional stage and/or stand age may influence the amount and distribution of carbon pools, our objective was to determine if the ratios and carbon pool location varied across different habitat types. To accomplish this we acquired a list, within each habitat and soil type, from forest

Table 1—Description of selected stands within each habitat type. Refer to figure 2 for study site locations.

Cover type-study site	Age	Aspect (°)	Slope (%)	Elevation (m)	Parent material[b]
The WH/CLUN[a] on the Idaho Panhandle National Forest—Priest Lake Ranger District (Priest River Experimental Forest)					
WH/DF/WL/WP-1[c]	100	310	45	1280	Ash/Belt
WH/DF/WL/WP-2	100	340	45	1340	Ash/Belt
WH/DF/WL/WP-3	100	340	45	1340	Ash/Belt
The SAF/VASC[a] on the Deerlodge National Forest—Butte Ranger District					
LP-4[c]	65	124	21	2073	Volcanic
LP-5	65	124	21	2073	Volcanic
LP-6	65	110	33	2073	Volcanic
The PP/QUGA[a] on the Coconino National Forest—Mormon Lake Ranger District					
PP-7[c]	141	0	0	2134	Basalt
PP-8	150	0	0	2134	Basalt
PP-9	145	0	0	2134	Basalt
The PP/FEAR[a] on the Kaibab National Forest—North Kaibab Ranger District					
PP-10[c]	127	0	0	2470	Limestone
PP-11	125	0	0	2470	Limestone
PP-12	123	0	0	2487	Limestone

[a] Habitat types and species for cover types: In northern Idaho (Cooper and others 1991) WH/CLUN = western hemlock (*Tsuga heterophylla* (Raf.) Sarg./queencup beadlily (*Clintonia uniflora* (Schult.) Kunth). In western Montana (Pfister and others 1977) SAF/VASC = subalpine fir (*Abies lasiocarpa* (Hook.) Nutt.)/dwarf huckleberry (*Vaccinium scoparium* Leib.). In northern Arizona (Larson and Moir 1986) PP/FEAR = ponderosa pine (*Pinus ponderosa* C. Larson/Arizona fescue (*Festuca arizonica* (Vasey) and PP/QUGA = ponderosa pine/gambel oak (*Quercus gambelii* Nutt.).

[b] Parent materials are (Alt and Hyndman 1989; Hunt 1972) **Ash**, fine shreds of lava blown from Mount Mazama; **Belt**, mildly metamorphosed sedimentary rocks, including argillites, siltites, quartzites, and dolomites; **Volcanic (Rhyolite)**, lava or shallow intrusion, fine grained, with composition equivalent to granite. **Basalt**: Black volcanic rock rich in iron, calcium, and magnesium, composed primarily of plagioclase; and **Limestone**, sedimentary rock or surface deposit of calcium carbonate.

[c] Species are WH = western hemlock, DF = Douglas fir (*Pseudotsuga menziesii* (Mirb.) Franco), WL = western larch (*Larix occidentalis* Nutt), WP = western white pine (*Pinus monticola* Dougl. ex D. Don.), LP = lodgepole pine (*Pinus contorta* Dougl. ax Loud.), and PP = ponderosa pine. The number following species cover type refers to the site number located on figure 2.

silviculturists and soil scientists of undisturbed stands consisting of mid-to late seral vegetation. From each list, three sites were randomly selected and then verified (table 1).

Data Collection

Twelve points were systematically located on a random transect bisecting the site. From these points, forest components and data for biomass estimates were sampled using five plot types: 1) variable, 2) fixed, 3) microsite, 4) soil core, and 5) line intersect (table 2). A variable plot using probability proportional to size was used to sample total height and d.b.h. (diameter at 4.5 ft; 1.4 m) for trees ≥12.7 cm (5 inches) d.b.h. Sapwood, heartwood, coarse roots, and overstory crown samples were collected for carbon analysis from each tree species. Increment cores at d.b.h. were used to sample sapwood and heartwood. Coarse roots (>1 cm; 0.5 inches diameter) were sampled 20 to 25 cm (7 to 10 inches) below the soil surface on the down-hill side of the tree. A sub-sample of overstory crown (branches and needles) was collected from three trees per species. For consistency, crown samples were removed from the third highest whorl and from the north side of the tree.

The second plot type was a 13.5 m^2 (1/300 acre) fixed-area circular plot (table 2). Trees <12.7 cm (5 inches) d.b.h. occurring on this plot were tallied and their heights were measured; for trees ≥1.4 m (4.5 ft) tall, d.b.h was measured, while basal diameter was measured on trees <1.4 m (4.5 ft) tall. In addition, foliage samples for carbon analysis were taken from the understory trees. Average basal diameter and number of stems occurring on the plot were also recorded for the following shrub

Table 2—Forest components and plot types where measurements and samples were obtained.

	Plot type[a]				
Forest component	**Variable**	**Fixed**	**Microsite**	**Line intersect**	**Soil core**
Trees					
Overstory crown	X				
Sapwood[b]	X				
Heartwood[b]	X				
Coarse roots (≥1 cm diameter)	X				
Fine roots (<1 cm diameter)					X
Understory crown (<12.7 cm tall)		X			
Understory vegetation[c]					
Small shrubs (<0.5 cm)			X		
Medium shrubs (0.5 to 2.0 cm)		X			
Forbs and grasses			X		
Woody residue[d]					
Sticks (<7.5 cm diameter)				X	
Solid logs (≥7.5 cm diameter)				X	
Rotten logs (≥7.5 cm diameter)				X	
Cones			X		
Organic soil					
Brown cubical rotten wood			X		X
Litter			X		X
Humus			X		X
Soil wood			X		X
Mineral soil[e]					
Shallow mineral (0-10 cm)					X
Deep mineral (>10, up to 30 cm)					X

[a] Plot types: 1) variable, based on probability, proportional to tree diameter. On the Idaho Panhandle, Kaibab, and Coconino National Forests, we used a 40 basal area factor (BAF); on the Deerlodge we used a 20 BAF; 2) fixed, a 0.0013 ha (24th acre) circular plot; 3) microsite, a 30 cm (12 inches) circular plot; 4) transect, 15.2 m (50 ft) in length (Brown 1974); and 5) soil core, l0 by 30 cm (4 by 12 inches).

[b] Sampled using increment cores at 1.4 m (4.5 ft).

[c] Shrub size classes are from Brown (1976).

[d] Size classes are from Brown (1974).

[e] Soil depths are from Jurgensen and others (1977).

size-classes from Brown (1976): low (0 to 0.5 cm; 0 to 0.2 inches) and medium (0.51 to 2.0 cm; 0.2 to 0.75 inches); tall shrubs (2.01 to 5.0 cm; 0.76 to 2 inches) were not present in any of the plots. When present, foliage samples for carbon analysis were taken for medium shrubs. The third plot type (microsite) was a 30 cm (12 inches) diameter hoop (table 2). All cones and organic soil components were collected for carbon analysis on this plot. Next to the microsite plot, a fourth type of plot consisted of extracting a 10 by 30 cm (4 by 12 inches) soil core. Litter, humus, soilwood (rotten wood buried in established humus or mineral soil horizon), and BCR were separated. Mineral soil was separated into two depths: shallow mineral (0 to 10 cm depth) and deep mineral (10-30 cm depth). Depth of each horizon was recorded and the entire sample was taken from the core from each horizon and placed in a cloth bag. Mineral soil bulk density was sampled at 0 to 10 cm (0 to 4 inches) and 10 to 20 cm (4 to 8 inches) depths using a 135.7 cm^3 (8.4 in^3) core sampler. Samples of BCR, litter, humus, soil wood, and mineral soils at two depths were collected for carbon analysis.

The fifth type of plot was a line intersect to determine woody residue biomass (Brown 1974) (table 2). Two 7.5 m (25 ft) transects were located in random directions from plot center. Woody residues were separated into stick (<7.5 cm; 3 inches), and solid and rotten logs ≥7.5 cm (3 inches). For carbon analysis, a sample was taken from each residue class.

Biomass Estimates

The Forest Vegetation Simulator (FVS) (Dixon 2002; Wykoff and others 1982) and field data were used to estimate tree biomass (tables 3 and 4). FVS provided a list that included total cubic foot volume, species, diameter, height, and number of trees per unit area for each sample tree represented. Published regression equations (tables 3 and 4) in combination with this tree list, provided estimates on crown, bole, bark, and coarse root weight (Baskerville 1965; Brown 1978; Feller 1992; Johnstone 1971; Kuiper and Coutts 1992; Whittaker and others 1974; Will 1966, cited in Santantonio 1977).

To estimate shrub biomass, we used regression equations (table 4) for the basal stem diameter-based size-classes (Brown 1976) described above. If grasses covered more than 10 percent of the site, their biomass was estimated using overstory basal area (Covington and Fox 1991) (table 4). Weight estimates of sticks and logs (solid and rotten) were determined using Brown's (1974) down woody debris transect methods. Cone and soil biomass were estimated directly from field sampling. Mineral soil biomass was estimated using core volume, percent coarse fragments, and bulk density. Oven-dry (60° C; 140° F for 12 hours) weights of cones, organic components, and fine roots were expanded to a per unit area basis.

Laboratory Analysis

Field collections were taken to the Forestry Sciences Laboratory in Moscow, Idaho, and prepared for carbon and organic matter analysis. Soils were oven-dried and sieved using screens with 2 mm (0.08 inches) openings. Roots were removed

Table 3—Regression equations used for estimating biomass for components of trees ≥5 cm (2 inches). Columns A, B, and C show coefficient values for specific species.

		Bole and bark[c]			Coarse roots		
		Wt (lb) =VolA +VolBC			Log_{10}WT (kg) =B(log_{10}DBH(cm) +Log_{10}A		
Species[a]	Crown[b]	A	B	C	B	Log_{10}A	Reference
PP	exp[0.2680+2.0740(lnd)]	25.0	0.24	21.8	2.445	–0.94	Will 1966[d]
ES	exp[l.0404+l.7096(lnd)]	21.8	0.19	30.6	2.151	–1.24	Whittaker and others 1974
GF	exp[l.3094+l.6076(lnd)]	23.1	0.20	37.4	2.445	–1.71	Baskerville 1965
SAF	7.345+1.255(diameter^2)	20.0	0.19	27.4	2.445	–1.71	Baskerville 1965
					Log_{l0}Wt (lb) =Blog_{10}DBH^2(in) Ht(ft) +log_{10}A		
WWP	exp(0.7276+l.5497(lnd)]	23.7	0.21	26.2	1.022	1.879	Johnstone 1971
WL	exp[0.4373+l.6786) (lnd)]	32.4	0.24	24.3	1.022	1.879	Johnstone 1971
LP	exp[0.1224+l.8820(lnd)]	25.6	0.11	26.5	1.022	1.879	Johnstone 1971
					Wt(kg)=A(DBH) (cm)		
					A	B	
DF <43 cm	exp[l.368+l.5819(lnd)]	30.0	0.19	27.4	0.01	2.630	Kuiper and Coutts 1992
DF ≥43 cm	1.0237(diameter^2) - 20.74						
					lnWt(kg)=B+Aln(DBH) (cm)		
WH	exp[0.72l8+l.7502(lnd)]	28.1	0.19	31.2	–4.159	2.519	Feller 1992
WRC	exp(0.88l5+1.6389(lnd)]	20.0	0.15	23.1	–4.159	2.519	Feller 1992

[a] Tree species: PP-ponderosa pine (*Pinus ponderosa* Dougl. ex Laws.); ES-Engelmann spruce (*Picea engelmannii* Parry ex Engelm.); GF-grand fir (*Abies grandis* Dougl. cx D. Don); SAF-subalpine fir (*Abies lasiocarpa* (Dougl. ex D. Don) Lindl.; WWP-western white pine (*Pinus monticola* Dougl. ex D. Don); WL-western larch (*Larix occidentalis* Butt.); LP-lodgepole pine (*Pinus contorta* Dougl. ex Loud.); DF-Douglas-fir (*Pseudotsuga menziesii* (Mirb.) Franco); WH-western hemlock (*Tsuga heterophylla* (Raf.) Sarg.); WRC-western red cedar (*Thuja plicata* Dorm ex D. Don).

[b] Ind = natural log diameter in inches (Brown 1978); d^2 = diameter^2.

[c] Wt = Weight; Vol = ft^3 ; A = specific gravity of wood; B = percentage of bark; C = specific gravity of bark.

[d] Cited in Santantonio and others 1977.

Table 4—Regression equations used to estimate forb, grass, shrub, and tree crown weight for trees less than 5 cm (2 inches). Columns A and B show coefficient values for specific species.

Species[a]	Crown weight[b]		Reference
Douglas-fir	exp[–4.212 + 2.7168 (lnht]		Brown 1978
Lodgepole pine	exp[0.0311(height^2)		Brown 1978
Western larch	exp(–5.126 + 2.5639(lnht)]		Brown 1978
Grand fir	0.0538 (height^2)		Brown 1978
Western redcedar	0.0307 (height^2)		Brown 1978
Ponderosa pine	exp[–2.7297 + 1.1707 (lnht)]		Brown 1978
	Shrubs, forbs, and grasses		
	lnWt(g)= A+ Bln basal stem diameter (cm)		Brown 1976
	A	B	
Vaccinium scoparium	2.113	2.148	
Juniperus communis	4.081	2.202	
Combined low shrub	3.565	3.565	
Forbs and grasses[c]	1nWt(kg/ha)=2.517 – 0.294 (10^{-1})BA[c]		Covington & Fox 1991

[a] Refer to table 3 for scientific names of species.

[b] lnht=natural log of height (ft)

[c] BA is basal area in m^2/ha

from the soils and coarse fragments greater than 2 mm (0.08 inches) diameter were weighed. The twelve samples from the litter, humus, BCR, soil wood, and mineral soils were each combined into four composites for each stand. Each composite was composed of three adjacent samples that were collected along the transect line. Similar to the soils, 12 samples were collected from the other forest components, were oven-dried and ground, and then placed into three composites. Before conducting any laboratory analyses, mineral soils were tested for carbonates using 10 percent hydrochloric acid (Soil Survey Staff 1992). The LECO Carbon, Hydrogen, Nitrogen (CHN-600) Autoanalyzer was used to determine the organic carbon concentration of soil and forest components.

Statistical Analysis

Analysis of variance for a randomized complete block design was used to analyze the data. Within each habitat type, a variety of data were collected from three individual randomly selected stands. These stands within each habitat type were used as replications in the analysis of variance (Steel and Torrie 1960). Mean values for each variable were computed for the stand prior to analysis. Therefore, the four habitat types described above served as the main effects in three replications (stands). Scheffé's (1953) S test was used to separate main effect means when more than two means were compared. Carbon proportional data (percent) were transformed using the square root of the arcsine prior to performing the analysis of variance. The analyses were conducted at P-level of $\leq$0.05.

Results

Organic Biomass

In all habitat types, outside the mineral soil, the highest amount of biomass occurred in tree boles and the least amount occurred in fine roots (table 5), while coarse roots contributed the second largest amount. Among habitat types, PP/FEAR and PP/QUGA tree bole biomass was greater than other habitat types

Table 5—Forest component biomass (Mg/ha) estimates (mean) and standard error ($S_{\bar{x}}$) by forest habitat type. The different letters (x, y, or z) indicate significant differences; if letters are the same there were no significant difference among the habitat types. Refer to table 1 for habitat type designation.

	Habitat type							
	PP/FEAR		PP/QUGA		WH/CLUN		SAF/VASC	
Forest component	Mg/ha	$S_{\bar{x}}$	Mg/ha	$S_{\bar{x}}$	Mg/ha	$S_{\bar{x}}$	Mg/ha	$S_{\bar{x}}$
Trees								
Overstory crown	99	7	113	10	51	3	31	4
Bole	415	16	388	46	268	21	160	31
Coarse roots (≥1 cm diameter)	215	10	252	22	84	11	68	13
Fine roots (<1 cm diameter)	0.l	0.1	0.3	0.1	9	2	5	4
Understory trees (<12.7 cm tall)	1	0.6	1	0.5	0.3	0.2	1	0.2
	x		x		y		y	
Total	730.1	33.7	754.3	78.6	412.3	37.2	265.3	52.2
Understory vegetation								
Medium shrub (0.5-2.0 cm)	—	—	—	—	—	—	9	4
Low shrub (<0.5 cm)	—	—	0.l	0.1	—	—	32	5
Forbs and grasses	0.1	0.01	0.3	0.2	—	—	—	—
	y		y				x	
Total	0.1	0.01	0.4	0.2	—	—	41	9
Woody residue								
Sticks (<7.5 cm diameter)	2	1	5	1	11	0.3	3	0.4
Solid Log (≥7.5 cm diameter)	4	3	0.l	0.1	42	10	1	1
Rotten log (≥7.5 cm diameter)	2	1	4	1	47	6	8	3
Cones	23	8	27	7	9	2	7	3
	y		y		x		y	
Total	31.4	13	36.1	9.1	109	18.3	19	7.4
Organic soil								
Brown cubicle rot	0.4	0.2	3	1	3	3	10	5
Litter	33	6	63	4	33	16	40	11
Humus	65	14	87	8	127	29	98	22
Soil wood	1	0.4	18	8	103	65	36	18
	x		x		x		x	
Total	99.4	20.6	171	21	266	113	184	56
Mineral soil								
	x		x		z		y	
Shallow mineral (0-10 cm)	1198	16	1170	52	512	48	756	41
	X		x		Y		y	
Deep mineral (>10, up to cm)	2992	18	2757	45	1306	121	1637	20

(table 5) and coarse root biomass was significantly greater (>215 Mg/ha; 100 tons/acre) than the other habitat types (<85 Mg/ha; 40 tons/acre). However, fine root biomass was greater in the WH/CLUN and SAF/VASC habitat types.

In addition to tree components, table 5 shows biomass estimates for understory vegetation, woody residue, organic soil, and mineral soil. With the exception of SAF/VASC, where shrubs were a significant component (>40 Mg/ha; 18 tons/acre), understory vegetation did not contribute large amounts of biomass. PP/QUGA had only 0.1 Mg/ha (0.05 tons/acre), while WH/CLUN did not have any. Among habitat types, there were significant differences in biomass across woody debris classes. For example, solid and rotten logs contributed most of the woody residue biomass for WH/CLUN, while cones contributed significantly in the ponderosa pine habitat types. Mineral soil had the greatest total weight, with deep mineral having more mass than shallow mineral.

Several tree species contributed to total tree biomass (no table shown). Eight species occurred in the WH/CLUN habitat type, with the majority consisting of western redcedar (134.5 Mg/ha; 62 tons/acre) and western hemlock (129.1 Mg/ha; 60 tons/acre), along with a minor component of western white pine (17.5 Mg/ha; 8 tons/acre) and subalpine fir (4.7 Mg/ha; 2 tons/acre). Tree biomass in the ponderosa pine habitat types consisted of only ponderosa pine and for SAF/VASC, only lodgepole pine.

Carbon Concentrations

Carbon concentrations were compared among forest components and habitat types (table 6). In all habitat types, carbon concentrations in fine and coarse roots were lower (44 percent to 47 percent) than the concentrations in overstory crown, bole, and understory trees (47 percent to 50 percent). Some of the lowest carbon concentrations (32 percent to 42 percent) occurred in the forbs and grasses component of understory vegetation. Among habitat types, the only significant differences occurred in the understory trees, where WH/CLUN had a significantly lower carbon concentration (47 percent) than the understory trees in both the ponderosa pine (50 percent) and SAF/VASC (49 percent) habitat types.

For woody residue, among the habitat types, solid log carbon concentrations were significantly lower in the SAF/VASC habitat type than the other habitat types; however, this was not statistically significant (table 7). We also had non-statistically significant results concerning rotten logs. However, those created from ponderosa pine had some of the lowest carbon concentrations (46 percent) compared to the other habitat types. Sticks in the ponderosa pine habitat types had significantly more carbon (49 percent) than sticks in the WH/CLUN (47 percent) and SAF/VASC (48 percent) habitat types. A similar trend occurred with cones, with 49 percent for ponderosa pine habitat types, 46 percent for WH/CLUN, and 47 percent for SAF/VASC.

Table 6—Carbon concentrations (%) are for vegetation by habitat type with the mean and standard error ($S_{\bar{x}}$). Significant differences among the means across habitat types are presented as x, y, and z located above the value. Significant differences among the means across different forest components within a habitat type are presented as “a” and “b” located next to the value. If the letter is the same no significant differences were identified. For habitat type designation, refer to table 1.

	PP/FEAR		PP/QUGA		WH/CLUN		SAF/VASC	
Forest component	**%**	**$S_{\bar{x}}$**	**%**	**$S_{\bar{x}}$**	**%**	**$S_{\bar{x}}$**	**%**	**$S_{\bar{x}}$**
			Trees					
Overstory crown	x 49.9 a	0.2	x 49.9 a	0.2	x 49.2 a	0.2	x 49.4 a	0.1
Bole	xy 49.0 a	0.4	x 49.5 a	0.2	yz 47.3 a	0.2	z 47.2 b	0.4
Coarse roots (≥1 cm diameter)	x 47.2 ab	0.7	x 48.6 a	0.6	x 46.8 a	0.3	x 47.l b	0.3
Fine roots (<1 cm diameter)	x 44.0 b	1.2	x 46.2 b	0.4	x 46.1 a	0.4	x 44.9 c	0.4
Understory trees (<12.7 cm dbh)	X 49.9 a	0.03	x 49.8 a	0.1	y 47.3 a	0.2	x 49.l ab	0.2
			Understory vegetation[a]					
Medium shrub (0.5-2.0 cm basal diameter)	—	—	—	—	—	—	48.3 a	0.2
Low shrub (<0.5 cm basal diameter)	x 47.l a	0.6	x 46.7 a	0.5	x 45.5 a	0.5	x 47.8 a	0.2
Forbs and grasses	x 42.7 b	0.4	x 41.5 b	0.6	x 41.6 b	1.3	x 42.0 b	0.4

[a]Shrub size classes are from Brown (1976).

Table 7—Carbon concentrations (%) for vegetation by habitat type with the mean and standard error ($S_{\bar{x}}$). Refer to table 1 for habitat type designation. Significant differences among the means across habitat types are presented as x, y, and z located above the value. Significant differences among the means across different forest components within a habitat type are presented as "a" and "b" located next to the value. If the letter is the same no significant differences were identified.

	PP/FEAR		PP/QUGA		WH/CLUN		SAF/VASC	
Forest component	**%**	**$S_{\bar{x}}$**	**%**	**$S_{\bar{x}}$**	**%**	**$S_{\bar{x}}$**	**%**	**$S_{\bar{x}}$**
			Woody Residue					
Sticks (<7.5 cm diam.)	x 49.l ab	0.3	x 49.0 a	0.1	y 46.9 a	0.3	y 47.6 ab	0.2
Solid log (≥7.5 cm diam.)	x 50.5 a	1.5	x 49.5 a	1.2	x 47.0 a	0.2	x 46.0 bc	0.1
Rotten log (≥7.5 cm diam.)	x 46.3 ab	0.9	x 50.8 a	1.6	x 47.9 a	0.7	x 48.9 a	0.6
Cones	x 48.6 ab	0.6	x 48.7 a	0.4	y 46.0 a	0.2	xy 46.5 abc	0.3
			Soil					
Brown cubical rot	x 48.2 a	0.9	x 47.l a	1.5	x 48.l a	0.9	x 49.0 a	0.7
Litter	x 41.3 ab	1.5	x 32.7 ab	3.4	x 34.0 b	2.6	x 43.6 ab	1.2
Humus	x 36.2b	3.6	xy 29.l b	4.4	y 21.9 c	2.0	xy 26.8 c	1.9
Soil wood	x 47.2 a	2.4	x 44.6 ab	3.0	x 42.9 ab	0.9	x 41.6 b	1.0
Shallow mineral (0-10 cm)	x 6.l c	0.5	x 4.5 c	0.6	x 4.4 d	0.6	y 1.4 d	0.1
Deep mineral (>10, up to 30 cm)	yz 1.9c	0.5	xy 2.5c	0.6	x 3.5d	0.3	z 0.7d	0.04

In the forest soil component, BCR tended to have statistically significant higher carbon concentrations (ranging from 47 percent to 49 percent) than the other soil components within each habitat type (table 7). More importantly, in all habitat types, the mineral soils had the lowest carbon concentrations, ranging from 1 percent to 6 percent. Within habitat types, concentrations for deep mineral soil were lower than those for shallow mineral soil, except WH/CLUN, which had 4 percent for both. The similarities that did occur were in the ponderosa pine habitat types where concentrations for humus were similar to those for litter.

Nine different tree species contributed to carbon concentrations for the tree components (table 8). For the overstory crowns, there were significant differences among species. For example, subalpine fir crowns had the highest (51 percent), while Engelmann spruce crowns had the lowest (48 percent). The results revealed no significant differences among species in sapwood carbon concentrations; however, significant differences were noted for heartwood, ranging from 46 percent for Engelmann spruce to 53 percent for ponderosa pine. For example, subalpine fir contained 46 percent while Douglas-fir contained 50 percent. Understory concentrations did not range as much (48 percent to 50 percent), with ponderosa pine having the highest (50 percent). For coarse root concentrations among species, Douglas fir had the highest (50 percent), while western redcedar and subalpine fir had the lowest concentrations (each had 46 percent).

Table 8—Carbon concentrations (%) for individual species with the mean and standard error ($S_{\bar{x}}$). Significant differences among the means across habitat types are presented as x, y, and z located above the value. Significant differences among the means across different forest components within a habitat type are presented as "a" and "b" located next to the value. If the letter is the same no significant differences were identified. Refer to table 1 for habitat type designation.

	Overstory crown				Bole wood		Coarse roots		Understory crown	
			Sap wood		Heart wood					
Species[a]	%	$S_{\bar{x}}$	%	$S_{\bar{x}}$	%	$S_{\bar{x}}$	%	$S_{\bar{x}}$	%	$S_{\bar{x}}$
WWP	x 50.0 ab	0.4	xy 48.5a	0.9	xy 48.9 b	0.8	xy 48.4 ab	0.1	y 47.3 ab	0.1
DF	y 48.7 bc	0.3	z 47.l a	0.2	z 47.4 bc	0.3	x 49.8 a	0.6	z 47.5 ab	0.4
GF	x 49.3 ab	0.3	y 47.7 a	0.1	y 47.7 bc	0.1	z 45.9 cd	0.2	y 47.8 ab	0.3
WH	x 49.9 ab	0.3	y 46.4 a	0.1	y 47.l bc	0.2	y 47.3 bcd	0.2	y 47.3 ab	0.2
WRC	x 48.8 bc	0.2	xy 47.l a	0.1	xy 48.0 bc	0.3	y 45.7 cd	0.1	xy 46.6 b	0.9
LP	x 49.5 ab	0.1	y 47.4 a	0.03	y 47.S bc	0.1	y 47.l bcd	0.3	x 49.l ab	0.2
ES	xy 47.6 c	0.1	y 46.7 a	0.2	y 46.4 c	0.2	y 46.5 bcd	0.3	x 48.0 ab	0.1
SAF	x 50.5 a	0.2	x 43.7 a	3.5	x 47.4 bc	0.2	x 45.7 d	0.1	—	—
PP	xy 49.9 ab	0.1	y 47.5 a	0.2	x 52.5 a	0.8	y 47.9 abc	0.4	xy 49.9 a	0.1

[a] Tree species are WWP-western white pine (*Pinus monticola* Dougl. ex D. Don); DF-Douglas fir (*Pseudotsuga menziesii* (Mirb.) Franco); GF-grand fir (*Abies grandis* Dougl. cx D. Don); WH-western hemlock (*Tsuga heterophylla* (Raf.) Sarg.); WRC-western red cedar (*Thuja plicata* Dorm cx D.Don>; LP-lodgepole pine (*Pinus contorta* oougl. cx Loud.); ES-Engelmann spruce (*Picea Engelmannii* Parry cx Engelm.); SAF-subalpine fir (*Abies lasiocarpa* (Dougl. cx D. Don) Lindl; PP-ponderosa pine (*Pinus ponderosa* Dougl. ex Laws.)

Carbon Content

We compared total carbon content in each classification among and within habitat types (fig. 3). The ponderosa pine types were considerably higher than others, with 562 Mg/ha (251 tons/acre) for PP/QUGA and 533 Mg/ha (238 tons/acre) for PP/FEAR; WH/CLUN had 394 Mg/ha (176 tons/acre) and SAF/VASC had 239 Mg/ha (107 tons/acre).

Within all habitat types, the highest proportion of carbon content was in trees; PP/FEAR had 62 percent, PP/QUGA 61 percent, SAF/VASC 42 percent, and WH/CLUN 40 percent. The component with the lowest proportion of the total carbon content for all four habitat types was woody residue, with WH/CLUN having the highest proportion (11 percent) compared to the others. Carbon content in the soils was not significantly different among habitat types.

In all habitat types (no figure shown), carbon content in trees was dominated by the boles, followed by coarse roots and understory crowns; fine roots had the least amount of carbon (fig. 3). Interestingly, carbon content in fine roots tended to be higher in the WH/CLUN (4.0 Mg/ha; 1.8 tons/acre) and SAF/VASC (2.2 Mg/ha; 0.9 tons/acre) habitat types compared to the ponderosa pine types (0.1 to 0.2 Mg/ha; 0.04 to 0.09 tons/acre). In SAF/VASC, shrubs had higher carbon content (15.0 Mg/ha; 7 tons/acre) compared to the forbs and grasses. In contrast, grasses and forbs had higher carbon content in PP/FEAR.

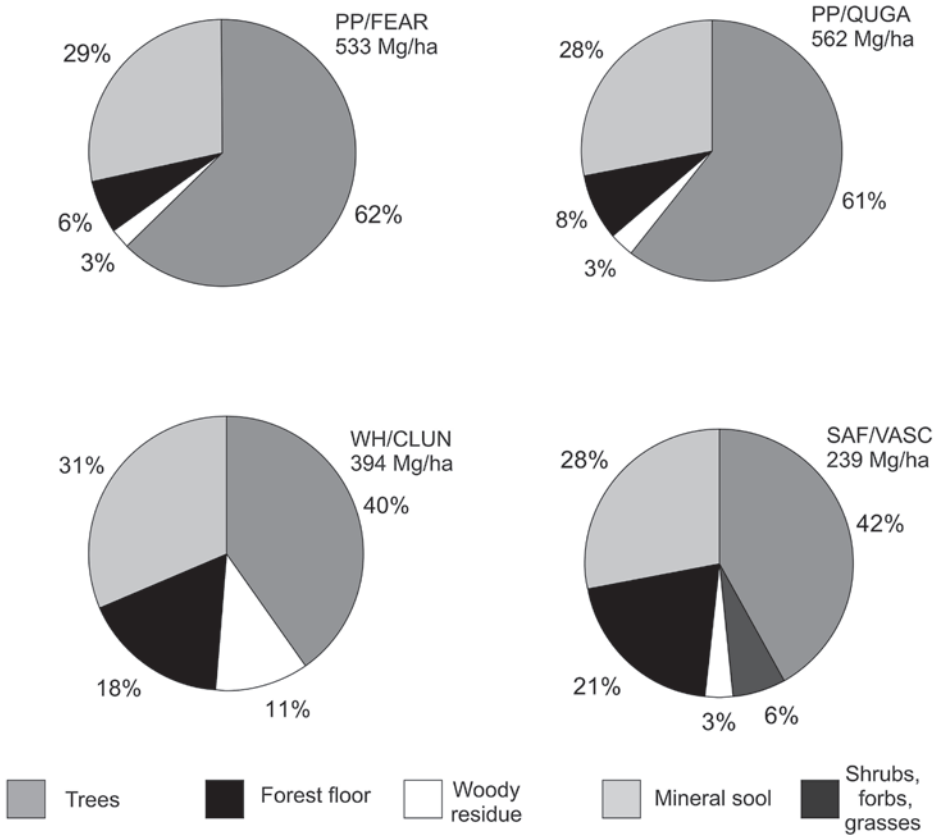

Figure 3—The distribution of carbon for trees, forest floor, woody residue, mineral soils, and understory vegetation (shrubs, forbs, and grasses) in the four habitat types (see figure 2 for habitat type definitions). The ponderosa pine sites (PP/FEAR and PP/QUGA) had significantly more carbon in the trees than the WH/CLUN or SAF/VASC sites. Woody residue, although a minor component, was significantly more abundant in WH/CLUN than the other sites. Shrubs and grasses tended to contribute more towards total carbon content in WH/CLUN and SAF/VASC compared to the ponderosa pine sites.

For woody residue components, cones comprised a rather large proportion of total carbon content in the ponderosa pine habitat types; PP/FEAR had 73 percent of the 15 Mg/ha in cones and PP/QUGA had 74 percent of the 18 Mg/ha in cones (fig. 4). In the WH/CLUN habitat type rotten (44 percent) and solid logs (38 percent) contributed the most carbon. In SAF/VASC, no significant differences occurred among woody residue components but the proportion of carbon content tended to be higher in cones (35 percent) and rotten logs (44 percent). Significant differences did occur among habitat types for sticks; SAF/VASC (15 percent) and PP/QUGA (14 percent) had more than WH/CLUN (10 percent) and PP/FEAR (7 percent).

Carbon content within the forest floor was dominated by mineral soil in the PP/FEAR habitat type (fig. 5). Within the organic soil components, humus and litter in PP/FEAR had significantly more than BCR and soil wood, while in PP/QUGA organic components (litter, humus, soil wood, and BCR) were not a significant contribution. In the SAF/VASC and WH/CLUN habitat types, no significant differences occurred among any of the soil components, indicating that carbon was well distributed among the different soil components. Comparisons among habitat types showed no significant differences in carbon content across the soil components, except in mineral soils. Generally, the content in shallow mineral soils for the ponderosa pine habitat types was higher than WH/CLUN or SAF/VASC.

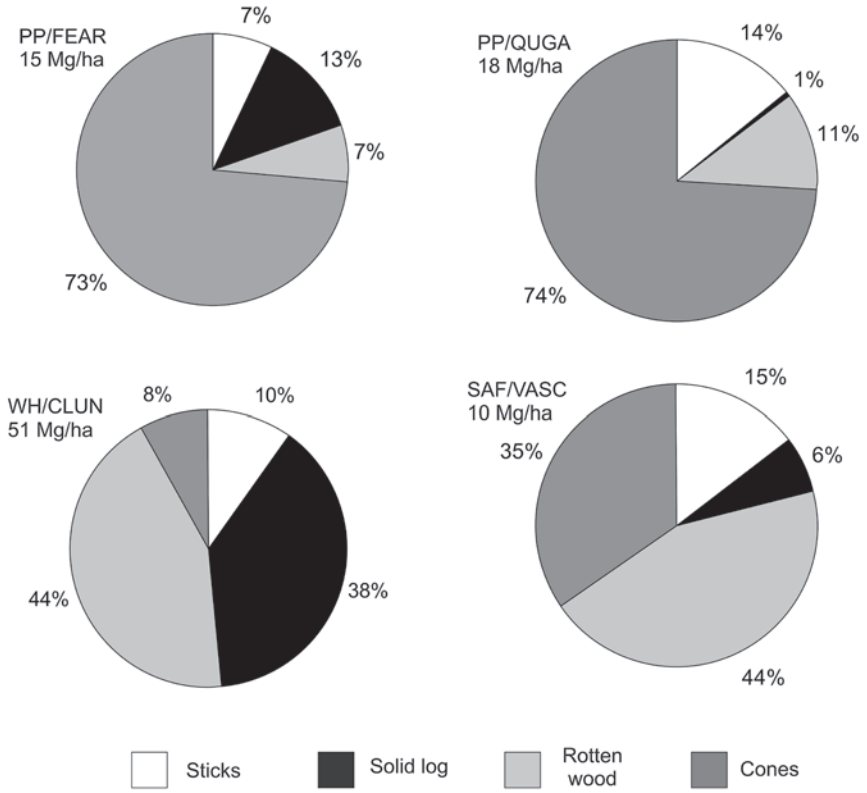

Figure 4—The distribution of woody residue in each of the habitat types. For the PP/FEAR and PP/QUGA, most of the woody residue was in cones, with 73% and 74% respectively. WH/CLUN had the highest amount of carbon in woody residue, with 51 Mg/ha; the greatest proportions of this occurred in solid and rotten logs. SAF/VASC had the lowest amount of carbon in woody residue; a large percentage of this was in cones (35%) and rotten wood (44%).

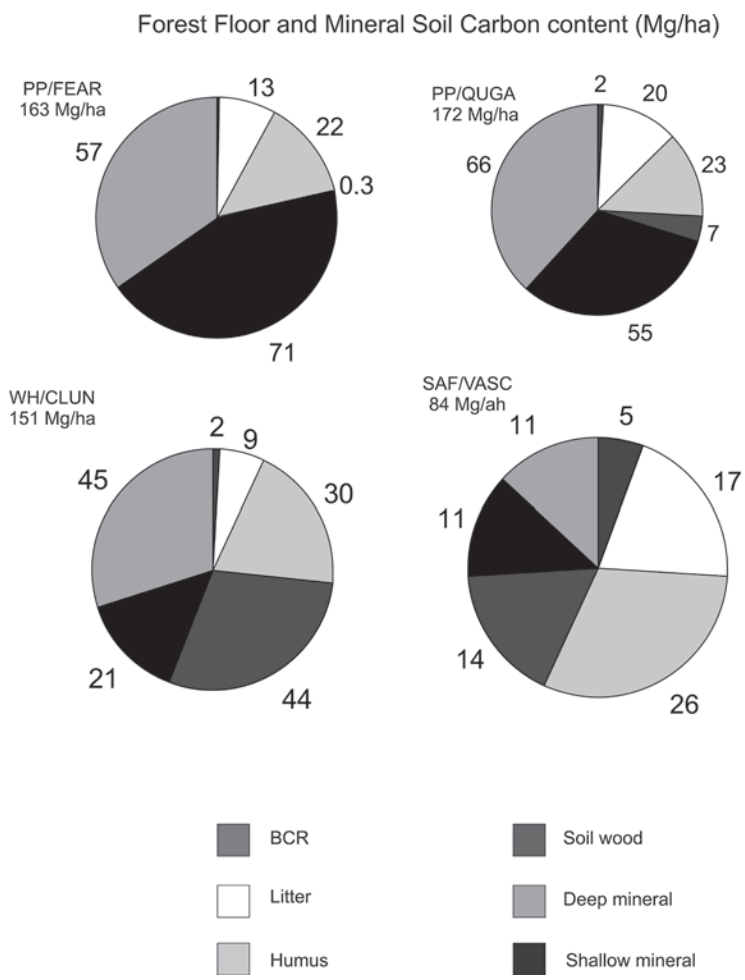

Figure 5—The distribution of carbon content varied widely among the different habitat types. In the ponderosa pine habitat types (PP/FEAR and PP/QUGA), mineral soil (shallow and deep) dominated. In contrast, for WH/CLUN and SAF/VASC, carbon was located in many more places besides litter (for example, humus, BCR, and soil wood). In addition, shallow mineral soil contained more carbon in WH/CLUN than in SAF/VASC.

In the deep mineral layers, carbon content (10.9 Mg/ha; 5 tons/acre) in SAF/VASC was significantly less than the carbon content among the other habitat types.

Discussion

Carbon is a critical element that plants accrue and use to support their structure and sustain physiological processes in temperate forest ecosystems (Waring and Schlesinger 1985). Besides being a key element in forest ecosystems, carbon is also essential for sustaining life on a global scale (Bolin and others 1979). Estimating carbon reserves in all ecosystems is critical if we are to understand the role carbon plays in climate change (Post and others 1990; Schlesinger 1977). It is also essential that we understand the potential human impacts on carbon storage in forest ecosystems and the resulting effects on the global carbon cycle. Although there are gross estimates available, additional information on carbon concentrations in forest components could improve those estimates.

Carbon Concentrations

To help improve the accuracy of estimates of carbon reserves in Rocky Mountain forests, this study quantified carbon concentrations for forest components in four habitat types (table 1). These results are comparable to other studies on carbon in the Rocky Mountains. For example, Klemmedson's (1975) carbon concentrations for tree, understory, coarse woody debris, and other components in southwest ponderosa pine forests ranged from 39 percent to 50 percent. The forest components from the ponderosa pine sites in our study had similar concentrations (table 9). In addition, concentrations in all forest types sampled were similar to those of Klemmedson's (1975) and Lamlom and Savidge (2003).

Traditionally, a concentration of 50 percent was used for calculating carbon content from tree biomass, as evaluated by Lamlom and Savidge (2003). This estimate is reasonable if the objective is to provide approximate estimates of total carbon. In this study, we found there to be significant differences in carbon concentrations of coarse roots and overstory crowns among species; however, the maximum differences were only 1.7 percent for the overstory and 2.7 percent for coarse roots. These small differences would have an insignificant impact on carbon estimates when the variations in biomass estimates are included. For example, the amount of error introduced in root biomass estimations outweighs the small differences detected in carbon concentrations. On the other hand, to improve carbon content estimates for trees, an average carbon concentration for each tree component could be used. Based on this study, we recommend 49.5 percent for overstory crown, 47.6 percent for boles, 47.2 percent for coarse roots, and 48.4 percent for understory trees (table 9). Similarly, better carbon content estimates for other forest components can be achieved by using more precise carbon concentrations.

Carbon Storage

In the ponderosa pine habitat types, carbon weights in trees were 20 percent higher than those reported by Klemmedson (1975), who conducted a similar study near Flagstaff, Arizona. This discrepancy is probably due to the difference in tree size between the two studies and our successional stage. Klemmedson sampled sapling and pole-sized trees, while trees sampled in this study were mature and ranged from 50 to 100 cm (20 to 40 inches) d.b.h. The amount of carbon storage in WH/CLUN trees was less than the amount stored on the ponderosa pine sites. This appears to suggest that the ponderosa pine sites are more productive than the WM/CLUN sites. However, this can be misleading because the sites sampled

Table 9—Carbon concentrations recommended for converting organic biomass to carbon content for habitat types evaluated in this study. Because significant differences occurred among habitat types, separate carbon concentrations are recommended. Refer to table 1 for habitat type designation.

Forest component	Carbon concentration (%)
Trees	
Overstory crown	49.5
Bole	47.6
Coarse roots (>1 cm diameter.)	47.2
Fine roots (≤1 cm diameter)	45.3
Understory trees (<12.7 cm tall)	48.4
Understory vegetation	
Shrubs	47.2
Forbs and grasses	41.4
Woody residue	
Sticks (<7.5 cm diameter)—PP/FEAR and PP/QUGA	49.1
Sticks (<7.5 cm diameter)—WR/CLUN and SAF/VASC	47.2
Solid log (≥7.5 cm diameter)	48.2
Rotten log (≥7.5 cm diameter)	48.7
Cones—PP/FEAR and PP/QUGA	47.9
Cones—WR/CLUN and SAF/VASC	46.0
Soils	
Brown cubicle rotten wood	48.0
Litter	37.9
Soil wood	44.2

on the WH/CLUN habitat type were not at their maximum growth potential and relatively young, while the ponderosa pine sites were older and maximum growth potential may have been reached (Pearson 1950). When western hemlock habitat types are at their full growing capacity, the carbon storage potential could be much higher (Haig 1932).

Several studies have reported the importance of shrubs, forbs, and grasses for nutrient cycling (Chapin 1983; Jorgensen and Wells 1986), yet rarely quantify the amount of carbon they can store. Dwarf huckleberry in the SAF/VASC is a small shrub rarely considered for its ability to store carbon; in this study, 15 Mg/ha (6.7 tons/acre) of carbon were stored in this component. Forbs and grasses may also have the potential to be important for carbon storage. This study, however, found a maximum of only 0.1 Mg/ha (.04 tons/acre) across all habitat types. Before fire suppression, forbs and grasses were abundant in the ponderosa pine types. However, due to lack of fire as well as over-grazing, in this study, they were an insignificant element for carbon storage (Covington and Moore 1994). However, with the advent of more wild and prescribed fire, these pools could shift and thus grass and forbs could play a greater role in storing carbon both above and below (rapid root turnover may increase carbon concentration in mineral soil) the soils surface. Therefore, depending on forest type and forest history, small components within forest ecosystems should not be overlooked when estimating carbon pools.

Coarse roots have the potential to store large amounts of carbon. In the ponderosa pine habitat types, we found that coarse roots stored more carbon than in the WH/CLUN and SAF/VASC habitat types. Carbon allocation to roots varies widely among sites, depending on growing season, nutrient availability, climate, tree species, age, and genetic materials (Kramer and Kozlowski 1979). For example, the ponderosa pine forests sampled in this study were 25 to 50 years older than the WH/CLUN forest and 60 years older than the SAF/VASC forest.

This difference in age most likely influenced the amount of carbon stored in roots and other carbon pools. The Northern Rocky Mountain habitat types had more fine woody roots than the ponderosa pine forests of the southwest. This may be because moisture and nutrients are located in the surface layers of Northern Rocky Mountain forests, causing trees to allocate more carbon to fine root growth (Aber and Melillo 1991; Kramer and Kozlowski 1979; Page-Dumroese and others 1990). Schlesinger (1977) discussed the importance detritus plays in ecosystem function and carbon cycling, referring to root turnover, undecomposed litter, and soil humus, but did not mention the contribution of woody residue. Keenan and others (1993) recognized the importance of woody residue, reporting 161 Mg/ha (71.8 tons/acre) of carbon in woody material in western redcedar and western hemlock forests in northern Vancouver Island. Similarly, in the younger inland western hemlock forest we sampled, 51 Mg/ha (23.8 tons/acre) was found in woody materials. Although woody material is beginning to be recognized as a carbon sink, usually only coarse woody debris is considered, while other components that may be important for storing carbon are ignored. For this reason, we quantified where carbon is located within some of these other woody components (fig. 4).

This study found that cones are a major component of the woody materials of three (PP/FEAR, PP/QUGA, and SAF/VASC) of the four habitat types (fig. 4). In the ponderosa pine habitat types, greater than 70 percent of the carbon in the woody residue was in cones. In the SAF/VASC habitat type, cones also stored a significant proportion (35 percent); WH/CLUN had the least amount (8 percent). In vegetation types such as ponderosa and lodgepole pine, where cones represent a large portion of the woody residue, it is important to consider these components when estimating total carbon reserves. Other types of woody residue also store large amounts of carbon. For example, in the WH/CLUN habitat type, large and small woody residue contributed 11 percent of the total carbon on the site (fig. 3), with more than 80 percent in solid and rotten logs (fig. 4). These results show that CWD plays a major role in storing carbon in WH/CLUN habitat types while sticks are important in ponderosa pine habitat types.

As snags, CWD, sticks, cones, and coarse roots decompose, they form soil wood and BCR, important soil components of Rocky Mountain forest ecosystems (Graham and others 1994; Harvey and others 1987). Graham and others (1994) suggested that forest floors may consist of 30 to 60 percent woody material. In this study, we found that 25 to 30 percent of the soil carbon was in soil wood and BCR (fig. 5). Although in the ponderosa pine sites less than 7 Mg/ha (3.1 tons/acre) of the soil carbon consisted of soil wood, there is a large potential for soil wood recruitment after trees die and root biomass becomes soil wood. In Rocky Mountain forests, soil wood and BCR are important carbon sinks that are often overlooked.

Litter and humus also store large amounts of carbon. The large amounts of litter in the ponderosa pine and SAF/VASC habitat types are the result of the continuous shedding of needles (Kilgore 1981; Olson 1981). The proportion of carbon in the litter and humus located in the soils of the ponderosa pine habitat types in this study ranged from 20 percent to 25 percent. These proportions were larger than the 10 percent reported by Klemmedson (1975). This is probably due again to the differences in stand ages or successional stage between the two studies. Klemmedson's (1975) stands were younger and did not produce as much litter and surface humus, while this study's stands were over 200 years old.

Carbon content in mineral soils varied among habitat types. The results from this study show that in the ponderosa pine habitat types, greater than 70 percent of the forest soil carbon was stored in the mineral soils (fig. 5). In comparison, Klemmedson (1975) found that 89 percent of the total soil carbon content was in mineral soils. This amount was the result of root turnover from

the grass component (Buol and others 1989). In the WH/CLUN and SAF/VASC habitat types, litter and humus contained 39 to 43 Mg/ha (17.4 to 19.8 tons/acre) of carbon (fig. 5). These results did not differ greatly from the 50 to 60 Mg/ha (22.3 to 26.8 tons/acre) in litter and humus reported by Keenan and others (1993) from sites in northern Vancouver Island. The ponderosa pine habitat types had more carbon in mineral soils than WH/CLUN and SAF/VASC (fig. 5), which may be due to higher clay content in soils of the ponderosa pine habitat types.

Other reasons for the variation in carbon storage among the forest types may be differences in climate and decomposition (Aber and Melillo 1991; Harmon and Hua 1991; Harvey and others 1987). For example, warm temperatures in the southwest coupled with summer rains provide favorable conditions for microbes to decompose woody material (Clark 1957), while WH/CLUN and SAF/VASC have colder temperatures, thus slowing decomposition rates (Harmon and others 1986). These environmental factors controlling decomposition contribute to the differences in carbon storage among the habitat types.

On a global scale, researchers have theorized that the major carbon sink is in mineral soils (Post and others 1982; Schlesinger 1986). However, most global carbon estimates ignore many other forest components. In this study, we found that other forest components such as shrubs, cones, CWD, BCR, and soil wood can be major carbon sinks in Rocky Mountain forest ecosystems.

Conclusions

Typically, the conversion of 0.50 is used to provide estimates of carbon pools; however, this study provided a suite of values that vary depending on the species, substrate, and location. Moreover, these values tended to be less than the conventional value, leading one to overestimate total carbon amounts in these forest types if the typical conversion is used. In addition to carbon concentrations, we showed the variability in carbon content as a function of forest type and that minor elements such as cones, shrubs, and brown cubical rotten wood can contribute to the total carbon pool. Given these results, a series of additional research questions could be pursued such as the effect of successional stage on carbon pool distributions. For example, young forests may not contain the brown cubical rotten wood mid- to late-seral moist forests contain. Also, as forests grow and develop, do carbon concentrations change or does only biomass distribution change over time? Thus, determining if carbon concentrations vary as a function of successional change could provide invaluable information concerning variation of carbon over time and space.

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The content of this paper reflects the views of the authors, who are responsible for the facts and accuracy of the information presented herein.

Carbon Benefits from Fuel Treatments

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Abstract—Landscape simulation modeling is used to examine whether fuel treatments result in a carbon offset from avoided wildfire emissions. The study landscape was a 169,200-acre watershed located in south-central Oregon. Burn probability modeling was employed under extreme weather and fuel moisture conditions. Expected carbon stocks post-treatment, post-wildfire were calculated for all stands on the treated landscape; post-wildfire on the untreated landscape. Results show a negative carbon offset initially—the known reduction of carbon stocks from treatment is greater than expected carbon benefit from reduced wildfire emissions. Treatment may break even as a carbon offset after 9 years.

Introduction

Many of the forests in the western United States (U.S.) are significantly altered as a result of decades of continuous fire suppression activity. The absence of wildfire in many forest types that historically have had frequent fire return intervals results in increased density of overstory trees and understory vegetation. Competition for limited moisture and nutrients further reduces tree growth and vigor. Add drought, insects or disease and the result is a perfect storm for uncharacteristically severe wildfire events—overstocked stands containing dead trees, dying crowns, ladder fuels, and high surface fuel loadings.

In response to these trends, the U.S. has increased appropriations for wildfire fuel treatment projects through the National Fire Plan and complementary programs (Healthy Forest Restoration Act 2003; Sexton 2006; USDA Forest Service and USDI Bureau of Land Management 2001). Appropriations for fuel treatments have also been awarded through the 2009 American Recovery and Reinvestment Act as an economic stimulus to the national recession. The goals of fuel treatment projects are to reduce wildfire fuels within stands, to provide defendable space around homes and communities, and to create fuel breaks that allow for more aggressive and effective initial attack suppression efforts (Finney and Cohen 2003; Reinhardt and others 2008). Mechanical methods include slash busting, chipping, bundling, piling and burning, as well as pre-commercial and commercial thinning. Prescribed fire is also used as a treatment, often in combination with mechanical methods. Effective treatments result in the removal of woody biomass from treated areas and improved residual stand vigor, which promote resiliency to fire, insect and disease. Fuel treatment projects also reduce fire spread and intensity both inside and outside the portions of the landscape treated (Finney 2001; Finney and others 2007). In most cases, fuel treatment projects are completed at a financial loss; the stumpage value of merchantable trees is usually insufficient to cover the cost of the overall treatment. Moreover, fuel treatment projects are not without controversy with respect to their impact on forest ecological values, even though there is general agreement as to their need (Brown 2008).

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Carbon Dioxide Emissions and Wildfires

For the period 2002-2006, wild and prescribed fire emissions from forests, range and agriculture in the U.S, averaged 4-6 percent of human derived carbon dioxide (CO_2) emissions from fossil fuel sources (Wiedinmyer and Neff 2007). However, the level of emissions varies widely among states and time of year. In Oregon, fire emissions are about 50 percent of the fossil fuel CO_2 emissions (Wiedinmyer and Neff 2007). A similar study estimates emissions from catastrophic wildfire in California during 2001-2007 as equivalent to three and a half years worth of CO_2 emissions from 14 million cars (Bonnicksen 2009). Wiedinmyer and Neff (2007) conclude that very large wildfires in a severe fire season lasting only one or two months can release as much carbon as the annual emissions from the entire transportation or energy sector of an individual state.

Increases in wildfire severity and extent will lead to increases in carbon dioxide (CO_2) emissions to the atmosphere. Simulations of how Oregon's forests will be affected by climate change in Bachelet and others (2001) show that Oregon's forests will accumulate more woody biomass, not less, especially in the eastern region of the state where continued biomass accumulation in the understory increases the risk of uncharacteristically severe wildfire. Nielson (2004) suggests that Oregon's current wildfire risk problem is not only the result of fire suppression efforts, but also the result of an underlying climate signal that has increased biomass accumulation. This signal is expected to continue into the future. In the Pacific Northwest, area burned can be expected to increase despite management efforts to reduce fuel loadings through treatment in fire prone forests (Gedalof and others 2005). As a result, the risk of high severity wildfires in western forests will increase, not decrease, in the future (McKenzie and others 2004). The area in need of fuel treatment far exceeds what is being planned for and accomplished despite increases in federal appropriations. This situation only stands to get worse as climate change leads to more biomass accumulation in western forests (USDA Forest Service 2005). Law and others (2004) conclude that large fire events, such as Oregon's 2002 Biscuit fire can significantly reverse the role forests play in the carbon cycle from being net carbon sink that sequesters atmospheric CO_2 to being a net source of CO_2 to the atmosphere, not only in the year of the fire, but in subsequent years due to continued decomposition of fire-killed material.

The wildfire risk problem can in part be attributed to an excess of stored carbon in the forest as a result of many decades of fire suppression (Hurteau and North 2009; North and others 2009; Stephens and others 2009a). Fuels reduction treatments in general reduce the amount of stored carbon on treated areas and are thus a source of carbon to the atmosphere. Girod and others (2007) concluded that strategies that promote both carbon storage and reduced wildfire risk will need to involve increased fuel treatment efforts that are spatially adjusted to avoid unnecessary reductions in carbon stocks. Despite the fact that fuel treatment practices result in carbon releases back to the atmosphere, several studies conclude that fuel treatment projects can result in a net gain in carbon storage landscape-wide because of the treatment's effect on reducing and avoiding CO_2 emissions from severe wildfires (Bonnicksen 2009; Hurteau and others 2008; North and others 2009; Stephens and others 2009a). Other studies disagree; fuel treatment leads to a net loss of carbon on the forest landscape even when the avoided emissions from uncharacteristically severe wildfire are factored in (Krankina and others 2008; Mitchell and others 2009).

These differing conclusions can be attributed to the fire regime and disturbance return interval of the forest vegetation type being studied and how wildfire risk is assessed. For example, Mitchell and others (2009) concluded that for hemlock-spruce-Douglas-fir forest types of western Oregon, fuel treatment projects fell

at odds with effectively managing forests for long-term carbon storage. This is because extreme weather conditions, not fuel loadings, are the leading cause of high severity wildfires in these forest types. Krankina and others (2008) found a net carbon benefit to treated stands, but only in short fire return interval dry ponderosa pine forests where the removal of non-merchantable understory material was sufficient to reduce wildfire risk.

Unless the surface fuels, understory and ladder fuels found in small, unmerchantable trees are all treated to reduce wildfire risk, the treatment projects will not be effective in reducing wildfire severity and extent (Cram and others 2006; Stephens and others 2009b). There are not many facilities that will take this type of material and when they do, the delivered value of the material is usually not sufficient to cover the costs of its removal and transportation (Mason Bruce & Girard Inc. and others 2006). Recognizing the potential of utilizing forest slash and other residues as a source of biomass energy, thereby displacing fossil fuel use, Oregon's strategy for reducing greenhouse gases included the creation of markets for forest biomass as a significant greenhouse gas mitigation action (Governor's Advisory Group on Global Warming 2004). The question remains though on whether the implementation of fuel treatment projects creates an additional carbon benefit accruing to the treated forest landscape (as the Advisory Group hypothesized), or results in an overall decrease in carbon stores that should be deducted from any displaced CO_2 emissions benefit claims being attributed to biomass energy.

Purpose

The purpose of this study is to properly account for the net balance of carbon storage resulting from a landscape fuel treatment project. On the treated acres themselves, the goal is to reduce or remove biomass, which may reduce on-site carbon stocks, even when continued carbon storage in long-lived wood products and avoided wildfire emissions is accounted for. However, strategically located fuel treatments can reduce fire spread and intensity both inside and outside the portions of the landscape treated. Therefore, the accounting of carbon gains and losses resulting from the fuel treatment project must be calculated in a way that includes the reduction in fire risk to and corresponding change in carbon stores within untreated areas of the landscape.

Quantifying whether a treatment's effect on stored carbon is positive or negative is especially important if the fuel treatment project attempts to seek funding from entities investing in such a project as a means to generate carbon offsets to their CO_2 emissions. If the treatment effect is positive (more carbon is stored on the landscape as a result of treatment), then a carbon offset might be able to be claimed by the project[6]. Clearly, if the opposite is true, then no carbon offset can be claimed. Such a "negative offset" means that there is an emission cost, in terms of carbon released to the atmosphere. But this result does not mean the fuel treatment project is any less effective in its primary purpose—to make forested landscapes more healthy and resilient to wildfire—and as a result, more suited for providing clean water, fish and wildlife habitat, and protecting Oregon's communities from the risk of wildfire.

[6] Whether or not the amount of carbon benefit can actually be claimed as on offset in a voluntary carbon market or compliance carbon trading program is beyond the study's scope.

Methods

Because wildfire is a chance event, this study estimates the expected carbon stored in forests based on the probability of wildfire on the landscape. Using a simulation modeling approach, expected carbon loss from wildfire on the untreated landscape is estimated and compared to the expected carbon loss from wildfire and treatment for the treated landscape. There are two ways to look at carbon loss. The first is to look at the expected change in carbon stocks; the second is to look at the expected carbon emissions from the landscape. If all affected carbon pools are accounted for, the two measures of carbon loss mirror one another. The expected change in carbon stocks equals expected carbon emissions when both changes are measured in the same units such as CO_2 equivalent. At the time of a wildfire event, carbon loss increases as the intensity of wildfire increases and in all cases, regardless of intensity, wildfire reduces carbon stocks. Only in the case where wildfire does not occur can carbon stocks increase. This study's approach is to simulate all possible wildfire intensities on all possible stand conditions (both treated and untreated) to develop carbon loss functions for each stand condition. Then, by simulating thousands of wildfires on the landscape, a probability surface of wildfire is developed both for the treated landscape and the untreated landscape. The difference in the probability surfaces (treated minus untreated) is a measure of the treatment effect on reducing both the likelihood of wildfire on the landscape given an ignition, and the intensity of wildfire given that wildfire occurs. Simulating carbon loss functions using this probabilistic approach and comparing the results from treated and untreated landscapes quantifies the expected change in carbon stocks resulting from treatment. Table 1 summarizes the major steps for the analysis.

Study Area

The study area is the Drews Creek watershed located within the Goose Lake basin of southern Oregon (fig. 1). The Drews Creek watershed was selected because it contains dry ponderosa pine and mixed conifer forest types at risk of uncharacteristically severe wildfire, is at the beginning stages of fuel treatment planning by the Fremont-Winema National Forest, and is within a collaborative terrestrial sequestration research pilot through the West Coast Regional Carbon Sequestration Partnership[7]. The watershed is approximately 169,200 acres of which approximately 77,500 acres are privately owned and the remaining 91,700 acres are owned and managed by the U.S. Department of Agriculture (USDA) Forest Service.

The Drews Creek watershed encompasses a relatively narrow band of topographical relief. Elevations of the watershed range from 6,400 - 7,900 ft in the northeast portion to 4,200 - 5,000 ft in the valley adjacent to Goose Lake in the southwest. Although slope ranges from 0 - 200 percent, much of the watershed slopes relatively gently with the average slope being 14 percent. The forested area of the watershed is 140,526 acres. Stands dominated by ponderosa pine (*Pinus ponderosa* Laws.) account for about 68 percent of the forest land in the watershed. Approximately 17 percent of the area is in juniper woodlands, and western juniper (*Juniperus occidentalis* Hook.) dominates 26 percent of all forested types, encroaching on the hot dry ponderosa pine sites. Stands dominated by white fir (*Abies* spp.) represent a minor contingent of the landscape, at about 6 percent of

[7] For more information on WESTCARB, see: http://www.westcarb.org/.

Table 1—Steps for modeling wildfire risk, fuel treatments and expected carbon.

Step	Description	Data source	Model
1	Calculate the amount of stored carbon by carbon pool for each untreated stand condition under all possible wildfire intensity conditions, including no fire.	GNN[a] treelist and LANDFIRE fuel model data.	FFE-FVS[b] with fuel model override and FlameAdj keyword.
2	Calculate the amount of stored carbon by carbon pool for each treated stand condition under all possible wildfire intensity conditions, including no fire.	Post treatment FVS treelist output.	FFE-FVS with fuel model override and FlameAdj keyword.
3	Create untreated landscape file.	Slope, Aspect, Elevation, GNN treelist, and LANDFIRE fuel model data; FVS outputs (initial conditions) for Canopy Bulk Density, Canopy Base Height, Crown Closure and Canopy Height.	ArcFuels
4	Create treated landscape file.	Same as Step 3 except use FVS outputs from Step 2 for Canopy Bulk Density, Canopy Base Height, Crown Closure and Canopy Height and switch fuel model to 181 for treated stands.	ArcFuels
5	Conduct repeated landscape simulations of wildfire from a random wildfire ignition for both the untreated and treated Drews Creek landscapes.	Landscape files from Steps 3 and 4.	Randig
6	Calculate conditional burn intensity probabilities for each 90- by 90-meter pixel for both the untreated and treated landscapes.	Output from Step 5.	Randig
7	Calculate the amount of expected stored carbon for each 90- by 90-meter pixel for both the untreated and treated landscapes.	Outputs from Step 1, Step 2 and Step 6.	ArcFuels
8	Calculate and map the expected carbon offset for each 90- by 90-meter pixel; summarize results to get the carbon offset accruing to the treated landscape.	Outputs from Step 7; the offset is treated landscape results minus untreated landscape results.	ArcFuels

[a] GNN—Gradient nearest neighbor.

[b] FFE-FVS—The Fire and Fuels Extension to the Forest Vegetation Simulator.

the forested acres. Smaller stands of lodgepole pine (*Pinus contorta* Dougl. ex Loud.) as well as small stands of aspen (*Populus tremuloides* Michx.) are scattered across the landscape. Past harvesting activities have significantly reduced the area dominated by large diameter old growth ponderosa pine. Dry grasslands, dry shrub lands and dry meadows comprise nearly one-half of non-forested lands (tree cover less than 10 percent) with the balance being agricultural lands and wet meadows associated with the major streams.

Fuel loadings are variable across the drainage, but follow various gradients. As elevation increases, stands types move from pure low elevation ponderosa pine, to ponderosa pine dominated mixed conifer, in association with white fir, western white pine (*Pinus monticola* Douglas ex D. Don), sugar pine (*Pinus lambertiana* Dougl), and incense-cedar (*Libocedrus decurrens* Torr.), to upper elevation fringes where white fir dominates or more frequently co-dominates with ponderosa pine.

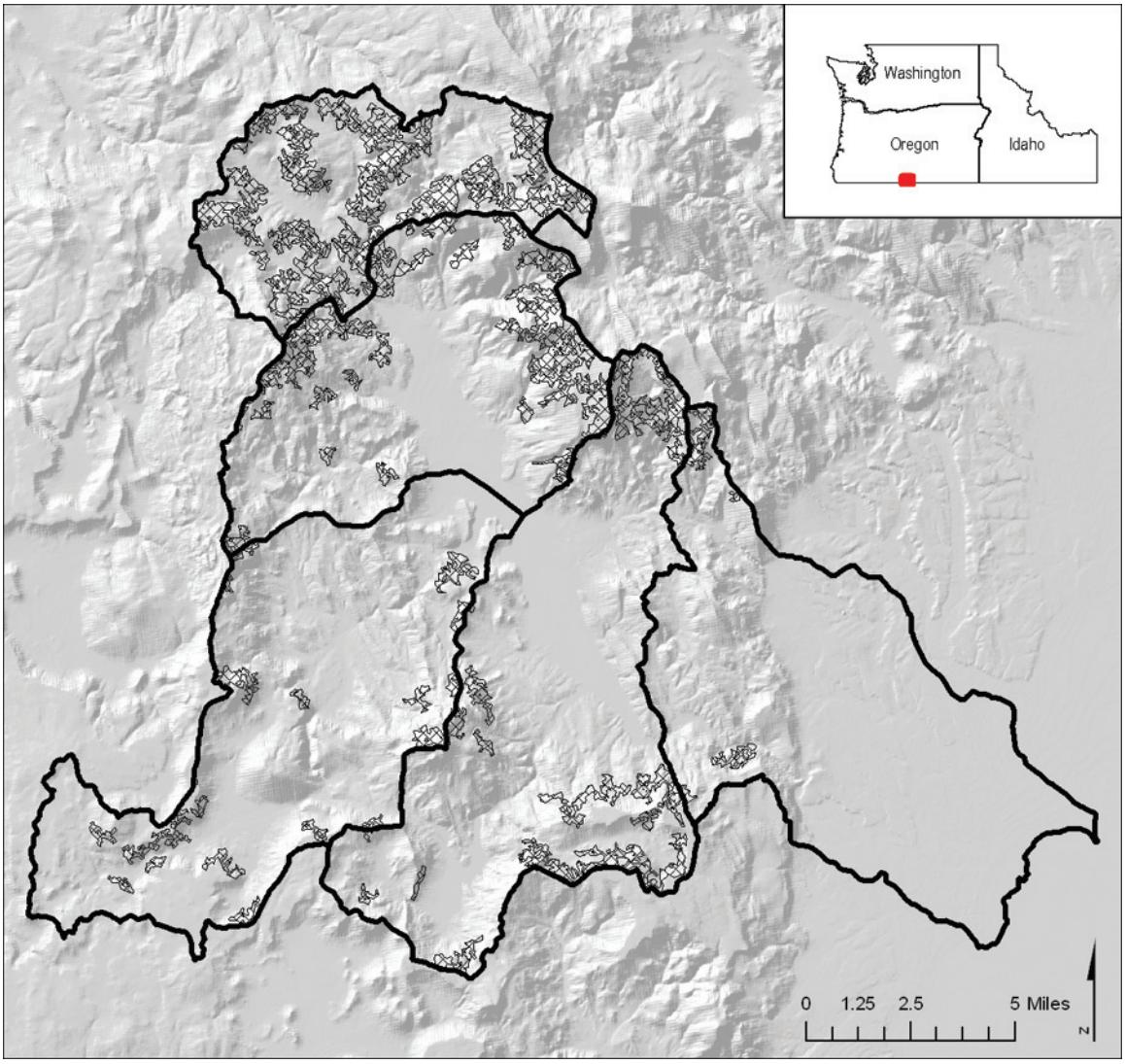

Figure 1—Vicinity and relief map for the Drews Creek Watershed, Fremont-Winema National Forest showing watershed boundaries and treatment units.

Another gradient is aspect, and a third is whether there has been a prescribed burning treatment. Most of the pine dominated stands below 5,500 feet in elevation have received at least one prescribed burn treatment, the exceptions being the Quartz Creek sub-watershed and the Chandler Creek drainage, forming the northern tier of the watershed.

In the low elevation treated pine stands, typical fuel loadings range from 2-5 tons per acre. Untreated pine stands tend to be more variable, averaging 3-15 tons per acre. As white fir joins the stands at low elevations, loadings increase rapidly, particularly where root disease is present. Typical loadings here can range from 15 tons to as high as 50 tons per acre or more if there has been recent disturbance without fuel treatment. For the period of 1949-1999 the watershed has had 688 wildfire ignitions, with an average of fourteen fire starts per year. The high was 38 in 1977 and the low was one in 1963. All fires were actively suppressed. Of these, 88 percent were suppressed at less than 0.25 acres, 10 percent between 0.26 to 9.9 acres with the balance larger. Forty-four fires larger than 10 acres occurred over this period; the total burned for these fires is approximately 9,000 acres.

Spatial Modeling of Wildfire, Fuel Treatments and Expected Carbon Loss

The study used ArcFuels[8] to automate much of the analyses (Ager and others 2006). ArcFuels is a library of ArcGIS[9] macros developed to streamline spatial modeling of wildfire behavior, stand growth and yield, and fuel treatments for planning purposes. ArcFuels brings together various data layers (e.g., gradient nearest neighbor (GNN[10]) treelists, digital elevation grids, stand polygons, Forest Vegetation Simulator[11] (FVS) growth and yield outputs, LANDFIRE[12] fuel model data, slope, and aspect) and processes them in ways that facilitate communication between fire simulation, stand growth and yield, and spatial modeling programs. ArcFuels links the FlamMap[13] wildfire behavior models with fuels and vegetation data through a Microsoft© Office and ArcGIS platform. Carbon stocks were modeled through the Fire and Fuels Extension (FFE) of FVS[14]. Specifically, FFE-FVS accounts for the following carbon pools: aboveground total and merchantable live, below ground live and dead, standing dead, dead and down woody debris, forest floor (litter and duff), and understory (shrub and herb). FFE-FVS also accounts for the fate of carbon stored in merchantable material removed—specifically the amount of continued storage in wood products and landfills—as well as the amount emitted for energy and non-energy use (Rebain 2009).

Stand treatments—Stands were selected for treatment based on criteria developed by staff on the Fremont-Winema National Forest. Virtually all stands eligible based on basal area (70 ft^2 per acre), also met additional distance to road and slope criteria. Pixels selected for treatment were then aggregated into treatment units having a minimum size of 15 acres by first kernel density smoothing the selected pixels, and then retaining only those pixels that exceeded a density threshold. The result was 94 treatment polygons, consisting of 17,740 acres, averaging 175 acres each (fig. 1). The treatment units covered approximately 12.6 percent of the watershed's forestland. Of the 17,740 acres, only 12,825 acres met thresholds for treatment (9.1 percent of the watershed's forestland).

Stands (represented by the imputed GNN treelist data to 30- by 30-meter pixels) that were within treatment polygons were assigned a treatment prescription and the treatment was then simulated in FVS. The treatment prescriptions called for thinning from below to a residual basal area of 70 ft^2 per acre for mixed conifer or fir dominated stands and 50 ft^2 per acre for pine dominated stands, followed by slash removal and under burning. The specific parameters for the fuel reduction prescription were chosen based on operational guidelines from the Fremont-Winema National Forest. The treatments were simulated with FVS and consisted of a three-year sequence of thinning from below, site removal of surface fuels, and under burning. Under burning and mechanical treatment of surface fuels was simulated with the FFE-FVS keywords SIMFIRE and FUELMOVE (Reinhardt and Crookston 2003). Fuel treatment prescriptions for thinning from below had no upper diameter limit and specified retention of fire tolerant ponderosa pine and

[8] For more information on ArcFuels, see: http://www.fs.fed.us/wwetac/arcfuels/.
[9] For more information on ArcGIS, see: http://resources.esri.com/gateway/index.cfm.
[10] For more information on gradient nearest neighbor data, see Ohmann and Gregory (2002).
[11] For more information on the Forest Vegetation Simulator, see: http://www.fs.fed.us/fmsc/fvs/.
[12] Landscape Fire and Resource Management Planning Tools Project. For more information see: http://www.landfire.gov/.
[13] For more information on FlamMap, see Finney (2006).
[14] See pp. 3 through 7 in Rebain (2009) and Hoover and Rebain (2008).

favored the removal of white fir. Surface fuel treatments simulated the removal of 90 percent of the material up to 12 inches. Under burning was then simulated using weather conditions and fuel moisture guidelines provided by fuels specialists on the Fremont-Winema National Forest. The prescription was supported by empirical studies as effective for reducing potential wildfire behavior (Peterson and others 2003; Stephens and Moghaddas 2005; Stephens and others 2009b; van Wagtendonk 1996).

Burn probability modeling—We simulated wildfires using the minimum travel time (MTT) fire spread algorithm of Finney (2002) as implemented in a command line version of FlamMap called "Randig" (Finney 2006). The MTT algorithm replicates fire growth by Huygens' Principle where the growth and behavior of the fire edge acts as a vector or wave front (Finney 2002). Extensive testing over the years has demonstrated that the Huygens' Principle can accurately predict fire spread and replicate large fire boundaries on heterogeneous wildlands (Knight and Coleman 1993; LaCroix and others 2006; Yohay and others 2009). The MTT algorithm was parallelized for multi-threaded processing making it computationally feasible to perform Monte Carlo simulations of many fires (>100,000). The MTT algorithm is now being applied daily for operational wildfire problems throughout the U.S[15] and can generate burn probability surfaces for very large (> 2 million ha) landscapes (Ager and Finney 2009; Ager and others 2006; Ager and others 2007). The MTT algorithm assumes constant weather and is used to model individual burn periods within a wildfire rather than continuous spread of a wildfire over many days and weather scenarios.

Relatively few burn periods generally account for the majority of the total area burned in large (e.g. >5,000 ha) wildfires in the western U.S., and wildfire suppression efforts have little influence of fire perimeters during these extreme events. Based on input from forest staff, and supported by historical data from remote automated weather stations, each fire event was simulated as an 8-hour burn period with a 25 mph wind under the fuel moisture conditions listed in table 2. Wind was randomly simulated from three directions (225, 235, and 245 degrees) for each burn period. Ignition locations were random. The target simulated problem fire under these conditions, on average, was 11,000 acres. Randig outputs a vector of conditional burn probabilities (BP_i) for each pixel that represents the probability of a fire at the ith 0.5 m flame length category. Different flame lengths

Table 2—Fuel moisture values used in the wildfire simulations[a].

Parameter	Percent moisture
1 hour	4
10 hour	5
100 hour	7
Live herbaceous	60
Live woody	62

[a]Personal communication. Fire management and silviculture specialists, Lakeview Ranger District, Fremont-Winema National Forest.

[15] See: http://www.fpa.nifc.gov and http://wfdss.usgs.gov/wfdss/WFDSS_About.shtml.

are predicted by the MTT fire spread algorithm depending on the direction the fire encounters a pixel relative to the major direction of spread (i.e. heading, flanking, or backing fire) (Finney 2002). The conditional burn probability for a given pixel is an estimate of the likelihood that a pixel will burn given a random ignition somewhere in the watershed under the weather conditions represented in the simulation. Random ignitions were also allowed to originate outside the watershed to include wildfire events that burned into the watershed. The treated and untreated landscapes were simulated with 10,000 wildfires to generate burn probability and intensity (flame length) surfaces at 90 m resolution.

Carbon loss—To quantify the potential effects of wildfire on carbon stocks, each possible stand condition (as represented by the GNN treelist data—both treated and untreated) for each possible wildfire intensity (as represented by flame length category) was burned through FFE-FVS. Each stand condition in the study area was burned within FFE-FVS under a pre-defined surface fire flame length ranging from 0.5 m to 10 m in 0.5 m increments (SIMFIRE and FLAMEADJ keywords). The post-wildfire carbon reports in FFE (FVS_Carbon and FVS_Hrv_Carbon reports) were then examined to determine the amount of carbon in each carbon pool post burning at each flame length. Treatments and wildfire were simulated within the first four years of the simulation; the FFE-FVS outputs from the fifth simulation year were used as the results. The result was a carbon loss function for each stand condition representing all the possible post-wildfire carbon stocks by wildfire intensity class including no wildfire. Carbon loss is defined here as the reduction in post-wildfire carbon stocks for a given wildfire intensity class when compared to the carbon stocks present if no wildfire occurred; the amount reduced being equivalent to CO_2 emissions to the atmosphere from the fire. For treated stand conditions, carbon loss included the sum of carbon loss from treatment and from wildfire. Treatment carbon losses occur as a result of the disposal of non-merchantable material either removed or left on site, carbon losses associated with the end-use and fate of merchantable material removed and from CO_2 emissions from the under burns. The fate of the non-merchantable material removed was not accounted for; it was treated as a CO_2 emission at the time of treatment.

The current configurations of FVS and Randig do not allow for exact matching of fire behaviors. Randig reports total flame length of the surface and, if initiated, crown fire. In contrast, the FVS FLAMEADJ keyword does not allow for specifying a total flame length (surface and crown); rather it allows specification of a flame length for surface fires only. Moreover, FVS FLAMEADJ will not simulate crown fire initiation if it is parameterized with only a fire flame length (except as reported in the potential fire report). To simulate crown fires in FVS, we calculated a critical flame length (representing the threshold flame length between a surface fire and a crown fire) and imposed 100 percent crown consumption (via parameter 3 of the FLAMEADJ keyword) when the surface fire flame length exceeded the critical flame length.

Expected Carbon

The carbon stocks representing the amount of stored carbon post-wildfire (for untreated stand conditions) and post-treatment and post-wildfire (for treated stands) was matched with the burn probability data to calculate expected carbon for each 90- by 90-meter pixel as follows:

$$E[C]_{LS_j} = \left[\sum_{i=0}^{20}[BP_{ij} \times SC_{ij}]\right] + WPC_j \qquad (1)$$

where:

$E[C]_{LSj}$ = Expected carbon (mass per unit area) post-wildfire for the jth pixel and LS = TRT for the treated landscape and NO-TRT for the untreated landscape.

BP_{ij} = Conditional burn probability of wildfire intensity class *i* reaching the pixel *j*; where:

$\sum_{i=1}^{20} BP_{ij} = BP_j$ where BP_j is the overall burn probability of wildfire reaching pixel *j*

$BP_{0\,j}$ = Conditional probability of no fire = 1-BP_j;

and $\sum_{i=0}^{20} BP_{ij} = 1$

$SC_{i\,j}$ = total stand carbon, post-wildfire of wildfire intensity class *i* burning in pixel *j*; i=1 to 20.

SC_{0j} = total stand carbon in pixel *j* if no wildfire occurs.

WPC_j = carbon stored in wood products from treatment in pixel *j*.

For the untreated landscape, $WPC_j = 0$ for all *j*. For treated pixels on the treated landscape, SC_{ij} represents total stand carbon post treatment and post-wildfire for intensity class *i* burning in pixel *j* for *i* = 1 to 20 and for treated pixels on the treated landscape, SC_{0j} represents total stand carbon post treatment if no wildfire occurred.

Carbon offset calculations—The expected carbon offset is calculated for each 90- by 90-meter pixel by comparing the expected post-wildfire amount of carbon stored in the pixel post-treatment (if the pixel is treated) for the treated landscape with the amount of carbon stored in the same pixel post-wildfire on the untreated landscape. If the carbon offset is positive—meaning that the amount of carbon stored on the treated landscape is greater than the amount of carbon stored on the untreated landscape—then a positive CO_2 emission reduction benefit occurs as a result of undertaking the treatments.

The expected carbon offset, $E[(\Delta C)]$, for the treated landscape is calculated as follows:

$$E[(\triangle C)] = \sum_{j=1}^{n} (E[C]_{TRT_j} - E[C]_{NO-TRT_j}) \tag{2}$$

where:

n = is the total number of pixels in the landscape

$E[C]_{TRT_j}$ = is the expected carbon post treatment and wildfire in pixel j; treated landscape

$E[C]_{NO\text{-}TRT_j}$ = is the expected carbon post-wildfire, pixel j; untreated landscape, and

$E[C]_{TRT_j} - E[C]_{NO\text{-}TRT_j}$ = the carbon offset occurring in pixel j as a result of treatment.

$E[(\Delta C)] > 0$ is a necessary condition for the offset to be used as mitigation for CO_2 emissions from an unrelated source. However, if $E[(\Delta C)] < 0$, then the fuel treatment project resulted in more CO_2 emissions (less stored carbon) than if the landscape remained untreated. This does not mean the fuel treatment project should not be implemented; rather, those other benefits of conducting the fuel treatment project (e.g., resource protection, lowering community wildfire risk) come with a net loss of carbon from the landscape to the atmosphere.

Results

Fuels treatment had the desired effect of reducing the likelihood of fire reaching a given pixel as measured by burn probability (figs. 2 and 3; table 3). For untreated stands on the treated landscape, the likelihood of wildfire spreading to untreated stands was also reduced as a result of applying the treatments (fig. 4a). For treated stands, there was a shift in the conditional burn probability distribution making low intensity fires much more likely than if the stands had not been treated, as well as reducing the overall likelihood of wildfire in those stands (fig. 4b). This lowered the average per acre carbon loss from wildfire in post-treated stands when compared to untreated stand conditions (fig. 4b). Comparing the frequency distribution of the wildfire simulations for both landscapes also shows that the fuel treatments were effective in reducing the intensity and extent of wildfire (fig. 5). Average fire size on the treated landscape was 32 percent lower than average fire size on the untreated landscape. Secondly, the largest fire simulated on the treated landscape was 15,000 acres compared to over 19,000 acres for the untreated landscape. In general, the treated landscape experienced a greater number of smaller sized wildfires when compared to the untreated landscape.

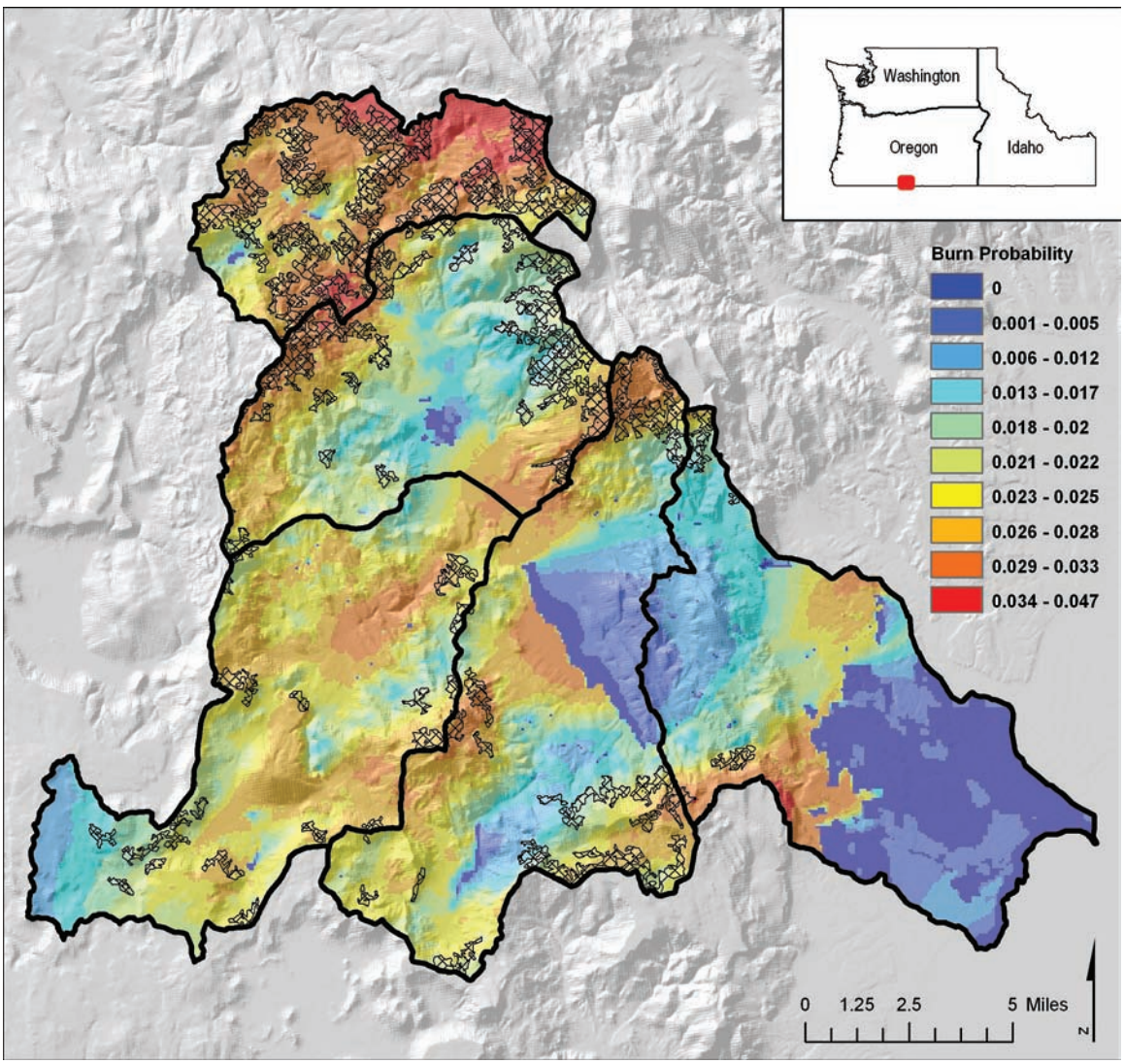

Figure 2—Conditional burn probability map from 10,000 randomly located ignitions and 8-hour burn periods (untreated landscape).

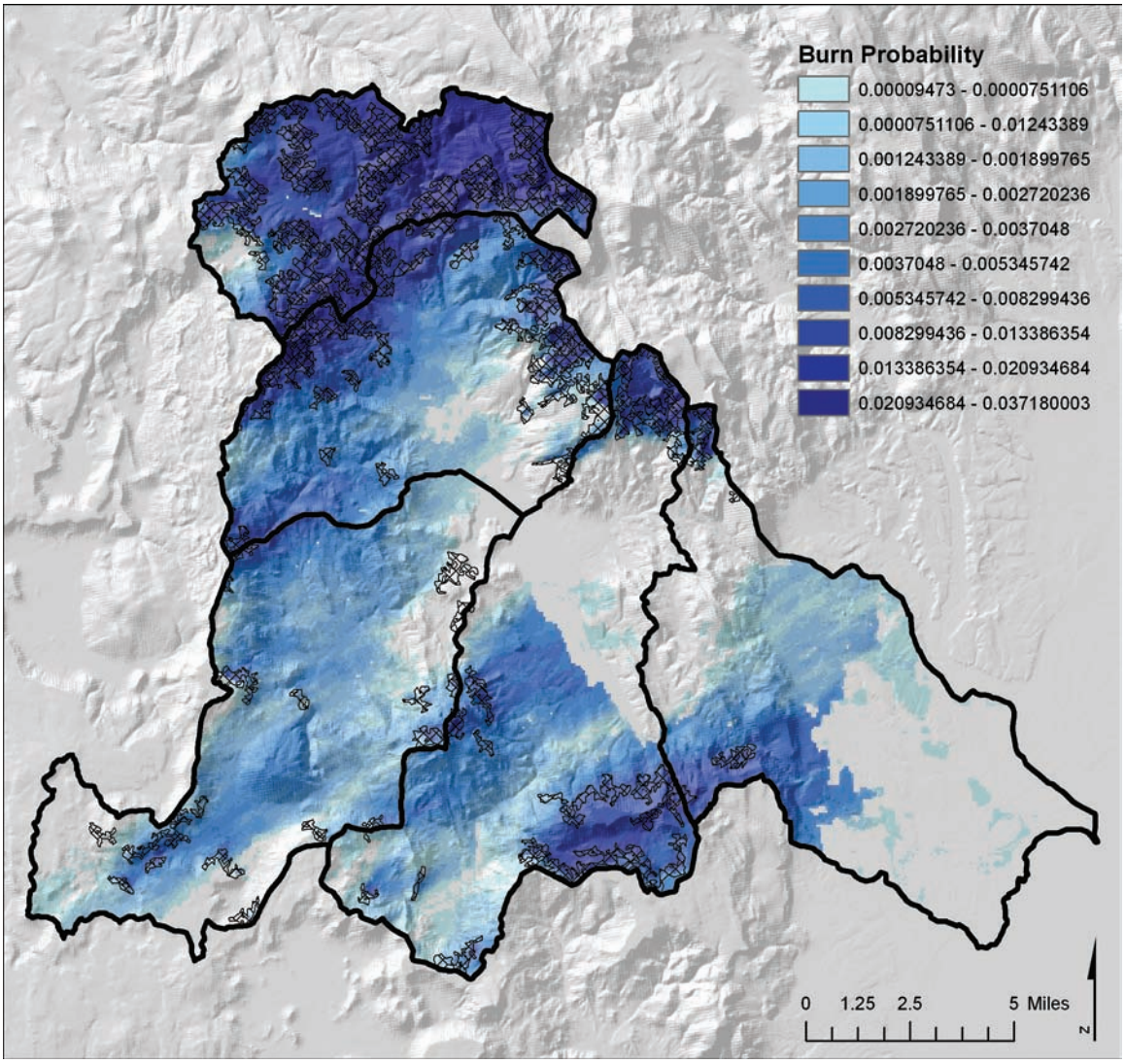

Figure 3—Difference in burn probability between untreated and treated landscape. Areas not shaded within watershed boundary had differences less than 0.0025.

Table 3—Mean overall conditional burn probabilities for Drews Creek—forestland.

	Treated landscape	Untreated landscape	Difference
Treated stands	0.01235	0.02602	–0.01367
Untreated stands	0.01709	0.02106	–0.00397
All stands	0.01665	0.02152	–0.00487

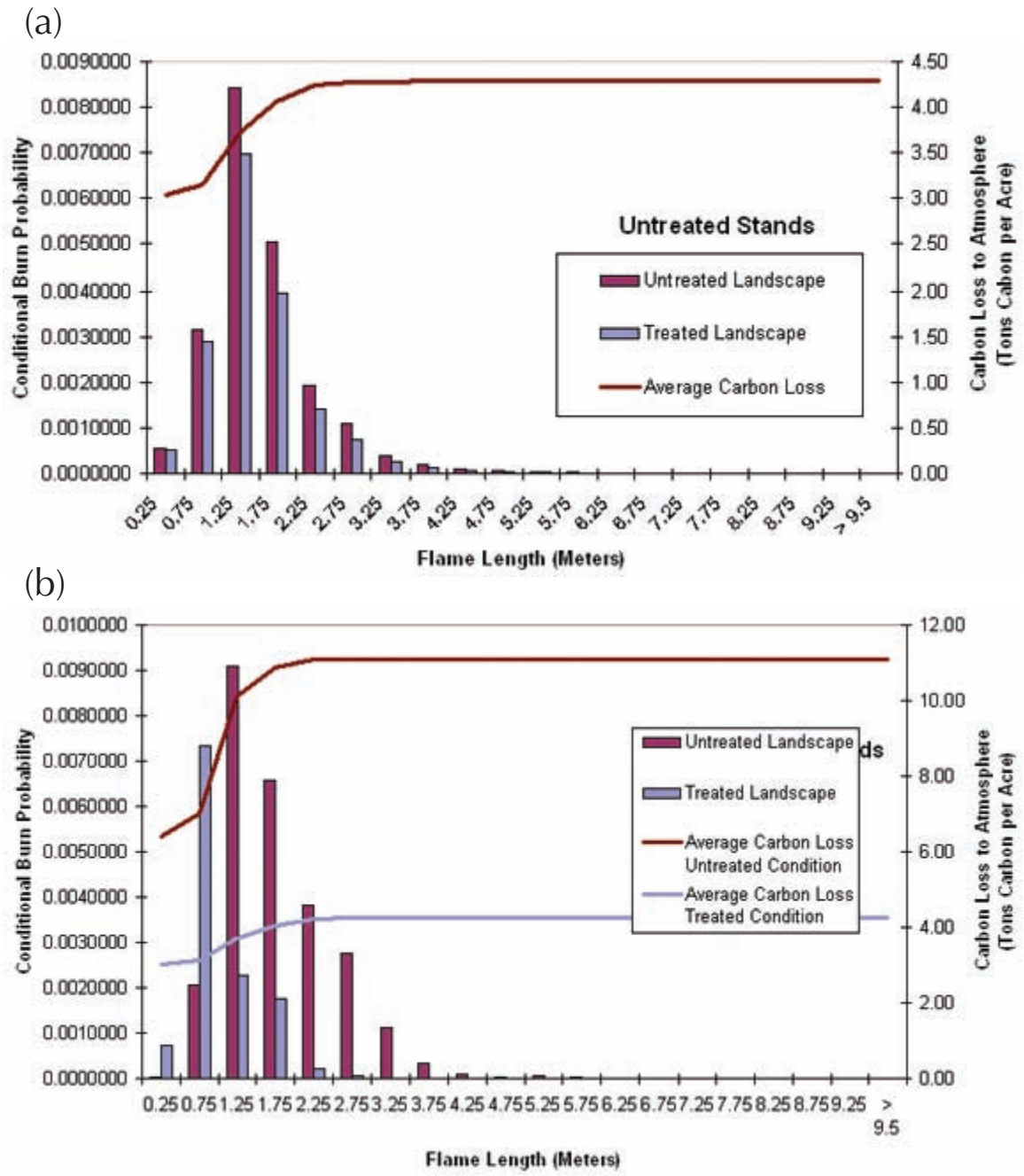

Figure 4—Frequency histograms for burn probabilities for untreated (a) and treated (b) scenarios for the Drews Creek watershed study area.

When compared to the carbon stocks post-wildfire for the same areas on the untreated landscape, fuels treatment resulted in a net carbon loss to the atmosphere of –303,458 tons (table 4) (–23.7 tons per treated acre). The expected carbon stocks within the untreated area of the treated landscape did increase to a small degree (3,536 tons; or 0.027 tons per untreated acre) as a result of the treatment's effect of reducing the likelihood that wildfire reaches the untreated stands. However, the cost in terms of carbon released to the atmosphere as a result of treatment was overwhelming with a net, negative carbon offset, $E[(\Delta C)] = -299{,}622$ tons (table 4) (–2.13 tons per forested acre).

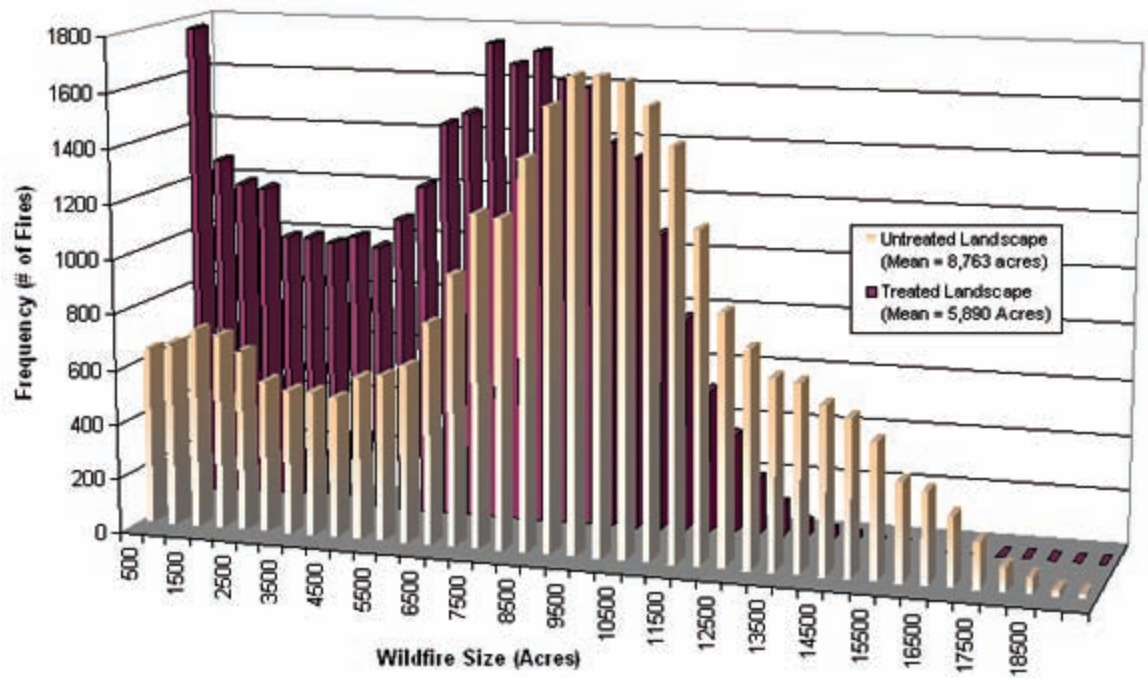

Figure 5—Frequency histogram of simulated fire sizes for untreated and treated scenarios for the Drews Creek watershed study area.

Table 4—Expected stored carbon immediately after wildfire—Drews Creek—forestland

	Treated landscape	Untreated landscape	Difference
		short tons carbon	
Treated stands[a]	538,940	842,398	–303,458
Untreated stands	2,961,484	2,957,948	3,536
		OFFSET	–299,622

[a]This is the area selected for treatment on the treated landscape but remained untreated on the untreated landscape. The treated stand area plus the untreated stand area equals the total forestland area.

Biomass removed through thinning from below and movement of the excess material (slash) resulting from harvest activities removed 215,071 tons of carbon (16.8 tons per treated acre) representing 19.1 percent of the total biomass in treated stands. Of this amount, 159,440 tons (74 percent) was emitted to the atmosphere (12.4 tons per treated acre) with the remaining 55,631 tons (26 percent) remaining stored in long-lived wood products (4.3 tons per treated acre). Under burning emitted another 111,893 tons carbon (8.7 tons per treated acre) representing13.3 percent of the total biomass in treated stands. In total, carbon lost to the atmosphere from the fuel treatment activity totaled –271,333 tons carbon (–21.2 tons per treated acre) (table 5). In comparison, only an expected 3,700 tons (0.21 tons per acre) of avoided carbon loss accrued to the treatment polygons as a result of the treatment's effect of reducing both the likelihood and intensity of wildfire in treated stands. Similarly, only 3,087 tons of expected avoided carbon loss accrued to the untreated polygons (0.025 tons per acre) as a result of the treatment's effect of reducing the likelihood of wildfire, for a total benefit of 6,787 tons of expected

Table 5—Expected carbon emissions[a]—Drews Creek—forestland.

Emission source	Treated landscape	Untreated landscape	Difference
		short tons carbon	
Merchantable material removed From treatment, but not stored	–41,884	0	–41,884
Non-merchantable material removed from thinning treatment	–62,796	0	–62,796
Non-merchantable material removed from FUELMOVE treatment	–54,760	0	–54,760
Total, thinning activity	–159,440		–159,440
Prescribed fire in treatments	–111,893	0	–111,893
Total, all treatment practices	–271,333		–271,333
Wildfire treated[b] stands	–157	–3,857	3,700
Wildfire non-treated stands	–8,936	–12,023	3,087
		OFFSET	–264,546

[a] Negative sign indicates a loss of stored carbon to the atmosphere.

[b] This is the area selected for treatment on the treated landscape but remained untreated on the untreated landscape. The treated stand area plus the untreated stand area equals the total forestland area.

carbon loss avoided (0.048 tons per forested acre). The net expected carbon loss accruing to the treated landscape when compared to the untreated landscape is –264,546 tons (–1.9 tons per forested acre) carbon (table 5)16.

Discussion

The Law of Averages

The question is—if the implementation of fuels treatments within the Drews Creek watershed had the beneficial effect of reducing the likelihood of wildfire intensity and extent as simulated in this study, why is the expected carbon offset from fuels treatment so negative? The answer lies in the probabilistic nature of wildfire. Fuels treatment comes with a carbon loss from biomass removal and prescribed fire with a probability of 1. In contrast, the benefit of avoided wildfire emissions is probabilistic. The law of averages is heavily influenced that given

[16] FVS does not account for all carbon losses as emissions and as a result the offset as calculated in table 4 (from comparing carbon stock changes) does not balance with the offset calculated in table 5 (from comparing carbon emission changes).

a wildfire ignition somewhere within the watershed, the probability that a stand is not burned by the corresponding wildfire is 0.98 (1 minus the average overall conditional burn probability in table 3). The simulations also included all possible wildfire scenarios from one random ignition within the Drews Creek watershed under the severe weather and fuel moisture conditions that *could* result in the problem fire. But, in many of the simulations, the problem fire did not occur (fig. 5).

Thus, the expected benefit of avoided wildfire emissions is an average that includes the predominant scenario that no wildfire reaches the stand. And if the predominate scenario for each stand is that the fire never reaches it, there is no avoided CO_2 emissions benefit to be had from treatment. So even though severe wildfire can be a significant CO_2 emissions event, its chance of occurring and reaching a given stand relative to where the wildfire started is still very low, with or without fuel treatments on the landscape. Further, when wildfire does occur, the immediate effect is to transfer the stored carbon on the landscape from the live tree pool to the standing dead tree pool. So, most of the carbon stored before wildfire is still there after wildfire, and the amount of carbon loss compared to the stored carbon before wildfire is still relatively small. Had this study included the continued carbon loss over time from the decomposition of the dead tree pool post-wildfire, then the total avoided wildfire emissions would have been greater.

Life Expectancy of Fuel Treatments

The carbon offset result of –299,622 tons carbon (–2.13 tons per forested acre) is the carbon offset of avoiding the possibility of a severe wildfire based on a single wildfire ignition under extreme weather and fuel moisture conditions. Our study results show that for any given ignition in the year after the completion of fuel treatments, the expected avoided carbon loss from one wildfire ignition is 6,787 tons carbon. But the Drews Creek watershed experiences on average, 14 wildfire ignitions a year. For example, if 1/3 (i.e., say 5) of the 14 ignitions per year for Drews Creek occurred during the severe weather and fuel moisture conditions used in this study and independence is assumed in the wildfire outcomes from one ignition to another, then the expected avoided carbon loss from the same fuel treatment investment (and treatment carbon loss) is 33,935 (=5*6,787) tons carbon[17]. This amount is still not enough benefit to make the carbon offset positive for the year following the completion of the fuel treatment project.

But, the fuel treatment project is actually an investment in reducing wildfire risk over an extended period of time beyond the year following treatment. If the life expectancy of the implemented fuel treatments is at least 9 years, then the expected carbon offset becomes positive. That is, if for each year, the fuel treatments avoid an expected 33,935 tons of carbon loss to the atmosphere, then 305,415 (=9*33,935) tons of expected carbon loss is avoided after 9 years for the same –299,622 treatment carbon loss investment[18]. While the carbon offset from fuel treatments is clearly negative for any one ignition in the year immediately

[17] This is a simplifying assumption used for illustrative purposes. If each random ignition wildfire outcome is an independent event—meaning the spread and intensity of the wildfire from the second ignition is not influenced by how the wildfire burned from the first ignition—the overall burn probability of wildfire reaching the stands from 5 random ignitions on the landscape is 5 times the probability of it reaching the stand from one ignition. Any dependency would lower this result, but the concept still holds.

[18] Again, this assumes independence between wildfire outcomes from one year to the next. In this case, this is a conservative estimate. It may be that the burn probability landscape intensifies—meaning the probability of wildfire not reaching a stand (1-*BP*) decreases if the problem fire event does not occur in the preceding year.

following the completion of treatments, it may become positive within the course of the shelf life of the treatment's effectiveness in reducing wildfire risk. Further, accounting for the consequences of continued decomposition from the in-stand dead tree pools and continued sequestration from the live tree pool not lost to wildfire as a result of treatment improves the carbon offset potential of the fuel treatment. While not accounted for in this study, the expected carbon benefit from the fuel treatment project would be better reflected as the cumulative expected avoided carbon loss and improved carbon sequestration gain over time from avoiding the problem fire from multiple ignitions in multiple years after treatment.

Future Work

This study had several limitations. First, the treatment design did not explicitly minimize the carbon loss from conducting the fuel treatment while at the same time maintaining the same level of effectiveness in reducing the risk of catastrophic wildfire. Better utilization of the non-merchantable material removed in products that continue to store carbon would reduce the carbon cost of treatment. Second, the analysis does not account for the continued loss of carbon post-wildfire from decomposition of burned material; nor does the study account for the continued growth and corresponding sequestration that occurs both in treated stands (where fire severity and extent is reduced) and in untreated stands (where the chance of wildfire reaching them is reduced) outside of the treatment areas. Third, we did not convert the conditional burn probabilities of wildfire occurring given an ignition on the landscape to the absolute probability of wildfire occurring. This step would take into account the probability of ignition and could take into account other factors such as the improved effectiveness of suppression afforded by treatment. Finally, modeling the shelf life of fuel treatments using burn probability maps for more than one year following treatment that take into account the wildfire outcomes in previous years might better reveal the cumulative carbon offset of conducting the fuel treatment project.

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The content of this paper reflects the views of the authors, who are responsible for the facts and accuracy of the information presented herein.

Forest Biomass and Tree Planting for Fossil Fuel Offsets in the Colorado Front Range

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Abstract—This study estimates the amount of carbon available for removal in fuel reduction and reforestation treatments in montane forests of the Colorado Front Range based on site productivity, pre-treatment basal area, and planting density. Thinning dense stands will yield the greatest offsets for biomass fuel. However, this will also yield the greatest carbon losses, if the removals remain on site to decompose. Stands that regrow the biomass removed will recover the carbon. However, if these treatments are maintained at low basal areas, then the treatments may permanently convert a large quantity of carbon from the forest to the atmosphere.

Keywords: ponderosa pine, carbon, wildfire risk, FVS, biomass

Introduction

Over the past century, the exclusion of wildfires in ponderosa pine/Douglas-fir (*Pinus ponderosa* / *Pseudotsuga menziesii*) forests has increased tree density in most landscapes. While this increase in tree density serves as a carbon sink, these same areas are at higher risk for a high severity fire, which could lead to a substantial initial release of carbon if trees are killed and the fire consumes the forest floor, foliage, and wood (Hurteau and others 2008; Hurteau and North 2009). Several recent high severity wildfires in the ponderosa pine/Douglas-fir forests of the Colorado Front Range have converted dense forests into large areas with no living trees and minimal ability to regenerate (Romme and others 2003). Since these areas have little to no regeneration, natural forest regrowth will not replace the carbon lost to combustion and decomposition and the area will become a carbon source (Dore and others 2008; Kashian and others 2006) for the foreseeable future. These fires have prompted the implementation of fuel reduction treatments intended to reduce crown fire risk. These treatments often involve removing small-diameter, non-merchantable trees that established over the past century. Since the biomass removed is non-merchantable, it is typically piled and burned or masticated, which moves carbon dioxide (CO_2) stored in the forest to the atmosphere. Furthermore, the lack of removable merchantable material makes these treatments costly.

The increase in atmospheric CO_2 and its role in global climate change have encouraged the search for ways to reduce or offset greenhouse-gas emissions. Emissions cap-and-trade programs or voluntary carbon trading programs can reduce CO_2 input to the atmosphere by creating a market aimed at decreasing fossil fuel emissions (Ellerman and others 1998; Petty and Ball 2001; Zhang and Folmer 1995). Carbon credit trading allows industries to buy credits from

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industries or entities that have reduced their emissions. Companies can choose to invest in reforestation projects that remove CO_2 from the atmosphere, or to use alternative energy sources for fossil fuels in heat and power generation. The large area currently storing carbon but in need of thinning for fire hazard reduction could benefit from these programs.

Biomass harvested from fuel reduction treatments could substitute for fossil fuels currently used for heat and power generation. The credits generated by the use of the biomass fuel could help compensate for the costs of the management activities, depending on the price of the credits. Also, planting trees in burned areas can help restore forests and promote carbon sequestration. While the initial harvests to reduce crown fire risk may result in immediate CO_2 offsets if used for biomass energy, the maintenance of low forest density to maintain low crown fire risk will reduce potential carbon stored on the landscape. This study estimates the amount of carbon available now for removal in fuel reduction and reforestation treatments in the ponderosa pine/Douglas-fir forests of the Colorado Front Range based on site productivity, pre-treatment basal area, and planting density. We also estimate the carbon storage potential of tree planting on areas not expected to regenerate naturally. Finally, we estimate the potential carbon offsets and associated crown fire risk for different management scenarios.

Methods

Study Area

Our study area was located within the Pike and San Isabel National Forest (PSINF) in Colorado, southwest of Denver (fig. 1). Tree species are ponderosa pine, Douglas-fir, quaking aspen (*Populus tremuloides* Michx.), lodgepole pine (*Pinus contorta* Douglas ex Louden), Engelmann Spruce (*Picea engelmannii* Parry ex Engelm.), and subalpine fir (*Abies lasiocarpa* [Hook.] Nutt.). Site index on the PSINF varies from 30 to 80 ft (index age of 100 years).

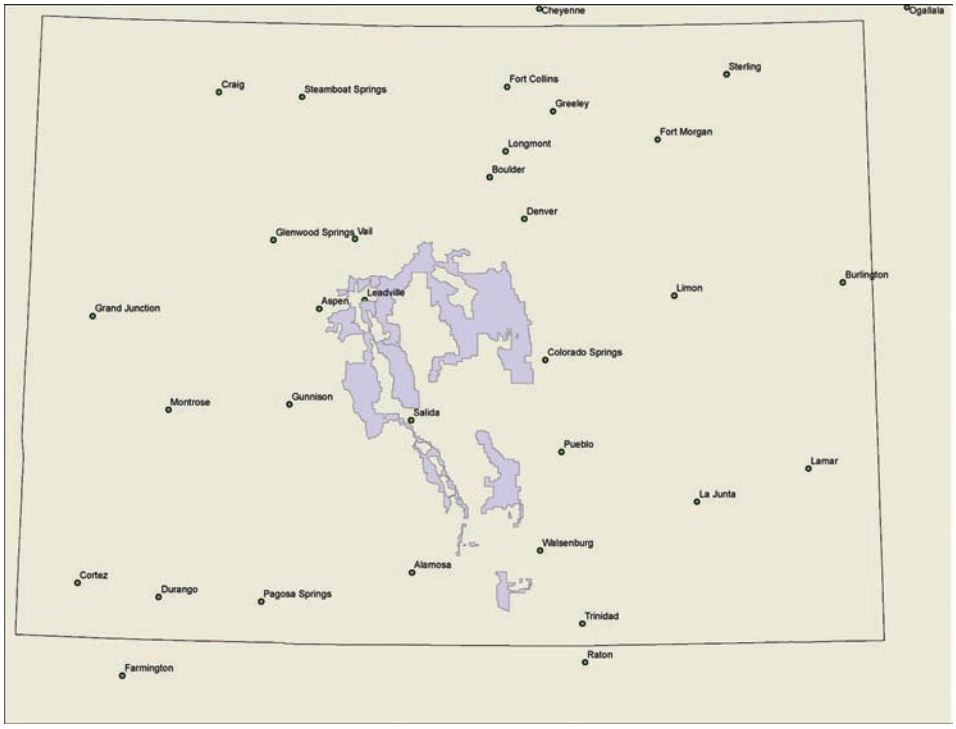

Figure 1—Location of the Pike-San Isabel National Forest (shaded) within the state of Colorado.

We studied the dry ponderosa pine/Douglas-fir forest because this is the forest type that has experienced severe wildfires and would likely be the area treated for fuel reduction. The dry ponderosa pine/Douglas-fir forests of the PSINF are found from 6,500 to 9,000 ft in elevation. Ponderosa pine dominates the south and west facing slopes and the Douglas-fir dominates the northern slopes. The understories are typically grassy, shrubby, or both. Soils are gravelly coarse sandy loams derived from weathered Pikes Peak granite (Moore 1992).

Overview

We modeled two scenarios (Fuel Reduction Treatment and Reforestation) to develop estimates for the amount of carbon that is currently and potentially available. We calibrated the Forest Vegetation Simulator (FVS) forest growth model to match growth conditions found on the PSINF and modeled carbon accumulation for different management and site index scenarios. Specifically, carbon accumulation was estimated by 1) using USDA Forest Service inventory data to determine the range of site indices and actual forest growth for areas within the PSINF either in need of fuels reduction or in areas burned in the Hayman wildfire, 2) calibrating FVS to match actual forest growth for the different site index values, 3) using the FVS model to model organic material accumulation for a range of site index values, and 4) applying literature-based values to estimate carbon content of biomass and carbon dioxide equivalent in metric tons per acre (MT/ac). We included belowground biomass and organic material accumulation (forest floor) in our carbon estimates.

The Forest Vegetation Simulator

The Forest Vegetation Simulator (Dixon 2002) is an individual tree and stand level growth and yield model that is based on the Prognosis model (Stage 1973). The Extensions to Prognosis began in 1978, and GENGYM (Edminster and others 1991), today known as the "Central Rockies Variant," was developed in 1991. Like the Prognosis model, the Central Rockies Variant is a set of allometric equations that defines growth rates for tree species in New Mexico, Arizona, Colorado, and South Dakota. This variant is broken into five sub-variants. The sub-variant specific to the PSINF region was used to generate the information in this study.

Scenarios

Current management guidelines in the PSINF for fuels reduction and forest restoration involve reducing standing forest biomass to a target basal area of 40 to 60 ft^2/ac. This guideline is based on research done by Merrill Kaufmann and others (Kaufmann and others 2000; Kaufmann and others 2001; Romme and others 2003) in the area surrounding Cheeseman Reservoir, Deckers, Colorado.

Fuel reduction treatment—We downloaded inventory data from the PSINF Resource Information System (RIS) database. The RIS database was queried for stand identification numbers of suitable ponderosa pine/Douglas-fir sites. Stands were sorted into 4 site index groups (45, 55, 65, and 75). Using FSVeg, specific stand exams were downloaded and entered into the FVS program.

We used FVS to grow each site index group until a specific management criterion was met and then had the model simulate thinning. For each of these groups, the FVS model was set up to grow to 6 different basal areas (90, 100, 110, 120, 130, and 140 ft^2/ac). Each basal area class was thinned to three post-treatment basal area targets (40, 50 and 60 ft^2/ac). These values encompass the range of possible management targets that a manager may chose, given the flexibility in

potential management practices. The *ThinBBA* keyword (thin from below to a specific basal area target) was used in the FVS model to allow the model stand to grow to and be thinned to a specified basal area target.

We simulated two additional scenarios to assess the potential carbon offsets and associated fire risk for stands managed to maintain fuel treatment effectiveness. For both scenarios, we chose to model the stands with a site index of 45. In the first scenario, the initial basal area was 100 ft^2/ac, and then it was thinned from below to 50 ft^2/ac and allowed to grow back to 100 ft^2/ac and thinned again to 50 ft^2/ac. In the second scenario, the initial basal area was 160 ft^2/ac, and then it was thinned from below to 50 ft^2/ac and only allowed to grow to 100 ft^2/ac and thinned again to 50 ft^2/ac. In both scenarios, the fire risk associated with the management regime was assessed with the fire and fuels extension (FFE) to FVS. FFE provides estimates of the torching and crowning index based on canopy base height, canopy bulk density, and an estimated fuel model. The torching index is the windspeed at 20 ft required to initiate a passive crown fire. The crowning index is the windspeed at 20 ft required to sustain an active crown fire (Scott and Reinhardt 2001). These scenarios were simulated for a 200 year period.

Reforestation—We selected ponderosa pine stands from the post-Hayman fire reforestation needs GIS coverage map. The polygons identified areas within the Hayman wildfire that had no living trees and lacked a seed source. These polygons had records that include stand inventory codes that allow the user to download stand exams from the PSINF database using the Forest Service's FSVeg network application.

The reforestation scenario used a matrix of site indices and planting densities to compare the differences in carbon storage for these scenarios. Site index values were taken from PSINF stand inventories completed between 1978 and 2000. Site index values ranged from 30 to 58, with only a few stands having a site index >55. We modeled growth for site index values of 35 to 55 in increments of 5. We modeled *surviving* tree density from a planting density of 100, 200, and 300 trees per acre for each site index value. The "surviving planting number" is the number of seedlings per acre remaining on a site after any post-planting mortality. These surviving planting densities were chosen to represent the tradeoff between high density with rapid carbon accumulation and frequent thinning to reduce fire risk, to low density with slow carbon accumulation but few necessary thinnings. These scenarios were simulated for one harvest cycle (typically 80 to 100 years).

Similar to the 'biofuels' simulations, two additional scenarios were simulated to assess the potential carbon offsets and associated fire risk for reforested stands that would be managed to maintain fuel treatment effectiveness. For both scenarios, we chose to model the stands with a site index of 45 and initial planting density of 450 trees per acre. In the first scenario, we allowed the stand to grow with no thinning activities for 400 years. In the second scenario, we allowed the stand to grow to 100 ft^2/ac and thinned from below to 50 ft^2/ac. We repeated this management regime for 400 years. In both scenarios, the fire risk associated with the management regime was assessed with the fire and fuels extension (FFE) to FVS.

Calculating Carbon Stores

To determine the amount of carbon stored for each scenario, we used the 'FuelOut' keyword in FVS to request a report of biomass in all forest carbon pools. Forest floor (sum of surface litter, surface duff, and surface wood—mostly litter and duff) and total standing biomass (sum of standing live wood, large roots, standing snags, and foliage) were calculated and summed as the total biomass accumulation in a forest stand. Although FVS now has the ability to calculate forest carbon, when the simulations were run, the extension was

still in development. We estimated the belowground biomass (large roots) as 25 percent of the aboveground biomass (Jackson and Chittenden 1981). We assumed that biomass was 50 percent carbon (Schlesinger 1997). Carbon trading uses CO_2 instead of carbon, so we multiplied total carbon by 3.67, the ratio of the molecular weights of CO_2 to carbon (Chicago Climate Exchange, Inc. 2004; US Department of Energy 2007). We report our values in metric tons CO_2 equivalent per acre (MT CO_2/ac)

Results

Fuel Reduction Treatment: Initial CO_2 offsets

Biomass available from fire hazard reduction activities in ponderosa pine/Douglas-fir forests of the PSINF varied by pre-treatment basal area, site index, and the post-treatment basal area. Dense stands will yield the most biomass and offsets, especially if they are thinned to the lower target basal area of 40 ft^2/ac (table 1). For example, stands with a site index of 45 and pre-treatment basal areas between 90 and 140 ft^2/ac will generate fossil fuel offsets between 18.6 to 43.6 MT CO_2/ac if thinned to a target basal area of 40 ft^2/ac (table 1). If these same stands were thinned to a target basal area of 60 ft^2/ac, they will generate fossil fuel offsets between 9.0 to 32.5 MT CO_2/ac (table 1). If these stands had a site index of 75, they would generate fossil fuel offsets between 22 to 55.3 CO_2 MT/ac if thinned to a target basal area of 40 ft^2/ac and 10.5 to 41.3 CO_2 MT/ac if thinned to a target basal area of 60 ft^2/ac (table 1).

Maintenance of Fuel Reduction Treatment: Future CO_2 Offsets and Fire Risk

Maintaining ponderosa pine/Douglas-fir forests between 50 and 100 ft^2/ac reduces fire risk by increasing the windspeed required to sustain an active crown fire. Before the initial harvest, the 100 ft^2/ac and the 160 ft^2/ac stands had low crowning indices, 26 (fig. 2) and 13 (fig. 3) mph, respectively. After the initial entry, crowning indices increased substantially (figs. 2 and 3). As the stands accu-

Table 1—Average (± standard error, in parentheses) metric tons of CO_2 equivalent per acre available for immediate biomass removal for different site productivities, pre-treatment basal area[a], and residual post-treatment basal area of ponderosa pine/Douglas-fir stands in need of fuel reduction treatments on the Pike and San Isabel National Forest, Colorado.

Pre-trt BA [a]		Site index = 45				Site index = 55				Site index = 65				Site index = 75		
	Target basal area (ft^2/ac)															
	0	40	50	60	0	40	50	60	0	40	50	60	0	40	50	60
ft^2/ac	----------Metric tons of CO_2 equivalent per acre (± standard error)----------															
90	39.1 (1.2)	18.6 (0.77)	13.7 (0.57)	9.0 (0.54)	42.4 (1.2)	20.1 (0.75)	14.8 (0.70)	9.9 (0.59)	47.5 (2.0)	22.8 (1.12)	17.2 (1.04)	10.9 (0.65)	46.5 (1.7)	22.0 (1.18)	15.7 (0.96)	10.5 (0.87)
100	45.3 (1.4)	23.9 (0.90)	19.9 (0.79)	13.9 (0.63)	48.0 (1.2)	25.1 (0.81)	19.9 (0.81)	14.7 (0.70)	52.1 (1.7)	27.4 (1.10)	21.4 (1.03)	16.2 (0.81)	53.5 (1.9)	28.4 (1.14)	21.7 (0.94)	16.2 (0.75)
110	50.4 (1.6)	28.9 (1.07)	23.6 (0.97)	18.3 (0.86)	52.9 (1.2)	30.0 (0.87)	24.4 (0.84)	19.1 (0.77)	57.4 (1.7)	32.7 (1.23)	26.3 (1.10)	20.9 (0.91)	60.1 (2.1)	34.1 (1.37)	27.8 (1.25)	21.4 (1.00)
120	55.5 (1.8)	33.8 (1.22)	28.4 (1.11)	23.1 (0.99)	59.1 (1.2)	35.6 (0.91)	29.7 (0.86)	24.4 (0.84)	63.9 (1.8)	38.6 (1.29)	32.1 (1.20)	26.9 (0.96)	66.4 (3.0)	40.1 (2.07)	33.8 (1.87)	27.2 (1.64)
130	60.0 (1.8)	38.4 (1.36)	32.8 (1.22)	27.5 (1.22)	63.8 (1.4)	40.4 (1.05)	34.4 (0.98)	28.7 (0.88)	71.0 (2.3)	45.4 (1.75)	38.2 (1.28)	32.2 (1.25)	75.3 (3.3)	48.1 (2.33)	41.2 (2.13)	34.5 (1.96)
140	65.6 (2.1)	43.6 (1.63)	38.0 (1.50)	32.5 (1.43)	70.4 (1.5)	46.8 (1.12)	40.4 (1.02)	34.7 (0.94)	77.9 (2.2)	51.7 (1.63)	45.2 (1.46)	38.5 (1.32)	83.0 (3.8)	55.3 (2.74)	48.1 (2.6)	41.3 (2.34)

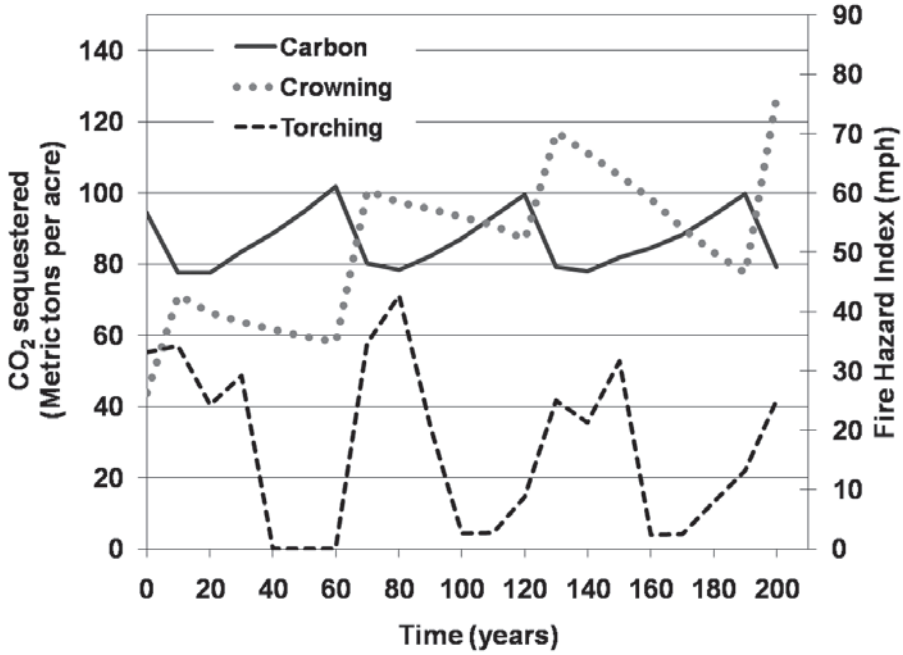

Figure 2—CO_2 equivalent per acre accumulated, potential CO_2 offsets, and the associated fire hazard for a stand maintained between 50 and 100 ft^2/ac. The pre-treatment stand basal area was 100 ft^2/ac and site index=45.

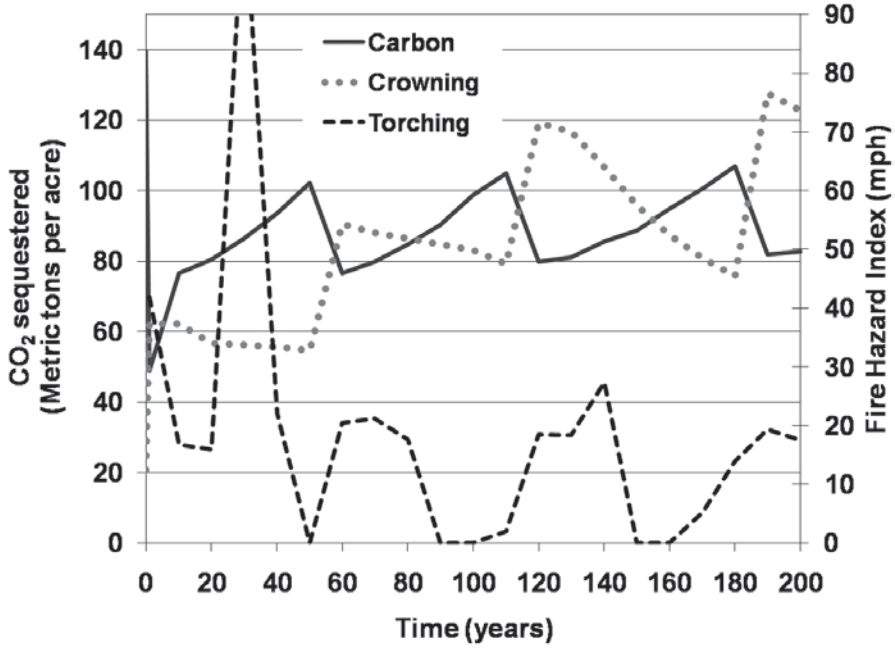

Figure 3—CO_2 equivalent per acre accumulated, potential CO_2 offsets, and the associated fire hazard for a stand maintained between 50 and 100 ft^2/ac. The pre-treatment stand basal area was 160 ft^2/ac and site index=45.

mulated more carbon, the crowning index decreased slightly, but never decreased to the pre-treatment value (figs. 2 and 3). This pattern continued throughout the simulation.

Although active crown fire risk was decreased, the success of thinning from below to prevent passive crown fire (crown fire initiation) was variable for both maintenance scenarios (figs. 2 and 3). For both scenarios, the torching index would increase substantially after each entry for about 30 years, but it would decrease due to the ingrowth of new tree regeneration.

The carbon consequences of maintaining stands at tree densities that reduce crown fire risk differed depending on the initial pre-treatment basal area. In the scenario where initial basal area was 100 ft^2/ac, thinning from below to 50 ft^2/ac would provide approximately 18 CO_2 MT/ac in the initial harvest (fig. 2). However, the scenario where initial basal area was 160 ft^2/ac and was thinned from below to 50 ft^2/ac would provide approximately 38 CO_2 MT/ac in the initial harvest (fig. 3). Throughout the next several rotations, thinning from below occurred whenever the basal area reached 100 ft^2/ac. For the scenario that was initially at 100 ft^2/ac, each maintenance harvest yielded approximately the same amount of CO_2 MT/ac as the initial harvest (fig. 2), replacing the carbon harvested each rotation. However, the scenario that was initially at 160 ft^2/ac only yielded approximately 20 CO_2 MT/ac in the maintenance harvests, never fully replacing the carbon harvested in the initial harvest (fig. 3).

Reforestation of Hayman Fire

The CO_2 equivalent accumulation rates for tree planting within the Hayman wildfire burn area varied with site index and planting density, but planting density was more important in determining carbon accumulation (table 2). After 90 years, forest CO_2 equivalent ranged from 22.5 to 50.1 MT CO_2/ac for a planting density of 100 trees per acre, 34.2 to 80.2 MT CO_2/ac for a planting density of 200 trees per acre, and 42.8 to 102.3 MT CO_2/ac for 300 trees per acre.

Reforestation: Future CO_2 Offsets and Fire Risk

The potential carbon offsets and associated fire risk for reforested stands differed depending on the management regime. The amount of CO_2 that accumulated after 400 years and available for an offset at the end of a 400-year rotation was 143 CO_2 MT/ac (fig. 4). For the stands that were maintained to reduce fire hazard, the CO_2 offsets from the harvests combined with the end of the 400-year rotation would yield 108 CO_2 MT/ac (fig. 5). The consequence of increased carbon sequestered in the *no thin* scenario was a higher risk for active crown fire with crowning indices ranging between 20 and 40 mph for the majority of the rotation (fig. 4).

Table 2—Metric tons of CO_2 equivalent per acre accumulated 90 years after a reforestation effort for different site productivities and ponderosa pine planting densities within the Hayman Fire on the Pike and San Isabel National Forest, Colorado.

	Planting densities (Trees per acre)		
Site Index	**100**	**200**	**300**
	Metric tons of CO_2 equivalent per acre		
35	27.5	41.3	51.9
40	29.7	46.1	57.3
45	34.8	54.0	67.4
50	39.6	63.6	79.8
55	50.1	80.2	102.3

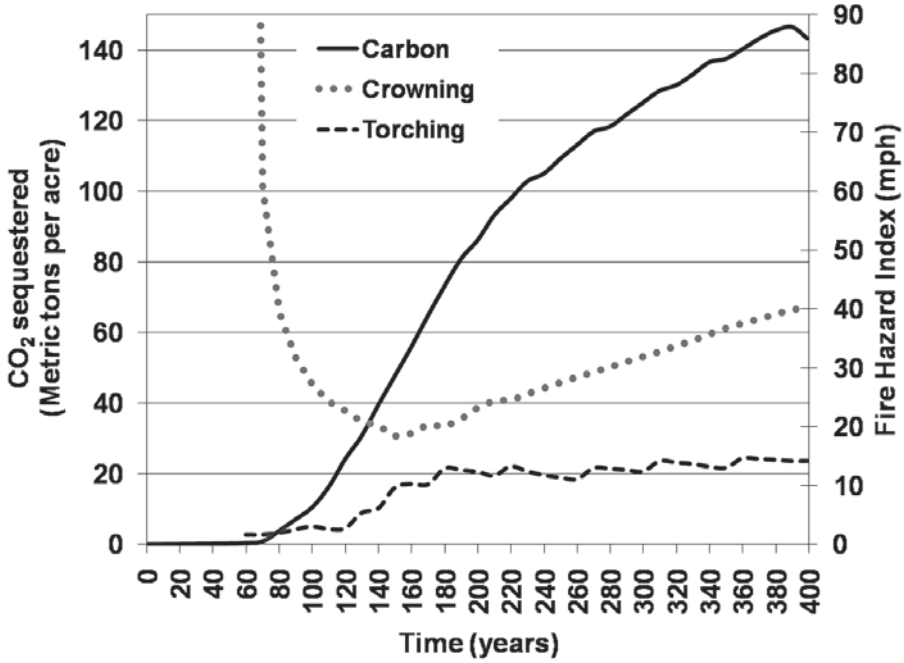

Figure 4—CO_2 equivalent per acre accumulated, potential CO_2 offsets, and the associated fire hazard for a reforested stand (site index=45) planted at a density of 450 trees per acre.

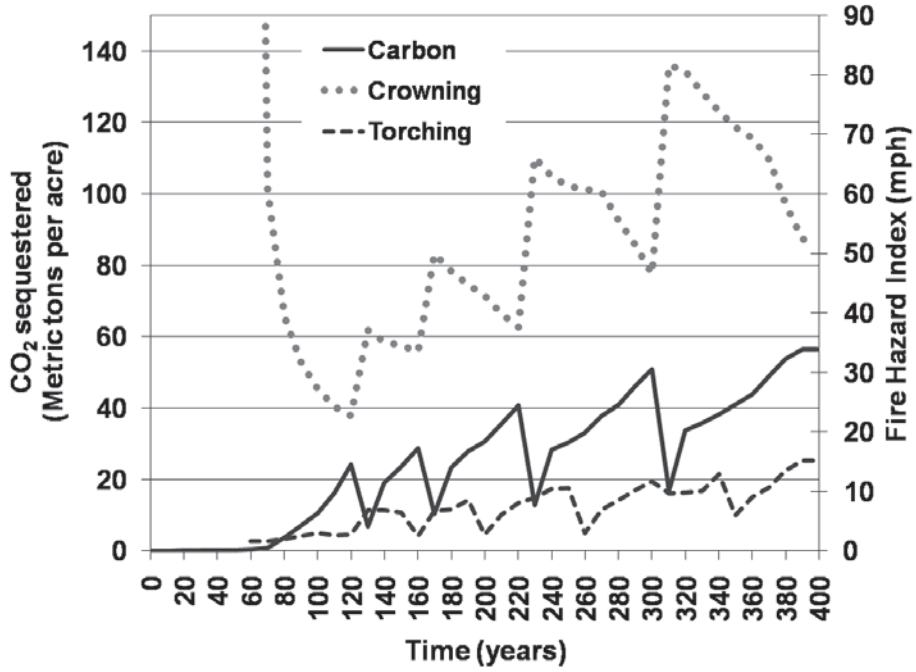

Figure 5—CO_2 equivalent per acre accumulated, potential CO_2 offsets, and the associated fire hazard for a reforested stand (site index=45) planted at a density of 450 trees per acre and maintained between 50 and 100 ft^2/ac.

In contrast, maintaining basal areas between 50 and 100 ft^2/ac resulted in lower active crown fire risk with crowning indices ranging between 40 and 80 mph (fig. 5). However, the risk of passive crown fire was high for both scenarios, with torching indices never exceeding 20 mph (figs. 4 and 5).

Discussion

The dense ponderosa pine/Douglas-fir forests of the Colorado Front Range currently serve as a carbon sink, but are at high risk for crown fire. Residents that live within the wildland urban interface and their urban counterparts recognize the increased wildfire risk (Kaval 2009) and are willing to pay to reduce the risk (Kaval 2009; Walker and others 2007). These residents favor thinning over prescribed fire (Walker and others 2007) to reduce crown fire hazard which will produce large quantities of non-merchantable material. If the biomass removed is not used as a substitute for fossil fuels then the area will become a carbon source. Also, maintenance of forests with low crown fire risk will result in lower carbon storage on the landscape.

Fuel treatments can reduce CO_2 emissions during a wildfire (Hurteau and others 2008) by reducing tree mortality and maintaining an intact post-fire forested stand. In our simulations, maintaining stand basal area between 50 and 100 ft^2/ac reduced the risk of a severe wildfire by substantially increasing the windspeed estimated to initiate a passive crown fire (torching event) or sustain an active crown fire. Other studies found similar results. In simulations of East Cascade ponderosa pine forests, the removal of understory vegetation led to a reduction in potential fire severity that consequently lowered overall biomass combustion (Mitchell and others 2009). A thin from below and whole-tree harvests were implemented, which resulted in several stand characteristics that reduce undesirable fire behavior (Agee and Skinner 2005). Thinning from below focuses on reducing the ladder fuel component of the stand, which raises the canopy base height of the stand. Taller canopy base heights place the flammable foliage further from the surface fuels and require taller and more intense flames for crown fire ignition. However, thinning from below may not accomplish target basal area reductions and some overstory may also need to be removed (Agee and Skinner 2005). In stands without a ladder fuel component, thinning would be focused in the overstory to reduce canopy bulk density in order to reduce crown fire risk.

As expected, in our simulations, as the residual trees grew and new regeneration established, stands added more foliage biomass, canopy bulk density increased, and canopy base height decreased. These changes in stand characteristics increased the fire risk until the next entry. Two things are notable. First, although the crowning index decreased throughout the rotation, it never reached the pretreatment values due to the lack of vegetation in the midstory, and met the objective of reducing active crown fire risk. Second, the torching index was variable throughout each rotation due to the ingrowth of regeneration that evidently reduced the average stand crown base height. It should be noted, however, that FVS is not a spatially explicit model and the placement of the regeneration in the understory is continuous. In reality, the regeneration could be quite patchy, creating situations in the stand where torching is more probable in one area versus another. Nevertheless, the lowering in torching index suggests that managers will need to have additional, more frequent entries that address the regeneration issue to maintain low crown fire hazard (Battaglia and others 2008).

While fuel treatments reduce the risk of severe wildfire and increase stand resiliency to fire, the removal of biomass from the stand also reduces the amount of carbon sequestered and produces carbon emissions (Finkral and Evans 2008;

Mitchell and others 2009; North and others in press). From a carbon standpoint, the thinning of forests to reduce fire hazard is only effective if a wildfire were to burn over the treated area within the rotation period. Therefore, if a wildfire doesn't burn over the fuel treatment, then that area will become a carbon source if the biomass removed is not used to replace fossil fuels to generate energy. The magnitude of the CO_2 emitted will depend on the equipment used for harvesting, the amount of biomass removed, the utilization of the material removed, and the biomass replaced during the rotation.

The potential amounts of CO_2 equivalent per acre emitted by fuel reduction treatments in the ponderosa pine/Douglas-fir forests of the PSINF will vary depending upon site productivity and thinning intensity if the biomass is not utilized to offset fossil fuels (table 1). In our simulations, stands with a high basal area (160 ft^2/ac) did not replace the carbon removed in the first thinning, because they were maintained at a lower basal area. Stands with lower basal area (100 ft^2/ac) would regrow the biomass removed, because the target for the next entry was 100 ft^2/ac. In the 160 ft^2/ac basal area stand, even if all the removed biomass was used as a substitute for fossil fuels, this treatment would still emit about 18 MT CO_2/ac, because the stand did not fully replace the carbon removed in the first entry. However, if a wildfire burned either of these treated areas, the stands would likely remain intact and CO_2 emissions would be lower than if not treated.

Assessing the carbon value of fuel reductions weighed against potential losses in a stand-replacing fire is complicated, but requires consideration of the total area treated and the probability of fire in that area if not treated. For any individual stand that burns, the calculation is fairly simple, and depends on the probability of regeneration. If a wildfire does burn over an untreated area the amount of CO_2 emitted over time would be substantial (Hurteau and others 2008), with initial losses from combustion of the foliage, small twigs, forest floor, and some dead wood, and subsequent losses from decomposition of the killed trees. If the burned area was small enough to be near live trees with seeds, natural regeneration would establish a new forest, and the area would eventually recover most or all of the carbon lost. Contemporary wildfires in ponderosa pine forests often produce large patches far from seed sources that prevent natural regeneration (Bonnet and others 2005; Lentile and others 2005; Romme and others 2003). In this case, unless replanted, the forest will change to a meadow and lose substantial carbon. But, fires do not always conveniently burn only the treated areas, and thinned areas are a substantial carbon source unless the biomass removed is used instead of fossil fuel and the trees allowed to regrow the carbon lost. If the entire PSFNF ponderosa pine/Douglas-fir zone were treated for fuels reduction, we estimate that about 60 percent of the biomass carbon would be moved from the forest to the atmosphere *unless* it was used for biomass fuel and the forest allowed to regrow. Contrast this with a historic probability of wildfire of ~2 percent of forested area per year, with a lower amount of area in a large, high severity burn that would preclude timely natural regeneration. From a carbon perspective, large-scale fuel treatments are likely to be a carbon source, *unless* biomass is used for fuel.

In our reforestation simulations, more carbon accumulated within 90 years as site productivity and planting densities increased (table 2). At sites with low productivity (SI=35), planting densities of 300 TPA would yield similar CO_2 equivalent sequestration as planting densities of 100 TPA on a higher site productivity of 55. These differences in CO_2 equivalent accumulated based on planting densities and site productivity should help guide managers with strategic reforestation planning. However, high planting densities would recreate forests with high active crown fire and bark beetle risk and more need for thinning, while low planting densities would not. Instead, low planting densities would develop forests with lower crown

base heights more susceptible to passive crown fire. In our reforestation simulation, the *no thin* scenario sequestered more CO_2 than the *fuel treatment* scenario. After a 120 years of stand development, the active crown fire hazard was high (30 mph) and the stands were in potential danger of high mortality and substantial CO_2 release. The *fuel treatment* scenario follows the same story as mentioned in the fuel treatment maintenance scenario above; increase in CO_2 sequestered and lower fire risk over several rotations. The *no thin* scenario continues to accumulate CO_2 throughout the simulation, but density-dependent mortality begins around 180 years. Also around this time, basal areas are exceeding 180 ft^2/ac, making this stand highly susceptible to mountain pine beetle (*Dendroctonus ponderosae*) attack (Schmid and others 1994). Crown fire risk is moderate, with windspeeds less than 40 mph required to sustain an active crown fire due to the dense overstory canopy. The risk of passive crown fire is high in both scenarios, suggesting that some additional treatments such as pruning in the *no thin* and addressing the regeneration issue in the *fuel treatment* scenario is needed.

Conclusions

Many acres in the ponderosa pine/Douglas-fir forests of the Colorado Front Range are currently sequestering substantial amounts of carbon, but are also at high risk of sustaining a crown fire if ignited. Thinning these stands to reduce crown fire risk will result in CO_2 emissions if the biomass is not used as a substitute for fossil fuels. Furthermore, if stands are maintained at densities that are lower than the pre-treatment basal area, carbon will move from the forest to the atmosphere because the carbon from the total biomass removed will not be recovered. From a carbon standpoint, the argument that thinning a forest to reduce crown fire risk and CO_2 emissions is valid, but only if a fire actually burns the dense stand. Regardless of the carbon impact, thinning ponderosa pine/Douglas-fir forests to reduce crown fire risk has other ecological benefits.

A recent report by the Front Range Fuel Treatment Partnership concluded that heating with woody biomass in institutional buildings along the Colorado Front Range appears to be the best opportunity to lower fuel reduction treatment costs while utilizing woody biomass (FRFTP 2006). The FRFTP estimates that institutional buildings require only ~300 tons of woody biomass a year, which is feasible in terms of the level of forest treatment and transportation activity that is sustainable and desirable. Furthermore, the demand for institutions heated by biomass along the Colorado Front Range is expected to rise due to the building of new schools and existing schools that will replace aging boilers in the next few decades. Since most of the public institutions along the Colorado Front Range are dispersed within the same forests that need treatment, transportation costs to individual boilers would likely be cheaper than transporting it to a centralized biopower facility. However, until the infrastructure is developed, the utilization of biomass generated from fuel reduction treatments will remain a challenge.

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The content of this paper reflects the views of the authors, who are responsible for the facts and accuracy of the information presented herein.

To Manage or Not to Manage: The Role of Silviculture in Sequestering Carbon in the Specter of Climate Change

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Abstract—Forests and the soils beneath them are a major sink for atmospheric CO_2 and play a significant role in offsetting CO_2 emissions by converting CO_2 into wood through photosynthesis and storing it for an extended period. However, forest fires counter carbon sequestration because pyrolysis converts organic C to CO and CO_2, releasing decades or centuries of bound C to the atmosphere as a pulse, exacerbating the greenhouse gas effect. With global warming, the probability of fire has increased. Silviculture is an important tool for reducing wildfire risk and enhancing long-term carbon sequestration and—through this—mitigating the effect of climate change. Using the data collected from three studies over the last several decades, we compared treatment effects (density manipulation, fertilization, vegetation control, and interactions among some of them) on tree growth and subsequently carbon accumulation, fire risks predicted with fire behavior simulations, and responses of stand to future climate changes modeled by a process-based model (3-PG). With these case studies, we found that (1) intensive management (vegetation control and fertilization) increased C sequestration 400 percent and decreased fire caused tree mortality 50 percent compared to control at age 21 (Whitmore Garden of Eden study). (2) Density manipulation and vegetation control increased C sequestration 30 percent and decreased fire caused tree mortality 50 percent compared to control at age 40 (Challenge Initial Spacing study). (3) Density manipulation increased C sequestration 9 percent and decreased fire caused tree mortality 40 percent compared to control at age 55 (Elliot Ranch LOGS study). In addition, bark beetles killed significantly more trees in the control (high density plots) than in the lower density plots. (4) The 3-PG model predicts that global warming impacts carbon sequestration more in unmanaged than managed stands. These findings suggest that if carbon sequestration and storage are goals, our forests should be managed more aggressively in the future.

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Introduction

Global climate is changing at an unprecedented rate. The latest assessment from the Intergovernmental Panel on Climate Change (IPCC 2007) states that the global average surface temperature has increased 0.74 °C from 1906 to 2005. By 2100, increases of 1.1-6.4 °C are projected over the 1990 level using different models with various scenarios. Warming trends are believed to be due to the anthropogenic increase of greenhouse gases (GHG), with an increase of 70 percent between 1970 and 2004. Carbon dioxide (CO_2) is one of the most important anthropogenic GHGs. Annual emissions grew by about 80 percent between 1970 and 2004. Not only have CO_2 and GHG concentration increased greatly since 1750, but the rate of increase far exceeds pre-industrial values determined from ice cores spanning many thousands of years (IPCC 2007).

Forests play a significant role in offsetting CO_2 emissions by converting CO_2 into organic C through photosynthesis. Much of the product of photosynthesis

is stored in the forest for decades or centuries. Despite uncertainties, annual carbon sequestration is estimated to vary between 149 and 330 million tonnes C by forests in the United States (Woodbury and others 2007), which offsets about 10 percent of US CO_2 emissions. Wildland fires have annually affected about 1.7 million ha of forests across the United States in recent decades with the area increasing in the last 10 years, thereby releasing vast pulses of ecosystem carbon back to the atmosphere.

Fire was historically an integral ecosystem process in the forests, especially forests in the Interior West. Prior to Euro-American settlement, forests of the region were more open, containing fewer trees and wider crown spacing than today (Agee 1993; Cooper 1960; Covington and Moore 1994; Skinner and Taylor 2006). Fires historically burned every 4-25 years (Graham and others 2004), thinning forest stands of small trees but also creating bare mineral soil environments favored by seedlings of some tree species, such as light-demanding pines. Starting in the late 19th and early 20th century, logging, livestock overgrazing, and fire suppression created conditions more suitable for tree regeneration and survival. Today, fire suppression and a lack of density management in both young and old stands have resulted in forests dominated by dense thickets of saplings and pole-sized trees, often with a higher proportion of shade-tolerant species. Furthermore, long intervals between fire events have led to heavy accumulations of litter, duff, and woody fuels in many areas. Therefore, today's forests are more susceptible to stand-replacing crown fires (Agee and Skinner 2005).

Managing these forests has become a great challenge to forest managers. In recent decades, Federal land management agencies and private land owners have treated millions of acres of hazardous fuels using mechanical thinning, prescribed fires, and other means. These treatments are absolutely necessary because forest structure and function would not be restored without them (Agee 2007). However, there is limited information on how these treatments affect carbon sequestration and storage and how treated stands are likely to respond to future climate. On one hand, fuel treatment may remove carbon by harvesting small-size trees and shrubs and by disturbing soils that may stimulate soil respiration. On the other, treatment may increase the vigor of remaining trees that will sequester more carbon. At the stand level, we would reallocate more carbon to residual living trees.

Manipulating stand density through thinning is not new. Since the birth of silviculture, thinning has been a major means for controlling stand density, structure, and composition (Smith and others 1996). Standing fuel reduction is merely a modern extension of density manipulation. The results from growth and yield studies established across the US in the past century can be interpreted to answer some of today's questions such as effect of density and competing vegetation control on carbon sequestration and storage, as well as fuel accumulations.

In this paper, we present three case studies that were conducted over several decades by the PSW Redding Silviculture Laboratory demonstrating how forest vegetation management has affected the fate of carbon and how silviculture may help mitigate climate change effects on our forests (table 1). These long-term permanent installations are the (1) Whitmore "Garden of Eden" study of the effects of understory vegetation control and fertilization on stand dynamics, established by the second author; (2) Challenge Initial Spacing study of the effects of stand density and understory vegetation control on stand growth; and (3) Elliot Ranch Levels-of-Growing Stock study of the effect of density on stand growth. Both established by retired Research Silviculturist William W. Oliver. These plantations represent 21, 40, and 55 years of development respectively and include a range of silvicultural treatments.

Table 1—Study site, silvicultural treatments, and climate change effect (percentage reduction) relative to normal climate on aboveground live tree carbon stock modeled with 3-PG for three plantations by assuming three scenarios (25% precipitation [ppt] reduction, 2 °C increase, and both) since the trees were planted at these sites.

Study site	Spacing/ density	Treatment	–25% ppt	+2 °C	–25% ppt +2 °C
Whitmore	2.4 x 2.4 m	Control (C)	–1.6	–14.5	–15.2
		Fertilizer (F)	–2.6	–9.6	–10.9
		Herbicide (H)	–3.3	–7.2	–9.7
		HF	–2.4	–4.5	–6.9
Challenge	1.8 x 1.8 m	Shrub present	–4.7	–3.4	–6.0
		Shrub absent	–2.3	–1.3	–3.7
	3.7 x 3.7 m	Shrub present	–1.5	–1.9	–3.7
		Shrub absent	–0.4	–1.7	–2.2
	5.5 x 5.5 m	Shrub present	–0.1	–1.8	–1.9
		Shrub absent	0.0	–1.8	–1.8
Elliot Ranch	38 m^2 ha^{-1}	Shrub present	–21.0	–16.0	–18.9
	16 m^2 ha^{-1}	Shrub present	–8.4	–0.3	–6.2

General Procedures

Aboveground Carbon Comparison

Using historical inventory data for each tree at each plot, we calculated and compared aboveground biomass among silvicultural treatments and controls. A diameter-based biomass regression (fig. 1) was established with 36 ponderosa pine (*Pinus ponderosa* C. Lawson) trees harvested at the Whitmore Garden of Eden (5-24 cm DBH) and 40 smaller diameter trees harvested from the Long-Term Soil Productivity Study installations (4-8 cm) in northern California (R.F. Powers, unpublished data). The biomass estimate for trees with DBH smaller than 4 cm must be used with caution. The allometric equation for trees with DBH of 25 cm and above was developed from 110 mature ponderosa pine trees harvested in natural stands prior to the Long-Term Soil Productivity Study installations (15 – 152 cm) at 11 sites in Northern California (fig. 2; R.F. Powers, unpublished data). After plotting predicted values from two equations, we found that both were remarkably similar for trees with DBH smaller than 25 cm (inset of fig. 2). For trees that are larger than 25 cm DBH, the equation extrapolated from small diameter trees (fig. 1) clearly underestimates the aboveground biomass. We concentrated on aboveground biomass because (1) it is more sensitive to disturbance such as wildfires and insect infestation and (2) it is relatively easier to measure the aboveground biomass than the below-ground biomass, which is not available for the larger-diameter trees. The biomass data were converted to carbon stock assuming a carbon content of approximately 50 percent by dry weight.

Fire Effects

Fire Family Plus (FFP) (http://www.fs.fed.us/fire/planning/nist/distribu.htm) software was employed to derive weather variables to use in fire simulations. We used data for Station 040615-Whitmore (http://nwcg.gov/fam-web/) to represent

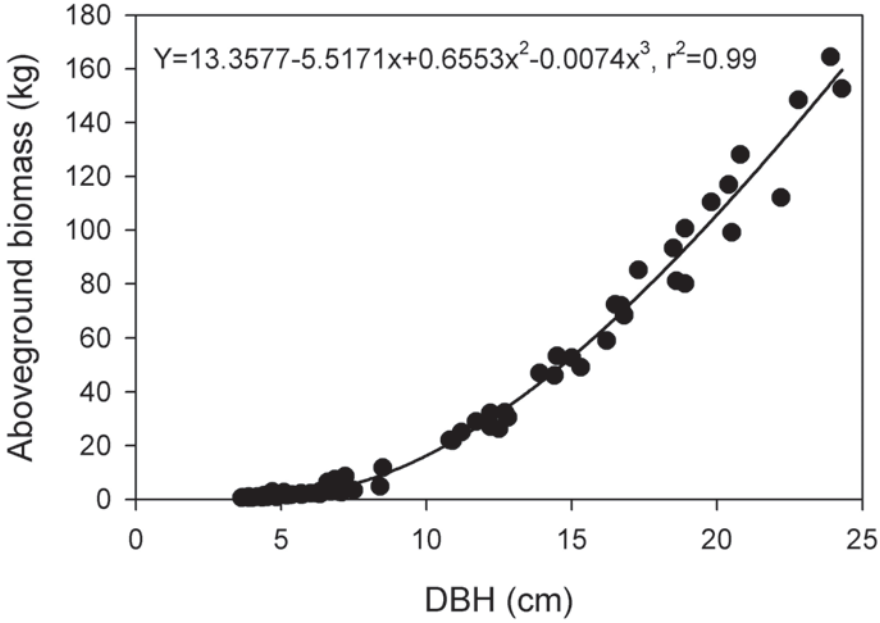

Figure 1—Ponderosa pine diameter-based biomass regression for 21-yr-old trees at Whitmore and some small diameter trees harvested at the Long-Term Soil Productivity installation in northern California. The range of DBH is between 3.6 cm and 24.3 cm.

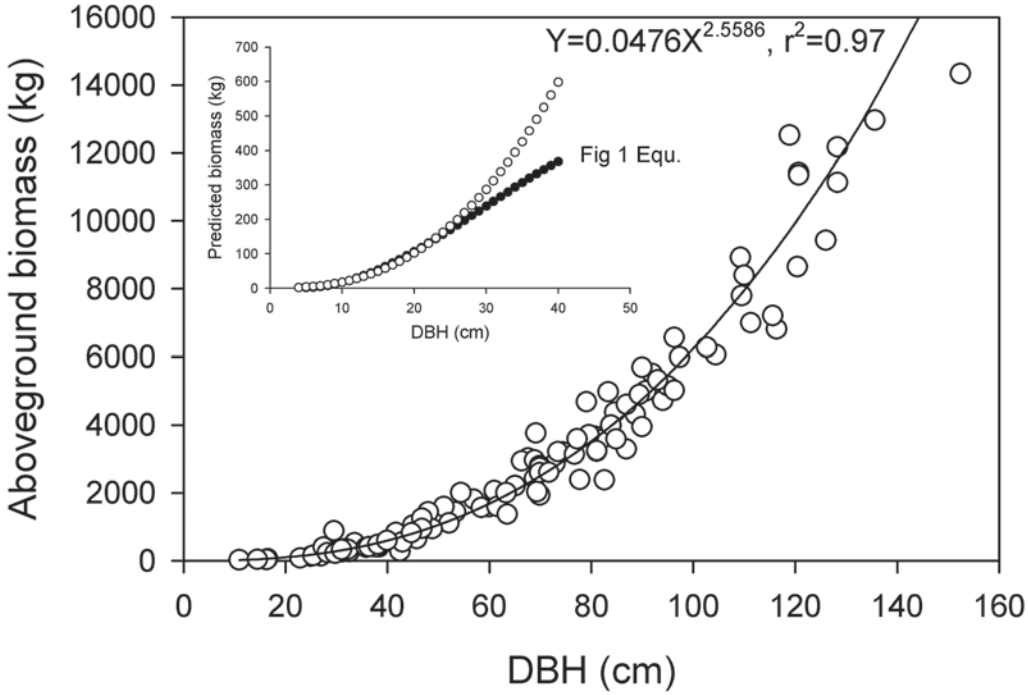

Figure 2—Ponderosa pine diameter-based biomass regression for mature trees (15-152 cm) harvested prior to the Long-Term Soil Productivity installations in northern California. The predictions from this power equation are remarkably similar with predictions from the polynomial equation established for trees with DBH less than 25 cm (inset figure).

conditions for all of our study sites in order to facilitate cross-site comparisons. Environmental conditions for fire behavior simulations were derived from climatological reports in FFP. Most variables are for 97.5 percent burning conditions (conditions exceeded only 2.5 percent of the time each year). Windspeed was set at the 97 percent condition of 8 km/hr (conservative for windspeeds in high-intensity wildfires in our study areas), but temperature and moisture conditions are mean high/low August values (35 °C, 10 percent relative humidity). Moisture contents of ground fuels were set at 2 percent, 4 percent, and 4 percent for 1-, 10-, and 100-hour fuels, respectively. Live fuel moistures were set at 30 percent, 56 percent, and 80 percent for herbaceous, woody, and foliage materials, respectively. For each simulation we used a standard fuel model from one of three sources: Rothermel (1983), Fire Program Solutions (FPS) (2005), or Scott and Burgan (2005). We did not attempt to create a custom fuel model since we did not have an opportunity to calibrate fire behavior output with an actual fire.

All fire behavior simulations were performed using the CrownMass routine of the Fuels Management Analyst Suite 3.01 (FPS 2005). This program allows the entry of a tree list from the site to estimate the canopy fuel conditions. This is then combined with a standard surface fuel model in the scenario to simulate surface fire spread and intensity as well as the potential for torching and crowning as described by Scott and Reinhardt (2001).

Calibrating the 3-PG Model

To examine climate change effects on aboveground biomass, we needed a process-based model to predict stand dynamics by varying temperature and precipitation. Because the Whitmore "Garden of Eden" study includes vegetation control and fertilization treatments and was intensively measured with the detailed records for 20 years, we used these records to calibrate the 3-PG (Physiological Principles Predicting Growth) model (Landsberg and Waring 1997) for ponderosa pine grown in northern California. 3-PG is a process-based model to predict forest performance with climatic variables as drivers. It holds several advantages for managers. First, 3-PG is a relatively simple stand-level model. Like other process-based models, it includes subroutines to calculate photosynthesis, transpiration, respiration, growth allocation, and litter production. Notably, 3-PG differs from most process-based models in that it requires only readily available site and climatic data as inputs, and predicts the time-course of stand development in a form familiar to the forest manager. Second, it is very user-friendly; a Microsoft Excel workbook includes everything—input and code that you can change, and output results in a normal spreadsheet. Finally, it has been extensively tested in the world's major forest species, including *Eucalyptus* spp., *Picea sitchensis* (Bong.) Carrière, *Pinus ponderosa* C. Lawson, *P. radiata* D. Don, *P. taeda* L., *Pseudotsuga menziesii* (Mirb.) Franco, etc. (Landsberg and others 2001; Law and others 2000; Sands and Landsberg 2002; Waring 2000). After calibrating key parameters with one treatment at Whitmore, we applied the model to other treatments at this site, the Challenge Initial Spacing study, and the Elliot Ranch LOGS study. We used the aboveground biomass output to compare the aboveground biomass estimation from the allometric equations for the silvicultural treatments at each site. Similarly, as calculated biomass from observed DBH, biomass was converted to carbon stock.

During model calibration, we used all parameters given by Law and others (2000) as our prototype blueprint for ponderosa plantations in northern California. Based on our biomass data at Whitmore, we found that the mathematical relationship between stem mass and diameter was:

$$W_s = 0.0456*(DBH)^{2.5687},\ r^2 = 0.99$$

Where: W_s is the stem and branch biomass in kg and DBH is diameter at breast height in cm. Therefore, both the constant and power differ from the parameters used by Law and others (2000) in Oregon.

In addition, ratios of foliage to stem partitioning differed from those for mature natural stands in Oregon. Our ratios were about 0.85 at DBH = 2 cm and 0.14 at DBH = 20 cm.

Maximum quantum use efficiency also differed from mature Oregon stands. By measuring sun and shade leaves from 36 trees grown from the different treatments at three Garden of Eden sites, quantum use efficiency was found to be 0.05 (Liang Wei, unpublished data, University of Idaho). Remarkably, there was no difference between sun and shade leaves, among treatments, or among sites.

We also added in maximum stand density of 365 as a constraint for tree mortality because stand density is strongly influenced by *Dendroctonus* bark beetles in northern California (Oliver 1995).

Climatic Data and Climate Change Scenarios

Climatic data for the model were obtained from weather stations nearest to each site. Temperature and precipitation are commonly available and are must-have variables. Incident solar radiation was calculated following Coops and others (2000). The 3-PG simulator can run with monthly data for specific years (Whitmore and Challenge) or with mean monthly data for many years (Elliot Ranch) if yearly data are not available. Whitmore Garden of Eden data were from a Remote Automated Weather Station (RAWS) at Whitmore, about a km from the study site (http://www.wrcc.dri.edu/cgi-bin/rawMAIN.pl?caCWHT). The data before 1990 when the station was installed were obtained from the average of 1990-2007. Two weather stations were used to make a complete climatic data for the Challenge Initial Spacing study since 1966: Strawberry Valley (http://www.wrcc.dri.edu/cgi-bin/cliMAIN.pl?ca8606) and Challenge Ranger Station (http://www.wrcc.dri.edu/cgi-bin/cliMAIN.pl?ca1653). Due to a lack of data for the last 60 years, we used station mean data for Elliot Ranch site; two stations were used: Foresthill (http://www.wrcc.dri.edu/cgi-bin/rawMAIN.pl?caCFOR) and Iowa Hill, CA (http://www.wrcc.dri.edu/cgi-bin/cliMAIN.pl?ca4288).

After the model was calibrated, we made projection runs for each treatment with the past climate. The modeled values were compared with calculated values from the allometric equations based on periodic DBH measurements. Then, we ran the model again by increasing temperature 2 ºC, reducing precipitation by 25 percent, or both. The final changes relative to the first run were presented as the climate change effect for each treatment itself. Lastly, impact of silvicultural treatments was compared.

Case Study I: Whitmore "Garden of Eden"

Site Characteristics and Experimental Design

The Garden of Eden study was established to determine the biological response of ponderosa pine to a broad array of silvicultural treatments over a range of sites typifying plantation management in northern California. These plantations were established on lands cleared of brushfields or natural forest from 1986 to 1988. Each plot measuring 19.7 by 21.9 m was hand-planted at a standard square spacing of 2.4 m with seedlings from superior genetic stock known to perform well at each seed zone and elevation. Following planting, 8 treatments were applied to 24 plots in a completely randomized design. The standard suite of treatments

consisted of a control (C: planting and no further treatment); fertilization only (F: eight nutrients applied at planting, and at an exponential rate over the next 6 years); vegetation control only (H: glyphosate herbicide applied annually to control all understory vegetation for the first 5 years); and fertilization and vegetation control combined (HF). An additional silvicultural treatment, systemic insecticide (I), was applied as well, producing 3 replications each of 24 plots. Each treatment plot of 0.04 ha consisted of 72 trees, but only the innermost 20 were used for measurement. Treatment details can be found in Powers and Ferrell (1996) and Powers and Reynolds (1999).

The Whitmore Garden of Eden plantation, the oldest in the series, was planted in spring 1986 on land managed by W.M. Beaty and Associates in eastern Shasta County on the southwestern slope of the Cascade Range (Lat. 40.62 N; Long. 121.90 W). Elevation of the Whitmore plantation is 730 m and precipitation averages 1,140 mm annually. The soil is the Aiken soil series (clayey, mesic Xeric Haplohumults), a widespread soil developed from a Pleistocene volcanic mudflow and typical of the west slope pine forest. Prior to clearing, the site supported a brushfield dominated by whiteleaf and common manzanitas (*Arctostaphylos viscida* Parry and *A. manzanita* Parry, respectively) that originated from a 1967 wildfire. Site index (Powers and Oliver 1978), based on measurements of older trees in surrounding stands, averages 23 m (78 ft) at 50 years.

After installation, all measurement plots were inventoried at ages 2, 5, 6, 10, 15, 21 years. Each tree was measured for DBH, height to the base of the live crown, and total tree height. In the earlier years when trees were less than 1.4 m tall, diameter at 10 cm above ground was measured. All understory vegetation was measured for height and percent cover by four line intercept transects per plot, each 10 m long. Sample trees spanning the range of tree sizes were felled at ages 15 and 21 for biomass analyses (3 trees per treatment plot). Each tree stem was measured and sectioned at several stem positions and rounds taken for determining oven-dry weights after drying to a constant weight at 70 °C. Bole mass was then estimated by applying mass/volume ratios of each round to the bole volume in each sequential sector. Crowns were divided into 5 equal-length sections and the branch of average basal area was taken from each section for dry weight analysis after separation into wood and needles. Crown mass per sector was estimated as:

$$\text{Crown sector mass} = \sum(\text{mass/basal area of sample branch x basal area of all branches})$$

Crown sector masses for each sample tree were summed for a single estimate of individual crown mass. This, added to the mass estimate for the bole, produced a mass estimate for each individual sample tree. Due to lack of an allometric equation to calculate biomass for trees that are under 1.4-m tall, we used data collected at ages 5 and older to tune the 3-PG model.

Aboveground Tree Carbon

At age 21, aboveground tree carbon stock was 73.3, 55.1, 44.2, and 20.0 Mg ha^{-1} on HF, H, F, and C, respectively (fig. 3), and treatment effects were highly significant ($P<0.001$). Relative to control trees (that is, do nothing after trees were planted), trees in the fertilizer only treatment (F) accumulated twice the mass of carbon, while the herbicide only treatment (H) nearly trebled the amount and HF treatment almost quadrupled the amount, compared with control trees. Therefore, managed stands stored much more carbon in trees than unmanaged stands.

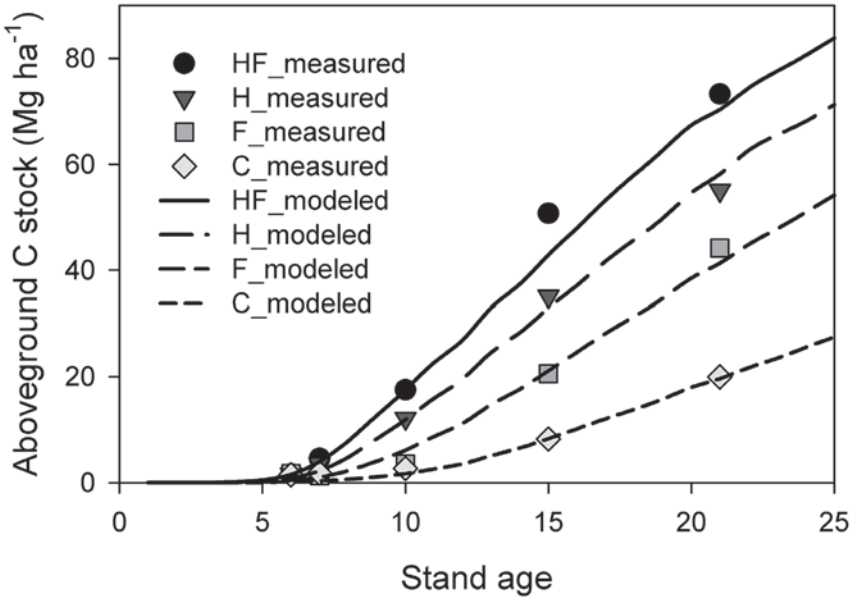

Figure 3—Age-related aboveground tree carbon stock accumulated on HF, H, F, and control plots at the Whitmore Garden of Eden study, Whitmore, California.

Fire Effect

By age 21, shrub cover was 57 percent and average height was 1.3 m (table 2) in the control plots so that fire behavior was modeled as a shrubfield fuelbed. Flames exceeded tree height leading to a crown fire. Mortality was projected as close to 100 percent if fire occurred at this age. Similar results would have occurred in the fertilization treatment with 99 percent mortality because fertilization stimulated the growth of understory shrubs before pine canopies had closed (Powers and Ferrell 1996) and left a continuum of dead and dying fuels that reached to the live crown once canopies had closed. In contrast, because herbicides eliminated understory in the H and HF plots, a surface litter model was used for fire simulation, which projected that flame length was less than 1 m and fire was confined to the surface. No crowns were ignited in the simulations, but scorch was sufficient to cause 40-50 percent mortality in lower crown classes at age 21. These results clearly suggest that managed stands are more resilient to wildfire even early in stand development.

Modeled Effects

Projections made with 3-PG predicts aboveground biomass well; an intercept and a slope of regression between modeled and measured values did not differ from zero and one, respectively (fig. 4). After applying three climate change scenarios in the 3-PG model, we found that a reduction of 25 percent precipitation for each of 21 years would have reduced aboveground tree carbon only 1.6 to 3.3 percent (table 1). Yet, a temperature increase of 2 °C would have reduced carbon accumulation by 7.2-14.5 percent. This result seems surprising because water is always considered a limiting factor during late growing season under the Mediterranean climate as at these sites in California. A possible, but unlikely, explanation is that these trees would not consume all precipitation so that the

Table 2—Results of fire behavior simulations for ponderosa pine plantations at Whitmore Garden of Eden treatment plots at age 21, Challenge Initial Spacing plots at age 30 when shrub data were available, and Elliot Ranch LOGS study at age 55. All fire behavior fuel models are standard fuel models from either (1) Rothermel (1983), (2) Scott and Burgan (2001), or (3) FPS (2005). Fire types are: PC = passive crown fire; SURF = surface fire.

Study site	Spacing/ Density	Treatment	Fuel model (source)	Stand HT (m)	Shrub HT (m)	Shrub cover (%)	Fire type	Flame length (m)	Mortality (%)
Whitmore	2.4 x 2.4 m	Control (C)	SH4 (2)	6.7	1.3	57.4	PC	7.3	100
		Fertilizer (F)	TU3 (2)	9.4	1.4	28.8	PC	6.1	99
		Herbicide (H)	TL8 (2)	9.8	0.0	0.0	SURF	0.6	48
		HF	FB9 (1)	11.6	0.0	0.0	SURF	0.5	42
Challenge	1.8 x 1.8 m	Shrub present	5A (3)	15.3	3.1	54.9	SURF	1.5	91.3
		Shrub absent	FB9 (1)	16.5	0.0	0.0	SURF	0.6	42.1
	3.7 x 3.7 m	Shrub present	5M (3)	16.8	2.3	64.0	SURF	1.4	77.6
		Shrub absent	FB9 (1)	19.3	0.0	0.0	SURF	0.6	24.8
	5.5 x 5.5 m	Shrub present	5Z (3)	20.2	2.3	86.2	SURF	1.8	84.8
		Shrub absent	FB9 (1)	22.9	0.0	0.0	SURF	0.4	12.6
Elliot Ranch	38 m^2 ha^{-1}	Shrub present	6M (3)	20.2	4.3	38.8	SURF	1.8	61.5
	16 m^2 ha^{-1}	Shrub present	6M (3)	18.0	2.4	22.1	SURF	1.4	36.6

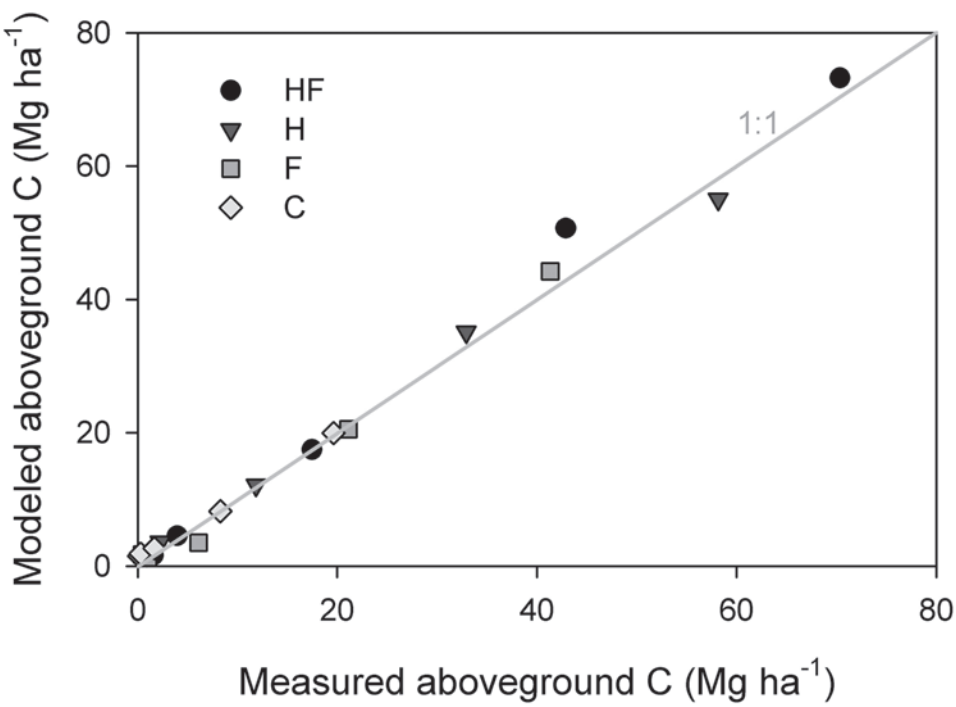

Figure 4—The relationship in aboveground tree carbon between modeled with 3-PG and calculated from measured DBH on HF, H, F, and control plots at the Whitmore Garden of Eden study, Whitmore, California.

25 percent reduction represents water lost in the system but with little effect on tree growth. Otherwise, the water relation submodel within 3-PG may need to be further improved concerning water use and transpiration. Because 3-PG is a stand model and does not account for leaf area in understory vegetation, it probably underestimates water use by trees in a droughty climate. By assuming both 25 percent reduction of precipitation and a temperature increase of 2 °C since 1986, we found that carbon storage would have reduced 6.9 to 15.2 percent. Carbon accumulation aboveground was reduced more in the control plot trees than in any managed plots, suggesting that untended plantations are particularly sensitive to climate change.

Case Study II: Challenge Initial Spacing Study

Site Characteristics and Experimental Design

This study was established with planted seedlings near the lower edge of the mixed-conifer forest, on the west slope of the northern Sierra Nevada (Lat. 39.48 N; Long. 121.22 W). Elevation is 810 m and precipitation averages 1,730 mm annually. Soil, an Aiken clayey, mesic Xeric Haplohumult, is more than 1.5 m deep. Dominant shrub species are whiteleaf manzanita, deerbrush (*Ceanothus integerrimus* Hook. & Arn.), squaw carpet (*C. prostratus* Benth.), Indian manzanita (*A. mewukka* Merriam), small numbers of Sierra gooseberry (*Ribes roezlii* Regel) and sprouts of California black oak (*Quercus kelloggii* Newberry) and tanoak (*Lithocarpus densiflorus* [Hook. & Aan.] Rehd.).

The original stand was 70-year-old ponderosa pine with a site index of 30 m at 50 years, which was clearcut for this experiment. Logging slash was raked, piled, and burned off the unit. In March 1966, ponderosa pine seedlings were planted in two randomized blocks. Each block contains five plots that were planted at square spacings of 1.8, 2.7, 3.7, 4.6, and 5.5 m. Each plot was split into two adjacent subplots. On one subplot, brush seedlings were grubbed out by hand for the first year and then herbicide 2,4,5-T (2,4,5-trichlorophenoxy acetic acid) was applied by hand sprayer in the second and fourth years after planting. Subsequent shrub seedlings were removed by hand for about five more years. On the other subplot, shrubs were allowed to develop naturally. Each subplot contained 12 measurement trees that were buffered from adjacent plots by at least 7 m, minimally two rows of trees. Because the same number of trees was used among plots, subplot size with buffer varied among treatments, covering 0.05 ha for 1.8 m spacing plots to 0.15 ha for 5.5 m spacing plots.

Height and DBH (if tree height reached 1.37 m) were measured every year from 1968 to 1975, every two years from 1975 to 1985, and every four years from 1985 to 2006. Other measurements include height to live crown, crown width, and tree condition.

Aboveground Tree Carbon

Based on our biomass equation, aboveground tree carbon stocks for shrub present and shrub absent treatments at 40 years were 78.0 and 70.3 Mg ha^{-1} on 1.8 m spacing, 80.8 and 99.2 Mg ha^{-1} on 3.7 m spacing, and 72.6 and 92.8 Mg ha^{-1} on 5.5 m spacing, respectively (fig. 5). The results suggest that control of competing vegetation enhances tree growth and carbon storage compared to the plots with shrubs present at the wider spacings. At the narrowest spacing of 1.8 x 1.8 m, the plots with shrubs absent developed so quickly that mortality occurred much earlier than the treatments leaving shrubs present. As a result, carbon stocks are higher on the shrub present plots.

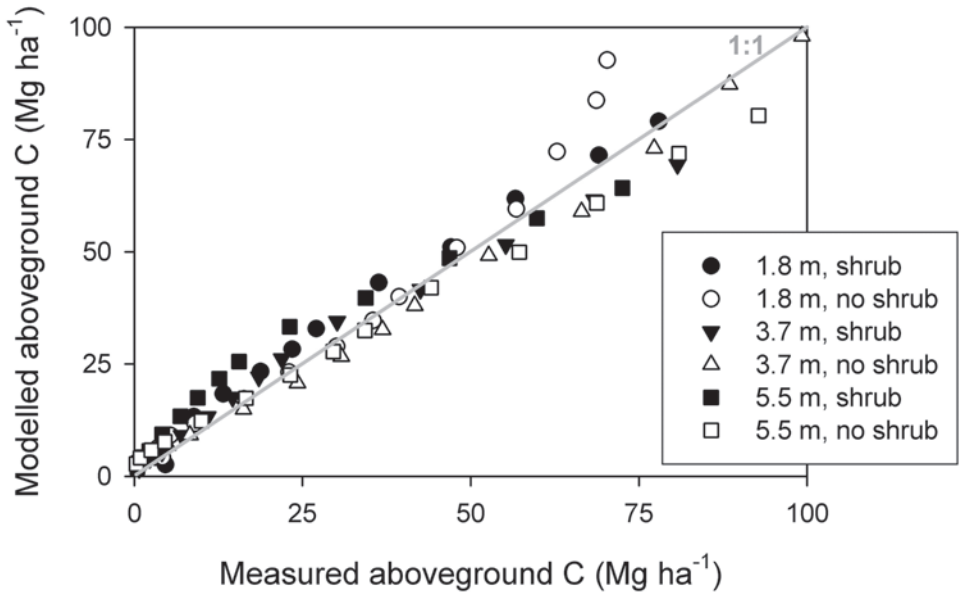

Figure 5—The relationship in aboveground tree carbon between modeled with 3-PG and calculated from measured DBH on three spacings split with either shrub present or shrub absent plots at the Challenge Initial Spacing study, Challenge Experiment Forest, California.

Fire Effect

By age 30, in plots where shrubs were not controlled, shrub cover ranged from 54.9 percent to 86.2 percent and average height varied from 2.3 to 3.1 m (table 2). These plots were modeled as a shrub-dominated fuelbed. Fire intensity in these plots was estimated to cause between 77.6 percent and 91.3 percent mortality depending on density of plantings (table 2). Where shrubs were controlled, a surface litter model was used for fire simulation, which projected fire mortality to range only from 12.6 percent to 42.1 percent—again depending on tree density. No crowns were ignited in the simulations, and mortality was caused primarily by scorch and cambium damage. Again, these results suggest that more intensively managed stands are more likely to be resilient to wildfire.

Modeled Effects

Overall, aboveground tree carbon stock based on DBH measurements was strongly correlated to 3-PG modeled aboveground tree carbon (fig. 5). Because our allometric equation was developed from trees with DBHs 3.6 cm and greater, a weaker relationship was expected at young ages. At later years, heavy mortality at 1.8 m spacing with shrub absence yielded a C stock overestimated by 3-PG.

After applying the three climate change scenarios in the 3-PG model, we found that a reduction of 25 percent precipitation for each of 40 years would have reduced aboveground tree carbon only 0 to 4.7 percent (table 1). A climate warming of 2 ºC would reduce carbon 1.3-3.4 percent and the combination of both changes would reduce carbon by 1.9 to 6.0 percent, varying with densities and whether shrubs were controlled. In general, high density stands are more sensitive to temperature increase and precipitation reduction because of greater competition for soil resources. Similarly, shrub present-plots are more sensitive to climate change.

Case Study III: Elliot Ranch Levels-of-Growing Stock Study

Site Characteristics and Experimental Design

These plots were established in a plantation that originated after the Elliot Ranch Fire, which burned a deerbrush shrub and snag field that had developed following the 1949 Elliot Ranch Burn. The area was planted at 1.8 by 2.4 m spacing in 1950 with 1-1 ponderosa pine stock from the appropriate seed zone. The plantation is located on the Foresthill Ranger District, Tahoe National Forest (Lat. 39.16 N; Long. 120.74 W) on the western slope of the Sierra Nevada. Elevation is about 1,200 m and precipitation averages 1,524 mm annually. Three clay-loam soils, Cohasset and Horseshoe Series (loamy, mesic Ultic Haploxeralfs) and an unclassified alluvium, underlie the study area (Oliver 1979). The average site index is estimated to be 35 m at 50 years. Cohasset Series is slightly more productive than Horseshoe, and trees growing in Cohasset soil in the study area are estimated to be 36.5 m tall at 50 years. Trees on the alluvial soil express a site index similar to Horseshoe Series.

The study plots were established in 1969 when the plantation was 20 years old. This is one of six installations in the west-wide levels-of-growing-stock study for even-aged ponderosa pine guided by Myers' (1967) study plan. Portions of the Elliot Ranch plantation were used in developing yield tables for managed stands of ponderosa pine (Oliver and Powers 1978).

All plots are buffered with a 9-m isolation strip. The study design is fully randomized with three replications. All plots are 0.2 ha in size, exclusive of buffer. Five thinning treatments of 9, 16, 23, 30, and 37 m^2 basal area per ha were applied in 1969. Rethinnings in 1974 and 1979 restored the original basal area stand densities. The third rethinning in 1989 used Stand Density Index as the measure of stand density and resulted in an increase of approximately 10 percent in growing stock for each density treatment.

All trees within the plots were measured for DBH. Stem deformities and evidence of insect and disease attack were also noted. Total height and height to live crown were measured on a 20 percent systematic sample of the trees during all but the last measurement in 2004. At that time, all trees were measured for total height. A probability-proportional-to-size sample of six trees per plot was measured with an optical dendrometer for stem volumes during several remeasurements.

Aboveground Tree Carbon

To demonstrate effect of treatments, we only calculated tree carbon for two stand densities: 16 m^2 ha^{-1} (70 ft^2 ac^{-1}) and 38 m^2 ha^{-1} (160 ft^2 ac^{-1}); the latter similar to that of the natural untreated stand. Treatment plots were installed when the plantation was 20 years old and stand densities were achieved with repeated thinning. After each thinning, only bole wood was removed from the site. Thinning slash from the original thinning was piled and burned in the isolation strips between the measurement plots, while that from subsequent thinnings was lopped and scattered. We calculated live-tree carbon using a specific gravity of 0.38 Mg m^{-3} in converting wood volume to biomass. Collectively, there was 201 Mg ha^{-1} carbon stock (current live trees, 156; initial thin, 16; repeated thins, 30) on 16 m^2 ha^{-1} plots and 185 Mg ha^{-1} (current live trees, 166; initial thin, 2.4; repeated thins, 17) on 38 m^2 ha^{-1} plots. Not only did moderately thinned plots produce large-sized healthy trees within 55 years, but also they stored about 16 Mg ha^{-1} more carbon than lightly thinned plots. The National Forests across the Sierra Nevada consider 76 cm (30 in.) trees as must-keep "old growth" trees

during fuel reduction projects (fig. 6). With an appropriate silvicultural treatment, some trees can reach that category of apparent "old growth" in only 55 years. In addition, higher density plots suffered greater mortality, mainly caused by bark beetles, than lower density plots (fig. 6). These dead materials have become hazardous fuels for the forests.

Fire Effect

Shrubs were present in both stand conditions at this site and cover varied from 22.1 percent to 38.8 percent. Where shrubs were present, the fire simulation estimated flame lengths exceeding 1 m. Fire mortality was estimated to be 36.6 percent in the lower density plots and 61.5 percent in the higher density plots (table 2). This difference is due primarily to greater shrub height and cover, and smaller trees in the later plots leading to greater area experiencing the higher intensity fire. As at Challenge, the fire simulation indicated mortality was due mainly to crown scorch and cambium damage.

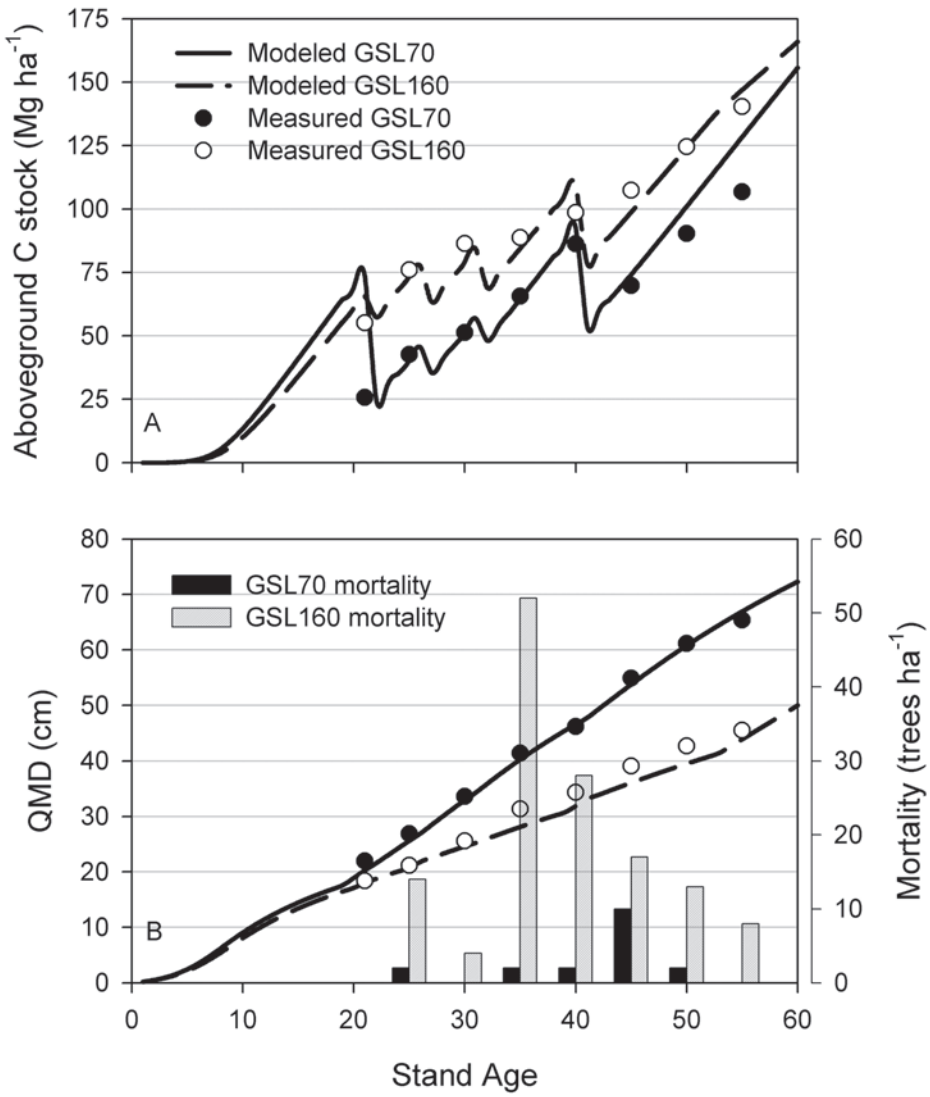

Figure 6—Measured and modeled aboveground tree carbon stock (A) and quadratic mean diameter (QMD) and tree mortality (B) on two growing stock levels at 16 m^2 ha^{-1} (70 ft^2 ac^{-1}) and 38 m^2 ha^{-1} (160 ft^2 ac^{-1}) plots across the last 55 years at the Elliot Ranch Level-of-Growing-Stock study, near Foresthill, California.

Modeled Effects

The 3-PG projection also tracked the thinning events well (fig. 6). We performed 3-PG runs based on the three scenarios of climate change that included the initial and repeated thinning treatments. Interestingly, the 25 percent precipitation reduction yielded the most reduction of carbon stocks with 8.4 percent for 16 m^2 ha^{-1} plots and 21 percent for 38 m^2 ha^{-1} plots, respectively (table 1). After 55 years, 2 ºC increases would have reduced C by 0.3-16 percent. Together, lower precipitation and higher temperature would have reduced C storage by 6.2 percent and 18.9 percent for the two densities. However, the higher density plots are more sensitive to climate change.

Concluding Remarks

Results from these case studies indicated that silviculture can play a significant role on managing forests for carbon and for mitigating the deleterious effects of climate change. Although what we presented here are results from plantations, the concept should hold for natural stands, especially when we consider wildfire as a part of ecosystem processes.

Managed stands accumulated more aboveground tree carbon than unmanaged stands. The result should not surprise forest managers because the goal of silviculture in earlier years was to maximize the highest wood production on a given land unit (Fernow 1914) and carbon stock is directly related to stem volume. We concentrated only on the aboveground tree carbon in this chapter because this represents stabilized carbon that varies considerably with stand ages. Understory vegetation and forest floor components are also important, but relatively unstable carbon because they are more susceptible to wildfire.

Managed stands are more resilient to wildfires or bark beetle infestation than unmanaged stands. This result supports the experience of seasoned forest managers, and echoes a major conclusion reached by Agee (2007) in his keynote at an earlier National Silviculture Workshop. In the past few decades, federal land management agencies and private land owners have treated millions of hectares of hazardous fuels using mechanical thinning, prescribed fires, and other means in order to create forests resilient to intensified wildfires or insect infestation. The challenge facing us is how we can use silviculture in meeting the multiple-use objectives for our forests.

Unmanaged stands are more sensitive to global climate change than managed stands in terms of carbon accumulations. Our three study sites are located near the lower edge of the mixed-conifer forest, on the western side of the Sierra Nevada and are dominated by a Mediterranean climate with wet and mild winters and hot and dry summers. Weather patterns suggest that growing season is controlled by water availability. Any management tools that improve availability of water and other resources during the growing season will benefit individual trees as well as stands.

Western ponderosa pine ecosystems have changed dramatically in structure and composition over the past century. More open forests with fewer trees and wider crown spacing have often been replaced by forests dominated by dense thickets of saplings and pole-sized trees due to various reasons. More than ever, these forests need to be managed in order to preserve their ecosystem services for this and future generations.

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The content of this paper reflects the views of the authors, who are responsible for the facts and accuracy of the information presented herein.

Estimating Long-Term Carbon Sequestration Patterns in Even- and Uneven-Aged Southern Pine Stands

Don C. Bragg[1] and James M. Guldin[2]

Abstract—Carbon (C) sequestration has become an increasingly important consideration for forest management in North America, and has particular potential in pine-dominated forests of the southern United States. Using existing literature on plantations and long-term studies of naturally regenerated loblolly (*Pinus taeda*) and shortleaf (*Pinus echinata*) pine-dominated stands on the Crossett Experimental Forest, allometric biomass equations, and reasonable assumptions about forest product life cycles, we projected the net C pools of the following silvicultural systems over a 100-year period: a short rotation loblolly pine plantation (4 rotations); a seed tree-based (natural origin) even-aged loblolly/shortleaf pine stand (2 rotations); and an uneven-aged loblolly/shortleaf pine stand (20 cutting-cycle harvests under the selection method). Both the seed tree stand and the intensively managed pine plantation produced large (if fluctuating) quantities (up to almost 190 tons/ha) of aboveground live biomass. Though not as productive as the even-aged treatments, the uneven-aged pine stands produced a steady stream of sequestered C in the form of high quality sawtimber while simultaneously maintaining a steady stock of 61.5 to 78.5 tons/ha of live aboveground biomass. Belowground C sequestration was also substantial in uneven-aged stands, with a fairly constant 13.3 to 16.9 tons/ha of coarse roots in the standing live pine crop. While shorter rotation even-aged stands tend to produce smaller coarse roots, their higher stocking levels more than offset this, and hence these stands have more live belowground biomass during most of the rotation (up to almost 41 tons/ha). By the end of the 100-year simulation, the even-aged stands had sequestered approximately 120 tons/ha of C in live tree and product pools, or about 50 percent more than the uneven-aged stand. The uneven-aged stand, however, maintained a more stable residual live tree C store, and fluctuated (only ± 2 tons/ha/yr) far less than either even-aged treatment. Averaged over the period, annual C storage ranged from 0.38 to 1.11 to 1.16 tons/ha for the uneven-aged, seed tree, and plantation, respectively. Though it is difficult to compare these values to other simulations, the data show that managing loblolly pine stands is an effective way to sequester C, even if their patterns differ appreciably.

Keywords: Crossett Experimental Forest, Loblolly pine, shortleaf pine, silviculture

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Introduction

Bioenergy production and carbon (C) sequestration have become a major emphasis for silviculture in recent years (for example, Gan and Smith 2007; Mead and Pimentel 2006). To date, afforestation has garnered the bulk of the economic interest because current commercial markets for C sequestration require the long-term accumulation of atmospheric C on locations presently lacking tree cover (Birdsey 2006). However, foresters and policy makers are trying to modify the nature of these markets to get credit for C accumulated in and the forest products

generated from existing stands. If successful, this broadening of C credits for silvicultural practices other than afforestation may significantly boost the income potential on millions of hectares of productive timberlands.

The possibility of C credits is fueling research into understanding the differences between management practices and their ability to sequester C, which in turn has raised a number of questions. For instance, are there quantifiable differences between long-term C sequestration patterns by silvicultural system? Many people assume that productivity is a reasonable measure of C accumulation, and therefore, fast growing pine plantations may be considered most desirable. However, much of the long-term C storage benefits may be found in end product usage. Commodities such as dimensional lumber or plywood last much longer than short-lived products (such as paper) that are often the primary output of short-rotation plantations (Johnsen and others 2001; Markewitz 2006). Would naturally regenerated southern pine stands geared towards quality sawlog production sequester more C over the long-term than a short-rotation, intensively managed pine plantation that produces more fiber than boards?

We will address these questions using long-term data accumulated on the USDA Forest Service's Crossett Experimental Forest (CEF) and adjacent industrial lands in extreme southern Arkansas. The naturally regenerated forests of the CEF are predominantly loblolly pine (*Pinus taeda* L.), with lesser amounts of shortleaf pine (*Pinus echinata* Mill.) and limited quantities of oaks (*Quercus* spp.), gums (*Liquidambar styraciflua* L. and *Nyssa sylvatica* Marsh.), and other hardwoods. The CEF is characteristic of mesic, relatively productive Upper West Gulf Coastal Plain upland sites, most of which have been in timber production for decades, and many of which have been recently converted to intensively cultured loblolly pine plantations (Wear and Greis 2002). This makes the results from studies on the CEF an appropriate source of information to compare long-term C sequestration patterns under different silvicultural systems.

Methods

Silvicultural Systems

To consider broad-scale differences between silvicultural systems rather than specific real-world stands, a number of "model" systems will be used to represent stand dynamics related to C sequestration. These idealized systems are based on examples of long-term growth and yield from southern pine stands in the Upper West Gulf Coastal Plain, which has a rich history of diverse and sustainable forestry varying from even-aged plantations to uneven-aged selection. As with all silvicultural systems, there are a number of possible conditions and management objectives—the treatments described below follow typical standards and practices for loblolly pine-dominated forests in southern Arkansas.

Uneven-aged stand—The loblolly pine-dominated uneven-aged stand in this exercise was modeled after examples of the selection method using periodic cutting-cycle harvests provided in Baker and others (1996), primarily adapted from data on the CEF's Good and Poor Farm Forestry Forties. Our simulated stand has a reverse J-shaped distribution, with a residual basal area of 13.8 m^2/ha and 345 stems/ha (merchantable pines only), a maximum d.b.h. of 53.3 cm, and a q-factor of approximately 1.2 for 2.5 cm d.b.h. classes (table 1). In the 5 years following any given cutting-cycle harvest, this stand is assumed to reach 17.2 m^2/ha of basal area and just under 400 merchantable pines/ha, producing 22.0 m^3/ha of sawtimber.

Table 1—Structure of uneven-aged loblolly pine-dominated stand immediately post-harvest (year = 0) and immediately prior to the next harvest (year = 5).

D.b.h. class midpoint	Live pine stocking by d.b.h. class		Live pine basal area by d.b.h. class		5-year harvested sawtimber volume
	Year = 0	Year = 5	Year = 0	Year = 5	
cm	*- - - - - trees/ha - - - - -*		*- - - - - m^2/ha - - - - - -*		*- - - - m^3/ha [a]- - - -*
			Submerchantable stems		
1.3	579.4	695.2	0.07	0.09	—
2.5	331.1	397.3	0.17	0.20	—
5.1	189.2	227.0	0.38	0.46	—
7.6	108.1	129.7	0.49	0.59	—
Subtotals:	1207.8	1449.2	1.11	1.34	—
			Merchantable stems		
10.2	61.8	74.1	0.50	0.60	—
12.7	46.2	49.4	0.59	0.63	—
15.2	40.8	44.5	0.74	0.81	—
17.8	35.8	39.5	0.89	0.98	—
20.3	30.9	34.6	1.00	1.12	—
22.8	25.9	29.7	1.07	1.22	—
25.4	24.5	24.7	1.24	1.25	0.05
27.9	18.5	20.0	1.14	1.23	0.55
30.5	13.6	17.3	0.99	1.26	1.65
33.0	11.4	14.6	0.97	1.25	1.84
35.6	9.4	12.1	0.93	1.20	1.92
38.1	7.7	10.4	0.87	1.18	2.31
40.6	6.7	9.6	0.87	1.25	2.98
43.2	5.4	7.4	0.80	1.09	2.31
45.7	3.5	4.9	0.57	0.81	2.00
48.3	2.0	3.7	0.36	0.68	2.68
50.8	1.2	2.5	0.25	0.50	2.17
53.3	0.0	0.7	0.00	0.17	1.47
Subtotals:	345.3	399.7	13.78	17.23	21.93
Grand totals:	1553.1	1848.9	14.89	18.57	21.93

[a] Calculated from equations in Farrar and others (1984) for sawtimber-sized pines only.

Seed tree stand—There are many possible permutations in how to conduct seed tree management in southern pine stands, depending on initial conditions, desired rotation length, the number of seed trees to retain following the regeneration harvest, the duration of overwood retention, etc. For convenience, we chose a strategy that incorporated a 50-year rotation (hence, 2 full rotations during our 100-year simulation period) and retained just under 20 seed trees/ha. We assume that 12,355 new loblolly pines/ha are successfully established after the regeneration cut, and all of the seed trees are harvested in year 5 in conjunction with a precommercial thinning to reduce the number of pines to 1,347 stems/ha by year 6. Commercial thinnings were performed in years 18, 25, 35, and 43, reducing stand density down to 494, 309, 222, and 124 pines/ha, respectively. After 50 years under this prescription, loblolly pines in the simulation were assumed to reach an average of 47.8 cm in d.b.h. An excellent description of the shelterwood system in loblolly pine-dominated stands in this region is presented in Zeide and Sharer (2000). Specific data used to develop this silvicultural regime were adapted from a variety of studies in even-aged, naturally regenerated, loblolly pine-dominated stands on the CEF, including Cain (1996), Cain and Shelton (2001, 2003), and Bragg (2010).

Plantation—The plantation used to exemplify artificial regeneration typical of this region assumed that 1,347 seedlings/ha of 1-0 genetically improved loblolly pine stock were planted on 2.4- by 3.0-meter spacing and were managed on a 25-year rotation. Although plantations in this region commonly recruit volunteer pines from nearby seed sources, we assumed that site preparation and release eliminated these unwanted volunteers. At 11 and 17 years, commercial thinning treatments were implemented to reduce this stocking to 445 and 222 pines/ha, respectively. At 25 years, when these improved loblolly pines were assumed to average 37.1 cm at d.b.h., all trees were cut and the site was replanted. For convenience of the calculation, we assumed that loblolly pines were immediately planted to 1,347 seedlings/ha after logging was completed so that no year went without the presence of a plantation (though in the real world, plantation re-establishment often occurs months to a couple years after harvest).

Model Design and Assumptions

Modularity—This modeling exercise assumes modularity of the treatments, with each silvicultural system module based on either cutting cycle (5-year cutting cycle for the uneven-aged example) or rotation length (25-year rotation for the plantation example or 50-year rotation for the seed tree example). Each module within each treatment is modeled exactly the same, with identical stand developmental trajectories and treatment implementations. Furthermore, no mortality or other stochastic events are assumed, making all of the modules in this system deterministic. However, biomass and sequestered C values are carried over from one module to the next, producing a running total. Hence, the results reported are for a single projection in Excel™ (in other words, no replication). While this focuses on the silvicultural treatments under idealized circumstances, it also likely produces overly optimistic growth and yield predictions.

Tree allometry—To standardize biomass values, a nationally derived set of estimators was applied. What the equations of Jenkins and others (2003) sacrifice in terms of local accuracy is offset by the needs of this simulation to segment individual trees into their primary biomass components (in other words, coarse roots, stem wood, stem bark, branches, and foliage) in a compatible system. Undoubtedly, in any given stand (even intensely managed pine plantations) there will be individual-based differences in attributes such as species, wood density, shoot:root ratio, bark thickness, leaf area efficiency, decay presence, among many other variables. We do not have the ability to account for all of these differences—hence, we are better off recognizing this inadequacy and emulating a "standard" tree following well-described factors we can control (for example, silvicultural system, rotation or cutting cycle length, stand density).

Jenkins and others (2003) used a series of published equations on biomass for different tree species groups to derive "pseudodata" that were then used to generate a set of equations for species groups based on a number of factors, including phylogenetic relationship, adequacy of the original equations and data, and similarity of wood specific gravity. Each species group equation was fit to a common logarithmic model form:

$$B = \frac{e^{b_0 + b_1 \ln d.b.h.}}{1000} \qquad [1]$$

where B = total aboveground biomass (in metric tons), $d.b.h.$ = diameter at breast height, and b_0 and b_1 are group-specific coefficients. Note that in this paper, all biomass or C sequestration weights are given in terms of oven-dry metric tons (1 metric ton = 1,000 kilograms = 1 megagram). Because we considered only loblolly and shortleaf pine-dominated natural stands and loblolly pine plantations,

we used their pine species group coefficients (b_0 = -2.5356 and b_1 = 2.4349, table 4 in Jenkins and others (2003)).

Jenkins and others (2003, their table 6) also developed a series of coefficients to estimate the proportions of different tree components for conifers based on the following ratio equation:

$$R_i = e^{b_{0,i}+b_{1,i}/d.b.h.} \quad [2]$$

where R_i = the ratio (0.0 to 1.0) of component i to total aboveground biomass. Note that the coarse root component is in addition to the biomass calculated in equation [1], and that the proportion of branch biomass (R_{BRANCH}) = 1.0 – R_{WOOD} – R_{BARK} – $R_{FOLIAGE}$. The ratios predicted by equation [2] do not explain a lot of the variation in the data they were derived from, but follow logical proportioning patterns, and vary reasonably as a function of tree diameter. Future iterations of this effort may be improved with the adaptation of more conceptually robust allometric relationships (for example, Enquist 2002; Enquist and Niklas 2002).

Biomass and post-harvest C storage pools—Live tree biomass (both above- and belowground) is tracked every year from the start of the even-aged stands. For uneven-aged stands, the residual contains the live trees retained to meet the minimum basal area target (in this case, 13.8 m^2/ha)—neither the seed tree stand nor the plantation has such a consistent base C storage.

Once harvested, there are two post-harvest biomass pools that each tree is proportioned to—the fast (short-term) and slow (long-term) pools. These pools represent the dynamics of felled trees post-harvest whether utilized as forest products or left on the site as slash. Fast pool biomass consists of finer materials such as smaller roots, bark, foliage, and certain portions of the bole that either decompose quickly or are converted into short-lived consumer goods such as paper products. Slow pool biomass is assumed to be larger portions of the bole and large coarse roots that are left on-site as coarse woody debris or converted into long-lived consumer goods such as boards or structural panels. These pools assume half-lives of 1 year for the fast pool and 50 years for the slow pool (Birdsey 1996) using the following exponential decay function:

$$\Psi_t = \Psi_0 e^{-\lambda t} \quad [3]$$

where the original harvested biomass (Ψ_0, fast or slow) reduces to Ψ_t after t years. For equation [3], the decay coefficient λ = 0.69315 for the fast pool and 0.01386 for the slow pool.

All stems less than 25.4 cm d.b.h. are automatically assigned to the fast pool because of their rapid decay or use as short-lived consumer products. Large bolewood was partitioned into slow and fast pools based on the lumber recovery work of Fonseca (2005, his table 6.6). For 25 cm d.b.h. stems, only 30 percent of their stem wood is assigned to the slow pool because the rest is either converted to sawdust, slabs, chips, or planer shavings, or is in part of the bole that is too small to decay slowly. The proportion of slow pool biomass from stem wood gradually increases to 45 percent when the tree reaches 40 cm d.b.h., and is held constant at this ratio no matter how much larger the stem gets. The 55 to 70 percent of bole biomass not assigned to the slow pool is transferred to the fast pool for time-related decay, as were all of the foliage, bark, and branches.

To convert from biomass (in both the live biomass and post-harvest pools) to weight of sequestered C only, we simply multiplied the biomass total by 0.5 (Johnsen and others 2004; Skog 2008). For this paper, sequestered C weight was defined as the sum of the above- and belowground live C weight plus the weight of the C in the current fast and slow post-harvest pools for any given year.

Other assumptions—Because this research is initial and exploratory, certain significant C pools are not expressly included in this analysis. For instance, the only vegetation being tracked are the crop loblolly pines that are either planted or explicitly included in the modular natural regeneration scenario. No other non-pine vegetation (for example, grasses, forbs, shrubs, vines, hardwoods, etc.) is tracked in the C pools. Likewise, no "volunteer" pines are incorporated in plantations, nor are pines that would otherwise seed in following the thinnings of even-aged natural stands. Though all of these components are noticeable parts of the ecosystem C pools, we lacked good, quantifiable information on how they would respond to the specific treatments presented over time, and therefore, we chose not to include them. Note, however, that the uneven-aged stand did assume continuous pine regeneration since this is how this stand condition perpetuates itself (rather than relying on discrete establishment events).

Two other dynamic components likely to be very important but poorly described are the soil and forest floor C pools (Birdsey 2006). Rather than trying to estimate these values and simulate their behavior, we will assume there are no significant differences between the size of the pools in any of the silvicultural treatments we are comparing. Studies performed on loblolly pine stands in the southeastern US have provided mixed results regarding these components, with some treatments decreasing and others increasing soil C (for example, Laiho and others 2003; Samuelson and others 2004). Most of these studies are relatively short-term (less than 10 years) and often focus on soil C dynamics for plantations managed under a gradient of treatment intensity rather than mature loblolly pine stands of natural origin. So, given the paucity of reliable information, we believed the bulk of the belowground contribution to C sequestration in these pine stands would be best dealt with in a later analysis. Hence, the only subsurface sequestration components in this paper are the coarse roots.

There are also C emission impacts related to harvest system and equipment type/usage by silvicultural system (Eriksson and others 2007; Markewitz 2006). Because this particular part of the C cycle depends strongly on the type of equipment and how it is used, coupled with the quantities of fossil fuel-based fertilizers and other chemicals applied and the nature of the site being treated, we have chosen to assume that there are no significant differences between our silvicultural systems. Ignoring this component is not likely to have a marked influence on overall C dynamics, anyhow—Markewitz (2006) estimated that cumulative C emissions for all silvicultural activities on an intensive fiber farming system using southern pines over an entire 25 year rotation was only about 3 tons/ha.

Finally, we did not explore the economic ramifications of these treatments (or any variations of them), so any conclusions on the efficacy of these silvicultural regimes are based on their ability to store C in either slow or fast post-harvest pools or residual (live) crop tree biomass on the site. We also did not assume any post-harvest consumer products were recycled, nor did we include C offsets due to product displacement or substitution.

Results

Aboveground Live Tree Biomass

Marked differences arose between the biomass patterns between these silvicultural systems in loblolly pine dominated stands (fig. 1). The seed tree (natural origin) stand, starting from the regeneration cut, contained more biomass (19.2 tons versus 0.0 tons/ha) than the plantation, which started from a true clearcut. Aboveground biomass in the seed tree increased rapidly as both the newly established

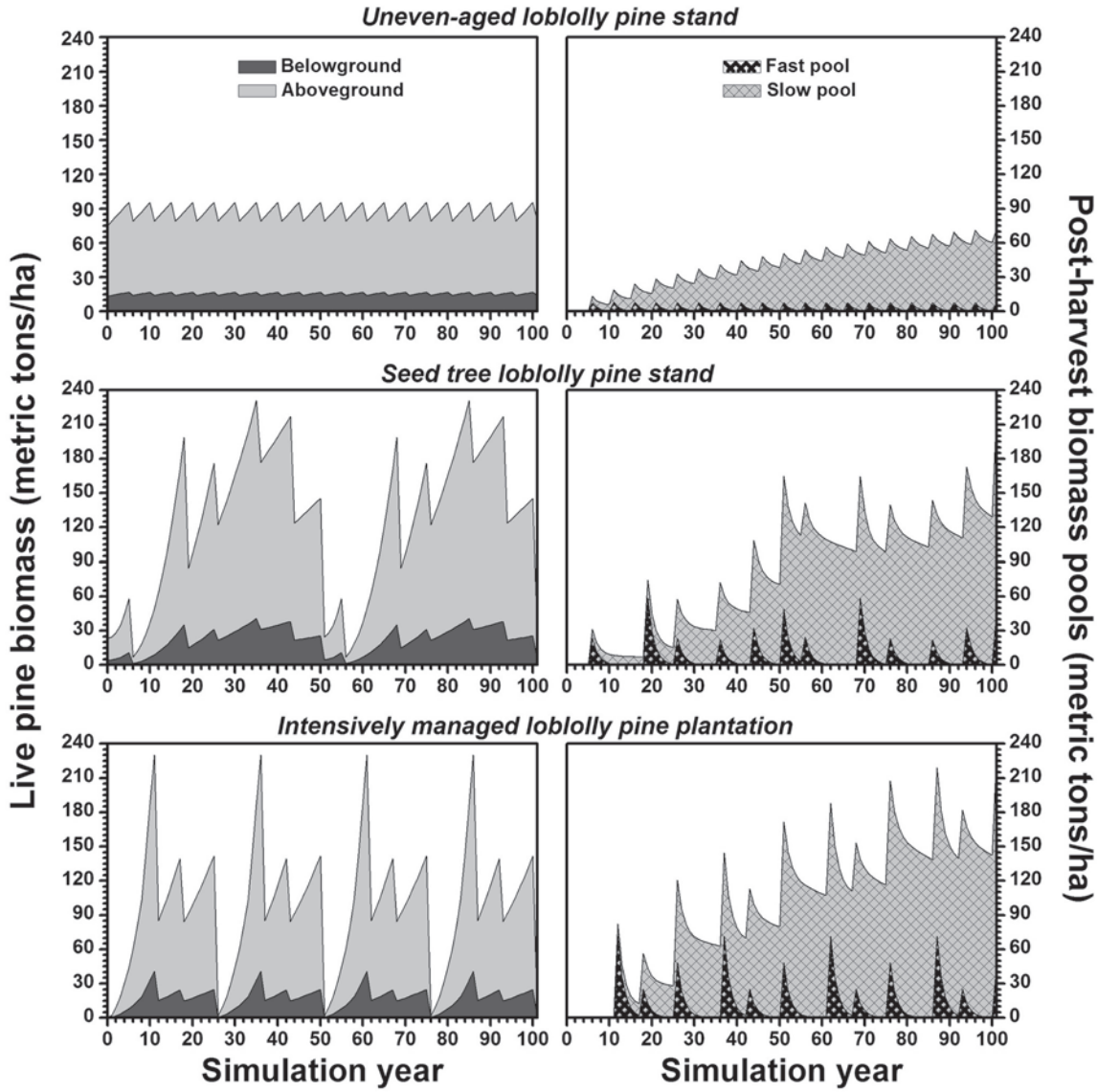

Figure 1—Biomass fractions in live loblolly pines (aboveground biomass + coarse roots) and contributions to different post-harvest product biomass based on fast (1-year half-life) and slow (50-year half-life) decomposition (loss) pools for three silvicultural systems.

pine seedlings and overwood pine grew quickly, reaching 47.2 tons/ha in year 5. The harvest of the seed trees and concurrent precommercial thinning resulted in a sharp decrease in biomass, noticeably below the level of the similarly stocked loblolly pine plantation. This biomass difference after 6 years (5.6 versus 49.5 tons/ha) is attributable to the significantly larger size of the improved pine seedlings in the plantation, which had been growing at a low density since planting compared to the considerably higher stocking of the natural origin seedlings during the first 5 years of their life.

The seed tree stand and the plantation both rapidly added biomass during the remainder of their rotations, periodically experiencing sharp drops as thinning operations and regeneration harvests removed biomass (fig. 1). Not surprisingly, the more intensively managed even-aged stands experienced substantially higher peak live biomass totals than the uneven-aged stand. Both even-aged stands

approached 190 tons/ha of aboveground live biomass—the plantation reached just under 190 tons/ha first, immediately prior to the first commercial thinning at age 11, before fluctuating between 69 and 117 tons/ha for the rest of the rotation. The seed tree stand peaked much later in its rotation, just exceeding 190 tons/ha in year 35 (right before the third thinning). The seed tree stand contained between 100 and 150 tons/ha of live pine biomass during the last 3 decades.

In contrast to the even-aged stands, biomass in the well-regulated uneven-aged stand managed under the selection system (Baker and others 1996) varied little over time. Immediately following each cutting-cycle harvest, the simulated uneven-aged loblolly pine-dominated stand started out at 61.5 tons/ha of live pine aboveground biomass and quickly increased to 78.5 tons/ha 5 years later (fig. 1). The closely controlled stand density of the uneven-aged stand kept any of the high or low aboveground live pine biomass amounts from fluctuating nearly as much as the even-aged stands.

Belowground Live Tree Biomass

Because belowground live tree biomass was determined as a relatively fixed proportion of aboveground live pine biomass (a function of diameter), this component closely paralleled the aboveground patterns. Even-aged stands peaked near 41 tons/ha of belowground biomass when aboveground biomass was peaking at 190 tons/ha, and would decline to less than 2 tons/ha at the end of each cutting cycle while the new stands were establishing themselves (fig. 1). The uneven-aged stand varied between 13.3 and 16.9 tons/ha of live pine belowground biomass across the entire 100-year simulation period.

Post-harvest Biomass Pools

The harvested biomass showed a steady increase in quantity for the slow pools (fig. 1). In all cases, the slow pools decayed at a rate that did not reach equilibrium with new inputs during the 100-year simulation period. Thus, all of the silvicultural treatments continually added to their respective slow biomass pools. Fast biomass pools, however, decayed quickly enough so that even the fairly substantial (greater than 60 tons/ha) periodic contributions by either even-aged treatment failed to build upon other pulses of fast pool biomass, and typically remained at less than 1 ton/ha during most of the simulation period.

C Sequestration Patterns

Long-term C stocks varied considerably over time, with the highest fluctuations once again being found in the even-aged treatments (fig. 2). The combination of above- and belowground biomass with the post-harvest product pools helped to dampen some of the more pronounced oscillations in the even-aged stands, but both still experienced dramatic changes. At their peaks, both even-aged stands exceeded 160 tons of C in their respective simulated stand developmental trajectories (before settling down to about 120 tons C at 100 years), and seem destined to continue to accumulate C well into the future.

The uneven-aged stand surpassed the other even-aged treatments in C storage only briefly during the first 3 simulated decades—from the first 8 to 14 years for the plantation and seed tree stands, respectively, and then again between 26 and 31 years after the logging and reestablishment of the plantation (fig. 2). C steadily accumulated in the uneven-aged stand and its related post-harvest pools from a low of 37.4 tons/ha to about 76 tons/ha at the end of the 100-year simulation period. Variation from year to year in the uneven-aged stand showed a much more stable pattern, with fluctuations of less than 2 tons/ha typical during the simulation.

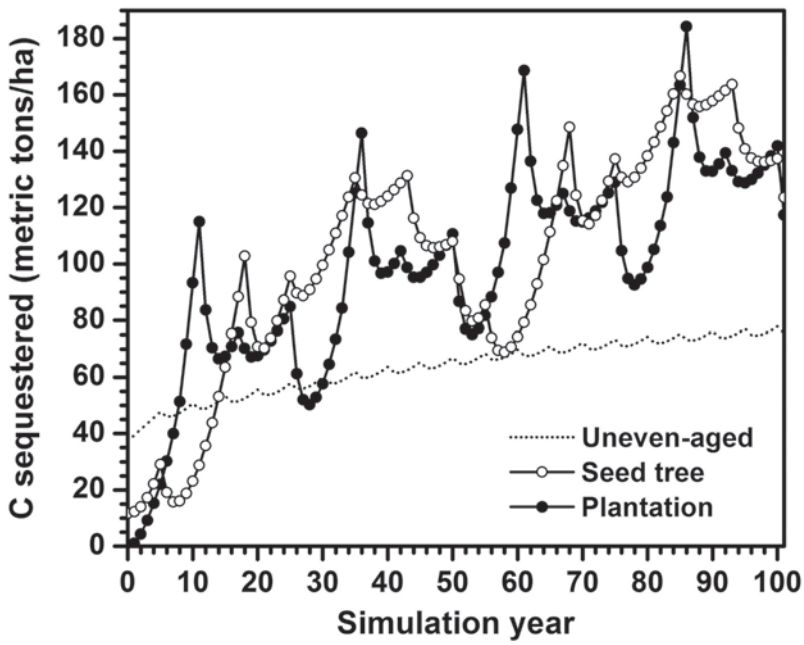

Figure 2—Sequestered carbon (C) in above- and belowground live trees and fast and slow product pools over a 100-year simulation of three silvicultural systems in loblolly pine stands.

Considerable variation appears in C storage patterns from one year to the next (fig. 3), although this pattern was far less pronounced for the uneven-aged stand. The uneven-aged scenario varied by ± 2.2 tons/ha/yr (standard deviation (SD) = 1.36 tons/ha/yr)), compared to between +14.4 and -24.2 (SD = 7.25) tons/ha/yr for the seed tree stand and +21.7 and -32.3 (SD = 11.90) tons/ha/yr in the pine plantation. Over the 100-year simulation period, average annual C sequestration was positive for all treatments, ranging from 0.38 tons/ha/yr in the uneven-aged stand to 1.11 tons/ha/yr for the seed tree stand to 1.16 tons/ha/yr for the plantation.

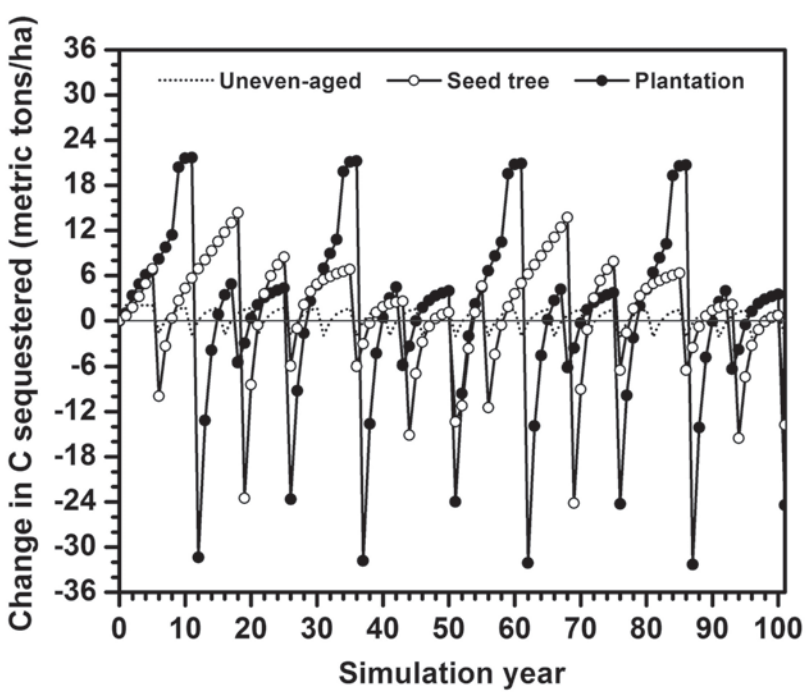

Figure 3—Change in sequestered carbon (C) from one year to the next during the 100-year simulation period as a function of silvicultural system in loblolly pine-dominated stands.

Discussion

This preliminary study makes a number of key oversimplifications in order to examine silvicultural systems in isolation. For instance, some of our assumptions (such as the decay coefficients for the fast and slow pools) apply commonly accepted values, but no sensitivity testing is presented here to evaluate the consequences of altering the rates of change on C sequestration patterns. Given the wide range of differences between fast and slow pool behavior with the two extremes of decay rates, we would expect this to vary considerably based on the values used. For instance, it may be more appropriate to use a slower decay rate for the belowground slow pool for the large sawtimber trees produced in the natural origin stands, as these stumps are often resin-soaked and therefore much more resistant to decay than younger, faster-grown plantation stumps of comparable size. The multitudes of possible management activities, coupled with the likelihood of stochastic mortality events, changing climate patterns, and even developments in the genetic improvement of loblolly pine may also drastically affect the outcomes of the silvicultural scenarios presented. It is simply not possible to determine every possible interaction and predict their influence on the outcomes.

The aforementioned qualifications notwithstanding, it is obvious that C sequestration patterns in southern pine stands, when the fate of the materials produced from them is included, show considerable potential to offset some atmospheric CO_2 increases. The sustained accumulation of C in this study is comparable to other simulation-based research (for example, Baral and Guha 2004; Johnsen and others 2001) and suggests that the active management of southern pinelands may increase atmospheric CO_2 sequestration over no-treatment options.

C storage patterns fluctuate dramatically in managed ecosystems, especially those under even-aged management (fig. 3). Because of how this exercise was designed, large negative values in the live biomass portion of this metric only occur in a single year, as regeneration and residual tree growth immediately begin accumulating new C. Any prolonged deficits in C storage shown in figure 3 reflect periods after major timber harvests, when new growth fails to offset C losses due to post-harvest product decay. Even with these prolonged deficits, stand-level C accumulation over the whole simulation period was positive for each treatment, ranging from 0.38 to 1.16 metric tons/ha/yr.

However, it is hard to compare these results with other published studies because different components are often used to derive C accumulation. For instance, Luyssaert and others (2008) used carbon flux estimates to find that old-growth temperate and boreal forests sequestered an average of 2.4 tons of C per hectare per year, a majority (83 percent) of which was attributed to C stored in coarse woody debris (CWD), roots, and soil organic matter. While the CWD pool in their study is analogous to our post-harvest product pools, Luyssaert and others (2008) also incorporated fine root biomass in their root category (we only considered coarse roots) and our study has no information on soil organic matter. It is also unclear how Luyssaert and others (2008) accounted for the decomposition of wood. Other estimates of C sequestration by temperate forests (for example, Baral and Guha 2004; Hall and others 1991) consider only C accumulation via growth, and do not account for simultaneous losses to the system. If couched in accretion-only terms, average annual C accumulation values of 0.81, 3.36, and 4.82 tons/ha/yr were calculated for the uneven-aged stand, the seed tree stand, and the plantation in this study, respectively. Undoubtedly, these values would increase even more if soil C storage was also factored into these estimates.

The long-term C sequestration benefits of southern pine forests under management also depend on the end-use of the biomass produced. Shifting products

from slow pool into fast pool consumer goods (for example, dimension lumber or plywood into paper products) would have a marked effect on C sequestration, especially if this happens at large scales. While this study has focused on C storage based on either biomass retained on the landscape or preserved within long-term commercial products, net C benefits may also be derived if woody biomass (rather than fossil fuels) are used to produce energy or other derived products (for example, Baral and Guha 2004; Frederick and others 2008; Gan and Smith 2007; Hall and others 1991), replaces corn-based ethanol with cellulosic ethanol (Piñeiro and others 2009), or is substituted for higher C-emitting building materials such as steel or concrete (Perez-Garcia and others 2005).

Conclusions

Undoubtedly, our results are sensitive to a number of possible variables, including the decay coefficient used for both fast and slow pools, mortality of individual pines as a function of treatment and random factors, regeneration success or failure, and differences in wood density related to genetics and growth habit. Our goal was not to consider all possible conditions and permutations related to the C cycle, but to explore the role of silvicultural practices on an aspect of stand management (C sequestration) only poorly considered to date.

Hence, we believe our results show that C sequestration patterns in southern pine-dominated forests depend considerably on the silvicultural system being applied. According to our projections and the work of others (for example, Johnsen and others 2001; Smith and others 2006), all of the conventional management practices used in loblolly pine ecosystems of the southern US have potential to accumulate C in standing timber, the soil, and long-term end products. Intensively managed loblolly pine plantations, due to their high rate of fiber production (and assuming that most of their products end up in slow decaying uses), were predicted to accumulate C at the highest rate over the course of a 100-year simulation.

However, it is possible to have mature, natural origin southern pine forests produce significant amounts of C storage. Loblolly pine-dominated stands managed under a seed tree regeneration approach accumulated C at a rate very comparable to a loblolly pine plantation. The timing of harvest and regeneration in the two even-aged harvest regimes produced some differences in C sequestration patterns, but both were predicted to store around 120 metric tons of C per hectare after 100 years of growth and harvesting. This total is at least 50 percent higher than that accumulated under uneven-aged management of loblolly pine-dominated stands at the end of the same period.

The perpetual understocking of uneven-aged southern pine stands to ensure adequate regeneration also assures lower C sequestration. Uneven-aged pine stands are also much less variable in their C accumulation patterns, since they always retain a large quantity of live biomass on the site. However, it may be possible to manage uneven-aged southern pine stands on a more irregular basis with cutting cycles longer than conventional 5-year intervals. Doing so would likely result in increased rates of C accumulation somewhat similar to that seen in the seed tree method, and concurrently would increase C sequestration while retaining the continuous cover canopy attributes sought by managers who utilize this silvicultural system.

The opportunities presented by bioenergy and C sequestration may have particular appeal for public lands management in the southeastern US. Given that few governmental agencies can engage in the large-scale industrial forestry of intensively cultured loblolly pine plantations, the ability to produce significant C storage while harvesting high-value timber products under naturally regenerated,

sustainable forest conditions would seem to fulfill multiple resource objectives simultaneously. To ensure this, it is vital that work (for example, Johnsen and others 2004; Mayfield and others 2007) considering the possibility of using forests for bioenergy and C sequestration in the southeastern US be expanded to include the unique statutory, regulatory, and policy obstacles that may supersede these opportunities on federal lands.

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The Role of Forests in Energy and Climate Change—Integrating Objectives

Dave Atkins[1]

Abstract—Woody biomass utilization presents a tremendous opportunity to address energy independence directly through the use of domestic wood as an energy source that offsets fossil fuel. It also works indirectly through conservation by the substitution of wood for higher embodied energy construction materials such as concrete and steel. Both the direct and indirect measures mitigate climate change by reducing greenhouse gas emissions. By integrating these goals with treatments of forests to make them more resilient to disturbances, longer term C sequestration is accomplished in the live forest on the landscape. This paper explores future and current technologies that are available to accomplish utilization and sustainable treatments in the field.

Introduction

Woody biomass utilization presents a tremendous opportunity to address energy independence directly through the use of domestic wood as an energy source that offsets fossil fuel and indirectly by the substitution of wood for higher embodied energy construction materials such as concrete and steel; moreover, it mitigates climate change by reducing greenhouse gas emissions. Integrating forest treatments that use woody biomass can also fulfill other objectives, such as creating forests that are resilient to disturbances and able to provide long-term C sequestration. In this paper, I present current and future technologies that are available to accomplish utilization and sustainable treatments in the field.

There is a need to move toward more sustainable systems over the next 30-40 years to achieve the goal of reducing C emissions. Forests are natural solar collectors that store energy and filter carbon from the atmosphere and store it in "wood batteries." Given that forests can store carbon for long periods (decades to centuries), conducting sustainable forest management activities and techniques to insure continued forest growth and development is one method of reducing carbon emissions in addition to sequestering carbon. One measure of resiliency is the post-wildfire environment. For example, in Colorado, the 1989 Sheep Rock fuel treatment area burned more variable and moderate compared to surrounding untreated sites. Because there were more abundant live trees, these sites continue to sequester C in the trees.

Wood as a Conservation Tool

The use of wood in place of steel and concrete results in C being sequestered in the wood for the life of the building; importantly, it can also save significant amounts of fossil C from being emitted during production. Figure 1 shows an example of the net energy consumed to extract, transport, manufacture, and erect 100 square feet of interior wall using various framing materials (fig. 1a)

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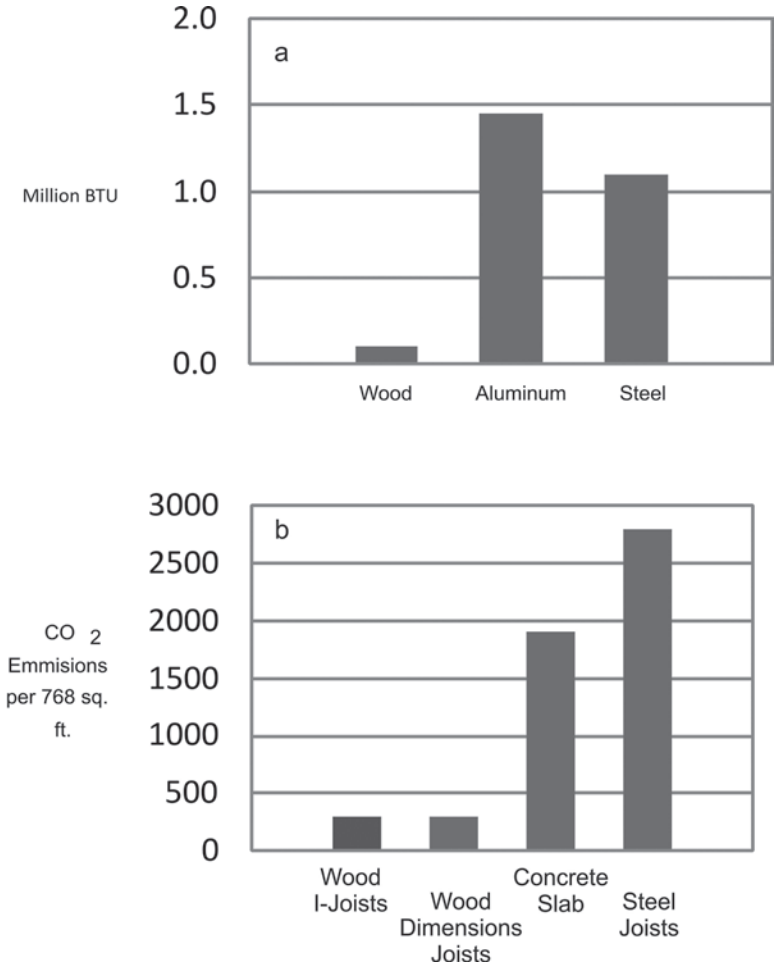

Figure 1—(a) The net energy consumed to extract, transport, manufacture, and erect an interior wall that is 100 square feet using wood, aluminum, and steel (CORRIM 1976). (b) The CO_2 emissions from floors constructed from wood I-joists, wood dimensional joists, concrete slab, and steel joists (CORRIM 2009).

and CO_2 emissions from different materials used for floor construction (fig. 1b). Products created from small wood, such as Glulam beams, and the use of wood for flooring, furniture, and engineering trusses are ways to store C and reduce energy consumption.

Energy Technologies With Wood

There are numerous technologies under research and development at various stages, such as fast pyrolysis to make bio-oil and biochar, gasification to liquid fuels, fermentation processing of ligno-cellulose to liquid fuel, and small-scale gasification for producing heat and power for power plants. However these new products are not commercially available. Because of a lack of existing markets for the product, they are untested in the marketplace; thus there is reluctance on the part of buyers to convert to the new product. Most significantly, the volatile nature of fossil fuels results in these products being more competitive during times when the price of fossil fuels is high compared to when it is low. Therefore, investors

are reluctant to make large investments given the risk that the low price times will kill the viability of the product. Along with a move towards a more sustainable set of energy sources, the efficiency of various uses of woody biomass should be considered. Current policies in the United States provide incentives to electrical and liquid production that are only 25 and 40 percent, respectively, efficient at converting the energy in the wood to useable energy for the consumer. On the other hand, thermal and combined heat and power are 65-90 percent efficient, yet have no incentives at the federal level. Policies to encourage efficient use of energy would certainly be better in the long-run.

Efficient Production and Transportation of Fuel

Slash and small trees have traditionally been viewed as a disposal problem; a cost problem rather than a potential product and revenue stream. The negative value of the material results in it being treated like trash with dirt and rocks being mixed into the fuel. While industrial systems can tolerate the dirt and rocks, albeit with a higher operating and maintenance cost, smaller commercial and institutional systems often cannot handle the low quality fuel without significant costs. Chipped material is preferable to ground material because it is more efficient in smaller combustion systems. However, chipping is typically more expensive than grinding, road systems to access many projects that generate potential fuel cannot handle highway chip vans, and chipping and grinding in the woods can be expensive due to significant lost production while moving to and from concentrations of slash. Many different methods and types of equipment are being developed and tested to improve the efficiency of production and transportation of material.

Conclusion

Wood utilization by an integrated industry can provide a wide range of products from solid wood and composites that can conserve energy as a result of lower energy content compared to concrete, steel, and plastics. It can provide a direct substitution for fossil fuel and thus reduce fossil C emissions. The sustainable use of forest products can provide for greater resilience of the forest to disturbances and thus continue to sequester more C in live forests as well as in forest products. The integration of management objectives for watershed protection, wildlife habitat enhancements, reduction of the potential for greater severity of wildfires in the context of climate change, and production of renewable energy adds to the considerations a silviculturist must make in developing landscape and stand level prescriptions. To achieve this level of integration requires consideration and communication of the various trade-offs.

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The Forest—Bioenergy—Carbon Connection

Jay O'Laughlin[1]

Extended Abstract—Burning wood for energy is a back-to-the-future approach for solving modern problems. The burning of fossil fuels for energy and resultant carbon emissions are global concerns: "The world needs ever increasing energy supplies to sustain economic growth and development. But energy resources are under pressure and carbon dioxide (CO_2) emissions from today's energy use already threaten our climate. What options do we have for switching to a cleaner and more efficient energy future?" (IEA 2008). The basic choice is between action and delay, and forest management is among the set of options that have the capacity to provide "stabilization wedges" and solve the climate problem for the next half-century (Pacala & Socolow 2004).

Wood bioenergy is a proven, cost-effective technology for producing homegrown, reliable baseload energy (O'Laughlin, in press). In 2007 wood provided 1.8 percent of the energy consumed in the United States (EIA 2009). Wood bioenergy use is generally higher in states with extensive forest resources and a substantial forest products business sector, like Idaho, where 4.7 percent of the energy consumed is from wood (Idaho Legislature 2007). Forest products manufacturing mill residues are the low hanging fruit for wood bioenergy, as they have already been transported out of the woods, but almost all mill residues are already used to make energy (Nicholls and others 2008).

Forest health thinnings and logging residues are a potential source of feedstocks for wood bioenergy, whether for space heating with thermal energy, electric biopower, or liquid biofuels. The benefits of producing thermal energy and biopower from wood can be substantial; e.g., the University of Idaho (2008) saves on average $1.5 million per year burning wood residues from local sawmills instead of natural gas, and Fuels for Schools projects save $1.8 million per year in fuel costs. These thermal heating solutions are the most efficient use of wood bioenergy and widely used in some European countries (see Richter and others 2009).

Using forest residues (thinnings and logging slash) for bioenergy is an opportunity to restore forest health, wildfire resiliency, and wildlife habitat. Silvicultural operations to improve forest conditions can help revitalize rural economies while providing renewable energy feedstocks (Cloughesy & Lord 2006). In addition to this "triple win" (IFPC 2009; OFRI 2009), forests play a key role in the global carbon cycle by capturing, storing, and cycling carbon (EPA 2009a, see also California Forest Foundation 2009). Forests in the conterminous U.S. sequestered, on average, 162 million metric tons of carbon per year during 1990-2005 (Woodbury and others 2007). This is sufficient to offset at least 10 percent of all U.S. carbon dioxide emissions (data from EPA 2009b). However, from 2002-2006 wildfires in the conterminous U.S. emitted, on an annual average, 59 million metric tons of carbon as CO_2 and two million metric tons as particulate matter (Wiedinmyer & Neff 2007).

Considerations for safe storage of carbon on, in, or deep under the soil create roles for forestry (Read 2009). Five types of carbon reservoirs are preferable to storing carbon in the atmosphere, and only the last item does not have an obvious role for forestry: 1) new forestry plantations; 2) new timber structures and other durable wood products from harvested wood; 3) underground wood burial, perhaps in abandoned mines; 4) biochar storage in soil reservoirs with co-produced bio-oil; and 5) carbon capture and storage in deep geological strata or as bicarbonates in the ocean or insoluble carbonates on land in played-out coal mines. In addition, the

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existing fossil carbon reservoir is maintained *in situ* through technology chains that involve bioenergy and other renewable sources of energy that substitute for fossil fuel (Read 2009).

Silviculture designed to reduce stand-replacing wildfires is the most important forestry strategy for mitigating climate change, followed by keeping forest lands in forest cover, putting trees back on the land through afforestation and reforestation, using wood products instead of substitutes like concrete and steel, and modifying harvest rotation length (Cloughesy 2006). Beginning in the mid-1980s, the acreage burned by wildfires in the 11 western states began to increase considerably from levels of the previous 50 years. According to information provided to the U.S. Government Accountability Office by the U.S. Forest Service, "The most extensive and serious problems related to the health of national forests in the Interior West is the over-accumulation of vegetation, which has caused an increasing number of large, intense, uncontrollable, and catastrophically destructive wildfires" (GAO 1999).

Climate change concerns heighten the issue: "The overall importance of climate in wildfire activity underscores the urgency of ecological restoration and fuels management to reduce wildfire hazards to human communities and to mitigate ecological impacts of climate change" (Westerling and others 2006). According to the Intergovernmental Panel on Climate Change, "…a sustainable forest management strategy aimed at maintaining or increasing forest carbon stocks, while producing an annual sustained yield of timber fibre or energy from the forest, will generate the largest sustained mitigation benefit" (Nabuurs and others 2007). Two types of barriers impede implementing this strategy on federal lands.

One of the two primary challenges to utilization of wood biomass that could provide energy feedstocks are high harvesting and transportation costs (GAO 2005, 2006). Perhaps the best way to create more favorable economics that can make biomass removal projects feasible is harvesting higher value timber along with biomass removals (Evans 2008; Nicholls and others 2008). Furthermore, in dry forest types comprehensive restoration treatments are not only more effective at reducing hazard than thin-from-below approaches designed to remove smaller trees only, the economics are more favorable and in many situations can return more revenue than the treatment costs, whereas thin-from-below requires out-of-pocket expenditures (Fiedler and others 2004).

Besides harvesting high value timber along with biomass, the only other way to deal with unfavorable economics of biomass harvesting is through public subsidies (Nicholls and others 2008). A rationale for subsidizing fuel treatments is that the benefits to society from hazardous fuel treatments are worth more than the value of electricity produced by wood biopower plants. These include the uncompensated benefits of reduced costs of fire suppression costs and avoided costs of site rehabilitation as well as avoided costs of carbon emissions, calculated at a conservative estimate of $10 per metric ton (WGA 2006). Other researchers have also quantified the value of these silvicultural benefits (see Mason and others 2006). In part to support bioenergy, the Western Governors' Association "call[s] on the federal government to create a substantial, long-term national public investment on the scale of tens of billions of dollars annually, and encourage at least the same investment from the private sector, to support the kind of basic and applied research and deployment of clean energy technology and infrastructure" (WGA 2009).

The other primary challenge to using wood as an energy feedstock is the lack of a long-term reliable supply (GAO 2005, 2006). To deal with that, some have suggested public programs that create a demand for biomass material (e.g., Williams 2005). One such program is "Fuels for Schools and Beyond" (see Nicholls and others 2008). However, unless entrepreneurs can demonstrate that they have reliable feedstock supplies for 10 or 20 years they will have difficulty attracting capital to wood bioenergy projects. Whether demand will pull out supply or supply will push new demand is a chicken-and-egg argument; both are necessary.

Since the advent of the National Fire Plan in 2000 there has been much positive activity to reduce hazardous fuels, notes Doug Crandall (2006), currently the legislative liaison for the U.S. Forest Service. Referring to the relationship of forest growth,

mortality, and removals he said, "Momentum has shifted from conflict and neglect to a recognition that there's actually a gorilla in the room." He opined that with 190 million acres of forests needing treatment, much more than the current level of four to five million acres per year nationwide needs to be accomplished (Crandall 2006). While serving as Chief of the U.S. Forest Service, Dale Bosworth described the situation as unsustainable: "We have some 73 million acres of national forest land at risk from wildland fires that could compromise human safety and ecosystem integrity. . . . The situation is simply not sustainable—not socially, not economically, not ecologically" (Bosworth 2003).

Idaho and Montana face forest health and wildfire risks from overstocked forests. Together the two states have 7 million acres of forests in the high-risk Fire Regime Condition Class (FRCC) 3 category and another 10.5 million acres in FRCC 2 (Schmidt and others 2002). Federal land managers are reducing fuels on an average of 250,000 acres per year (Healthy Forests and Rangelands 2009). At that rate it would take 28 years to treat just the FRCC 3 lands, at which time vegetation would have grown back and retreatment would be necessary to maintain fuel reduction benefits. Forest managers, stakeholders, and policy-makers need to consider whether passively waiting for the inevitable wildfire to burn is better that actively managing fuels, and whether piling and burning biomass onsite is preferable to moving it offsite and burning it in boilers that produce bioenergy. The State of Oregon subsidizes biomass removal for energy production at $10 per green ton (ODE 2007). A similar program failed to pass the Montana legislature in 2009 (O'Laughlin, in press).

In conclusion, wood bioenergy opportunities are substantial and sustainable. Many communities are interested in installing wood bioenergy facilities to reduce costs to heat public buildings and provide local jobs. In addition, the uncompensated social benefits of reduced wildfire suppression costs, plus the avoided costs of site rehabilitation and carbon emissions, exceed the value of bioenergy and create a rationale for subsidizing hazardous fuel treatments. The implementation question in the forestry sector is whether the subsidy should be in the form of timber from the forest or cash payments from the public treasury and taxpayers' pockets. In the short term hazardous fuel reduction provides a triple win: improved forest conditions, renewable energy feedstocks, and revitalized rural communities. The reduction of carbon emissions from burning wood in a boiler instead of in the woods is a substantial bonus. The long-term payoff from wood bioenergy will be enhanced energy security.

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Biomass Utilization Opportunities to Achieve Diverse Silvicultural Goals

Barry Wynsma[1] and Christopher R. Keyes[2]

Abstract—Silviculturists and ecologists may recommend land management prescriptions that are designed to be resilient to changing climatic conditions. When considering biomass utilization opportunities that may result from climate-change treatments, it really doesn't matter what species mix or stocking levels are to be retained: if there are trees that need to be harvested, there will usually be opportunities for utilizing woody biomass.

Keywords: biomass utilization, harvesting systems, designation by description, silvicultural prescriptions

Introduction

This paper focuses on biomass utilization opportunities in the context of Goals 2 and 3 of the recently released Forest Service Strategic Framework for Responding to Climate Change. Goal 2 is Adaptation—"Enhance capacity of forests and grasslands to adapt." Goal 3 is Mitigation—"Promote management of forests and grasslands to reduce greenhouse gases while sustaining the multiple benefits and services of these ecosystems." The take-home message is simple: it doesn't matter what management prescription you want to apply or habitat type you are working in—as long as you are proposing to cut trees, there will be opportunities to utilize biomass.

Biomass Utilization to Achieve Mitigation in Young Stands

Goal #3, Mitigation, is to promote the management of forests and grasslands to reduce the buildup of greenhouse gases, while sustaining the multiple benefits and services of these ecosystems. There is a wide variety of management scenarios that can integrate biomass utilization solutions while meeting climate change objectives. One possible way to meet the goal is to find ways to utilize logging slash that is excess to other resource needs and is normally disposed of by open burning, which releases CO_2 into the atmosphere. CO_2 emissions from open burning could be vastly reduced if the slash were instead burned in very-low emission biomass-electric generation facilities, which could convert an average-sized grapple pile of slash into about 2,000 kilowatt-hours of electricity. A single grapple pile may also contain enough biomass to convert to between 40 and 80 gallons of ethanol.

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Operational Alternatives to Piling and Burning

One of the most high-tech, and expensive, ways to remove logging slash from harvest units is to employ a "slash bundler" such as the John Deere 1490 (fig. 1). The bundler is capable of operating on slopes up to around 40 percent and can maneuver through residual stands with a leave-tree spacing of about 20 to 30 feet without excessive damage to residual trees. The cost of this machine approaches $500,000, so it needs to be used in locations that have a high volume of biomass removed on a steady basis in order to be economically feasible.

Figure 1—Slash can be densely packaged into movable units (top) with the John Deere 1490 slash bundler (middle), and transported with a typical forwarder (bottom).

Some bundling systems include the capability of removing the bundler attachment to convert the machine to a forwarder, while others require the use of a second piece of equipment that can transport the bundles to a roadside for trucking or on-site chipping. If logging slash is needed to be left on-site in a scattered fashion for a period of time to permit nutrient cycling for instance, this system is useful because slash can be bundled at any time after the timber harvest occurs. A forwarder is the most likely equipment for yarding bundles (fig. 1, bottom), but grapple skidders or other equipment could also be used. Still other bundlers or "balers" are designed to either be pulled through the woods behind a machine or set up at a landing. Data gathered by the Southern Research Station and Forest Products Lab during a slash bundler study in 2003 (Rummer and others 2004) indicate that slash bundles contain about 750 to 1,000 kilowatt-hours worth of convertible energy.

An alternative to using the slash bundler technology is to simply remove loose slash from harvest areas on forwarders (fig. 2). Some entrepreneurs are experimenting with making "grapple piles on wheels" that can be gathered in the woods and transported to roadside for chipping or hauled off in loose form in something like a roll-off container (i.e. large dumpster). An advantage to this system over the slash bundler is that if local loggers don't have the money to invest in the bundler, they can make do with this. Disadvantages may include slower production rates and less capability for terrain than the bundler. Probably more cost effective than gathering scattered slash from within harvest units, chipping at landings in combination with whole-tree-yarding is yet another way to utilize biomass.

Perhaps the ultimate way to utilize biomass while managing for climate change objectives is to stop thinking about biomass as a "forest residue" and start thinking about it as an energy resource to be integrated in management activities. We can begin doing this by integrating biomass utilization in young age-class timber stands and plantations to accomplish what in the past has been referred to as "pre-commercial thinning" treatments. The ultimate goal is to decide what tree species, stocking levels, coarse woody debris and other biomass we want to leave in a harvest unit to meet our climate change management prescriptions, and then remove and utilize all the excess wood for the full range of forest products, including biomass for energy.

Figure 2—Loose slash transported from harvest area on a small-scale forwarder.

Case study: Templemental Stewardship contract—The Templemental Stewardship contract "biomass thinning" unit is being conducted in a 300 acre plantation that was established in 1981 after the Templeman Lake fire in 1978. The average tree diameter is 5 1/2" inches dbh and average tree height is about 40 feet (fig. 3, top). This plantation is being thinned to spacings between 12 and 17 feet (variable-density thinning; Designation by Prescription) and

Figure 3—Young plantation (Templemental Stewardship Project, top) is subjected to variable-density thinning using Cat 314 (middle) and Rottne 2002 (bottom) mini-harvesters (shown working recently in private property areas).

requires mandatory removal of trees as small as 1-inch dbh and 4 feet tall, with the exception of leaving coarse down woody debris based on guidelines by Graham and others (1994) and also some material for slash mats. This biomass thinning treatment also includes long-term soil productivity research to be conducted by the Rocky Mountain Research Station (RMRS) and production rate and recovery research on the equipment to be used (mini-harvesters and small-scale forwarders) to be conducted by the Southern Research Station (SRS). Two of the machines to be used in the Templemental Stewardship Project are the Cat 314 and Rottne 2002 "mini-harvesters" (fig. 3). Both the Cat 314 and Rottne 2002 can cut trees as small as 1-inch diameter. The Cat 314 is less than 10 feet wide; the Rottne 2002 is less than 9 feet wide and can reach about 23 feet with the harvester head.

Biomass Utilization to Achieve Resilience in Mature Stands

Improved biomass utilization can be achieved while harvesting mature timber stands using conventional and cut-to-length harvest systems. Goal #2, Adaptive Climate Change strategies, can be achieved through biomass utilization by restoring resiliency in timber stands.

This section provides examples wherein biomass utilization advances Goal #2. Requiring smaller diameter trees, smaller top diameters and shorter minimum pieces to be removed during initial harvest treatments not only improves timber stand resilience, but also increases biomass supplies and could also reduce the footprint of management activities by eliminating the need to have follow-up slash disposal treatments.

Case Study: Deerskin Roundwood Timber Sale Unit 26

This case study of the Deerskin Roundwood timber sale is to illustrate how a prescription for a small diameter timber sale was implemented using Designation by Description (DxD), weight-scale contract and utilization of smallwood material having minimum specs of 5 inch dbh to a 3 inch top on a 16 foot piece. This treatment would probably be considered a "restoration" treatment, promoting a condition that is more resilient to climate change.

The long-term objective for Unit 26 is to develop large diameter western larch and white pine, while maintaining a mix of other species and also maintaining coarse woody debris on-site. A reconnaissance cruise collected current stand composition data within the proposed harvest unit in order to determine the best mix of species and diameters to be left in the residual timber stand. Contract provisions for Designation by Description were then designed to meet the objectives of the silvicultural prescription, while at the same time providing a cost savings for sale preparation of about $50 per acre.

Extremely dense understory (fig. 4) illustrates why DxD is a necessary cost-saving tool for managing these kinds of stands. DxD should be considered the "leatherman tool" of sale preparation foresters, in that there are many ways to use it, but site-specific stands usually require only a combination of 2 or 3 tools (i.e. designations). This stand used two designations, which included a diameter limit cut (understory removal) along with a requirement to maintain a minimum spacing of 15 feet between leave trees if 8-inch trees were available.

Because resource protection measures were built into the contract to protect soils from excessive compaction and disturbance, harvesting was accomplished in a low-impact manner. Soil compaction and disturbance in these types of stands with these types of equipment can be kept to a minimum. An in-woods processor

Figure 4—The dense understories of some restoration treatments make the Designation by Description contract provision approach invaluable (Deerskin Roundwood Timber Sale).

(fig. 5, top) not only sorted small sawlogs and small pulplogs, but also provided a slash mat for operating on along with grapple skidders. The system's grapple skidder (fig. 5, middle) operated on slash mat and elevated most logs completely off the ground, further reducing soil disturbance. Even with utilization set at 5 inch dbh to a 3-inch top on a 16-foot piece, substantial slash was left after grapple piling and leaving the recommended level of coarse woody debris (fig. 5, bottom).

Two More Scenarios

An example is the 2001 Kat Tail 2 timber sale on the Bonners Ferry RD Idaho Panhandle National Forests (IPNF) (fig. 6). The minimum size tree to be cut and removed in this project was 4 inches dbh having a minimum top diameter of 2½ inches on a 13-foot piece. Small trees were removed from the understory, leaving larger overstory trees in a more resilient condition.

Perhaps one of the most challenging opportunities for biomass utilization is being accomplished on the Apache-Sitgreaves NF White Mountain Stewardship project. This project includes a range restoration treatment area designed to reduce the grossly overstocked condition in a pinyon-juniper habitat type, with utilization of the "trees" for biomass-electric energy at the nearby Renegy electric facility near Snowflake, Arizona. Trees were whole-tree harvested, roots and all, and were to be chipped on-site and then transported to the Renegy facility (fig. 7).

Figure 5—Slash and soil management at the Deerskin Roundwood Timber Sale. Slash mat created by in-woods processor (top) cushioned the grapple skidder (middle). Most slash was grapple-piled (bottom) yet sufficient coarse woody debris was retained.

Figure 6—Kit Tail 2 Timber Sale on the Bonners Ferry RD before (top), during (middle), and after thinning of smallwood to promote stand resilience.

Figure 7—Biomass utilization as part of range restoration on the Apache-Sitgreaves NF White Mountain Stewardship Project. Biomass in this project is utilized as feedstocks for the Renegy electric facility.

Conclusion

Silviculturists and ecologists may recommend land management prescriptions that are designed to be resilient to changing climatic conditions. When considering biomass utilization opportunities that may result from climate-change treatments, it really doesn't matter what species mix or stocking levels are to be retained. If there are trees that need to be cut as a result of the prescription, there will usually be opportunities for recovering these as woody biofuels.

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Emissions, Energy Return and Economics From Utilizing Forest Residues for Thermal Energy Compared to Onsite Pile Burning

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Abstract—The emissions from delivering and burning forest treatment residue biomass in a boiler for thermal energy were compared with onsite disposal by pile-burning and using fossil fuels for the equivalent energy. Using biomass for thermal energy reduced carbon dioxide emissions on average by 39 percent and particulate matter emissions by 89 percent for boilers with emission control. Over 21 units of bioenergy were produced for each unit of diesel energy used to collect, grind, and haul biomass. At prices in place at the time of the study, utilizing biomass was economically viable on 49 percent of the study area.

Keywords: biomass energy, bioenergy, carbon emissions, greenhouse gases, logging residues

Introduction

In the western U.S. approximately 16.8 million acres of accessible forestland could benefit from mechanical fuel treatments that reduce hazardous fuels (Rummer and others 2003). Such treatments have the potential to produce significant quantities of forest residue biomass, which includes the tops and limbs from merchantable trees and smaller trees removed by prescription (Barbour and others 2004; Loeffler and others 2006; Perlack and others 2005, Rummer and others 2003). The common practice of disposing of these residues via onsite open burning has drawbacks, however, including negative effects on air quality, potential for escaped fires, and seasonal limits on burning. Open burning also releases atmospheric carbon dioxide and methane, two internationally recognized greenhouse gases and prominent compounds of interest in the global warming literature (IPCC 2007a; US Environmental Protection Agency 2009a). Furthermore, no energy is captured by open burning.

An alternative to onsite, open burning of forest residues is to utilize them instead as feedstock for energy production. Most of the wood-based energy in the US has historically been generated from industrial mill residues (Malmsheimer and others 2008), but there is increasing interest in generating energy directly from forest treatment residue biomass. Additionally, new research is investigating different methods for expanding the use of forest residues as a feedstock for various approaches to energy production. There are a number of potential advantages to utilizing forest residues for energy including: reducing smoke from onsite burning, providing a source of energy for offsetting fossil fuel consumption, promoting new industries in rural economies, and improving the balance sheet for forest fuel reduction and forest restoration treatments by opportunities to add product value.

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Many questions remain regarding the contributions that expanding the use of forest residues for energy can make toward offsetting fossil fuel consumption or for meeting objectives for carbon and particulate matter emissions and sequestration (Tilman and others 2009). Forest residues are often dispersed over forested landscapes, sometimes requiring long haul distances for energy utilization to occur. We contend that the spatial configuration of forest residues will influence their energetic and economic contribution to management or policy goals. In this analysis we consider the following questions: How much fossil fuel is required to harvest, grind, and haul these forest residues from various landscape locations, and how does it compare with the amount of bioenergy that can be produced? What are the net emissions of key greenhouse gases and particulate matter produced by utilizing forest residues from various landscape locations? How do these emissions compare with the common practice of burning these forest residues onsite? Under what conditions is it economically viable to utilize these forest residues?

To address these questions, we considered the case of collecting, grinding, and hauling forest residue biomass from potential treatment units (74,352 acres) spread across a forested 1.3 million-acre landscape in western Montana. We computed the consumption of diesel fuel needed to utilize these forest residues and compared it with the thermal energy that they would produce in a boiler. In addition, the total greenhouse gas and particulate matter emissions from delivering and burning forest residues in a boiler for thermal energy were compared with onsite disposal by pile-burning and then using fossil fuels to produce the equivalent amount of useable energy (fig. 1). We also compared the fossil fuel requirements to use this forest biomass in a boiler for thermal energy with the fossil fuels needed to provide the equivalent heat in a boiler. Finally, we analyzed where biomass utilization is economically viable within the study area for various diesel and delivered biomass prices.

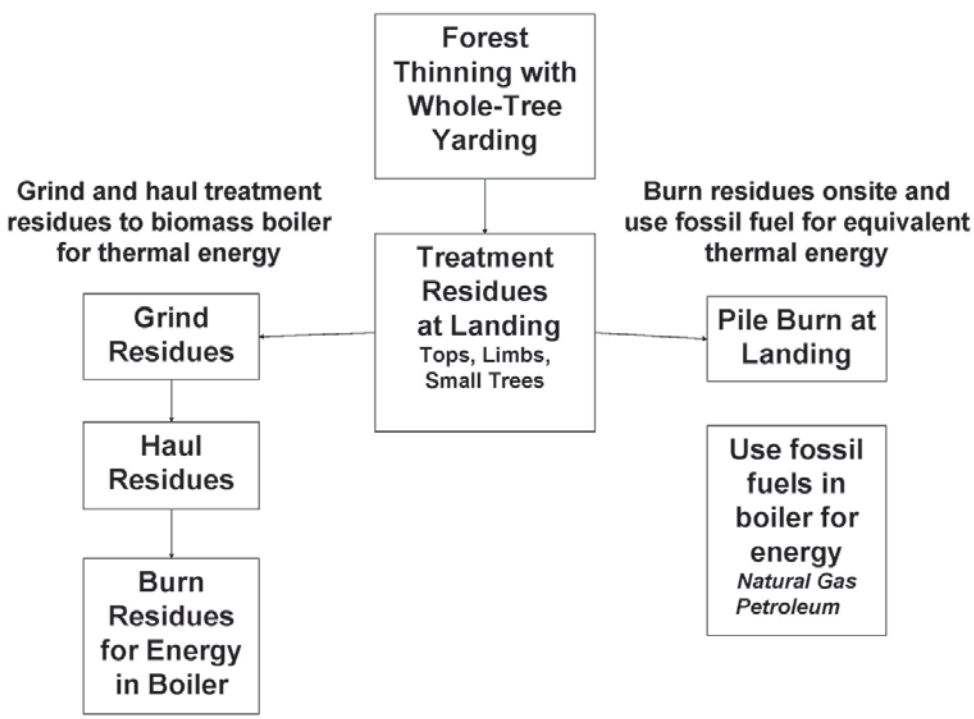

Figure 1—Comparison of burning forest residues in a boiler for thermal energy with onsite disposal by pile-burning and then using fossil fuels to produce the equivalent amount of useable energy.

Methods

Study Area

The study area included the Bitterroot National Forest and adjacent forested lands in the Bitterroot Valley of western Montana, comprising a total of 1.3 million acres (fig. 2). Past fire suppression, together with other factors, has contributed to increased densities of shade-tolerant trees over much of the study area. This forest cohort creates "ladder fuels," which can increase the risk of crown fire and can reduce the growth and vigor of larger trees via competitive stress. Thinning and other density reduction treatments offer ways to accomplish forest and fuels management objectives of reducing fire severity, promoting tree growth, and fostering natural regeneration. We examined two options for disposal of forest residues produced by mechanical fuel treatments, onsite burning and removal for producing energy. Disposal of these forest residues is important to accomplish the treatment objectives of reducing forest fuels to in turn reduce the risk of wildfire.

A GIS-based forest vegetation classification system, R1-VMP (Brewer and others 2004), was used to identify the locations for mechanical fuel treatments within the mapped study area. R1-VMP categorizes polygons based on dominant and co-dominant tree species, stand size class, and stand density as measured by percent canopy cover. The R1-VMP polygons selected as candidates for treatment

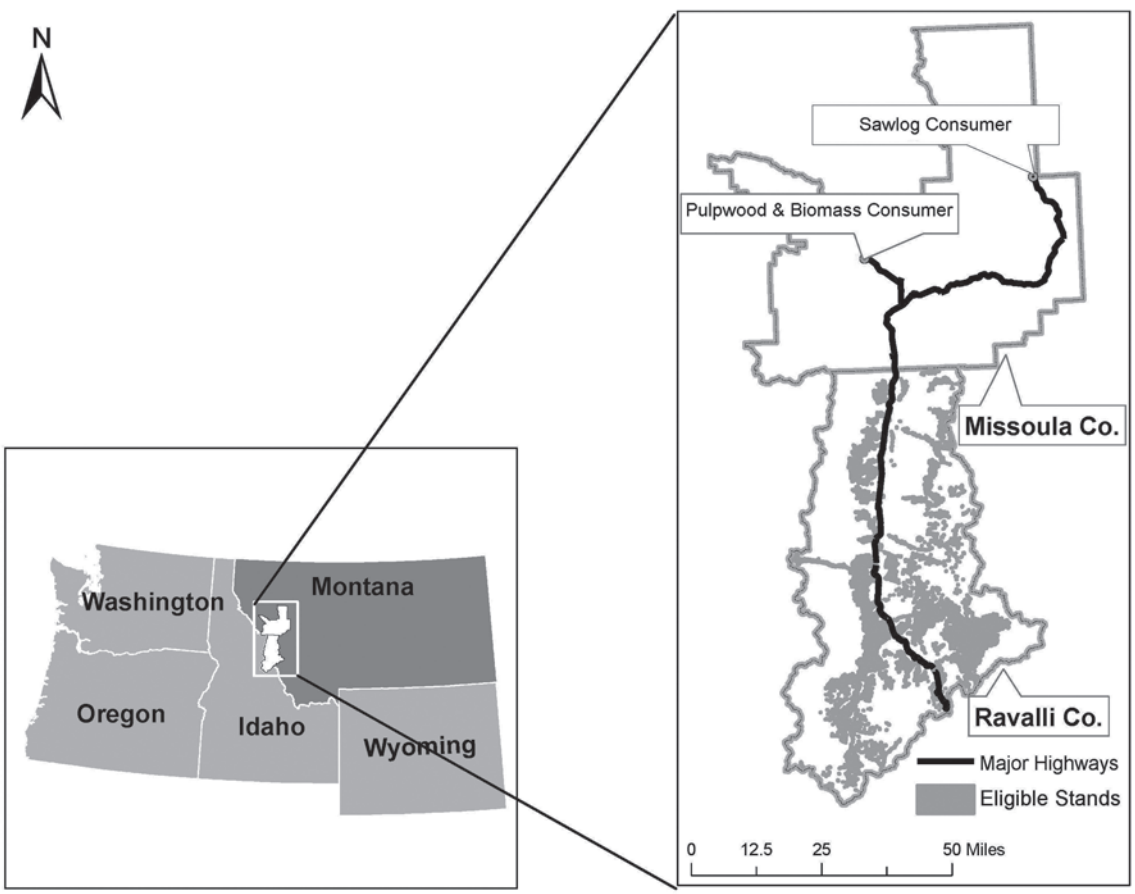

Figure 2—Study area showing treatment polygons and mill locations for consuming sawlogs, pulpwood, and forest treatment residue biomass.

contained species that are associated with low-elevation, frequent low-intensity fire regimes (ponderosa pine [*Pinus ponderosa* C. Lawson] and mixtures of *Pinus ponderosa,* western larch [*Larix occidentalis* Nutt.], and Douglas-fir [*Pseudotsuga menziessi* (Mirb.) Franco] and miscellaneous shade-tolerant species) and fell into fire regime condition classes 2 and 3. Land categorized as condition classes 2 and 3 contain fuel loading that places these forests at the greatest risk of environmental damage from uncharacteristic wildfire (Hardy and others 2001, Schmidt and others 2002). Candidate polygons were further restricted to those with average slopes less than 35 percent, that lie within 1500 feet from polygon center to existing roads, and are classified as Forest Service non-reserved or non-industrial privately owned land. This resulted in 15,800 polygons (average size is five acres) comprising 74,352 acres.

The treatment residues were assumed to go to a wood residue boiler 17 road miles north of the study area boundary that produces electricity and heat for a commercial manufacturing plant. Pulpwood was assumed to go to the same facility and sawlogs to a mill 67 road miles north of the study area boundary. Transportation to these mills is over forest roads and secondary roads that feed into a main highway that exits the north end of the study area.

Modeling Silvicultural Treatments

A variety of silvicultural treatments are available to land managers to achieve differing fuel treatment and/or forest health restoration objectives. For this analysis we focused on a mechanical treatment called "comprehensive restoration" that was designed to reduce ladder and crown fuels, thereby mitigating severe wildfire effects and restoring forests to historical conditions (Fiedler and others 1999). This mechanical treatment removes all trees below seven inches diameter at breast height plus some larger diameter trees with a target residual stand basal area in the range of 40-60 ft^2 per acre comprised of fire resistant tree species such as ponderosa pine and western larch. This treatment is designed to produce an open stand of trees that reduces the potential for crown fire and promotes health of the residual trees by removing competition for moisture and nutrients.

We assumed that whole tree harvesting is used to cut and skid trees to a landing accessible by road. Further, we assumed the tree boles that are suitable for sawlogs and pulpwood are removed and the portion that remains is the residue available for bioenergy. This residue consists of the tops and limbs of the commercial trees, and all of the smaller, noncommercial trees that are skidded to the landing to meet treatment objectives. This green biomass typically has a moisture content around 50 percent and is allowed to air dry to 30 percent moisture content prior to grinding and hauling offsite (Han and others 2008).

Volumes of logs and treatment residues produced by this treatment were estimated using the method described in Loeffler and others (2006). The Northern Idaho/Inland Empire variant of the Forest Vegetation Simulator (FVS, www.fs.fed.us/fmsc/fvs) was used to model the outcome of applying the comprehensive treatment prescription to Forest Inventory and Analysis (FIA, http://www.fs.fed.us/rm/ogden) plot data. To ensure adequate data, we supplemented the FIA plots from within the study area with similar inventory plots from outside the study boundary. Analyzing all plots provided estimates of merchantable timber volumes and non-merchantable biomass volumes that would be removed per acre, assuming that all cut trees are whole tree skidded to the landing (table 1). Quadratic mean diameter (QMD) and trees cut per FIA plot were tallied for both the merchantable and non-merchantable categories. The Fire and Fuels Extension of FVS was used to estimate the weight of the total biomass removed. Subtracting the removed merchantable log weight from the weight of the total biomass removed yielded the

Table 1—Summary statistics from modeling application of the comprehensive restoration treatment on 1-acre plots (n=458).

	Treatment residue biomass	QMD[a] of merchantable[b] trees removed	Number of merchantable trees removed	Volume of merchantable trees removed
	(dry tons)	*(inches)*	*(number)*	*(cubic feet)*
Mean	11.6	10.5	96.0	1,371.1
Median	10.3	10.3	82.5	1,091.7
Standard Deviation	7.3	2.2	64.0	1,162.7
Minimum	.5	7.0	1.9	24.1
Maximum	47.6	24.4	364.0	6,556.3

[a] Quadratic mean diameter.

[b] Merchantable trees are greater than four inches diameter at breast height.

weight of the non-merchantable biomass. Based on the default residue recovery fraction in the Fuel Reduction Cost Simulator (FRCS; Fight and others 2006), we assumed 80 percent of the non-merchantable biomass was skidded to a landing; the remaining 20 percent represented breakage that stays in the treatment unit.

The volumes estimated from analyzing the FIA plots were assigned to the R1-VMP vegetation categories based on dominant species, tree size class, and stand canopy cover. The results from analyzing the plots were averaged within the R1-VMP categories such that each R1-VMP category contained the average tree attributes calculated from the FIA plots in the corresponding category.

Treatment costs (excluding administrative and planning) were modeled for each application of the comprehensive treatment using the FRCS. Required FRCS input variables include trees per acre removed, QMD, average tree volume, green wood weight, and residue weight to bole weight fractions. These were calculated from the FVS-generated cut tree lists (table 1), regression equations from Jenkins and others (2003) and dry wood weights from Reinhardt and Crookston (2003) adjusted to 50 percent wood fiber moisture content. We classified the treatment polygons into three slope categories and assumed an average skidding distance of 1,000 feet. Average skidding distance is approximately 2/3 of the maximum skid distance assuming logs are skidded to a centralized landing for a triangular treatment unit (Matthews 1942). The model was calibrated to reflect western Montana wage rates – $14.72 per hour (ACINET 2008). The model's default labor benefit rate of 35 percent was retained and move-in costs were included.

Mill-delivered prices at the time of the analysis were used to value the products produced by the comprehensive treatment: $28 per ton at 30 percent moisture content for ground biomass, $40 per ton for pulpwood, and $425 per MBF for sawlogs.

Modeling Transportation

A GIS roads coverage obtained from the Bitterroot National Forest (www.fs.fed.us/r1/bitterroot) provided the road network for modeling haul of treatment residue biomass, pulpwood, and sawlogs from the candidate treatment polygons to the respective processing facilities. This GIS coverage contains road segments separated by nodes, which were placed at every road intersection and in the vicinity of candidate treatment polygons. The location where biomass volume from each polygon enters the road system was approximated by choosing the nearest down-slope node.

Many of the treatment polygons are next to roads inaccessible by large chip vans, which are generally considered the most cost-effective way of trucking biomass on paved surfaces. Therefore, we assumed the biomass was hauled from the polygons to the bioenergy facility by hook-lift trucks hauling roll-on/off containers resembling extremely large trash bins (Han and others 2008). These trucks are suitable for low-standard mountain roads and have essentially the same access capabilities as a logging truck. These hook-lift trucks haul one roll-on/off container and pull a pup trailer with a second container, providing a total payload of approximately 25 tons (Thomas, personal communication). This compares with 27 to 30 ton payloads for a chip van. We assumed that the biomass is ground into these roll-off containers at the landings. The hook-lift trucks then pick up the loaded containers and haul them to the biomass utilization facility. Empty containers are returned to the landing on the return trip.

Haul costs were estimated on a per mile basis for each of two types of roads, paved and non-paved, using the Forest Residue Trucking Model (FoRTS; http://www.srs.fs.usda.gov/forestops/). Costs were calibrated to reflect local wages and conditions and various diesel fuel prices. Standard log trucks were assumed for haul of pulpwood and sawlogs. Log trucks were assumed to haul 30 tons of pulpwood and five MBF of sawlogs. The average haul distance from all the potential treatment polygons to the biomass utilization facility was 85 miles (fig. 2).

Fossil Fuel Consumption Associated with Utilization of Forest Biomass

Diesel fuel is used in cutting, skidding, and processing the whole trees at the landing into merchantable logs, for grinding the biomass into the roll-on/off containers, and for hauling the ground biomass to the energy utilization site. Diesel consumption for cutting, skidding, and processing was estimated at 0.022 gallon per cubic foot of harvested timber (CORRIM 2004). We assumed the diesel attributable to biomass removal was proportional to the biomass percentage of the total weight of material delivered to the landing, which based on FVS analysis averaged 25 percent of total weight.

Diesel consumption for grinding into the roll-on/off containers was estimated using the FoRTS model at 0.42 gallon per ton of biomass, which had been allowed to dry in piles to an average 30 percent moisture content. In addition, we used FoRTS to estimate the diesel consumed during a 20 minute idle time for each hook-lift truck and pup trailer to be loaded at 0.21 gallon. The diesel consumption for trucks hauling biomass was estimated at four miles per gallon (Thomas, personal communication). This consumption rate was applied to the loaded haul distance as well as the return trips with empty containers.

Spatial Modeling of Components

MAGIS, a spatial decision support system for scheduling vegetation treatments and road-related activities (www.fs.fed.us/rm/econ/magis) was used to simulate the treatments on the study area. The spatial R1-VMP polygons and road network data, vegetation treatment data, costs, delivered product prices, and fossil fuel consumption data served as inputs in the MAGIS model. MAGIS was then applied to simulate the application of the comprehensive restoration treatment on the relevant polygons on the landscape, load the biomass residue, pulpwood, and sawlogs onto the road network, and route the loaded trucks over the shortest path to their respective mill facility locations. In this process MAGIS calculated the acres receiving treatment, tons of biomass produced by the treatments and either hauled for energy production or burned onsite, the truck-miles required to haul the biomass for energy production, and the diesel consumption involved

in collecting, grinding, and hauling the biomass. The emission factors discussed below were applied to the model results. The scheduling capability in MAGIS was used to analyze applying the comprehensive treatment to incremental portions of the study area having increasing average haul distances.

Emission Factors

This paper focuses on two greenhouse gases, carbon dioxide and methane, as well as particulate matter emissions less than 10 microns in size (PM10). PM10 is one of several measurements of air quality used by the US Environment Protection Agency (2009b). For the alternative of utilizing forest residues in a boiler, we include emissions from internal combustion diesel engines, and stack emissions produced by burning biomass in a boiler for generating electricity and/or thermal energy. For the alternative of onsite disposal of forest residues by pile-burning and using fossil fuels to produce the equivalent amount of useable energy we include the emissions from pile-burning as well as the stack emissions from using either #2 distillate oil or natural gas to produce the equivalent usable energy in a boiler. The pile-burn emission calculations assume 95 percent of the residues in the piles are burned based on the assumption that unburned material at the edge of the piles is manually thrown into the fire (Hardy 1998; Fox, personal communication; Parks, personal communication)

Carbon dioxide, methane, and PM10 emissions for internal combustion diesel engines were estimated using the US Environmental Protection Agency AP-42 report (US Environmental Protection Agency 1995) and data from the US Energy Information Administration (US Energy Information Administration 2008). Stack emissions from burning biomass in a boiler both with and without a wet scrubber were estimated using the AP-42 report and data from the USDA Forest Service Forest Products Lab (USDA Forest Service Forest Products Laboratory 2004). AP-42 factors were also used for the stack emissions from burning either #2 distillate oil or natural gas in a boiler and emission factors for pile-burning the biomass in the forest came from published fuel management data (Hardy and others 2001). We assume boiler efficiency ratings of 83 percent, 80 percent, and 74 percent respectively, for distillate oil, natural gas, and biomass at 30 percent moisture content (USDA Forest Service Forest Products Laboratory 2004) to calculate the amounts of distillate oil and natural gas required in the pile-burn alternatives to produce the equivalent heating value of bioenergy.

A fossil energy ratio factor was incorporated into our estimates of fossil energy used in the alternatives. The fossil energy ratio is the useable fuel energy divided by the total fossil energy inputs required to collect, refine, and deliver the fossil fuel to market (National Renewable Energy Laboratory 1998). The direct consumption of diesel, #2 distillate oil, and natural gas was divided by the fossil energy ratio of 0.8337 (National Renewable Energy Laboratory 1998) to include the fossil fuel energy required to deliver the fossil fuels to the final market as well as the direct usage of fossil fuels in the alternatives analyzed.

Results

Emissions

Figure 3 compares the total carbon dioxide emissions from using forest treatment residues for thermal energy (the bioenergy alternative) with disposal of treatment residues by on-site pile burning and using fossil fuels in a boiler to produce the equivalent amount of usable thermal energy. Carbon dioxide emissions from the bioenergy alternative are only 57 percent of the pile-burn alternative

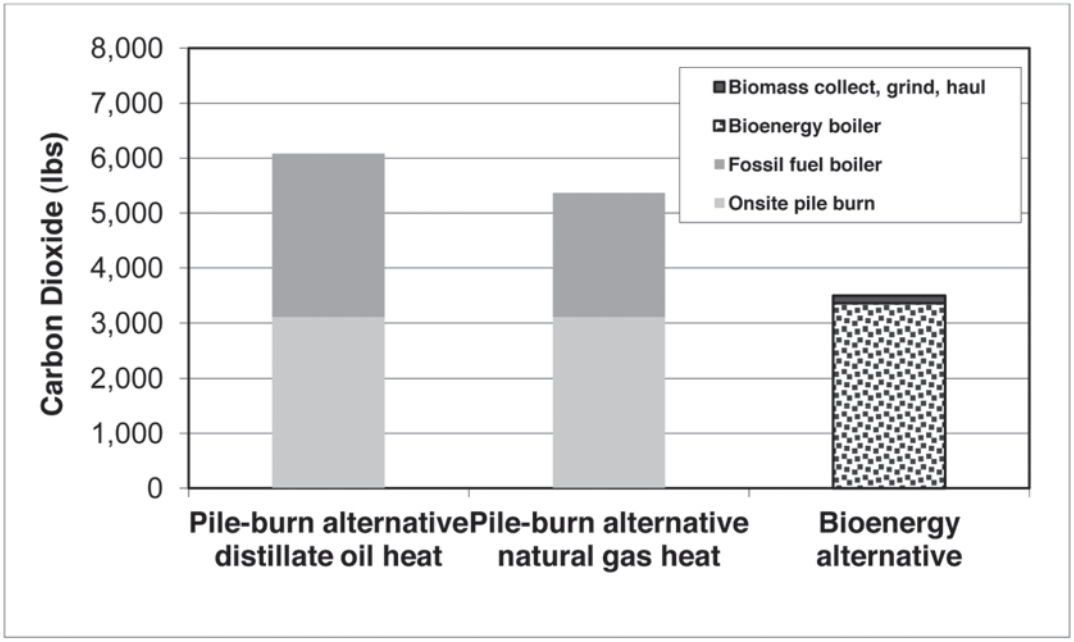

Figure 3—Carbon dioxide emissions per dry ton of forest treatment residues utilized in the bioenergy alternative compared with disposal by on-site pile burning and using either distillate oil or natural gas to provide the equivalent thermal heat in a boiler.

using distillate oil and 65 percent of the pile-burn alternative using natural gas. Notice that the carbon dioxide emissions from the consumption of diesel fuel to collect, grind, and haul the biomass to the boiler facility represents only a very small percentage of the total carbon dioxide emissions associated with using fossil fuels in boilers to provide the equivalent heat in the pile-burn alternatives.

The reductions in methane emissions (fig. 4) are much greater than the reductions calculated for carbon dioxide, with the methane emissions from the bioenergy alternative representing only about 3 percent of the pile-burn alternatives. Methane is not produced in appreciable amounts by burning fossil fuels in a boiler or in diesel engines. The methane production, while small compared to carbon dioxide, is important because the global warming potential of methane is about 21 times that of carbon dioxide (IPCC 2007b, US Environmental Protection Agency 2009c).

For the PM10 comparison, stack emissions were computed for biomass boilers both with and without wet scrubber particulate matter emission control (fig. 5). Although large biomass boilers would be expected to have particulate matter emission controls, we were also interested in comparing emissions from small boilers without these controls. PM10 emissions from the bioenergy alternative with wet scrubber emission control were 11 percent of the pile-burn alternatives, and without the emission control were 44 percent of the pile-burn alternatives. For the pile-burn alternatives, PM10 emissions are almost entirely produced by pile-burning, very little is produced by burning either distillate oil or natural gas in a boiler.

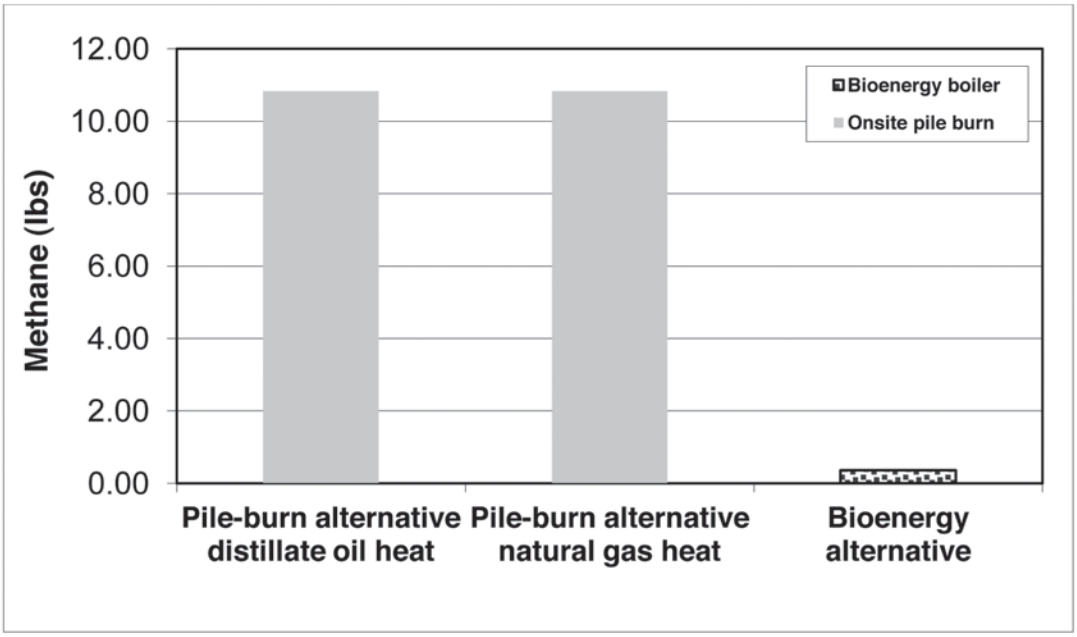

Figure 4—Methane emissions per dry ton of forest treatment residues utilized in the bioenergy alternative compared with disposal by on-site pile burning and using either distillate oil or natural gas to provide the equivalent thermal heat in a boiler.

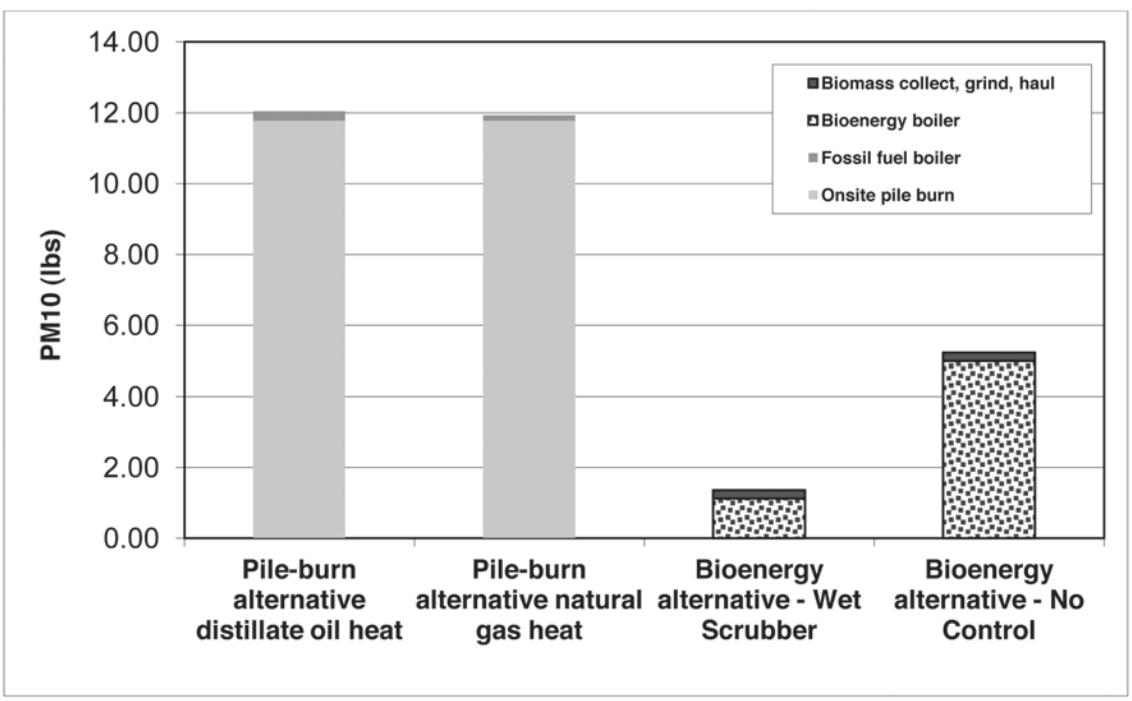

Figure 5—PM10 emissions per dry ton of forest treatment residues utilized in the bioenergy alternatives compared with disposal by on-site pile burning and using either distillate oil or natural gas to provide the equivalent thermal heat in a boiler.

Biomass Energy Returns

The scheduling capability in MAGIS was used to analyze energy returns from delivering forest residues in ten percent increments of total potential residues available on the study area. The first ten percent cost the least to haul to the bioenergy consumer; the second ten percent costs the next least, and so on. Figure 6 shows the average haul distances for each of the ten increments and the units of biomass energy obtained for each unit of diesel energy expended to collect, grind, and haul the biomass over these ten percent increments. As in the boiler emission calculations, the fossil energy ratio of 0.8337 was applied to our estimate of the amount of diesel consumed by these activities to account for the total amount of energy required for a gallon of fuel. At the 47-mile average haul distance, 26 units of energy are obtained for each unit of diesel fuel energy required to deliver the ground biomass to the energy facility. This ratio decreases to 21 units of energy per unit of diesel fuel energy consumed at the 85-mile average haul distance.

These bioenergy returns compare well with other bioenergy alternatives. For example, in a survey of literature of energy return Hammerschlag (2006) reported ratios for corn ethanol energy produced per unit of nonrenewable energy expended ranging from 0.84 to 1.65. For cellulosic ethanol Hammerschlag reported ratios of 6.61 for a mixed feedstock, 4.55 for poplar, 4.40 for corn stover, and 0.69 for switchgrass. Wu and others (2008) estimated year 2030 production cellulosic ethanol energy returns per unit of nonrenewable energy expended at 6.25 for wood residue and 11.11 for corn stover.

The fossil fuel energy consumed to collect, grind, and haul one dry ton of biomass in the bioenergy alternative is on average four percent of the fossil fuel energy required to provide the equivalent usable thermal energy in a boiler. In other words, the fossil fuel energy required in a boiler to provide the equivalent heat in the pile burn alternatives is many times greater than the fossil energy consumed in the bioenergy option.

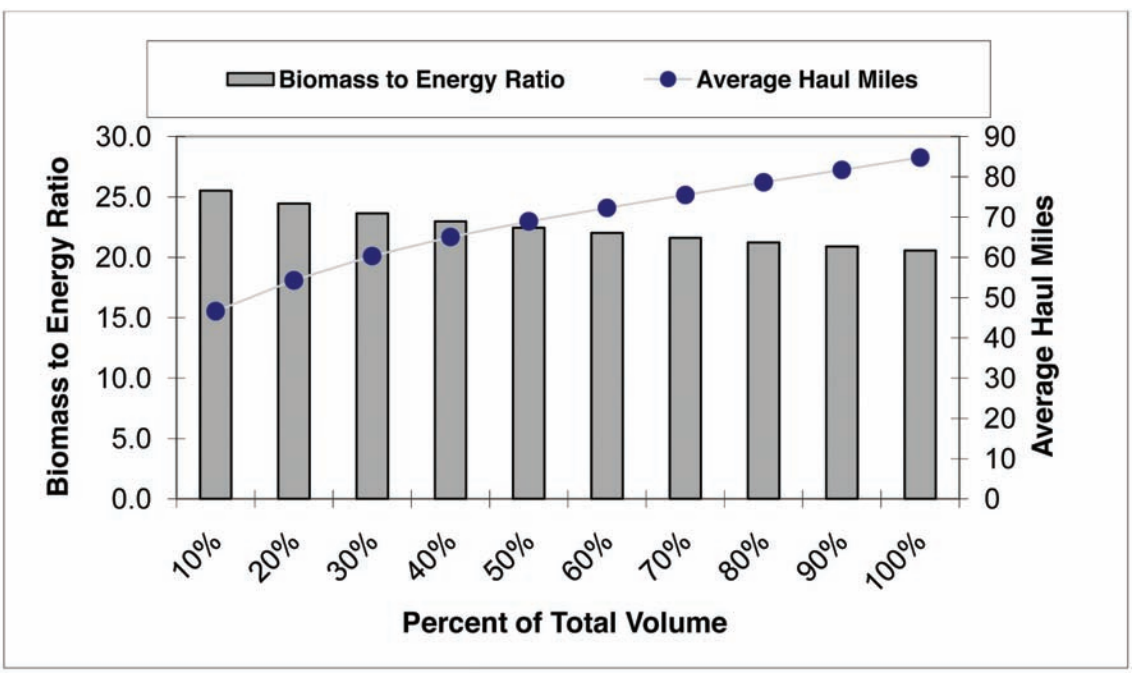

Figure 6—Biomass energy obtained for each unit of diesel fuel used to collect, grind, and haul forest treatment residue biomass across increasing average haul distances.

Economics of Biomass Utilization

Figure 7 identifies the candidate treatment units within the study area where utilization of biomass for thermal energy production is economically feasible. For these units, the delivered value of removed treatment residue biomass is greater than or equal to the cost of handing, grinding, and hauling biomass to that mill location. We assumed whole tree harvesting, so the biomass costs apply to piled treatment residues either at a landing or at road-side.

When diesel price is $4 per gallon and the delivered biomass price is $28 per ton at 30 percent moisture content, 36,447 acres (49 percent of the 74,352 total acres in polygons analyzed for potential treatment) are economically viable (left-most map in fig. 7). If the diesel price were to increase 50 percent to six dollars per gallon (center map in fig. 7), the number of economically viable acres drops to 23,445 (31 percent of the potential acres). The polygons that drop out at this higher diesel price are those with the longer hauling distances and/or more un-paved road hauling distance. If both the diesel and delivered biomass price were to increase 50 percent from the base case (right-most map in fig. 7), then the economically viable acres increases to 37,915 acres (51 percent of the potential acres). This suggests that changes in the delivered price of biomass are slightly more important in economic feasibility than changes in the price of diesel fuel.

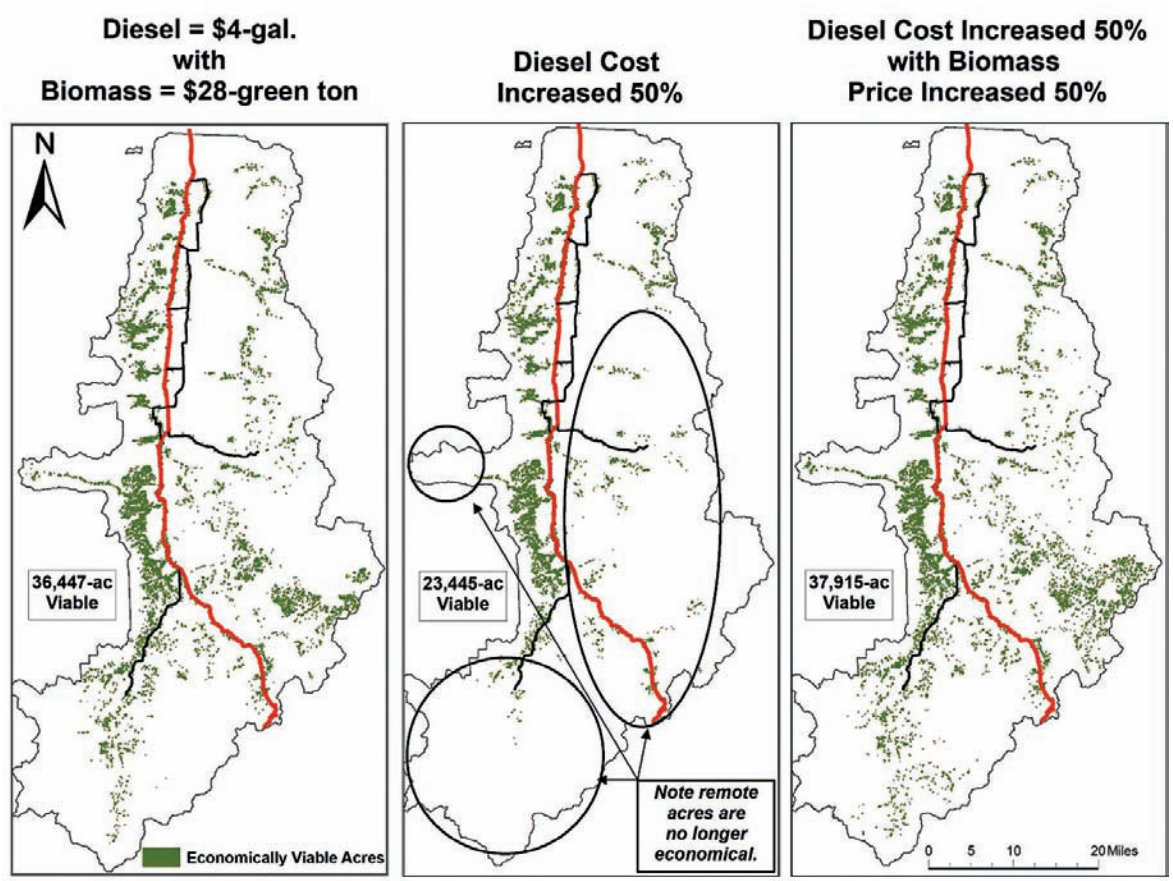

Figure 7—Where biomass utilization is economically viable within the study area across various diesel and delivered biomass prices.

Conclusions

These results suggest that when a bioenergy alternative to onsite pile-burning is available, far fewer carbon dioxide, methane, and particulate matter emissions would be generated and useable energy is produced that could offset the use of fossil fuels for thermal energy production. In addition, the fossil fuel energy required for the bioenergy alternative is small compared to the energy produced in the bioenergy alternative. Based on the economics of biomass utilization results, these relationships hold for haul distances that are many times longer than what are financially feasible.

The analysis we present in this paper is based on whole tree harvesting, grinding the skidded biomass residue into containers, and trucking these containers to the location where the biomass is burned for heat energy. We expect that other wood utilization standard and ground-based harvesting or biomass handling methods would produce different emission trade-offs and energy consumption ratios.

Our results indicate that utilizing woody residues for thermal heat can contribute to generating energy while also reducing greenhouse gas and particulate matter emissions compared to alternative methods of residue disposal. The reduction in particulate matter emissions may also provide an advantage in areas where open burning is restricted by air quality standards.

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The content of this paper reflects the views of the authors, who are responsible for the facts and accuracy of the information presented herein.

Can Portable Pyrolysis Units Make Biomass Utilization Affordable While Using Bio-Char to Enhance Soil Productivity and Sequester Carbon?

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Abstract—We describe a portable pyrolysis system for bioenergy production from forest biomass that minimizes long-distance transport costs and provides for nutrient return and long-term soil carbon storage. The cost for transporting biomass to conversion facilities is a major impediment to utilizing forest biomass. If forest biomass could be converted into bio-oil in the field, it may be more profitable to utilize forest biomass for bioenergy. Bio-oil can substitute for fuel oil, or be used as a crude oil and further refined into additional products. Transporting energy-dense bio-oil is more cost effective than transporting bulky, low-value biomass. In-woods pyrolysis can also address concerns over removing nutrients and carbon from forest sites through reapplication of bio-char, a pyrolysis byproduct, which is equivalent to the charcoal found in all fire ecosystems. Bio-char is 70-80 percent carbon and retains most nutrients contained in biomass. It can be used as a soil amendment to enhance soil productivity through a liming effect, which improves cation exchange capacity and base saturation, increasing anion availability, improving water holding capacity and decreasing bulk density. Charcoal is known to remain stable in soils for hundreds to thousands of years. Long charcoal residence times provide a way to quickly sequester atmospheric carbon by assimilating it into a recalcitrant form that can be applied to soils. In total the portable pyrolysis approach has the potential to improve the economic efficiency of biomass removal from overstocked forests through the in-woods conversion of biomass to bio-oil that avoids the costs and emissions of transportation to central facilities. Bio-char can be returned to the forest economically if pyrolysis occurs at or near the site of biomass removal. Reapplication of bio-char will sequester carbon in soil and may enhance site productivity.

Keywords: bioenergy, bio-oil, carbon sequestration, fuels reduction, soil productivity

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Introduction

Forest biomass accumulation is both a problem and an opportunity. Increasing forest biomass is a consequence of continuous forest growth, effective fire suppression tactics, lack of harvest activities, and other management practices. Young growing forest stands quickly become overstocked with numerous small diameter tree stems, slowing individual tree growth and causing stem exclusion processes to initiate (Oliver and Larson 1990). Prior to implementing effective fire suppression tactics, some fire-adapted ecosystems (i.e. low-elevation, frequent fire regime forests), burned regularly, often as cooler understory fires or moderate severity fires that served to limit biomass accumulation, release nutrients

and improve stand vigor (Agee 1996; Stanturf and others 2002). Pre-commercial thinning is also used to restore ecological function (Thibodeau and others 2000). Without frequent burning or thinning such overstocked forests contain abundant ground level biomass and experience considerable mortality of subordinate trees as dominant stems emerge. This fuel buildup has resulted in high-severity stand replacing fires, which captures the attention of those living in developments at the wild land interface. As a consequence of increasing wildfire occurrence and intensity, public land managers in fire-prone areas have once again begun to thin overstocked stands with a focus on fuels reduction, even though the area actually being treated is small relative to that in need of treatment.

Removed biomass adds to the equally large volume of biomass that is commonly found at landings of logging operations where whole-tree yarding is practiced (Perlack and others 2005). This accumulated biomass from thinning and harvesting practices is typically flared to avoid continued risk of fire as the slash piles dry. Onsite flaring releases greenhouse gases, energy and carbon captured by natural forest processes, and concentrates nutrients at burn pile locations.

Opportunity for Bioenergy

Utilization of biomass offers a potential solution to the problem of hazard fuel accumulation. Developing markets for biomass may provide managers and land owners a way to achieve management objectives if forest operators have a viable opportunity to sell biomass and land managers have the ability to contract for product removal. Potential markets for biomass utilization include products such as small-wood furniture and structures, garden mulch, bioenergy, chemicals, and other products (Hakkila 1989; LeVan-Green and Livingston 2003). Of particular interest at this time is abundant energy contained in biomass that can be tapped as an alternative to fossil fuels and avoid greenhouse gas emissions. Bioenergy production is most attractive when fossil fuel energy prices increase, but as greenhouse gas emissions become an increasing concern it also causes us to look for alternative, renewable and low emissions energy sources. Bioenergy production from forests may meet that need. It is particularly interesting with the coincident occurrence of enhanced energy security needs, requirements to reduce emissions of carbon, and the requirement to remove biomass from forest stands.

Biomass utilization for bioenergy has a long history. Much of the nation's energy needs were met by wood fuel prior to widespread use of coal and petroleum. Even now it is common to find combined heat and power production operations where there are abundant biomass supplies such as in pulp mills and lumber yards. Recent interest has also been spurred by government programs promoting alternatives such as heat for schools, prisons, hospitals, etc. (Richter and others 2009). Even with this level of utilization, there is still over 300 million tons of unused biomass coming available annually nationwide (Perlack and others 2005). However, adoption of bioenergy production practices typically occurs only where there is a ready biomass supply on site, such as forest product facilities, or where modest feedstock requirements are met within close proximity to the energy conversion facility, such as a low-demand educational heating facility in a forested region.

The importance of the biomass supply being localized to minimize transport costs cannot be overstated. While there are significant costs for biomass removal, those costs may be exceeded by revenue gained through the sale of that material to a local conversion facility (Evans 2008). However, delivery to distant conversion facilities frequently causes the delivered cost to exceed revenues making the biomass utilization process economically unviable (Stokes and others 1993).

Consequently, despite abundant supply, biomass is commonly not removed for utilization due to expenditures exceeding potential revenues and instead is cut, piled and burned at significant expense and with important consequences to consider.

Biomass Disposal Concerns

Both off- and on-site consequences occur from pile-burning biomass. Dried biomass is about 50 percent carbon and when biomass slash piles are flared that carbon is oxidized and released back to the atmosphere as carbon dioxide or other organic compounds. Such disposal is questionable in light of efforts to reduce greenhouse gas emissions and sequester carbon. In addition to volatilizing carbon, other essential plant nutrients are also lost from the site by burning. These losses include several processes such as oxidation, vaporization, convective ash losses, leaching and erosion (Fisher and Binkley 2000). The two main inorganic nutrients lost to oxidation are nitrogen and sulfur, which are typically released as air pollutants in the smoke produced by open-air burning. Phosphorous can also be lost, but in lower quantities than nitrogen and sulfur. In hot fires, such as in well seasoned slash piles, oxidative losses of nitrogen can be 25-65 percent and for sulfur they can be 25-90 percent. These nutrients are frequently growth limiting in forest environments (Fox and others 2007; Kishchuk and Brockley 2002), so it is equally unwise to cause such losses rather than conserving onsite stores. Nutrients are also lost from site in smoke emissions. Convective losses of particulates occur during burning that contain the full range of mineral nutrients found in biomass, many of which are concentrated in ash (Fisher and Binkley 2000). Finally, other pollutants including particulates, carbon monoxide, and a variety of volatile aromatic carbon compounds are also released in smoke (U.S. National Research Council 2004). These pollutants are typically regulated in urban and agricultural areas requiring permits to release. Smoke management procedures are also in place to limit forest biomass pile burning to favorable atmospheric conditions (e.g. http://www.smokemu.org/).

Piling and burning slash concentrates nutrients in the fire ring, which may lead to lower average site productivity. The site preparation practice of shearing, piling and burning was discontinued in southern pine plantations after it was recognized that the redistribution of nutrients resulted in productivity declines (Carter and Foster 2006). Similar results were observed in other regions (Binkley 1986), some of which may be explained by topsoil displacement as well as biomass redistribution. Regardless, the concentration of biomass into piles and release of nutrients localizes nutrients and can potentially saturate nutrient exchange capacity in the burned area, leading to greater leaching loss.

While utilization of abundant forest biomass for bioenergy is appealing, it too may result in removal of nutrients from sites. Environmental critics of forest bioenergy production systems frequently cite the concern of nutrient removal and over-exploitation of the resource as an expected negative consequence of biomass harvesting for energy production (Kimmins 1997). We know from timber harvesting that bole-only removal has an undetectable impact on the regrowth of subsequent forest stands; however, if we remove whole trees from nutrient poor sites, impacts on growth of the next forest rotation have been detected (Kimmins 2004). More certainly we know that removing litter and displacing soil will have significant impact on the next rotation (Fleming and others 2006; Van Miegroet and Johnson 2009). But we have little or no information on the impacts of removing small diameter biomass material, such as tops, branches and needles that contain high concentrations of nutrients (Evans 2008; Palviainen and others 2004). We do not know if those removals will impact subsequent forest

productivity, but it will likely depend on the inherent site quality, the frequency and intensity of harvest and the ability of the site to replenish nutrients removed (Kimmins 2004). The forest system is resilient and maintains large stocks of nutrients that, given adequate time, can meet the requirements of forest growth, but an accelerated frequency of removal may exceed the replenishment capacity. Consequently, there is an urgent need to understand the implications of biomass removal. A sustained bioenergy production system might include removing the energy and not the nutrients, or returning the nutrients after energy is extracted from the biomass.

Pyrolytic Biomass Conversion Solution

Both profit and sustainability are essential where financial analysis controls the viability of alternative energy projects and the feedstock derives from venerated forested ecosystems. The mobile fast pyrolysis bioenergy production system (Badger and Fransham 2006) may be one approach to profitable and sustainable biomass utilization. The mobile pyrolysis unit has potential to cover the cost of biomass removal through the production of a crude oil product known as "bio-oil" that has higher density and energy content than biomass. In addition to the bio-oil, there is also a "bio-char" byproduct that has market value of its own, but might best be used by returning it to the site of energy extraction as a soil amendment and as a means of soil carbon sequestration. Such an approach has recently been advocated for agricultural systems (Laird 2008; Lehmann and others 2006), but it makes even greater sense for forest ecosystems when the bio-char is produced at and immediately returned to the site of energy extraction.

Table 1 shows value comparisons for fast-pyrolysis products. The pyrolysis actually has three product phases: gas, liquid and solid (Bridgwater 2004). The flammable gas is used to fuel the pyrolysis process in a self sustaining combustion. So although in some situations the heating value of the gas can be quantified as a product, in this case it provides the energy for producing the other products. The gas amounts to ~48 percent of the energy in dry wood (Raveendran and Ganesh 1996). The bio-oil is the liquid phase product and fast pyrolysis will produce more than 120 gallons per dry ton of biomass (Mohan and others 2006). We determined the value of bio-oil by comparing it to substitute market products. Bio-oil is discounted by 60 percent in this analysis to account for the lower heating value relative to the petroleum products. Minor furnace or boiler modifications are also

Table 1—Value of pyrolysis products from one air dry ton of biomass.

1. Syngas (fuel for Pyrolysis)
2. Bio-oil = 120 gal of bio-oil
   - $64 ($0.89 / gal[1] Bunker Fuel Houston, TX, Bunkerworld.com)
   - $94 ($1.30 / gal[1] Wholesale fuel oil, tonto.eia.doe.gov)
3. Bio-char = 500 lbs of bio-char
   - $65 ($260/ton, author market survey)
   - $9-$18 ($35-$70 / ton[1] EU carbon trading EU ETS, www.pointcarbon.com)

One ton Forest Biomass = $73-$159 (sum of bio-oil and bio-char products

[1] Prices as of 20 April 2009

required in handling and burner/boiler design to allow for unique chemical and physical bio-oil properties (Mohan and others 2006). This comparison gives a value of $64 - $78 of bio-oil produced per dry ton of biomass. The third product of pyrolysis is the solid bio-char and it is similarly valued by substitute market products. Bio-char can be sold for horticulture or barbeque charcoal at a value of ~$65 of bio-char per dry ton of biomass. Although bio-char does have this wholesale market, the real benefit of the bio-char produced from forest biomass using a portable pyrolyis unit might be in leaving it on the site from where the biomass was extracted and using it for soil conditioning and carbon sequestration. As with biomass, the bio-char is a low-density, bulky material (0.35 specific gravity, (Antal and Gronli 2003)) and transport cost may overcome the value and favor leaving it on site. Carbon sequestration might provide a value of $9 and $18 per air dry ton. If ten air dry tons of biomass can be removed from an acre, the potential market value of bio-oil plus bio-char might result in revenue of $730 to $1430 per acre. In comparison to the median cost of biomass removal of $625 per acre (Evans 2008), there appears to be a reasonable potential for profit considering production, relocation, and transport costs must still be accounted.

One of the key features of the mobile pyrolysis approach is the ability to take the conversion unit to the biomass source and avoid biomass transport. In-woods pyrolysis operations allows us to convert biomass into an energy rich high-density bio oil. Transporting a value-added high-density product not only decreases transportation costs, but also decreases fossil fuel emissions required for transport. Therefore, life-cycle analysis is another aspect of the portable vs. centralized pyrolysis plant for which accounting should occur.

The capital and operating costs of small scale conversion units are high relative to larger units (Bridgwater 2004). Greater efficiencies are created by using higher capacity pre-processing and handling equipment: relatively fewer personnel requirements, lower maintenance and greater operating hours per year. For instance, moving the mobile pyrolysis unit into the woods, conducting startup procedures, consuming available biomass, shutting down and relocating may have a significant impact on operating efficiency. It is likely that the portable pyrolysis unit will be located at a single central location within one or more project area(s) and operated at that one location for considerable time, requiring minimal transport of biomass, but still incurring some short-distance biomass transport costs. Consequently, mobile pyrolysis units have both the advantage of limiting transport distance over that of the centralized fixed-location conversion facility and the disadvantage of having greater capital, operating, and relocation costs. Our research is evaluating these operational and economic tradeoffs.

Figure 1 demonstrates the hypothetical operating range of mobile pyrolysis units within the Umpqua and Willamette National Forest woodshed. Biomass from the Umpqua would otherwise be transported to a centralized plant located in Roseberg, OR. The central plant draws from a broader region beyond the indicated National Forests, including surrounding Bureau of Land Management ground as well as other public and private lands in and beyond the area illustrated. Travel routes affect the efficiency with which biomass can be moved to Roseburg and road networks are being used to calculate transportation requirements. Operational efficiency of fixed and mobile pyrolysis units is being evaluated. Capital costs and operational requirements of fixed location units are known through commercial applications (Bridgwater 2004) and are being compared to information from development-stage mobile units.

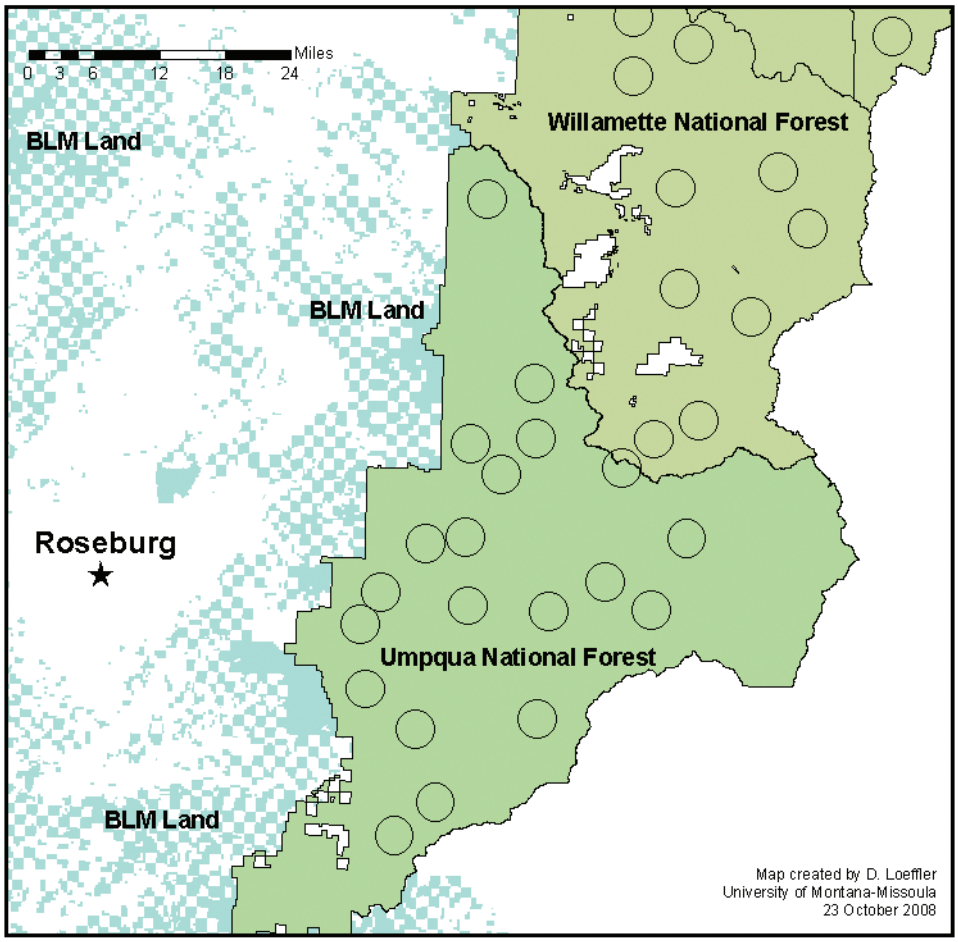

Figure 1—Map of biomass supply area with circles representing portable pyrolyzer supply areas within the Willamette and Umpqua National Forests and a centralized processing facility located in Roseburg, OR with a supply area extending beyond the map area.

Bio-Char Advantage

The bio-char produced through these mobile units is equivalent to charcoal that is manufactured for numerous other purposes through traditional and modern pyrolysis techniques. Charcoal manufacture has been used throughout human history including fuel for iron and bronze metallurgy starting 4,000 years ago and lasting until the use of fossil fuel became widespread during the 19th century (Rackham 1980). Modern charcoal uses include air and water filtration, cooking charcoal, horticultural media, bioremediation, medicinal purposes, among others. As an equivalent to charcoal, bio-char is also an artificially produced analog to charcoal found in many fire ecosystems. This black carbon has been defined as a natural component of fire ecosystems that lends favorable properties to soils and enhances soil productivity (DeLuca and others 2008; Pietikainen and others 2000; Zackrisson and others 1996). Therefore, it can be applied to native ecosystems without concerns of contamination. Bio-char presents an opportunity to return nutrients removed in the biomass from project locations, and as mentioned above, reapplication of bio-char to project sites also has potential value in carbon

sequestration. Both the nutrient return and carbon sequestration values of bio-char reapplication to project sites may outweigh other potential uses. Segments of the public are increasing demands for limits on forest product utilization from public land, which may prompt requirements for nutrient conservation. Geopolitical decisions are expected to expand limits on carbon emissions and reward carbon sequestration. On-site retention may be the best option in light of these social pressures.

Charcoal also has important horticultural values and soil enhancement characteristics. It can be used in greenhouses as a plant growth media. Figure 2 compares poplar trees growing in potting soil blends with increasing bio-char proportions. Poplar was used as a bioassay because of its responsiveness to variable growing conditions and sensitivity to soil growth media. In this case, each is growing equally well regardless of the amount of char included. Bio-char can be used as an effective soil media in the greenhouse and at forest sites because of the favorable properties provided to the soil.

Bio-char contains the majority of nutrients found in biomass feedstock (Gaskin and others 2008). Nutrients such as nitrogen and sulfur can be volatilized during the pyrolysis process, but the bio-char produced may also contain significant amounts of these nutrients. This means that the bio-char resulting from extracting energy in bio-oil production can be returned to the site to replenish soil nutrient stocks.

Returning the bio-char to the site can also enhance soil organic matter. Bio-char is mainly carbon held in aromatic form, which results in it being inert when added as an amendment. As a consequence, it quickly builds the recalcitrant soil carbon fraction of soil. We know from research on wildfire occurrence and the development of anthrosols that charcoal-derived carbon can remain in the soil for hundreds to thousands of years (Agee 1996; Lehmann and Rondon 2006). Enhancement of the soil organic matter pool with charcoal provides the numerous benefits of other organic matter including large surface area for exchange of water and nutrients; however bio-char also has other characteristics that create additional soil improvements.

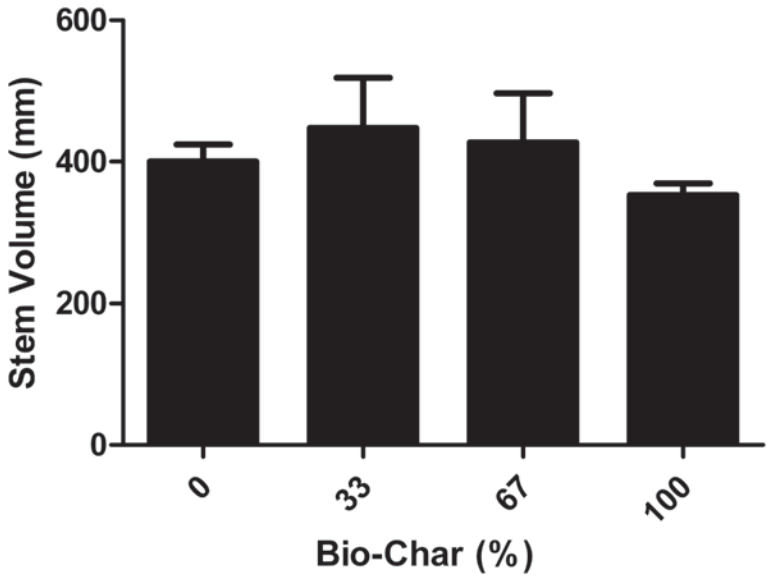

Figure 2—Poplar trees growing for 12 weeks in potting soil with different proportions of bio-char. Each pot received 1.5 g slow release fertilizer (18-6-12). Differences between treatment were not significant ($P = 0.63$). Error bars are standard errors.

Bio-char acts as a liming agent resulting in increased soil pH and nutrient availability for a number of different soil types (Glaser and others 2002; Lehmann and Rondon 2006). Soil liming results in pH increases of one-half to one pH units. The liming of acidic soils decreases Al saturation, while increasing cation exchange capacity and base saturation. These responses following bio-char additions are common soils responses to lime additions (Tisdale and Nelson 1975) indicating that the effects of bio-char are similar to those of other liming agents. Nutrient availability may actually increase beyond the amount expected by cation exchange sites due to soluble salts available in the char. Anion availability may also increase suggesting that anion exchange may be enhanced by bio-char additions to soils (Glaser and others 2002). Microbial biomass and diversity is also known to increase with greater bio-char including more abundant mycorrhizal associations and enhanced biological nitrogen fixation (Lehmann and Rondon 2006). Therefore, when bio-char is added to soil it "sweetens" the soil by raising the pH, improving the fertility level through additions of nutrient ions commonly associated with ash additions, and enhances symbiotic soil microbe populations.

Bio-char may also increase the water holding capacity of forest soils. This is especially important on western soils where the growing season is determined by the length of time into seasonal summer droughts where soil moisture remains favorable to growth. It may become more important in other forest ecosystems where extended summer drought can significantly decrease growth and the frequency and amount of summer rain events are expected to decrease with predicted climate change. Improved water holding capacity through char additions is most commonly observed in coarse textured or sandy soils (Gaskin and others 2007; Glaser and others 2002). Just as increased surface area improves water holding capacity of ash deposits (Dahlgren and others 2004; McDaniel and Wilson 2007), the impact of bio-char additions on moisture content may be due to increased surface area relative to that found in coarse textured soils (Glaser and others 2002).

The residence time of bio-char in soils may be in excess of 1000 years making it a potential tool for carbon sequestration. Bio-char consists of highly aromatic organic material having carbon concentrations of 70 to 80 percent (Lehmann and others 2006), and it is highly resistant to decay by common soil saprophytes. Evidence for the residence time of bio-char comes from several lines of research. Fire ecology typically makes use of the long residence times of bio-char in dating fire events through the latest interglacial period (Agee 1996). Archeologists similarly have demonstrated the use of coppiced woodlands for prehistoric metallurgy by dating the charcoal remains of historic operations back some four millennium (Rackham 1980). Furthermore, the rich Terra Preta soils produced through charcoal additions by a poorly understood Amazonian society occur in a matrix of highly weathered tropical Oxisols (Mann 2008). These soils were developed over 2000 years ago as the agricultural basis of this sophisticated society and are still regarded today as high quality top-soils with charcoal as the vital component (Glaser and others 2001).

The potential to sequester carbon by char additions to soils creates an important possibility to mitigate greenhouse gas emissions. This idea is not new (Seifritz 1993), but has recently gained interest with greater public awareness of the effect of greenhouse gas emissions on climate. The portable pyrolysis units at scattered locations throughout the forest may create greater opportunity to sequester carbon than pyrolysis conversion at a centralized plant. For large fixed-location pyrolysis plants, the economic incentive to return bio-char back to the woods is low because of high transport costs and alternative uses for filtration, clean energy, cooking, horticulture, etc. Furthermore, biomass moved from the woods is just as likely to be used by any number of other processes in addition to pyrolysis including fueling industrial boilers where char would not be a significant byproduct. From

a forest management perspective, the preferred use for bio-char may not be for transport to alternative use markets, but as an on-site soil amendment. Bio-char reapplication represents the middle ground that might make biomass utilization a reality.

Conclusion

The portable pyrolysis system offers a solution to biomass accumulation in forest ecosystems. By utilizing the abundant forest biomass that is annually produced through forest harvest residues and hazard fuel reduction projects it may be possible to produce a liquid fuel that will reduce dependence on foreign energy sources. If biomass conversion can occur in the woods it will improve the economic and environmental impact of biomass utilization for energy production. In addition, the bio-char byproduct can be redistributed to the site of energy extraction and thereby return nutrients to the site to maintain site quality. The additional properties of char additions, including liming, microbial enhancement and improved water holding capacity, create the opportunity to maintain or improve soil quality. Furthermore, bio-char's recalcitrance can sequester carbon for centuries. Such an approach is advocated for agricultural systems (Laird 2008; Lehmann and others 2006), but the arguments are even stronger for portable pyrolysis units used in forestry systems where long distances make onsite reapplication a better option than long-distance transport of biomass to and return of char from a centralized processing facility.

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The content of this paper reflects the views of the authors, who are responsible for the facts and accuracy of the information presented herein.

Ethanol Production from Woody Biomass: Silvicultural Opportunities for Suppressed Western Conifers

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Abstract—The 2007 Energy Security and Independence Act (ESIA) requires 16 billion gallons of ethanol to be produced from lignocellulose biomass by 2022 in the United States. Forests can be a key source of renewable lignocellulose for ethanol production if cost and conversion efficiency barriers can be overcome. We explored opportunities for using woody biomass from thinning western conifers as source materials for conversion to biofuel. We present preliminary results using suppressed lodgepole (*Pinus contorta* Douglas ex Louden var. *latifolia* Engelm. ex S. Watson) and ponderosa pine (*Pinus ponderosa* C. Lawson var. *ponderosa)* from Pringle Falls Experimental Forest in central Oregon. We first examined growth rates of suppressed and presumably unsuppressed lodgepole pine and ponderosa pine planned for removal during thinning operations, and determined that all sampled trees were equally suppressed. We found component polysaccharides in relatively high concentrations among all sample trees. Finally, we used a new sulfite pretreatment technique for biochemical conversion of lignocellulose in wood chips from our sampled lodgepole pine and ponderosa pine to ethanol, and discuss the efficacy of the sulfite pretreatment in terms of dissolved carbohydrate composition, hydrolysis reaction rates, and sugar yield after hydrolysis. Using biomass from forest thinning to make biofuel may help mitigate the cost of fuel reduction treatments and potentially offset the cost of sustaining healthy forests and reducing the risk of catastrophic fires.

Introduction

In the United States, grain-ethanol, a biofuel, is mainly produced from the starch in kernels of field corn. Ethanol production from starches is limited by supplies of agricultural crops and costs associated with production. Currently, grain ethanol supplies 3 percent of US demand for fuel (about 4.5 billion gallons of ethanol produced annually and about 140 billion gallons of fuel used for ground transportation annually) (Somerville 2006). Using lignocellulose as biomass to produce ethanol could contribute to meeting US transportation fuel demands and help the US to achieve its goal of energy independence. Forests can be a key source of renewable biomass feedstock for ethanol production if cost and conversion efficiency barriers can be overcome. Using woody biomass in the form of small diameter trees removed during thinning projects as source materials for biofuel may mitigate the cost of thinning treatments and potentially offset the cost of sustaining healthy forests and reducing the risk of catastrophic fires.

The biochemical conversion of lignocellulose to ethanol typically consists of four major unit operations: pretreatment/size reduction, enzymatic hydrolysis, fermentation, and product separation/purification. Most pretreatment processes are designed to alter the structure of biomass cellulose by increasing the accessibility of cellulose to enzymes that convert the carbohydrate polymers into

In: Jain, Theresa B.; Graham, Russell T.; and Sandquist, Jonathan, tech. eds. 2010. Integrated management of carbon sequestration and biomass utilization opportunities in a changing climate: Proceedings of the 2009 National Silviculture Workshop; 2009 June 15-18; Boise, ID. Proceedings RMRS-P-61. Fort Collins, CO: U.S. Department of Agriculture, Forest Service, Rocky Mountain Research Station. 351 p.

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fermentable sugars (Lynd 1996). The pretreatment stage can account for as much as 30 percent of the total production cost in the conversion of cellulosic biomass to ethanol (Aden and others 2002). Despite extensive research and development, limited progress has been achieved in the economic development of an effective pretreatment method for woody biomass, especially conifer species.

A novel pretreatment process under development at the Forest Products Laboratory has the capability of removing woody biomass recalcitrance (the natural resistance of plant cell walls to decomposition from microbes and enzymes) for subsequent hydrolysis to glucose (Zhu and others 2009a). This process utilizes a Sulfite Pretreatment to Overcome the Recalcitrance of Lignocellulose (SPORL). The SPORL pretreatment can be applied directly to wood chips in a digester providing a low liquid to wood ratio of 3 or less in aqueous pretreatment to significantly reduce thermal energy requirements (Zhu and others 2009b). After pretreatment, size reduction by disk milling results in a coarse wood pulp at greatly reduced specific energy levels (~50 W-hr/kg) (Zhu and others 2009b). Furthermore, the resulting wood substrate has excellent digestibility in subsequent enzymatic hydrolysis. This SPORL pretreatment, however, has yet to be evaluated with western conifers.

Recent work at the Forest Products Laboratory indicated that red pine (*Pinus resinosa* Aiton), grown at high stand density, produced wood with higher concentrations of glucan and xylan (both polysaccharides) compared to wood produced by red pine grown at more open stand densities. This suggests that trees grown under suppressed conditions, especially small diameter trees in dense, even-aged stands, may be useful for biofuel production. In this paper, we expand this work to further consider two conifers, lodgepole pine (*Pinus contorta* Douglas ex Louden var. *latifolia* Engelm. ex S. Watson) and ponderosa pine (*Pinus ponderosa* C. Lawson var. *ponderosa).* Together these species represent a large proportion of the timber targeted for thinning in overstocked stands throughout the western US.

The overall goal of this study was to evaluate the potential of western conifers, particularly trees targeted for thinning, for biochemical conversion to biofuel. We used the SPORL pretreatment process, disk milling, and enzymatic hydrolysis to determine the effective yield of glucose from wood substrate samples. Specific objectives of the study were:

1. Quantify the growth rate of lodgepole pine and ponderosa pine from areas selected for thinning.
2. Relate the growth rate to the chemical composition of lodgepole pine and ponderosa pine.
3. Quantify the efficacy of the SPORL pretreatment process when combined with enzymatic hydrolysis for lodgepole pine and ponderosa pine chips.

In this paper, we introduce the study area, and present preliminary results in quantifying growth rates, relating the growth rates to chemical composition, and quantifying cellulose conversion to glucose through enzymatic hydrolysis.

Methods

Study Area

Wood was obtained from Pringle Falls Experimental Forest (lat. 43°42’ N, long. 121°37’ W). Pringle Falls Experimental Forest lies within the Deschutes National Forest in central Oregon about 48 kilometers southwest of Bend, Oregon, and was established in May 1931 as a center for silviculture, forest management, and insect and disease research in ponderosa pine forests east of the Oregon Cascade

Range. The entire experimental forest is addressed in the Deschutes National Forest Management Plan as a single management area, with Forest-level standards and guides applied when the management activities do not conflict with research.

The 1430 ha Lookout Mountain Unit of the experimental forest was added to the Pringle Butte Unit in 1936. The Lookout Mountain Unit lies on an ancient shield volcano with an upper elevation of 1900 m above a relatively flat ancient lakebed surface at 1300 m. Currently, this relatively large block of closed-canopy forest has undergone little major disturbance since stand-replacement fires occurred in 1845 and 1890, resulting in the establishment of dense lodgepole pine and ponderosa pine at lower elevations and Douglas-fir (*Pseudotsuga menziesii* (Mirb.) Franco), grand fir (*Abies grandis* Douglas ex D. Don) Lindl.), white fir (*Abies concolor* (Gord. & Glend.) Lindl. ex Hildebr.), sugar pine (*Pinus lambertiana* Douglas), western white pine (*Pinus monticola* Douglas ex D. Don), and mountain hemlock (*Tsuga mertensiana* (Bong.) Carrière) at higher elevations. These stands may represent some of the most productive ponderosa pine sites in central Oregon. Limited low thinning occurred in 1969 when the road network was established. Lookout Mountain is the site of limited past research, but several silvicultural studies continue as long-term projects with regional and national significance, such as the Lookout Mountain installation of the Levels-of-Growing Stock study (Oliver 2005). Compared to structural characteristics of nearby old-growth stands (at least 600 years old) (Youngblood and others 2004), ponderosa pine in the Lookout Mountain stands are relatively young, have grown exceptionally well but have declined in radial growth over the past several decades, and currently have structural characteristics that place them at imminent risk of catastrophic loss to wildfire or bark beetles. Thinning and fuel reduction treatments are planned for 1413 ha of Lookout Mountain to restore tree vigor, reduce susceptibility to insect infestation, and reduce the risk of stand-replacement wildfire (Youngblood 2009). Wood for this study came from trees targeted for removal during thinning treatments. The study area is similar in environmental characteristics and stand structures to other low elevation ponderosa pine forest-dominated landscapes throughout the pumice zone of eastern Washington, central Oregon, and northern California.

Field Methods

Lodgepole pine and ponderosa pine wood samples were collected from two stands near the lower elevation of Lookout Mountain (1370 meters above sea level). In each case, an initial assessment was conducted to determine the diameter distribution of live trees. The plant community at both sample sites is best described as belonging to the ponderosa pine/bitterbrush-snowbrush/sedge plant association (CPS312) (Volland 1985). Soils are well-drained, relatively undeveloped, and derived from aerial deposits of pumice or scoria flow that exceed 2 m in depth after the explosion of ancient Mount Mazama. Surface horizons are loamy coarse sand to fine sandy loam. The closest weather station is 40 m lower in elevation at Wickiup Dam, OR (latitude 43°41'N, longitude 121°41'W), about 11 km to the southwest, with continuous records since 1941. The mean annual temperature at Wickiup Dam is 6.3 °C, and the mean annual precipitation totals 534 mm, mostly as snow during winter or rain during early summer convection storms.

Measurements of stand structure were conducted in circular 0.04-ha plots, with two plots at each sampling site. Trees selected for growth analysis and wood chemistry sampling were between about 10 and 36 cm diameter at breast height (1.37 m), were representative of the lower 75 percent of the stand diameter distribution, had low crown ratios and low to mid canopy positions, and were presumed to be suppressed. In addition, a single tree of both species with large

diameter, high crown ratio, and dominant or codominant crown position was selected to represent relatively unsuppressed growth. A total of 13 lodgepole pines and 16 ponderosa pines were felled by hand using Deschutes National Forest fire crews during the last half of June, 2008. Trees were delimbed, total bole length measured, and then the bole was bucked into sections. From each tree, a 10-cm thick cross-sectional disk was cut from immediately below breast height, and a 71-cm short bolt was cut from below this cross-sectional disk. Additional 71-cm short bolts with minimum small-end diameter greater than 10 cm were cut at 6.1, 12.2, 18.3, and 24.4 m on all trees. All short bolts and cross-sectional disks were marked with numbered metal tags for identification. All short bolts were wrapped in plastic to minimize drying, and were shipped to the Forest Products Laboratory in Madison, WI, for analysis.

Stand Structure and Tree Growth

Stand structure was characterized by basal area, stem density, diameter class distribution, stand density index, and tree height. Basal area was computed separately by live and dead trees for each species as a summation of the cross-sectional areas of all trees equal to or greater than 1.37 m in height. Trees were grouped into 10 cm diameter size classes to give size-frequency distributions for each species. Stand density index (SDI) is a relative density measure based on the relationship between mean tree size and number of trees per unit area in a stand (Reineke 1933). This statistic has proved useful for quantifying relative density across a wide variety of stand conditions because it is independent of site quality and stand age (Long and Daniel 1990). We used the individual tree summation approach rather than the more easily applied but biased approach based on uniform diameter classes (Woodall and others 2003). Because the diameter distribution was unknown or was not normal, SDI was calculated as a summation of individual tree values as:

$$SDI = \sum (T/A)(dbh_i/10)^b,$$

where SDI is stand density index, T/A is trees per acre, dbh_i is diameter in inches at breast height of the i_{th} tree in the plot, and the exponent b is a species-specific value (Shaw 2000). Values of the exponent b were 1.77 for ponderosa pine and 1.74 for lodgepole pine (Cochran and others 1994). For comparison, SDI at full stocking is 365 for ponderosa pine and 277 for lodgepole pine (Cochran and others 1994).

The cross-sectional disks cut at breast height were sanded with increasingly fine grit sandpaper (#150, 220, and 320) to clarify ring structure, and annual ring widths measured by two people under a binocular microscope by using a Velmex measuring stage with an AccuRite linear encoder (resolution 0.001 mm) with a Metronics digital readout unit coupled to a computer with MEASURE J2X software (version 4.1.2). On each cross-sectional disk, ring widths along the largest radii and the radii opposite were measured. Crossdating and measurement quality control were completed for each radii by using the computer program COFECHA version 6.06 (Holmes 1986) (accessed at the Tree-Ring Lab, Lamont-Doherty Earth Observatory, Columbia University). Undated ring width series were correlated with a master dated series (Lookout Mountain Lower, OR046; downloaded from the International Tree-ring Data Bank, World Data Center for Paleoclimatology) collected in 1995 from an adjacent ponderosa pine old-growth stand for identifying pandora moth outbreak frequency (Speer and others 2000) with emphasis on the most recent century. For example, the OR046 chronology documented exceptionally low radial growth in 1930 and 1995 associated with insect herbivory, and also low radial growth in 1977.

Trees growing at a constant rate under uniform environmental conditions will increase in diameter by producing increasingly smaller growth rings as

the circumference of the bole increases, and this pattern of ring widths is easily modeled as a negative exponential growth curve. All tree ring width series were standardized in the R statistical programming environment (dplR) (Bunn 2008) by first fitting each ring width series to the model:

$$G_t = ae^{-bt} + k,$$

where the growth trend G_t is estimated as a function of time t with coefficients a, b, and k. Annual values from each ring width series were then divided by the corresponding value of the growth trend G_t. The resulting ring width indices have no residual age effect, a mean of 1.0, and equal variances. Finally, ring-width indices were averaged across trees from each site to build a common chronology. Deviations above or below this standardized index represent periods when growth was influenced by exogenous factors including climate, insect herbivory, and density-influenced suppression. While climate and insect herbivory may cause short-term changes in radial growth, suppression is more chronic. Ring width indices were examined for periods when growth indices were ≤ 0.9 for at least 5 years. Descriptive statistics were used to compare suppressed and reputed unsuppressed chronologies. These statistics include the mean ring width, mean sensitivity (MS), the first-order serial autocorrelation (AC) to detect eventual persistence retained after the standardization, the mean correlation between trees ($\bar{r}_{wt}$) and within trees ($\bar{r}_{wt}$), and the expressed population signal (EPS) to estimate the amount of year-to-year growth variations shared among trees of the same chronology (Briffa and Jones 1990; Wigley and others 1984). The EPS is based on the mean correlation between all tree-ring series included in the chronology and increases from zero to one with sample size and with the strength of the mean correlation.

Growth Analysis and Wood Chemistry

Thin disk samples were cut from each short bolt, oven dried, and sanded to expose the ring structure. A representative pith-to-bark wedge sample with companion strip (fig. 1) was then split from each disk for analysis of chemical composition and growth characteristics, respectively. Strips were then progressively sanded with 600-, 800-, 1000-, 1200-, and 1500-grit sandpaper and polished on clean copier paper. To determine the growth characteristics of each strip, portions were viewed under a stereo-microscope at various magnifications to measure rings, and in some cases, individual tracheids. Once the growth characteristics were known, the wedge samples were split into sections of relatively uniform growth by first scoring the row of earlywood cells with a razor blade at the beginning of the section of interest and then fracturing the section from the remainder of the wedge (fig. 2). This technique resulted in a very clean fracture between the latewood band of the previous section and the earlywood band of the next section. The fractured sections were then chipped into smaller fragments with a sharp chisel and chopped into 20-mesh particles in a Wiley Mill in preparation for chemical analysis. The samples were first disintegrated in concentrated sulfuric acid and then centrifuged to produce an aliquot that could be injected into a Dionex ion chromatograph with pulsed amperometric detection to determine polysaccharide composition (Davis 1998).

To date, analysis of growth and chemical composition has been conducted on only a subset of short bolts selected to represent extreme differences between trees. For lodgepole pine, three short bolts with the smallest breast-height diameters were analyzed in addition to the reputed “unsuppressed” tree. For ponderosa pine, three samples with the smallest breast-height diameters, one additional sample with larger diameter yet a discrete and extreme period of suppressed growth, and the short bolt from the reputed “unsuppressed” tree were analyzed.

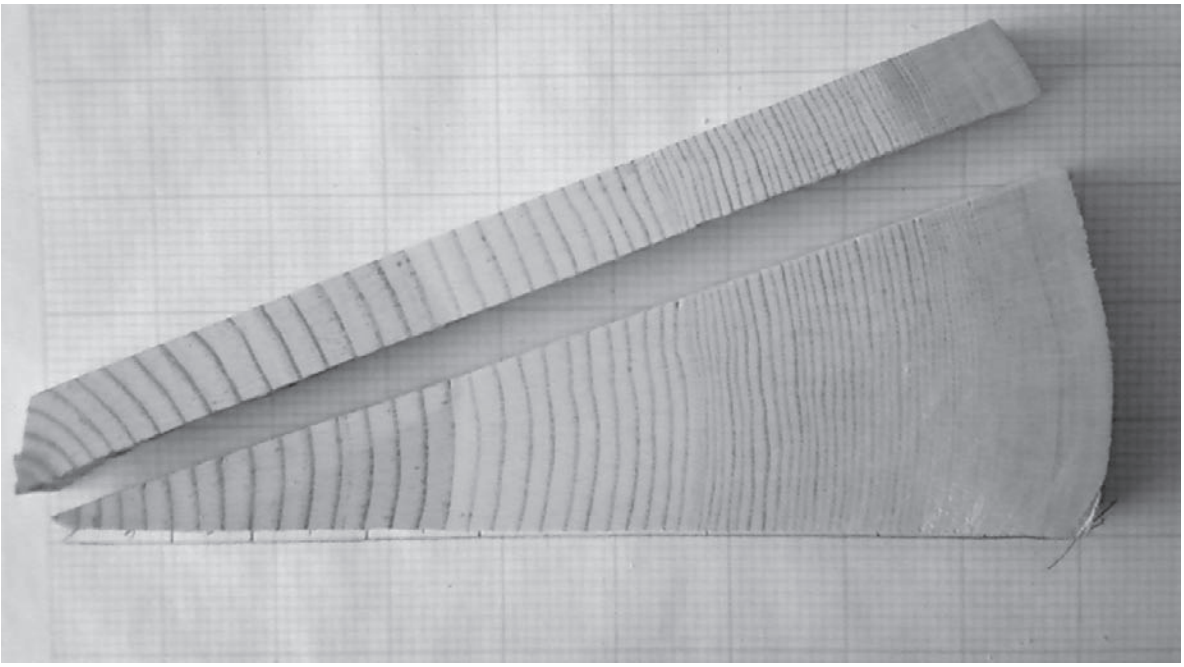

Figure 1—Breast height wedge section with companion strip removed from the reputed unsuppressed tree in a ponderosa-dominated stand sampled for biofuel potential at Lookout Mountain, Pringle Falls Experimental Forest, central Oregon.

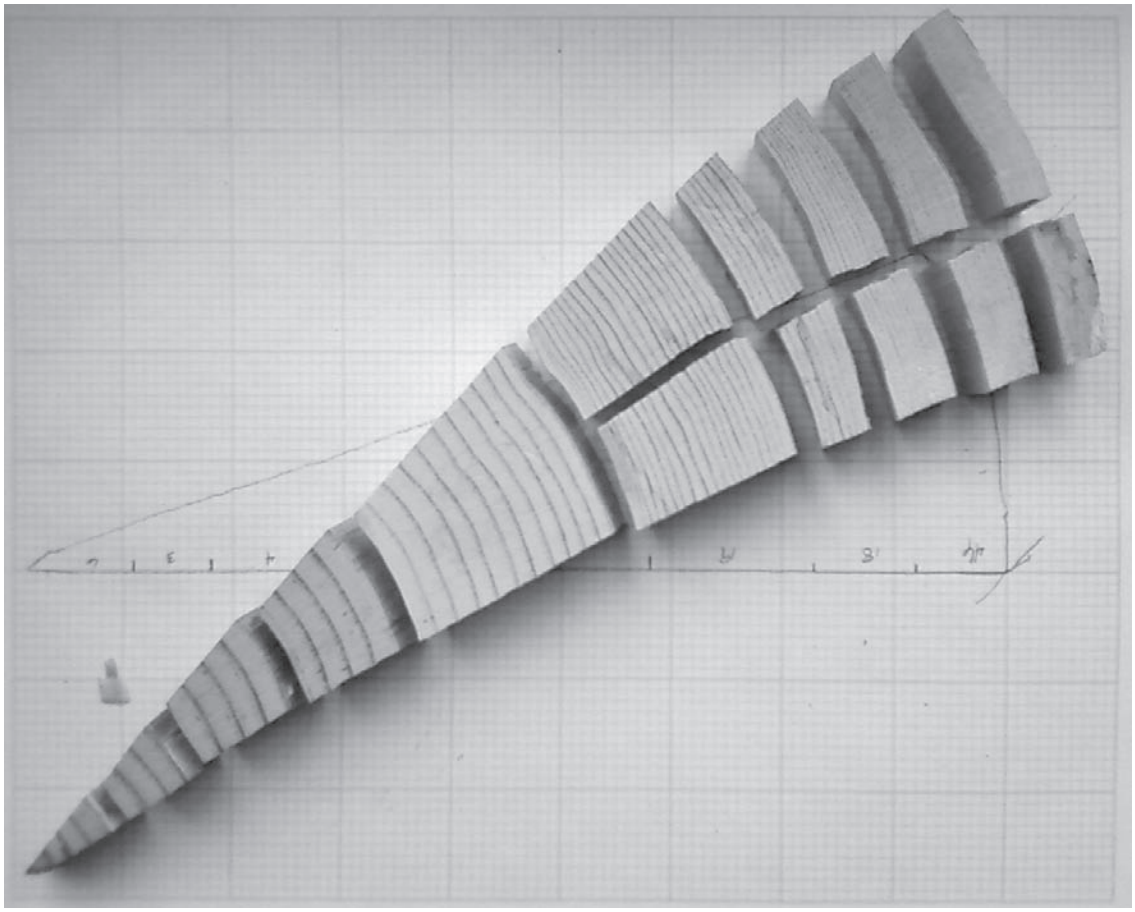

Figure 2— Breast height wedge section, divided into separate parts based on radial growth patterns, in preparation for polysaccharide composition analysis of a reputed unsuppressed tree in a ponderosa-dominated stand sampled for biofuel potential at Lookout Mountain, Pringle Falls Experimental Forest, central Oregon.

Enzymatic Cellulose Conversion

Short bolts were hand debarked and chipped. The wood chips were screened, retaining only particles greater than 6 mm and less than 38 mm in length to ensure smooth operation in disk milling. The thickness of the chips ranged from 3 to 8 mm.

The wood chips were conditioned to 30 wt% solids, then loaded into a laboratory batch digester (capacity 23 liters) and subjected to a SPORL pretreatment at 180 °C for 30 minutes. An 8 percent sodium sulfite charge on wood (dry basis) was added to the pretreatment liquor for an initial sodium sulfite concentration of 2.67 percent. A second pretreatment was conducted in more acidic conditions (pH 1.9) by adding sulfuric acid to the liquor at a dosage of 2.21 percent on wood (dry base) or acid concentration of 0.4 percent (volume-to-volume). After the SPORL pretreatments, the chips were directly fed into a 305 mm atmospheric disk refiner (plate pattern D2B-505, Andritz Sprout-Bauer, Springfield, OH) for fiberization. The chemical compositions of the untreated wood and pretreated wood substrates were analyzed using an improved high-performance anion exchange chromatograph with pulsed amperometric detection (HPAEC-PAD) (Davis 1998). Reported data averages of duplicate measurements conducted three weeks apart.

Enzymatic hydrolysis was conducted using commercial enzymes at 2 percent of substrate solid (weight-to-volume percent) in 50-mL sodium acetate buffer using a shaker/incubator (Thermo Fisher Scientific, Model 4450, Waltham, MA) at 200 rpm. The pH and temperature were adjusted to 4.8 and 50 °C, respectively. A mixture of Novozyme Celluclast 1.5 L cellulose with an activity loading of approximately 15 filter-paper units per gram (FPU/g) substrate and Novozyme 188 (β-glucosidase) cellobiase with an activity loading of approximately 22.5 cellulose-binding module per gram (CBU/g) substrate were used for enzymatic hydrolysis[1]. Hydrolysates were sampled periodically for glucose analysis using a Glucose Analyzer (YSI 2700S, YSI Inc., Yellow Springs, OH).

Results

Stand Structure and Tree Growth

Both the lodgepole pine stand and the ponderosa pine stand were overstocked, with high density, high SDI, and high basal area (table 1). These values are consistent with the outcome of more extensive field sampling of ponderosa pine stand structure throughout the Lookout Mountain Unit[2], indicating that basal area generally ranged from 34.4 to 41.3 m^2 ha^{-1} and SDI generally ranged from 223 to 257. Cochran and Barrett (1999) showed that continued mortality in ponderosa pine from mountain pine beetle occurred when the SDI was above 240, and set an upper management zone (UMZ) for SDI to sustain healthy stand conditions for forest stands based on the dominant plant association. Upper management zones defined specifically for the Deschutes National Forest represent the level of stocking or relative density beyond which there is imminent risk of catastrophic loss of overstory trees to bark beetles[3]. Values for the UMZ for the lower portion of Lookout Mountain containing the lodgepole pine and ponderosa pine stands

1 Enzymes were generously donated by Novozymes (Franklinton, NC).

2 Data on file, LaGrande Forestry Sciences Laboratory, LaGrande, OR.

3 Booser, J, and J. White. Calculating maximum stand density indexes (SDI) for Deschutes National Forest plant associations. Unpublished paper, on file, Deschutes National Forest.

Table 1—Structural attributes of density, mean diameter at breast height, stand density index, and basal area by species in two stands sampled for biofuel potential at Lookout Mountain, Pringle Falls Experimental Forest, central Oregon.

Site	Density	Diameter	Stand density index	Basal area
	stems ha^{-1}	*cm*		*m^2 ha^{-1}*
Lodgepole stand				
Live lodgepole pine	375	20.0	109.2	13.72
Dead lodgepole pine	50	12.6	21.2	2.74
Live ponderosa pine	538	14.6	132.5	18.58
Dead ponderosa pine	87	2.8	2.7	0.26
Total	1050	16.7	265.6	35.28
Ponderosa stand				
Live lodgepole pine	112	8.2	15.8	2.02
Dead lodgepole pine	0	0	0	0
Live ponderosa pine	938	18.3	293.5	39.97
Dead ponderosa pine	100	6.6	4.5	0.44
Total	1150	16.3	313.8	42.44

sampled in this study fall between an SDI of 102 and 156, thus these two stands are 225 percent above the UMZ and are in imminent risk of complete overstory mortality from bark beetles.

Sampled diameter distributions for both stands had steeply decreasing or reverse J shapes, indicating that these stands were relatively young and individual trees were experiencing strong competition for space and resources (fig. 3). Diameter distributions and stem densities for both stands did not resemble diameter distributions and stem densities of nearby old-growth ponderosa pine stands (Youngblood and others 2004).

Diameter-total tree height relations were consistent with relatively young stand structure; lodgepole pine showed steeper height growth with diameter compared to ponderosa pine (fig. 4). Total height of lodgepole pine likely is near the upper limit for this region, while ponderosa pine on similar relatively productive sites will reach 45 m in total height (Youngblood and others 2004).

As expected, breast-height ages in both the lodgepole pine- and ponderosa pine-dominated stands were consistent with single-cohort structure originating from a single stand-replacement event (fig. 5). Lodgepole pine breast-height ages ranged from 91 to 107 years. Assuming 10 years to reach breast height, this suggests a stand replacement disturbance such as a fire and a re-establishment date of 1890. Ponderosa pine breast-height ages ranged from 73 to 107 years. This wider period of re-establishment is not unusual, especially since ponderosa pine seedlings are known to be more sensitive to low temperature than lodgepole pine in this region (Cochran 1972).

Both the lodgepole pine and ponderosa pine ring-width analysis indicated that the reputed "unsuppressed" trees were actually suppressed. Reputed unsuppressed trees had similar establishment dates, sensitivity, and first-order autocorrelation as suppressed trees at the same location (table 2). In addition, the chronology developed for the reputed unsuppressed trees showed little difference from the chronology developed for all other suppressed trees, as shown for lodgepole pine in figure 6. For lodgepole, all chronologies contained periods with less than expected growth from 1944 through 1957, from 1994 through 1998, and since 2002. Ponderosa pine chronologies contained extended periods with less than expected growth from 1960 through 1977 and from 1995 through 1999.

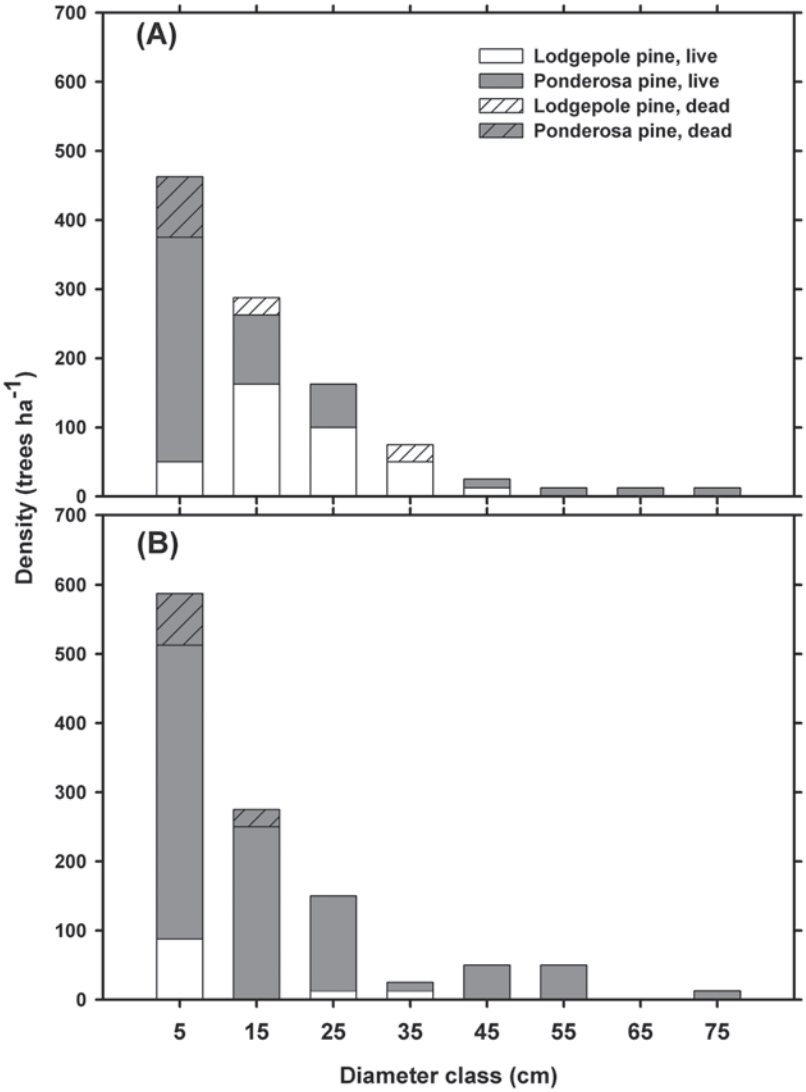

Figure 3—Diameter distribution of lodgepole pine and ponderosa pine at (A) a lodgepole-dominated stand and (B) a ponderosa-dominated stand sampled for biofuel potential at Lookout Mountain, Pringle Falls Experimental Forest, central Oregon.

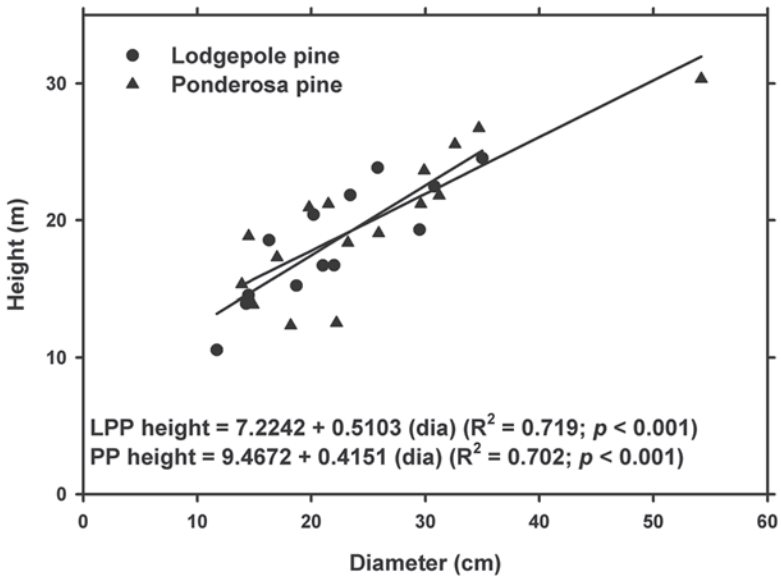

Figure 4—Breast-height diameter-total height distribution in a lodgepole-dominated stand (n = 13) and a ponderosa-dominated stand (n = 16) sampled for biofuel potential at Lookout Mountain, Pringle Falls Experimental Forest, central Oregon.

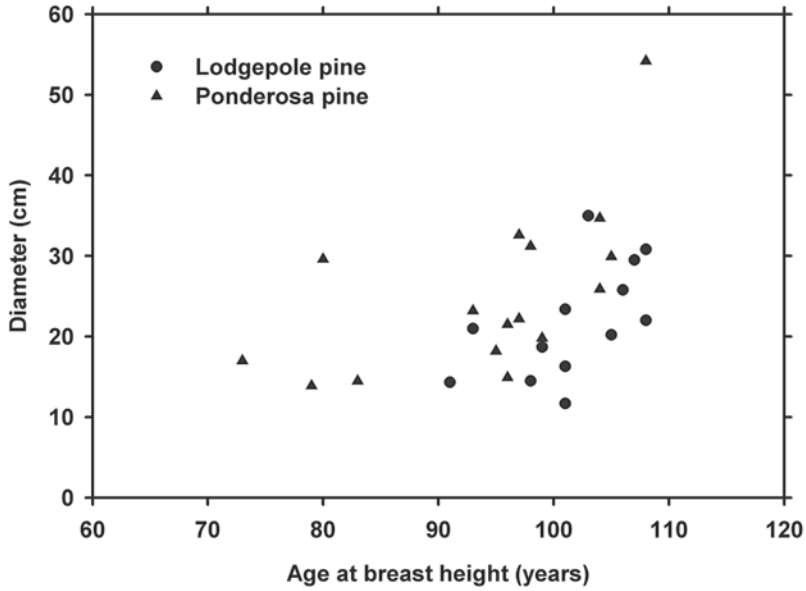

Figure 5—Breast-height age-diameter distribution in a lodgepole-dominated stand (n = 13) and a ponderosa-dominated stand (n = 16) sampled for biofuel potential at Lookout Mountain, Pringle Falls Experimental Forest, central Oregon.

Table 2—Statistics of tree ring chronologies by species in two stands sampled for biofuel potential at Lookout Mountain, Pringle Falls Experimental Forest, central Oregon.

Site	Sample size	First year	Ring width	MS	AC	$\overline{r}_{bt}$	$\overline{r}_{wt}$	EPS
			mm					
Lodgepole pine stand								
Suppressed	24	1900	0.94	0.179	0.858	0.144	0.608	0.94
Unsuppressed	2	1905	1.59	0.132	0.918	NA	0.701	NA
All trees	26	1900	0.99	0.176	0.862	0.162	0.615	0.87
Ponderosa pine stand								
Suppressed	30	1900	1.04	0.221	0.902	0.034	0.523	0.59
Unsuppressed	2	1900	1.99	0.209	0.964	NA	–0.015	NA
All cores	32	1900	1.11	0.220	0.906	0.034	0.489	0.61

Mean ring width; MS, Mean sensitivity; AC, first-order autocorrelation; $\overline{r}_{bt}$, mean correlation between trees; $\overline{r}_{wt}$, mean correlation within trees; EPS, expressed population signal, a measure of the amount of year-to-year growth variations shared among trees of the same chronology.

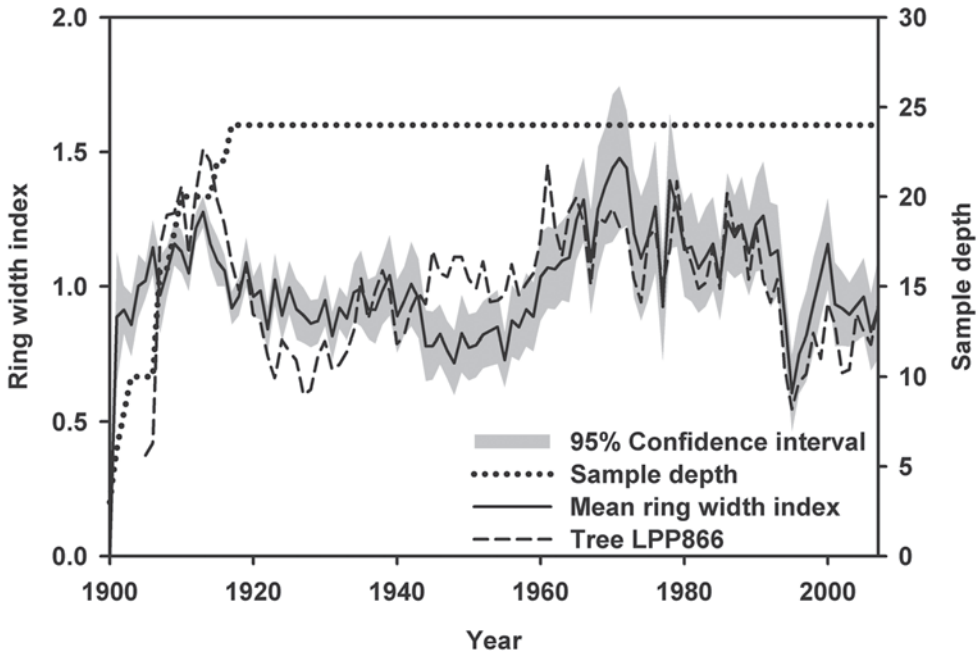

Figure 6—Standardized ring-width chronology with 95% confidence interval (left axis) and sample depth (right axis) from a suppressed lodgepole pine-dominated stand sampled for biofuel potential at Lookout Mountain, Pringle Falls Experimental Forest, central Oregon. Standardized ring-width index for the "unsuppressed" tree is overlaid.

Growth Analysis and Wood Chemistry

Preliminary analysis relating wood chemistry to tree growth focused on the reputed unsuppressed ponderosa pine. Ten discrete sections with unique growth characteristics were identified in the disk extracted at breast height (table 3). The first four sections (A through D) contained the first 17 years of growth and include rings formed under relatively rapid radial growth. Section E began beyond the apparent extractives region of heartwood and contained rings formed by rapid and

Table 3—Statistics of radial growth patterns by section in a reputed unsuppressed ponderosa pine sampled for biofuel potential at Lookout Mountain, Pringle Falls Experimental Forest, central Oregon.

Section	Number rings	Cumulative age	Section width	Cumulative radius	Mean ring width	Section area	Disk area	Ring area	Growth rate
		year	*mm*	*mm*	*μm*	*cm²*	*cm²*	*%*	*cm²yr⁻¹*
A [a]	6	6	23.6	23.6	3937	18	18	1.1	2.9
B	3	9	17.5	41.1	5842	36	53	2.3	11.9
C	4	13	25.4	66.5	6350	86	139	5.6	21.5
D	4	17	21.1	87.6	5271	102	241	6.6	25.5
E	11	28	49.0	136.6	4457	345	587	22.3	31.4
F	20	48	39.1	175.7	1956	383	971	24.8	19.2
G	7	55	12.7	188.4	1814	145	1116	9.4	20.7
H	11	66	12.7	201.1	1155	155	1271	10.0	14.1
I	18	84	10.5	211.6	586	137	1408	8.8	7.6
J	28	112	10.2	221.8	363	138	1546	8.9	4.9

[a] Section A closest to pith; section J closest to bark

consistent growth over the next 11 years, with mean ring width of 4.5 mm. This section likely represents normal or unsuppressed growth. Radial growth slowed over the next 27 years (sections F and G) and then rapidly declined. Over the next 57 years (sections H through J), annual radial growth dropped from about 1814 μm year^{-1} to 363 μm year^{-1} and the corresponding disk area only increased by 27 percent. Over a period of 7 years from 1994 to 2000, annual rings averaged only 3 earlywood fibers and 2 latewood fibers in the radial direction. These sectional results are consistent with the full ponderosa pine chronology results presented above. Despite its outward appearance, this reputed "unsuppressed" tree was as suppressed as other trees selected on the basis of physical appearance to represent suppression.

Numerous polysaccharides, including glucan, were identified in the wood of the reputed unsuppressed ponderosa pine (table 4). Only modest differences in glucan concentration were found among the various sections of the reputed unsuppressed ponderosa pine (sections E through I described above). In addition, sampling of short bolts from higher in the same tree, representing areas of recent growth, failed to establish a clear trend of increasing or decreasing glucan concentration. Other samples, corresponding to the most recent 60 years from several trees with the smaller diameters and thus representing potentially the more suppressed trees of those sampled, also contained glucan in similar amounts.

A similar preliminary evaluation of polysaccharide composition of lodgepole indicated that component polysaccharides occur in similar concentrations to those found in ponderosa pine, with perhaps slightly higher concentrations of galactan and xylan.

Enzymatic Cellulose Conversion

Substrate chemical composition after SPORL pretreatment differed from untreated samples at pH 4.2 and pH 1.9 (table 5). For both pretreatment levels of pH, lignin loss was about 10 percent while most of the hemicellulose (xylan and mannan) was removed. Glucan loss, undesirable for our purposes, was undetected at the pH 4.2 level (measurement uncertainty resulted in no loss and repeated measurements showed consistent results), and was moderate (10 percent) at the pH 1.9 level. Enzymatic cellulose conversion of fiberized lodgepole pine using the SPORL pretreatment process was time dependent as expected (fig. 7). Preliminary results suggest that near complete cellulose conversion to glucose can

Table 4—Composite polysaccharide composition (%) by component and glucan/carbohydrate ratio (G/C ratio) of lodgepole pine and ponderosa pine cross-sectional samples analyzed for biofuel potential at Lookout Mountain, Pringle Falls Experimental Forest, central Oregon, and reported values from other sources.

	Component composition									
	Ash	Lignin	Arabinan	Galactan	Glucan	Xylan	Mannan	Carbohydrates	Yield	G/C ratio
	percent									
Lodgepole pine	0.0	27.9	1.6	2.1	42.5	5.5	11.6	63.4	91.1	0.67
Ponderosa pine	0.0	26.9	1.8	3.9	41.7	6.3	10.8	64.6	91.4	0.65
Aspen[1]		23.0	0.0	0.0	45.9	16.7	1.2	63.8	86.8	0.72
Corn stover[2]	13.3	19.7	2.1	0.8	34.0	19.5	0.5	56.9	89.9	0.60
Hybrid poplar[2]	1.0	23.5	0.5	0.7	43.7	16.6	2.8	64.3	88.8	0.68
Switch grass[2]	5.8	19.5	2.9	1.2	33.6	23.1	0.4	61.2	86.5	0.55
Wheat straw[2]	10.2	16.9	2.4	0.8	32.6	19.2	0.3	55.3	82.4	0.59

[1] Wang et al., 2009.

[2] DOE.

Table 5—Chemical composition of lodgepole pine wood, untreated and pretreated using a Sulfite Pretreatment to Overcome the Recalcitrance of Lignocellulose (SPORL) process at two initial levels of pH, evaluated for biofuel potential as suppressed trees sampled at Lookout Mountain, Pringle Falls Experimental Forest, central Oregon.

	Liquor		Component weight								
Pretreatment	Initial	Final	Klason lignin	Arabinan	Galactan	Rhamnan	Glucan	Xylan	Mannan	Sum	Total
	- - - - pH - - - -		- - - - g - - - -								
Untreated			27.09	1.56	2.23	0.07	42.55	6.93	10.99	91.42	100.0
SPORL	4.2	2.5	25.26	0.0	0.02	0.0	46.03	1.80	0.86	73.97	76.9
SPORL	1.9	1.5	25.43	0.01	0.05	0.0	38.19	0.32	0.25	64.25	66.7

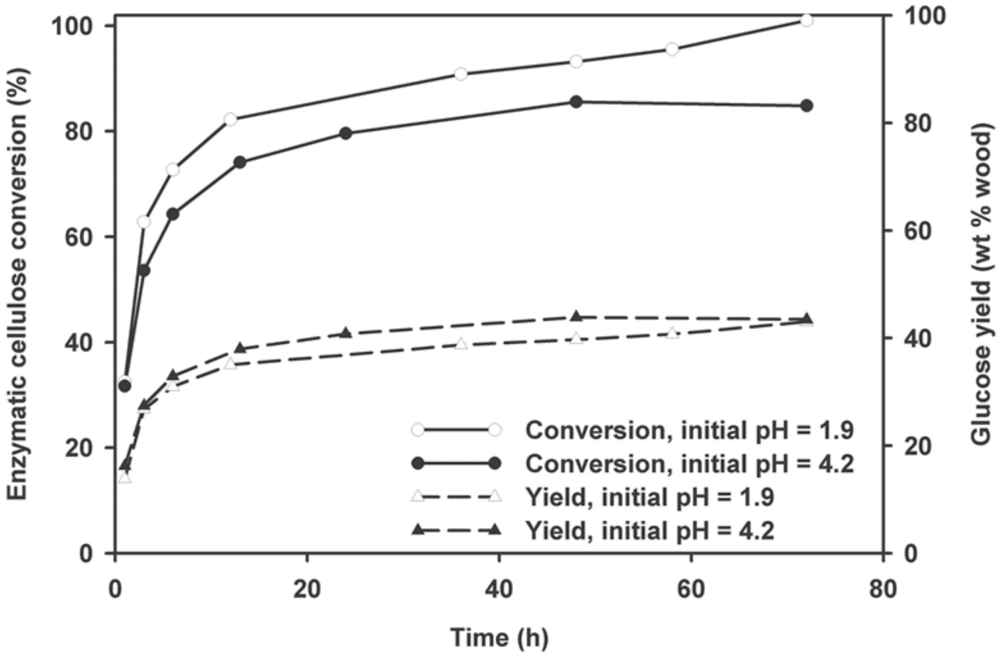

Figure 7—Mean time-dependent enzymatic cellulose conversion (%) (left axis) and glucose yield (wt % wood) (right axis) from substrates of lodgepole pine pretreated using a Sulfite Pretreatment to Overcome the Recalcitrance of Lignocellulose (SPORL) process to evaluate biofuel potential of suppressed trees sampled at Lookout Mountain, Pringle Falls Experimental Forest, central Oregon.

be achieved in 48 hours at an enzyme dosage of 15 FPU/g substrate (equivalent to about 25 FPU/g cellulose). Considering an initial glucan content of about 42 percent in untreated chips, the SPORL pretreatment, with subsequent disk milling and enzymatic hydrolysis, produced high glucose yields from the lodgepole pine chips, suggesting that highly suppressed lodgepole pine could be a viable source of lignocellulosic biomass for conversion to biofuel.

Discussion

In this work, we explored opportunities to use small diameter lodgepole pine and ponderosa pine, scheduled for removal as part of a landscape-scale thinning and fuel reduction project in Pringle Falls Experimental Forest, as biomass to

support biofuel production. Using dendrochronological techniques, we confirmed that trees selected in the field representing suppressed crown classes were severely suppressed based on the criterion of reduced diameter growth for at least five years. In addition, we found that both the reputed "unsuppressed" lodgepole pine and the reputed "unsuppressed" ponderosa pine were also suppressed. This limited our comparison of glucose conversion.

Analysis of polysaccharide components was limited to a portion of our ponderosa pine and lodgepole pine samples. While subtle differences were detected among the samples, the apparent differences that were expected to occur in relation to bole height were not readily distinguishable. Overall, we found similar amounts of glucan in the selected wood samples. The amounts we found suggest that both lodgepole pine and ponderosa pine, when suppressed, contain sufficient glucan and other polysaccharides that conversion to biofuel is possible.

Pretreatment is one of the most expensive operations in the conversion of cellulosic biomass to fermentable sugars. Other pretreatment processes have yet to effectively overcome recalcitrance of woody biomass, especially conifer biomass, and thus fail to enhance the conversion of cellulose to glucose through enzymatic saccharification. For example, weak acid pretreatment can achieve only about 40 percent cellulose conversion to glucose (Wyman and others 2009; Zhu and others 2009a). Ammonia pretreatment is not effective for biomass with high lignin content (Sun and Cheng 2002; Wyman and others 2009). Steam catalyzed-steam explosion produced slightly better conversions of cellulose to glucose (up to about 70 percent) with greater energy demands (Tengborg and others 2001; Wyman 2009). Organosolv pretreatment can result in over 90 percent enzymatic cellulose conversion to glucose, but hemicellulose recovery from the solvent is low. The economic production of ethanol from woody biomass using organosolv process requires that high value lignin products be developed and marketed. A proposed $80 million ethanol demonstration plant in Colorado that would use the organosolv process was recently terminated due to the instability of energy prices and uncertainty in the capital markets, before markets for lignin products could be developed. In contrast, pretreatment of lodgepole pine and ponderosa pine chips with the SPORL proved both cost and time effective. The SPORL pretreatment resulted in excellent conversion of cellulose to glucose as demonstrated in this study, has excellent fermentable hemicellulose sugar recovery (Zhu and others 2009a), low energy consumption (Zhu and others 2009b), and commercial scalability. We believe the SPORL pretreatment offers an enormous potential for cellulosic ethanol production using small diameter woody biomass resulting from forest thinning and fuels reduction activities.

While field corn currently is the most common source of biofuel, our preliminary results indicate that the vast area in the western US currently supporting high density stands of lodgepole pine and ponderosa pine could contribute to energy demands by providing sources of woody biomass for conversion to biofuel. Lodgepole pine and ponderosa pine occur across the western US at low elevations as dry forests; few areas in the western US have the potential to support production of field corn at the same extent. Woody biomass in these stands has far greater mass per unit area than field corn, thus transportation and processing costs should be evaluated in more detail. Further, field corn is harvested within a relatively short window of time and requires storage prior to processing. Woody biomass is relatively easy to store, and could be harvested from western forests and transported throughout the year, thus providing greater stability in workforce demands and equipment needs. Finally, the SPORL pretreatment process and other technological advances have removed any remaining technological barrier to bioconversion of woody biomass to biofuel.

Although the two western conifers tested in this study proved capable of contributing to ethanol production through SPORL pretreatment technology, major research and development will be needed before silvicultural treatments with ethanol objectives might be implemented. First, additional effort is essential to ensure that the technology, developed with relatively small digesters and incubators, is scalable in terms of sugar and ethanol yields and chemical and enzyme dosages. Pilot or full-scale production facilities are needed to confirm that energy savings associated with wood size reduction observed in our laboratory study can be expected across a range of commercial facilities. The commercial marketability of dissolved lignosulfonate, as a byproduct of the SPORL process, is largely unknown. While the technology builds on existing equipment, technology, and infrastructure that have long been used in the pulp and paper industry, this infrastructure is not readily available throughout much of the western US where lodgepole pine and ponderosa pine occur. In addition, operational-scale ethanol production would likely require a similar scale logging infrastructure, yet throughout much of the western US the logging infrastructure has disappeared over the past decade. Finally, our work is limited to two tree species, and much of their distribution in the western US is on federally administered lands. Management of forests dominated by lodgepole pine and ponderosa pine on these public lands for ethanol conversion must be balanced with management of other resource objectives.

Acknowledgments

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The content of this paper reflects the views of the authors, who are responsible for the facts and accuracy of the information presented herein.

Part 3: Alternative Silvicultural Strategies

Estimating Site Index From Tree Species Composition in Mixed Stands of Upland Eastern Hardwoods: Should Shrubs be Included?

W. Henry McNab[1]

Abstract—Site index is the most widely used method for site quality assessment in hardwood forests of the eastern United States. Its application in most oak (*Quercus* sp. L.) dominated stands is often problematic, however, because available sample trees usually do not meet important underlying assumptions of the method. A prototype method for predicting site index from tree species composition has shown promising results in the southern Appalachian Mountains. The objective of this study was to determine if upland oak site index was associated with two common understory shrubs: mountain laurel (*Kalmia latifolia* (L) and spice bush (*Lindera benzoin* (L) Blume). Regression analysis indicated that including these shrubs in the model increased r^2 from 0.63 to 0.67. Predicted upland oak site index decreased by 5.4 ft if mountain laurel was present on a sample plot and increased by 7.4 ft if spice bush was present. Results from this exploratory study conducted in a small watershed should be validated elsewhere, but the findings do suggest that estimates of upland oak site index using the species composition method can be improved by observing the presence of mountain laurel and spice bush.

Keywords: mesophytic, site quality, soil moisture, upland oaks, xerophytic, yellow-poplar.

Introduction

The productive capacity of forest stands strongly influences the response to silvicultural treatments (Smith 1962). Except for short rotation plantation biomass culture, however, estimation of forest site productivity by direct measurement of product yields is generally not feasible. Indirect estimation of forest site quality using site index is the method most used in upland hardwoods (Spurr and Barnes 1973). Beck and Trousdell (1973) provide a thorough description of the method, particularly its underlying assumptions and limitations. Estimation of site index in mixed hardwoods is often problematic, however, because suitable sample trees are generally lacking particularly on intermediate and lower quality sites, resulting in biased estimates of site quality. Also, site index is a timber measure and cannot be readily adapted for other applications, such as ecological studies or assessment of wildlife habitat. Replacement of conventional site index estimation with an alternate method is particularly desirable for ecosystems where oaks (*Quercus* sp. L.) are a dominant component of the overstory because many stands lack trees suitable for determination of site quality. One such alternative method to site index is a procedure reported by Whittaker (1956) for arraying stands on environmental gradients based on composition of the tree stratum.

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Forest productivity in the southern Appalachian Mountains is associated primarily with temperature and moisture gradients (Whittaker 1966) and somewhat with fertility. Whittaker (1966) reported that "… an index of site moisture conditions based on weighted averages of stand composition…" was highly correlated with forest production. He subdivided the topographic-soil moisture gradient within an elevation zone into four soil moisture classes (mesic, submesic, subxeric, and xeric) and assigned a weight (0, 1, 2, or 3) to each class. Each tree species was assigned to a soil moisture class based on its modal frequency of occurrence along the gradient. Whittaker used the average weight of the frequency of each species present >1 in diameter at breast height (dbh) as an index of the soil moisture conditions for a site. The index, which was a means for quantifying the relative position of sites on the moisture gradient, was highly correlated with primary forest production for vegetative communities occupying environments ranging from xeric to mesic in the Great Smoky Mountains National Park (GSMNP) (Whittaker 1966). The simplicity of such a site classification system is appealing for several reasons: it can be applied with data typically collected from sample plots in a systematic inventory of stand conditions, it is easily adapted to other ecosystems with their associated species, and the system can be extended to other environmental gradients of temperature and nutrients.

Whittaker's (1966) methods were used in a previous study (McNab and Loftis, in press) to array forested sites along a moisture gradient that was quantified with a moisture regime index (MRI) calculated from arborescent vegetation present on each plot. Results of that study indicated that upland oak site index was correlated with the MRI. Shrubs were not included in that study, but a number of species occur in the Southern Appalachians and appear to be associated with sites of varying environmental conditions (Stupka 1964). Whittaker (1966) reported about a dozen shrubs species were common in the GSMNP and reported the "… distributions of undergrowth species do not appear closely related to those of dominant [tree] species." The independence of tree and shrub species distributions observed by Whittaker (1966) suggests that each form of vegetation was responding to different environmental conditions and could account for different sources of variation that affect site quality. My study investigated the question: Are common species of forest shrubs associated with forest site quality? This study is an extension of a previous study of site quality that was based only on tree species (McNab and Loftis, in press). The purpose of this exploratory study was primarily to determine if additional study on this topic is warranted and not to develop an application for immediate use by managers.

Methods

Study Area

The study was conducted in the Bent Creek Experimental Forest; a 6,000-ac watershed located about 10 miles southwest of Asheville, North Carolina. This area is characterized by short, mild winters and long, warm summers; elevation ranges from 2000 to 4000 ft. Annual precipitation averages about 45 in and is evenly distributed throughout the year. Geologic formations consist of gneisses and schists of Precambrian Age that have weathered to form a complex, dissected land surface consisting of ridges and coves. Soils, which consist mainly of Ultisols with lesser areas of Inceptisols, are generally deep (>40 in), highly acidic (pH<5.5) and range in moisture regimes from xeric to hydric.

The arborescent canopy of xeric to subxeric sites on upper slopes and ridges typically consists of communities dominated by scarlet oak (*Quercus coccinea*

Muenchh.), white oak (*Q. alba* L.) chestnut oak (*Q. prinus* L.), and black oak (*Q. velutina* Lam.). Typical mesophytic species occurring on moist slopes and coves include yellow-poplar (*Liriodendron tulipifera* L.), northern red oak (*Q. rubra* L.), white ash (*Fraxinus americana* L.), eastern hemlock (*Tsuga canadensis* (L.) Carr.), cucumbertree (*Magnolia acuminata* L.), and black locust (*Robinia pseudoacacia* L.). Midstory species include, dogwood (*Cornus florida* L.), blackgum (*Nyssa sylvatica* Marsh.), sourwood (*Oxydendrum arboreum* (L.) DC.); shortleaf pine (*Pinus echinata* L.) or pitch pine (*P. rigida* L.) may occur on disturbed dry sites. Various hickory species occur on dry and moist sites. Red maple (*Acer rubrum* L.) is usually present on sites of all moisture regimes. American chestnut (*Castanea dentata* L.) was a major component of most stands before it was lost as a canopy species as a result of the introduced blight disease (*Cryphonectria parisitica*) in the 1920s; it was typically replaced by oaks (*Quercus* spp. L.) on many sites. Common shrubs of dry sites include mountain laurel (*Kalmia latifolia* L.) and deerberry (*Vaccinnium staminium* L.); moist sites support rosebay rhododendron (*Rhododendron maximum* L.) and spicebush (*Lindera benzoin* (L.), Blume). More than 250 trees, shrubs, and vines are native to this area of the southern Appalachian Mountains (Stupka 1964).

Principal types of disturbance in the watershed have resulted from natural and human causes. Most stands in Bent Creek Experimental Forest have been affected by past land-use. Extensive areas of gentle slopes were cleared for agricultural purposes from 1800 until about 1900, when land abandonment resulted in conversion to pine-hardwood mixtures on dry sites and yellow-poplar on moist sites. Stands on areas of steeper slopes, which were not cultivated or cleared for pasture, were typically burned to promote sprouting for woodland grazing by livestock, and periodically harvested for timber by high grading. Following acquisition of the watershed by the USDA Forest Service around 1915, timber stand improvement work was done in selected areas to reduce stocking of non-commercial species such as red maple, sourwood, and dogwood. Lightning-caused fires are seldom a source of extensive disturbance in this humid region.

Field Plots and Vegetation Inventory

I used field data from several earlier silvicultural studies installed in the experimental forest for my exploratory investigation. Three requirements were necessary for the previously collected data to be suitable for use: (1) inventory of all tree and shrub species present on each plot, (2) accurate estimation of site index for stands on each sample plot, and (3) the sample plots represented the range of site qualities in the experimental forest. Field data from two previous studies satisfied these requirements: one study dealing with prediction of site index for xerophytic upland oaks and the other study to investigate growth and yield of mesophytic yellow-poplar after thinning. In the site index study, Doolittle (1957) established 114, 0.2-acre plots in even-aged stands of upland oaks ranging in age from 30 to 120 years. Arborescent and shrub vegetation in the sapling and tree strata had been re-inventoried by species in 1970 (unpublished data on file). Vegetation on this group of plots is typically described as xerophytic. The yellow-poplar growth and yield study consisted of 34, 0.25-ac plots that had been established in even-aged stands, ranging in age from 30 to 76 years, for which site index had been determined (Beck and Della-Bianca 1970). Vegetation associated with this group of plots is characterized as mesophytic. I pooled the inventories from these two studies to obtain a data set consisting of 148 sample plots that extended over the range of site qualities present on upland sites in the Bent Creek Experimental Forest.

Site index relationships developed by Schnur (1937) for upland oak stands in the central hardwood region were used as the standard measure of timber productivity. I converted yellow-poplar site index to upland oak site index using the relationship presented by Doolittle (1958). Upland oak site index overlapped between the two groups of plots characterized by composition of the arborescent vegetative community (Stupka 1964):

Group of plots (Source of data)	Range of oak site index (ft)
Mesophytic (Beck and Della-Bianca 1970)	71 - 95
Xerophytic (Doolittle 1957)	36 - 87

Across all 148 plots, oak site index averaged 65 ft. The plots established for the growth and yield study are dominated by yellow-poplar and other mesophytic species and are considered representative of high to moderately productive stands on mesic and submesic sites. The plots established for the site index study are dominated by oaks and other xerophytic species and are considered representative of moderate to low productivity on xeric and subxeric sites.

Woody perennial vegetation on each sample plot was recorded by species in two physiognomic groups of plants (i.e. tree or shrub) and two size classes (i.e. overstory or midstory). A tree was a plant with a well-defined stem carrying a definite crown; shrubs were plants without a well-defined stem (Helms 1998). Each tree on a sample plot was recorded by species in two arbitrary size classes: overstory (>4.5-in dbh), or midstory (0.01 to 4.5 in dbh). Each shrub was recorded by species without regard to size. The number of tree and shrub species present on each plot was determined for each size class from the inventory data.

Moisture Regime Index

Following the rationale of Whittaker (1956) I subdivided the apparent soil moisture gradient on upland sites in the Bent Creek watershed into four classes: xeric, subxeric, submesic, and mesic. A fifth class, hydric, was not included in this study because the tree species of interest do not occur on soils of that moisture regime. The xeric class of sites was typically associated with upper slopes of ridges; mesic sites occurred on lower slopes of valleys. The subxeric and submesic classes were on sites perceived as somewhat less dry or less moist than xeric or mesic, respectively. I assigned each tree species occurring in the watershed to a moisture regime class based on its perceived modal position (i.e. distributional center) along the gradient (e.g. post oak (*Quercus stellata* Wangenh.), xeric; chestnut oak, subxeric; northern red oak, submesic, etc). Soil moisture deficits would likely develop on xeric sites in the mid to late growing season and on subxeric sites during the late season of most years. On the submesic class of sites, soil moisture deficits would occur occasionally during the mid to late growing season and on mesic sites, lack of moisture for plant growth would likely occur only rarely during an interval of 30 years. Each moisture class was assigned a weight, ranging from 1 to 4 (e.g. xeric = 1; subxeric = 2; etc), which represented the perceived relative availability of soil moisture for plant growth during the frost-free season. Moisture class weights were refined by assigning half values to some species where their modal position appeared to occur between two classes (e.g. scarlet oak = 1.5; white oak = 2.5, etc).

The combined information provided by all tree species present on a sample plot is used to determine the location of the site on the moisture gradient. The numerical location of the plot (e.g. site) on the moisture gradient, which ranges in magnitude from 1 to 4, is defined as the moisture regime index (MRI). The plot MRI is calculated as the average moisture weight of all species present on the plot:

$$MRI = \sum(Species_i * MW_i + \ldots + species_j * MW_j) / N\ species$$

where: MRI is moisture regime index, species is each tree species present, MW is moisture weight value for each species (values 1 to 4), and N is number of species used in calculation of the index.

Because MRI can be calculated with various combinations of vegetation size classes and frequencies of occurrence, I used the method of calculation reported as best in an earlier study (McNab and Loftis, in press). I combined the midstory and overstory size classes for each plot and calculated MRI using tree species represented by more than one individual (N > 1). I evaluated the effect of shrubs on estimation of site index by including them as binary variables (e.g. 0 or 1) in regression analyses, where a species was present on ten or more sample plots.

Data Analysis

I used a completely randomized design where sample plots had been established previously in stands meeting criteria for study of timber productivity and without regard to topographic or soil characteristics.

Regression model without shrubs—Simple linear regression was used to determine the relationship of upland oak site index with MRI by the model:

$$Y_i = \alpha + \beta X_i + \varepsilon$$

where Yi are observed values of upland oak site index, α and β are population parameters, X_i are observed values of MRI, and ε is unexplained error.

Regression model with shrubs—A Spearman rank correlation test was used to determine the relationship between the frequencies of occurrence of the inventoried tree and shrub species in relation to the xeric (oak dominated) plots and the mesic (yellow-poplar dominated) plots. T-tests were used to determine if site index was significantly associated with the presence of each species of shrub. For those shrubs significantly associated with oak site index, multiple regression was used to determine the combined effect of moisture regime index and shrub species by the relationship:

$$Y_i = \alpha + \beta X_i + \beta Z_i + \varepsilon$$

where Y_i are observed values of upland oak site index, X_i are observed values of MRI, Z_i are species of shrubs absent or present (represented as 0 or 1), α and β are population parameters, and ε is unexplained error. Relative performance of the two types of models (e.g. site index estimated from MRI without shrubs, and with shrubs) was evaluated using the proportion of variation of the dependent variable explained by the independent variables (adjusted r^2). Statistical tests of significance were made at the probability level of 0.05.

Results and Discussion

Thirty-one arborescent species were present with frequency of occurrence of n >1 stems on the 148 sample plots (table 1) and were used in calculation of MRI for each plot. Although dry-site and moist-site species occurred on plots of all moisture regimes, their frequencies of occurrence were generally consistent with the perceived soil moisture conditions (e.g. xerophytic oaks tended to dominate plots characterized by xeric and subxeric soil moisture regimes). Spearman

Table 1—Arborescent species present (>0.05 inch dbh) on plots, moisture weight assigned to each species, and their frequencies of occurrence (percent of total plots sampled) in two studies installed previously for investigations of productivity for xerophytic and mesophytic vegetation types in the Bent Creek Experimental Forest.

Common name	Scientific name	Moisture weight	Plot moisture regime	
			Xerophytic[a]	Mesophytic[b]
			- - - - - - *Percent*- - - - - - -	
Post oak	*Quercus stellata*	1.0	1	0
Virginia pine	*Pinus virginiana*	1.0	0	6
Scarlet oak	*Quercus coccinea*	1.5	75	0
Black oak	*Quercus velutina*	2.0	60	32
Black gum	*Nyssa sylvatica*	2.0	32	21
Chestnut oak	*Quercus prinus*	2.0	71	15
White pine	*Pinus strobus* L.	2.0	0	18
Persimmon	*Diosporis virginiana*	2.0	1	0
Sassafras	*Sassafras albiumn*	2.0	2	15
Shortleaf pine	*Pinus echinata*	2.0	32	9
Sourwood	*Oxydendrum arboreum*	2.0	86	56
Southern red oak	*Quercus falcata*	2.0	2	0
Hickory spp	*Carya* spp.	2.3	0	62
Holly	*Ixex opaca*	2.5	0	3
White oak	*Quercus alba*	2.5	80	32
Red maple	*Acer rubrum*	2.5	74	82
American beech	*Fagus grandifolia*	3.0	3	3
Black locust	*Robinia pseudoacacia*	3.0	37	71
Dogwood	*Cornus florida*	3.0	42	47
Frazier magnolia	*Magnolia fraserii*	3.0	0	6
Mulberry	*Morus alba*	3.0	0	3
Northern red oak	*Quercus rubra*	3.0	31	35
Black cherry	*Prunus serotina*	3.5	0	3
Cucumber	*Magnolia acuminata*	3.5	2	0
Eastern hemlock	*Tsuga canadensis*	3.5	3	21
Hornbeam	*Ostraya virginiana*	3.5	1	0
Sweet birch	*Betula lenta*	3.5	3	74
Yellow-poplar	*Liriodendron tulipifera*	3.5	39	100
Black walnut	*Juglans nigra*	4.0	0	3
White ash	*Fraxinus americana*	4.0	4	26
Yellow birch	*Betula lutea*	4.0	0	3

[a] 114, 0.2-ac plots situated in stands dominated by oaks.

[b] 34, 0.25-ac plots situated in stands dominated by yellow-poplar.

$r = 0.49$ ($n = 31$, $p < 0.01$), however, indicated a moderate similarity of tree species occurrence between the two sets of plots.

Six species of shrubs were present on the sample plots, but the principal species were mountain laurel and spice bush (table 2). There was little correlation between shrub species occurring on the two sets of plots (Spearman $r = 0.11$, $n = 6$, $p > 0.05$). Four shrubs were present on less than ten plots, leaving only mountain laurel and spice bush present with frequency judged adequate for analysis. These two species rarely occurred on the same plot.

Table 2—Shrub species present on sample plots, and their frequencies of occurrence (percent of total plots sampled) in two studies installed previously for investigations of productivity of xerophytic and mesophytic vegetation types in the Bent Creek Experimental Forest.

Common name[a]	Scientific name	Plots sampled	Plot moisture regime	
			Xerophytic[b]	Mesophytic[c]
		N	- - - - - - *Percent*- - - - - - -	
Mountain laurel	*Kalmia latifolia* (L.)	32	25	2
Spice bush	*Lindera benzoin* (L., Blume)	21	6	47
Rhododendron	*Rhododendron maximum* (L.)	2	2	0
Flame azalea	*R. calendulaceum* (Michx., Torr.)	4	3	3
Deerberry	*Vaccinium stamineum* (L.)	1	0	3
Black huckleberry	*Gaylussacia baccata* (Wang., K. Koch)	6	4	0

[a] 95 plots had no shrubs and two or more species occurred on 13 plots.

[b] 114 plots were located in stands dominated primarily by oaks.

[c] 34 plots were located in stands dominated primarily by yellow-poplar.

Site Index Model With Shrubs Excluded

Regression analysis of site index as a function of the moisture index produced the relationship:

$$\text{Oak SI (ft)} = -21.972 + 36.524*(\text{MRI})$$

where SI is site index (50 years) in feet for mixed oaks and MRI is moisture regime index based on midstory and overstory tree species with plot density >1. This equation has an r^2 of 0.63 and mean square error of 8.10 ft. The MRI variable was highly significant ($p<0.001$).

The distribution of upland oak site index in relation to moisture regime index without shrubs is shown in figure 1. The simple regression model fitted the field data reasonably well, and the pattern of residuals from the regression appeared to be uniformly distributed, suggesting a prediction equation with little bias.

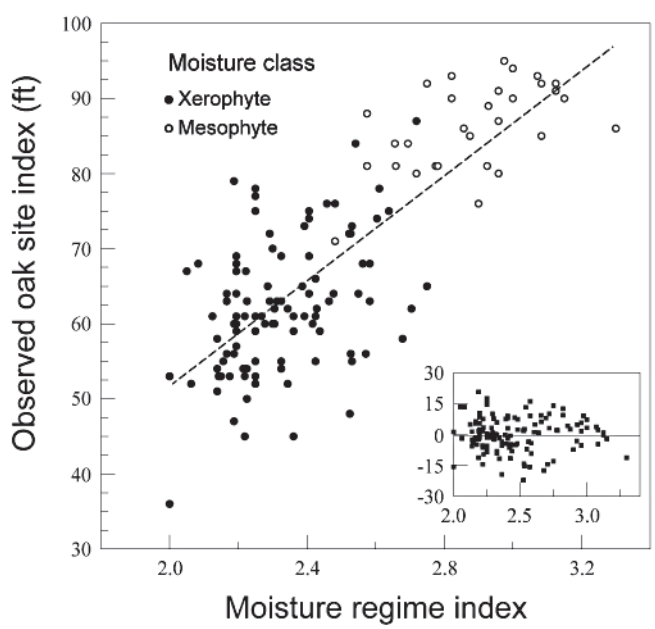

Figure 1—Observed site index measured on 148 sample plots (plotted points) dominated by either xerophytic or mesophytic species and predicted site index (dashed line) in relation to the moisture regime index. The inset shows residuals (actual - predicted) of predicted site index in relation to the moisture regime index.

Observed site index varied by about ±20 feet from predicted values for the xerophytic plots and about ±10 feet from the mesophytic plots, however, suggesting that other variables might explain part of the variation in site index not accounted for by the tree species and their perceived association with a moisture gradient.

Site fertility could account for additional variation of site index not explained by moisture (Kimmins 1987), but relatively little has been reported on the effects of nutrients on height growth for most tree species. Gilmore and others (1968) found yellow-poplar height was better correlated with soil pH than nutrients. Carmean (1975) reported that for various species of oaks, fertility (particularly pH) had a stronger influence on the distribution of some species (e.g. chinquapin oak (*Quercus muehlenbergii* Engelm.) than on height growth.

Site Index Model with Shrubs Included

The two predominant shrubs were each associated with oak site index (table 3). Compared to plots with no shrubs, average oak site index was lower by 8.2 ft where mountain laurel was present ($t = -3.70$, $P<0.003$) and site index was higher by 19.6 ft on plots were spice bush was present ($t = 7.48$, $P <0.001$). Regression analysis of moisture regime index and the two significant shrub variables produced the following relationship:

$$\text{Oak SI (ft)} = 15.564 + 21.918*(\text{MRI}) - 5.432*(\text{KALA}) + 7.368 * (\text{LIBE})$$

where: SI is site index (50 years) in feet for mixed upland oaks, MRI is moisture regime index calculated from tree species, KALA is absence (0) or presence (1) of mountain laurel, and LIBE is absence (0) or presence (1) of spice bush. This equation has an r^2 of 0.67 and mean square error of 7.7 ft. All variables were highly significant ($p<0.001$). The relationships between observed and predicted upland oak site index for the two models is shown in figure 2. The practical importance of the shrub variables is to reduce upland oak site index predicted from MRI by 5.4 feet if mountain laurel is present or increase site index by 7.4 feet if spice bush is present (fig. 3).

Table 3—Mean and SD of upland oak site index observed on sample plots where either mountain laurel, spice bush, or neither species, was present in the Bent Creek Experimental Forest.

Shrub present	Sample plots	Upland oak site index	
		Mean	SD
	N	- - - - - - -feet - - - - - -	
None	95	66.8	11.2
Mountain laurel	32	58.6[a]	9.4
Spice bush	21	86.4[b]	8.7
All	148	67.8	13.2

[a]Significantly lower compared to plots where the species was not present.

[b]Significantly higher compared to plots where the species was not present.

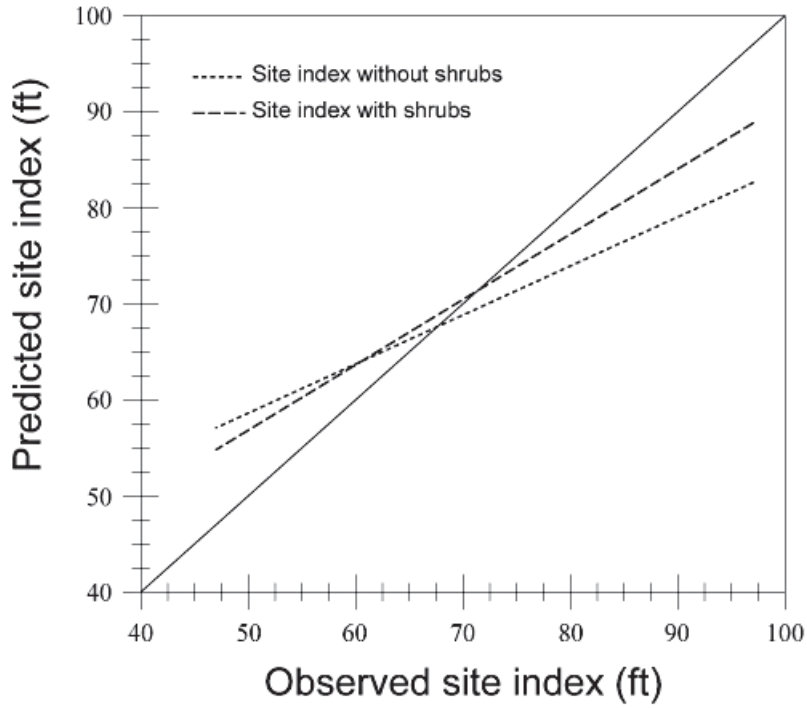

Figure 2—Relationships between observed and predicted upland oak site index for two models: (1) site index as a function of moisture regime index without shrubs (dotted line) and (2) site index as a function of moisture regime index with the occurrence of two shrubs (dashed line). Both models under predict site index on high quality sites (SI > 70) and over predict site index on lower quality sites. The diagonal line represents perfect fit between observed and predicted upland oak site index.

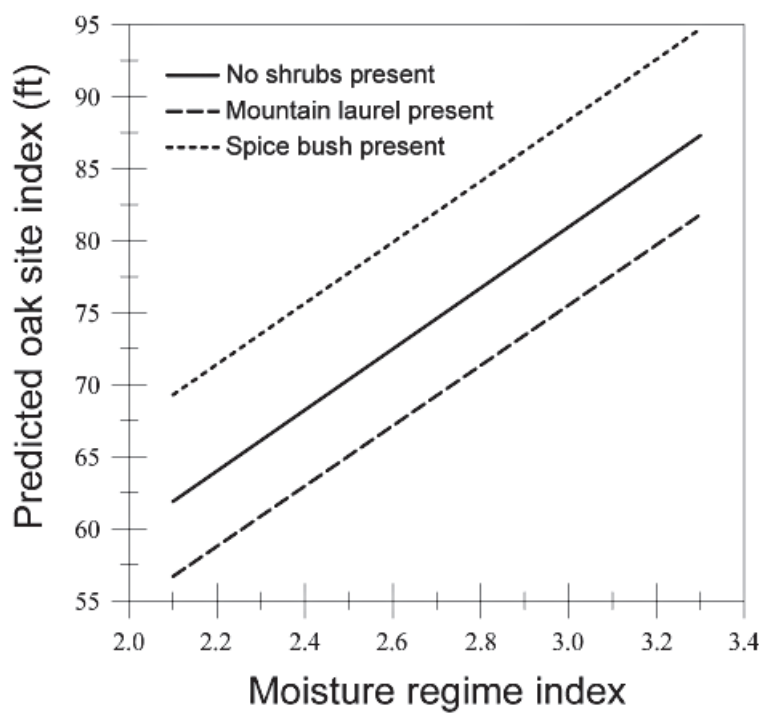

Figure 3—Relationship between moisture regime index and predicted upland oak site index in relation to the presence or absence of two species of shrubs in the Bent Creek Experimental Forest.

Results of this exploratory study in a small watershed suggest that both trees and shrubs are beneficial for estimating site index from species composition. These findings are not surprising because shrubs have different rooting zones compared to trees and may have different environmental requirements for their establishment, growth, and reproduction (Kimmins 1987). Including shrubs with MRI for estimation of site index was an unplanned part of the overall study that resulted from serendipitous observations in the study area. Mountain laurel, normally a dry-site species, occurred as scattered plants in several yellow-poplar stands that had become established on sites with low site index, which were located on moist flood plains. This unusual composition of tree and shrub vegetation provided a clue to the value of non-arborescent data for assessment of site quality. A number of shrub species are common in this region, mostly xerophytic species, although several mesophytic species occur also, such as rosebay rhododendron (*Rhododendron maximum* L.) and sweet shrub (*Calycanthus floridus* L.). Additional study is needed to determine the value of other shrubs for estimation of site quality based on species composition. These results should be applicable within similar temperature regimes of the southern Appalachian Mountains, similar to where my study was conducted.

Application of the method requires several considerations by the resource manager. First, an adequate design must be derived for adequate inventory of the stand. A systematic grid of sample points is recommended similar in intensity to that used for estimation of the timber resources. Next, the tree species list must be completed for the area of application. The lists of trees in table 1 and shrubs in table 2 are limited to species encountered in the study area, which is about a third of the species occurring throughout the southern Appalachian Mountains. Finally, moisture weights assigned to each species must be adjusted for the region in which the method will be applied. The location of some species on the moisture gradient could change somewhat if the area of application is near the limits of the natural range or if there are compensating factors for the area, such as temperature or fertility. For example, Whittaker (1956) assigned northern red oak a value weight of 4.0 for submesic in the overstory. In my area of application, which is somewhat lower and drier, northern red oak was assigned a value of 3.5. As this example illustrates, one strength of the MRI method of site classification is the ease by which it can be adapted and extended to other areas.

In summary, results of this exploratory study demonstrate that at least two species of shrubs may be useful for classification of forest site productivity in the southern Appalachian Mountains. More specifically, I found that upland oak site index is associated with the moisture regime index and the occurrence of either mountain laurel or spice bush. An equation based only on the moisture index accounted for 63 percent of the variation in oak site index on sample plots in the study area. Including the presence of either mountain laurel or spice bush accounted for an additional four percent of variation in site index. Although this system of site classification has been under intermittent development for over ten years, this is the first formal test of the method using nonarborescent species. The relationships with shrubs should be evaluated for application beyond the limited area of this test. Many shrubs occur throughout the Southern Appalachians, and the benefits of using other species for estimation of site index should be tested. Considerable additional work is needed to refine the system and particularly to establish a direct relationship of the moisture regime index with site quality based on volume increment.

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The content of this paper reflects the views of the authors, who are responsible for the facts and accuracy of the information presented herein.

Consistent Definition and Application of Reineke's Stand Density Index in Silviculture and Stand Projection

John D. Shaw[1] and James N. Long[2]

Abstract—Reineke's Stand Density Index (SDI) has been available to silviculturists for over 75 years, but application of this stand metric has been inconsistent. Originally described as a measurement of relative density in single-species, even-aged stands, it has since been generalized for use in uneven-aged stands and mixed-species stands. However, methods used to establish the maximum SDI for various forest types have varied widely. As a result, there are maximum SDI values for some forest types that do not appear to be supported by adequate analysis. This situation has led to confusion and lack of confidence in SDI among some practitioners. We describe several issues related to the determination of maximum SDI, and propose guidelines for future research and application.

Keywords: density management diagram, Forest Vegetation Simulator (FVS), Forest Inventory and Analysis (FIA), SDI, self-thinning, stand development

Introduction

Reineke's Stand Density index (SDI; Reineke 1933) is a useful measure of relative density that, in silvicultural practice today, forms the basis for most density management diagrams (Jack and Long 1996; Newton 1997). SDI is based on the relationship between tree size and number, and the changes in both as a stand develops and self-thins as it matures. As such, SDI and density management are grounded in the so-called -3/2 self thinning law, although the "law" itself was not well-described until the 1950s and 1960s by Japanese researchers (Kira and others 1953; Shinozaki and Kira 1956; Yoda and others 1963). In its original form, the law describes the inverse relationship between the average mass of plants in a population and their number (Zeide 1987):

$$w = kN^{-3/2} \quad [1]$$

where: w is average plant mass,
N is the number of individuals per unit area,
and k is a constant, generally varying by species.

In common forestry application, including SDI, the quadratic mean diameter is substituted for average mass. Although the universality of the law was affirmed by subsequent investigation (White and Harper 1970), it has been a source of controversy among some researchers (Osawa and Sugita 1989; Pretzsch and Biber 2005; Sackville Hamilton and others 1995; Weller 1987; Weller 1990) and the discussion continues today.

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Although many of the issues surrounding the characteristics and use of SDI have been synthesized elsewhere (Shaw 2006), several issues are particularly relevant to the use of SDI in silviculture, stand characterization, and stand projection within the Forest Service. Some of these issues are as fundamental as a lack of agreement on the maximum SDI value that should be used for a particular species or forest type across different Forest Service Regions. Indeed, at this workshop Basford (this proceedings) described how the lack of a suitable maximum SDI led to adoption of a different, although related approach to density management in ponderosa pine forests of Idaho. Guidelines specified a maximum SDI of 830, when the appropriate maximum should not have exceeded 450 (Long and Shaw 2005).

The maximum SDI of 450, which is used in other parts of the Interior West, was supported by a range-wide analysis of ponderosa pine (Long and Shaw 2005). However, the 830 maximum used in Idaho should have been questioned much earlier because it can be traced back to Reineke's (1933) original paper. Close examination of Reineke's graph of ponderosa pine data reveals that the ponderosa pine maximum of 830 was based on fewer than two dozen data points, and in none of the cases did mean diameter exceed 6 inches. Surely, the persistence of this unsupportable value in Forest Service documentation suggests that it was never examined closely. Through experience, practitioners suspected there was a problem, but it was not caught by the research community. As a result, experiences such as Basford's have led to diminished trust in SDI as a useful index, in turn causing some to abandon its use. Because of controversy in the literature and perceived shortcomings in practical use, other practitioners have been hesitant to adopt the use of SDI or density management diagrams (fig. 1).

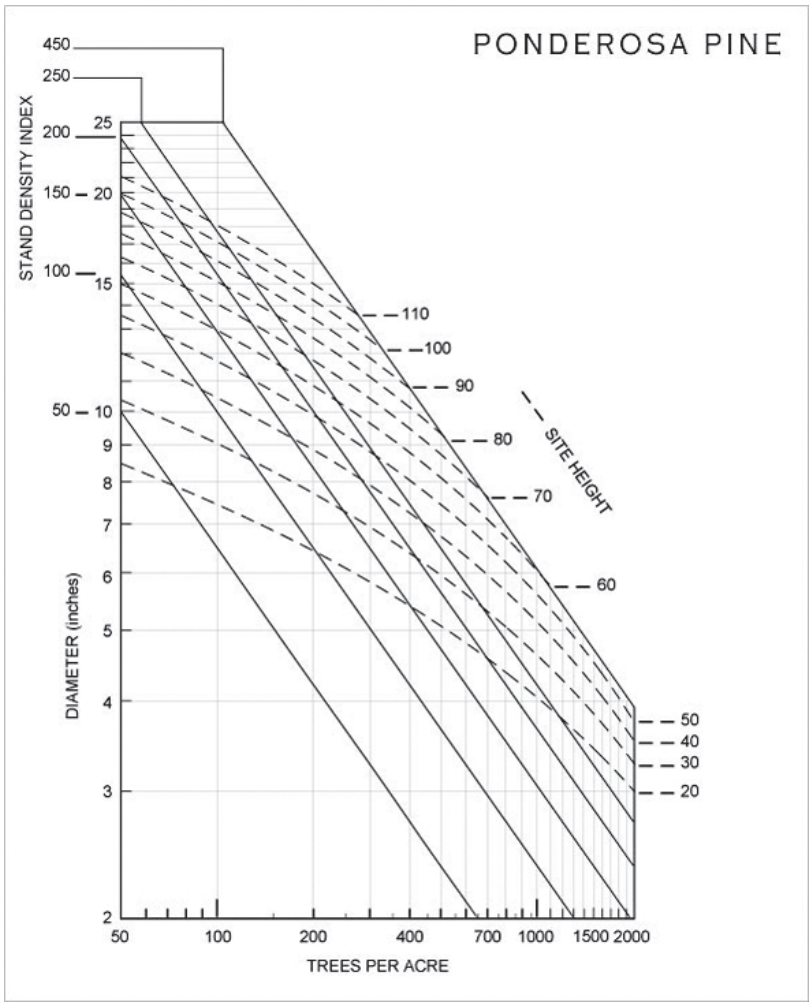

Figure 1—Density management diagram for ponderosa pine. Analysis of data from across the range of ponderosa pine in the western United States supported the use of 450 as the maximum SDI (Long and Shaw 2005). The slope of the maximum SDI line is based on the self-thinning relationship.

However, we have found that many of the "problems" with SDI may be traceable to inadequate data, inconsistent methods, or even the perpetuation of weakly supported results. From the perspective of the silviculturist, these problems represent a failure of the research and development community to address the needs of the practitioner—i.e., a lack of clarity in research leads to confusion in application. In this paper we highlight the most important issues and propose a path to resolution. The need for resolution is driven by the need for consistent definition and use of SDI within the Forest Service:

- Forest Service silviculturists need defensible measures of relative stand density to use as the basis for silvicultural prescriptions; these may be in the form of a maximum SDI for a given forest type or a well-constructed density management diagram. Research that relates certain stand characteristics, such as risk of insect infestation or suitability as wildlife habitat, to SDI, must measure stand density in a way that is consistent with the values used by the silviculturist.
- The Forest Vegetation Simulator (FVS) uses SDI as a driving variable during projections of stand development (Dixon 2003). FVS requires generalized rules for using SDI in models that predict stand response to changing composition, structure, and density. Values used by these models should be consistent, in terms of the data and analysis used, with the methods employed by model users—e.g., silviculturists.
- The Forest Inventory and Analysis (FIA) program has a need to assess and interpret stand characteristics, including relative density, in all forest types of the U.S. Regional inconsistencies in the characterization and use of SDI create difficulties with FIA analysis and reporting. The FIA program needs a sound basis for calculation of relative density.

Although some of these needs apply more broadly than to the Forest Service alone, it is important for the agency to strive for internal consistency among various tools and programs, such as FVS and FIA. Once this consistency is accomplished it will benefit the silviculture community at large.

The Issues

Some of the issues surrounding the use of SDI can be related back to one of the first questions that a silviculturist might ask in the development of a density management regime—"What is the appropriate maximum SDI to use for this stand?" This question requires a definition of the term "maximum SDI," and implies some population to which it applies—i.e., the stand. The establishment of a maximum SDI for a particular forest also establishes the variable $\boldsymbol{k}$ for that type. In our experience, it is necessary to address four important issues in order to achieve consistent definition and application of SDI:

- Sampling methodology and error—stands at maximum SDI are rarely observed in the field, so it is necessary to understand the role of sampling in the estimation of maximum SDI.
- The scale issue—to what entity should a stated maximum SDI apply—the plot, the stand, or something else?
- Assumptions of universal slope and linearity—is the slope of the size-density relationship truly constant when plotted in log-log space, is it nonlinear, or are there other limitations to size-density?
- Lack of consistent definition—is there consistency between the data and methods used to determine maximum SDI and the application of the index in the field?

Sampling Error

Maximum SDI is generally understood to be a hypothetical limit of relative density that is very rarely observed in the field. However, it is common for data to include observations of higher SDI (sometimes much higher) than would be expected for well-studied forest types. These seemingly contradictory data are typically plot-based measurements that sample a relatively small portion of the population in question (i.e., the stand). One reasonable explanation for these "excessive" density observations could simply be that they represent a form of sampling error—i.e., the sample design captures a high-density portion of the stand but does not account for lower-density areas that affect the competitive status of the measured area. This possibility is related to the scale issue, which is discussed in more detail below.

Another possibility is that the methods used to establish maximum SDI for the forest type were different from the methods used to assess the stand at hand. In effect, the researcher who documents maximum SDI and the practitioner who assesses the stand are using different measurement scales. For example, consider a situation where the maximum SDI for a species was determined using plot-level measurements as the sample data. Although various methods may be used to establish the maximum, most methods allow for a small percentage of "outliers"—high-density plots that are considered to be the result of sampling error—and establish the maximum based on the remainder of observations or some subset.

In practice, however, density is assessed at the stand level. As a result, a potential problem arises because plot-level density and stand-level density are not equivalent measures. This can be illustrated with a simple example. Given an acre of forest (representative of a single stand) that is sampled by a series of four plots (fig. 2), SDI can be calculated five times—once for each plot and again at the stand level.

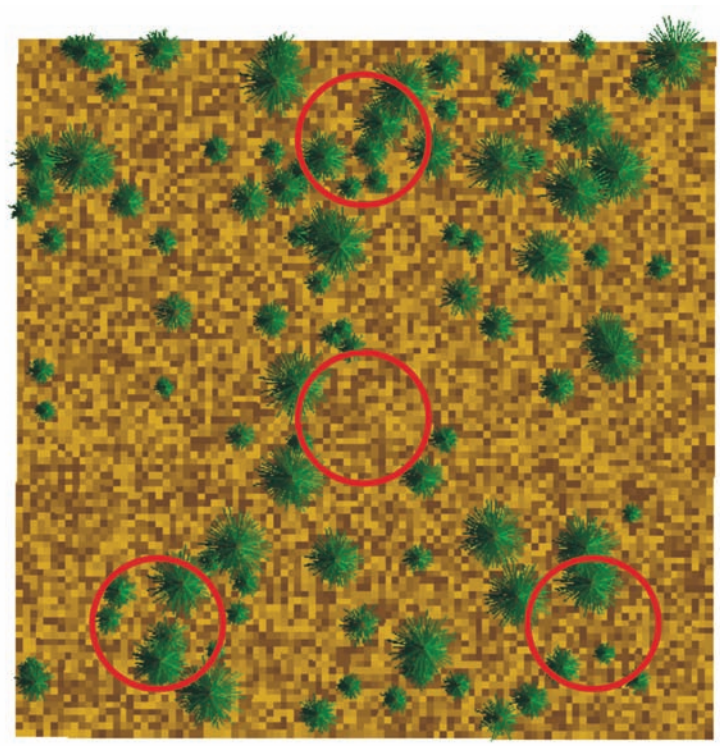

Figure 2—Example of within-stand variability that results in widely varying SDI at the plot level. SDI is 37 percent of maximum at the stand level, but the SDI of individual plots may be nearly double that.

Note that even in a relatively homogeneous stand, there is variability in SDI among the plots. In this example, SDI ranges from 0 to 245 among the plots, with a stand average of 137. Stand-level SDI is always lower than the maximum plot-level SDI, except in the special case that SDI is identical on all plots measured within a stand, because it is an average of the plot-level measurements. If we assume the example stand to be ponderosa pine, then plot-level SDI ranges from 0 percent to 54 percent of maximum SDI for the forest type, with stand-level SDI at just over 30 percent of maximum. From a silvicultural perspective, some parts of the stand have open growing space, some parts are on the verge of self-thinning (which is expected to begin at around 50% of maximum SDI), and the stand, on average, is relatively well-stocked. How then should the silviculturist assess this stand, when within-stand variation suggests a need for regeneration in some areas, thinning in others, and the average stand condition suggests that no intervention is needed? We suggest that the answer partly depends on how well-matched the assessment methods are to the methods used to determine the reference condition (i.e., maximum SDI). This situation changes the issue from one of sampling error (accurately measuring the condition) to one of plot scale (measuring the same thing in research and application).

The Scale Issue

When the scale issue is examined using a large sample of plots and stands, such as would be used to develop maximum SDI values, differences in the characteristics of the sample become apparent. Data from the Forest Inventory and Analysis (FIA) program are used to illustrate the effect of using plot-level vs stand-level data for analysis. The FIA plot design uses four 1/24-acre subplots, arranged as in figure 2. In most cases, all four subplots sample a single "condition," which is equivalent to a stand in most cases. Where multiple conditions (stands) occur across the plot footprint, the changes are mapped and the proportions of plot area occupied by each condition are recorded. For the purpose of this analysis, data from single-condition FIA plots were compiled two ways: 1) treating each subplot as a separate sample, using subplot data as the observation, and 2) averaging all subplots on a plot to represent a stand-level observation. The two estimates represent the exact same sampled area, with the plot-based data (1) having four times the number of observations as the stand-based data (2). The results of this comparison for two common forest types are shown in table 1.

Note that the measures of central tendency are relatively close for each forest type when the plot-level data are compared with stand-level data. However, the discrepancy between the two data sets increases substantially as the upper limits of apparent density are approached in each sample. The magnitude of this discrepancy is extremely important with respect to the estimation of maximum SDI, because

Table 1–Summary statistics for two forest types, using FIA data compiled at the plot level (FIA subplot) and at the stand level (FIA plot).

Sample Statistic for SDI	FIA Forest Type			
	Douglas-fir		Aspen	
	Stand-based	Plot-based	Stand-based	Plot-based
Mean	198	207	220	231
Median	191	186	211	207
Maximum	581	987	639	1683

all of the methods that have been used to estimate the maximum—from Reineke's pencil and ruler to RMA regression (Leduc 1987) to binning methods (Bi 2001; Bi and Turvey 1997)—manipulate the upper limits of the data in order to arrive at an estimate. As a result, it is possible for two studies to sample the same area of one population, apply the same analysis methods, but estimate substantially different maximum SDIs solely due to the sample design.

Assumptions of Universal Slope and Linearity

Another approach to SDI analysis that can be a source of inconsistency involves the determination of the slope of the self-thinning trajectory. As mentioned earlier, this has been a source of debate. However, this debate is more generally focused on the merits of the -3/2 self-thinning law as opposed to Reineke's SDI in particular (although the two are closely linked). The slope representing constant SDI is fixed by definition as reflected in the equation used to calculate it (equation 2):

$$SDI = TPA \bullet \left(\frac{Dq}{10}\right)^{1.6} \qquad [2]$$

where SDI is stand density index,
Dq is quadratic mean stand diameter in inches at breast height,
TPA is the number of trees per acre,
10 is the reference diameter in inches, and
1.6 is the slope factor.

In our view, it is necessary to consider the characteristics of slope as two separate issues: 1) the self-thinning trajectory based on a universal slope factor, and 2) the upper boundary of the size-density relationship as indicated by observed stand conditions. We separate these characteristics on the basis that the former may be a manifestation of density-dependent mortality, whereas the latter may represent a combination of density-dependent and density-independent stand dynamics. This is an important distinction, because with few exceptions (e.g., Pretsch and Biber 2005), most analyses of the size-density relationship are based on one-time observational data, and not on long-term repeat measurements. As a result, many SDI analyses employ space-for-time substitution.

The issue of a universal slope factor (i.e., the exponent that represents the self-thinning trajectory is always 1.6) will largely be avoided in this discussion for three reasons: 1) it can be argued that a slope factor of 1.6 is inherent in Reineke's (1933) definition of SDI, and therefore immutable; 2) like the maximum SDI example above, it can be shown that some differences in slope estimation are possible solely due to the characteristics of the data used in the analysis (e.g., finding different slopes for two populations of the same forest type, one of which consists of relatively young stands and the other of which includes a broad range of age classes); and 3) although the self-thinning trajectory is generally assumed to be in effect throughout stand development, broad-scale data suggest that this assumption may be incorrect (Shaw and Long 2007). For the sake of further discussion, we will assume that the "self-thinning law" is indeed a law, but we also consider the possibility that it may not be in operation at all stages of stand development or under all stand conditions (e.g., in stands at low relative densities or experiencing non-density-related mortality).

Reasons #2 and #3, then, are related to the observation that most stands fail to maintain the expected maximum density in a mature state. This situation may be more common than is currently appreciated. FIA data collected over a wide range of stand conditions have shown that relative density begins to fall off in many forest types after quadratic mean diameter reaches about 10 inches (fig. 3, left panels). Reineke (1933) suggested that most species conformed to the "universal"

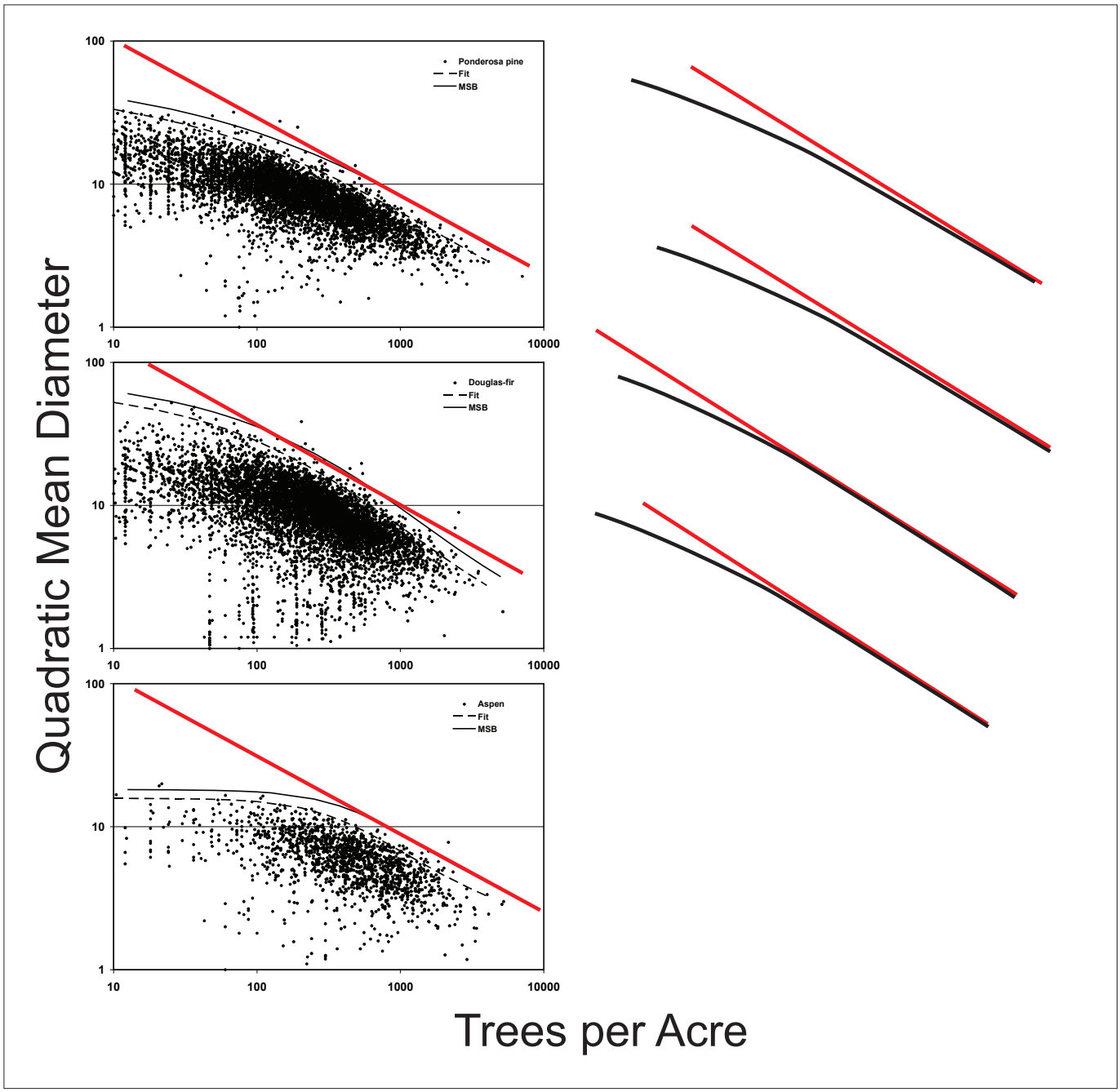

Figure 3—Size-density data showing relative density fall-off for several forest types. Left panels are Forest Inventory and Analysis data for ponderosa pine, Douglas-fir, and aspen. Solid black lines represent the approximate limit of size-density observed in natural stands, described by Shaw and Long (2007) as the mature stand boundary. Red lines represent the expected self-thinning line as defined by SDI. The right panel is taken from figure 7 in Reineke (1933), where the image has been rotated and mirrored to match the orientation of the left-side graphs. From top to bottom the graphs show data from shortleaf pine, slash pine, longleaf pine, loblolly pine from temporary plot, and loblolly pine from remeasured plots. Reineke only acknowledged divergence from the expected slope in the shortleaf pine and slash pine data (thin black lines); longleaf pine and loblolly pine data were considered to conform to the common self-thinning slope. However, in this orientation it appears that relative density falls off when quadratic mean diameter is greater than 10 inches (thick black lines) for loblolly and longleaf pines.

self-thinning trajectory, but acknowledged that data for a few species showed some degree of divergence (fig. 3, right panels). He interpreted this divergence by presenting an altered linear slope. However, when nonlinear curves are applied to the frontier of Reineke's data, it is apparent that the data he used revealed the same fall-off in relative density seen in FIA data. Reineke's (1933) data for longleaf pine therefore appear to be consistent with the mature stand boundary described by Shaw and Long (2007).

The evidence for fall-off of relative density suggests that the assumption of linearity over the entire range of size-density is incorrect. A more precise characterization would be to say that density-dependent mortality (self-thinning) may only be in effect during a certain period of stand development, and that there is a transition point where other factors limit relative density. One hypothesis for the cause of the mature stand boundary is that the mortality rate of large trees exceeds the capacity of the residual stand to capture the available growing space. This characterization of relative density transition is compatible with the Oliver and Larson (1990) stand development model, with maximum SDI limiting stand density during the stem exclusion stage and the mature stand boundary occurring during the understory re-initiation and old-growth stages.

Lack of Consistent Definitions

Another question that might be asked by the silviculturist is: What SDI should be applied in the case of irregularly structured or mixed-species stands? Because the title of Reineke's (1933) paper specified "even-aged forests," the purist approach suggests that the index is inappropriate for use in irregularly structured stands. However, Stage (1968) illustrated how SDI could be partitioned, and this approach was further developed for irregularly structured stands by Long and Daniel (1990). More recent literature on the subject (Ducey 2009; Ducey and Larson 2003; Shaw 2000) has dealt with the implications of calculating SDI by summation (equation 3), so there appears to be adequate information available to guide practitioners on how to use SDI with irregular stand structures. The alternative formulation of SDI in equation [3] therefore provides a more flexible application of the index, and avoids some issues related to the comparison of relative density in even-aged and irregularly structured stands (Shaw 2006).

$$SDI_{sum} = \sum\left(TPA_i \cdot \left(\frac{D_i}{10}\right)^{1.6}\right) \qquad [3]$$

where Di is the breast height diameter of the ith tally tree on the plot and
TPAi is the number of trees per acre represented by the ith tree.
10 is the reference diameter in inches, and
1.6 is the slope factor.

Less attention has been paid to the question of species mixtures, although various approaches to weighting maximum SDI by stand composition have been discussed by Puettmann and others (1992), Dean and Baldwin (1996), Torres-Rojo and Martinez (2000), and Woodall and others (2005). Although the approaches vary, these studies and examination of FIA data indicate that composition is a factor in the determination of potential maximum SDI for a stand. All of these studies suggest that the appropriate "adjustment" for SDI in mixed-species stands should be based on weighting of the individual species maximum SDI, as determined through analysis of pure stands, by the relative composition of each species in the mixed stand. In other words, the expected maximum SDI for a 50:50 mixture of two species should be approximately the average of the individual species maxima. If such a stand is a candidate for thinning and one species will become

more dominant than the other after treatment, then a new maximum should be calculated for the resulting stand based on the post-thinning proportions.

The questions about effects of structure and mixtures are relevant to the issue of consistency in the development and use of SDI. Most studies on SDI are silent with respect to the ranges of composition and structure that are represented in the analyzed data. Because both stand characteristics have an effect on SDI calculations, they should be considered in research used to estimate maximum SDI or develop tools such as density management diagrams, and they should be considered by the silviculturist when designing density management regimes. As with the plot scale issue, problems may be caused for SDI users simply because of a lack of stated or consistent definitions as part of research methods. For example, if two studies are designed to determine the maximum SDI for a species, but one uses data largely acquired from pure stands (e.g., >90% composition of the target species) and the other uses data acquired from stands where the species is merely dominant (e.g., >50% composition of the target species), there is a high likelihood that they will find two different SDI maxima for the same species. Without explicit definition of the analyzed population, users may be confused by these results.

Proposals for Future Research and Application

In the discussion above we describe four factors that can cause inconsistencies between values of SDI as determined in research studies and field application: 1) sampling error, 2) scale at which the data are obtained (plot scale), 3) assumption of linearity of the self-thinning dynamic during advanced stages of stand development, and 4) lack of consistent definitions or a mismatch between the population used to develop SDI-related values and the population being assessed in management practice. We will not discuss sampling error here, because it is an inevitable artifact of estimation and a minor issue in comparison to the others. We believe that the remainder of the issues can be resolved through the adoption of a few basic guidelines for use during research and field application of SDI.

Researchers should ensure the data used in SDI-related analyses are compatible with the scale expected to be used by the practitioner when the results of research are applied in the field. Methods or recommendations for application should state the appropriate scale of application. This is not to say that data of different scale are not useful in the research process. Small-scale (plot-level) data may be useful to determine the biological capacity of a species for packing on the site, whereas larger scale (stand-level) data are more likely to describe the range and variability of density that would be measured at management scales. Both scales of information might be useful to the silviculturist, who might simultaneously seek to manage for dense pockets and more moderate stand-level density. In addition to matching of source data and application scales, it is important to define the population to which the results apply. The population of interest is commonly thought of in terms of geographic extent, but the description should include both the range of composition and stand structure. Scale, composition, and structure have the potential to affect determination of maximum SDI, and it is likely that one or more of these characteristics have been a factor in cases where there are apparent regional differences among maximum SDI values. If the population of interest is adequately identified, then apparent regional differences should be minimal.

The upper boundary of the size-density relationship is complex, so there is little to be gained through continued evaluation of the self-thinning slope where a linear relationship is assumed. Instead, research should focus on the mature stand boundary, including mechanisms underlying the fall-off phenomenon and stand

dynamics when stem exclusion is not the driving factor. Increasing knowledge in this area should substantially improve the realism of stand projection models such as the Forest Vegetation Simulator.

Finally, a definition of maximum SDI should be consistent with "benchmark" percentages of the maximum that correspond to canopy closure, the onset of competition, and the zone of imminent competition mortality (Drew and Flewelling 1977). This compatibility is required in order for density management regimes to produce desired results. If density is scaled incorrectly in tools and growth models, then silviculturists risk missing management objectives because density may inadvertently be managed higher or lower than the density target would indicate. This is the case that led Basford (this proceedings) to abandon SDI in favor of an alternative. Anyone attempting to manage ponderosa pine using a maximum SDI of 830, when 450 is the appropriate maximum, is likely to fail to meet their objectives because the true relative density (and by extension, the state of competition) in the managed stands would always be much higher than suggested by the SDI percentage.

Conclusion

In this paper we have attempted to highlight inconsistencies in research and application of SDI, and suggested opportunities for resolution of the issues causing the most confusion when applying the index in silvicultural practice. Most of these issues may be resolved simply by full disclosure of the data and methods used for analysis, coupled with use by the practitioner that is consistent with the results. In some cases where inconsistencies have been found, they may be traced back to weakly supported conclusions. For some reason, an incorrect maximum SDI for one of the most important timber species in the West, ponderosa pine, went unquestioned for over 75 years. This might not be surprising, considering the foundational nature of Reineke's work, but the persistence of this unsupportable value in Forest Service documentation suggests that it was never examined closely. We believe that situations like this can be avoided in the future, if SDI research and application are conducted with the adoption of a few simple guidelines:

- Density management guidelines should be developed using spatial scales comparable to what would be used in application.
- Research documentation should explicitly describe the population from which the data were drawn, in terms of composition and structure, so that practitioners understand the range of conditions represented in the analysis.
- Definitions of terms used in research should be consistent with accepted definitions, or explicitly defined in cases where they are open to interpretation.

Adoption of standards and guidelines such as these, especially in cases where the same concept may be described in many different geographic areas and forest types using multiple methods, should do much to avoid confusion during technology transfer from researcher to the practitioner.

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The Role of Strategic Forest Inventories in Aiding Land Management Decision Making: Examples From the U.S. Forest Inventory and Analysis Program

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Abstract—A major challenge for today's resource managers is the linking of stand- and landscape-scale dynamics. The U.S. Forest Service has made major investments in programs at both the stand- (national forest project) and landscape/regional (Forest Inventory and Analysis [FIA] program) levels. FIA produces the only comprehensive and consistent statistical information on the status, changes, and trends in the condition and health of all forest ecosystems in the USA. Intended to be a strategic inventory, FIA data have not been used very much for small area inventory, planning, and analysis due to the high variation associated with the estimates. Yet, trends observed over landscape and regional scales can help managers making decisions at the local level. FIA data can be used to assist with project-level decision making, adding scientifically defensible data and framing local management in a larger context. FIA data are helpful in understanding stocking and density limitations and inter-species interactions. They also can provide insight into future growth and yield. FIA data provide opportunities to conduct scale-independent analyses to examine relationships between stand characteristics and forest health and invasive species, as well as methods for establishing ecological benchmarks and prioritizing restoration opportunities.

This paper explains how FIA data can be used at three different levels of analysis. At the project level, FIA data can be used for Forest Plan revision, monitoring conditions and trends at mid- to broad spatial scales over time, and setting the context for proposed projects. One step beyond management decision making, FIA data can provide input for stocking guides and other quantitative tools that aid silvicultural planning. Finally, FIA data can help detect and analyze patterns across broader geographic areas that reveal (or illustrate) relationships and processes that can be applied at the local level or provide policy guidelines that can guide prioritization and allocation of scarce management resources.

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Introduction

The original objective of the Forest Inventory and Analysis Program was to estimate resource availability at the strategic or state-level as an aid to understanding trends in resource utilization potential. Early inventories were referred to as "Forest Survey" and focused on growing stock volume and productive timberland area, both reflecting an emphasis on the potential for wood utilization. With the evolution of our nation's attitudes toward the environment and the role of forest resources in providing ecosystem services, the purview of the "Forest Survey" has expanded to include not only productive and accessible timberland, but also (protected) forest land reserved from timber harvest and forest land that may not be considered productive enough to grow crops of timber. Additionally, selected

criteria and indicators of ecological sustainability, such as estimates of coarse woody debris, understory vegetation, and lichens, are now incorporated into the sample design. A robust forest inventory requires a large network of sample plots measured consistently over time. Over the past decade, the USDA Forest Service, Forest Inventory and Analysis (FIA) program has implemented an inventory system that seeks to achieve national and international consistency[4]. This system utilizes three phases of inventory designed to make estimates of forest extent, composition, structure, health, and sustainability: Phase 1 – remote sensing, Phase 2 – the actual on-the-ground sampling in a systematic grid, and Phase 3 – a subset of Phase 2 that examines certain variables that are considered to be indicators of forest health (McRoberts 1999). The design of the FIA program gives greater certitude to estimates across larger geographic extents. While FIA does publish estimates of county-level attributes, the uncertainty at such scales often is high, especially in sparsely-forested locales. The paucity of plots for small geographic areas and elements of stratification by ownership or forest type can limit usefulness at the local level. Nonetheless, FIA data can still provide great information value to forest managers.

The potential benefits of FIA data for forest resource managers exist at three levels that are loosely represented by their distance from the project-level, on-the-ground decision-making process. At the most immediate level is the use of FIA data for formulating or updating land management plans for large tracts, monitoring forest conditions and trends at the broad level over time, and setting the context for proposed projects. We present three examples: monitoring trends of target species or forest types, managing snag habitat, and estimating the extent of old-growth forest.

Further removed from the management decision, FIA data provide input for stocking guides and other quantitative tools that aid silvicultural planning. In the past, many of these tools were developed using limited data, and they were regional in scope. As a result, some tools have been used in areas outside those for which they were developed, perhaps without local validation. The fact that FIA data are geographically unbiased and represent a range of conditions that exist within a given forest type in proportion to the abundance of those conditions on the landscape permits regional comparisons of modeling results and geographically appropriate application of silvicultural tools. In addition, the data permit analysis and generalizations of stand dynamics and growth-growing stock relationships that are not possible or appropriate with more limited data sets. To illustrate this point, we provide an example of using FIA data to develop density management diagrams.

Finally, FIA data can detect and analyze patterns at large geographic extents that reveal relationships and processes that can be applied at the local level. The data also can inform policy guidelines for prioritizing and allocating scarce management resources. We present two examples of this capability, one that provides guidance in prioritizing restoration activities by comparing FIA data to historic estimates of structure and abundance, and the other that examines the extent of non-native invasive plants in forest land and the factors that appear to influence their presence. At all three scales, decision makers will find FIA data to be a valuable information resource.

[4] Complete documentation of the plot design and all measurements is at http://socrates.lv-hrc.nevada.edu/fia/dab/databandindex.html.

Use of FIA Data for Project Level Planning

Statistically sound estimates of the current condition and trends of vegetation and associated attributes are fundamental to developing forest plan components, monitoring in relation to forest plan standards, guidelines, and desired conditions, and managing wildlife habitat, including cumulative effects analysis for project-level planning. Furthermore, statistically reliable data are needed to address controversial management issues, such as climate change effects and carbon accounting.

FIA data allow for regional monitoring based on an unbiased, representative sample of forest lands that are remeasured regularly. Many attributes of trees and the site are collected on an FIA plot. The FIA sampling frame uniformly covers all forested lands, regardless of ownership status or management emphasis; thus, wilderness and roadless areas, as well as more intensively managed lands, have equivalent sampling probabilities. As a result, spatial data sets can be intersected with FIA plot locations to describe vegetation characteristics within various map strata. This section contains 3 examples that incorporate these capabilities.

Example 1 — Identifying Target Species in Silvicultural Prescriptions

Within a national forest's planning area, FIA data can be used to assess current condition. Since plots are remeasured on a 5- to 10-year cycle (depending upon geographic area), progress towards desired condition or compliance with achievement standards can be monitored. Furthermore, comparing current condition to future condition allows prioritizing project-level vegetation treatments needed to achieve management objectives. For example, FIA data can be used to compare current estimates to desired conditions for the historically most-common forest types found on the Idaho Panhandle National Forests (IPNF) in the Northern Region (Region 1, which covers Montana and parts of Idaho and North Dakota). The comparison suggests that projects that encourage restoration and regeneration of specific forest types, such as white pine, western larch, and ponderosa pine, should take priority other forest types.

Example 2 — Managing Snag Habitat

In another example, the Northern Region analyzed snag densities for planning development of project-level snag retention and recruitment options. FIA data were used to assess the density and distribution of snags within and outside of wilderness/roadless areas and categorized by vegetation classifications, such as habitat type groups and seral stages (table 1). This analysis took into consideration recent findings on the effects that timber harvest and human access have on snag density, how snag density relates to stand succession and disturbances, and the spatial pattern of snags. After obtaining these results, all national forests in the Northern Region began to monitor snag densities over time at the broad level and use the data to adaptively manage at the project-level.

Example 3 — Managing Old Growth

Additionally, the Northern Region uses FIA data to estimate and monitor forest plan standards such as the percentage of a national forest in old growth status (table 2). The Region has a documented definition of old growth based on geographic area, old growth forest type, and habitat type groups that are applied to FIA inventory data. Using FIA data, one can estimate that 9.4 percent of the Clearwater National Forest (CNF) is in an old growth condition; an estimate with a

Table 1—Mean snag densities per acre with 90% confidence interval, by diameter classes, inside and outside of wilderness/roadless areas for all northern Idaho Forests and for each Forest.

Area	Wilderness/ Roadless	Snags per acre 10"+			Snags per acre 15"+			Snags per acre 20"+			Total Number PSUs	Number Forester PSUs
		Mean	90% CI - Lower bound	90% CI - Upper bound	Mean	90% CI - Lower bound	90% CI - Upper bound	Mean	90% CI - Lower bound	90% CI - Upper bound		
North Idaho Forests	IN	10.3	9.3	11.4	3.9	3.4	4.4	1.6	1.4	1.9	514	514
Idaho Panhandle		10.4	8.4	12.6	3.7	2.8	4.7	1.6	1.1	2.1	133	133
Clearwater		8.9	7.3	10.5	3.7	2.9	4.5	1.6	1.2	2.1	189	189
Nez Perce		11.7	9.9	13.7	4.2	3.3	5.2	1.7	1.3	2.1	192	192
North Idaho Forests	OUT	11.7	10.5	12.9	4.3	3.7	4.9	1.6	1.4	1.9	478	478
Idaho Panhandle		12.7	11.1	14.5	4.2	3.5	4.9	1.4	1.1	1.7	260	260
Clearwater		9.6	7.5	11.9	4.4	3.3	5.7	2.1	1.4	2.8	106	106
Nez Perce		11.2	8.6	14.0	4.3	3.0	5.9	1.8	1.2	2.4	112	112

Table 2—Northern Region and individual National Forest estimates of percent of old growth and 90%-confidence intervals.

Unit	Percent old growth estimate	90 percent confidence interval—Lower bound	90 percent confidence interval—Upper bound	Total number of PSUs	Number of forested PSUs
Northern Region	13.7	12.9	14.4	3883	3423
Beaverhead—Deerlodge	22.9	20.5	25.4	547	442
Bitterroot	12.8	10.1	15.6	252	226
Idaho Panhandle	11.8	9.6	14.0	413	397
Clearwater	9.4	7.3	11.8	305	300

90 percent confidence interval and a lower bound of 7.3 percent. The CNF's Forest Plan Standard is to retain at least 10 percent old growth forest-wide. Currently, the CNF is not proposing any treatments that impact old growth stands or stands approaching old-growth status. When FIA annual data show an increase in the amount of old growth on the Forest, the Forest will reconsider its current position.

FIA data can be associated with various map products to describe and explore various map strata. For example, FIA inventory data were analyzed to determine whether or not the Region's old growth criteria were met. Forests collapse their existing vegetation layers into several cover types. The percentage of plots within each cover type, which meet old growth criteria can then be extracted and compared (fig. 1).

Use of FIA Data for Development of Silvicultural Tools

Although the FIA inventory system was designed to estimate population characteristics and trends across broad geographic areas, plot-level FIA data are increasingly being used to describe and model stand growth and structural characteristics. This use of FIA data is partly driven by the fact that FIA reporting, by design, requires wall-to-wall coverage (i.e., all forest types in all ownerships

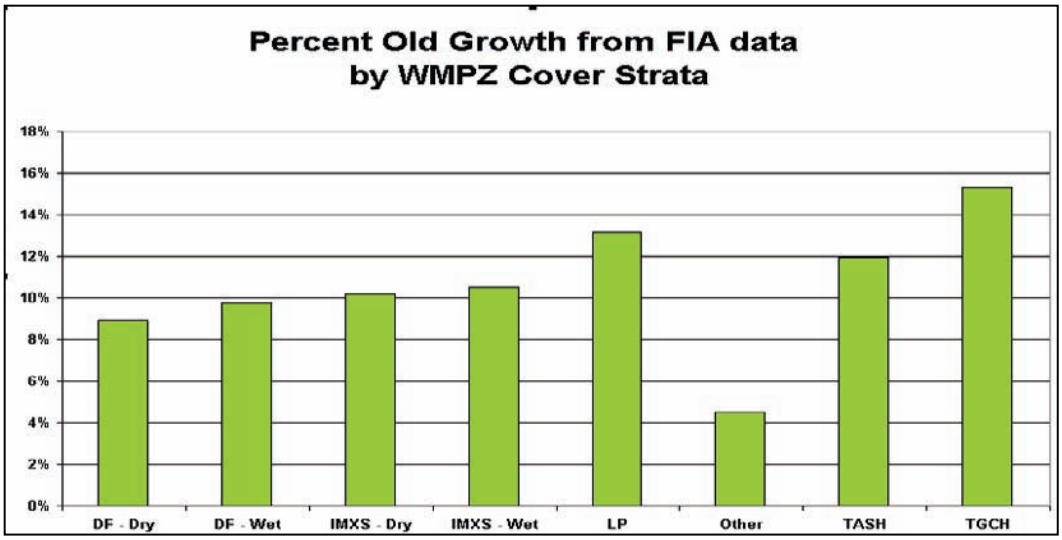

Figure 1—Percent of old growth, based on FIA data, by Western Montana Planning Zone (WPMZ) Cover Strata.

under all management scenarios). Unfortunately, some of the most basic tools and models, such as stocking charts, yield equations, and site index curves, have been developed locally using relatively small data sets. Often, the geographic extent is not well documented. Because of the extensive demand for these tools, the FIA program applies these models over geographic areas that may be greater than appropriate for the scope of the original research. This extrapolation may occur without substantial validation, because the process for adopting models typically relies on expert opinion of model developers, or in some cases, model users.

Because of this situation, the FIA program has relied largely on third parties (such as non-FIA Forest Service and university-based researchers) to develop the tools and models underlying many FIA variables. Problems arise where there are large knowledge gaps, such as for species that are less studied because they are not commercially important in all or part of their range. These gaps tend to persist for a long time because there has generally been no systematic mechanism available to fill them. Until recently, the FIA program had little capacity to develop new models and so used the "best available" tools and models, based on limited testing and expert opinion. As a result, many "surrogate" models persist in the FIA computation system.

However, with the advent of the current, nationally consistent, mapped-plot design, the FIA program began compiling data that can be used to validate existing tools and models or to develop new ones from scratch. Plot-level FIA data can be compiled to describe characteristics and conditions, such as stand structure and composition, individual tree- and stand-level growth, and mortality rates. In comparison to the studies that produced many models and tools in use today, data developed on FIA plots offer several potential advantages.

First, many older studies were geographically limited, or one species may have been represented by multiple studies across its range, with differing experimental procedures and analysis being used among studies. FIA data cover the entire ranges of species, or at least their range within the boundaries of the United States, and sample all parts of the range with one core protocol. As a result, the data are consistent across large geographic areas. Sub-regions of a species' range can be compared without the introduction of noise from varying methods.

Second, older studies were frequently based on relatively small sample sizes, usually confined to one geographic area, and with unknown geographic bias. For common species and forest types, FIA plots available for analysis typically number in the thousands, depending on the criteria imposed by the research question. FIA data are geographically unbiased because FIA uses a systematic sampling design.

Finally, many older studies relied on temporary plot data. FIA data from the legacy periodic inventories and initial measurement of annual inventory plots are also treated as temporary plot data for many current analyses. However, annual inventory plots are permanent and will be remeasured on a 5- to 10-year cycle, depending on the state (Bechtold and Patterson 2005). The annual inventories eventually will yield long-term data on growth, mortality, successional change, and other tree- and stand-level characteristics. These data can serve as validation data for existing models, or time-series data for the development of new models.

Although the points above are broad generalities, they highlight some of the potential advantages of using plot-level FIA data in a variety of analyses. Of course, the data collected on FIA plots cannot satisfy every research question, so the need for "experimentally based" research will always exist. What the FIA program offers is a well documented, statistically sound sample design that produces the kind of data used in many observational studies, without the pitfalls associated with ad-hoc selection of plots. In the longer term, FIA data will produce valuable time-series data with unparalleled geographic scope.

Example 4 — FIA Data and Development of Density Management Diagrams

An example of the broad applicability of FIA data is the development of density management diagrams for ponderosa and longleaf pines (Long and Shaw 2005; Shaw and Long 2007). Density management diagrams relate yield and density and allow forest managers to use current stand density to project what the future stand would look like (Kershaw and Fischer 1991). Both of these species are commercially and ecologically important, and many aspects of their ecology and associated silvicultural practices have been studied. Density management diagrams, which are graphic models of stand structure and development (Jack and Long 1996; Newton 1997), are commonly used in the western U.S. and much of Canada. They are less commonly used in eastern states, but availability and use there is gradually increasing.

The lack of a density management diagram (DMD) for ponderosa pine represented a major knowledge gap for that species. Even the maximum relative density of ponderosa pine, expressed as stand density index (Reineke 1933), was not well understood across the species' range. The need for a ponderosa pine DMD and the availability of FIA data presented an opportunity to create a "test case." One aspect of this test would be the use of "off-the-shelf" FIA data — i.e., the data that are freely available to the public through the FIA Datamart (http://fiatools.fs.fed.us/fiadb-downloads/datamart.html). Given this approach, it would be possible for users of the DMD to independently reproduce the results of the research using publicly available data and the methods described by Long and Shaw (2005). This feature would add a level of transparency to the study that is relatively rare in research today.

The number of plots needed to construct a DMD is about 300 (J.N. Long, pers. comm.). FIA data provided 766 plots (out of 8,183 plots with ponderosa pine) for development of the ponderosa pine DMD based on compositional and structural criteria. The DMD appeared to be robust across several sub-regions of the range of ponderosa pine, suggesting that the tool can be used over a large geographic area without concern for local bias (Long and Shaw 2005). Although the data used

in construction of the diagram were necessarily treated as temporary plot data, many of the plots used in the original analysis will be remeasured in the future, providing data that can be used for periodic validation of the stand dynamics represented in the DMD.

From a management perspective, the lack of a DMD for longleaf pine did not represent as great an unmet need as did the lack of a DMD for ponderosa pine. However, two issues related to longleaf pine could be addressed with the development of a DMD. The first was an apparent fall-off in the relative density of mature stands that resulted in gross over-prediction of potential basal area and volume when a full-stocking scenario was extended beyond a certain limit. The second was the need for tools that would assist silviculturists in the development of treatments for the benefit of the endangered Red-cockaded Woodpecker (RCW *Picoides borealis*). Although management goals for the RCW were well established by recovery guidelines (U.S. Fish and Wildlife Service 2003), the fall-off issue in mature stands was not well understood. Were the apparent limits of relative density that were derived from local data universal for the longleaf forest type or were they only local?

Again, the broad-ranging availability of FIA data was able to provide a defensible answer. The FIA database yielded 5,222 plots with longleaf pine present, leaving 343 available for development of the DMD after applying somewhat stricter compositional and structural evaluation criteria than were used in the case of ponderosa pine. A somewhat larger pool of plots was used to characterize the "fall-off" phenomenon, which appeared to be common to the longleaf forest type throughout its range. As a result, the longleaf pine DMD was published with the inclusion of a "mature stand boundary" (fig. 2) that represented the practical upper limit of stand-level management in size-density space (Shaw and Long 2007). Finally, FIA data were used to analyze the relationships among mean stand diameter, stand basal area, and cumulative basal area by diameter class. These relationships were used to translate stand structural characteristics specified in the RCW recovery guidelines (U.S. Fish and Wildlife Service 2003) into a "suitable habitat zone" on the DMD (fig. 2). Addition of the mature stand boundary and the RCW habitat suitability zone considerably enhance the utility of the basic longleaf pine DMD.

Although the construction of density management diagrams has been used to illustrate the usefulness of FIA data for development of new tools, the list of potential applications is long. There are many species and forest types for which DMDs are in demand; DMDs for the Sierra mixed conifer and aspen forest types are under development using FIA data, and several more are planned. In addition, FIA data are being brought to bear on a variety of questions related to forest growth and yield and stand dynamics. In contrast to past studies that may have been limited in scope and developed from data collected at one point in time, it will be possible to establish a cycle of development, validation, and revision as the flow of annual FIA data continues.

Use of FIA Data at the Landscape Level

Because of the extent of the FIA inventory design, at least 1 plot per 6,000 acres, patterns and trends that are not obvious at a project level might reveal themselves at a larger scale. Some of the questions are explicitly a function of this landscape view, like the restoration example below. Others merely need a large number of data points to examine specific cause-and-effect relationships, like the invasive plant example that follows.

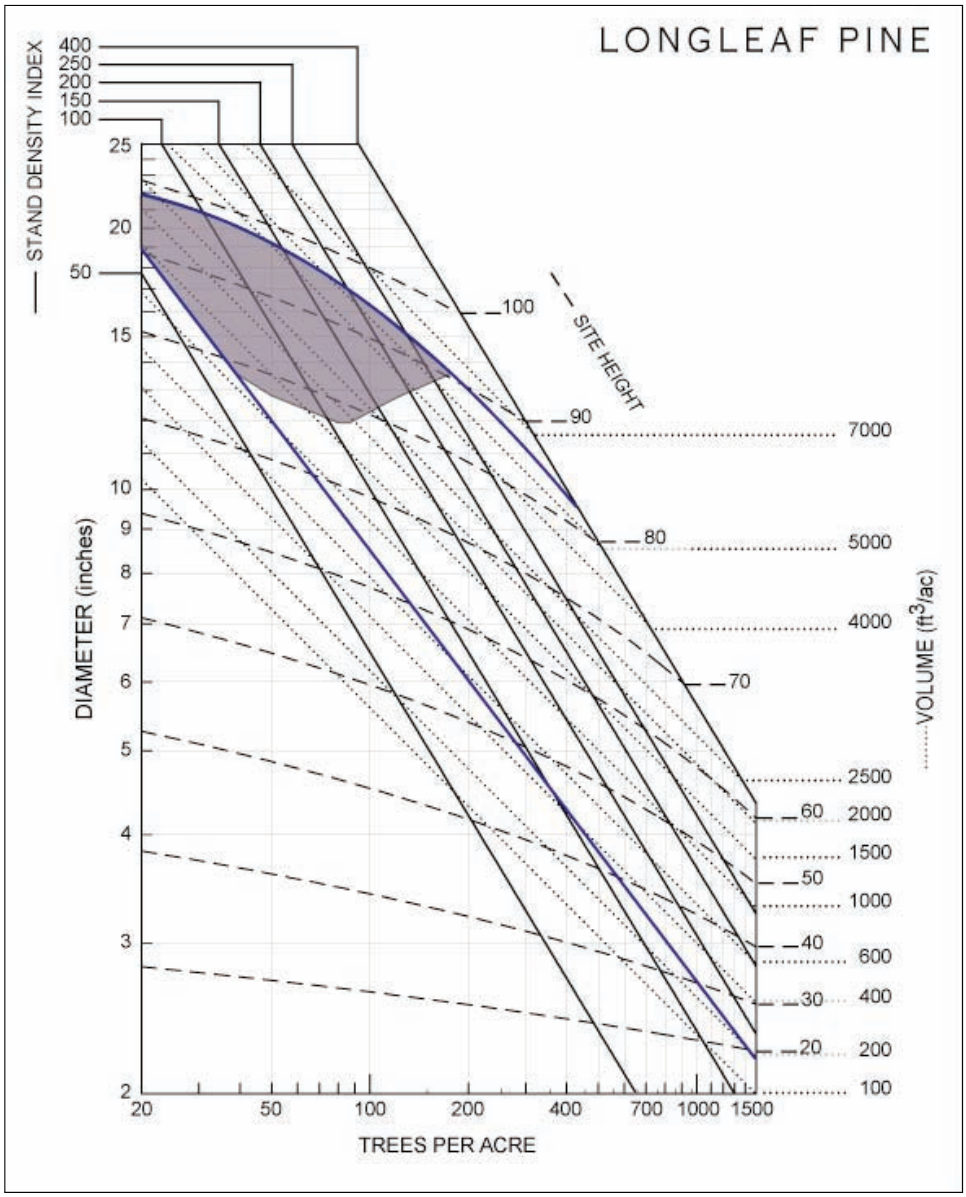

Figure 2—Density management diagram for longleaf pine, with the mature stand boundary (in blue) and zone of Red-cockaded Woodpecker habitat suitability (shaded polygon) (based on Shaw and Long 2007).

Example 5 — Restoration Ecology

As a counterpoint to the highly altered landscapes of today, some ecologists and resource professionals trying to establish criteria for sustainability have pointed to the pre-Euro-American-settlement landscape as a benchmark for restoration efforts (Bragg 2003; Foti 2004; Landres and others 1999; Swetnam and others 1999). To prioritize restoration efforts, a robust method for comparing current and historical landscapes would be extremely useful. In the western two-thirds of the United States, public land managers are fortunate to have an historical inventory in the Public Land Survey (PLS) and the FIA Program's inventory of current forest conditions.

The U.S. government's General Land Office (GLO) conducted public land surveys (PLS) across most of the country during the 19th century. Most of the lands west of the 13 original colonies were subject to the GLO surveys. These public records are an excellent source of historical landscape conditions when properly interpreted. The PLS is a rectangular, rule-based system that divided the landscape into a series of townships and ranges associated with a point of origin and a series of meridians and baselines (usually by state). These north-south and east-west running demarcations divided the land into nominal 36 mi^2 townships, which were then further subdivided into 36 640-acre sections (National Atlas 2006). These townships were replicated across the landscape. Along this hierarchical grid, survey markers (posts) were set at 1/2 mile and 1 mile intervals. At these locations, information such as species, estimated diameter, and distance were collected on two to four trees near the posts and recorded in survey notebooks. Despite significant deficiencies (e.g., a not particularly intensive sampling regime, and numerous biases, ambiguities, and inconsistencies [Bourdo 1956; Bragg 2004; Mladenoff and others 2002; Nelson 1997; Schulte and Mladenoff 2001]), the PLS records still provide landscape-to-regional information on the vegetation of the period due to their detail, wide extent, and resolution.

To test the effectiveness of a system comparing PLS to FIA data, Moser and others (2006a) compared historic and current data covering the southeast Missouri Ozarks. Missouri was surveyed by General Land Office surveyors from 1816 through 1855. FIA data have been collected annually in Missouri since 1999. Moser and others (2006a) used data from the 1999-2003 FIA inventory of the State of Missouri to depict current forest conditions. The two datasets – historic (PLS) and current (FIA) — had to be reduced to a common data structure to compare species and structures and determine potential for restoration.

During a review in 2002 of current forest management technical specifications by the Missouri Department of Conservation, a team produced a table of current and potential hardwood forest type groups with the suitability (and, by implication, the ease) of conversion based on site index. This table (updated to include pine forest types) became the basis for comparing the PLS data to the FIA data to determine how easy it might be to "convert" current forest types to historic ones.

In order to create common variables of structure and composition between the historic and current data, Moser and others (2006a) used a simple moving window to classify each pixel in the study area to a structure and forest type, using data from land survey plot corners (PLS) and inventory plots (FIA). The moving window centers on a target pixel and then looks at all points within a specified distance (window) of the target pixel. It assigns the value of most of the points in the window to the target pixel and moves to the next pixel. The size of the moving window was adjusted to reflect the spacing of the data points, using a smaller window for the PLS data and a larger window for the FIA data.

After the moving window maps were combined with the forest conversion table, a restoration difficulty/suitability map was created. While there were combinations of past/present that had no information, a substantial amount of acreage was classified into restoration categories: Low Effort, Medium Effort, High Effort, Maximum Effort, and Not Possible.

A map that identifies the effort required to restore a landscape to pre-settlement condition (fig. 3) was produced by combining PLS and FIA data.

Of the 3 million acres in the study area, 14 percent was classified as low-effort sites, 17 percent as medium-effort sites, 11 percent as high-effort sites, 21 percent as non-forest, and 11 percent as not possible (table 3). The remaining 25 percent was classified as no information.

Available funds and time are usually not sufficient to restore all deserving sites, so choices must be made. Resource managers can use this methodology

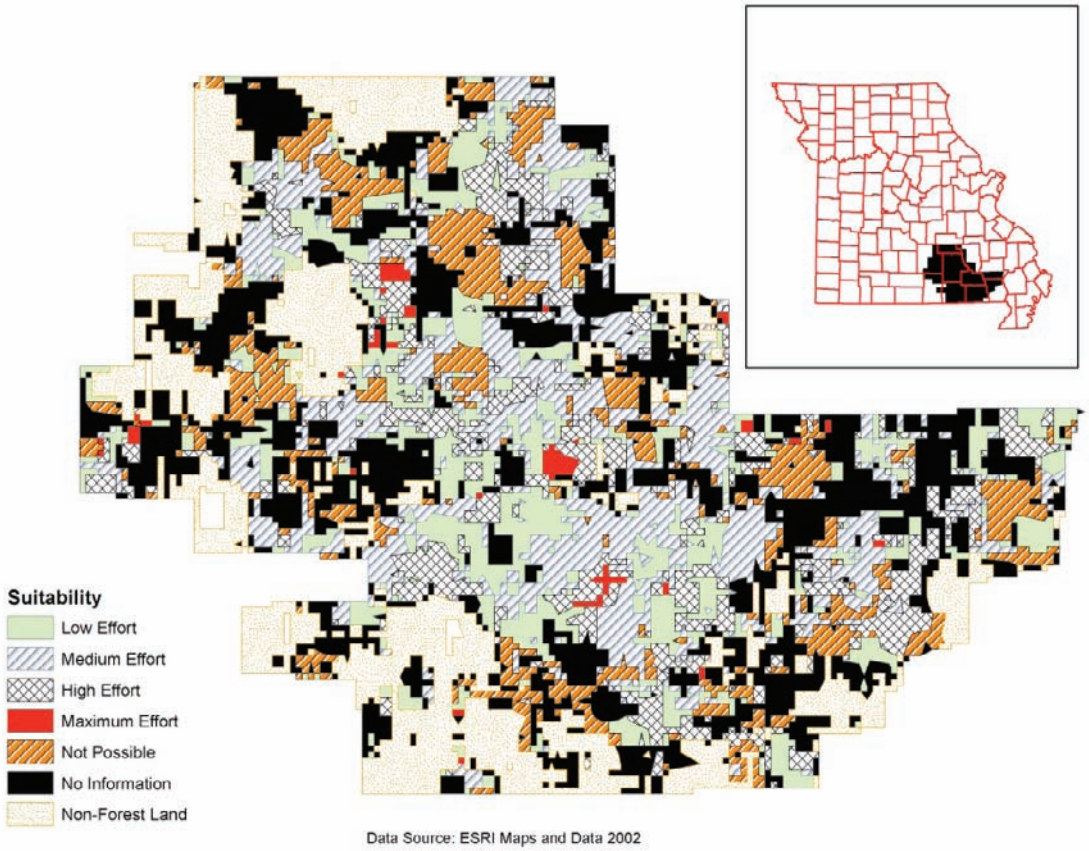

Figure 3—Map of categories of restoration suitability and effort. Suitability is inversely related to effort, e.g., high suitability for restoration is assumed to equal low effort required to do so. The "Not possible" category" represents those binary combinations that were deemed highly unsuitable. "No information" represents those pairs of current and historic forest types that were not considered.

Table 3—Summary of categories of conversion suitability, in 1000s of acres.

Suitability	Acres (1000s)	Percentage of total
Low Effort	406	14
Medium Effort	503	17
High Effort	333	11
Maximum Effort	25	1
Non-Forest	609	21
Not Possible	334	11
No Information	752	25
Grand Total	2,962	

to prioritize restoration opportunities as part of a larger management plan. This methodology is limited by the spatial distribution of past and present species groups and the simplification of a fairly complex vegetation pattern into a more manageable number of categories. Nested within these broad geographic categories will be individual, stand-level decisions that will be based on site conditions and the local manager's individual knowledge and expertise.

Example 6 — Invasive Plants

Non-native invasive plants (NNIP)[5] are expanding across the U.S. Once established, NNIP threaten the sustainability of native forest composition, structure, function, and resource productivity (Moser and others 2009; Webster and others 2006). Factors influencing exotic plants' invasion of forests include: disturbance, competitive release, resource availability, and competitive pressure (Richardson and Pyšek 2006). Moser and others (2008) analyzed FIA plot data with three objectives in mind: 1) document the distribution of species, 2) compare invasive presence to site characteristics, and 3) determine the role of disturbance in invasive species presence and coverage in Missouri. This study was more a snapshot than a trend analysis, as the full extent of NNIP had not previously been documented statewide on FIA plots in Missouri.

Located at the juncture of several ecoregions, Missouri's pre-settlement landscape ranged from upland forest in the Ozarks to bottomland ecosystems of the Mississippi Embayment in the southeast of the state to savannas and prairies in the north and west. The fertile soils of northern Missouri were ideal for farming and settlers quickly cleared the land for agriculture and grazing. In the heavily timbered areas of southern Missouri, commercial harvesting exploited the magnificent stands of shortleaf pine and other species (Beilmann and Brenner 1951). The combination of clearing, settlement, and timber harvesting resulted in a highly fragmented landscape, creating many opportunities for non-native invasive plants to become established in forests.

During 2005-2006, Phase 2 FIA plots were assessed for presence and cover of any of 25 non-native invasive woody, vine, grass, and herbaceous species of interest[6] (table 4). Moser and others (2008) used these NNIP data, along with a geographic information system and geospatial data about road location (ESRI, Redlands, CA, 2006 version) and density and summaries of forest fragmentation data from the Conservation Biology Institute (Heilmann and others 2001), to look for relationships among evidence of human disturbance, forest structure and composition, and invasive species presence.

Of the 25 NNIP species sampled for in the 2005 and 2006 annual inventory panels, only 13 were observed in Missouri and only three—multiflora rose (*Rosa multiflora* Thunb.), non-native bush honeysuckles (*Lonicera* spp.), and Japanese honeysuckle *Lonicera japonica* Thunb.)—were recorded in substantial number. Of the 1,264 plots sampled in this study, 42 percent had at least one invasive species of interest. Multiflora rose was the most frequently recorded species, being observed on 36 percent of the plots (fig. 4). Woody invasive species were especially prominent.

Like all plants, NNIPs benefit from higher site productivity. FIA measures elements of site productivity in three ways: 1) site index, based on representative trees near the plot, 2) aspect, based on measurements taken by the field crews, and 3) physiographic class code, a determination made by field crews based on their assessment of land form, topographic position, and soil type.

[5] This paper will also refer to NNIP as "exotics" or "invasives."

[6] This list was not exhaustive but represented those species likely to have a significant impact somewhere in the 11-state Upper Midwest.

Table 4—Non-native invasive plants surveyed on FIA plots in the Upper Midwest of the U.S., 2005-2006.

Common name	Scientific name
Woody species	
Multiflora rose	*Rosa multiflora*
Japanese barberry	*Berberis thunbergii*
Common buckthorn	*Rhamnus cathartica*
Glossy buckthorn	*Frangula alnus*
Autumn olive	*Elaeagnus umbellata*
Non-native bush honeysuckles	*Lonicera* spp.
European privet	*Ligustrum vulgare*
Vines	
Kudzu	*Pueraria montana*
Porcelain berry	*Ampelopsis brevipendunculata*
Asian bittersweet	*Celastrus orbiculatus*
Japanese honeysuckle	*Lonicera japonica*
Chinese yam	*Dioscorea oppositifolia*
Black swallowwort	*Cynanchum louiseae*
Wintercreeper	*Euonymus fortunei*
Grasses	
Reed canary grass	*Phalaris arundiacea*
Phragmites, Common reed	*Phragmites australis*
Nepalese browntop, Japanese stiltgrass	*Microstegium vimineum*
Herbaceous	
Garlic mustard	*Alliaria petiolata*
Leafy spurge	*Euphorbia esula*
Spotted knapweed	*Centaurea bierbersteinii*
Dame's rocket	*Hesperis matronalis*
Mile-a-minute weed, Asiatic tearthumb	*Polygonum perfoliatum*
Common burdock	*Arctium minus*
Japanese knotweed	*Polygonum cuspidatum*
Marsh thistle	*Cirsium palustre*

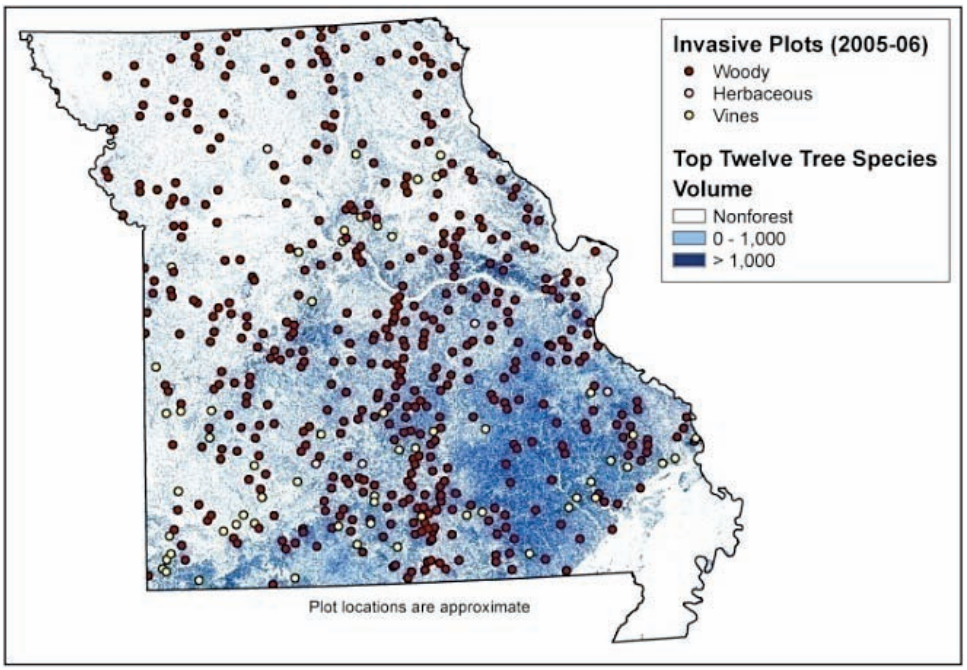

Figure 4—Distribution of plots containing non-native invasive plants in Missouri in 2005-2006 inventory years.

Moser and others (2008) reported the following significant relationships:

- site index and the presence of multiflora rose and non-native bush honeysuckles,
- level aspects and multiflora rose and Japanese honeysuckle, and
- mesic physiographic class and multiflora rose and Japanese honeysuckle.

Like young tree seedlings, ground flora—both native and exotic—are influenced by the presence of trees in the overstory. Not only is the presence of a particular basal area an indicator of the likely microenvironment below the canopy, but in addition it is likely the result of past disturbance events that may also have facilitated the establishment of NNIP. Of the three most prominent invasive plants that Moser and others (2008) reported in Missouri, only multiflora rose appeared to benefit from reduced basal area.

Analysis of the presence and cover of invasive species at a single point in time does not usually provide enough data to evaluate trends in regeneration, expansion, or growth (Rejmánek 1989). The FIA database can elicit evidence of disturbances and/or management activities, but only in the interval since the previous inventory. Non-native invasive plant sampling has only recently been initiated in certain parts of the country; remeasurements of these plots will provide information about the extent and trends of these unwanted guests.

Summary

Strategic forest inventories—such as FIA in the United States—have traditionally focused on estimating the total number of trees or the total tree volume in a state, and they have performed that task well. Such inventories can also be valuable to land managers working at smaller scales, such as management projects on national forests. However, the increased level of statistical variation when using FIA data for small area estimation tends to dissuade people from considering its usefulness altogether at the local level. Yet, the unbiased statistical design of FIA can provide valuable information to support planning and decision making at the project, landscape, and regional levels. To provide useful and defensible information at the local and regional levels, the analysis must be focused and take into account both the opportunities and limitations of the data.

In this paper, we presented examples of how FIA data aids managers at the project level (target tree species, snag management, old-growth estimation), the planning level (density management diagrams), and the landscape level (forest restoration prioritization and factors influencing the presence of exotic plants). While the examples we presented were not explicitly linked across scales, it is quite possible to use the same subset of FIA data to aid decision-making at each of these levels.

The FIA program is designed to estimate resource availability and forest health trends at broad scales. National estimates of carbon storage, forest health indicators, and wood product utilization potential depend on these data. But FIA data also have great potential to assist in achieving management objectives at the local level, while still satisfying broad-level, long-term goals.

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The content of this paper reflects the views of the authors, who are responsible for the facts and accuracy of the information presented herein.

Variable-Density Thinning for Parks and Reserves: An Experimental Case Study at Humboldt Redwoods State Park, California

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Abstract—Variable-density thinning is emerging as a valuable tool for the silvicultural promotion of old-growth conditions in second-growth forests of the Pacific Coast. This paper reports on an experimental variable-density thinning prescription applied between 2006 and 2007 at north coastal California's Humboldt Redwoods State Park. The prescription strategy relied on known patterns of second-growth stand development during the stem exclusion phase, and was designed to alter current stand development trajectories in order to promote reference forest conditions. Prescription outcomes are described and tradeoffs are discussed, with management constraints unique to parks and reserves providing the context for this analysis.

Keywords: ecological restoration, forest stand dynamics, disturbance, stand structure.

Introduction

Silvicultural practices and systems in timberlands management are increasingly modeled after patterns of natural disturbance (Puettmann and others 2009), but the converse is also occurring in parks and reserves, where the potential of silviculture as a disturbance force to restore natural conditions in re-growth forests is being developed. In the coastal areas of the western United States, thinning to promote gaps and stand complexity has become a dominant management paradigm for the active restoration of old-forest attributes to second-growth forests (Carey 2003; Carey and Curtis 1996). In this region, vegetation composition and structure in old and young forests differ widely (Bailey and others 1998; Lindh and Muir 2004), and biodiversity is favored by the structural complexity that thinning promotes in second-growth forests (Bailey and Tappeiner 1998; Carey and Wilson 2001; Lindh and Muir 2004; Schowalter and others 2003). New silvicultural techniques are being devised to optimally promote that diversity (O'Hara and Waring 2005).

For these forests of the Pacific Coast, restoration practices are conducted to accelerate and make more certain a developmental pathway that will eventually result in stand structures and compositions comparable to old-growth forests. In younger stands, silvicultural restoration treatments are conducted to manipulate stand structure and composition in order to promote a subset of trees with structural attributes (low height:diameter ratios, high live crown ratios) that enhance their potential for resilience and persistence, and to promote patterns of development that promote the sustained dominance of those individuals (Chittick and Keyes 2007; Plummer 2008). Treatments may also be conducted to remediate

In: Jain, Theresa B.; Graham, Russell T.; and Sandquist, Jonathan, tech. eds. 2010. Integrated management of carbon sequestration and biomass utilization opportunities in a changing climate: Proceedings of the 2009 National Silviculture Workshop; 2009 June 15-18; Boise, ID. Proceedings RMRS-P-61. Fort Collins, CO: U.S. Department of Agriculture, Forest Service, Rocky Mountain Research Station. 351 p.

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composition imbalances. In older stands, restoration treatments may be conducted to expedite the development of biodiversity and structural complexity by establishing canopy gaps that promote understory plant recruitment and the initiation of a new tree cohort (Harrington and others 2005; Thysell and Carey 2001). Parks and reserves represent a unique context for the restoration of old-forest features to second-growth forests; in many cases the restoration strategy is constrained to a single-entry opportunity that must maximize ecological benefits while minimizing the potential for negative consequences.

This paper presents a case study in forest restoration at Humboldt Redwoods State Park, where an experimental variable-density thinning prescription was applied during 2006 and 2007. The treatment is discussed from both ecological and operational standpoints.

Methods

Project Area

Humboldt Redwoods State Park was established in 1921 to preserve massive alluvial stands of old-growth redwood along lower Bull Creek and the Eel River. Such forests account for more than 17,000 of the park's nearly 53,000 acres. The remaining 36,000 acres consist primarily of second-growth upland forests. In the aftermath of flooding in 1955, which damaged Rockefeller Forest, a need was recognized to claim and control the upland forests that surround the old-growth stands at higher stream orders. The park was steadily expanded between 1963 and 1984, until the whole of the Bull Creek watershed was annexed (Rohde and Rohde 1992). Most of the upland forests of the Bull Creek watershed (including Panther Creek), however, had been logged between 1950 and 1962, and are now in varying conditions of ecological impairment.

Stands within the Panther Creek watershed are aptly described by three impaired forest condition classes defined in an earlier assessment of second-growth forests in Humboldt Redwoods State Park (Keyes 2005). The project area did not have distinct stand boundaries, but instead was comprised of a mosaic of conditions typical of second-growth forests in this region: namely, compositionally diverse but structurally homogenous even-aged mixed stands (figs. 1 and 2). Due to management history, overstory density was very high, the proportional composition of Douglas-fir (*Pseudotsuga menziessi* (Mirb.) Franco) was very low, and spatial heterogeneity and vertical stand structure were minimal. Remediation of those conditions was the objective of the prescription. The pre-treatment conditions found in these stands were likely beyond the natural range of variability, and given our understanding of second-growth forest stand development patterns (Oliver 1980; Oliver 1981; Oliver and Larson 1996), we believe they are unlikely to achieve the pre-disturbance forest structure that defines the area's reference condition.

Restoration Prescription

A form of variable-density thinning (VDT) was the recommended technique to achieve ecological restoration goals for the forests of Panther Creek. VDT differs from most traditional methods of thinning, which typically emphasize uniformity in tree spacing and form (Nyland 2007; Smith and others 1997), in that it promotes spatial heterogeneity and height differentiation among canopy trees to enhance structural complexity. While several approaches to VDT have been proposed and tested in experimental settings (e.g. Carey 2003), techniques for the efficient implementation of VDT as an operational practice have not yet

Figure 1—Panther Creek watershed, 1959. Panther Creek drains from southwest to northeast through the middle of this aerial photo. Landslides resulting from cutting are evident at bottom center of photo. The photo illustrates the intensity of harvesting activity and high density of skid roads.

been developed or standardized. Existing experimental efforts that have been documented generally introduce spatial heterogeneity in a random manner, without capitalizing on variability that already occurs within the stand. Approaches relying on randomness or stand-level targets (such as inter-tree spacing levels) fail to permit flexibility in tree selection that allows for adaptation to existing heterogeneity, and hence can delay the achievement of treatment goals.

The VDT technique developed for the forests of Panther Creek was designed to capitalize upon differentiation that had already occurred. The prescription consists of a basic size-constrained, species-specific, diameter-based multiplier cutting rule (Dx Rule; see fig. 3 for a schematic):

A) Identify the largest Douglas-fir or redwood tree larger than five inches dbh in the vicinity, multiply the diameter by two, and cut trees for that many feet around the Douglas-fir in a radius up to, but not greater than, twenty feet (the maximum treatment radius is twenty feet).
B) Within the cutting radius, cut no trees smaller than five inches dbh, or conifers greater than ten inches dbh, or hardwoods (broadleaf trees) over fifteen inches dbh.
C) Move on to next closest Douglas-fir or redwood release tree (including Douglas-firs retained in Step B), and repeat Steps A and B.

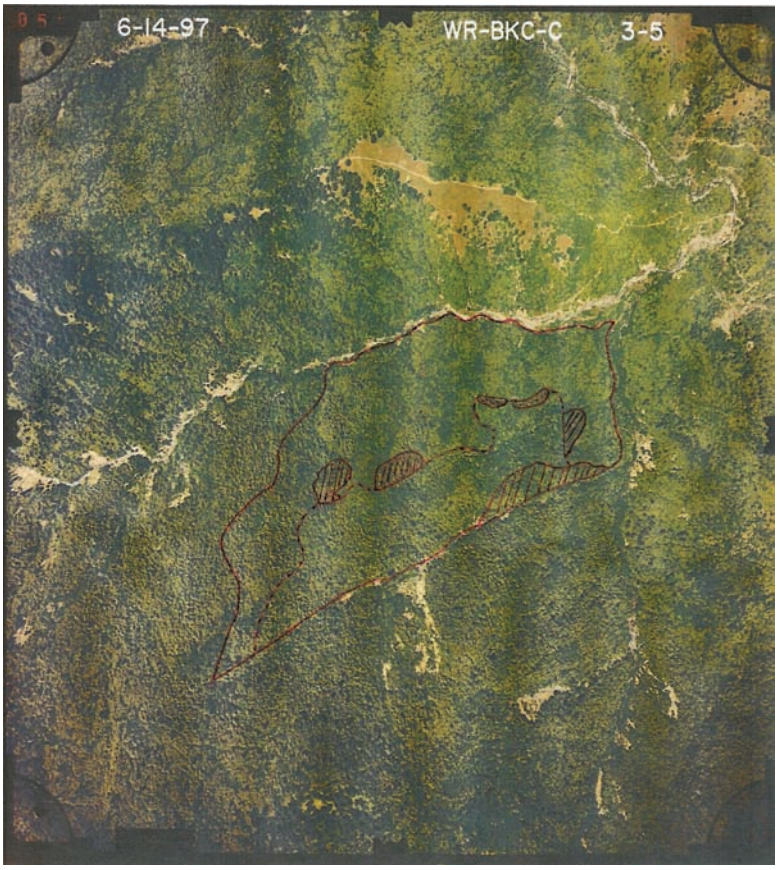

Figure 2—Panther Creek watershed, 1997. Photo reveals the widespread distribution of hardwood re-growth (light green) amidst lesser conifer re-growth (dark green) (red lines reference project area boundaries and landmarks). Although spatially uniform in age structure, the area is compositionally diverse, posing difficulty to identification of stands with distinct boundaries, and difficulty in preparing stand-based prescriptions. Such variability lends itself to tree-scale prescriptions that operate adequately and efficiently across a spatial range of compositions.

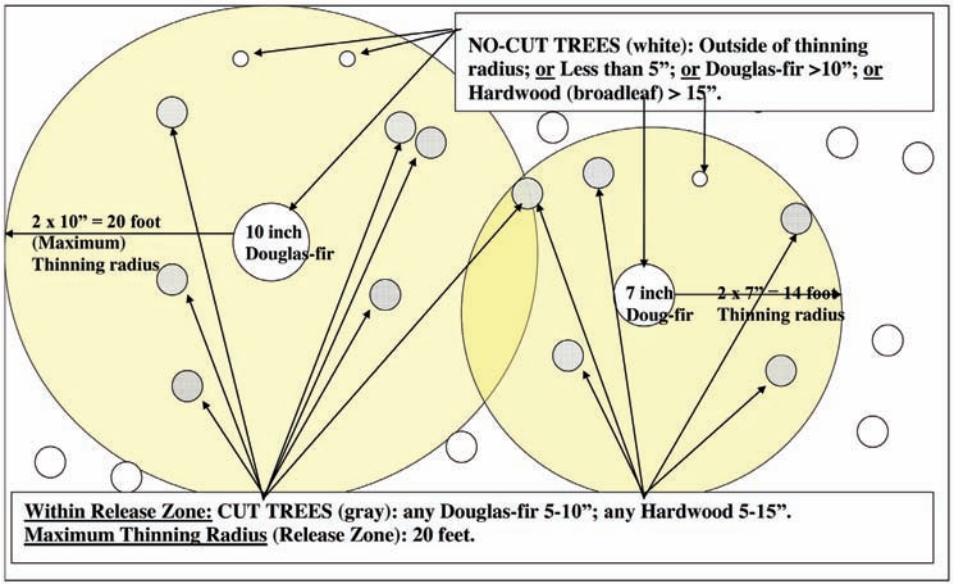

Figure 3—Conceptual diagram of variable-density thinning prescription.

For stands with little structural variation, the Dx Rule does not significantly alter spatial heterogeneity. But in partly-differentiated stands, the Dx Rule exacerbates inequalities in growing space by appropriating proportionally larger gaps around larger trees than around smaller trees. Moreover, two goals in prescription development were simplicity and universality; this necessitated to some extent a compromise that worked across the range of forest conditions. An advantage of the approach is its universal applicability across forest conditions, and an ability to remain effective as contractors move through unanticipated changes in those conditions.

Analysis

We utilized pre-treatment and post-treatment data (1790 tree records) from a semi-permanent plot network of 84 plots. This dataset captured tree species, diameter at breast height, and harvest history (cut versus uncut). Both compositional and structural changes were analyzed to quantify or evaluate general stand characteristics, re-distribution of stand volume, prescription effectiveness, and operator adherence to the prescription.

Prescription implementation effectiveness was indicated by a redistribution of basal area from smaller hardwoods to conifers greater than 10” and hardwoods greater than 15”. Along with a species composition shift to a higher proportion of conifers in the post-treatment stands, an overall reduction in trees per acre, with the greatest reduction in hardwood stems less than 15” is also indicative of effective implementation of the prescription. Maintenance or exacerbation of spatial variability in stand density was another desirable outcome. Adherence of the operator to the prescription was evaluated by focusing on the percentage of plots in which the prescription was correctly versus incorrectly implemented. Consistent operator errors would have indicated either an overly complex prescription or a lower level of operator professionalism.

In addition to the quantitative analysis of plot data, we made numerous qualitative observations during and after the implementation of the prescription. Our goal here was to understand the operational linkages between the prescription and its implementation by forest workers, in order to improve and simplify future prescriptions for similar complex forest objectives.

Results and Discussion

Quantitative Analysis

Figures 4 and 5 reveal that the prescription reduced overall stand density while shifting proportional composition from hardwood to softwood dominance. Hardwoods accounted for 52.6 percent of the original stand basal area and only 5.4 percent of the residual stands basal area. In terms of trees per acre, hardwoods accounted for 71.9 percent of the stems in the pre-cut stand and only 17.8 percent of the residual stand. The greatest reduction in both basal area and trees per acre came from the tanoak component of the stand. Although a desired compositional shift occurred, the removal intensity on this site may be higher than desired given the potential for hardwood sprout regeneration. The intention of the cutting was to release existing conifers, not create new space for stem initiation that would favor those sprouting hardwoods.

Figure 6 displays the pre-cut and post-cut stand structure in terms of both basal area and trees per acre. Cutters by-and-large accurately observed the diameter and species guidelines defined by the prescription, but they did improperly remove some hardwoods above the 10” diameter limit. Such cuttings represented only

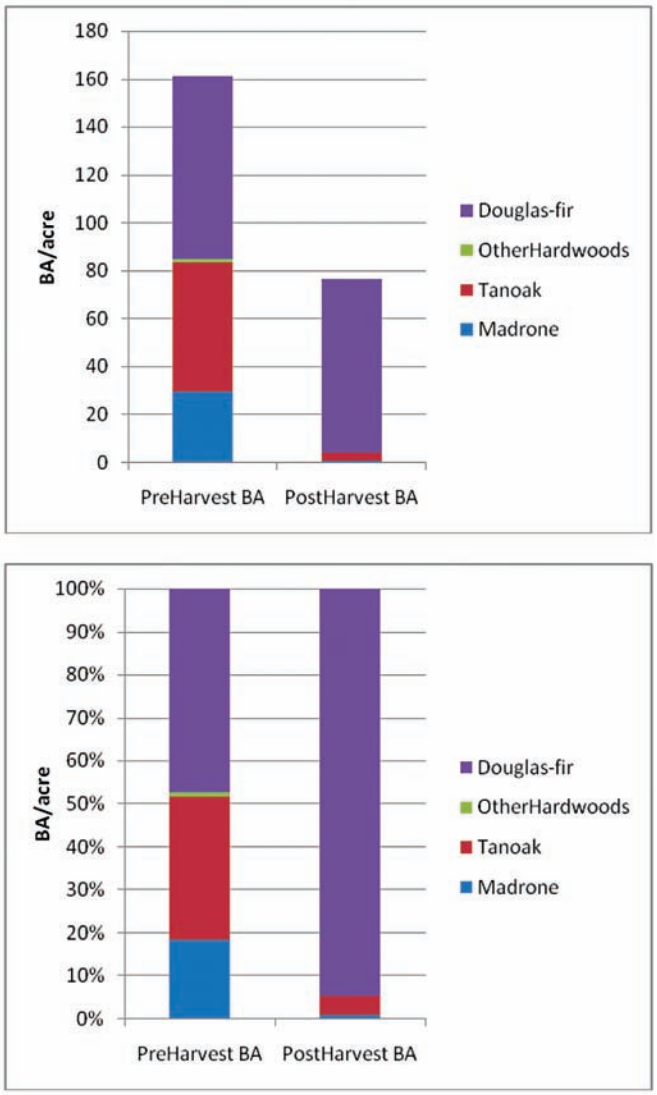

Figure 4—Pre-harvest and post-harvest species compositions expressed in basal area (ft^2 per acre) and Tanoak (*Lithocarpus densiflorus*) Blume, Madrone (*Arbutus* L.) as a percentage of the total basal area.

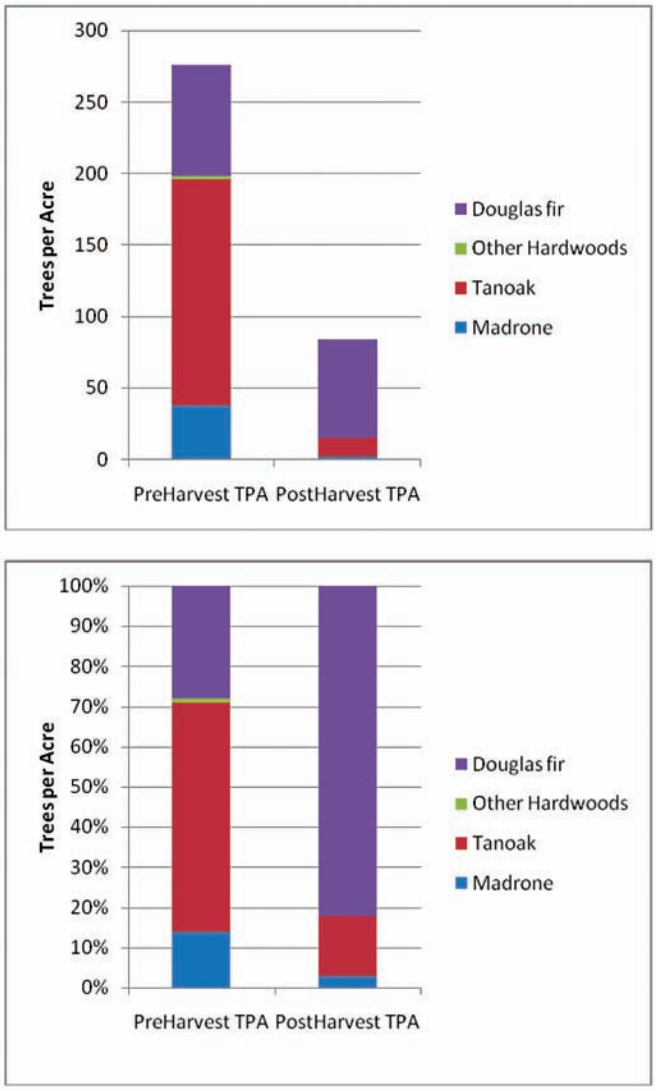

Figure 5—Pre-harvest and post-harvest species compositions expressed in trees per acre and as a percentage of the total trees per acre.

4 trees per acre, but removed a substantial amount of hardwood basal area in these larger diameter classes. This removal was a deviation from the prescribed treatment rather than a failing of the prescription itself; correct implementation would have retained these large diameter hardwoods.

Cutters accurately distinguished between conifers and hardwoods, but did not distinguish between hardwoods. All of the species in the "other hardwoods" group (California laurel [*Umbellularia californica* (Hook & Arn.) Nutt.], coastal live oak [*Quercus agrifolia* Née var. *oxyadenia* (Torr.) J.T. Howell], and California black oak [*Quercus kelloggii*] Newberry) were eliminated by the cutting. Those trees were few in number, representing an average of less than one tree per acre. However, maintaining the site's species richness was an inherent goal of the prescription. Hence, proper species identification proved an important criterion for proper prescription implementation.

To concisely explore the prescription's effect of spatial heterogeneity in stand density, we conducted analysis of grouped decile classes (representing 1/10th of the range of pre-cut density) for trees per acre and basal area (fig. 7), similar to

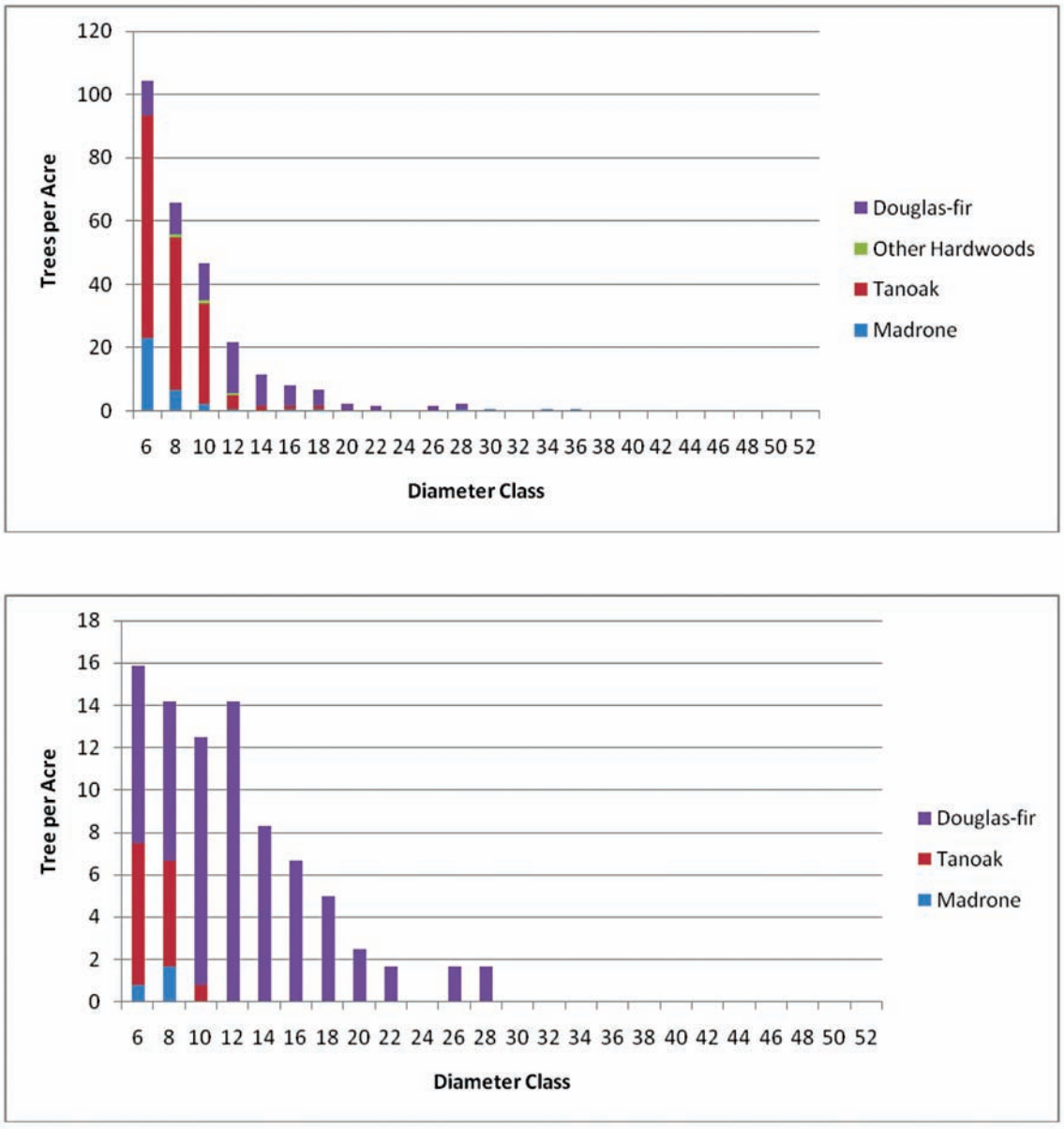

Figure 6—Species-designated diameter distributions displaying pre-cut (top) and post-cut (bottom) trees per acre.

diameter classes. This approach allowed for a contrast of the pre-cut and post-cut structural variation in density. The range of stem densities (in terms of trees per acre) did not change as a result of the prescription; in both the pre-cut and post-cut stand, eight of the ten density classes were occupied. There was a notable redistribution of plots falling into the lowest stem density class. The basal area density classes exhibited the same shift towards the lowest class; however, seven of the eight density classes in the original stand were still occupied. Some structural diversity in basal area was lost, since no post-treatment plots occupied the three highest basal area density classes observed prior to treatment.

According to the rule, no cutting was to be conducted in pure hardwood areas where releasable conifers were not present. Analysis of cut and uncut areas combined revealed that the overall species composition shift was tempered by high hardwood and low softwood stem densities in the uncut areas. Generally, the uncut areas either did not contain releasable softwoods, or if softwoods were present, the hardwood density was low enough to preclude thinning. These untreated plots

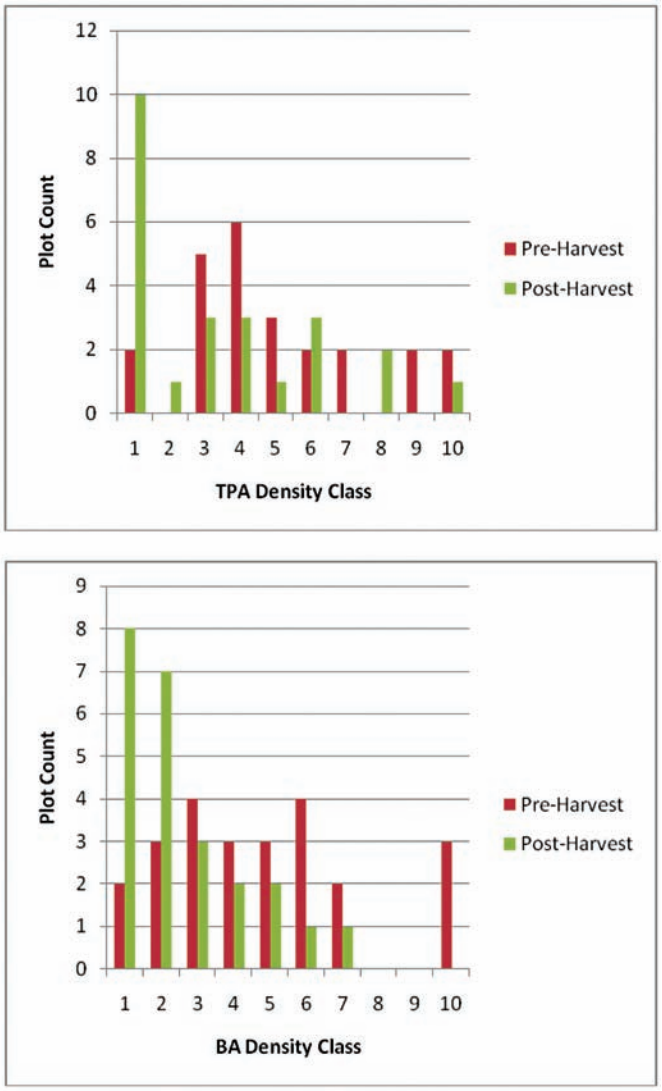

Figure 7—Pre-cut and post-cut plot counts grouped by decile density classes for trees per acre (top) and basal area (bottom). The range of stand densities was reduced since highest density areas no longer exist; however, substantial variation remains in the treated stand, with greater representation in the lowest density classes. Variation in stand density classes indicates spatial heterogeneity in stand structure.

contribute to the large diameter hardwood component and maintained some areas of high stem density, retaining the upper end of the overall range of densities within the stand.

Two different contractors implemented the prescriptions in separate parts of the project area, yielding different trends in removal intensity and the structural retention. Contractor implementation of the prescription proved to be a strong determinant of post-treatment structure, and a critical aspect of a successfully designed and implemented restoration treatment (discussed below).

Development of the stand structures established by this treatment will reveal whether the treatment's objectives will be met over longer timeframes. Occlusion of overstory space by released Douglas-firs and their dominance over hardwood

sprouts was the intended result, but whether this will be achieved depends upon rates of crown expansion among residual trees and rates of height growth among hardwood stump sprouts

Qualitative Observations

Thinning crews failed to thin small pockets within the designated project area. It is estimated that the skipped areas comprised less than five percent of the total project area. These mostly occurred in small pockets, apparently between steep stream channels. They were small enough not to be detected during the project implementation, hence even the presence of an on-site compliance worker would unlikely detect them in real-time. At lower slope positions throughout the project area, class I and II streams were impassable chasms. This made contour-based foot-travel impossible, and required substantial upslope and downslope travel during project reconnaissance; it was also probably responsible for many of the pockets that were missed by thinning crews.

Communications between foresters and forest workers posed a potential challenge to the correct implementation of the thinning rule. Communications with the leader of the migrant crew appeared to suffer from both language and cultural differences. The fortunate presence of a compliance forester that was fluent in both English and Spanish, and with a good understanding of forestry principles and the prescription itself, is credited with enhancing treatment implementation at Panther Creek. This element is a vital consideration for similar situations where communication is difficult, the prescription is by description, and the prescription objective and post-treatment structure is unlike that which forest workers are accustomed.

Workers were impatient to start cutting even prior to receiving instructions on the thinning rule. Workers are believed to have been accustomed to the simple instructions associated with pre-commercial thinning's regular spacing method. In retrospect we would have planned and scheduled with the contractor a formal saw-free instruction with all crews present and attentive prior to removal of cutting equipment and prior to entering treatment area. In the absence of tree marking, it is vital that crews recognize that VDT thinning differs in important ways from traditional thinning methods. We provided a demonstration area for the contractor bidding and to familiarize workers with the prescription, but should have contractually required its use in a tutorial provided by us at the outset.

With our makeshift reactions to these apparent deficiencies, immediate post-treatment impressions of the VDT proved positive in regard to achieving project objectives of shifted species composition and reduced density. Residual spacing was variable throughout the area, hence crews did not thin in regular spacing patterns as might have been feared with communications limitations. Workers appeared capable of identifying tree diameters correctly. Diameter-based constraints appeared to be adhered to quite well. Large trees were not cut inadvertently. However, it was difficult to determine whether the diameter multiplier (2x) was being implemented by crews or whether a constant 20-foot radius was applied to all focus trees. Many residual trees by prescription had large diameters, hence had target thinning radii that were capped at 20 feet. In future efforts, we would like to test the radius-multiplier in a stand with smaller pre-treatment dbh's; if the multiplier is not being adhered to by crews, yet the outcomes achieve the treatment objectives, then a standard thinning radius would offer greater simplicity.

Special treatment areas of reserve forest were flagged for exclusion. Such areas included pockets that were not logged during the mid-century, did not require restoration, and could potentially be damaged by cutters. However, this special treatment was not necessary. A pocket that went unnoticed during pre-treatment

reconnaissance, and which was not flagged off as an exclusion area, was subjected to the crews and the thinning rule. The thinning rule as written prohibited cutting the large trees that occurred in that area, and it was correctly implemented. We believe that crews with a proven track record in prior contracts can be trusted to adhere to and be able to distinguish accurately diameters and diameter-based constraints for this type of prescription. Hence, the expense and potential for confusion associated with flagging off reserve pockets can be avoided, unless they are not specifically protected by the language of the thinning rule.

Conclusions

The outcomes of restoration treatment applied to Panther Creek were successful in most regards. Concerns regarded the prescription implementation aspect, rather than the prescription formulation itself. Future efforts can benefit by reducing worker confusion, enhancing worker understanding of the prescription, and reducing pre-treatment expense. Structurally and compositionally, the prescription appeared to achieve forest objectives. Since implementation appeared to be a limiting factor to project success, further simplifications to the prescription would prove beneficial if they can be shown to reduce worker confusion and expense while yet resulting in similar post-treatment forest complexity.

Acknowledgments

Staff at the North Coast Sector of California State Parks proposed the opportunity, provided logistical and financial support, and implemented the prescription. Humboldt State University students were instrumental in data collection. The study was made possible in part by the Applied Forest Management Program at the University of Montana, a research and outreach program of the Montana Forest & Conservation Experiment Station.

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The content of this paper reflects the views of the authors, who are responsible for the facts and accuracy of the information presented herein.

Site Quality Changes in Response to Slash Retention and Prescribed Fire in Thinned Ponderosa Pine Forests

Matt Busse[1]

Abstract—The ecological effects of post-thinning slash retention on vegetation, wildlife browse, and soil were evaluated in sixty-year-old stands of second-growth pine in central Oregon. Three slash-retention treatments were compared: whole-tree removal, bole-only removal, and thin no removal (boles and slash scattered on site). The study intent was to create a wide gradient of surface organic matter mass among treatments and assess any ensuing changes in site quality. No differences in site or soil productivity indices were found among the slash-retention treatments after 20 years. Tree growth, understory plant production and diversity, wildlife browse cover, litter decay, soil nutrients, and soil biological activity were similar among the treatments, suggesting that the retention of thinning slash is trivial to the health of these forests. In general, thinning alone, regardless of slash treatment, and thinning with subsequent burning were sound options for reducing wildfire hazard and maintaining site quality in these pine ecosystems.

Keywords: site productivity, thinning slash, prescribed fire, organic matter, pumice soil

Introduction

Forest management is a contentious topic. Unresolved issues are numerous and they range in scope from local concerns about public safety and wildfire to the politically charged world of protecting old-growth ecosystems and to the scientific unknown of global climate change. Should forests be managed as carbon stores in order to slow the warming of our planet? Is it unacceptable to harvest moderate-to-large trees on public lands, or should such practices be encouraged on occasion in order to maintain healthy, long-lived forests? Which forests should be actively managed and which should be allowed to develop "naturally" with minimal or no human intrusion? These are complex questions whose practical solutions have evaded some of the brightest minds of today despite, at times, reasonably clear policy and legal direction.

But the beauty of today's forestry issues is that they require discourse, debate, and scientific inquiry. They require the involvement of assorted individuals, groups, and agencies; they require understanding of complex ecosystems; and, perhaps, they require compromise. However, as current issues, many will likely be replaced in time by other contentious issues, just as some of the key concerns of today were not on the radar screen 20 years ago. And this is where the current story begins, with a forestry issue that was on people's minds 20 to 25 years ago, but has since been dropped from many dance cards: long-term site productivity. Within the forest management and research community, this topic was crackling with energy well before biofuel production, biochar, or hybrid cars gained their notoriety.

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The notion of long-term site productivity on public forests owes much to the congressional mandate set by the Multiple Use—Sustained Yield Act of 1960 (National Forests shall provide "sustained yield of the several products and services… without impairment of the productivity of the land") and the National Forest Management Act of 1976 ("Soil will not be irreversibly damaged"). The responsibility of sustaining our nation's forests for future generations was set into law and placed in the hands of forest managers, with open public participation and with full reliance on professionals from a multitude of forestry and ecosystem-related disciplines. Soil scientists, for example, spoke for the land with their concerns about detrimental erosion, compaction, and nutrient losses. By the 1980s, numerous soils-related studies were installed on public forests in an attempt to better understand how forest management practices affected the long-term productivity of the land. Studies such as the Long-Term Soil Productivity study (Powers 2006) and the Long-term Ecosystem Productivity experiment (Bormann and others 2008) steamrolled into view. It was during this time period that the Bend Long-term Site Productivity study (LTSP) was conceived.

The Bend LTSP study began as a collaboration between the Deschutes National Forest, located in the rainshadow of the central Oregon Cascades, and Pat Cochran, a research soil scientist with the Pacific Northwest Research Station. The National Forest was alarmed about bark beetle infestations in young, dense stands of ponderosa pine, and needed to monitor the effects of proposed thinning operations on forest health. Was wide-scale thinning an effective practice to limit bark beetle attack? And how would pine vegetation, fuels, soils, and wildlife respond to an aggressive thinning regime? For Pat Cochran, a golden opportunity was presented to study the fate of thinning residues in pine forests. Should slash material be left on site in order to maintain or enhance site productivity? Soil scientists had long recognized the quintessential quality of soil organic matter as a cornerstone of productive ecosystems. So the definitive question was asked: how important is it to retain site organic matter in the form of thinning slash?

Preliminary evidence from central Oregon suggested that thinning residues were, indeed, important to site productivity. Little and Shainsky (1995) assessed the organic carbon, nitrogen, and phosphorus pools in trees and soils and recommended bole-only harvesting as the best management practice for maintaining site productivity. They suggested that whole-tree harvesting could remove a substantial percentage of the above-ground nitrogen pool. This finding is supported by studies in the boreal forests of Scandinavia, in which growth declines resulted when harvest slash was removed from site (Jacobson and others 2000). And as a general recommendation, Page-Dumroese and others (2010) suggested leaving thinning residues on site, particularly in areas with low fertility soils, to help (1) maintain soil organic matter levels, (2) increase N cycling, (3) promote mycorrhizae development, and (4) minimize detrimental soil compaction.

Evidence to support such recommendations is far from definitive, however. In an effort to resolve this issue, LTSP study plots were installed in 1988, and the effects of retaining thinning residues were evaluated during the succeeding 20 years. The objective of this work was to determine whether surface residues are advantageous to tree growth, understory plant composition and production, wildlife browse, or soil fertility.

Methods

Site Description

The study is located on the Deschutes National Forest, in the pumice plateau region of central Oregon between the Cascade Range to the west and the Great

Basin to the east. Ponderosa pine (*Pinus ponderosa* C. Lawson) forests are common throughout the plateau, with their production limited by cold winters and dry summers. Annual precipitation ranges from about 30 to 50 cm and occurs primarily as snow during the winter months. The growing season for pine typically lasts from mid-May to mid-August.

Three sites were selected in 1988 along a precipitation gradient that extends east from the Cascade crest to the desert fringe. East Fort Rock, the least productive of the three sites (site index of 25 m at 100 y), is located near the desert fringe, 17 km southeast of Bend, OR; Sugar Cast, a medium site (site index of 31 m at 100 y), is located 5 km east of Sunriver, OR; and Swede Ridge, the most productive of the three sites (site index of 35 at 100y), is located 20 km west of Bend, OR. These sites had been clearcut logged in the 1930s and allowed to regenerate naturally. By 1988, stand densities ranged from 480 to 780 trees per hectare (195-315 trees per acre), mean tree diameter at breast height (DBH) was 26 cm (10.2 inches), and basal areas ranged from 24-33 m^2 ha^{-1} (105-144 ft^2 ac^{-1}).

The soil at the three sites is coarse textured (sandy loam), developing from wind blown deposits of pumice and ash from the eruption of Mt. Mazama. Soil fertility is low, with both organic matter and total nitrogen contents particularly sparse at soil depths below 10 cm. Understory vegetation is comprised primarily of woody shrubs, including antelope bitterbrush (*Purshia tridentata* [Pursh] DC.), snowbrush (*Ceanothus velutinus* Douglas ex Hook.), and greenleaf manzanita (*Arctostaphylos patula* Greene). Herbaceous species such as bottlebrush squirreltail (*Elymus elymoides* [Raf.] Swezey), western needlegrass (*Achnatherum occidentale* [Thurb.] Barkworth), Idaho fescue (*Festuca idahoensis* Elmer), Ross' sedge (*Carex rossii* Boott), Virginia strawberry (*Fragaria virginiana* Duchesne), cryptantha (*Cryptantha affinis* [A. Gray] Greene), and silverleaf phacelia (*Phacelia hastata* Douglas ex Lehm.) are present at low coverage.

Study Design

Four slash-retention treatments, components of the larger Bend LTSP study (see Busse and others 2009 for overall experimental design), were compared: whole-tree removal, bole-only removal, thin no removal, plus a no-thin control. The experiment was a randomized complete block with three replications (one replication per site). Treatment plots were 0.4 ha (1 ac) with 20 m (66 ft) or greater between adjacent plots. Thinning guidelines followed the prescriptions used by the Deschutes National Forest in 1988. Target basal area was 13.7 m^2 ha^{-1} (60 ft^2 ac^{-1}) or less with a tree spacing of approximately 5.5 x 6.1 m (18 x 20 ft), favoring the removal of damaged or smaller trees. Trees marked for thinning were cut by chainsaw between November 1988 and October 1989, and felled trees were removed using either a rubber-tire or track grapple skidder. All harvest material was removed from whole-tree removal plots; boles were removed from bole-only removal plots, with tree crowns lopped and scattered across the plots; and all harvest material was retained on the thin no removal plots, with tree crowns lopped and scattered and boles left intact on the ground (fig. 1). Surface residues at the start of the experiment averaged 12 Mg ha^{-1} (5.3 tons ac^{-1}) for both whole-tree removal and no thin plots, 26 Mg ha^{-1} (11.6 tons ac^{-1}) for bole-only removal plots, and 48 Mg ha^{-1} (21.4 tons ac^{-1}) for thin no removal plots.

Two additional LTSP treatments, repeated prescribed fire (fig. 2) and fertilization, will be introduced briefly in the results of this report. Three replicate plots of each slash-retention treatment were (1) burned in spring 1991 and spring 2002 by low-to moderate-intensity prescribed fire, (2) fertilized in 1991 and 1996 with nitrogen (224 kg ha^{-1}), phosphorus (112 kg ha^{-1}), and sulfur (37 kg ha^{-1}), or (3) left unburned and unfertilized. Details of the burn conditions, fire behavior, and fertilizer application rates are presented by Busse and others (2009).

Figure 1—Thinning and slash-removal treatments, clockwise from upper left: whole-tree removal; bolewood-only removal; thin, no-removal; and no thin.

Figure 2—Typical low- to moderate-intensity prescribed burning in spring 2002. Average flame lengths were 0.4 to 0.8 m, with litter and duff depths reduced by about 50 percent.

Site Productivity Measurements

Trees—Diameter at breast height (DBH) of all trees on all plots was measured in 1988 (pre-treatment), 1991, 1996, 2001, and 2006. Any mortality or damage to trees was recorded at each date. Total tree height was measured in 1991 and 1996 using an optical dendrometer, with a subset of 15 trees per plot (representing a cross-section of tree sizes) measured for volume. Regression equations were then developed that predicted tree volume in 2001 and 2006 as a function of field measurements of DBH. Periodic annual increment diameter and volume growth were calculated for live trees only.

Understory vegetation—Shrub cover was estimated ocularly prior to treatment (1988), and then measured using belt transects in 1993, 1996, 1999, 2003, and 2006 (Busse and Riegel 2009). Three belt transects (5 x 20 m) were located systematically in each plot, and each shrub within a belt transect was measured for canopy length and width. Coverage of an individual shrub was calculated assuming an ellipse-shaped canopy. Total cover on a plot basis was calculated as the sum of the individual shrubs within the three transects.

Herbaceous plants were clipped at ground level for biomass determination during peak season (mid-June to mid-July) in 1992, 1994, 1997, 1998, and 2003. Circular plot frames (1.5 m diam.) were used in 1992 and 1994, and rectangular-shaped frames (0.5 x 0.5 m) were used subsequently. All plants within 8 systematically located plot frames per plot were clipped at each sample date. Plant species were identified for richness and diversity indices, clipped, dried, and weighed separately for dry matter production for 1992 and 1994 samples only. Species diversity was estimated by Simpson's Diversity Index. After 1994, plants were clipped separately by lifeform (forb, graminoid) prior to drying and weighing, and no attempt was made to determine species diversity.

Fuel load—Surface woody fuel mass was measured in 1990 (pre-burn), 1991, 1996, 2001 (preburn), 2002, and 2007 using a modified planar-intercept method (Brown 1974). Downed wood was counted on 12 transect lines per plot. Measurement length on each transect line was 3 m for the 1-h and 10-h timelag fuels; 10 m for 100-h timelag fuels; and 15 m for 1000-h timelag fuels and larger. Litter and duff mass was estimated by collecting 12, 50 x 50 cm samples per plot, located adjacent to each transect line. Dry weights were determined following oven drying at 70 °C for 48 h.

Soil—Soil samples were collected periodically from 1989 to 2006 for measurements of organic matter content, total carbon, nutrient content, pH, fertility index (carbon:nitrogen ratio), microbial biomass and activity, and microbial diversity (phospholipid fatty acids) using standard analytical procedures. Ten samples from the surface 10 cm of mineral soil were composited per plot at each sample date. In addition, a two-year litter decay study examined differences in decomposition rates among the slash-retention treatments. Briefly, litterfall was collected in fall 1992 from areas outside of the treatment plots. A pre-weighed amount of litter (about 5 g) was placed in 3 mm-mesh litterbags, and 8 replicate litterbags were placed on the forest floor surface of each plot. Two litterbags were collected from each plot semi-annually, and the litter was dried and weighed for mass loss.

Data Analyses

The main effects of thinning and slash-retention treatments on vegetation, fuel load, and litter decay were analyzed by repeated measures analysis (PROC MIXED in SAS 9.1). Treatment effects on soil properties for 2003 samples only were tested using ANOVA. Significance for all statistical analyses was set at $\alpha = 0.05$.

Findings and Implications

Twenty years is a brief period in the life of most ponderosa pine forests. Thus, the findings here may represent little more than an extended snapshot in time, far from a complete story. However, 20 years is most likely sufficient to draw a few conclusions about the fate of thinning residues and their contribution to site productivity. Woody residues decay surprisingly fast in this region (Busse 1994), a fact that is evident on the LTSP plots: the organic residues have either been incorporated in the duff layer or they have decayed sufficiently that their presence is visually undetected. Any measurable effect from this pulse addition of foliage and woody material should have registered by now. Of course, whether the pulse effect registers in succeeding years remains to be seen. Nevertheless, the results from the study were unambiguous, no changes in site productivity indices resulted from the gradient of thinning slash left on site. Whole-tree harvesting, bole-only harvesting, and leaving all material on site produced similar vegetation and soil responses in these forests. The following sections document this observation.

Tree Growth and Mortality

No differences in tree diameter or volume growth were detected among the 3 slash-retention treatments during the 15-year span between 1991 and 2006 (fig. 3). This trend held whether the plots were burned or not. In comparison, unthinned plots had a 50 percent lower rate of diameter growth compared to the thinned treatments, as expected following a thinning. The thinning release was expressed in diameter growth only, however. Volume growth was comparable between thinned and unthinned plots, reflecting the greater number of trees of smaller diameter on the unthinned plots.

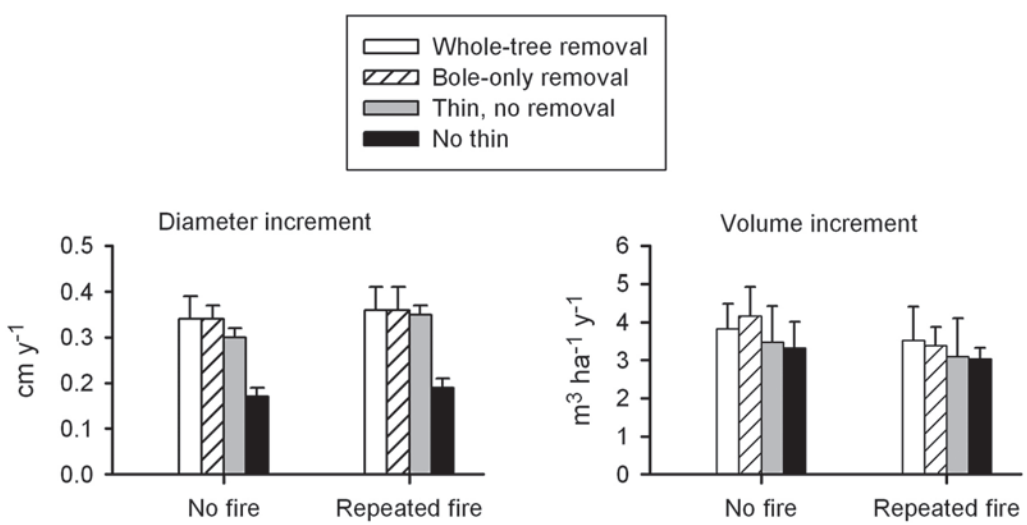

Figure 3—Tree-growth increment between 1991 and 2006. Thinning and slash-removal treatments were completed in 1989, and repeated prescribed burning occurred in spring 1991 and spring 2002. Error bars represent the standard error of the mean ($n = 3$).

Tree mortality was nominal in thinned plots in the absence of fire, with no differences found among the slash treatments (table 1). Repeated fire, in comparison, resulted in a reasonably high rate of mortality in the thin no removal treatment. This finding was triggered by high-velocity, swirling winds that hit a single plot at the time of ignition, killing numerous trees. Still, unthinned plots had the highest rate of mortality during the study due to bark beetles (without fire) and crown scorch (with fire).

Shrub Cover

Bitterbrush, an important wildlife browse species in central Oregon (Gay 1998), was the dominant shrub at two sites, while snowbrush was dominant at Swede Ridge, the highest elevation site. No differences in total cover (fig. 4) or cover of individual shrub species (data not shown) resulted between 1993 and 2006 due to the slash treatments. An initial decline in shrub presence was noted after thinning (pre-thin cover averaged 28 percent), followed by a steady increase in shrub cover which, again, was independent of the slash treatment. Without thinning, the majority of shrubs were gradually eliminated by the end of the study.

Table 1—Stand density (trees per hectare) in 1991 and 2006 and percent mortality during the 15-year growth period for whole-tree removal (WT), bole-only removal (BR), thin no removal (NR), and no thin (NT) treatments. Plots were thinned in 1989 and burned in spring 1991 and 2002. Values are means (n = 3) plus standard errors in parentheses.

	No fire				Repeated fire			
	WT	BR	NR	NT	WT	BR	NR	NT
1991	276 (53)	278 (59)	256 (56)	615 (75)	298 (92)	266 (37)	274 (51)	603 (77)
2006	274 (51)	276 (57)	256 (56)	561 (42)	288 (92)	262 (30)	220 (42)	458 (23)
Percent mortality	1	1	0	9	3	2	20	24

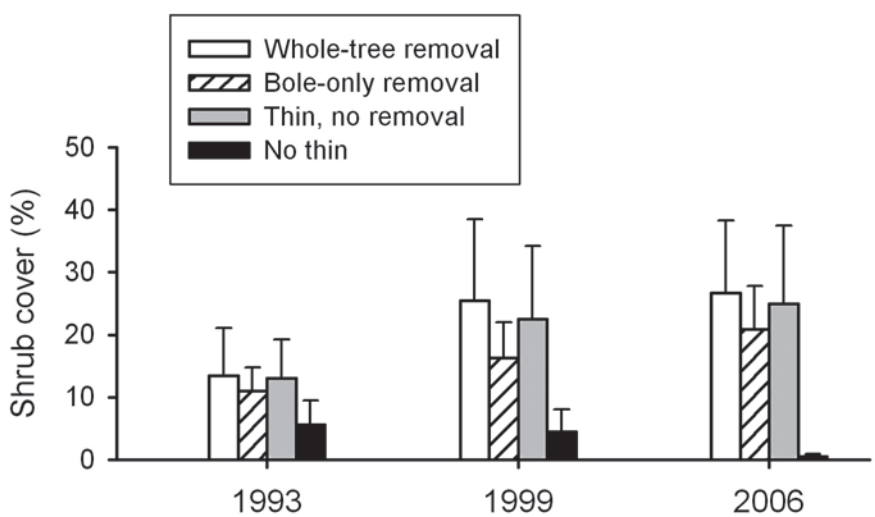

Figure 4—Effect of thinning and slash removal on shrub cover. Average shrub cover in 1988 prior to thinning was 28 percent. Error bars represent the standard error of the mean (n = 3).

Thinning, regardless of slash treatment, was particularly effective at stimulating bitterbrush cover (fig. 5). In comparison, thinning plus repeated fire led to a rollercoaster-like response by bitterbrush. Bitterbrush recovery was fairly rapid after the initial burn, with no statistical difference in cover detected between burned and unburned plots by 6 years after burning (Busse and Riegel 2009). However, the second entry of fire resulted in a near collapse of bitterbrush presence (fig. 5). Busse and Riegel (2009) surmised that this resulted both from extensive seed germination after the first burn, depleting the seed bank, and from the inability of bitterbrush germinants to produce a sizable seed crop in the 11 years between burns. This finding has relevance to the management of bitterbrush as both a desirable wildlife browse and as an unwanted ladder fuel, and it points to a few interesting possibilities for land managers to consider. If the presence of bitterbrush for wildlife habitat is preferred, then thinning alone may offer the best option. Alternatively, repeated burning offers the best option to eliminate bitterbrush from the understory and promote low wildfire hazard. For an ecological balance, a landscape-scale strategy that incorporates a mosaic pattern of thinning alone, thinning plus repeated fire, and occasional unthinned stands based on local knowledge of wildlife needs and wildfire hazard may be preferred.

Herbaceous Plants

Grasses and forbs were largely missing-in-action at the LTSP study sites. Low annual precipitation, the presence of competitive shrubs, and a pumice soil that challenges even the hardiest of seedbanks to survive with its diurnal temperature flux all contribute to low herbaceous production throughout much of the pumice

Figure 5—Stand conditions in 2006, 17 years after thinning, with (a) no thinning, (b) thinning, (c) thinning plus 1 burn, and (d) thinning plus 2 burns.

plateau. As testimony, herbaceous biomass was exceedingly low for every LTSP treatment except one during the study. This response held for the 3 slash-retention treatments, no thin plots, and thin plus burn plots (fig. 6). In concurrence, plant diversity was low and no differences were found among LTSP treatments (Busse and others 2009). Only the fertilizer plots (nitrogen + phosphorus + sulfur) produced sizeable herbaceous biomass for 2 to 3 years following application in 1991 and 1996. This observation suggests a few fact-busting possibilities: (1) fire is not essential to stimulating herbaceous plants in these forests, (2) poor herbaceous production on the pumice plateau is not necessarily related to an impoverished seedbank; instead, a large nutrient pulse is required to stimulate plant germination and growth.

Fuel Loading

Downed woody fuels were measured periodically between 1991 and 2007 to determine if sizeable or long-lasting differences in fuel loads existed between whole-tree removal and bole-only removal thinning practices. Does bole-only harvesting result in undesirable fuel loads from a wildfire hazard perspective? By the second growing season after thinning (1991), the fuel load of small-diameter wood (< 7.5 cm) on bole-only removal plots was about double the amount found on whole-tree removal plots (fig. 7). Five years later, however, the difference between treatments had subsided and, importantly, the absolute quantity of fuel for all treatments was low, and remained low throughout the study. The important results to consider are, (1) whole-tree removal was an effective means to maintain low fuel loads without prescribed burning, and (2) there was not a long-lasting difference in fuel loads between treatments, suggesting that the wildfire hazard due to retention of thinning slash is transient.

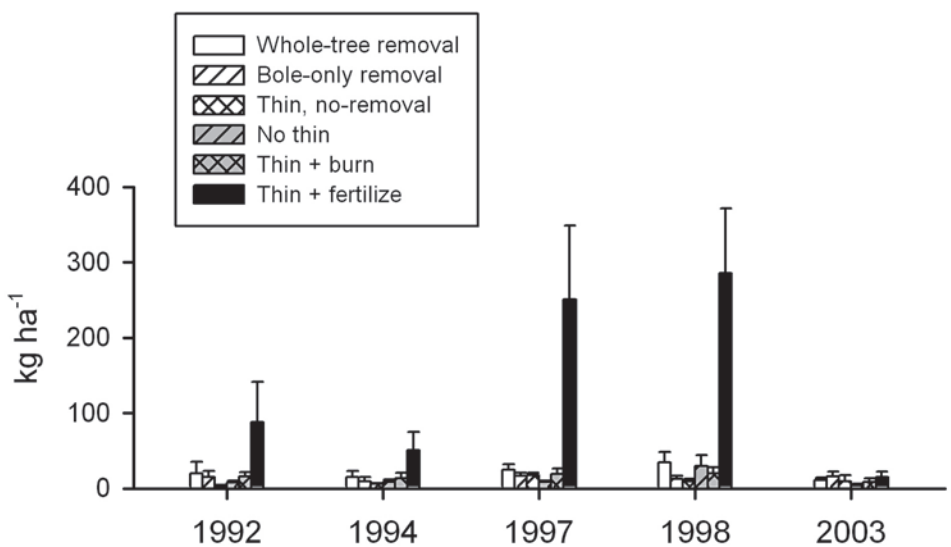

Figure 6—Herbaceous biomass production during peak growing season. Plots were thinned in 1989 and burned in spring 1991 and spring 2002. Error bars represent the standard error of the mean ($n = 3$).

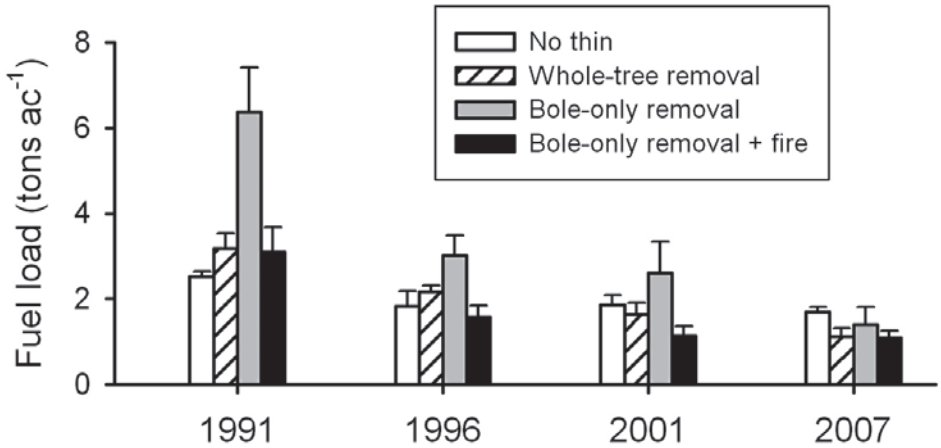

Figure 7—Changes in fine woody fuel mass (0 to 7.5 cm diameter) between 1991 and 2007. The fuel load in 1991 for "Bole-only removal + fire" was measured after the initial prescribed burn. Error bars represent the standard error of the mean (n = 3).

Soil Productivity

The slash-retention treatments had little or no effect on the soil properties by 2003, 14 years after thinning. In addition to the basic measures reported in table 2, no differences among treatments were detected in nutrient concentrations (Ca, Mg, K), microbial respiration and diversity, or duff depth. Similar results showing little effect of slash treatment were noted for soils collected earlier in the study (1993 and 1998). Some soil compaction due to the thinning operation was found on whole-tree removal plots (Parker and others 2007). However, the effects were not extensive, as no detrimental effects to trees or understory vegetation were detected on a plot basis (Busse and others 2009).

Table 2—Effect of thinning and slash removal on selected soil quality characteristics. Samples were collected from the surface 10 cm of mineral soil in 2003, fourteen years after thinning. Means plus standard errors in parentheses within a row followed by different letters are significantly different at α = 0.05 (Tukey's HSD).

Soil property	Whole-tree removal	Bole-only removal	Thin, no removal	No thin
pH	6.5 (0.2)	6.5 (0.1)	6.4 (0.1)	6.5 (0.1)
Carbon (g kg^{-1})	25.3 (3.6)	29.3 (4.0)	26.0 (3.6)	23.4 (3.2)
Nitrogen (g kg^{-1})	0.6 (0.1)	0.7 (0.1)	0.6 (0.1)	0.6 (0.1)
C:N	40 (2) ab	40 (1) ab	43 (2)a	37 (1)b
Phosphorus (mg kg^{-1})	39 (10)	47 (8)	44 (8)	47 (11)
Litter decay (percent mass loss in 2 years)	36 (3)	37 (8)	40 (2)	37 (4)
Microbial biomass (mg kg^{-1})	772 (106)	902 (174)	818 (124)	652 (28)

Conclusions

No differences in vegetation, wildlife browse, or soils due to whole-tree harvesting versus either bole-only harvesting or leaving all material on site were identified in 20 years of study. Thus, to answer Pat Cochran's original question about the importance of retaining site organic matter: thinning slash apparently does not need to be left on site in order to maintain site productivity. This suggests that single-entry thinning operations on the pumice plateau in central Oregon should treat slash residues based on practical concerns (type of logging equipment available, budget constraints), not based on ecological concerns for site productivity. The rapid reduction in fuel load found for the bole-only removal treatment additionally suggests that wildfire hazard due to post-thinning slash is a transient concern.

How universal are the study results? Certainly any inferences made beyond the geographical constraints of the Bend LTSP study in central Oregon should be made with caution. Differences in forest type, soil, climate, thinning prescription, or harvest equipment may all contribute to unique responses to slash treatment among forests and regions. In this regard, Page-Dumroese and others (2010) suggest that the value of retaining thinning slash is greatest for low fertility soils, those that would benefit from added sources of organic matter and nutrients. However, this recommendation was not supported by the LTSP results despite the low fertility of the region's pumice soils. Examination of the nutrient pools in central Oregon pine forests provides clarification for this assertion. Based on the nutrient profile reported by Little and Shainsky (1995) for the three LTSP sites, only about 4 percent of the total site nutrient pool was removed by whole-tree thinning compared to 1 percent by bole-only harvesting. Thus, removal of foliage and limbs during thinning accounted for only a 3 percent loss of the site nutrient pool, helping explain the lack of vegetation and soil responses to the slash treatments.

To summarize the key findings of the Bend LTSP study, (1) the retention of thinning slash was trivial to the health of central Oregon pine forests, (2) increased fuel loads for bole-only compared to whole-tree harvesting were transient, (3) thinning alone had a positive effect on wildlife browse and tree vigor without causing detrimental changes to understory vegetation or soil productivity, and (4) thinning plus repeated fire eliminated wildlife browse (bitterbrush), yet was an effective treatment for reducing wildfire hazard and maintaining site and soil productivity.

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The content of this paper reflects the views of the authors, who are responsible for the facts and accuracy of the information presented herein.

Stand Density Guides for Predicting Growth of Forest Trees of Southwest Idaho

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Abstract—This paper presents a method for estimating stand growth from stand density and average diameter in stands of pure and mixed species in Southwest Idaho. The methods are adapted from a model developed for Douglas-fir, ponderosa pine, and lodgepole pine on the Salmon National Forest. Growth data were derived from ponderosa pine increment cores taken from sample plots on the Boise and Payette National Forests. The model was tested against growth plots and permanent inventory plots located on the Boise and Payette National Forests. Model results closely fitted observed dbh's on these managed and unmanaged stands. These stands were summarized by group with average errors ranging from 0.8 to 2.9 percent (r^2 = 0.90 to 0.96).

Introduction

Stand density is a critical element in managing forests of the Western United States. Although managing stand density is commonly associated with timber production, it also affects wildlife habitat, water yield, and other forest resources. To manage forests, resource managers need accurate tools to predict how stand density affects future growth, which in turn affects various forest uses and values. The Relative Stand Density Index (Drew and Flewelling 1979) and Reineke's Stand Density Index (Gingrich 1967; Reineke 1933) provide useful formats in some situations.

Basford and others (1991) discussed the stand density considerations in forest treatments and the procedure of establishing and measuring permanent growth plots over time. Subjective decision processes based only on experience or related published data, are less time consuming but can lead to less accurate results. Consequently, inappropriate growing space applied during thinning treatments, or a delay in thinning, has caused reduced canopies and slower stand growth than expected.

To address these questions, a 1991 study on the Salmon NF developed stocking guides for Douglas-fir (*Pseudotsuga menziesii* var. *glauca* [Beissn.] Franco), ponderosa pine (*Pinus ponderosa* Dougl. Ex Laws.), and lodgepole pine (*Pinus contorta* var. *latifolia* Engelm.) (Basford and others 1991). The model developed for ponderosa pine is called the Salmon-Challis ponderosa pine model (SCPP), and using this model helped foresters on the Salmon NF make better predictions about growth expected after thinning.

In 1999, personnel from the timber staff on the Boise and Payette National Forests (NFs) requested that a study be undertaken to determine stand density recommendations for ponderosa pine, because current recommendations were not working. The majority of all 30 to 50 year old plantations on the Boise and Payette NFs had been thinned at least once in the last 20 to 30 years. Most did

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not respond to thinning treatments as desired due to inadequate spacing to allow stand release.

As a result of this concern, staffers adopted the approach that had worked to build the SCPP model. That result, the Boise-Payette ponderosa pine model (BPPP), predicts the future diameter growth and density of both thinned and unthinned stands in southwest Idaho. The model allows managers to closely replicate stand growth rates using an automated process that otherwise would be acquired only through repetitive permanent sample plots.

Development of the BPPP stand growth models for the southwest Idaho forests included the use of computer program (BKTHIN) that allows for data input, model processing and tabular output. The program is easy to use and does not require large amounts of complex data. The growth models are based on measurements from individual trees and their observed growth patterns. To validate the accuracy of the models, tests were run using stand data from permanent growth and inventory plots. The predicted growth was compared to actual measured growth. This computer program, BKTHIN, is an important and efficient tool to help manage forest stands to meet identified objectives.

Methods, Results, and Discussion

The following discussion is organized to first summarize data collection of sample, released, and "free-to-grow" trees, followed by a description of the development of mathematical models (BPPP and SCPP, within the BKTHIN computer program), and ending with a summary of model validation.

Data Collection

The growth equations for the Boise and Payette NFs were derived by analyzing increment cores, rather than by measuring permanent sample plots over time. Increment cores were taken from ponderosa pine trees in plots of different density levels on the two National Forests. A total of 133 plots were selected, representing an average diameter range of 2 inches up to 22 inches. Each plot contained a sample tree, surrounded by other trees of the same species, uniform in size and spacing, so that each sample tree experienced consistent competition on all four sides (fig. 1). All sample plots were free from mortality and from insects and disease. It was assumed that any reductions in growth rates would be related to density.

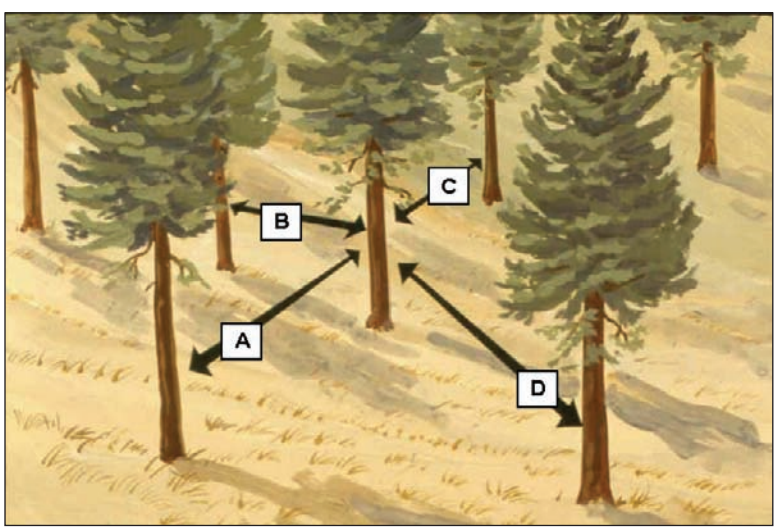

Figure 1—Distribution of sample tree to adjacent trees (A through D)

Sample tree selection and data collection followed procedures outlined in Basford and others (1991). Sample plots of uniform condition were located. Sample trees were selected from stands with a relatively narrow range of tree diameters and spacing, with at least four adjacent competing trees (fig. 1). Sample trees were selected so that the diameter differences between the sample tree and adjacent competing trees did not exceed one third of the diameter of the sample tree (for example, if the sample tree diameter at breast height was 9 inches, the diameter of the adjacent trees was between 6 and 12 inches.). Also, the distance between the sample tree and the four adjacent trees could not differ by more than 25 percent.

Increment cores were taken from the uphill and downhill side of each sample tree. Measurements on each increment core to record radial distance inside bark to each identified growth period were taken and then averaged (fig. 2). Diameter outside bark was calculated using the equation described by Husch and others (1972).

The average number of rings per inch (rpi) for each growth period was calculated by dividing the number of rings by the length of the core within that period, and then averaging the measurements from the uphill and downhill cores.

On each sample plot, site and tree data were collected. Data from sample plots of similar site quality (habitat type) were grouped, and the distances between the boles of the sample tree and adjacent trees were averaged. A linear regression line for the transition points between each growth period was drawn. For all regressions, graphs of residuals versus predicted values were examined, and standard deviations about the regressions were calculated as measures of reliability.

"Free-to-grow" trees—To establish potential diameter growth rates for trees without competition, large open-grown ponderosa pine trees were sampled. These were called "free-to-grow" trees. A total of 17 of these trees were sampled, 11 on the Idaho City Ranger District, Boise NF, and 6 on the Council Ranger District, Payette NF. Increment cores and bark thickness were taken from each sample tree, and rpi was calculated using the same tree sampling procedures as identified earlier.

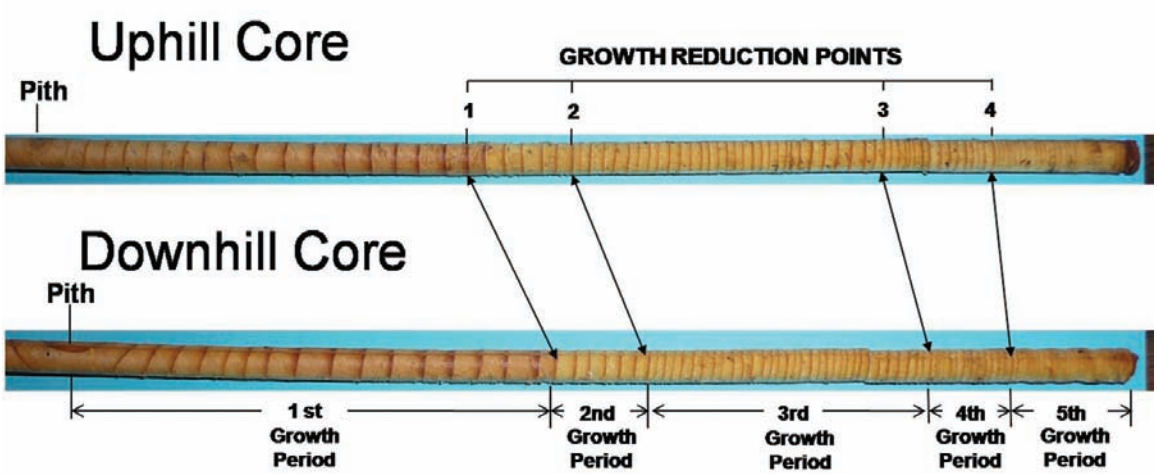

Figure 2—These cores are from a sample tree to illustrate growth rates, periods of uniform growth, and points at which growth reduction occurs. An additional 6th growth period was identified on other core samples. Rings per inch (RPI) are computed for each growth period.

Live crown ratio of released trees—Released trees are those that have been under competition at some previous point but, that at the time of measurement, were not competing with other trees. These trees had a smaller live crown ratio than open-grown trees of similar diameter, due to competition before thinning. Live crown ratios were measured and growth rates calculated in order to determine potential growth rates when a stand had been thinned and released from competition (table 1). This aspect is important to the model because it will determine what the maximum growth rate will be after thinning.

Analysis of Data

Sample tree—The cores from trees that met the strict sample criteria all showed identifiable transitions, usually over a period of two to three rings where growth slowed. After each identifiable growth reduction point (transition), the rings in the core will again grow at a fairly constant, although slower, rate. The cores in figure 2 show these growth reduction points and growth periods. If the strict sampling criteria were not met, these key transition points between growth periods, which are very gradual, and could be misidentified. The analysis also shows that the timing and diameter at these reduction points can be predicted (fig. 3).

Each of these five growth reduction points was grouped. From these groups, a linear regression was computed between average tree spacing (Y) and diameter (X) of the sample trees. The resulting equations represent lines at which the growth reduction is expected to take place as a stand grows, at any given density and average diameter (table 2).

Table 1—Crown ratio measurements. The spacing and average diameters were grouped into crown ratio classes and a regression calculated. The independent variable, X, is the tree diameter (inches). The dependent variable, Y, is tree spacing (ft).

Species	Crown ratio	Sample size	Crown competition equations	r^2	Predicted rings/inch after release	10 year diameter growth after release
	(percent)					*(inches)*
Douglas-fir	60 - 70	17	Y = 1.86 + 1.15X	0.85	8.4	2.4
(Salmon-Challis N.F.)	50 - 59	20	Y = 1.42 + 0.98X	0.79	8.8	2.3
	40 - 49	26	Y = 0.89 + 0.94X	0.68	10.4	1.9
	30 - 39	10	Y = 0.88 + 0.88X	0.81	12.8	1.6
	< 30		—	—	17.2	1.2
Ponderosa pine	60 - 70	7	Y = 0.46 + 1.14X	0.90	8.3	2.4
(Salmon-Challis N.F.)	50 - 59	12	Y = 0.45 + 1.04X	0.92	9.4	2.1
	40 - 49	18	Y = –0.22 + 1.02X	0.78	10.8	1.9
	30 - 39	11	Y = –1.29 + 1.04X	0.87	13.0	1.5
Lodgepole pine	50 - 70	38	Y = –0.39 + 1.32X	0.52	No Data	No Data
(Salmon-Challis N.F.)	30 - 40	15	Y = –1.26 + 1.26X	0.65	No Data	No Data
Ponderosa pine	60 - 80	55	Y = –0.50 + 1.35X	0.84	5.4	3.7
(Boise-Payette N.F.'s)	50 - 59	57	Y = –0.48 + 1.18X	0.71	6.9	2.9
	40 - 49	24	Y = –1.44 + 1.17X	0.82	6.9	2.9
	30 - 39	11	Y = –1.46 + 1.05X	0.85	7.4	2.7
	< 30		—	—	10.7	1.9

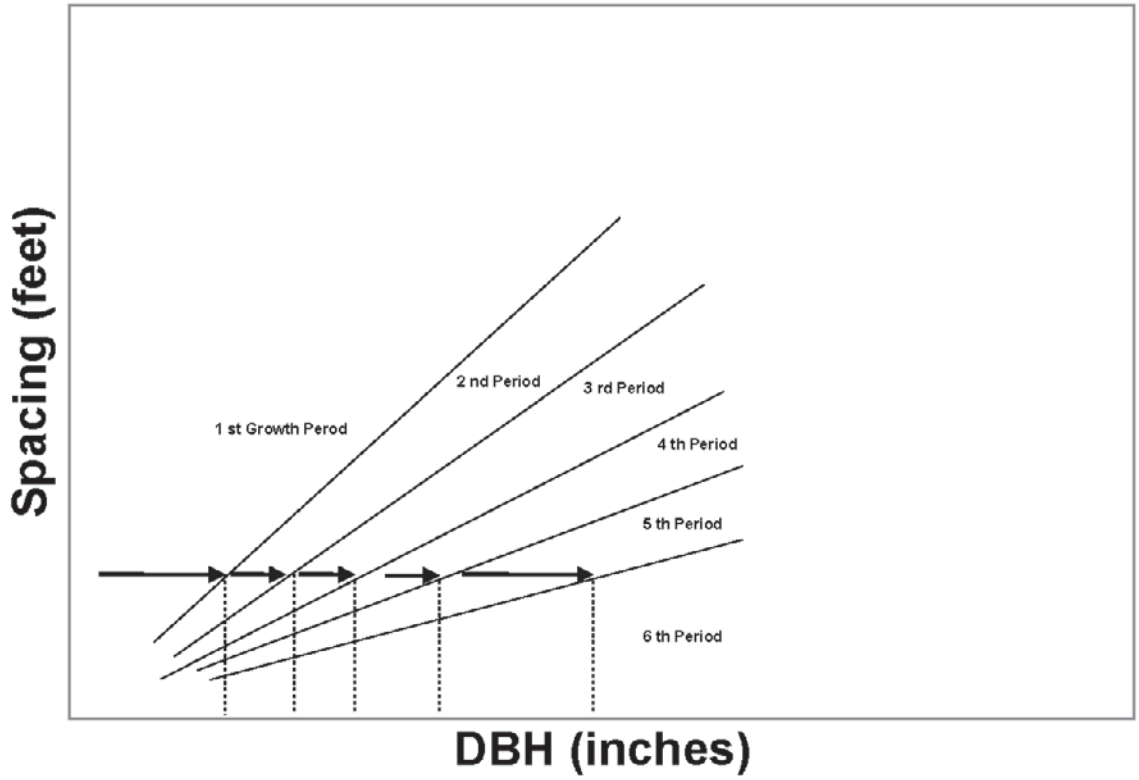

Figure 3—Spacing and diameter related to growth reduction lines and growth periods.

Table 2—Comparison of tree spacing / diameter regression equations. Regression equations represent the transitions between the maximun average growth rates. The independent variable, X, is the tree diameter. The dependent variable, Y, is tree spacing. The Boise-Payette equations are presented graphically in figure 4.

Species	Average growth rate (rings/in)	10 year diameter growth (inches)	Sample size	Range of data (dbh) (inches)	Equations (X = Diameter) (Y = Spacing)	r^2
Douglas-fir						
(Salmon-Challis N.F.)	8.4	2.4	36	1.1-14.3	Y = 1.11 + 1.85X	0.91
	16.0	1.3	29	2.4-21.2	Y = 1.10 + 1.25X	0.88
	29.6	0.7	7	5.6-14.2	Y = 0.08 + 1.01X	0.92
Ponderosa pine						
(Salmon-Challis N.F.)						
	8.3	2.4	25	1.0-11.9	Y = 2.68 + 1.58X	0.86
	16.4	1.2	21	2.2-16.2	Y = 1.09 + 1.23X	0.91
	31.6	0.6	8	4.1-20.8	Y = 0.59 + 1.02X	0.96
Lodgepole pine						
(Salmon-Challis N.F.)						
	8.9	2.2	39	1.6-12.4	Y = 1.63 +1.80X	0.9
	19.4	1.0	37	2.2-10.6	Y = 0.62 + 1.39X	0.81
	35.1	0.6	22	2.7-12.6	Y = –1.10 + 1.41X	0.89
Ponderosa pine						
(Boise and Payette N.F.'s)	5.4	3.7	63	2.3-16.4	Y = 1.72 + 1.72X	0.88
	9.6	2.1	30	4.4-22.1	Y = 1.72 + 1.43X	0.81
	13.4	1.5	19	2.5-17.5	Y = 1.57 + 1.24X	0.77
	23.1	0.9	14	3.2-19.7	Y = 1.48 + 1.01X	0.99
	44.3	0.5	7	3.6-19.7	Y = 0.60 + 0.93X	0.96

Free-to-grow—Data from the free-to-grow trees were used to determine the potential diameter growth rates in the absence of intertree competition as described above. Increment core analysis of the free-to-grow trees showed periods of relatively uniform diameter growth separated by points of identifiable growth reduction (table 3). Average diameter was computed at each growth reduction point and average growth rate was computed for each growth period. However, in these trees, growth reductions were due to tree physiology and occurred at larger diameters than those under competition.

The potential radial growth rates from free-to-grow trees were used to validate the average rpi for the first growth period of the sample trees. Sample trees that showed growth rates that were not within three rings of the potential rate for the site, as indicated by the free-to-grow trees, were rejected because their growth rates exceeded the identified variance. The reduced growth rates were likely due to an unidentified cause such as competition from nearby trees that had since disappeared.

Crown ratio of released trees—The last, and important, part of the analysis is the live crown ratio component. As stand competition increases, live crown ratios become smaller. Crown ratios and diameter growth rates were measured on trees that had been released from competition through thinning. The live crown ratios were grouped and the growth rates (rpi) within each group were averaged. It was found that when the live crown ratios are 60 percent or greater, the predicted growth rate closely reflected potential diameter growth rate (table 1). When the live crown ratio is less than 60 percent, the reduced photosynthetic base prevents the trees from reaching its predicted growth potential. When live crown ratio falls below 20 percent, little or no response is expected from a tree retained after thinning.

After calculating the predicted radial growth rate following thinning, the model compares it to the growth rate based on the crown ratio of release trees, and chooses the slower of the two. In this way, the model prediction would not allow a stand to grow faster than a reduced crown ratio would allow.

Computer Program Development

The mathematical models were programmed into a framework to facilitate user input, data processing and tabular output. The program was compiled using Microsoft Visual Basic®6.0. This computer application called BKTHIN can be run on most computers using Windows operating systems. The software and a user's guide can be obtained upon request. The minimum inputs are stand density and diameter. Remaining inputs can be used to further refine the results.

The BPPP Model

We compared results for the Boise-Payette model (BPPP) with three similar models from the Salmon-Challis National Forest (Salmon-Challis ponderosa pine = (SCPP), Salmon-Challis Douglas-fir = (SCDF), and Salmon-Challis lodgepole pine = (SCLP) (Basford and others 1991). The Boise-Payette model is based on more sample plots than the Salmon- Challis models, but the range of sample tree size is very close. The BPPP model generates faster growth rates than the other three, and has five growth reduction regression lines compared to three regression lines for the other models (table 2). The growth rates are slower in the Salmon-Challis models because they were developed from drier sites with 100-year site indices ranging from 60 to 80, whereas Boise and Payette NFs sites have 100-year site indices in the range of 80 to 115. These groups of growth reduction lines are limited by the size of sample trees available to sample. If larger sample

Table 3—Mean diameter growth rates during each growth period and the mean diameter at the growth reduction points for free-to-grow trees. The formula to convert rings/inch to diameter growth per decade = 20/rings per inch.

Ranger District/ Forest	Species[a]	Habitat Type	Site index mean[b]	No. of trees in sample	Growth reduction points						
					Rings/inch before 1st	dbh at 1st	Rings/inch between 1st and 2nd	dbh at 2nd	Rings/inch between 2nd and 3rd	dbh at 3rd	Rings/inch between 3rd and 4th
North Fork/SCNF[c]	PP	Psme/Caru-Pipo Psme/Cage-Pipo Psme/Syal-Pipo	75	11	7.5	13.2	12.4	19.8	19.8	26.1	26.8
Idaho City/BNF[de]	PP	Psme/Caru-Pipo Psme/Spbe-Pipo Psme/Syal-Pipo Psme/Cage-Pipo Psme/Phma-Pipo	109	11	7.4	22.3	11.5	30.9	20.2	36.4	32.1
Council/PNF[f]	PP	Abgr/Spbe Abgr/Caru	115	6	6.9	25.8	11.6	31.7	17.7	35.2	36.3
Salmon/SCNF	DF	Psme/Feid-Feid Psme/Caru-Feid	65	4	9.2	12.8	16.3	21.9	21.2		
Leadore/SCNF	DF	Psme/Caru-Caru	65	3	8.9	11.5	12.4	18.3	25.9		
Salmon/SCNF	DF	Psme/Caru-Caru	75	4	7.8	16.4	10.8	26.6	17.0	33.2	29.2
North Fork/SCNF	DF	Psme/Caru-Pipo	82	3	8.1	20.5	13.7	28.5	23.9	30.6	44.6
Salmon/SCNF	LPP	Psme/Caru-Caru	68	6	7.1	9.5	11.8				

[a] PP = ponderosa pine, DF = Douglas-fir, LPP = Lodgepole pine

[b] of 100-year base age

[c] SCNF = Salmon-Challis National Forest

[d] BNF = Boise National Forest

[e] Tree diameters on the BNF were larger than the SCNF resulting in the additional growth reduction points listed as follows: dbh at 4th = 42.2, rpi between 4th and 5th = 43.3, dbh at 5th = 46.6, rpi between 5th and 6th = 60.5, dbh at 6th = 52.4, rpi after 6th reduction = 77.5

[f] PNF = Payette National Forest

trees could have been found that met the sample criteria, more growth reduction equations would have been developed. The r^2 values for the BPPP model reduction equations for all models show that the data points (Y=spacing and X=dbh) form acceptable linear relationships.

The growth reduction equations in table 2 for ponderosa pine on the Boise-Payette NFs are graphed in figure 4. This figure shows the simplified results of the BPPP model. The solid lines represent the range of data. Dotted lines indicate where growth rates have been extrapolated. Using the average spacing and the average diameter, the growth of a ponderosa pine stand can be predicted using the graph. For instance, a stand grown for 80 years, starting with a mean diameter of 4 inches and a spacing of 16 feet, will grow at a rate of 5.4 rpi (3.7 inches/decade), reaching a mean diameter of 8.3 inches in the first growth period. In the second growth period, growth will slow to 9.6 rpi (2.08 inches/decade), and the stand will reach a mean diameter of 9.9 inches. Growth will then slow to 13.4 rpi (1.49 inches/decade) in the third growth period, and the stand will attain a mean diameter of 11.5 inches. Growth then slows to 23.1 rpi (0.87 inches/decade) in the fourth growth period with a mean diameter of 14.4 inches. Finally, in the fifth growth period the rpi slows to 44.3 rpi (0.45 inches/decade), reaching a mean diameter of 16.8 inches within the 80 years allowed by the model. The stand will have grown from 4 inches to 16.8 inches in an 80 year period without thinning. To predict the results of a thinning to increase the growth rate, simply move up the spacing axis to the new spacing and continue from there.

The other component of the BPPP model is live crown ratio. The live crown ratio observations were averaged and show that as crown decreases, tree diameter growth slows. This average is used in BPPP to determine the predicted growth of

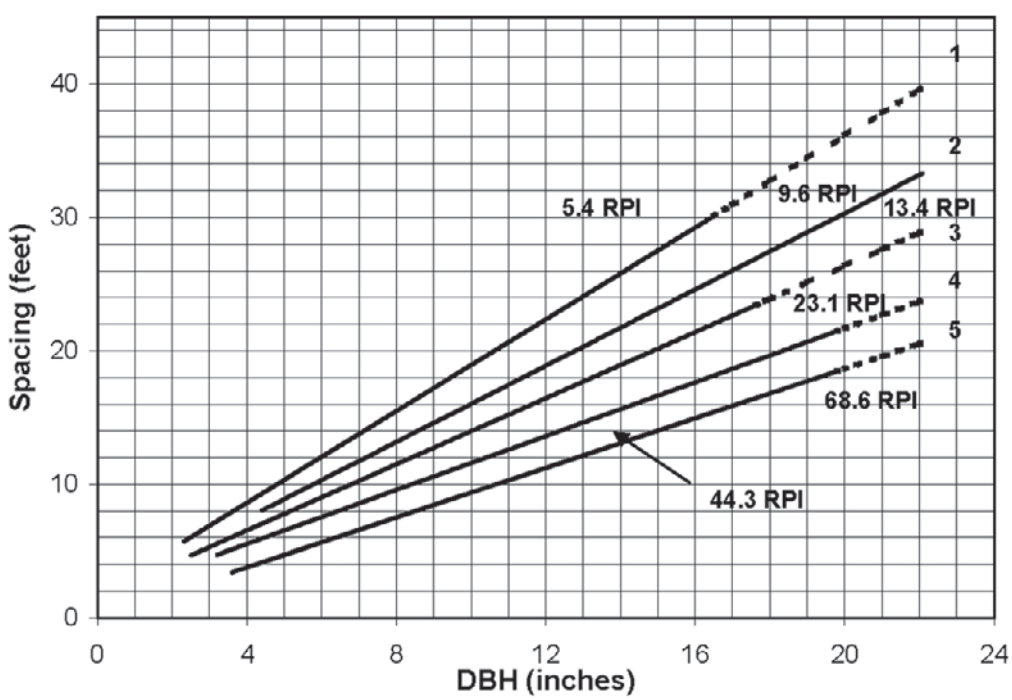

Figure 4—Ponderosa pine diameter growth rates (rings-per-inch, rpi) associated with spacing and tree size in the Boise-Payette National Forests. Equations for each regression line are presented in table 2. The solid lines represent the range of data and the dotted lines where growth rates have been extrapolated. The diameter growth before the first regression line (first growth period) is 5.4 rpi, second growth period 9.6 rpi, third growth period 13.4 rpi, fourth growth period 23.1 rpi, fifth growth period 44.3 rpi, and the sixth growth period 68.6 rpi.

a stand after its release from intertree competition. For example, if the live crown ratio is above 60 percent, then the predicted growth rate in the first growth period will be 5.4 rpi. Below 60 percent the growth rate slows. Below 30 percent live crown ratio, the expected growth rate should be only 10.7 rpi (table 1).

BKTHIN Output

Table 4 presents an example of the BKTHIN program output. All of the settings that have been entered are displayed in the heading of the output. In this example, using the Boise/Payette model, the beginning stand diameter was 4.2 inches at

Table 4—Example of an output from the Boise-Payette BPPP model. YR = Year; AGE = average stand age; DIAMETER = average diameter at breast height (inches); D-Gr RPI = diameter growth in rings per inch; T/A = trees per acre; HT = average height (ft); CU-FT = stand volume in cubic feet per acre; MAI = mean annual increment in cubic feet per acre; MBF = stand volume in 1000 board feet per acre; MAIBF = mean annual increment in board feet per acre; BA/A = basal area per acre; R-SDI = Reinke's stand density index; MCA= maximum crown area, CCF = crown competition factor; CR-% = crown ratio percent/10; CW = crown width in feet; CSEP = crown separation distance in feet; WPB = western pine beetle index from low (0), to high (11).

7/9/2005 Rev 4.0.9 8/24/2004 BOISE/PAYETTE NF BASFORD-KEENE PROGRAM
INITIAL STAND CONDITIONS
Stand Description: PP/DF Site Stand Table
Productivity Class: Medium
Height Relation: DBH/HT
Average Stand Diameter: 4.2
Optimum Diameter Growth: 5.4
Initial Spacing: 9.0 X 9.0 ft.
Stockability %: 100
Stand Age: 30
Years Projected: 27

Mortality Factor for 6 Growth Periods: .05, 0.1, 0.3, 0.3, 0.5, 0.5

THINNING INFORMATION
Distribution: User Specified
Thinning Diameters: 4.2, 12.1, 0.0, 0.0, 0.0 in.
After Thin Spacing: 18.0, 24.0, 0.0 ft
Minimum Cut MBF: 0.0

YR	AGE	DBH	D-G RPI	T/A	HT	CU-FT	MAI	MBF	MAIBF	BA/A	R-SDI	MCA	CR-%	CW	CSEP	WPB
0	30	4.2	5.4	538	24	0	0.0	0.0	0	52	134	51	7-9	7	1.9	0.0
COMM THIN REMOVAL				403		0		0.0	Thinned to: 18.0 X 18.0 Ft.							
1	31	4.6	5.4	134	26	0	0.0	0.0	0	16	39	14	7-9	8	10.5	1.0
2	32	5.1	5.4	134	28	0	0.0	0.0	0	19	45	16	7-9	8	10.1	1.2
3	33	5.5	5.4	134	30	0	0.0	0.0	0	22	51	18	7-9	8	9.7	1.2
4	34	5.9	5.4	134	32	0	0.0	0.0	0	26	58	20	7-9	9	9.2	1.2
5	35	6.3	5.4	134	34	0	0.0	0.0	0	29	64	23	7-9	9	8.8	1.2
6	36	6.8	5.4	134	35	0	0.0	0.0	0	33	71	25	7-9	10	8.4	1.2
7	37	7.2	5.4	134	37	301	13.7	1.3	244	38	79	27	7-9	10	8.0	1.2
8	38	7.6	5.4	134	39	392	17.1	1.3	233	42	86	30	7-9	10	7.6	1.2
9	39	8.0	5.4	134	41	494	20.6	1.3	224	47	94	32	7-9	11	7.1	1.2
10	40	8.5	5.4	134	43	606	24.2	1.3	215	52	102	35	7-9	11	6.7	1.2
11	41	8.9	5.4	134	44	728	28.0	1.3	206	58	111	38	7-9	12	6.3	1.2
12	42	9.3	5.4	134	46	861	31.9	1.3	199	63	119	41	7-9	12	5.9	1.2
13	43	9.7	9.6	133	48	1006	35.9	1.3	192	69	128	44	7-9	13	5.5	1.2
14	44	10.0	9.6	133	49	1092	37.6	1.3	185	72	133	46	7-9	13	5.2	1.2
15	45	10.2	9.6	133	50	1181	39.4	1.6	187	76	138	47	7-9	13	5.0	1.4
16	46	10.5	9.6	133	51	1274	41.1	2.1	199	79	143	49	7-9	13	4.8	1.4
17	47	10.7	9.6	133	52	1371	42.8	2.7	211	83	148	51	7-9	13	4.5	1.4
18	48	10.9	9.6	133	52	1471	44.6	3.3	223	87	153	53	7-9	14	4.3	1.4
19	49	11.2	9.6	133	53	1576	46.3	3.9	234	90	159	55	7-9	14	4.1	4.8
20	50	11.4	13.4	132	54	1684	48.1	4.6	246	94	164	57	7-9	14	3.8	4.8
21	51	11.6	13.4	132	55	1760	48.9	5.1	252	97	167	58	7-9	14	3.7	4.8
22	52	11.8	13.4	131	56	1838	49.7	5.5	258	99	171	59	7-9	15	3.5	4.8
23	53	11.9	13.4	131	56	1917	50.5	6.0	264	102	174	61	7-9	15	3.3	4.8
24	54	12.1	13.4	131	57	1999	51.2	6.5	270	105	178	62	7-9	15	3.2	4.8
COMM THIN REMOVAL					55	839		2.7	Thinned to: 24.0 X 24.0 Ft.							
25	55	12.5	5.4	76	58	1283	53.0	4.5	181	65	109	38	7-9	15	8.7	1.4
26	56	13.0	5.4	76	60	1420	55.1	5.3	197	69	114	40	7-9	16	8.3	1.4
27	57	13.4	9.6	75	62	1565	57.2	6.2	213	74	120	42	7-9	16	7.9	1.4

age 30, and rate of growth was 5.4 rpi. Initial spacing was 9- by 9-feet with two projected thinnings (using user-specified spacing): one initially at 4.2 inches (18- by 18-feet spacing), and the second at 12.1 inches in diameter (24- by 24-feet spacing). The output is given in a year by year summary. After "Year 0" the stand was thinned to a spacing of 18- by 18-feet removing 403 trees per acre. Yearly updates of stand conditions are reported. After "Year 24," at stand age 54, and mean diameter 12.1 inches, a second thinning removed 55 trees per acre and the final spacing was 24- by 24-feet. The table ends in "Year 27" with a mean stand diameter of 13.4. If no thinning had taken place in this example the resulting average stand diameter would have been 7.7 instead of 13.4 inches.

Model Validation

The model results were validated using data from four sets of inventory and permanent growth plots in thinned and unthinned stands. The data used for validation include average density (trees per acre), number of years to last measurement, and average beginning diameter. This data was entered into the BPPP to calculate growth. The computed ending diameter was then compared to the measured mean diameter and percent error was calculated. For thinned stands, the measurement periods went from the time of thinning to the final inventory. For the unthinned plots, the measurement period ran from age 0 (dbh) to the age of the stand at final measurement. To validate, the actual average diameters were compared to those predicted by the model. Each line in tables 5 to 8 represents an individual stand and a model prediction for that stand.

The first set of data came from permanent growth plots on the Payette NF (Unpublished data on file at the Supervisors Office, Payette National Forest, McCall, Idaho. 2003a). These 61 plots contained mixed species and pure stands of ponderosa pine. They were thinned then re-measured 12 to 21 years later. The average thinned spacing ranged from 13.5 to 28.7 feet. Comparing the computed

Table 5—Summary of actual versus predicted diameter growth for a 12 to 21 period on 61 thinned permanent growth plots of Payette NF ponderosa pine and mixed stands using the Boise-Payette model. Percent growth error was calculated according to (predicted diameter - ending diameter)/ending diameter. The r^2 value for actual diameter vs. predicted diameter is 0.91.

Species	Number of stands	Trees/acre	Spacing (ft)	Number of years for run	Diameter spread (in)	Average dbh (inches) Initial	Ending	Predicted	Percent error (%)
PP	34	85 - 213	13.5 - 22.6	15	3.0 - 23.3	7.2	11.4	11.3	–0.9
PP,DF	10	67 - 200	14.8 - 25.5	15	3.5 - 22.9	8.0	11.5	11.8	2.6
PP,DF	3	113 - 167	16.2 - 19.6	12	4.3 - 19.0	11.8	12.8	13.7	7.0
PP,DF	1	93	21.6	21	7.7 - 27.4	14.8	18.2	17.5	–3.8
PP,DF,LP	1	167	16.2	15	3.7 - 17.2	8.9	11.9	11.8	–0.8
PP,WL	1	213	14.3	15	5.3 - 11.2	4.7	8.9	9.4	5.6
PP,LP	1	180	15.6	15	4.0 - 17.1	8.2	12.7	11.3	–11.0
PP,GF	1	140	17.6	15	7.2 - 16.2	5.5	10.3	10.8	4.9
PP,WL,GF	1	160	16.5	15	4.4 - 9.8	5.3	9.6	10.4	8.3
GF,DF	4	53 - 127	18.5 - 28.7	15	6.7 - 27.1	15.1	17.0	17.1	0.6
ES,WL	2	100 - 127	18.5 - 20.9	15	5.5 - 20.0	12.0	14.0	14.6	4.2
ES	1	87	22.4	15	9.5 - 16.8	11.9	13.6	15.3	12.5
ES,DF,PP,GF,WL	1	147	17.2	15	4.1 - 13.1	3.7	8.9	9.7	9.0
Average across all stands	61	53 - 213	13.5 - 28.7	12 - 21	3.0 - 27.4	8.3	12.0	12.1	0.8

Table 6—Actual versus predicted diameter growth on 24 permanent control growth plots of Payette NF ponderosa pine and mixed stands using the Boise-Payette model. These untreated stands are control plots for the treated stands in table 5. Percent error was calculated according to (predicted diameter - ending diameter)/ending diameter. The r^2 value for actual vs. predicted diameter is 0.97.

Species	Trees/acre	Spacing (ft)	Number of years	Diameter spread (in)	Average dbh (in) Initial	Ending	Predicted	Error (%)
PP	253	13.1	12	3.5-7.8	0	5.1	5.1	0.0
PP	367	10.9	11	3.5-7.4	0	5.0	4.7	–6.0
PP	520	9.2	11	3.4-7.4	0	4.6	4.7	2.2
PP	167	16.2	11	3.4-5.4	0	4.4	4.7	6.8
PP,DF,ES	427	10.1	28	3.3-13.4	0	8.0	7.6	–5.0
PP	380	10.7	14	3.5-8.4	0	5.6	5.8	3.6
PP	293	12.2	21	3.7-10.1	0	7.3	7.7	5.5
PP,DF	493	9.4	60	3.4-16.6	0	9.1	9.2	1.1
PP	327	11.5	11	3.5-7.0	0	4.6	4.7	2.2
PP,GF,DF	327	11.5	58	3.6-19.4	0	11.2	10.7	–4.5
PP,DF	213	14.3	81	5.4-27.1	0	13.9	14.0	0.7
PP,DF	380	10.7	24	3.3-9.6	0	6.6	7.6	15.2
PP	347	11.2	12	3.5-11.4	0	5.1	5.1	0.0
PP	293	12.2	10	3.4-7.4	0	4.5	4.3	–4.4
PP	320	11.7	29	3.4-12.0	0	7.9	8.6	8.9
ES,WL,LP.DF	527	9.1	69	3.6-17.7	0	9.3	9.3	0.0
GF,DF,ES	473	9.6	69	3.2-17.9	0	10.0	9.8	–2.0
PP	573	8.7	26	3.5-11.1	0	6.0	6.6	10.0
PP	620	8.4	30	3.4-14.0	0	7.2	6.9	–4.2
PP,DF	407	10.4	60	3.4-18.2	0	9.0	9.9	10.0
PP,GF	253	13.1	12	3.4-7.5	0	4.8	5.1	6.2
PP,GF	347	11.2	18	3.6-8.3	0	5.9	6.7	13.6
PP,WL	167	16.2	12	3.3-5.4	0	4.0	5.1	27.5
PP,DF	327	11.5	56	3.4-19.4	0	10.9	10.5	–2.8
Average	366.7	11.4	31.0		0	7.1	7.3	2.8

Table 7—Actual versus predicted diameter growth on 19 untreated mixed plots from Timber Inventory data collected on the Payette National Forest using the Boise-Payette model. The plot measurement interval ranges from 57 to 114 years. Percent error was calculated according to (predicted diameter - ending diameter)/ending diameter. The r^2 value for actual vs. predicted diameter is 0.90.

Species	Trees/acre	Spacing (ft)	Number of years	Diameter spread	Average dbh (in) Initial	Ending	Predicted	Error (%)
GF,DF,PP,ES	213	14.3	57	5.5 - 19.3	0	12.3	12.8	4.8
DF,PP	235	13.6	74	4.0 - 38.0	0	11.7	13.2	11.9
DF	263	12.9	97	7.0 - 40.1	0	11.8	13.6	15.1
PP,DF,GF	226	13.9	81	0.0 - 46.0	0	13.9	13.7	–0.7
DF,PP	277	12.5	70	6.7 - 24.4	0	11.9	12.1	1.7
AF,ES,LP,DF	423	10.2	69	5.5 - 28.8	0	10.0	10.2	2.0
GF,PP,DF	143	17.5	77	5.9 - 34.1	0	16.8	16.2	–3.0
GF,DF,PP	550	8.9	76	0.5 - 33.0	0	9.3	9.4	1.0
GF,PP,DF	363	11.0	82	0.0 - 25.1	0	11.4	11.4	0.0
ES,LP,DF,WL,AF	654	8.2	66	0.0 - 16.1	0	8.1	8.4	3.7
GF,PP,ES,DF	120	19.1	62	0.0 - 34.8	0	16.8	16.0	–4.2
GF,PP,DF	255	13.1	68	0.0 - 23.0	0	11.5	12.4	7.8
GF,DF,PP	490	9.4	89	0.0 - 28.5	0	10.0	10.3	3.0
GF,LP,ES,DF	403	10.4	63	0.0 - 47.8	0	10.9	10.1	–7.3
GF,ES,DF	242	13.4	109	1.0 - 38.8	0	14.1	14.5	2.8
GF,DF	502	9.3	70	1.0 - 23.0	0	10.1	9.6	–5.0
GF,ES,LP,WL	647	8.2	59	0.0 - 25.4	0	7.6	8.2	7.9
ES,AF,LP	793	7.4	83	5.6 - 18.7	0	9.8	8.3	–15.3
GF,DF,ES,PP	273	12.6	114	2.0 - 31.7	0	13.2	13.9	5.3
Average	372.2	11.9	77.2		0	11.6	11.8	1.7

Table 8—Actual diameter growth on 18 treated ponderosa pine permanent growth plots on the Boise National Forest is compared to predicted growth using the Boise-Payette model. Percent error was calculated according to (predicted diameter-ending diameter)/ending diameter. The r^2 value for actual vs. predicted diameter is 0.96.

Species	Trees/acre	Spacing (ft)	Number of years	Diameter spread	Average dbh (in) Initial	Ending	Predicted	Error (%)
PP	380	10.7	14	3.4-7.4	0.0	5.0	5.8	16.0
PP	480	9.5	15	4.8-18.4	5.1	7.2	7.2	0.0
PP	140	17.6	20	6.9-14.3	5.8	11.7	12.0	2.6
PP	440	10.0	20	5.5-14.4	4.6	8.7	7.8	–10.3
PP	220	14.1	15	4.4-11.6	5.6	9.1	9.6	5.5
PP	340	11.3	15	4.5-10.8	5.8	8.3	8.4	1.2
PP,DF	300	12.1	20	4.2-14.2	5.2	8.6	9.1	5.8
PP	380	10.7	20	5.0-10.8	4.7	8.0	8.3	3.8
PP	280	12.5	12	3.5-5.8	0.0	4.5	5.1	13.3
PP	160	16.5	9	3.1-5.7	0.0	3.9	3.8	–2.6
PP	220	14.1	12	7.0-11.1	5.0	8.3	8.9	7.2
PP	140	17.6	10	3.4-4.9	0.0	4.3	4.3	0.0
PP	80	23.3	9	3.0-5.3	0.0	4.0	3.8	–5.0
PP	120	19.1	12	5.0-8.3	4.5	8.8	9.6	9.1
PP	60	26.9	12	8.2-8.8	4.3	8.6	9.4	9.3
PP	60	26.9	9	3.2-5.2	0.0	4.3	3.8	–11.6
PP	120	19.1	12	5.2-9.8	4.2	8.2	9.3	13.4
PP	80	23.3	11	3.7-5.4	0.0	4.4	4.7	6.8
Average	222.2	16.4	13.7		3.0	7.0	7.3	2.9

stand growth output to the Payette permanent growth plot data revealed the model did a very good job of predicting stand growth in thinned stands, with an error between predicted and observed data of less than one percent (table 5).

The second set of plots used for validation was a group of 24 permanent growth plots from the same area (Unpublished data on file at the Supervisors Office, Payette National Forest, McCall, Idaho. 2003b). These had similar composition but they were unthinned control plots. Average spacing in these plots was between 9.1 and 16.2 feet. Comparison of the computed diameter to the actual measured diameter resulted in a percent error of less than 3 percent (table 6).

The third set of data from the Payette NF inventory plots contained 19 stands of mixed composition of which 11 plots contained ponderosa pine (Unpublished data on file at the Supervisors Office, Payette National Forest, McCall, Idaho. 2003c). None were pure ponderosa pine. The other species were Douglas-fir, grand fir (*Abies grandis* (Douglas ex D. Don) Lindl.), Engelmann spruce (*Picea engelmannii* Parry ex Engelm), subalpine fir (*Abies lasiocarpa* (Hook.) Nutt. var. *lasiocarpa*), western larch (*Larix occidentalis* Nutt.), and lodgepole pine. These stands have never been thinned. Again, the predicted average stand diameter was close to the actual measured stand diameter, with an error of less than 2 percent (table 7).

The final data set was from Boise NF permanent growth plots (Unpublished data on file at the Supervisors Office, Boise National Forest, Boise, Idaho. 2003). These 18 ponderosa pine stands were thinned to a range of 9.5 to 26.9 feet and re-measured anywhere from 9 to 20 years later (table 8). The results were similar to those from the Payette NF, with an error between observed and predicted dbh of less than 3 percent.

BPPP seems to work well in mixed forest types of southwest Idaho, even when ponderosa pine is a minor component of the stand. Stand density (range, 7.4 ft to 28.7 ft) and number of years of stand growth did not seem to affect the error.

It could be expected that the error would increase over time; however, the comparison showed little difference between short and long term projections. One stand grew for 109 years and the error was 2.8 percent. Another BPPP projection of 82 years predicted diameter exactly as observed. Also, there did not seem to be much difference in error when predicting small versus large average diameter stands (the initial diameter range was from 0 to 17.9 in). Comparing all four tables together, only 17 percent of the plots had an error between observed and predicted dbh greater than 10 percent, whereas 51 percent were within 5 percent of the observed diameter.

Other Growth Models

It was shown by Basford and others (1991) that the SCPP model could effectively be used to predict ponderosa pine growth in other dry locations outside the southwest Idaho region. A direct comparison between actual growth data and predicted growth of ponderosa pine on 82 plots produced an r^2 of 0.98 in northern Arizona (Ronco and others 1985), the Blackhills of South Dakota (Edminster, C.B. 1988. [Unpublished data]. Fort Collins, CO: USDA Forest Service Rocky Mountain Research Station), central Oregon (Barrett 1982), Pringle Falls, Oregon (Cochran and Barrett 1999a), Lookout Mountain in Oregon (Cochran and Barrett 1999b), the Methow Valley in Washington (Cochran and Barrett 1998), and the Malheur NF in Oregon (Cochran and Barrett 1995). Only 10 percent of the plots showed an error of more than 10 percent. Sixty-eight percent of the plots were within 5 percent. The data is summarized by location in table 9.

Another computer modeling system called the Forest Vegetation Simulator (FV*S*) (Dixon 2006) also allows users to predict densities and diameters. It is reasonable for users to compare BKTHIN to FVS, but this should be done cautiously as these tools have different purposes with different levels of input. The FVS is a stand level, growth and yield system, using a variety of site and tree conditions within specific vegetative classes such as habitat type or site index within a variant region. Individual tree sample diameters are input, incremented, and compared with other tree samples in the plot. Each cycle interval (i.e. 10 years) is based on all tree samples in the plot. Tree specific information as well as stand summary information is reported.

Table 9—Actual versus predicted diameter growth (inches) on 82 ponderosa pine growth plots using the Salmon-Challis model. It was tested against the following studies: northern Arizona (Ronco and others 1985), Black Hills, South Dakota (Edmininster 1988), Pringle Falls, Oregon (Cochran and Barrett 1999), Lookout Mountain, Oregon (Cochran and Barrett 1999), Methow Valley, Washington (Cochran and Barrett 1998), and Malheur N.F., Oregon (Cochran and Barrett 1995). Percent error was calculated according to (predicted diameter - ending diameter)/starting diameter. The r^2 value for actual vs. predicted diameter is 0.98.

Study area	Species	Number of stands	Range of trees/acre	Range of spacing (ft)	Number of years	Average dbh (in)			Error (%)
						Initial	Ending	Predicted	
Arizona	PP	12	61 - 681	8.0 - 26.7	10	6.6	8.5	8.2	–3.5
South Dakota	PP	24	41 - 617	8.4 - 32.5	10	6.0	7.8	8.1	3.8
Pringle Falls, Oregon	PP	5	62.5 - 1000	6.6 - 26.4	35	2.0	9.0	9.1	1.1
Lookout Mountain, Oregon	PP	18	22 - 308	11.9 - 44.5	10	12.6	14.4	14	–2.8
Methow Valley, Washington	PP	5	62.5 - 2387	4.3 - 26.3	30 to 35	4.3	9.1	10.0	9.9
Malheur N.F, Oregon	PP	18	56 - 455	9.8 - 27.5	4 to 10	8.7	9.8	10.0	2.0
Average	PP	82	22 - 2387	4.3 - 44.5		7.8	9.9	9.9	1.0

Conversely, BKTHIN is a tree species level method, driven by spacing and diameter criteria that can be applied to broader regions. The average stand diameter is used as the initial input for the model and that diameter is incremented yearly over the planning horizon based on growth regression equations. Only stand summary information is reported at each one-year cycle.

A simple comparison was completed to see how both methods predict diameter and growth (diameter increment) for managed permanent growth plots on the Payette NF. These plots were initially inventoried in 1983 and 1988 with three to four 5-year re-measurements. The model was projected for a 15-20 year period using both BKTHIN and FVS. Table 10 displays the results of these projections as compared to the actual beginning data and the actual last re-measurement. Both methods started with the same initial QMD for a given plot. The diameter increment results are recorded for the actual, FVS, and BKTHIN, which are computed by calculating the change from the beginning to ending diameter and then dividing the result by the total years. Finally, all plot results were averaged by habitat type group. It may not be clear which method predicts most closely to actual. This is not a large data set, but both approaches appear to be reasonably similar. It would be interesting to conduct this comparison between FVS and the empirical approach reported here across a larger and more comprehensive data set.

Management Application

The BPPP model is based on average size/density measurements. It allows thinning of stands to a given density, assuming growing stock is fairly uniform throughout the total area. It also uses crown ratio when considering whether the trees can fully respond to release. The model works best with even-aged stands, but data in table 7 suggest that it also works in uneven-aged stands. It predicts growth in unmanaged stands and has been found effective in pure or mixed stands of ponderosa pine, Douglas-fir, grand fir, subalpine fir, Engelmann spruce, western larch, and lodgepole pine.

BPPP can meet the needs of managers on the Boise and Payette NFs. The BKTHIN program can be used to maximize tree diameter through thinning at the right times before growth rates slow. It can also maximize stand volume by accepting slower growth rates but allowing more trees to better utilize the site.

Table 10—Actual versus predicted diameters (inches) from permanent plot data using FVS and BKTHIN are summarized by habitat type and number of years between measurements. “Average diameter increment” represents the average annual growth rate.

Habitat type	Years projected	Number of stands	Ave. actual diameter		Ave. predicted QMD 2003		Ave. diameter increment		
			Start	End	FVS	BKTHIN	Actual	FVS	BKTHIN
Abgr/Acgl/Phma	20	3	4.93	10.43	11.27	11.27	0.28	0.32	0.32
Abgr/Libo	15	3	8.90	12.60	11.73	11.40	0.25	0.19	0.17
Abgr/Spbe	15	3	9.60	13.67	12.37	11.70	0.27	0.18	0.14
Abgr/Spbe	20	8	5.39	12.64	12.24	11.76	0.37	0.33	0.32
Abgr/Vagl	15	3	7.37	11.03	11.10	10.63	0.24	0.25	0.22
Pipo/Syal	20	3	5.40	10.63	12.13	12.60	0.26	0.34	0.36
Psme/Caru/Pipo	15	3	9.30	13.13	12.23	11.90	0.26	0.20	0.17
Psme/Caru/Pipo	20	3	4.57	10.90	11.50	12.27	0.32	0.35	0.39
Psme/Spbe	15	3	7.17	11.00	10.97	11.40	0.25	0.26	0.28
Psme/Syal/Pipo	15	7	9.49	12.73	12.36	12.24	0.21	0.19	0.19
Psme/Syal/Pipo	20	3	8.83	13.20	12.43	12.90	0.22	0.18	0.21

When spacing is to be kept relatively uniform, the size of the trees at harvest dictates the number and spacing of trees left after each thinning. Lack of consideration for final tree size and the timing of each intermediate harvest could result in reduced stand growth.

This model has several advantages. It can accurately predict diameter growth in forest stands of southwest Idaho. The input required to operate the model is very simple and easily obtained from National Forest stand inventory plots. BKTHIN is fast, easy to use and can be used by managers in the field. The methods reported above can be easily replicated for use in other regions.

With BKTHIN it is possible to manage diameter growth in even-aged stands by scheduling treatments to meet management objectives. However, as others have reminded us (Ernst and Knapp 1985, Leak 1981), we must continue to evaluate recommended stocking levels (stand density) to see how close they come to satisfying our management objectives. Through this process, refinements can be made and the stocking guides can be improved. For a copy of the BKTHIN computer model, please contact one of the authors.

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Built for the Future: New Directions in Silviculture Research and Demonstration at Montana's Lubrecht Experimental Forest

Christopher R. Keyes[1] and Thomas E. Perry[1]

Abstract—Manipulative experiments at the University of Montana's Lubrecht Experimental Forest have long been set aside as permanent research and demonstration areas (RDA's) to communicate the tradeoffs among different stand management strategies. However, most of these have either degraded over time or have diminished relevance to contemporary forest management issues. An evaluation and rehabilitation of Lubrecht Forest's research and demonstration infrastructure is currently underway. Examples are presented of existing RDA's that are being refurbished, replaced, revised, or retired. New demonstration areas that exemplify the central theme of this rehabilitation effort—stand complexity and regeneration—are also described.

Keywords: silvicultural systems, stand structure, complexity, regeneration, reforestation.

Introduction

From its inception, the University of Montana's Lubrecht Experimental Forest (LEF) has included in its mission the study and demonstration of silvicultural practices and systems. At this 28,000-acre forest laboratory, manipulative experiments have been set aside as permanent reserves, or research/demonstration areas (RDA's), to communicate the tradeoffs among varying stand management strategies for ubiquitous forest types of the northern Rocky Mountains. Lubrecht Forest's RDA's have been widely utilized by many agencies, organizations, and landowner groups over the years, as well as legions of University of Montana students. But few new RDA's have been created since the 1980s. Many existing areas have suffered degradation, declined in their interpretative value, and lost much of their relevancy to contemporary issues.

An evaluation and rehabilitation of Lubrecht Forest's research and demonstration infrastructure is currently underway. Stand complexity and regeneration are the central themes that underlie this forest-wide rehabilitation effort. It is an effort that is designed to equip Lubrecht with a deliberately prepared suite of relevant RDA's, and a plan for their use and maintenance that will sustain this experimental forest's relevance into the future.

Lubrecht Forest Management

Lubrecht Forest was established in 1937, a gift from the Anaconda Company to the State of Montana under the auspices of the Montana Forest & Conservation Experiment Station (MFCES). Through subsequent additions, LEF now totals

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28,000 acres of forest and range. Permanent study areas date back to 1948, but most RDA's were established after 1981 when the state legislature established the Mission-Oriented Research Program, an MFCES program that would continue to 2007 and provide much of the impetus for research and demonstration occurring at LEF. Today the MFCES's Applied Forest Management Program, initiated in 2007, oversees the long-term studies and demonstration areas at Lubrecht and leads the development of new research and demonstration opportunities.

Lacking a full management plan, Lubrecht Forest's management framework is summarized in a 1996 brief, titled "LEF Management Guidelines." LEF's stated mission (according to that document) is, "Natural resource study, demonstration, and learning, and public use in a forest setting." This goal is to be met via five stated management goals (underlining and italics added for emphasis):

1) Serve Montana citizens through <u>research</u>, <u>demonstration</u>, instruction, and public use.
2) Be available for a wide variety of <u>research studies</u>, <u>demonstrations</u>, and learning opportunities.
3) <u>Demonstrate</u> management which leads to healthy, sustainable ecosystems
4) <u>Demonstrate</u> a variety of forest management techniques applicable to ecosystems in the northern Rocky Mountains.
5) Generate revenue to support MFCES activities through the sale of goods and services *in a manner consistent with other goals* for the forest.

Operational management at LEF is best described as a conservative custodial model, with origins as a young regrowth forest with limited economic value. Much of the forest consists of same-aged thinned or unthinned forests. Over time, this custodial management model has changed little, but the forest itself has changed dramatically through growth and succession. Young stands on the forest are rare, the result of infrequent regeneration harvests or occasional stand-replacing disturbance events. The limited emphasis on strategic regeneration harvesting has limited structural diversity across LEF and has fostered the development of unmanaged understory cohorts that are uniformly dominated by Douglas-fir. These unfortunate consequences demand an innovative silvicultural approach to their remediation. Recently, substantial and widespread bark beetle mortality is forcing a more deliberate management focus on the next generation forest. A dynamic approach to promoting resilience and adaptability in that forest is a top priority.

Lubrecht's Silviculture RDA's

Lubrecht has a notable collection of long-term installations for research and demonstration of silvicultural practices. Although Lubrecht has been the site of numerous studies, relatively few of these have been established as permanent installations. Some of Lubrecht's RDA highlights are frequently used by diverse users, and hence they are considered among the most important for protection and maintenance. Two examples are the Fire & Fire Surrogates study area and the Uneven-aged Silviculture study areas. Part of a national network, LEF's large Fire & Fire Surrogates study, established in 2002, is used to evaluate the long-term effects and tradeoffs of fire and thinning to reduce forest fuels and restore ponderosa pine (*Pinus ponderosa* C. Lawson) forests. A smaller example is the Uneven-aged Silviculture study, established in 1984 with the objective of demonstrating the potential of uneven-aged silvicultural methods to achieve forest health and restoration goals. With regular data collection on a 5-year basis since 1984, and located immediately adjacent to Highway 200, it lends itself well to frequent use.

These and a few other gems notwithstanding, examples also exist at Lubrecht of RDA's that are essentially demonstrating decay. Missing, outdated, and damaged signs minimize the interpretive value of otherwise useful sites. In other areas, normal growth and decay have put RDA's beyond their original shelf lives. Examples include the group selection cutting unit of the Silvicultural Systems area, where growth has made the group selection gaps indistinguishable from the forested matrix; in another RDA designed to illustrate thinning and slash management techniques, piles and twitches have deteriorated to indistinguishable mounds covered in needles. Most recently, major levels of mortality from a landscape scale bark beetle outbreak has substantially altered many RDA's, forcing rapid assessments of their usefulness and creative responses to their treatment.

Evaluation of Existing RDA's

To address these situations, a complete evaluation of Lubrecht's RDA's is underway. The effort began with an inventory and compilation of relevant data, and is now progressing in a continuous screening and sorting of RDA's into condition classes that will help dictate their future management. These categories are:

- Refurbish—useful and relevant, but requiring service; continue the original management strategy with periodic treatment updates consistent with the original intent of the RDA.
- Replace—still conceptually useful, but has outlived its research and/or demonstration shelf-life due to mortality or decay; conduct fresh treatments to revitalize lost elements, or else identify new location where original conditions can be re-established.
- Revise—useful stand structure but treatments reflect outdated strategies or are lacking in contemporary value; enhance the site with new hypotheses and/or treatments that best exploit the treatment histories and longitudinal data.
- Retire—beyond salvage; site is so deteriorated that its usefulness is minimal; return to operational landbase.

The most challenging of these is the "Revise" category. An example is provided by the Levels-of-Growing-Stock study areas. One-time low thinnings were conducted in 1982 (three pure stands, three mixtures) to demonstrate the effect of thinning on stand structure and tree growth. Unmanaged since 1982, they adequately exhibit the effects of a one-time thinning, but they are not representative of a silvicultural strategy for continued density management. For these RDA's, strategies of density management based on stand density index (SDI) are being developed to demonstrate the potential of long-rotation thinning regimes and to evaluate tradeoffs in growth, value, and stand structure among them. Alternatively, where understory development has been substantial, treatment options to address advanced regeneration may also be desirable, for example, overlaying a precommercial thinning of the understory cohort to promote vigor, or else conducting understory burn treatments to retard ladder fuels development.

New Directions

Even as existing studies are being evaluated and modified, new studies are being developed and installed. These are designed to address topical voids in the Lubrecht RDA infrastructure and to meet contemporary and projected needs. They mainly focus on structural complexity and regeneration. Examples include managing for spatially irregular stands using the experimental 'irregular selection' technique that has been applied in Idaho (Graham and Jain 2005, Graham and others 2007), and by an adaptation of the experimental 'Acadian Femelschlag'

method being tested in Maine (Seymour 2005). Others include the establishment of Nelder spacing wheels with pure and mixed species plantings; analysis of regeneration dynamics in salvaged and unsalvaged beetle-kill stands; spatially irregular planting to promote stand differentiation; and the transitional silviculture of aging, even-aged regrowth stands to develop pure and mixed two-age stands. Consistent with this focus on fostering resilience during the regeneration phase, Lubrecht Forest's operational capacity for reforestation will be enhanced with the addition of a micro-nursery over the next several years. This new infrastructure will provide the forest with the ability to better link reforestation to harvesting activities, and will also provide LEF with new opportunities for studying and demonstrating aspects of small-scale seedling production.

Conclusions

Understanding and forecasting Lubrecht Forest's current and potential future users is helping to guide this planning effort. In the past, Lubrecht's silviculture RDA's were used by many agencies, organizations, and landowner groups, as well as legions of University of Montana students. In the future, public participation in natural resource decision-making is expected to grow as it has during past decades. By providing effective, accessible research and demonstration areas with relevance to contemporary and anticipated forest management issues, by creating an array of stand structures and compositions for yet unknowable future uses, and by developing new and innovative opportunities for their utilization in public outreach and learning, a refreshed and revitalized Lubrecht Forest will be positioned for continued relevance and leadership in the study and promotion of sustainable silvicultural management.

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Part 4: Decision Support Tools

Addressing Climate Change in the Forest Vegetation Simulator to Assess Impacts on Landscape Forest Dynamics

Nicholas L. Crookston[1], Gerald E. Rehfeldt, (Retired)[2], Gary E. Dixon, (Retired)[3], and Aaron R. Weiskittel[4]

Abstract—To simulate stand-level impacts of climate change, predictors in the widely used Forest Vegetation Simulator (FVS) were adjusted to account for expected climate effects. This was accomplished by: (1) adding functions that link mortality and regeneration of species to climate variables expressing climatic suitability, (2) constructing a function linking site index to climate and using it to modify growth rates, and (3) adding functions accounting for changing growth rates due to climate-induced genetic responses. For three climatically diverse landscapes, simulations were used to explore the change in species composition and tree growth that should accompany climate change during the 21st century. The simulations illustrated the changes in forest composition that could accompany climate change. Projections were the most sensitive to mortality, as the loss of trees of a dominant species heavily influenced stand dynamics. While additional work is needed on fundamental plant–climate relationships, this work incorporates climatic effects into FVS to produce a new model called Climate–FVS. This model provides for managers a tool that allows climate change impacts to be incorporated in forest plans.

See this web site for access to the model and supporting literature: http://www.fs.fed.us/fmsc/fvs/description/climate-fvs.shtml.

Keywords: species climate relationships, stand dynamics, species composition, genetic adaptation, general circulation model, climate change, carbon loads, site index, growth and yield

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Calibration of State and Transition Models With FVS

Melinda Moeur[1] and Don Vandendriesche[2]

Abstract—The Interagency Mapping and Assessment Project (IMAP), a partnership between federal and state agencies, is developing mid-scale vegetation data and state and transition models (STM) for comparing the likely outcomes of alternative management policies on forested landscapes across the Pacific Northwest Region. In an STM, acres within a forested ecosystem transition between state classes defined by cover type and seral stage. Transitions resulting from natural and human-induced change can be modeled over long time horizons (centuries). Natural growth and mortality, silvicultural treatments, and stochastic disturbances such as wildfire and insect outbreaks can all be modeled in an STM. In Washington and Oregon, STM models are being used to explore proposed actions that affect wildfire risk, forest health and restoration, habitats for species whose populations are declining, and long-term timber supply. We describe how the Forest Vegetation Simulator (FVS) is being used with regional inventory data to empirically derive STM parameters (residence times in states, pathways between states, and transition probabilities between states), and to link outputs to vegetation states. For example, active fuel treatments in overstocked stands can be simulated to track how management might influence long term probabilities of wildfire risk. A range of outputs from FVS—for example, harvest volumes and carbon balances associated with thinning—can be captured and linked to treatments through the modeling framework. The ability to empirically forecast complex growth and disturbance scenarios using real data makes STM tools more applicable to real-world management questions.

Introduction

The diverse landscapes of the Pacific Northwest are managed by different owners for different objectives. Key considerations facing all forestland planners include managing fuel conditions and wildfire risk, old-growth, wildlife habitat, supply and demand of forest products, biomass supplies and carbon budgets, among others. Ideally, planning efforts on lands with a variety of management objectives would all share a common set of landscape data and modeling methods to help insure consistent answers to common questions. The Interagency Mapping and Assessment Project (IMAP) is responding to this need to foster cooperation on important landscape issues affecting different management entities at different scales. IMAP supports federal and state partners in Washington and Oregon by building integrated datasets, models, and tools for conducting mid-scale assessments (Hemstrom and others 2008).

IMAP uses a state and transition model (STM) approach to project the effects of disturbance and management on the forested landscape. We use the Vegetation Dynamics Development Tool (VDDT) developed by ESSA Technologies Ltd. (2005) as the IMAP STM. An STM treats vegetation as combinations of cover type and structural classes linked by pathways resulting from natural succession, management actions, and disturbances (example, fig. 1). The cover type/structural states (boxes) represent the most important developmental stages. Pathways

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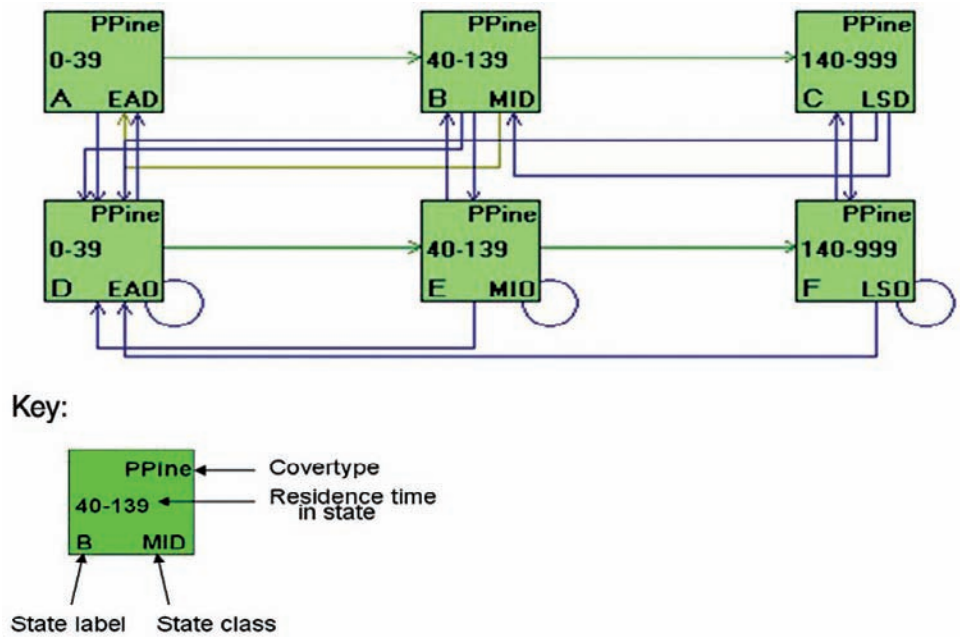


Figure 1—Successional pathways diagram for the Warm Dry Ponderosa Pine VDDT STM, Blue Mountains modeling region. Boxes represent cover type/structural state classes, and arrows represent transitions. Green arrows depict deterministic pathways resulting from natural growth dynamics. Brown or blue arrows depict stochastic pathways following disturbance events such as stand replacing wildfire, non-lethal underburning, or commercial thinning. A key to state box codes follows:

Cover type definitions:
PPine Ponderosa pine cover type

State class definitions:
EAD Early seral, Dense (0-10 inches qmd, > 40% canopy cover)
MID Mid seral, Dense (10-20 inches qmd, > 40% canopy cover)
LSD Late seral, Dense (> 20 inches qmd, > 40% canopy cover)
EAO Early seral, Open (0-10 inches qmd, 10-40% canopy cover)
MIO Mid seral, Open (10-20 inches qmd, 10-40% canopy cover)
LSO Late seral, Open (> 20 inches qmd, 10-40% canopy cover)

(arrows) are the linkages between the states. Residence time is the average length of time that vegetation typically remains in the same class before transitioning to another state through successional dynamics. For disturbance pathways, transition probabilities control the frequency with which movements between states occur.

An STM modeler specifies the model parameters that control states, pathways, and transition probabilities (see sidebar). Of course, model projections will be most realistic when the model parameters are informed by best science and relevant experience. Typically, model users obtain as much information as possible on vegetation dynamics and disturbance ecology from the literature; however, expert opinion is often heavily relied upon to specify model parameters.

Stand projections of inventory data with the Forest Vegetation Simulator (FVS) (Dixon 2003) provide another important source of information for helping make state and transition models behave in a more realistic way. This approach was first

SIDEBAR

Virtually any successional pathway, natural disturbance, or management prescription can be constructed and modeled within VDDT. The examples below are verbal descriptions of important vegetation dynamics within the cool moist mixed conifer model type.

Succession and ingrowth, with and without management

- Ingrowth in medium density stands grow to a closed stand after 20 years of no disturbance.
- Grass/forbs stage can last for as long as 100 years under the harshest conditions. However, under many conditions saplings will establish. These circumstances are represented by an alternate succession pathway with a 0.01 annual probability.
- Common harvesting activities include thinning in mature stands, shelterwood cutting, first entry, second entry, and clearcut with reserves. All have been entered in with a 0.01 annual probability.
- A sapling stand will naturally go to a closed canopy condition unless pre-commercial thinning keeps it in a medium density state.
- Ingrowth within the post-disturbance classes recognizes that snags are falling down and stands are recovering from stand replacing events. A time since disturbance value of 30 years is required before movement out of a post disturbance class.

Fires

- Stand replacing fires (mean fire return interval 300 yrs) transition the area to a grass/forbs post-disturbance state if the stand was in a Medium, Large, Very Large, or Giant size class.
- Mixed severity fires (mean fire return interval 500 yrs) convert 50 percent of the area to grass/forbs post disturbance and the other 50 percent remains in the same class.

Insect & Disease

- Beetle mortality and wind damage is lumped into an insect and disease disturbance with a low probability (0.0001). Only in open stands do these events not decimate the entire stand.

proposed by Stage (1997), as a way of providing "an empirical link between the Columbia River Basin Successional Model (CRBSUM) and the real world." The idea was that current vegetation inventories representing different stages of stand development could be integrated with insect, disease, management activities, and fire effects available in the FVS system to empirically inform parameters and outputs in STMs. In concept, pursuit of such an analysis in Region 6 is particularly appealing because large amounts of plot sample data from regional strategic inventories are available for projection. Thus, it is likely that representative samples exist for most or all modeled states in the STM.

Case Study

In this paper, we describe the process for using FVS in parallel with VDDT to help inform IMAP state and transition models. One objective of this dual modeling system is to test the assumptions made by the STM modeler—in some cases, this process may lead to modification of some STM model parameters. Another objective is to use FVS/empirical data in combination to more fully understand important vegetation pathways that may not have been adequately represented in the original STM—and perhaps, to expand the STM model. Conversely, a development pathway conceived to be important in the original STM may be shown through the FVS process to be not as prevalent as originally thought—perhaps leading to eliminating a particular pathway in a revised STM model. Finally, we know of no better way than an FVS analysis to estimate outputs for the many complex transitions that are likely to be modeled in an STM—FVS, especially when used with the Event Monitor, can be used to develop outputs such as standing and harvest volumes, fuel conditions, and stand structural attributes that can be linked to states and transitions in a VDDT model.

Warm Dry Ponderosa Pine in the Blue Mountains Modeling Region

Within IMAP, the STM work is organized hierarchically around geographic and ecological study regions. In Region 6, the entire project area has been divided into modeling regions scheduled around the Forest Plan Revision cycle (fig. 2). In subsequent examples, we'll refer to the 13.1-million acre Blue Mountains modeling region where the Wallowa Whitman, Umatilla, and Malheur National Forests are revising their forest plans. Within a modeling region, the landscape is divided into ecological strata called potential vegetation types (PVTs). A PVT represents a particular combination of environment, disturbance regimes, and vegetation growth potential. Unique VDDT state and transition models are designed for each PVT (Hemstrom and others 2006). For example, the Blue Mountains project area is stratified into eight PVTs that are depicted by separate models (table 1). Within each model, combinations of cover type and structure—standard classes of tree size, canopy density, and canopy layering (table 2)—define the state boxes.

Each acre in the landscape is initialized to its PVT, cover type, and structural state based on a map of current vegetation (fig. 3). These maps have been created using the gradient nearest neighbor imputation approach (GNN). GNN uses tree-level attributes collected on Current Vegetation Survey (CVS) and Forest Inventory and Analysis (FIA) inventory plots (Max and others 1996; FIA 2006), coupled with satellite imagery and other spatial data such as topography and climatic gradients, to populate 30-meter pixel maps (Ohmann and Gregory 2002). The resulting vegetation maps are used to define the initial conditions that assign acres to state classes according to the definitions in table 2. Other spatial layers are used to stratify the study region by ownership and land allocation.

In IMAP, the primary spatial unit for modeling is the watershed (5th *–order* hydrologic code, units averaging about 100,000 ac), and within the watershed, additional sub-strata are created for owner/land allocation groups defined primarily by distinct management goals. Examples are timber suitable public lands, wilderness areas, species habitat reserves, and private industrial lands managed for intensive wood production. Summarizing the acres and outputs for these sub-strata generates useful information about the spatial distribution of landscape characteristics without implying pixel or stand-level accuracy. Results at the watershed/owner/allocation scale are fine enough for most mid-scale assessments.

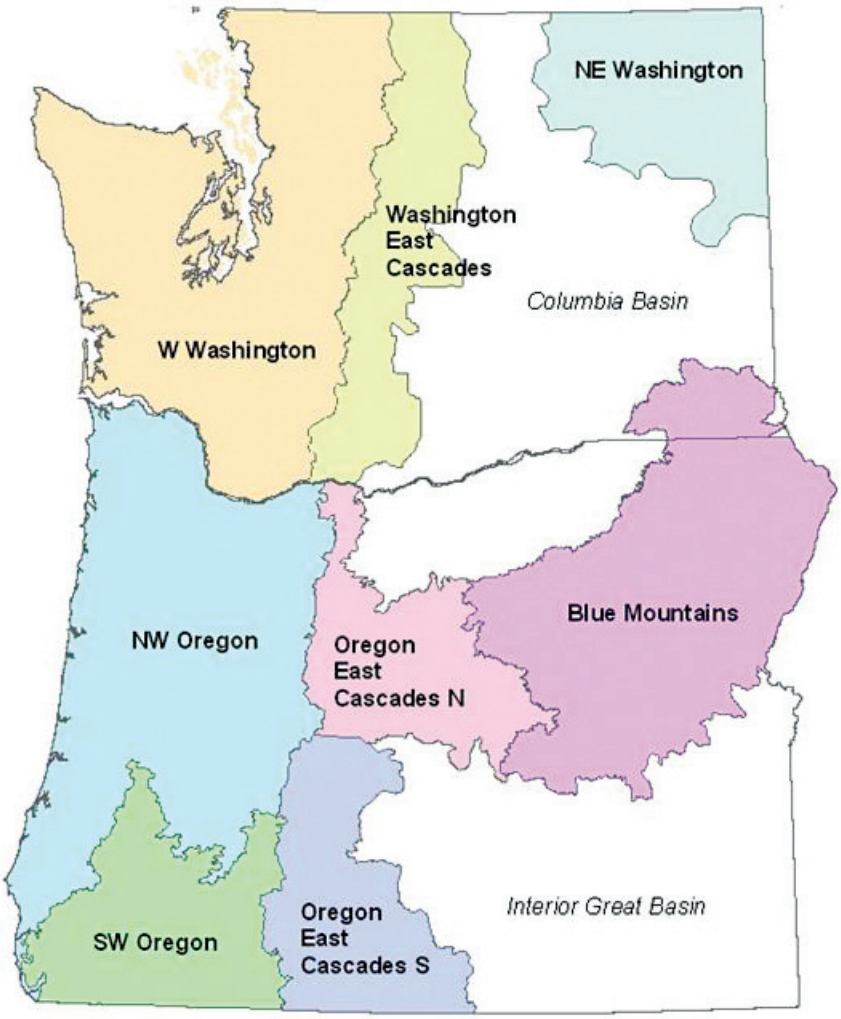


Figure 2—IMAP modeling regions in Washington and Oregon. Modeling regions are scheduled to coincide with National Forest Plan revisions. Because the Columbia Basin and Interior Great Basin have little forested land, they are not currently being modeled.

Table 1—Eight VDDT models are defined by forested potential vegetation type in the Blue Mountains modeling region. Shown are the numbers of inventory plots on National Forests by potential vegetation type. The Blue Mountains variant of FVS was used for modeling all plots.

VDDT potential vegetation type	VDDT code	Cover types[b]	Number of CVS plots by National Forest[a]			
			MAL	UMA	WAW	Total
Subalpine whitebark pine	sw	WB	16	8	46	70
Cold dry mixed conifer	cd	GF; ES/AF; WL/LP	22	126	282	430
Cool moist mixed conifer	cm	DF/GF/ES; WL/LP	206	432	426	1064
Warm dry grand fir	dg	PP; DF/GF	420	122	242	784
Warm dry Douglas-fir	dd	PP; DF	280	302	440	1022
Warm dry ponderosa pine[c]	**dp**	**PP**	**312**	**66**	**86**	**464**
Hot dry ponderosa pine	xp	PP	288	60	100	448
Woodland western juniper	jw	WJ	94	26	8	128
Non-forest	nf	none	120	128	444	692
Total:			1758	1270	2074	5102

[a] National Forest codes: MAL = Malheur; UMA = Umatilla; WAW = Wallowa-Whitman.

[b] Species codes: AF = subalpine fir; DF = Douglas-fir; ES = Engelmann spruce; GF = grand fir; LP = lodgepole pine; PP = ponderosa pine; WB = whitebark pine; WJ = western juniper.

[c] Used as the modeling example in the text.

Table 2—Standard structure classes that along with cover type (see table 1) and disturbance history, define states in the IMAP VDDT models.

Size class (DBH—inches) [a]	Class description
0.0 to 4.99	Seedling-Sapling
5.0 to 9.99	Small tree
10.0 to 14.99	Medium tree
15.0 to 19.99	Large tree
20.0 to 24.99	Very large tree
25.0 and larger	Giant tree
Canopy cover (percent)	**Class description**
0 to 9.9	Non-tree
10 to 39.9	Open
40 to 69.9	Medium
70 to 100	Closed
Canopy layers	**Class description**
1	Single
2 or more	Multiple

[a] Quadratic mean diameter of dominants and codominants.

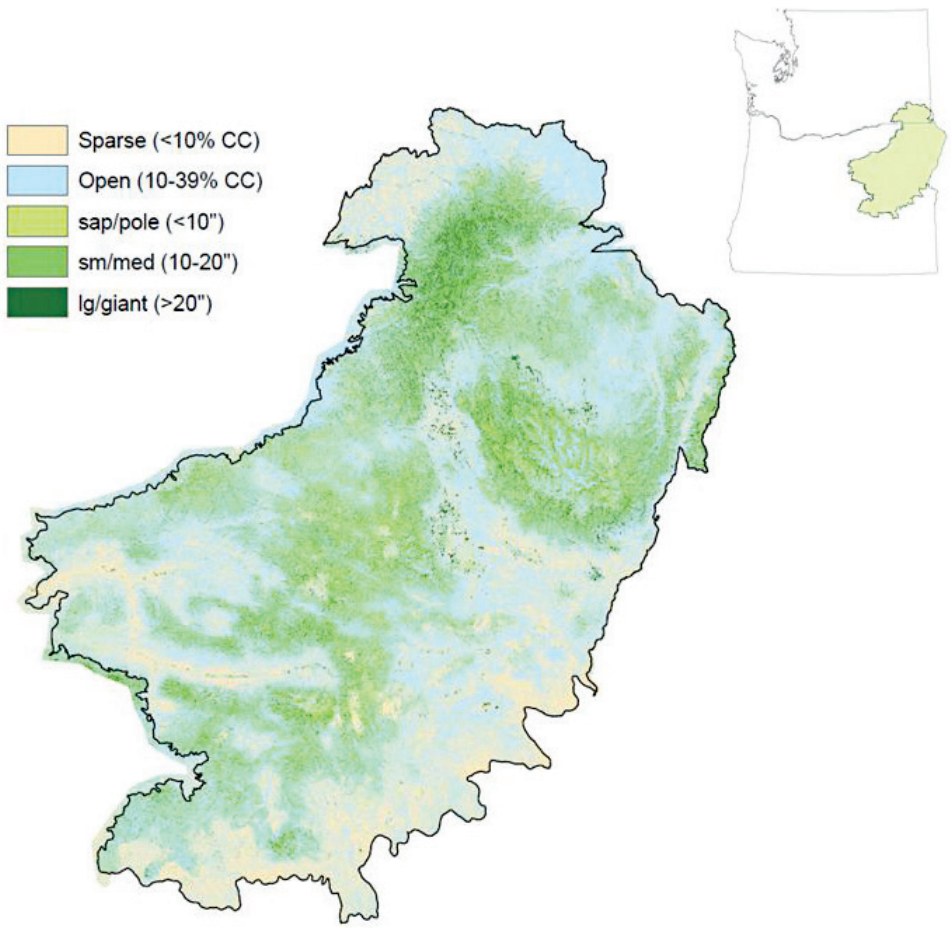


Figure 3—Existing vegetation for the Blue Mountains modeling region from Gradient Nearest Neighbor modeling (Ohmann and Gregory 2002).

FVS Analysis

In order to accomplish the integration of FVS within the STM approach, a computer program was developed to classify inventory data into vegetation classes (i.e. cover type, size class, canopy cover, canopy layers) for initial conditions and for subsequent projection cycles. The Preside program (Vandendriesche 2009a) summarizes various vegetation classes into states and provides average time in a particular state and the probability of movement to associated states. Armed with this tool, the general sequence of steps in an analysis process that integrates FVS projections with state and transition models is this:

1. Prepare the inventory data for projection by FVS.
2. Adjust FVS parameters to the current inventory.
3. Develop natural growth projections to estimate parameters for successional pathways (without disturbance).
4. Process the tree list output through the Preside program and accumulate the results into a matrix from which mean residence times within states and transition probabilities between states can be summarized.
5. Aggregate stand attributes by vegetation states.
6. Repeat steps for projections with management, insect and disease, and fire to estimate parameters for disturbance pathways.
7. Review state and transition empirical parameters in relation to conceptual parameters and adjust the STM where necessary.

Inventory data—Data for an FVS-VDDT analysis can be obtained from regional strategic inventories such as CVS and FIA. On U.S. Forest Service non-wilderness lands (as well as on BLM lands in western Oregon), one CVS plot is installed every 1.7 miles (1 plot per about 7,200 ac). On U.S. Forest Service wilderness lands, plots are installed at one-fourth the intensity of non-wilderness (one plot every 3.4 miles). On non-U.S. Forest Service or non-BLM lands, we are using FIA data from the periodic inventories collected between 1988 and 1999 in Oregon and Washington, and also from FIA's National Annual Inventory beginning in 2000 in Oregon and 2002 in Washington. FIA plots are installed at a sampling intensity of one plot every 6,000 acres. For the example analysis, we used data only from the Region 6 CVS inventory on the Wallowa Whitman, Umatilla, and Malheur National Forests. There were 5,102 samples (2,551 plots, each measured at two sampling occasions) available for the Blue Mountains modeling region, of which 464 samples fall in the warm dry ponderosa pine (*Pinus ponderosa* C. Lawson) PVT (table 1).

CVS data are prepared for projection using a translator program developed at Region 6 (Gregg and other 2004). Before further processing, each plot must be assigned to the appropriate VDDT model. The ecoclass code (Hall 1998) recorded for the plot is used to assign the potential vegetation type. A GIS intersection of the plot location with ownership and land allocation coverages are used to assign the administrative classification.

FVS calibration—Before projecting the large set of inventory plots for a project area, it is important to adjust FVS default parameters for growth, mortality, and regeneration for each combination of PVT model type and owner/allocation stratum. Table 1 shows the number of inventory samples used for calibration and modeling for each PVT in the Blue Mountains study area. The purpose of performing these calibration steps is so that the projections more closely mimic the empirical conditions determined from the actual field measurements. One example of a situation where calibration is essential is for projecting old-growth stands. The sample base upon which the empirical growth and mortality equations

in FVS are built is not well suited to modeling old-growth forests over long time horizons, and yet typically VDDT simulations are performed for 200- to 300-year intervals. Thus, thoughtful calibration can greatly improve the realism of simulations when projecting stands over long time periods by attenuating height and diameter growth and mortality during stand senescence.

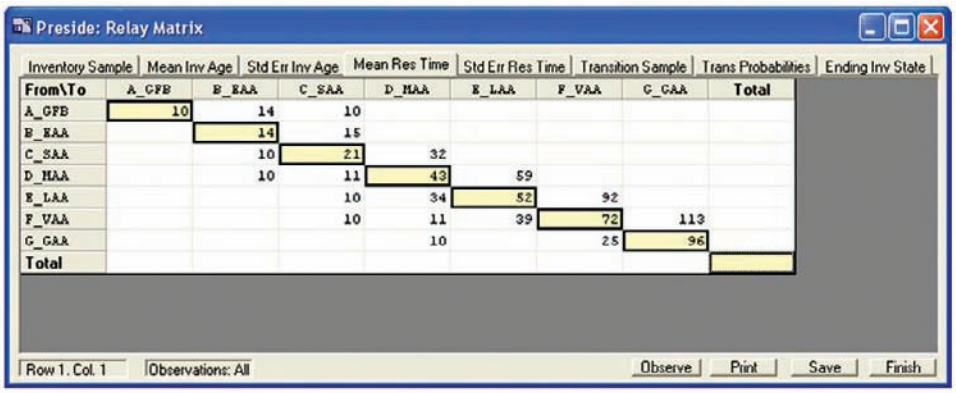
Calibration procedures include using the FVS self-calibrating feature (for example, altering the baseline estimate of the large-tree diameter growth models), estimating and inputting natural regeneration response (querying existing stands to tabulate their seedling component), accounting for tree defect for volume estimates (adjusting net merchantable volume from gross tree dimensions), and determining tree species size attainment and limiting stand maximum density. Another paper in this volume (Vandendriesche 2009b) deals with this topic in more detail, and so we will not elaborate further in this paper.

Natural growth projections—In VDDT, the successional classes, pathways, and transition probabilities are defined for each potential vegetation type. A single PVT may have more than one set of probabilities defined to represent different management regimes or ecological conditions. In general, two types of transitions can occur (fig. 1). One type is movement between states due to natural succession. This process integrates background disturbances that affect regeneration, growth, and self-thinning, but not extrinsic disturbances such as insect or disease outbreaks, wildfire, or silvicultural treatment. Transitions representing natural successional dynamics (or 'natural growth') are modeled deterministically in VDDT. What this means is that transitions from one class to the next class occur when the residence time (a surrogate for successional 'age') has exceeded the value set for the state. For transitions in VDDT related to disturbances, movement between states is determined stochastically according to probabilities set by the user.

Once the FVS calibration procedure has been completed, we use FVS commands (keywords) to adjust growth, mortality, and regeneration responses as outlined in the above section. To model natural succession in FVS, we track residence time in a state—the average length of time that vegetation typically remains in that state before transitioning to the next state along the successional pathway. We do this by projecting all the plots in the study region without invoking any extrinsic disturbance or pest effects in FVS. Then 300-yr projections are performed for every plot, outputting tree lists and stand summaries each cycle for completing the next two steps.

Classify the tree lists, calculate residence times—Preside classifies the current tree list for each plot at each cycle boundary into the cover type, size class, canopy closure, and canopy layer that define the possible states (tables 1 and 2). Following Stage's approach (1997), estimates of the residence times and pathways are summarized by use of an array of all possible transitions from one state to another, and indexed by PVT, owner, and allocation groups to which a plot belongs. For each plot at each cycle, its source (that is what state it began the cycle in) and destination (that is what state it ended the cycle in) are recorded. The length of time each plot remains within a state class between cycles is accumulated and the mean and variance of residence times is summarized over all the cycles and transitions in the projection (fig. 4). The pathways (direction of movement between source and destination) between states are also summarized using the array.

Accumulate and summarize outputs—At the end of an FVS projection, a set of FVS post-processing steps have been bundled together that produce aggregate summaries for each of the state classes, using the sample of plots populating each



From\To	A_GFB	B_EAA	C_SAA	D_MAA	E_LAA	F_VAA	G_GAA	Total
A_GFB	10	14	10					
B_EAA		14	15					
C_SAA		10	21	32				
D_MAA		10	11	43	59			
E_LAA			10	34	52	92		
F_VAA			10	11	39	72	113	
G_GAA				10		25	96	
Total								

Figure 4—Preside program relay matrix. Preside calculates mean residence times and transition probabilities for the sample of plots in a PVT projection.

State class definitions:

A_GFB VDDT-State A: Grass/Forbs/Brush
B_EAA VDDT-State B: Seed/Sap size class (0-5 inches qmd)
C_SAA VDDT-State C: Small Tree size class (5-10 inches qmd)
D_MAA VDDT-State D: Medium Tree size class (10-15 inches qmd)
E_LAA VDDT-State E: Large Tree size class (15-20 inches qmd)
F_VAA VDDT-State F: Very Large Tree size class (20-25 inches qmd)
G_GAA VDDT-State G: Giant Tree size class (25+ inches qmd)

state class during the projection. It is then easy to display graphics for communicating the STM results. For example, images from the Stand Visualization System (SVS) can be displayed for each vegetation state that is an aggregate of the plots in that state (fig. 5). The post-processing programs also index the aggregate state classes to summary values derived from the tree lists, attributes from standard FVS output reports, and variables computed from the Event Monitor. This feature is useful for tracking important values such as stand volume and biomass across states (example, fig. 6). Other attributes tracked by this process are shown for the warm dry ponderosa pine (example, table 3).

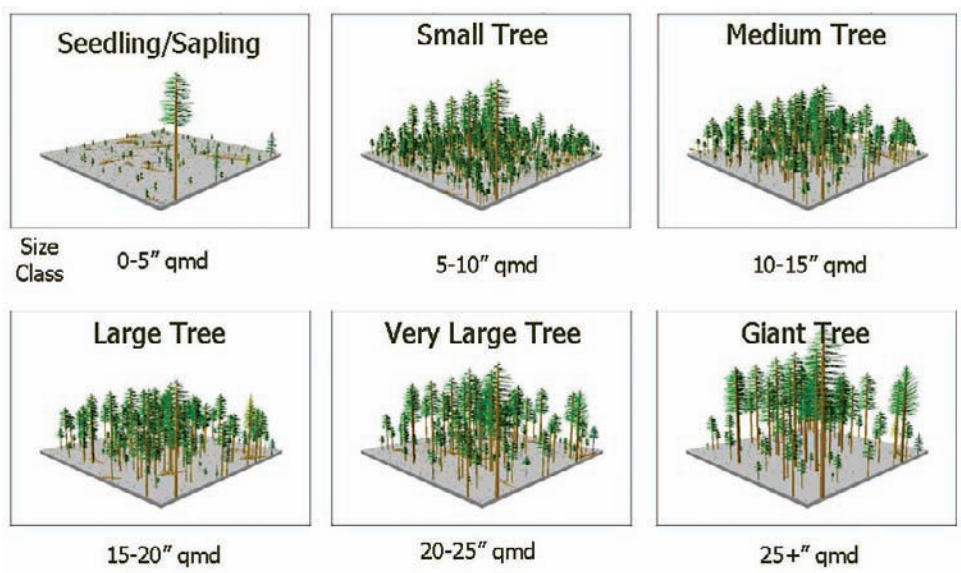


Figure 5—Stand Visualization System images of the dry ponderosa pine model arrayed by size class.

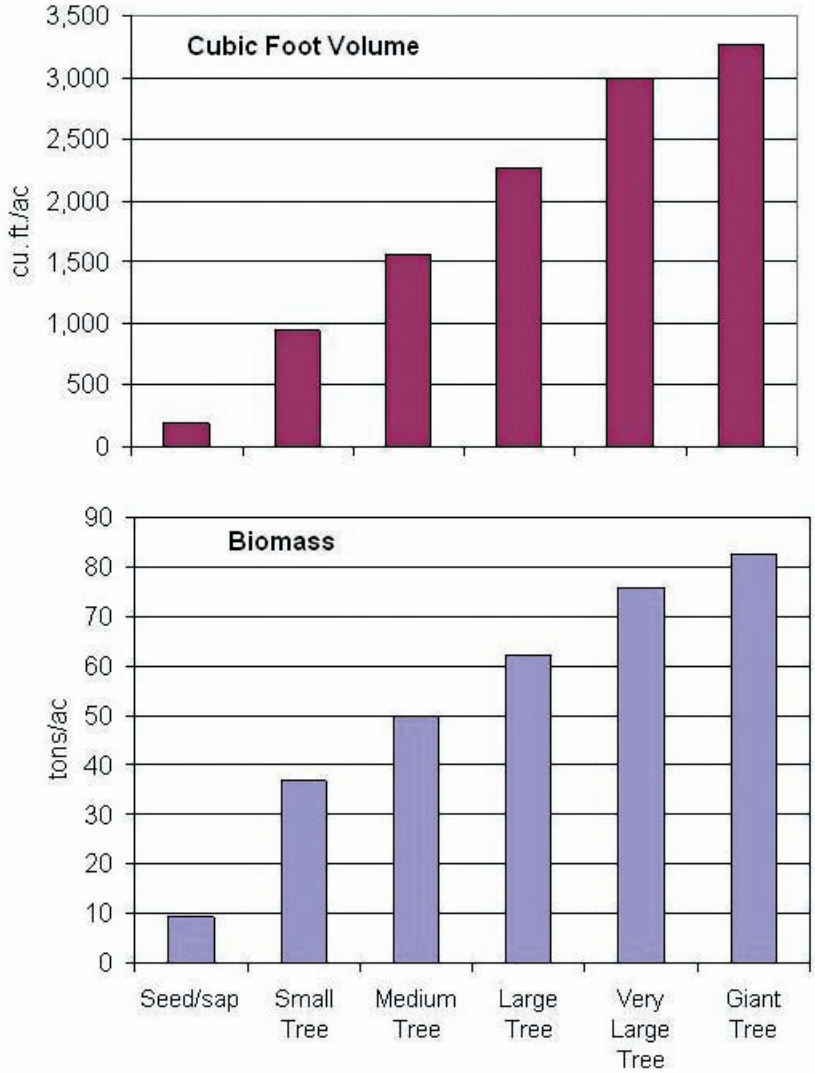


Figure 6—Summary attributes of aggregate state classes for the warm dry ponderosa pine model tracked by the FVS-VDDT post-processing programs.

Feedback loop between FVS and VDDT model—The mean residence times and transition probabilities that are computed by the Preside program can be used to substantiate assumptions built into the VDDT model by the user. In the warm dry ponderosa pine example, we compared the residence times computed from the 464 sample plots projected through the FVS system with VDDT parameter values obtained from the literature and from expert opinion (table 4). Recognizing five size classes spanning a 200-year time horizon for dry ponderosa pine sites in the Interior West, note the disparity of residence times by source method within the developmental stages. The most simplistic view renders an even splicing of the time distribution as cited by the literature review. A group of experts recognized an emphasis on enhanced early growth with a slow down occurring in later size classes. The FVS modeled runs conform to empirical knowledge gained from inventory data and indicate that the STM residence times should be further adjusted, especially shortening the time-in-state for the smallest size classes and lengthening it for the largest state classes.

Table 3—Aggregate values by VDDT vegetation state derived from FVS computed variables for the warm dry ponderosa pine model, by size class.

	Seed/sap (0 - 5")	Small tree (5-10")	Medium tree (10-15")	Large tree (15-20")	Very large tree (20-25")	Giant tree (25"+)
Vegetation Structure Variables:						
Dominance Type	PIPO	PIPO	PIPO	PIPO	PIPO	PIPO
Size Class	1	2	3	4	5	6
Canopy Class	1	2	2	2	2	2
Canopy Layers	1	1	1	2	2	2
Stand Age – Dominant Story	15	46	93	140	199	241
Total Plot Sample Count	23	558	3,406	4,242	2,561	394
Stand-Stock Variables:						
Basal Area/Acre, trees >= 1.0" diameter	19	91	112	122	131	130
QMD – Top 20 % trees, diameter	4.0	8.6	13.1	17.5	22.3	25.7
Percent Canopy Cover	24	53	51	46	43	40
Live – Cubic Feet/Acre	198	943	1563	2272	2997	3268
Live – Board Feet/Acre	1,139	5,043	7,868	11,661	16,136	18,534
Wildlife Habitat Variables:						
Standing Snags/Acre						
Small = 5-10" diameter	4.8	28.9	35.1	18.3	6.6	1.9
Large = 10"+ diameter	2.6	6.8	17.9	21.2	18.4	14.0
Extra-large = 18"+ diameter	0.6	2.0	3.2	5.0	8.0	9.7
Snag Recruitment/Acre						
Small = 5-10" diameter	2.2	43.1	30.5	12.4	3.8	1.2
Large = 10"+ diameter	1.0	4.9	11.3	10.8	7.4	4.3
Extra-large = 18"+ diameter	0.2	0.9	1.2	2.1	2.9	2.7
Wildfire Risk Variables:						
Crown Bulk Density	0.05	0.10	0.07	0.06	0.05	0.05
Fire Hazard Rating	3	4	4	3	3	2
Biomass-Carbon Variables:						
Tree Biomass – Dry weight live & dead/boles & crown	9	37	50	62	76	83
Stand Carbon – Total carbon above & below ground	10	31	49	61	73	80

Table 4—Mean residence time by size class for Dry Ponderosa Pine type.

	Seed/sap 0-5"	Small 5-10"	Medium 10-15"	Large 15-20"	Very large 20"+
Literature Review [a]	40	40	40	40	40
Expert Opinion [b]	35	35	45	45	45
Empirical Basis (FVS) [c]	14	21	43	52	72

[a] The Nature Conservancy, under contract with the U.S. Forest Service Southwestern Regional Office, for the Southwest Forest Assessment Project, values for Ponderosa Pine/Bunchgrass PVT.

[b] U.S. Forest Service Pacific Northwest Regional Office and Fremont-Winema National Forests, Southwest Oregon, collaborative forest plan working group, values for Dry Ponderosa Pine PVT.

[c] Interagency Mapping and Assessment Project, U.S. Forest Service Pacific Northwest Regional Office, Blue Mountain Modeling Region, from the Forest Vegetation Simulator, values for Warm Dry Ponderosa Pine PVT.

For testing pathways, a simple test is to verify the correspondence between the FVS projection and the STM projection at 'equilibrium' over a long projection period for the natural growth scenario. To record the successional growth pattern, the inventory plots are forecasted for 200-300 years through FVS under endemic stand development (without disturbance). Ending vegetation states for the inventory plots as reported by Preside are compared with the equilibrium conditions projected by the STM. If the VDDT model is configured appropriately, it should predict reasonably well the ending inventory conditions projected by FVS. Appropriate adjustments can then be made.

Discussion

Note that both FVS and VDDT models predict state transitions (a change from one vegetation state to another). Transitions in VDDT can only occur along natural growth and disturbance pathways specified by the modeler. But transitions in FVS result from the detailed regeneration, growth, and mortality of individual trees, condensed to VDDT vegetation classes. As a result, FVS typically models transitional pathways not explicitly represented in the corresponding VDDT model. For example, if size class (calculated as the qmd of the dominant overstory) decreases between cycles in FVS as a result of the mortality of just a few large trees in the tree list, the resulting transition may indicate "retrogression" from a larger state class to a smaller state class. Unlike FVS, VDDT won't model retrogressions unless such pathways are explicitly specified by the user. For example, ESSA Technologies Ltd (Beukema & Robinson 2009) investigated a dry mixed conifer PVT from central Washington with a VDDT model having 32 transitions in total covering about 40 classes. For the same PVT, the FVS analysis using about 500 plot samples resulted in 174 transitions only of which 15 pathways were explicitly specified in the associated VDDT model. The remaining transitions were retrogressions, or 'jumps' between non-adjacent state classes.

FVS and STM can be viewed as models with similar objectives, but at different ends of the spectrum in terms of resolution. Both are designed to project vegetation changes due to successional dynamics, and changes driven by disturbances. An STM like VDDT represents vegetation change as movement between classes along pre-determined pathways. An individual tree model like FVS represents vegetation change along a continuum as the sum of regeneration, accretion, and mortality of individual trees. The objective of our paper has been to demonstrate how these models can be used in conjunction to provide an informed decision process. Building linkages between a strong empirical base represented by inventory data and FVS, and an STM calibrated largely by expert opinion, will lead to an improved modeling system.

FVS's expanded resolution accounts for complex changes in the stand development due to regeneration and mortality and the changing impacts of having multi-species stands. However, the challenge of building a comprehensive VDDT model is to represent the important successional, disturbance, and management dynamics in sufficient detail, but not too much detail. If the ultimate goal of landscape planning is to understand the underlying processes, coupling FVS with VDDT will aid the endeavor.

In building STMs, we recognize the importance of striking a balance between FVS projection results, expert judgment (i.e., the collective experience of silviculturists, ecologists, and others about how forest vegetation is likely to develop and respond to management and disturbance), and additional scientific knowledge (i.e., vegetation responses drawn from the literature). The VDDT modeler should review all three types of information and add best professional judgment in selecting the final transition times.

More information about the data, models, and tools mentioned in this paper can be obtained on-line. FVS can be obtained from the FVS website (http://www.fs.fed.us/fmsc/fvs/index.shtml) and Preside, and associated post-processors can be obtained by contacting the authors. A comprehensive user guide that documents the primary computer processing steps is also available (Moeur & Vandendriesche 2009). The Vegetation Dynamics Development Tool can be acquired from ESSA Technologies Ltd. (http://www.essa.com/tools/VDDT/index.html). IMAP data and more information about the IMAP program can be found at http://www.reo.gov/ecoshare/mapping/index-issues.asp.

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The content of this paper reflects the views of the authors, who are responsible for the facts and accuracy of the information presented herein.

FVS Out of the Box—Assembly Required

Don Vandendriesche[1]

Abstract—The Forest Vegetation Simulator (FVS) is a prominent growth and yield model used for forecasting stand dynamics. However, users need to be aware of model behavior regarding stocking density, tree senescence, and understory recruitment; otherwise over long projections, FVS tends to concentrate substantial growth on few survivor trees. If the intent is to forecast endemic conditions, model performance can be modified by establishing baseline trends from measured inventory data and configuring FVS to stay within the observed ranges. This paper will present the steps needed to carefully craft an endemic FVS run. Techniques include using the FVS self-calibrating feature; accounting for tree defect; limiting stand maximum density; determining tree species size attainment; and, accounting for regeneration response. A case example from the Blue Mountains project area in northeast Oregon will be presented.

Introduction

Have you ever purchased an item from a store that comes in a box only to realize that once opened, all the components are there but you must carefully assemble the parts to properly construct the object? You may have a manual or diagram to consult, but getting from the raw materials to the final product can be challenging. This analogy can be applied to using the Forest Vegetation Simulator (Dixon 2002). Users may naively assume that after initially installing the software, they are fully ready to go. This can be problematic in matching measured trends to modeled projections.

Three projection scenarios are recognized in this paper: (1) full site occupancy, (2) endemic stand development, and (3) epidemic area disturbance[2]. By default, FVS will forecast stand development to full stocking. In contrast, compilation of large field inventory data sets void of significant impacts will typically render endemic conditions. Whereas, bark beetle outbreaks, dwarf mistletoe epicenters, and active crown fires render epidemic, stand replacing events. Users of FVS need to be cognizant of the modeling context and take steps necessary to ensure simulation results portray logical outcomes. The purpose of this paper will be to present the fundamental steps needed to project stand development under endemic conditions with FVS. The methodology prescribed will be based on the procedures used to construct vegetation pathways for regional assessments and forest planning.

[2]Endemic implies being constantly present in a particular region and generally considered under control. In contrast, epidemic refers to out of control situations where the vector is spreading rapidly among many individuals (Ref: Webster's New World Dictionary, Third College Edition).

In: Jain, Theresa B.; Graham, Russell T.; and Sandquist, Jonathan, tech. eds. 2010. Integrated management of carbon sequestration and biomass utilization opportunities in a changing climate: Proceedings of the 2009 National Silviculture Workshop; 2009 June 15-18; Boise, ID. Proceedings RMRS-P-61. Fort Collins, CO: U.S. Department of Agriculture, Forest Service, Rocky Mountain Research Station. 351 p.

[1] *Forester, U.S. Department of Agriculture, Forest Service, Forest Management Service Center, Vegetation Simulation Group, Natural Resources Research Center, 2150A Centre Avenue, Fort Collins, CO.*

Whether planning at the national, broad, mid, or base level, the analysis process for projecting vegetation should contain the following elementary steps:

P erform vegetation stratification
I dentify data sources
C alibrate FVS model
T ailor natural growth runs
U tilize treatment prescriptions
R eport vegetation pathways
E valuate output values

Having a good 'PICTURE' in mind before beginning a project will aid the endeavor. Each of these analysis topics will be elaborated upon relative to development of vegetation pathways for a landscape assessment of the Blue Mountains project area.

Vegetation Stratification

Forest planning efforts often require developing estimates of conditions and outcomes by stand type. A stand type is a combination of the physical, vegetative, and developmental characteristics used to identify homogeneous forest strata (Davis and Johnson 1987). Physical attributes describe the site aspects of the forest, such as topography, soils, and habitat type. Vegetative attributes characterize the flora aspects of the forest, such as overstory dominant species, its relative size and density. Developmental attributes portray the human aspects of the forest, such as roads, buildings, and administrative boundaries. In this context, stand types are non-spatial but are comprised of many geographically identifiable forest stands.

Biophysical settings of plant associations (habitat types) arrayed by prevailing tree species comprised the stand types used to describe the vegetation strata for the Blue Mountains project area. Figure 1 provides a relational schematic of ascending elevation adjoining to the potential vegetation types that represent the various temperature/hydrologic regimes.

The Warm Dry Ponderosa Pine (*Pinus ponderosa* C. Lawson) stand type will be used to demonstrate FVS assembly techniques employed in the construction of an endemic model run. Figure 2 displays a Stand Visualization System (SVS) (McGaughey 2004) rendering of this biome.

Figure 1—Biophysical settings of potential vegetation types within the Blue Mountains project area.

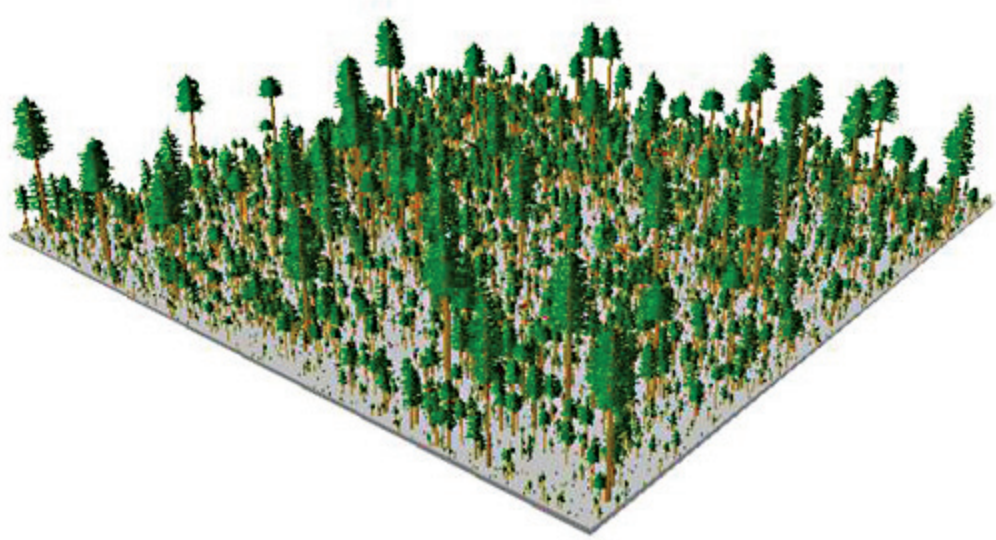

Figure 2—SVS ten-acre depiction of the Warm Dry Ponderosa Pine stand type.

Data Sources

Two types of data are generally used for planning projects: spatial and temporal. Spatial data is usually compiled from remote sensing imagery, and acreage compilation is accomplished by summing the various stand types residing within mapped polygons. Temporal data is collected during a field inventory, and place-in-time attributes are gathered to provide an estimate of forest conditions. Inventory values provide per acre estimates. When spatial data that complies with the vegetation stratification scheme is multiplied by temporal data obtained from field inventories, total strata estimates are produced. These values are then incorporated into the landscape assessment.

The spatial data source for the Blue Mountains was assembled using Gradient Nearest Neighbor technology (Ohmann and Gregory 2002). The primary data source available for the area was the USFS Pacific Northwest Region Current Vegetation Survey (CVS) (Max and others 1996) that predates the standardized installation of the Forest Inventory and Analysis (FIA) (Miles 2001) sample design. Table 1 summarizes the CVS inventory sample used to represent the Warm Dry Ponderosa Pine stand type on the Malheur, Umatilla, and Wallowa-Whitman National Forests.

Table 1—Data set used for the Warm Dry Ponderosa Pine stand type on the Blue Mountains Project Area.

National Forest	Inventory Occasion 1 [a]	Inventory Occasion 2 [b]	Total Sample
	- - - - - - - - number of plots - - - - - - - - -		
Malheur	156	120	276
Umatilla	33	33	66
Wallowa-Whitman	42	31	73
Total:	231	184	415

[a] Occasion 1: Initial Installation of Current Vegetation Survey

[b] Occasion 2: First Remeasurement of Current Vegetation Survey

Portioning the Warm Dry Ponderosa Pine stratum by size class resulted in the plot distribution listed in table 2. Stand age was incorporated into the definition of size class. Stand age provides a general measure of important ecological processes. Each successional stage provides structural components critical for particular plants and animals. Vegetation pathways developed for contemporary mid-scale plans account for distribution of stand structure across the forest landscape.

Table 2—Plot sample distribution by size class.

Stand type	Size class [a]	Stand age [b]	Plot sample
	Seedling-Sapling (0-5”)	0-20	7
Warm	Small Tree (5-10”)	20-50	31
Dry	Medium Tree (10-15”)	50-90	178
Ponderosa	Large Tree (15-20”)	90-140	143
Pine	Very Large Tree (20-25”)	140-200	31
	Giant Tree (25”+)	200-270+	6
Non-Vegetated [c]	Grass/Forbs/Brush	~	19
Total:			415

[a] Based on the quadratic mean diameter (qmd) of the largest 20 percent of the trees with a minimum of 20 trees.

[b] Origin date of the oldest cohort (i.e. qmd size class) inferred as time since the last stand replacement disturbance.

[c] Although non-vegetated with tree cover at the time of inventory, these plot samples reside in Warm Dry Ponderosa Pine plant associations.

Model Calibration

An essential step in appropriate use of the Forest Vegetation Simulator is calibration of the model. The FVS geographic variants are comprised of numerous mathematical relationships. One prediction equation may provide input to another. For long-term projections, users should validate virtual world estimates generated by FVS against real world values obtained from inventory data.

Determining the modeling context matters in regards to constructing a verifiable FVS projection. As conceived and implemented, stand development within FVS trends toward full site occupancy. If endemic or epidemic conditions are to be portrayed, users should be aware of existing software extensions and addfiles that account for insect, disease, and fire effects. These utilities address additional mortality impacts during long-term projections. In the absence of available disturbance model extensions, users assume the responsibility to ensure simulation runs provide reasonable results. Inventory data should be acquired that represents either full stocking, common conditions, or impacted landscapes to provide the sideboards for FVS model predictions. The focus of the remainder of this paper will be targeting endemic conditions that characterize vegetation pathways for landscape planning.

Figure 3 depicts measured trends in the Warm Dry Ponderosa Pine stand type as compared to modeled FVS projections of the seedling-sapling size class. The images were derived by averaging individual plots. The graphics portray structural trends from stand ages 10 to 260 years by size class increments. Note that the initial seedling-sapling size class images for the measured and modeled frames are identical. The modeled projections were produced by the Blue Mountain variant of FVS without and with user intervention. Comparing measured trends (left column in fig. 3) with FVS modeled projections, without adjustments (middle column in fig. 3), demonstrates vast differences in stand development.

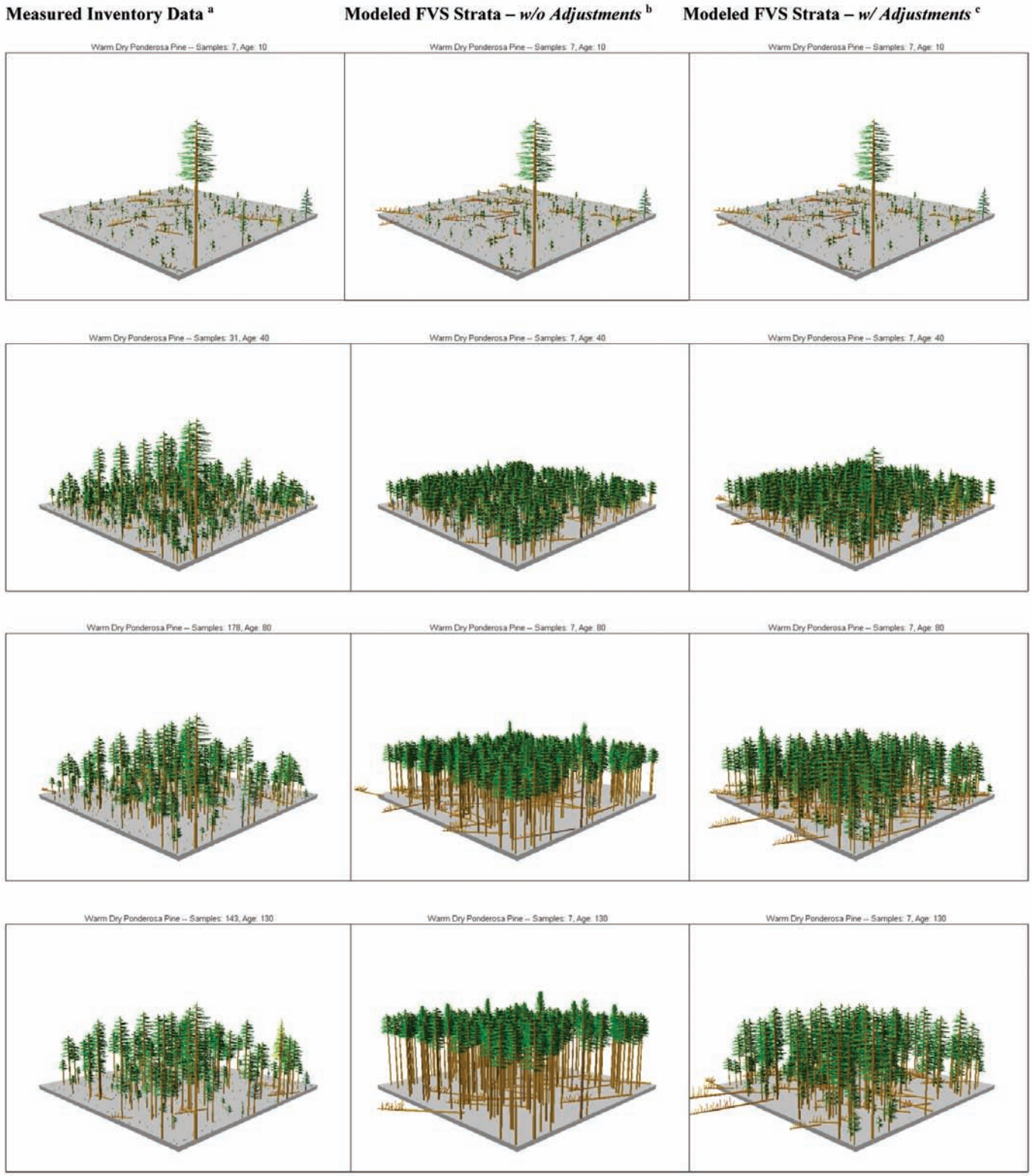

Figure 3—Measured versus modeled trends of Warm Dry Ponderosa Pine stratum.
[a] Left column: Measured inventory data from stand age 10 to 260 by size class increments.
[b] Middle column: Modeled strata projected from stand age 10 to 260 by size class increments without adjustments.
[c] Right column: Modeled strata projected from stand age 10 to 260 by size class increments with adjustments.

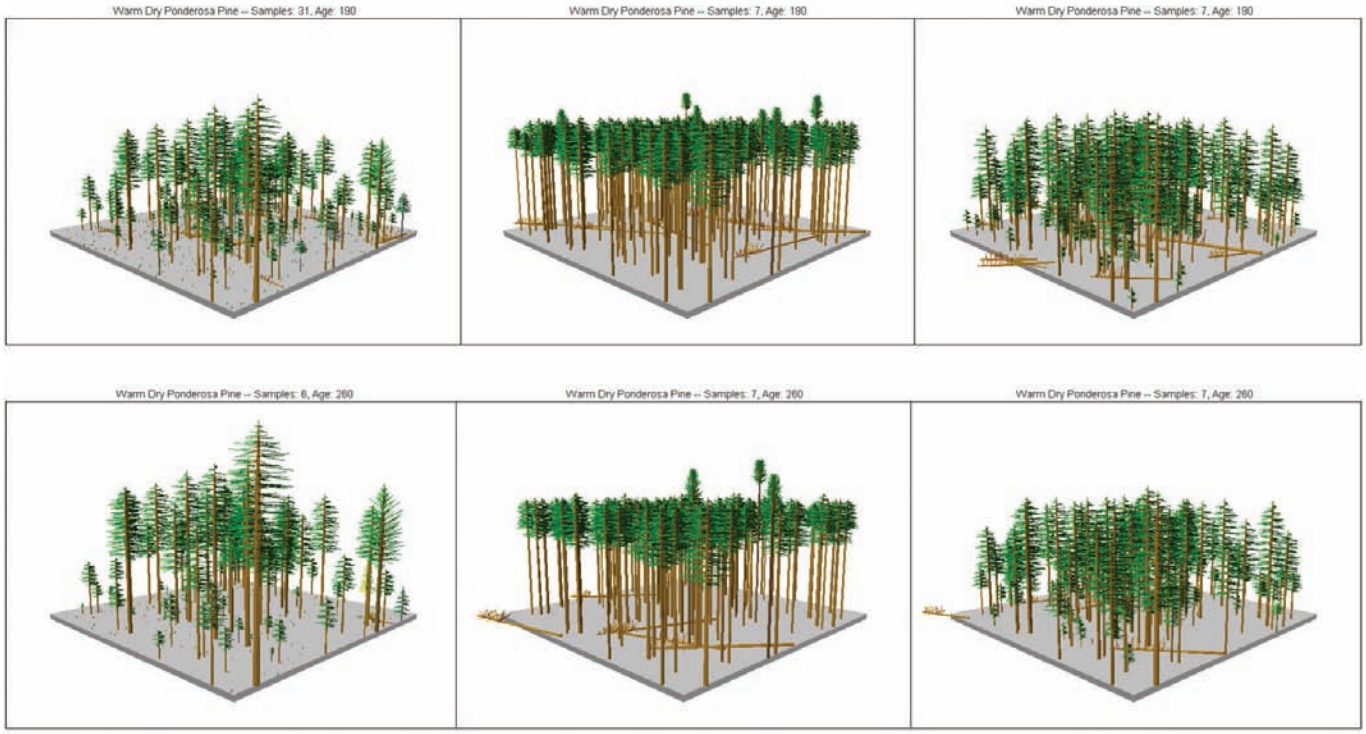

Figure 3—Continued.

Delving into two key attributes highlights the anomalies. Trees per acre plotted over stand age by size class increments are displayed in figure 4 (Inventory Data vs. FVS w/o Adjustment). Board foot volume per acre plotted over stand age by size class increments is displayed in figure 5 (Inventory Data vs. FVS w/o Adjustment). Contrasting the measured inventory data to the unadjusted modeled run over time, tree density is too low and stand volume is too high. Mortality and regeneration aspects are difficult components to model. One subtracts trees from the ecosystem; the other adds trees to it. The default mortality paradigm embedded in FVS allows stands to progress to full site occupancy (i.e. maximum stand density). However, this ceiling does not represent the generalized endemic growth pattern typified by inventory data sets used for landscape assessments. Consequently, densely stocked conditions are forecast. Note also that most FVS variants include only the partial establishment extension that lacks automatic natural regeneration features. Thus, tree frequencies steadily decline over time. Growth dynamics are accumulated in the survivor trees resulting in excessive stand volume.

FVS Self-Calibration

A unique feature of the Forest Vegetation Simulator is its ability to self-calibrate the small-tree height and large-tree diameter increment models based on measured growth rates per inventory plot (Stage 1981). This mechanism alters the interspecies tree competition from the basis that was used to build the model. Over long projections, species composition may differ from the base model forecast as a result of local observations.

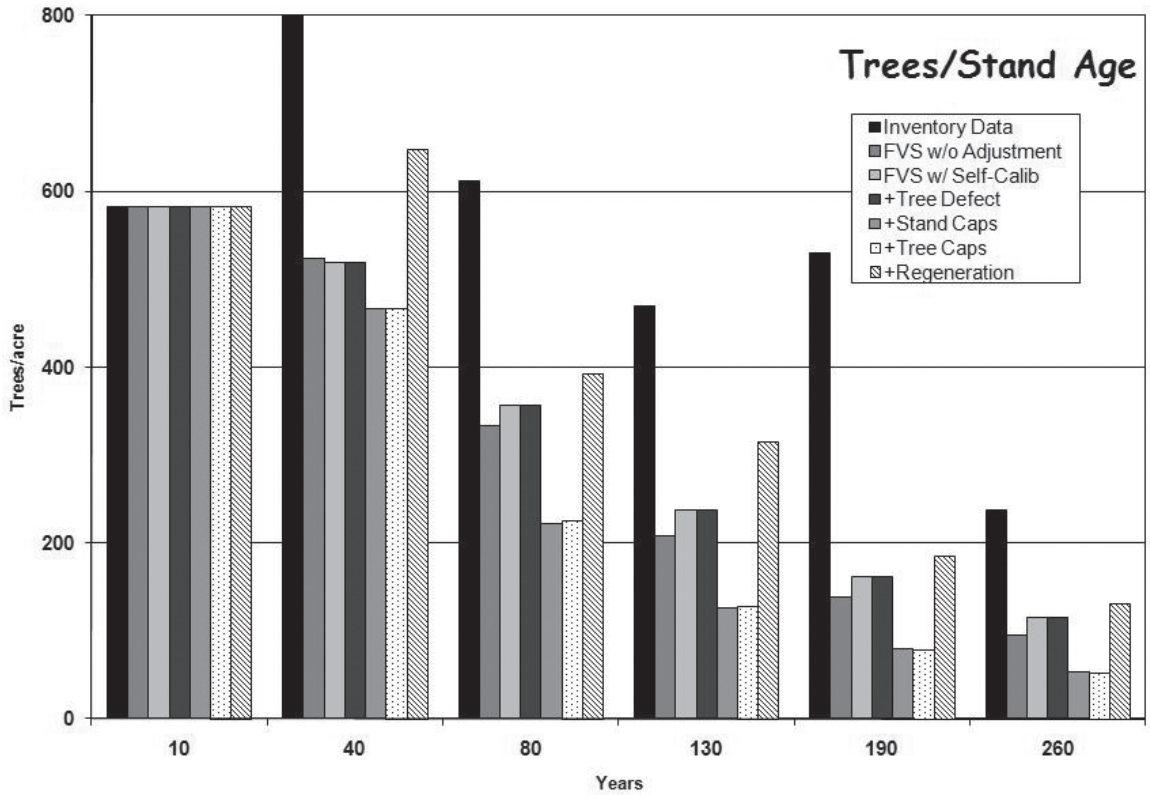

Figure 4—Comparison of trees per acre over stand age by size class increments for measured and modeled runs. Within grouping, recommended assembly steps are added to the unadjusted model run to properly configure FVS to conform to measured values.

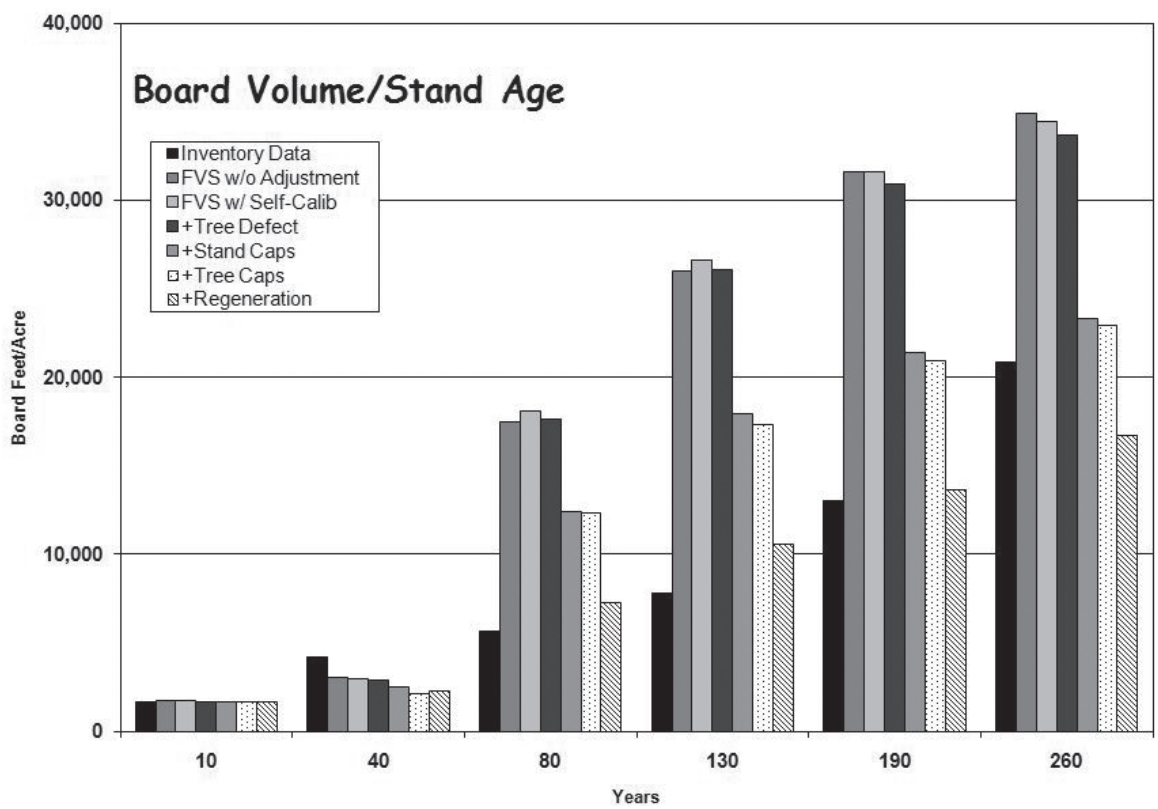

Figure 5—Comparison of board foot volume per acre over stand age by size class increments for measured and modeled runs. Within grouping, recommended assembly steps are added to the unadjusted model run to properly configure FVS to conform to measured values.

The FVS self-calibration process computes a scale factor that is used as a multiplier to the base growth equations. This scaling procedure is really quite simple. The affected models are linear with logarithmically scaled dependent variables. Therefore, the model intercepts are in effect growth multipliers. FVS predicts a growth increment to match each observed increment for a given species on a specific plot. The median difference is then added to the model for the species as an intercept term.

Growth multipliers can be developed across a large geographic area for a particular stand type. Using the CalbStat keyword in conjunction with the Calibration Summary Statistics post processing program produces average scale factors for all qualifying plots. ReadCorR (Readjust Correction for Regeneration) and ReadCorD (Readjust Correction for Diameter) keywords can be constructed from the mean multipliers listed in the Calibration Statistics Report. The ReadCorR and ReadCorD keywords alter the baseline estimate for small-tree height and large-tree diameter growth, respectively. For a particular species, the original baseline estimate is multiplied by the scale factor and the result becomes the new baseline estimate. These adjustments are done prior to the FVS self-calibration procedures per plot. Large-tree diameter increment scale factors attenuate toward the new baseline estimate at twenty-five year intervals as depicted in figure 6. Average scale factors computed for the Warm Dry Ponderosa Pine stand type within the Blue Mountains project area are displayed in table 3.

With respect to large-tree diameter growth, western larch (*Larix occidentalis* Nutt.) and Douglas-fir (*Pseudotsuga menziesii* [Mirb.] Franco) grow slightly faster whereas grand fir (*Abies grandis*), lodgepole pine (*Pinus contorta* Douglas ex Louden), and ponderosa pine (*Pinus ponderosa* C. Lawson) grow slightly slower (compared to a mean ReadCorD multiplier equal to 1.000) relative to the base model equations. Regarding this data set for long-term projections, the species composition differs slightly with the inclusion of the ReadCorD keyword than without.

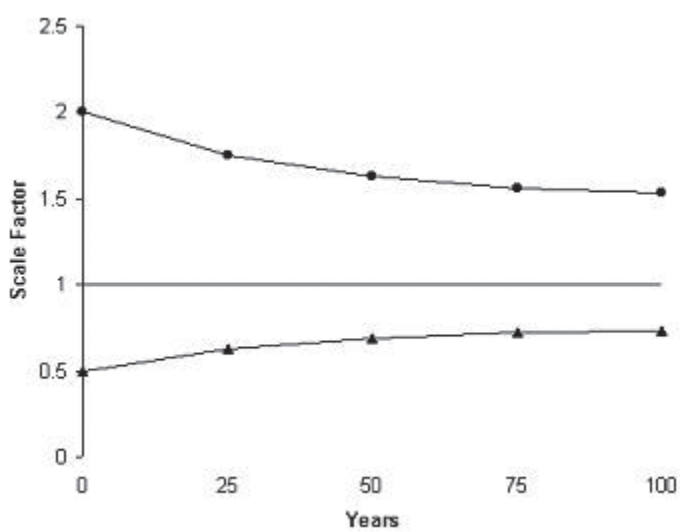

Figure 6—Scale factor attenuation occurs over time to the adjustment of the large-tree diameter growth equation.

Table 3—Calibration scale factors for the Warm Dry Ponderosa Pine stand type.

Tree species	Total tree records	Mean ReadCorR multiplier
Small-Tree Height Growth		
Ponderosa Pine	5	1.205
Large-Tree Diameter Growth		
Western Larch	32	1.096
Douglas-fir	118	1.001
Grand Fir	36	0.772
Lodgepole Pine	17	0.735
Ponderosa Pine	2741	0.813

FVS self-calibration does not address all 'measured' versus 'modeled' stand development discrepancies but rather only individual tree species small-tree height and large-tree diameter growth performance. In this case, due to the nominal differences in mean multipliers, board foot attainment was not dramatically improved by the inclusion of ReadCorR and ReadCorD keywords in the runstream. Refer to figure 5 (FVS w/ Self-Calib).

Tree Defect

Determining net merchantable volume from gross tree dimensions requires an estimate of tree defect. Values can be obtained from field inventory data or recent timber sales. For example, defect estimates obtained from CVS data for the Blue Mountains project area was arrayed by tree species, by 5-inch diameter classes, to populate the Defect keyword. Figure 7 displays the board foot defect trend for ponderosa pine and grand fir.

Consult the Keyword Reference Guide (Van Dyck 2006) or the Suppose (Crookston 1997) interface for specific parameter fields associated with the Defect keyword. Applying board foot defect factors normally affects larger diameter trees and aids in reining in runaway sawtimber volume. Given that the Warm Dry Ponderosa Pine strata is comprised primarily of ponderosa pine tree species, the magnitude of board foot reduction is slight as observed in figure 5 (+Tree Defect). Certainly, the Warm Dry Grand Fir strata would display a more dramatic effect from accounting for board foot volume defect.

Natural Growth Runs

Landscapes that have been heavily impacted by disturbances may require forest planners to estimate stand development beyond the pool of existing stand structures. Very large and giant tree size classes may be absent among currently inventoried stands. In these situations, projections of old growth development are needed. A reasonable assumption would be an extrapolation of existing circumstances toward a 'steady-state' condition extending into the future.

The goal of developing natural growth runs is to try to capture ecosystem processes that sustain stand types to their reasonable extent. Adjustments are taken into account for stand and tree level mortality components. As growing space opens, regeneration fills the void. Crafting an endemic natural growth profile requires addressing mortality and regeneration interactions.

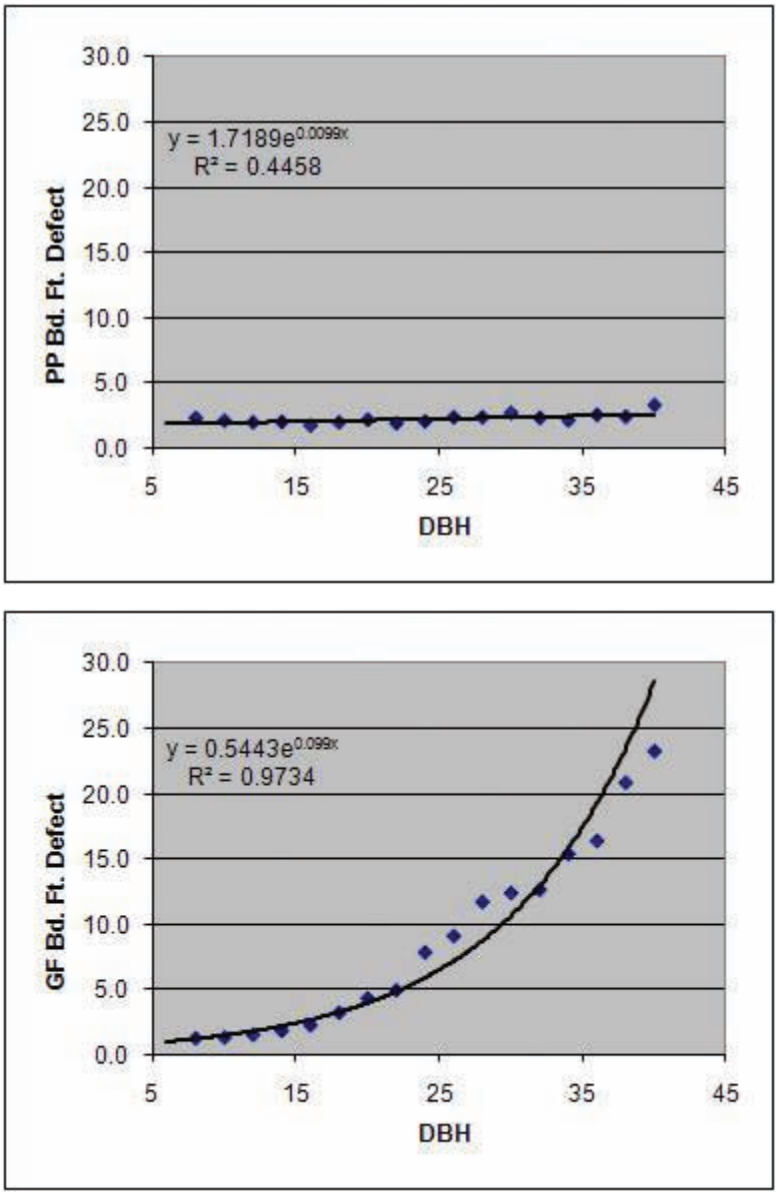

Figure 7—Measured board foot defect for ponderosa pine (PP) and grand fir (GF) on the Blue Mountains Project Area.

Stand Size Caps

The Forest Vegetation Simulator base model mortality predictions are intended to reflect mortality rates that allow for stand development to full site occupancy. Increases in mortality from insects, pathogens, and fire are accounted for in the various FVS model extensions. Mortality from other causes, such as logging operations, animal damage, or wind events, needs to be simulated with appropriate FVS keywords. There are three types of mortality base models used in FVS: (1) the original Prognosis type mortality model; (2) the Stand Density Index based mortality model; and (3) the Stand Density Index/TWIGS based mortality models.

The Prognosis type mortality model (Stage 1973) is used in western FVS variants where there were enough inventory data suitable for developing the associated equations. Two independent equations are involved. The first equation predicts an annual mortality rate as a function of habitat type, species, diameter, diameter

increment, estimated potential diameter increment, stand basal area, and relative diameter. The estimated annual mortality rate is multiplied by a factor based on Reineke's (1933) Stand Density Index (SDI) that accounts for expected differences in mortality rates on different habitat types and National Forests. The second equation estimates mortality loss and is dependent on the proximity of stand basal area (BA) to the assumed maximum for a site, and on the estimated rate of basal area increment. The mortality rate applied to a tree record is a weighted average between equation one and two. The weights applied to the respective estimates are dependent on the proximity of the stand basal area to the maximum basal area specified for a site or strata (BAMax).

The Stand Density Index (SDI) based mortality model (Dixon 1986) is used in western FVS variants where there were not enough inventory data suitable for developing the Prognosis type mortality model, and no other suitable mortality model existed. Mortality predictions for the Blue Mountains variant are SDI based. The model has two steps. In the first step, the number of mortality trees is determined; in the second step, this mortality is dispersed to the individual tree records in FVS. Two types of base model mortality are estimated: (1) background mortality and (2) density related mortality. Density related mortality accounts for mortality in stands that are dense enough for competition to be the causal agent. All other mortality is attributable to background mortality. Background mortality gives way to density related mortality based on the relationship between current and maximum Stand Density Index. By default within FVS, density related mortality begins when the stand SDI is above 55 percent of maximum SDI, and stand density peaks at 85 percent of maximum SDI. Background mortality is used when current stand SDI is below 55 percent of maximum SDI. In FVS terminology, the 55 percent value is referred to as the lower limit of density related mortality, and the 85 percent value is the upper limit.

The Stand Density Index/TWIGS based mortality models are used in all eastern FVS variants. Mortality losses are determined using the SDI base model method. Mortality values are then dispersed to individual tree records using relationships found in the TWIGS type mortality models (Buchman 1983; Buchman and Lentz 1984; Buchman and others 1983; Teck and Hilt 1990). These equations are variant dependent and actually predict survival rate, rather than mortality rate. Survival rate is predicted as a function of diameter, diameter growth, basal area in larger trees, and/or site index. The survival rate is converted to a mortality rate for FVS processing. In addition, background mortality is estimated as 1/10th of the calculated TWIGS mortality rate for each individual tree record.

Thus, basal area or stand density index maximum control mortality predictions and associated stand-stocking attainment in all FVS variants. The bases for the default maximum density relationships within FVS are research studies that principally include pure, fully stocked, uniformly even-aged stands. Since most forest stands on public lands are very heterogeneous in regard to stocking and structure, FVS without adjustment tends to overestimate the carrying capacity of common conditions. Model projections tend toward full site occupancy estimates. To account for endemic or epidemic mortality loss caused by insects, diseases, fires, or other disturbance agents, use of FVS model extensions should be investigated. In their absence, if exogenous data is available that can be used to parameterize mortality modifying keywords; this method ought to be explored. Lacking model extensions or mortality keyword support, a proxy for estimating the endemic average maximum density can be derived from the inventory data sets that are used for the landscape analysis.

Users can set maximum SDI and BA values that represent endemic conditions. The Forest Vegetation Simulator can be used to compute the measured SDI value for each inventory plot. Filtering the SDI calculation to include trees 1.0-inch and

greater in diameter enables excluding seedling tallies that can overwhelm the resultant value. Generally, the cluster of the top three percent of the inventory plots can then be averaged to determine the observed SDI maximum value[3]. To derive the average stand basal area maximum of the measured plots, multiply the observed SDI maximum value by 85 percent to represent an endemic maximum limit[4]. The observed basal area maximum is simply the basal area stocking most closely aligned with 85 percent of the observed Stand Density Index maximum value. The upper limit of density related mortality for the SDImax keyword should be set to 75 percent to represent the empirically derived endemic condition.

Table 4 contrasts the FVS default values for maximum SDI and BA (full site occupancy) against those derived from the inventory data (endemic conditions) for the Blue Mountains project area. Note that these two metrics have different basis and intended use. The FVS default values are indicative of theoretical stand density maximums. The data derived values are representative of average maximum stand density and have use for landscape planning projects.

The variant overview document should be consulted to determine the specific mortality model being employed for a given geographic area. The Blue Mountains variant uses the Stand Density Index based mortality model. Setting the stand density index maximum affects mortality predictions and stand-stocking attainment. Specifying a complementary basal area maximum is recommended. The effects of including an endemic stand density maximum that is more closely aligned to measurement data can be observed in figure 5 (+Stand Caps). Notice that as bars are being added to the right per stand age/size class grouping, the gap is closing between the inventory data and model projections with adjustments.

Tree Size Caps

The TreeSzCp keyword assists in setting the morphological limits for individual tree diameter and height development. The adjusted mortality rate is applied when a tree's diameter exceeds the threshold minimum diameter indicated for a given tree species. The threshold diameter acts as a surrogate for age to invoke senescence mortality. The process to parameterize the TreeSzCp keyword entails choosing the minimum diameter class that contains approximately one tree per acre (TPAmin) (the exact number is dependent on the relative abundance of a particular tree species) and targeting a maximum diameter class that contains

[3] Edminster (1988) stated that: "For each stand in the database (1400 stands from USFS Region 2 and 2939 stands from USFS Region 3), Stand Density Index (SDI) was calculated, and stands with SDI values in the upper two percent from each Region were selected for further analysis to develop the Average Maximum Density (AMD) line. There was nothing special about using the top two percent, but for the six forest types analyzed, the selection has resulted in AMD lines which represent 'average maximum' conditions". Given smaller data sets to work with for landscape assessments, using the top three percent of the inventory plot set to determine AMD is reasonable and has worked repeatedly well for determining "average maximum" stand density index for FVS.

[4] Powell (1999) indicated in "Table 3 – Characterization of selected stand development benchmarks or stocking thresholds as percentages of maximum density and full stocking" that: "*Maximum Density* is the maximum stand density observed for a tree species; although rare in nature, it represents an upper limit. *Full stocking* refers to 'normal yield table' values; it has also been termed as 'Average Maximum Density.'" As a "percent of maximum density," full stocking was cited by Powell at 80 percent. (Recall in FVS, that density related mortality trends toward full site occupancy and peaks at 85 percent of maximum SDI.) The "*Lower Limit of Self Thinning Zone*, also referred to as the 'zone of imminent competition mortality' (Drew and Flewelling 1979)" is referenced by Powell at 60 percent of maximum density. (Recall in FVS, that density related mortality begins when the stand SDI is above 55 percent of maximum SDI.) Use of 75 percent of average maximum density as represented by the top three percent of selected plots renders an upper mid-range value between full stocking and the lower limit of self thinning zone. This is the suggested target for the *Endemic Stand Density Index Maximum*. Through repeat application of this process, FVS projection results support using this methodology to set stand density maximums

Table 4—Stand density maximums: Default values from Blue Mountains variant; derived values from inventory data.

Stand type	SDI Maximum		BA Maximum (ft^2/acre)	
	FVS default	Data derived	FVS default	Data derived
Subalpine Whitebark Pine	700	535	325	205
Cold Dry Mixed Conifer	593	505	275	175
Cold Moist Mixed Conifer	597	460	277	170
Warm Dry Grand Fir	604	415	280	160
Warm Dry Douglas-fir	446	340	207	150
Warm Dry Ponderosa Pine	***416***	***330***	***193***	***135***
Hot Dry Ponderosa Pine	361	290	167	115
Woodland Western Juniper	420	170	195	75

approximately one-tenth tree per acre (TPAmax)[5]. Subtracting the associated diameter minimum from the diameter maximum, then dividing by the average diameter growth rate renders the number of FVS projection cycles needed to get from the minimum to the maximum size diameter. Determining the mortality rate is akin to computing the discount interest rate needed to pay off a capital sum. The mortality rate equals one minus the ratio of TPAmax to TPAmin raised to the power of one over the number of projection cycles. Note that the mortality rate compounds each projection cycle. Thus, this is the factor needed to diminish the tree count from one to one-tenth. The mortality rate becomes the proportion of trees to succumb to mortality agents during successive projection cycles.

Table 5 displays stand table values for ponderosa pine based on the inventory plots from the Blue Mountains project area. The goal of developing the TreeSzCp keyword is to ensure morphological senescence for a given tree species based on measured observations. On average, ponderosa pine tree frequencies diminish from one to one-tenth in the 22-inch to 36-inch diameter range. The weighted average annual diameter growth rate within this diameter range was measured to be 0.088 inches. This implies that it would take approximately 160 years (i.e. sixteen 10-year FVS projection cycles) for ponderosa pine trees to grow from 22 inches in diameter to 36 inches. This aligns with the stand age estimates associated with the progression from the *very large* to *giant* tree size class listed in table 2.

Table 6 presents the values used in the determination of the mortality rate for ponderosa pine. Since the TreeSzCp keyword is applied from projection cycle to projection cycle during simulation runs, its effect is exponential. The relationship between the mortality rate and surviving trees is asymptotic and therefore not all trees die. A few trees (or portions of tree records in FVS terms) remain and grow to larger diameter greater than the maximum diameter value.

[5] Shaw and others (2006) observed that: "However, the density-dependent self-thinning dynamic projected in the Southern variant of FVS may not be realistic for mature longleaf pine stands. Recent work suggests that the expected self-thinning trajectory does not hold for stands with a quadratic mean diameter greater than about 10 inches. Specifically, FVS projections of longleaf pine growth exceed the maximum limit of the size-density relationship, or "mature stand boundary," proposed by Shaw and Long (2007)." Application of the TreeSzCp keyword to the upper diameter range per tree species addresses this stand dynamic. Given that the mortality rate is applied exponentially over several projection cycles and across the "one-tenth" tree distribution target range, the TreeSzCp keyword implements the mature stand boundary concept. Repeat trials have supported methods used to construct the TreeSzCp keyword.

Table 5—Ponderosa Pine stand table, Blue Mountains Project Area, 415 inventory plots.

Diameter class (inches)	Trees/ acre	Average diameter (inches)	Diameter growth/year (inches)	Average height (ft)
<1.	261.518	0.10	0.302	1.0
2.	61.855	1.84	0.154	10.3
4.	42.051	3.87	0.111	18.4
6.	21.775	5.89	0.137	27.5
8.	18.171	7.92	0.132	36.3
10.	14.305	9.88	0.149	44.5
12.	10.320	11.87	0.141	52.5
14.	5.718	13.90	0.141	59.8
16.	3.727	15.86	0.133	66.6
18.	2.181	17.90	0.130	73.3
20.	1.538	19.86	0.100	79.6
22.	1.057	21.91	0.101	86.4
24.	0.715	23.84	0.092	91.6
26.	0.536	25.92	0.084	97.4
28.	0.406	27.89	0.082	100.7
30.	0.338	29.88	0.074	104.9
32.	0.193	31.91	0.072	109.3
34.	0.144	33.87	0.070	114.0
36.	0.091	35.93	0.079	118.2
38.	0.052	38.01	0.064	120.3
40.	0.088	42.12	0.063	128.0
Average: [a]			0.088	

[a] Weighted average annual diameter growth within specified range.

Table 6—Ponderosa Pine mortality rate computation.

Diameter class (inches)	Trees/ acre	Avg. diameter growth/year (inches)	10-Year project. cycles	Mortality rate[a]
22. — 36.	1.057 — 0.091	0.088	16	0.143

[a] Mortality Rate = 1 – $(TPAmax/TPAmin)^{1/Proj\_Cycles}$

Although the impact of including size caps per tree species appears minimal in regards to board foot volume achievement, the importance can be observed in terms of tree frequency per stand size class. Refer to figure 4 (+Tree Caps +Regeneration). Having larger trees die provides opportunity for new trees to become established. These young recruits then grow in diameter and progress through the various size classes as would be expected in natural stand development.

Regeneration Inference

In the last stand visualization image for the FVS modeled run "without adjustments" (fig. 3, middle column), the lack of understory trees is readily apparent. Two versions of the Regeneration Establishment extension are used in FVS variants. One version is referred to as the "full" establishment model (Ferguson and Carlson 1993). This version has been calibrated for western Montana, central and northern Idaho, and coastal Alaska. It includes the full array of establishment options. The other version is referred to as the "partial" establishment model. This version only simulates regeneration from planting or stump sprouting; users must provide estimates of natural regeneration using keywords. The partial establishment model is used in variants for which the full establishment model has not been calibrated.

Most FVS variants rely on the partial establishment model. For those variants that support the full establishment model, the "with adjustment" run would have shown improved results in overall stand structure. The Blue Mountains variant relies on the partial establishment model. An empirical approach was used for estimating the natural regeneration response over time. Note that the tree count and board foot volume in figures 4 and 5 (+Regeneration) respectively reveal a comparable trend between measured data and model results. When regeneration is excluded, tree counts per acre steadily decline. When regeneration is included, canopy gaps caused by mortality agents are readily filled. Standing board foot volume is maintained in older age classes.

There are several methods available to induce a natural regeneration response for FVS variants that support only the partial establishment model. A portal in the FVS source code allows external programs to interact during the regeneration processing sequence (Robinson 2007). An understory estimation procedure has also been developed to import seedling/sapling recruitment (Vandendriesche 2009). One last method that should not be overlooked is local sources of expert opinion. Sophisticated FVS keyword sets can be constructed that depict expected natural regeneration response. Regardless of the process used, whether automatically invoked or user supplied, appropriate regeneration inferences are needed to configure a realistic FVS projection.

Board foot volume displayed in figure 5 (+Regeneration) reveals a significant reduction as a result of including regeneration impulses into the FVS projections. In conjunction with the tree size cap that injects overstory senescence, small trees can then occupy open space and influence the mortality prediction/stand density dynamic. Large trees are not allowed to simply grow and accumulate board foot volume. Large trees succumb to morphologic processes, which makes room for small tree establishment.

Assembly Process Summarized

Natural growth runs are a common starting point in the development of vegetation pathways for landscape assessments. It is quite possible in this day of constrained management that many stands will be left to let grow through the planning horizon. From a vegetation modeling standpoint, this scenario may appear to be the simplest to construct. However, due to our limited knowledge of

older stand structures, this run stream may require the most time and imagination. Cultured stands are fairly straightforward with regard to stocking density at various stand ages. Also, the regeneration response may be highly regulated. Natural stands that are left to grow are more intricate to model. Forests are not static and in some cases are very dynamic over short periods of time.

Table 7 summarizes the assembly steps and associated FVS keywords that were employed to craft the endemic FVS runs for the Blue Mountains landscape assessment. Note that while the assembly steps are imperative, the suite of recommended FVS keywords is not exhaustive or conclusive. Use of the CrnMult or Prune keywords may be needed to adjust crown estimates during the model calibration phase. If priori information is available, perhaps the MortMult and FixMort keywords would be more appropriate for configuring the stand and tree size caps. If supportive expertise is accessible, use of existing insect and disease extensions and keyword component addfiles should also be explored.

Stand visualizations displayed in the right column of figure 3 show the improvements of applying the recommended assembly process. Comparing FVS "out of the box" with recommended adjustments, the difference between the measured trends and modeled projections is minimal. Individual plots have varying species compositions and structures but given sufficient time in the absence of disturbance will strive toward normality.

Looking again at two key characteristics highlights the similarities between the measured inventory data and the modeled FVS run with adjustments. Trees per acre plotted over stand age are displayed in figure 4. Board foot volume per acre plotted over stand age is presented in figure 5. Notice the magnitude in the inventory data bars on the left versus the fully assembly bars on the right of stand age/size class groupings. Reasonable similarity exists in terms of trees per acre and board foot volume attainment.

Table 7—Recommended checklist for assembling a FVS run.

Assembly steps:	FVS keywords:
Model calibration	
1. FVS Self-Calibration	{ReadCorR, ReadCorD}
2. Tree Volume Defect	{Defect}
Natural growth run	
3. Stand Stocking Attainment	{SDIMax, BAmax}
4. Tree Senescence Caps	{TreeSzCp}
5. Regeneration Response	{Natural}

Treatment Prescriptions

Once natural growth runs have been constructed using recommended FVS assembly techniques, various silvicultural prescription scenarios can be modeled. Management direction suggest action, be it passive or active. For planning projects, certain stand level treatments are postulated as potential activities to move the forest toward desired outcomes. For example, it may be proposed to reduce stocking densities to lessen insect impacts. Also, it may be recommended to provide remedial fuel treatments to minimize wildland fire intensity. Additionally, it may be advocated to produce a balance of stand size classes throughout the forest to furnish a full spectrum of wildlife habitats. Furthermore, it may be

beneficial to explore predicted changes in climate. For each proposed action, a stand treatment schedule needs to be formulated to achieve the stated goal. The natural growth runs described in the previous section are a de facto prescription option to let stands grow with minimal management intervention.

Vegetation Pathways

A fundamental step in landscape planning is the analysis of the management situation. Various alternatives are proposed to guide future programmatic direction. Inherent to the analysis process is the gathering of inventory data and the projection of potential outcomes. Computer models play an important role in the projection process and formulation of management alternatives. Generally, two types of computer models are used for mid-scale forest planning: yield forecasting models and decision support systems. Yield models summarize current conditions and project future developments thus providing point-in-time value estimates. Decision support models pull together the state and transition components of forest planning. Coefficients computed by yield models are used by decision support systems to address management issues.

Evaluate Output

Preparing vegetation profiles in support of landscape planning projects is dissimilar to processing inventory data through static software. You can not simply feed data in one end and produce meaningful output at the other. Professional talents, including those of a mensurationist, ecologist, silviculturist, and forest analyst, are required to construct valid trends in vegetation development. This is not a complete list of specialty skills. Possessing these abilities does not ensure proper integration of tasks. Formal experience on several projects aids in solidifying the corporate memory to conduct such analyses. There is as much art as science that goes into the process.

Conclusions

In the *ideal* world of vegetation modeling, each of the fundamental biological processes that occur on an individual tree basis (i.e. diameter, height, and crown development) should be examined to verify expected performance. Does the diameter increment seem reasonable; do dominant trees obtain site-height at given stand ages; does tree crown development appear correct in open and closed canopy situations? Each of these aspects should be questioned and verified. Stand dynamics should also be checked. Are average maximum stocking densities obtainable; are successional trends captured; are forest gaps reclaimed by regeneration? These are basic stand level tenets that should be confirmed.

In the *real* world of vegetation modeling, time constrains such rigorous substantiation of predicted outcomes. Regardless, if a project requires analysis of one stand or many stand types, inventory data should be acquired that depicts the condition of interest. Calibration and maximum values can be derived from the associated data set. Regeneration response can also be gleaned empirically from measurement data. These are the facets that need to be examined and incorporated into a fully assembled FVS projection.

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The content of this paper reflects the views of the authors, who are responsible for the facts and accuracy of the information presented herein.

An Empirical Approach for Estimating Natural Regeneration for the Forest Vegetation Simulator

Don Vandendriesche[1]

Abstract—The "partial" establishment model that is available for most Forest Vegetation Simulator (FVS) geographic variants does not provide an estimate of natural regeneration. Users are responsible for supplying this key aspect of stand development. The process presented for estimating natural regeneration begins by summarizing small tree components based on observations from local Forest Inventory and Analysis (FIA) data sets. Average frequency and height for seedlings and small saplings by tree species are calculated from the data. Seedling recruitment is determined by relationships to the maximum stand density index. Small saplings regeneration is derived from a distribution pattern computed from current stand size and density conditions. Apportioning the small sapling frequencies according to the observed distribution pattern provides an expected natural regeneration estimate by stand size/density condition, or vegetation state. Associated small sapling shade tolerance factors can be applied to govern progressions in species composition across the stand size/density continuum. Repute, an FVS post-processing program, has been developed to automate the estimation process. An example from the Southwest U.S. is presented to demonstrate the methodology.

Introduction

Predicting the abundance of natural regeneration, that is seedling and sapling recruitment beneath an existing overstory, is a difficult task required of forest growth modeling. Regeneration response is a stochastic event and the phenomena that trigger adequate natural regeneration are not easily determined. Inventory data sets that could be used to develop algorithms rarely cite the statistical parameters needed to effectively predict regeneration establishment (Ferguson 1997). As a result, prediction estimates for regeneration are generally weak in a system that must account for all facets of stand development. The objective of this paper is to present a process based on seedling and sapling counts as summarized from existing inventory conditions that can be used to provide understory recruitment for the Forest Vegetation Simulator (FVS) (Dixon 2002).

Forest models can be designed based on either *growth* or *yield* processes. The distinction is that growth architecture integrates growth processes over time to estimate future states whereas yield architecture directly estimates the future state as a function of time. Regeneration models that use a growth-based approach tend to be complex and predict too many poorly understood processes (Stage 2002). Such models are rare. Instead, most use a yield-based approach to assemble a picture of stand regeneration (Ferguson and others 1986). The estimation process described here follows the yield-based method. Observed regeneration attributes are drawn from FIA samples to estimate the natural regeneration component of a given plot of the same ecological stratum. This approach implicitly presumes that

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the inferences driving regeneration in the inventoried plots applies to regeneration of modeled plots (Ek and others 1997).

Relevant Questions

Simply assuming that regeneration will only occur following a disturbance is wholly inadequate for providing a fully functioning regeneration model. *Tree based mortality* resulting from morphological senescence creates canopy gaps enabling thrifty understory recruits to fill the void. *Stand based mortality* resulting from overstory crown closure provides a successional opportunity for shade tolerant understory tree species to take hold. Regeneration occupancy must be considered for each combination of stand size and density class. For every vegetation state, there are two basic questions to be addressed:

Quantity ⇢ How many seedlings/saplings should be established?
Quality[2] ⇢ What is the species composition of those established?

For this estimation procedure, both seedlings and small saplings are taken into account for natural regeneration. Flushes of seedlings pulse into stand projections on a twenty-year period. Mortality algorithms within FVS take their toll on the seedling component. To ensure recruitment into larger diameter size classes, regulation of small saplings (trees between one and three inches diameter) is pursued. A distribution pattern for small sapling abundance is computed relative to the current vegetation state based on stand size/density criteria.

Stand structure, in terms of size and density, is defined using overstory quadratic mean diameter (qmd) and percent canopy cover, respectively. There are five recognized stand size classes: 0-5 inch, 5-10 inch, 10-15 inch, 15-20 inch, and greater than 20 inches overstory qmd. Overstory qmd is comprised of the largest 20 percent of the trees by diameter, with a minimum of 20 trees needed for the classification. Otherwise, overstory qmd is determined by the largest diameter trees that constituted 25 percent of the canopy cover. Three stand density classes are defined: 10-40 percent, 40-70 percent, and greater than 70 percent canopy cover (Crookston and Stage 1999). If the canopy cover is less than 10 percent, the density class is classified as nonstocked.

The size and density classification system was derived from forest plan revision work for the Colville, Okanogan, and Wenatchee National Forests located in Washington State (Jahns 2006). Vegetation states were developed from National Standards for Geospatial Data (USDA Forest Service 2003) and supplemented with Standards for Mapping of Vegetation in the Pacific Northwest Region (USDA Forest Service 2004). The national/regional standards proposed a resolution of vegetation classification that was far more detailed than necessary to address forest plan revision efforts. However, the standards allowed flexibility for aggregation into appropriate vegetation classes. Regional wildlife specialists were consulted for advice on appropriate size and density aggregations for species viability and proposed the aforementioned classification scheme.

[2] Quality in terms of species composition may be ambiguous but is used in the context of regeneration establishment to imply a tree species' ability to survive relative to a given site potential or relative to its tolerance of overstory shade.

Pattern Revelation

A supporting premise of the proposed natural regeneration procedure is that there is a regular periodicity of good seed years and subsequent seedling establishment that is normally followed by equally impressive seedling mortality. This "burst" and "bust" dynamic relative to actual survivorship is difficult to capture in forest growth modeling. Consequently, the tactic devised for predicting adequate natural regeneration targets the more stable population of small saplings. These trees generally have an acceptable girth and height to indicate having a root system firmly entrenched in mineral soil. Furthermore, they possess the capacity either to fill gaps due to overstory tree mortality or to follow successional recruitment trends as dictated by species shade tolerance.

Conditionally, seedlings are input into an FVS projection once per twenty-year period. During that time, seedlings are subject to the various FVS mortality mechanisms. A complimentary check is added to evaluate whether sufficient small saplings survived from the pool of seedlings to ensure a viable stream of ingrowth into larger diameter trees. Thus, a fully functioning regeneration system is in-place. The following text will detail the process.

Southwest Prototype

Many of the concepts behind the natural regeneration estimation process arose from work for landscape analysis. An example is presented based on recent forest planning efforts from the U.S. Forest Service Southwest Region. There are six National Forests in Arizona and five National Forests in New Mexico encompassing 20.6 million acres (fig. 1). Several terrestrial life zones exist within this diverse geographic area. Recent periodic and annual Forest Inventory and Analysis data sets were assembled to represent this landscape. Forest cover types, as comprised of plant associations, were used to stratify the data. FIA plot counts by cover type are shown in table 1. The Forest Vegetation Simulator can be used to model stand development of timberland and woodland communities. Methods developed for estimating natural regeneration will be described for the *Mixed Conifer—Dry* type.

Figure 1—Geographic location of National Forests within the USFS Southwestern Region.

Table 1—FIA plot count by forest cover type.

Forest cover	Type descriptor	Plot sample
Spruce-fir	Upper subalpine	69
	Lower subalpine	72
Mixed conifer	Wet—infrequent fire	48
	Dry—frequent fire	***444***
Ponderosa pine	Grass	495
	Gambel oak	295
	Evergreen oak	132
Pinyon-juniper	Woodland	1,354
Oak	Woodland	308
Total:		3,217

Inspection of the observed small sapling density within the Mixed Conifer-Dry strata (table 2) reveals the following pattern: Starting with any canopy cover class (row), then reading from the largest to smallest stand size class (columns, right to left), each smaller size class contains two to three times as many saplings as the next larger size class. Applying the average observed small sapling distribution factor (2.6330) among the size and density classes as shown in the bottom half of table 2 does a good job of smoothing inordinate jumps between adjacent classes. Derivation of the small sapling distribution factor will be described in the following paragraphs.

The natural regeneration estimation process begins by determining the vegetation state (i.e. size/density class) of each plot in the inventory. A reference template is displayed in table 3. A two-digit number indexes the size/density class combination. The first digit represents columns (qmd stand size classes), and the second digit represents rows (canopy cover density classes). For example, vegetation state 22 would indicate a size class of 5-10 inch qmd and a density class of 40-70 percent canopy cover.

Table 2—Mixed conifer—dry: small sapling distribution.

Canopy cover class	Overstory QMD size class				
	0-5	**5-10**	**10-15**	**15-20**	**> 20**
(%)	- - - - - - - - - - - - -		*(inches)*	- - - - - - - - - - - - -	
Observed small sapling frequency (trees/acre)					
10-40	739	233	59	62	15
40-70	1816	417	179	36	25
> 70		545	256	112	
Predicted small sapling frequency (trees/acre)[a]					
10-40	417	158	60	23	9
40-70	1098	417	158	60	23
> 70	2891	1098	417	158	60

[a] Based on small sapling distribution pattern factor of 2.6330 derived from observed data.

Table 3—Mixed Conifer—Dry: Size/Density Template and FIA plot count.

Canopy cover class	Overstory QMD size class				
	0-5	5-10	10-15	15-20	> 20
(%)	- - - - - - - - - - - - - *(inches)*- - - - - - - - - - - - -				
	Code reference guide[a]				
10-40	11	21	31	41	51
40-70	12	22	32	42	52
> 70	13	23	33	43	53
	FIA plot sample[b]				
10-40	14	50	32	6	5
40-70	13	121	92	19	6
> 70		11	12	4	

[a] Code reference guide provides a template for the size/density classes. A 22 code indicates a 5-10 inch qmd size class and a 40-70 percent canopy cover density class.

[b] Of the 444 total FIA plots sampled within the Mixed Conifer-Dry strata, 59 were classified as nonstocked (canopy cover less than 10 percent) rendering 385 plots for small sapling analysis.

The six size/density classes used to compute the small sapling distribution factor are vegetation states: 21, 22, 31, 32, 41, and 42 (referred to as base cells). These cells contain the majority of FIA plots as displayed in the bottom half of table 3. Base cells are of primary interest in the natural regeneration estimation procedure. Note there is little need for additional natural regeneration in smaller size or closed density classes. These vegetation states are by definition either a "seedling/sapling" size class (i.e. 0-5 inches) or at "full site occupancy" in terms of stand density (> 70%). Accordingly, additional natural regeneration is not targeted for the 12, 13, 23, 33, 43, or 53 size/density classes.

The distribution factor is calculated as a weighted average of the base cells. Table 4 presents the calculation method. Vegetation states lying adjacent to base cells are used to determine a distribution factor for the base cell. The representative FIA plot samples serve as weights in the calculation. A composite distribution factor is determined by summing the base cells multiplied by their FIA plot sample and dividing by the total number of FIA plots residing in the base cells.

Table 5 displays composite small sapling distribution factors for each of the major forest cover types in the Southwestern Region. A trend can be noted related to the composition of timberland tree species compared to woodland. As timberland species diminish and woodland species increase in prevalence, the composite small sapling distribution factor increases. Given the relative proportional size and site occupancy of woodland tree species, this trend is reasonable.

Repute Program

Repute, an FVS post-processing program, has been written to implement the concepts of natural regeneration estimation process. The main Repute window is shown in figure 2. In brief, the Repute program processes output files from the Fvsstand Alone program (Vandendriesche 1997) to develop regeneration "addfiles" (i.e. auxiliary FVS keyword files). Within a FVS run, the Repute addfile determines the vegetation state (size/density class) as each projection cycle is processed; associates the composite distribution matrix that was based

Table 4—Mixed conifer—dry: calculation method for composite distribution factor.

Canopy cover class	Overstory QMD size class				
	0-5	5-10	10-15	15-20	> 20
(%)	- - - - - - - - - - - - - *(inches)*- - - - - - - - - - - - -				
	Size/density distribution factors[a]				
10-40		2.3270	3.2815	1.1006	
40-70		2.2573[b,c]	2.6764	4.5281	
> 70					
	Composite distribution factor: 2.6330[d,e,f,g]				

[a] Refer to Table 2 (T2) – Observed Small Samplings Frequency for tree per acres values; Table 3 – Code Reference Guide for indicated size/density class; and, Table 3 (T3) – FIA plot sample for associated weights.

[b] Size/Density 22 Distribution Factor = ((T2:12/T2:22)*T3:12+(T2:22/T2:21)*T3:21+(T2:22/T2:32)*T3:32+(T2:23/T2:22)*T3:23) / (T3:12+T3:21+T3:32+T3:23)

[c] 2.2573 = ((1816/417*13)+(417/233)*50+(417/179)*92+(545/417)*11) / (13+50+92+11)

[d] Refer to Table 3 – Code Reference Guide for indicated size/density class; Table 3 (T3) – FIA plot sample for associated weights; and, Table 4 (T4) – Size/Density Distribution Factors for class values.

[e] Composite Distribution Factor = (T4:21*T3:21+T4:22*T3:22+T4:31*T3:31+T4:32*T3:32+T4:41*T3:41+T4:42*T3:42) / (T3:21+T3:22+T3:31+T3:32+T3:41+T3:42)

[f] Base Cell sample size must contain 10 or more counts. Note that the plot sample for Base Cell 41 (T3:41) equaled six and therefore was reset to zero.

[g] 2.6330 = (2.3270*50+2.2573*121+3.2815*32+2.6764*92+1.1006*0+4.5281*19) / (50+121+32+92+0+19)

Table 5—Composite small sapling distribution factors by forest cover type.

Forest cover	Type descriptor	Distribution factor
Spruce-fir	Upper subalpine	2.1199
	Lower subalpine	1.9529
Mixed conifer	Wet—Infrequent fire	1.9700
	Dry—Frequent fire	***2.6330***
Ponderosa pine	Grass	2.9019
	Gambel oak	3.0383
	Evergreen oak	2.8769
Pinyon-juniper	Woodland	2.7820
Oak	Woodland	2.8083

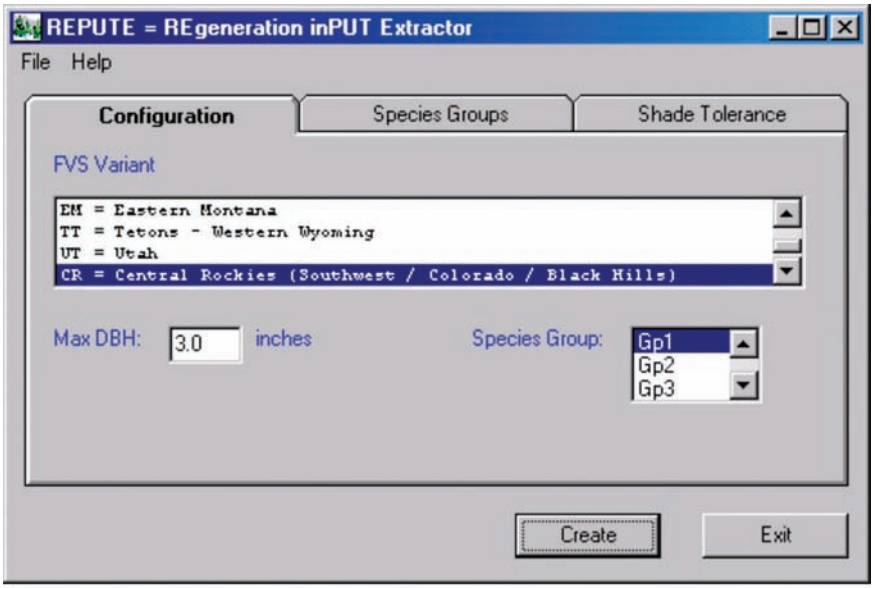

Figure 2—Repute user interface.

on observed small sapling frequencies; applies shade tolerance multipliers if applicable; inputs seedling reproduction; and, estimates small sapling ingrowth.

Implementation of the natural regeneration estimation procedure relies heavily on the Event Monitor (Crookston 1990) capabilities within the FVS model. There are limitations associated with using the Event Monitor that must be recognized such as the number of "compute variables" (199) and "conditional statements" (160). Conserving available resources of the Event Monitor is important. The natural regeneration estimation procedure was developed to be as efficient as possible. "Economies of Scale" principles were invoked. As developed, the vegetation classification system identifies fifteen combinations of size and density. Computing a composite distribution factor based on observed small sapling data allows rendering the fifteen size/density classes as a function of one multiplicative value.

Computed Size/Density Class

The natural regeneration estimation process begins by determining the vegetation state (i.e. size/density class) at each FVS projection cycle (fig. 3). The most important steps are outlined as follows. The Event Monitor DBHDist function is used to compute size class based on the largest 20 percent of the trees. Canopy cover, corrected for overlap, is used to calculate density class based on all trees. A qualifier variable (CCx) is computed to determine whether a plot is stocked (greater than or equal to 10 percent canopy cover) or nonstocked (less than 10 percent canopy cover) at the beginning of the projection cycle. If classified as nonstocked, the vegetation state is set to zero. A "regeneration state" variable is defined and set equal to the overstory vegetation state. However, there are times when a "bumper" regeneration pulse is allowed. Two additional statements are used to indicate the need. If a plot's current stocking level is less than 25 percent of the specified Stand Density Index maximum (Long 1985) and if there is a minimum of at least 10 trees greater than nine inches in diameter (seed trees), then the regeneration state is set to equal one resulting in a significant flush of natural regeneration injected into the projection.

```
Line Number   Column Ruler
              ----+----1----+----2----+----3----+----4----+----5----+----6----+----7----+----8
                 Keyword      fld1      fld2      fld3      fld4      fld5      fld6      fld7
1   ******************************************************************************
2   *   Compute Vegetation State                                                 *
3   *   - Composite: Size Class = QMD 5-inch class widths                        *
4   *                Overstory Density = Canopy Cover S=10-40%/O=40-70%/C=70+%   *
5   ******************************************************************************
6   Compute            0
7   _TotTrCt =Max(SpMcDBH(1,All,0,0.20,99.0,5.0,500.0,0),1)
8   _LrgTrCt =Max((SpMcDBH(1,All,0,0.20,99.0,5.0,500.0,0)*0.20),20)
9   _DiaL    =DBHDist(3,_LrgTrCt/_TotTrCt*100)
10  _DiaU    =DBHDist(3,6)
11  _QMDc    =SpMcDBH(5,All,0,_DiaL,_DiaU,5.0,500.0,0)
12  _CC00p   =SpMcDBH(7,All,0,00.00,99.0,0.0,500.0,0)
13  _CCx     =LinInt(_CC00p,0,10,10,999,0,0,1,1)
14  _SizClss =LinInt(_QMDc,00,05,05,10,10,15,15,20,20,99,1,1,2,2,3,3,4,4,5,5)*_CCx
15  _DenClss =LinInt(_CC00p,0,40,40,70,70,999,1,1,2,2,3,3)*_CCx
16  _VegSt   =(_SizClss*10)+_DenClss
17  _SDImin  =LinInt(SpMcDBH(11,ALL,0,0.2,200,1,500,0), &
18                   0,BSDIMAX*0.25,BSDIMAX*0.25,9999,0,0,10,10)
19  _SdTree  =LinInt(SpMcDBH(01,ALL,0,9.0,200,1,500,0), &
20                   0,10,10,99999,0,0,1,1)
21  _RgnSt   =LinInt(_SDImin+_SdTree,0,1,1,2,2,9, &
22                   0,0,1,1,(_SizClss*10+_DenClss),(_SizClss*10+_DenClss))
23  End
```

Figure 3—FVS keyword addfile segment that computes size/density class (vegetation state) for each projection cycle.

Multiplicative Distribution Factor

A predominant vegetation state is determined from the observed FIA sample and is simply the size/density class that contains the most plots. This state is designated as the "index" cell. For the Mixed Conifer—Dry strata, size/density class 22 is the predominant vegetation state as a result of being comprised of 121 FIA plots from a total of 444 FIA plots (bottom of table 3). Across all FIA plots within this vegetation state, the average small sapling frequency is 417 trees per acre. The composite distribution factor is applied relative to the index cell. In this example, the 22 size/density class receives a multiplicative factor equal to 1.0000 (fig. 4). Successive smaller size classes and denser canopy cover classes

```
Line Number   Column Ruler
              ----+----1----+----2----+----3----+----4----+----5----+----6----+----7----+----8
                 Keyword      fld1      fld2      fld3      fld4      fld5      fld6      fld7
1   ******************************************************************************
2   *   Small Sapling Composite Distribution Matrix                              *
3   *   - 2.6330 Factor based on RegSt 22                                        *
4   ******************************************************************************
5   Compute            0
6   _Sapp=LinInt(_RgnSt,      0,        1,        1,       11, &
7                            11,       12,       12,       13,       13,       21, &
8                            21,       22,       22,       23,       23,       31, &
9                            31,       32,       32,       33,       33,       41, &
10                           41,       42,       42,       43,       43,       51, &
11                           51,       52,       52,       53,       53,      100, &
12                       1.0000,   1.0000,   2.6330,   2.6330, &
13                       1.0000,   1.0000,   2.6330,   2.6330,   6.9326,   6.9326, &
14                       0.3798,   0.3798,   1.0000,   1.0000,   2.6330,   2.6330, &
15                       0.1442,   0.1442,   0.3798,   0.3798,   1.0000,   1.0000, &
16                       0.0548,   0.0548,   0.1442,   0.1442,   0.3798,   0.3798, &
17                       0.0208,   0.0208,   0.0548,   0.0548,   0.1442,   0.1442)
18  _Saps= 417
19  End
```

Figure 4—FVS keyword addfile segment that associates a multiple of the composite distribution factor based on dominant "22" size/density class.

are multiplied by the composite distribution factor. Successive larger size classes and less dense canopy cover classes are divided by the composite distribution factor. This allows apportioning the index cell's sapling frequency (417 trees per acre) across all size/density classes and renders the predicted values as displayed in the bottom portion of table 2. As such, the question of regeneration "quantity" is addressed.

Shade Tolerance Multipliers—Small Saplings

In the absence of disturbance events, either human or natural, tree species succession follows capacities for survival in light of overstory shading (Spurr and Barnes 1973). Transitions in species composition do not affect all forest cover types equally. For example, the species ponderosa pine (*Pinus ponderosa* Dougl. ex Laws) is considered shade "intolerant" (USDA 1990). If found in a Ponderosa Pine forest cover type, it is very resilient and will regenerate as opportunities arise. However, if ponderosa pine is found in a Mixed Conifer forest cover type, it will struggle to survive without disturbance. Shade tolerance considerations governing ponderosa pine regeneration are important in the latter situation, but not in the former. The Repute program allows the user the option to include shade tolerance effects for small saplings. This feature addresses the "quality" aspect regarding species composition of natural regeneration.

Figure 5 displays small sapling shade tolerance multipliers used for intolerant species such as ponderosa pine. Shade tolerance multipliers were developed using inventory data sources from throughout the Interior West (Colville-Okanogan-Wenatchee National Forests, Clearwater-Nez Perce National Forests,

```
Line Number   Column Ruler
              ----+----1----+----2----+----3----+----4----+----5----+----6----+----7----+----8
                 Keyword      fld1      fld2      fld3      fld4      fld5      fld6      fld7
1   ******************************************************************************
2   *   Determine Small Sapling Shade Tolerance Multiplier                         *
3   ******************************************************************************
4   Compute            0
5   _Shd1=LinInt(_RgnSt,  . . .
6   _Shd2=LinInt(_RgnSt,  0,    1,    1,   11, &
7                            11,   12,   12,   13,   13,   21, &
8                            21,   22,   22,   23,   23,   31, &
9                            31,   32,   32,   33,   33,   41, &
10                           41,   42,   42,   43,   43,   51, &
11                           51,   52,   52,   53,   53,  100, &
12                      100,  999,   &
13                        1,    1, 2.50, 2.50, &
14                           1.75, 1.75, 1.50, 1.50, 1.00, 1.00, &
15                           1.25, 1.25, 1.00, 1.00, 0.75, 0.75, &
16                           0.10, 0.10, 0.25, 0.25, 0.35, 0.35, &
17                           0.05, 0.05, 0.01, 0.01, 0.00, 0.00, &
18                           0.01, 0.01, 0.00, 0.00, 0.00, 0.00, &
19                        0,    0)
20  _Shd3=LinInt(_RgnSt,  . . .
21  _Shd4=LinInt(_RgnSt,  . . .
22  _Shd5=LinInt(_RgnSt,  . . .
23  End
```

* Shade Tolerance Rating:
_Shd1 = Very Intolerant
_Shd2 = Intolerant
_Shd3 = Intermediate
_Shd4 = Tolerant
_Shd5 = Very Tolerant

Figure 5—FVS keyword addfile segment that determines a multiplier relative to size/density class and associated overstory shade tolerance for small saplings.

and Southwestern Region National Forests). Tree species were sorted by five shade tolerance ratings: very intolerant, intolerant, intermediate, tolerant, and very tolerant. Within these groupings, small sapling frequencies were averaged in accordance with the size/density classification matrix. All cells were then divided by the index cell in order to scale the shade tolerance matrix relative to the composite distribution matrix. In this manner, the two matrices could be joined (i.e. multiplied) to account for regeneration quantity and quality.

Table 6 provides an example for ponderosa pine small saplings within the Mixed Conifer—Dry strata. On average, 26.34 trees per acre were observed on all FIA plots. This value is distributed by the shade tolerance multipliers displayed in the upper portion of table 6 to render the expected ponderosa pine small sapling frequency as presented in the bottom portion of table 6.

Table 6—Mixed conifer—dry: Ponderosa pine small saplings expected abundance.

Canopy cover class	Overstory QMD size class (inches)				
	0-5	5-10	10-15	15-20	> 20
(%)	Small sapling shade intolerant multipliers[a]				
10-40	1.75	1.25	0.10	0.05	0.01
40-70	1.50	1.00	0.25	0.01	0.00
> 70	1.00	0.75	0.35	0.00	0.00
(%)	Expected small sapling frequency (trees/acre)[b]				
10-40	46.09	32.92	2.63	1.32	0.26
40-70	39.51	26.34	6.59	0.26	0.00
> 70	26.34	19.75	9.22	0.00	0.00

[a]Comparative analysis using observed small sapling data from throughout the Interior West. Index cell was divided into all other size/density class cells to compute the multiplier.

[b]Observed ponderosa pine small sapling frequency dispersed from Index Cell to all other size/density cells based on shade intolerant multiplier. Values indicate expected small sapling frequency abundance.

Shade Tolerance Effects—Seedlings

Five phases of stand development based on relative stand density index have been postulated by Long (1985). They are:

- ***0-25% of SDI^{max}***—A stand or group of trees at this density level is open-grown. There is little competition for moisture, nutrients, and sunlight.
- ***25-35% of SDI^{max}***—A stand begins to experience tree-to-tree competition for moisture and nutrients. Less sunlight reaches the ground but site productivity is still increasing.
- ***35-60% of SDI^{max}***—This is the range in which full site occupancy has been reached. Tree-to-tree competition has resulted in individual trees sacrificing growth. Site productivity is still increasing but has begun to plateau.
- ***60-100% of SDI^{max}***—At this stage, tree-to-tree competition is so great that individual trees can no longer receive enough moisture, nutrients, and sunlight to survive. Self-thinning occurs but openings created by trees dying are quickly filled by the growth of surviving trees.

- ***100%+ of SDI^{max}***—Full site occupancy and growth are lost as overstory trees become weakened and die. As mature trees die, openings are created for more shade tolerant tree species if the site is capable of supporting those species.

By default within FVS, background mortality occurs when the current stand SDI is below 55 percent of maximum SDI. Density related mortality begins when the relative stand density is above 55 percent of maximum SDI. Stand density peaks at 85 percent of maximum SDI (Dixon 2002). These thresholds were incorporated into the survivability logic for seedling recruitment for three realms of shade tolerance in the following manner:

- Intolerant: 25-55% of SDI^{max}
- Mid-tolerant: 35-85% of SDI^{max}
- Tolerant: 55-100% of SDI^{max}

Within the specified range for a shade tolerance realm, input of natural regeneration for seedlings is tapered from 100 to 0 percent of the target value.

Natural Regeneration Estimates

Repute selects tree counts and average height by species for the seedling (< 1-inch) and small sapling (2-inch) diameter classes from the Fvsstand Alone 'stand tables' to create separate tree species regeneration keywords (figs. 6 and 7). On a twenty-year basis, the natural regeneration estimation process checks whether conditions warrant a pulse of seedling/sapling reproduction. Target

```
Line Number   Column Ruler
              ----+----1----+----2----+----3----+----4----+----5----+----6----+----7----+----8
                 Keyword      fld1      fld2      fld3      fld4      fld5      fld6      fld7
1             ******************************************************************************
2             *   Seedling Establishment Values                                            *
3             ******************************************************************************
4             If                 0
5             SpMcDBH(1,ALL,0,0,1,0,999,0) LE _SEED AND &
6             SpMcDBH(11,ALL,0,1.0,200,1.0,500,0) LE BSDIMAX*0.85 AND &
7             (_RGN EQ 1 OR _RgnSt LE 1) AND _RgnSt NE 12 AND _CANOPY LT 3 AND EVPHASE EQ 2
8             Then
9             Estab
10            Natural            0    Parms(WF, . . .
11            Natural            0    Parms(PP, &
12            Max(((40.690-SpMcDBH(1,PP,0,0,1,0,999,0)),0)* &
13            LinInt(SpMcDBH(11,ALL,0,0.2,200,1,500,0),BSDIMAX*0.25,BSDIMAX*0.55,1,0)* &
14            LININT(SpMcDBH(2,PP,0,3.0,99,0,999,0)/Max(SpMcDBH(2,All,0,3.0,99,0,999,0),1), &
15            0.00,0.02,0.02,0.05,0.05,1.99,0.40,0.40,0.80,0.80,1.00,1.00), &
16             100.00, 0.00,   1.0, 0)
17            Natural            0    Parms(DF, . . .
18            Natural            0    Parms(AS, . . .
19            End
20            Compute            0
21            _Rgn=0
22            End
23            Compute           20
24            _Rgn=1
25            End
26            EndIf
27            Compute            1
28            _SEED=858
29            End
```

Figure 6—FVS keyword addfile segment that establishes seedling reproduction on a 20-year basis. Values by species (PP: Ponderosa Pine) for tree count (40.690 trees/ac) and average height (1.0 ft) are extracted from Fvsstand Alone 'stand tables.'

```
Line Number   Column Ruler
              ----+----1----+----2----+----3----+----4----+----5----+----6----+----7----+----8
                 Keyword      fld1      fld2      fld3      fld4      fld5      fld6      fld7
1   ********************************************************************************
2   *   Small Sapling Ingrowth Values                                                *
3   ********************************************************************************
4   If                 0
5   SpMcDBH(1,ALL,0,1,3,0,999,0) LE _SAPS*_SAPP AND &
6   SpMcDBH(11,ALL,0,1.0,200,1.0,500,0) LE BSDIMAX*0.85 AND &
7   (_RGN EQ 1 OR _RgnSt LE 1) AND _RgnSt NE 12 AND _CANOPY LT 3 AND EVPHASE EQ 2
8   Then
9   Estab
10  Natural            0    Parms(WF, . . .
11  Natural            0    Parms(PP, &
12  Max(((26.339*_SAPS/_SAP2*_SAPP*_SAPF*_SHD2)-SpMcDBH(1,PP,0,1,3,0,999,0)),0)* &
13  LININT(SpMcDBH(2,PP,0,3.0,99,0,999,0)/Max(SpMcDBH(2,All,0,3.0,99,0,999,0),1), &
14  0.00,0.02,0.02,0.05,0.05,1.99,0.40,0.40,0.80,0.80,1.00,1.00), &
15   100.00, 0.00,  12.1, 0)
16  Natural            0    Parms(DF, . . .
17  Natural            0    Parms(AS, . . .
18  End
19  Compute            0
20  _Rgn=0
21  _SAPF=((_SAPS*_SAPP)/( &
22  +27.521*_SAPS/_SAP2*_SAPP*_SHD4 &
23  +26.339*_SAPS/_SAP2*_SAPP*_SHD2 &
24  +40.353*_SAPS/_SAP2*_SAPP*_SHD3 &
25  +22.287*_SAPS/_SAP2*_SAPP*_SHD1 & . . .
26  End
27  Compute           20
28  _Rgn=1
29  End
30  EndIf
31  Compute            1
32  _SAP2=312
33  End
```

* Example: Ponderosa Pine Small Sapling Target for Size/Density Class 32
Value = 26.339 (observed small saplings, ponderosa pine, all size/density classes)
_Saps = 417 (observed small saplings, all species, size/density class 22)
_Sap2 = 312 (observed small saplings, all species, all size/density classes)
_Sapp = 0.3798 (composite distribution factor for size/density class 32)
_Sapf = 2.3220 (shade tolerance factor, all shade tolerance classes)
_Shd2 = 0.25 (shade intolerant multiplier)

Target = 26.339*_SAPS/_SAP2*_SAPP*_SAPF*_SHD2
Target = 26.339*417/312*0.3798*2.3220*0.25 = 7.76 small saplings per acre

Figure 7—FVS keyword addfile segment that ensure small sapling regeneration on a 20-year basis.

thresholds are based on the classified vegetation state. If existing frequencies are less than specified thresholds, natural regeneration is invoked. Checks are included that account for existing seedlings and small saplings so as not to exceed expected targets. Also, a test is performed to inquire whether enough mature trees of a given species are present to produce an adequate seed source. For a more in-depth discussion of the Repute program, a detailed Users Guide is available (Vandendriesche 2005).

Recall that seedlings are input as pawns for small sapling recruitment. The crux of the natural regeneration estimation process is insuring that a stable small sapling ingrowth component is available that will transition into larger diameter classes in subsequent projection cycles. An example is provided in figure 7. Input values and the resultant small sapling target for size/density class 32 are displayed. If existing small ponderosa pine saplings exceed the target value of 7.54, then

the "Natural" keyword operation is not performed. Conversely, if existing small ponderosa pine saplings are lacking, the target frequency of 7.54 trees per acre will be regenerated so long as there is a sufficient number of mature ponderosa pine capable of providing an adequate seed source.

Conclusions

In the absence of having the "full" regeneration establishment model available to provide natural reproduction into an FVS projection, users need to account for this important aspect of forest growth modeling. External regeneration imputation models have been developed and should be explored where available (Robinson 2007). Lacking these resources, the natural regeneration estimation procedure presented in this paper may serve to fill the void. The procedure presented describes the process of inputting a stable supply of small saplings into projection runs to procure a vital source of recruitment into larger diameter trees. Flushes of seedlings are also input to account for the smallest form of forest reproduction. The regeneration estimation method is well suited for natural growth simulations. It may also be suited following certain silvicultural treatments but may need augmentation based on professional experience.

Special situations may require creative solutions when running the Repute program. For example, in areas with intense deer browsing, it may be necessary to input larger diameter trees, greater than the small sapling size class, to represent the stable recruitment component. Having a representative supply of natural regeneration to input into a simulation run is vital.

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The content of this paper reflects the views of the authors, who are responsible for the facts and accuracy of the information presented herein.

Evaluating the Ecological Sustainability of a Pinyon-Juniper Grassland Ecosystem in Northern Arizona

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Abstract—In order to develop strategic land management plans, managers must assess current and future ecological conditions. Climate change has expanded the need to assess the sustainability of ecosystems and predict their conditions under different climate change and management scenarios using landscape dynamics simulation models. We present a methodology for developing a state-and-transition model (STM) with the Vegetation Dynamics Development Tool (VDDT), using outputs from the Forest Vegetation Simulator (FVS). Preside, a recently developed accessory to the FVS program, is used to process and report FVS outputs in terms of succession probabilities and residence times for each STM. We've applied these tools with a case study based on the pinyon-juniper grassland ecosystem in northern Arizona. After applying local probability values for natural growth, contemporary fire, insect and disease, and management activities, VDDT simulations were conducted to project future ecosystem conditions including carbon accounting. Finally, we also describe how these models can be retooled with FVS support to reflect the effects of climate change so that managers can consider adaptation and mitigation strategies.

Introduction

The objective of this paper is to illustrate through a case study how State-and-Transition Models (STMs) can be developed and used to evaluate the ecological sustainability of ecosystems.

Projecting transitions in vegetation states (composition and structure) over time facilitates evaluating the ecological sustainability of ecosystems. Vegetation states can change in "the absence of disturbance" through natural regeneration, growth, competition and mortality; change also can result from disturbances and other discrete events in time such as fire, management activity, insect and disease outbreaks, etc. To facilitate projecting the effects of the interactions of these agents of change, landscape STMs such as the Vegetation Dynamics Development Tool (VDDT) developed by ESSA Technologies Limited (2006) can be used to quantify the dynamics of vegetation change (He 2008).

Ecological sustainability analysis evaluates both ecosystems (*ecosystem diversity*) and their associated species (*species diversity*). A guiding principle for ecosystem management (FEMAT 1993) is to use ecosystem reference conditions, the range of variation, as an inference of ecological sustainability to enable the persistence of ecosystem function and species diversity. In this paper, we focus on vegetation diversity and related ecological processes such as fire, and apply a case study assessing the ecological sustainability of the *pinyon-juniper grassland* ecosystem on the Coconino National Forest (NF) in northern Arizona.

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Methods

Framing the Analysis

We stratified the Coconino NF by potential natural vegetation types (PNVTs) (Schussman and Smith 2006), a coarse ecosystem framework defined by site potential and historic fire regimes, that provides a basic framework for analyzing ecosystem diversity. Although the same process was used on each PNVT, this paper documents the analysis process conducted on the 122,086 hectare (301,675 acres) pinyon-juniper grassland PNVT.

The pinyon-juniper grassland type occurs across the States of Arizona and New Mexico, in what was historically open woodlands with grassy understories (Ffolliott and Gottfried 2002). On the Coconino NF tree species include twoneedle pinyon (*Pinus edulis* Engelm.), oneseed juniper (*Juniperus monosperma* (Engelm.) Sarg.), Utah juniper (*Juniperus osteosperma* (Torr.) Little), and alligator juniper (*Juniperus deppeana* Steud.). On reference sites, native understories are made up of predominantly cool season perennial grasses including muttongrass (*Poa fendleriana* (Steud.) Vasey), squirreltail (*Elymus elymoides* (Raf.) Swezey ssp. *brevifolius* (J.G. Sm.) Barkworth) and western wheatgrass (*Pascopyrum smithii* (Rydb.) A. Löve), with both annual and perennial forbs, while shrubs are absent or scarce (<1 percent cover)(Miller and others 1995). Contemporary understories often include invasive grasses such as cheatgrass (*Bromus tectorum* L.) and a dominance of warm season species such as blue grama (*Bouteloua gracilis* (Willd. ex Kunth) Lag. ex Griffiths), and have uncharacteristically high shrub cover. This pinyon-juniper woodland type is typically found on sites with well-developed and moderately deep soils with loam and clay loam surface textures. Soil orders include mollisols derived from basaltic parent materials, andisols formed from cinder deposits and alfisols developed from sedimentary sources. Climate is characterized by a seasonal distribution of precipitation of which over half occurs between the months of April through September, with an annual rainfall ranging from 15-18 inches.

Information on the historic condition of this type is sparse. The ability to reconstruct historic stand structure and fire chronologies in pinyon-juniper is problematic, so the role of fire and the resulting vegetation structure is often speculative (Jacobs 2008). However, site productivity suggests that the development of a grass and fine fuels layer would have supported frequent fire, open forest dynamics, and perhaps uneven-aged conditions (Gottfried 2003).

We described historic (reference), current, and future structural conditions according to standard classification schemes based on average tree size (diameter) and canopy cover class (Brohman and Bryant 2005; table 1). Due to disparities in historic and current condition references, and how they were developed, it was necessary to develop crosswalks to normalize across references and enable comparisons between historic and current conditions. For instance, the U.S. Department of Agriculture (USDA), Forest Service mid-scale mapping, used to depict current conditions (Mellin and others 2008), uses a canopy cover break of 30 percent to distinguish open and closed, versus the LANDFIRE model (Havlina 2005) that employs a 40 percent break. We portrayed historic, current, and future *composition* conditions according to a southwestern regional classification of existing vegetation based on dominance types (Triepke and others 2005). Dominance types, defined by the relative abundance and dominance of tree species, are similar to Society of American Foresters or Society for Range Management cover types (Eyre 1980; Shiflet 1994), but are keyable, exhaustive, and mutually exclusive.

Table 1—Crosswalk to facilitate comparison of historic, current, and future conditions of the pinyon-juniper grassland ecosystem.

Reference Condition LANDFIRE RA JUPI1 Model			Current Condition USDA FS R3 Mid-Scale EV Map		Future Trends USDA FS R3 PJ Grassland Model	
State	**Mean[a]**	**Description**	**Dominance Unit**	**Structure[b]**	**State**	**Description**
A	5%	Post replacement	Non-tree: Recently burned and all shrub, grass, and forb types	Grass-forb-shrub	A	GFB/SHR
				Seed/sap-open	B	SSO
C	25%	Mid-open	—	Seed/sap-closed	E	SSC
			All pinyon, juniper, and mixed shade intolerant tree dominance types occurring within the pinyon-juniper grassland PNVT	Small-open	C	SMO
D	50%	Late-open		Med to very large-open	D	MVO
B	10%	Mid development closed		Small-closed	F	SMC
E	10%	Late-closed	—	Medium to very large-closed	G	MVC

[a] Average proportion of the landscape during the reference period (circa 880-1880 (Schussman and Smith 2006)).

[b] Size classes based on diameter at breast height for forest tree species and diameter at root collar for woodland species: seedling/sapling (< 13cm), small (13–24.9cm), medium (25–50cm), and very large (>50cm): overstory cover classes are sparse (<10% tree canopy cover), open (10 – 29.9% cover), and closed (>29.9% cover).

Specific combinations of dominance type, size class, and canopy cover that are characteristic to each PNVT are expressed in terms of *vegetation states* identified for each PNVT, and configured in PRESIDE (Process RESIDEnce Times), a recently developed ancillary program (Vandendriesche 2009) to the Forest Vegetation Simulator (FVS) model (Dixon 2002). Each vegetative state represents an important phase in the ecosystem dynamics of a PNVT. The historic pinyon-juniper grassland ecosystem has been described (LANDFIRE 2007) as a five-state model that includes a grass-forb state (A), two open forest states (C and D), and two closed forest states (B and E) (table 1). Frequent surface fires maintained the forest in these reference conditions. *Ecological process* reflects the ability of natural and anthropogenic events such as fire, forest insect and disease, and resource management activities to alter vegetation composition and structure and, in turn, wildlife habitat and species diversity (Perry and Amaranthus 1997). Along with site potential, the characteristic frequency and severity of fire are differentia of the PNVT classes themselves.

Describing Reference Conditions

The Vegetation Dynamic Development Tool (VDDT, ESSA 2006) has been used by the National LANDFIRE program (Ryan and others 2006) and others such as the Nature Conservancy (TNC) (Schussman and Smith 2006) to develop state-and-transition models that describe reference conditions. The VDDT software moves cells (representing a unit of area) from one state to another based on a set of "transitions." Traditionally, "deterministic transitions" describe succession (aging) in the absence of disturbance. Probabilistic transitions reflect the quantitative assessments of discrete natural and anthropogenic events including fire, insects, diseases, grazing, harvesting, and severe weather events. Each probabilistic transition typically has three characteristics that define its pathway: 1) its return frequency or probability, 2) its severity or impact on vegetation, and 3) the destination state in which the cell will reside after transition.

We retooled these models to project future conditions by replacing historic probabilities and transitions with contemporary transitions and attendant frequencies that reflect current land management. We also added contemporary and possible future vegetative states. We detail the development of the pinyon-juniper grassland model below. Reference condition descriptions and models typically are based on peer-reviewed journal articles as well as published conference proceedings, reports, theses, dissertations, and book chapters along with some consideration of professional judgment provided by model developers. In contrast, the models that we developed for projecting future conditions were more empirical, using Forest Inventory and Analysis (FIA) data, FVS simulation runs, and related software tools.

Describing Current Conditions

We mapped PNVTs using Terrestrial Ecosystem Survey (TES) data for the Coconino NF (Miller and others 1995). The TES is a terrestrial ecological unit inventory that formulates map units based on similarities in climate, soils, landform, and potential vegetation at the map scale of 1:24,000 (Winthers and others 2005). Among the map unit attributes, disclimax classes (zootic, fire) indicate historic disturbance regime, making TES map data the best available resource for PNVT mapping.

In 2004, the Southwestern Region initiated mid-scale mapping of existing vegetation at 1:100,000 across all National Forests and Grasslands (Mellin and others 2008). This mapping includes the three principle existing vegetation map components previously mentioned—dominance type, size class, canopy cover. With the description of vegetation states (table 1), these map data allowed for the quantitative analysis of current conditions within each PNVT. We intersected PNVT mapping in GIS with the existing dominance type, size class, and canopy cover layers from mid-scale vegetation mapping products to produce tabular summaries of current conditions within each PNVT class. These summaries were in turn synthesized to give hectares and percent of each vegetation state within each PNVT. We then compared these percents to historic and projected conditions for the ecosystem.

Along with each condition reported (historic, current, or projected), we calculated *ecosystem condition class* values using the same equation employed by LANDFIRE to compute Fire Regime Condition Class (FRCC) (Hann and others 2005). But unlike FRCC, which provides percentages for each departure class (1, 2, or 3), our own ecosystem condition class (ECC) provides one overall departure rating for a given analysis area (Weisz and others 2009). The ECC is computed for each comparison, either current vs. reference condition, or projected vs. reference condition (table 2) based on the departure of all states in total from their reference conditions. In each calculation, the sum of the lesser of percent values for each state, either reference or current, is subtracted from 100 to provide one overall departure index on a scale of 0 percent to 100 percent, higher values representing more departed conditions. From there, three classes make up the ECC rating system:

- ECC 1 (within reference condition) represents departure index values ≤ 33;
- ECC 2 (moderately departed) represents departure index values >33 and ≤ 66; and
- ECC 3 (severely departed) represents departure index values > 66.

Recently developed FRCC map data for LANDFIRE map zones in Arizona (LANDFIRE 2008) corroborate our findings, as do regional studies of these systems.

Table 2—Calculation of departure and ecosystem condition class based on the disparity between reference and current conditions for the PJ grassland ecosystem on the Coconino NF.

	Reference condition		Current condition	
State	Description	Mean[a]	Proportion	Calculation[b]
A	Post replacement	5%	23%	5%
C	Mid-open	25%	14%	14%
D	Late-open	50%	27%	27%
B	Mid development closed	10%	17%	10%
E	Late-closed	10%	19%	10%
			Sum	66%
			Departure index value = 100% - Sum	34
			Ecosystem condition class 1 (0–33), 2 (34–65), 3 (66+)	"2"

[a] Average proportion of the landscape during the reference period (circa 880-1880 (Schussman and Smith 2006)).

[b] Lesser of reference condition and current condition.

Projecting Future Conditions

Retooling models—Typically reference conditions models are based on a survey of the literature, supplemented by empirical data as well as expert opinion (LANDFIRE 2008). Often these models are applicable to a large map zone or to a large region like Arizona and New Mexico (Gori and Bate 2007; Havlina 2005). To retool these models to project conditions under existing or proposed management schemes, managers can modify reference condition models to: 1) include new states or modified states that reflect vegetation classes that did not exist under reference conditions; 2) incorporate current and projected natural and anthropogenic processes; and 3) incorporate current and projected transition probabilities. We illustrate by example how the Coconino NF retooled the pinyon-juniper grassland reference condition model for this purpose (see below), with the assumption of no climate change. Carbon accounting was also provided using the carbon extension of FVS (Havis and Crookston 2008; Hoover and Rebain 2008). The carbon extension provides values for dead and live standing trees, and dead and live belowground tree tissue (Hoover and Rebain 2008). Standard values are provided for carbon held in herbs and shrubs, downed wood, and litter and duff, based on similarly measured plant communities. The paper concludes with a description of how the model could be retooled in the future to consider climate change.

New or modified states—Typically, models for current and projected conditions contain as many or more states than reference condition models. In the case of the pinyon-juniper grassland PNVT, we developed a seven-state model to describe current conditions in contrast to the reference conditions model that had five states. Table 1 illustrates how we cross walked the reference conditions states with the current states.

Quantifying current transitions—To retool reference condition models to reflect contemporary processes, four steps are followed: 1) identify the contemporary transitions; 2) replace reference transitions with contemporary ones (tables 3, 4a and 4b); 3) model future conditions; and, 4) interpret the results. Each contemporary transition is identified in terms of its type, transition class (groups

Table 3—Canopy cover and fire mortality proportion table.

Canopy Cover / Fire Mortality Proportion Table		
Beginning Canopy Cover Class	Fire Severity Class	Ending Percentage by Canopy Cover Classes
10 – 30% (open)	non-lethal	9% → sparse (0 – 10%)
		91% → open (10 – 30%)
	mixed severity	55% → sparse (0 – 10%)
		45% → open (10 – 30%)
	stand replacement	100% → sparse (0 – 10%)
30 – 60% (closed)	non-lethal	16% → open (10 – 30%)
		84% → closed (30 – 60%)
	mixed severity	2% → sparse (0 – 10%)
		79% → open (10 – 30%)
		19% → closed (30 – 60%)
	stand replacement	87% → sparse (0 – 10%)
		13% → open (10 – 30%)

of transition types), frequency, and effects. We used four transition classes in our current model: wildland fire, management activities, insect and disease, and natural growth transitions in the absence of disturbance. Transition types within each transition class may have unique frequencies and effects unto themselves. The management activities transition class contains, for example, mechanical thinning, prescribed burning, etc.

Wildland fire transitions—We used LANDFIRE definitions of fire severity based on how much overstory canopy mortality would occur during a wildland fire: nonlethal (or low severity), <25 percent mortality; mixed severity fire, 25 percent to 75 percent mortality; and stand replacement fire, >75 percent mortality (Hann and others 2005). We generated fire frequencies for each of the transition classes using local fire history data on the planning unit for the period 1988 through 2006. Spatial data was available for approximately five wildland fires greater than 40 hectares (16 acres) in size for the period 1960 to 2005. Fire mortality mapping was available for three incidents including the Lizard (2003), Mormon (2003), and Jacket (2004) fires. For other fires that occurred after 1975, fire officials provided estimates of the percentage of non-lethal, mixed severity, and stand replacement fire that occurred. We corroborated fire mortality for the fires using orthophotos in GIS, estimating fire extent and mortality based on patterns of top-kill and regeneration.

We summarized these results as average annual probabilities per hectare for each fire type: nonlethal fire (0.0002), mixed severity fire (0.0021), and stand replacing fire (0.0032) and assigned these probabilities to each model state (table 4a). The mixed severity and stand replacement probabilities can be attributed to significant Pinyon Ips bark beetle activity since 1996. The effects of a fire on a cell within the model depend on pre-fire canopy cover and the severity of the fire (the fire mortality class; table 3).

Management activity transitions—We quantified management activities using the Forest Activity Tracking System (FACTS) database (M. Pitts, unpublished data). We queried all activities recorded on the planning unit from 1988 through 2006, and then eliminated activities that did not affect broad-scale vegetation

Table 4a—Pinyon-juniper grassland natural and anthropogenic disturbance transitions expressed as the average annual probability per hectare per year.

From State Code: Acronym: Description: Transition Type	Probability	Proportion[a]	To State Acronym
A: GFB: Grass/Forb/Brush			
Nonlethal Fire	0.0002	1.00	GFB
Stand Replacing Fire	0.0032	1.00	GFB
B: SSO: Seed/Sap, Open			
All Regeneration	0.0011	1.00	GFB
Insect and Disease	0.0100	1.00	SSO
Mixed Severity Fire	0.0021	0.55	GFB
Mixed Severity Fire	0.0021	0.45	SSO
Nonlethal Fire	0.0002	0.09	GFB
Nonlethal Fire	0.0002	0.91	SSO
Stand Replacing Fire	0.0032	1.00	GFB
C: SMO: Small, Open			
All Regeneration	0.0011	1.00	GFB
Insect and Disease	0.0100	1.00	SMO
Mixed Severity Fire	0.0021	0.55	GFB
Mixed Severity Fire	0.0021	0.45	SMO
Nonlethal Fire	0.0002	0.09	GFB
Nonlethal Fire	0.0002	0.91	SMO
Stand Replacing Fire	0.0032	1.00	GFB
D: MVO: Medium to Very Large Open			
All Regeneration	0.0011	1.00	GFB
Insect and Disease	0.0100	1.00	MVO
Mixed Severity Fire	0.0021	0.55	GFB
Mixed Severity Fire	0.0021	0.45	MVO
Nonlethal Fire	0.0002	0.09	GFB
Nonlethal Fire	0.0002	0.91	MVO
Stand Replacing Fire	0.0032	1.00	GFB
E: SSC: Seed/Sap Closed			
All Regeneration	0.0011	1.00	GFB
Insect and Disease	0.0100	1.00	SSO
Mixed Severity Fire	0.0021	0.02	GFB
Mixed Severity Fire	0.0021	0.19	SSC
Mixed Severity Fire	0.0021	0.79	SSO
Nonlethal Fire	0.0002	0.84	SSC
Nonlethal Fire	0.0002	0.16	SSO
Stand Replacing Fire	0.0032	0.87	GFB
Stand Replacing Fire	0.0032	0.13	SSO
F: SMC: Small, Closed			
All Regeneration	0.0011	1.00	GFB
Insect and Disease	0.0100	1.00	SMO
Mixed Severity Fire	0.0021	0.02	GFB
Mixed Severity Fire	0.0021	0.19	SMC
Mixed Severity Fire	0.0021	0.79	SMO
Nonlethal Fire	0.0002	0.84	SMC
Nonlethal Fire	0.0002	0.16	SMO
Stand Replacing Fire	0.0032	0.87	GFB
Stand Replacing Fire	0.0032	0.13	SMO
G: MVC: Medium to Very Large Closed			
All Regeneration	0.0011	1.00	GFB
Insect and Disease	0.0100	1.00	MVO
Mixed Severity Fire	0.0021	0.02	GFB
Mixed Severity Fire	0.0021	0.19	MVC
Mixed Severity Fire	0.0021	0.79	MVO
Nonlethal Fire	0.0002	0.84	MVC
Nonlethal Fire	0.0002	0.16	MVO
Stand Replacing Fire	0.0032	0.87	GFB
Stand Replacing Fire	0.0032	0.13	MVO

[a] Proportion of acres affected by a transition that will move to the destination state.

Table 4b—Pinyon-juniper grassland natural growth in the absence of disturbance successional transitions expressed as the average annual probability per hectare per year.

From State Code: Acronym: Description	Probability	To State Acronym
A: GFB: Grass/Forb/Brush	.9691	GFB
	.0136	SSO
	.0041	SMO
	.0132	MVO
B: SSO: Seed/Sap, Open	.9249	SSO
	.0269	SMO
	.0247	SSC
	.0236	SMC
C: SMO: Small, Open	.0045	SSO
	.9175	SMO
	.0193	MVO
	.0024	SSC
	.0494	SMC
	.0070	MVC
D: MVO: Medium to Very Large Open	.0078	SSO
	.0036	SMO
	.9714	MVO
	.0014	SSC
	.0016	SMC
	.0142	MVC
E: SSC: Seed/Sap Closed	.9093	SSC
	.0907	SMC
F: SMC: Small, Closed	.0004	SSC
	.9759	SMC
	.0237	MVC
G: MVC: Medium to Very Large Closed	.0003	MVO
	.0002	SSC
	.0036	SMC
	.9960	MVC

composition and structure from further analysis; thus, we eliminated wildlife inventories, mine reclamation activities, etc. We summarized the remaining 8,747 management activities into standardized transition classes such as prescribed burning, fuels treatment, and harvest thinning. As with the wildland fire transitions, we calculated average annual probability-per-hectare values for each PNVT and assigned these probabilities to each model state (table 4a). In a typical year during the sampled time period, 0.1 percent of the pinyon-juniper grassland PNVT was affected by these activities.

Insect and disease transitions—Both localized and widespread mortality events have occurred in the pinyon-juniper woodlands on the Coconino NF (Lynch and others 2007). These events have typically been pinyon ips outbreaks associated with periods of drought, such as occurred in the 1950s, and more recently in the mid-1990s and 2001-2003. Localized outbreaks resulted from range improvement projects that generated large amounts of fresh pinyon slash (Negrón and Wilson 2003; Yasinski and Pierce 1962). Although pinyon ips outbreaks can be severe, with pinyon mortality approaching 100 percent within a given stand, they are generally short lived (1-2 years). The pinyon ips outbreak during the late-1990s east of Flagstaff near Twin Arrows encompassed almost 5,261 hectares (13,000 acres) at its peak (Negrón and Wilson 2003).

At least within the historic period, the size and severity of the recent drought- and pinyon ips-related die-off is unprecedented for the Coconino NF and northern Arizona (Allen 2007; Mueller and others 2005). The contemporary pinyon die-off is 100 times as large (two orders of magnitude) as any previously recorded acreage for pinyon ips beetle mortality for the Coconino NF, Kaibab NF, and Grand Canyon National Park. High levels of pinyon mortality were detected by aerial survey during 2001 through 2003, with approximately 809,389 hectares (2,000,000 acres) impacted Region-wide and more than 60,704 hectares (150,000 acres) on the Coconino NF. The mortality was primarily attributed to pinyon ips attacking drought-stressed pinyon; however, twig beetles (*Pityophthorus* spp.) were also observed killing smaller pinyon in 2003. Pinyon mortality averaged 41.4 percent within an 80-km (50-mile) radius of Flagstaff, with mortality being significantly greater on southern aspects and shallow soils developed in volcanic cinders (Gitlin and others 2006).

Using data from the above insect and disease outbreaks, we calculated average annual probability-per-hectare values for each model state. These transition probabilities were assigned to each model state (table 4a).

Natural growth transitions in the absence of disturbance—To quantify the natural growth transitions that will occur in the absence of natural and anthropogenic disturbances, we used the PRESIDE software (Vandendriesche 2009) to process the outputs of FVS (Dixon 2002). In our case study, we show the results of applying this methodology in the Southwestern Region. The steps in this process include:

1. Prepare the FIA inventory data for projection by FVS: Each FIA plot for the PNVT in the Southwestern Region is assigned to the appropriate model state.
2. Perform FVS calibration steps: Calibration procedures include using the FVS self-calibrating feature, estimating and inputting natural regeneration response, accounting for tree defect for volume estimates, and determining tree species size attainment and limiting stand maximum density.
3. Run natural growth projections for each FIA plot using the calibrated FVS to simulate growth over a 250-year time period.
4. Process the tree list output through the PRESIDE post-processor classifier and accumulate the results into a matrix from which to estimate the average annual probability per hectare that in the absence of disturbance a plot will transition from one state to another state (table 4b).
5. Using the sample of plots populating each state at each point in time during the projection, summarize the vegetation characteristics of each model state (table 5). The post-processing software indexes the aggregate state classes to summary values derived from the tree lists and attributes from standard FVS outputs. Several dozen vegetation characteristics such as stand volume and stand carbon can be quantified for each model state.

Model runs—Model simulations from VDDT are non-spatial and reflect a summary of up to 50,000 sample units or *cells*. For our study, we opted for 1,000 sample units because our earlier work, and work conducted by TNC and LANDFIRE, indicated that this number produced reasonable and consistent projections (TNC and others 2006). If we increased the number of cells beyond 1000, results of the analysis would not be significantly changed, but running time would be increased significantly. In the next step of the modeling process, we initialized the starting hectares in each state based on current conditions indicated by mid-scale vegetation mapping data. We ran multiple simulations to estimate the long-term

Table 5—FVS Outputs.

Vegetation Structure Variables:							
VDDT State	**A _ GFB**	**B _ SSO**	**C _ SMO**	**D _ MVO**	**E _ SSC**	**F _ SMC**	**G _ MVC**
Dominance Type	PIED	PIED	PIED	PIED	PIED	PIED	PIED
Size Class	0	1	2	3	1	2	3
Canopy Class	0	1	1	1	2	2	2
Canopy Layers	1	2	1	1	1	1	1
Stand Age – Overstory	17	76	98	118	97	146	207
Stand Age – Dominant Story	0	55	92	130	68	121	196
Total Plot/Activity Count	323	277	317	1222	194	3084	9895
Stand-Stock Variables:							
Seedlings/Acre < 1.0” diameter	61	148	117	103	265	159	89
Trees/Acre = 1.0” diameter	206	441	395	286	777	493	315
Basal Area/Acre = 1.0” diameter	11	37	44	57	84	117	135
Quadratic Mean Diameter – Trees = 1.0” diameter	3	4	5	6	5	7	9
Quadratic Mean Diameter – Top 20 percent, diameter	0	7	9	14	8	11	15
Stand Density Index – SDI_Summation	20	90	94	99	195	233	235
Stand Density Index – SDI_Dq	30	106	111	128	221	260	263
Canopy Cover	6	21	22	21	40	48	48
Live – Cubic Feet/Acre	88	233	328	568	515	1176	1741
Live – Board Feet/Acre	0	5	12	16	30	18	11
Wildlife Habitat Variables:							
R3 – Vegetative Structural Stage	1	3ASS	4ASS	5ASS	1C	4CSS	5CSS
Standing Snags							
Small = 5-10” diameter	1.1	0.6	2.8	2.4	3.4	26.4	14.0
Large = 10”+ diameter	1.6	2.7	2.9	5.9	4.3	6.2	12.1
Extra-large = 18”+ diameter	0.7	1.2	1.3	2.8	1.5	2.0	2.6
Snag Recruitment							
Small = 5-10” diameter	0.2	0.4	2.0	0.9	4.9	26.0	11.4
Large = 10”+ diameter	0.5	1.3	1.0	2.5	2.4	3.5	7.5
Extra-large = 18”+ diameter	0.2	0.4	0.4	1.0	0.6	0.7	0.9
Pestilent Disturbance Variables:							
Dwarf Mistletoe Rating	0.04	0.09	0.14	0.15	0.22	0.34	0.41
Wildfire Risk Variables:							
Crown Bulk Density	0.00	0.02	0.02	0.01	0.04	0.04	0.03
Crown Base Height	4.5	4.7	6.2	7.6	4.9	5.7	7.5
Crowning Index	170.5	72.4	85.7	121.8	52.3	49.0	61.3
Torching Index	5.0	3.2	6.4	8.0	3.1	5.8	10.1
Fire Hazard Rating	3.0	3.0	3.0	3.0	3.0	3.0	3.0
Fuel Load – Coarse Woody Debris = 0-3” diameter	0.1	0.4	0.5	0.6	1.2	1.7	1.5
Fuel Load – Coarse Woody Debris = 3-12” diameter	0.2	0.7	1.3	2.1	1.9	5.1	9.4
Fuel Load – Coarse Woody Debris = 12”+ diameter	0.2	0.8	1.3	2.0	1.2	3.1	5.2
Biomass-Carbon Variables:							
Tree Biomass – Dry weight live & dead/boles & crown	2.4	7.6	8.6	14.3	16.7	27.8	36.4
Stand Carbon – Total carbon above & below ground	2.8	7.2	8.5	12.4	14.8	24.9	32.6

effects of continuing current management under the existing land management plan. We ran ten simulations with each simulation projecting conditions annually for 200 years based on data and assumptions described earlier. We compared the average annual results of these simulations with current conditions and reference conditions (table 6)

Table 6—Proportion of area in pinyon-juniper grassland states in current and projected conditions.

Vegetation state	A	B	C	D	E	F	G	Percent departure	ECC
Current condition	23	1	13	27	0	17	19	34	2
Projected trends									
10 years	20	4	9	25	1	18	23	36	2
50 years	15	6	6	22	1	18	32	40	2
200 years	13	4	5	22	2	17	37	42	2

Results

Here we provide results to answer the question, "How do current and projected conditions compare to reference conditions?" Again, reference conditions were derived from VDDT models, developed by LANDFIRE, to quantify the historical proportion of major vegetation states of the pinyon-juniper grassland system (table 2). Current conditions and projected conditions are summarized in table 6.

Current conditions represent existing vegetation mapping synthesized according to the vegetation states contained in the pinyon-juniper grassland model. An ECC value of 34 indicates a system that is somewhat departed from reference conditions, on the low end of the moderate range. Current conditions indicate an uncharacteristic excess of grass-forbs communities (state A), an excess of closed woodlands (states B and E), and a reciprocal deficit of open woodlands (states C and D).

Likewise future projections indicate a continuing trend towards departure, from an index value of 36 at 10 years, to 40 at 50 years. If current management continues, departure from reference conditions as measured by Ecosystem Condition Class will increase over time due to more acres moving to the closed states.

These results relate to the ecological sustainability concepts stated in our introduction and restated more simply here: Every species around today persisted over time in its environment under reference conditions. If current and proposed future management creates or approximates that environment, then the species is not likely or is less likely to be at risk. On the other hand, as in this case study, if the ecosystem is departed from reference conditions, and if that departure increases over time, then both the ecosystem and its associated species are less likely to be sustainable.

Discussion

The Analysis Process

As mentioned, TNC, LANDFIRE, and others have made a significant investment in the development of reference condition descriptions and models. We've complimented these models with the development of calibrated and more detailed models that depict current trends and project future conditions. Current and future conditions can be compared with reference conditions to answer two questions: 1) is there a current departure from reference conditions, and 2) will conditions remain static, trend towards, or trend away from reference conditions? Trends away from reference condition may indicate an ecosystem at risk and, if so, the model can be further tooled to evaluate the potential effectiveness of management strategies.

Also, as discussed under Methods, in retooling the models, several assumptions were necessary in the face of uncertainties concerning the historic condition. While additional information is needed to supplement and refine concepts for the pinyon-juniper grassland PNVT, working assumptions on fire frequency and stand dynamics were necessary to enable useful modeling of the system. For example, we assumed that a plurality of tree diameters existed to indicate one of four tree-dominated states, acknowledging that multiple tree cohorts within any one plant community were likely due to fire frequency and productivity.

Evaluation of Results

Southwestern pinyon-juniper woodlands span a wide range of environmental settings over 8.6 million hectares (3.5 million acres), yet historical descriptions are extremely limited. The pinyon-juniper grassland type is thought to have been maintained historically by frequent, low-severity surface fires that spread from and into adjacent systems including semi-desert grassland, juniper grassland, and ponderosa pine forest. Some references (e.g., Gottfried 2003) suggest that the resulting stand structure would have been uneven-aged, dominated by open grown trees. Modern fire suppression and grazing would have since favored closed canopy structures susceptible to drought-insect induced mortality and uncharacteristic fire (stand replacement). The current surplus of grass-forbs communities has likely resulted from stand clearing and pasture development, and from increased drought-insect mortality and fire activity. Long term VDDT modeling based on current practices, as reflected in management records from 1988 to 2006, indicates the perpetuation of dense canopies with regular conversion to a grass-forbs state.

The objective of this paper is to illustrate through a case study how State-and-Transition Models (STMs) can be developed and used to evaluate the ecological sustainability of ecosystems. We accomplished this objective by using an empirical approach to create and calibrate our models based on existing inventory data and FVS simulations based on existing data; this allowed us to compare and contrast reference conditions, existing conditions and projected conditions to quantify the departure of existing and projected conditions from reference conditions.

Our analysis indicates that the pinyon-juniper grassland ecosystem on the Coconino NF is moderately departed from reference conditions and that this trend will continue into the future under the existing land management plan. Fire suppression coupled with infrequent forest management activities contributes to an existing departure from reference conditions. Thus, the continued current implementation of the existing land management plan may pose a risk to the ecological sustainability of this ecosystem.

Others such as Arno and Fiedler (2005) have explored deteriorated forest and woodland conditions in western North America and reached similar conclusions. By developing empirically based landscape dynamics models, we can quantify woodland conditions with more reliability to assess the ecological sustainability of these ecosystems within a more credible, systematic framework for strategic land management plans.

Addressing Carbon Accounting and Climate Change

Future extensions of our methodology include projecting the effects of climate change on ecological sustainability and providing spatial simulations (Miller 2007). We also advocate evaluating adaptive and mitigation strategies as outlined by Millar and others (2007).

Carbon accounting for mitigation strategies was provided by the carbon extension of the Forest Vegetation Simulator (Havis and Crookston 2008). Carbon accounting attributes are shown in table 7. Our results indicate that as the ecosystem moves further away from reference conditions over time (due to more acres moving into the closed states), the ecosystem sequesters more total carbon (above and below ground), because the closed states contain more stand carbon per acre than do the open states. This represents a trade off that must be evaluated by land managers: managing toward reference conditions versus managing to maximize short-term carbon sequestration.

The long-term sustainability of these uncharacteristic closed states is dependent upon insect, drought, and fire occurrence. For example, as closed canopy states are removed by wildfire, the sequestered carbon is released to the atmosphere. Sequestration of excess carbon in closed canopy states in frequent-fire adapted forest types may result in a net long-term loss of carbon sequestration values when uncharacteristic stand-replacing fires occur (Hurteau and others 2008).

The current model does not provide for charcoal or soil organic carbon, though future analyses are likely to include these components (DeLuca and Aplet 2008; Jenkins and others 2003). The amount of soil carbon in the pinyon-juniper woodlands is significant; it can be up to 8 tons per acre in the A horizon and up to 12 tons per acre in the total solum (Meurisse and others 1991). The amount of organic carbon in soils within the pinyon-juniper woodlands is inherently lower than higher elevation montane forest types (Meurisse and others 1991).

Adaptation strategies necessitate predictions about future vegetation patterns and at this time, we are considering the advantages and disadvantages of alternative approaches to modeling climate change. The assumptions in our projection

Table 7—Thousands of tons of stand carbon occurring above and below ground.

Vegetation state	A	B	C	D	E	F	G	Total
Current condition	194	22	333	1,010	0	1,277	1,869	4,705
Projected trends								
10 years	169	87	231	935	45	1,352	2,262	5,081
50 years	127	130	154	823	45	1,352	3,147	5,778
200 years	110	87	128	823	89	1,277	3,639	6,153

[a] Stand carbon per acre is taken from table 5 and does not reflect charcoal or organic soil carbon. Acres are taken from table 6.

models can be modified in the following ways to incorporate the emerging evidence from climate research (Hemstrom and Merzenich, unpublished document):

1. Types of states: Climate change may result in the addition or removal of states within a PNVT as new vegetation composition and structural patterns are introduced with changing site potential and processes (such as the introduction of exotic species).
2. Types of transitions: Climate change may result in the addition or removal of transitions within a PNVT, with novel patterns of vegetation composition, structure, and process.
3. Rates of transitions: The rates of transitions between model states for existing transitions, for example, stand replacing fire, may change within the PNVT and planning area.
4. New (adventive) PNVTs: Adventive PNVTs may need to be modeled, depending on the climate scenario.
5. Transitions between PNVTs: In addition to transitions *within* PNVT models, transitions *between* PNVTs may be necessary to reflect the movement of area between PNVT classes as climate changes.
6. New management activities: New management activities may be necessary to respond to adaptive and mitigation strategies (Millar and others 2007), along with modification to the rates of existing transitions.
7. Projected climate variability: Changes in the annual variation of phenomena such as wet years, dry years, insect and disease incidence, etc. may be explicitly modeled within existing VDDT software.
8. Addressing multiple climate scenarios: Current assumptions can be modified to reflect each climate change scenario that needs to be considered by management; for example in scenario 1 the planning area may be getting warmer and drier, and in scenario 2 the planning area may be getting warmer and wetter.

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The content of this paper reflects the views of the authors, who are responsible for the facts and accuracy of the information presented herein.

Part 5: Summary

Summary of the 2009 National Silviculture Workshop

James M. Guldin[1]

Abstract—The theme of the 2009 National Silviculture Workshop held in Boise Idaho in June 2009 was, "Integrated management of carbon sequestration and biomass utilization opportunities in a changing climate." The session had a series of outstanding presentations and field tours focused on the theme of the meeting nationally, and with specific reference to the forests of the northern Rocky Mountains. There was consensus in the meeting that climate change will be the defining issue for this generation of resource managers. Silviculture will play a key role in the future of climate change, and it will be option-creating silviculture, not option-reducing silviculture. Silviculturists and decision-makers must use the best science regardless of how it tests the popular will and the politically easy decision—and this is no simple task in light of the administrative issues that govern forest management on Federal lands. An adaptation strategy for climate change will also require integrating the principles of landscape ecology modeled via landscape succession models with principles of forest ecology and silviculture modeled using tools such as the Forest Vegetation Simulator. Data from long-term experiments will be increasingly important to validate simulation outcomes. Finally, state-of-the-art science delivery programs will be needed to think about and develop silvicultural prescriptions that address climate change adaptation strategies in project-level decisions, and that are implemented spatially in a strategic way across the forested landscape.

Introduction

Approximately 150 attendees convened in Boise, Idaho, in early June 2009 for the 12th biennial meeting of the National Silviculture Workshop. Amid pleasant weather and a fine setting along the Boise River, the speakers and attendees met to discuss topics on the theme, "Integrated management of carbon sequestration and biomass utilization opportunities in a changing climate." The session had a series of outstanding presentations and field tours focused on the theme of the meeting nationally, and with specific reference to the forests of the northern Rocky Mountains.

The speakers welcoming the group succinctly summarized the challenges in science and management facing silviculturists. Mr. Bill LeVere, Director of Natural Resources for the USDA Forest Service (FS) Intermountain Region (Region 4) in Ogden, UT, noted that 12 of the warmest years on record have occurred since the 1990s; that we have experienced earlier snowpack melt, longer growing seasons, earlier greenup rates, and a 30 percent decline in August streamflows in the northern Rocky Mountains; that warmer climate provides longer periods for forests to be under stress from insects and disease; and that resource managers have lots of questions but few answers. Dr. Tom Crow, Program Manager with the FS Rocky Mountain Research Station in Fort Collins, CO, suggested that climate change will be the single defining issue facing the current generation of resource managers, that there are deep scientific questions underlying the concepts of managing forests for resilience and adaptation in the face of climate

In: Jain, Theresa B.; Graham, Russell T.; and Sandquist, Jonathan, tech. eds. 2010. Integrated management of carbon sequestration and biomass utilization opportunities in a changing climate: Proceedings of the 2009 National Silviculture Workshop; 2009 June 15-18; Boise, ID. Proceedings RMRS-P-61. Fort Collins, CO: U.S. Department of Agriculture, Forest Service, Rocky Mountain Research Station. 351 p.

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change—and that all of this is occurring in an era of declining Federal budgets. Ms. Cecelia Romero Seesholtz, the Forest Supervisor of the Boise National Forest in Boise, ID, offered a generous welcome, and thanked her staff for working so hard and successfully to convene the Workshop. To Supervisor Seesholtz, the organizing committee and attendees at the Workshop offer heartfelt thanks for her leadership and support.

Plenary Session

The keynote address was provided by the Hon. Cecil Andrus, Director of the Andrus Center for Public Policy in Boise, ID. He engaged the Workshop with a highly entertaining presence, and included two key concepts in his comments. First, he challenged attendees to manage National Forests for energy, economic development, and environmental quality for a changing climate—and suggested that these three objectives are important and they can be contradictory. And he challenged research scientists and resource managers alike to use the best science in making resource management decisions, regardless of how it tests the popular will and the politically easy decision.

Two other speakers were invited to participate in the Plenary Session. The first was Dr. Dave Cleaves, Acting Deputy Chief for FS Research and Development (R&D), who suggested a number of key elements from his position of research leadership in Washington, DC. He noted that silviculturally, there is "no sequestration without adaptation"; that the Forest Service mission under changing climate will not change, but we'll have to work in a different context to fulfill the mission; and that silviculture is a key role in the future of climate change—specifically, option-creating silviculture that expands opportunities available for future managers, not option-reducing silviculture that constrains what future managers can do in the field. The second was Mr. Mike DeBonis, Southwest Region Director for the Forest Guild in Santa Fe, NM, who reminded us of two key points: that the forester in the field is the eyes and ears, or the first respondent, when things go awry in the woods, and that collaboration and alliances are critical.

Technical Papers, Day 1

The first day of technical papers provided some perspective on the workshop theme using computer models, academic training, and plot data from the Forest Inventory and Analysis (FIA) program of the USDA Forest Service. Nick Crookston of the Rocky Mountain Research Station in Moscow, ID, described the first version of Climate-FVS, an extension of the widely used Forest Vegetation Simulator (FVS). A key component of this model is based on species-level climate profile models that predict the climatic range of species occurrence. He included some example outputs of this model that were quite sobering, suggesting that climate change will drastically alter at least some of our forest ecosystems during this century.

Bob Deal of the Pacific Northwest Research Station in Portland, OR, defined terms such as sequestration and ecosystem services; he then suggested that there will be opportunities in formulation of policy and in field practice to link sequestration, ecosystem services, and climate change—but cautioned that we have to get it right.

Keith Moser, with the North Central Research Station in St. Paul, MN, pointed out the extraordinary value of FIA data across the Nation. The FIA survey is

designed to describe forest resources at a large scale (States and Regions), but increasingly has value at small scales (such as National Forests, and large private land holdings), especially when supplemented with state-of-the-art aerial imagery.

Linda Nagel from Michigan Technological University, Houghton, MI, described a science delivery program for the consideration of climate change in project-level decisions that she and her colleagues have developed as part of the Forest Service National Advanced Silviculture Program. It was generally agreed in side conversations at the Workshop that this program is an outstanding model for other regions of the nation as well, not only from the perspective of content but also highlighting the cooperation between the academic community and the Forest Service to address a timely issue from a practical perspective.

Don Vandendriesche from the National Forest System (NFS) Forest Management Service Center in Fort Collins, CO, described the use of FVS to calibrate state and transition models (a Rube Goldberg-style modeling approach with algorithms for buckets, pipes, and valves) that are being used for landscape assessments to quantify long-term trends in forest structure. He posed the question: "Are analysts truly able to assist decision makers or just add complexity to already overburdened planning staffs?" Indeed, the methods presented support the effort by providing an empirical basis to an otherwise subjective process.

Reuben Weisz from the Southwest Region (Region 3), in Albuquerque, NM, reported on a pinyon-juniper (*Pinus edulis-Juniperus* spp.) grassland case study, and spoke to a practical tradeoff between restoration and sequestration—an inherent contradiction between managing for open canopy conditions in a restoration context versus on-site opportunities for carbon sequestration through accumulation of biomass. He cautioned that balancing such contradictory concepts requires a long-term, life-cycle-based perspective.

Technical Papers, Day 2

The second day of the Workshop was introduced by a panel addressing climate change at the local, regional, and national scales. Ed Gee, National Woody Biomass Utilization Team Leader, Forest Management Staff, in the Washington Office, brought the Workshop up to speed on national issues. Dave Atkins, Biomass Utilization Program Manager in the Northern Region (Region 1) office in Missoula, MT, introduced the concept of "negawatts"—efficiencies that feed power into the grid or conserve the drawing of power from the grid. Jay O'Laughlin, Professor of Forest Resources at the University of Idaho in Moscow, ID, noted that wood energy has its own byproducts—restoring forest health, providing renewable energy alternatives, restoring local economies, and yielding a bonus in carbon management. Barry Wynsma, a field forester on the Idaho Panhandle NF in Bonner's Ferry, ID, described the silvicultural tactic of "designation by description" as the "Leatherman tool" for foresters, who save $50/ac in time spent in sale preparation when using that tactic in combination with weight scaling, compared to individual tree scaling by volume.

Following the panel, three technical papers were included in the remainder of the morning session, and seemed to share a theme of carbon sequestration. Doug Basford with the Salmon-Challis NF in Salmon, ID, described a growth model for mixed conifer stands in southwestern Idaho without using FVS—and reported excellent results. This descriptive analysis shows that analog approaches to data analysis sometimes work as well as computer models, and point to the value of experience in interpreting data.

Matt Busse with the Pacific Southwest Research Station in Redding, CA, reminded the workshop of the "Miracle of Photosynthesis" in which carbon dioxide is taken in by growing trees, and stored as cellulose under the familiar equation:

$$6\ CO_2 + 6\ H_20 \Rightarrow C_6H_{12}O_6 + 6\ O_2$$

Busse then described what he called the lesson of mitigation: that managed stands store more carbon than unmanaged stands (in the short term); that managed stands are more stable in sequestration of carbon than unmanaged stands; and that as a result, forest management can help mitigate climate change.

Don Bragg with the Southern Research Station in Monticello, AR, noted that reality is a special case in the context of computer modeling. He suggested that southern pines have a role to play in carbon sequestration, and that data from the real world—specifically, from long-term experiments—are useful to validate simulation outcomes.

The afternoon session featured five papers continuing along the theme of carbon sequestration. Alan Ager with the Pacific Northwest Research Station in Prineville, OR, discussed risk analysis at a landscape scale from the perspective of catastrophic fires and then extended the idea to climate change. He suggested that the impact of climate change can be calculated as a risk analysis, or the probability of some event occurring coupled with the changing value of the loss associated with that event.

Mike Battaglia with the Rocky Mountain Research Station in Fort Collins, CO, discussed the practice of thinning, noting that thinning removes carbon from the site—but that removal is preferable to losing all the carbon on the site in a catastrophic fire. He further noted that there are substantial amounts of CO_2 offsets in areas that need fuel reduction, and that denser and more productive stands will provide greater benefits.

Terrie Jain with the Rocky Mountain Research Station in Moscow, ID, pondered the origins of the commonly used conversion:

$$[C] = \text{biomass} * 0.5$$

Her data show that [C] ~42-48 percent, which led her to wonder whether is it worth the effort to use actual carbon content rather than the commonly used conversion. Her reply suggested that it depends on the analysis, but her demeanor implied that of course using the real value of the conversion is appropriate, especially at the stand level.

Tara Keyser with the Southern Research Station in Asheville, NC, reported results from her dissertation research in Black Hills ponderosa pine (*P. ponderosa*) crown physiognomy. She observed that ponderosa pine there features an unusually large lollipop-sized crown that contributes to growth and to flammability; she concluded that it was important to properly model canopy physiognomy.

Finally, Chris Keyes with the University of Montana in Missoula, MT, gave an update on research and management of the school's renowned Lubrecht Experimental Forest (EF). He reported on the status and planning for continuing several classic long-term studies and for initiating new research, and observed that challenges facing the Lubrecht EF included concerns about infrastructural support and competing uses other than research.

Field Trip, Day 3

The Workshop always includes a field trip as part of the session, and this year the trip headed for the Boise Basin Experimental Forest, on the Idaho City Ranger District (RD) of the Boise National Forest. The tour guides were

Russ Graham, Terrie Jain, Bob Denner, and Jonathan Sandquist of the Rocky Mountain Research Station in Moscow, ID; Tom Martin, Regional Silviculturist with the Intermountain Region in Odgen, UT; Barry Stern of the Boise NF Supervisor's Office; John Sloan of the Lucky Peak Nursery in Boise, ID; Ray Eklund, Shannon Hitch, and Allyn Spanfellner of the Idaho City RD; and John Roberts with the Idaho State Department of Lands, Idaho City, ID. Thanks to all of these folks for an outstanding tour.

Tour stop 1 featured a 40-year-old ponderosa pine plantation established on terraces created by bulldozers (fig. 1). Terracing was a highly controversial practice in the 1960s and 1970s. It was designed to remove competition, stabilize erosion, and provide a more or less level planting site on steep slopes after clearcutting in the northern Rocky Mountains. The practice fell into disrepute in the 1970s, largely because of the cost and the critical response from the public about the aesthetic and ecological effects. However, after 40 years, this plantation appears to be growing at an acceptable rate, and the harshness of the terraces is somewhat diminished by time.

The second tour stop demonstrated the effects of fire exclusion in mixed conifer stands in the northern Rocky Mountains (fig. 2). Fire exclusion in mixed conifer stands in the northern Rocky Mountains results in high densities of seedlings and saplings, which can serve as hazardous ladder fuels in fire-adapted ecosystems.

One way to ameliorate that condition is to engage in a restoration treatment conducted using the free selection approach championed by Graham and Jain (2005) in these stands. The third tour stop illustrated a recently conducted free selection restoration treatment (fig. 3). This stand features the heterogeneous clumped and scattered distribution of trees that Graham and Jain seek when implementing the prescription. However, the Boise Basin EF also supports a number of more traditional long-term uneven-aged selection reproduction cutting studies in ponderosa pine stands, in which the classic reverse J-shaped distribution of stand structure are readily apparent (fig. 4).

Figure 1—A terraced 40-year-old ponderosa pine plantation on the Idaho City Ranger District, Boise NF, in southern Idaho. (USFS photo by James M. Guldin).

Figure 2—Fire exclusion in a mixed conifer stand in the northern Rocky Mountains results in high densities of seedlings and saplings, which can serve as hazardous ladder fuels in fire-adapted ecosystems, as illustrated in this image from the Boise Basin Experimental Forest in southern Idaho. (USFS photo by James M. Guldin).

Figure 3—Implementation of the free selection reproduction cutting method for old-growth ponderosa pine restoration on the Boise Basin Experimental Forest in southern Idaho. The stand featured a heterogeneous clumped and scattered distribution of trees—and ironically, of tour participants as well. (USFS photo by James M. Guldin).

Figure 4—Classic reverse J-shaped stand structure in an uneven-aged reproduction cutting study in ponderosa pine on the Boise Basin EF in southern Idaho. (USFS photo by James M. Guldin).

Wildfire is of course an important issue in the northern Rocky Mountains and Intermountain West, and managers seek information about the ecological effects of salvaging trees in burned areas. That led Jain and Graham to develop a research study that simulates the effects of wildfire and follows up the simulated fire with salvage, in a controlled context where soils and water can be monitored (fig. 5). The study was implemented by the Idaho City RD timber and fire staff, and workshop participants admired the deft creativity and attention to detail that the staff used to meet the researchers' needs.

Participants then viewed an operational ponderosa pine thinning study on the Boise Basin EF in southern Idaho (fig. 6). The size of the slash piles was impressive, not only from the perspective of the cost of conducting follow-up fuels treatments in forest operations on difficult terrain, but also as an indication of the potential of these stands to produce supplemental biomass associated with harvest of merchantable trees.

At the final tour stop, the group observed an application of the shelterwood reproduction cutting method on forest lands belonging to the State of Idaho (fig. 7). The differences between the shelterwood method as imposed by State foresters and the free selection method discussed earlier in the day were apparent, and largely as expected. The shelterwood method removed more merchantable

Figure 5—Simulated wildfire and salvage research study in northern Rocky Mountain mixed conifers, on the Boise Basin EF, southern Idaho. (USFS photo by James M. Guldin).

Figure 6—View through an operational ponderosa pine thinning study on the Boise Basin EF in southern Idaho; the slash piles contain unmerchantable material harvested but not hauled, and show the potential of these stands to produce supplemental biomass associated with harvest of merchantable trees. (USFS photo by James M. Guldin).

Figure 7—Application of the shelterwood reproduction cutting method on forest lands of the State of Idaho. As expected, the shelterwood method removes more merchantable stems and thus retains lower post-harvest residual basal area than was observed in the free selection reproduction cutting studies on the Boise Basin EF. (USFS photo by James M. Guldin).

stems and thus retained lower post-harvest residual basal area than was observed in stands marked using the free selection method.

The day was capped with the official banquet routinely held at the Workshop, with a long and detailed presentation on the history of the region presented by Susie Osgood, Forest Historian on the Boise NF. The highlight of the banquet was the presentation of the National Silviculturist of the Year awards. Honorees this year from the National Forest System were Joseph F. Myers with the Coeur d'Alene Nursery, Idaho Panhandle National Forests, in Coeur d'Alene, ID, Thomas Martin and Donald Vandendriesche. Honorees from FS Research and Development (R&D) were Marilyn Buford, National Program Leader for Silviculture on the R&D staff in Washington, DC., Daniel Dey, Research Forester with the Northern Research Station in Columbia, MO., and Henry Mcnab, Research Forester with the Southern Research Station in Asheville, NC. The names of these recipients have been added to the National Silviculturist of the Year recipient data table (table 1).

Table 1—National Silviculture Workshop Award Recipients, 2001-2009.

YEAR-Region (Location)	National Forest System	Research and Development
2001—Region 6 (Hood River, Oregon)	Fred Zensen, R6 TM	Ray Shearer, Rocky Mt. Res. Stn. Jim Jenkinson, Pacific Southwest Res. Stn. Bob Curtis, Pacific Northwest Res. Stn. Nicholas Crookston, Rocky Mt. Res. Stn.
2003—Region 2 (Granby, Colorado)	Brian Ferguson, R-4 Mary Frances Maholovich, R-1 Monty Maldonado, WO-FM Tom Tibbs, R-8	Jim Guldin, Southern Res. Stn. Kurt W. Gottschalk, Northeastern Res. Stn. Paul Johnson, North Central Res. Stn.
2005—Region 5 (Tahoe City, CA)	William "Bill" Jones, R-9 Glenda L. Scott, R-1 Tom Landis, S&PF Jim Russell, R-10	Jim Barnett, Southern Res. Stn.
2007—Region 10 (Ketchikan, Alaska)	Marlin Johnson, R-3 Kathy Sleavin, WO-Ft. Collins Dave Evans, R-5 Bill McArthur, R-6	Terrie Jain, Rocky Mt. Res. Stn. Steve Shifley, North Central Res. Stn.
2009—Region 4 (Boise, Idaho)	Thomas Martin, R1 Donald Vandendriesche, WO-Ft. Collins Joseph F. Myers, R1	Henry Mcnab, Southern Res. Stn. Marilyn Buford, WO Daniel Dey, Northern Res. Stn.

Technical Papers, Day 4

The final day of the meeting featured papers on the topic of biomass and bioenergy, and a broad spectrum of research was reported. Matt Busse summarized 20-year results of the Long-Term Site Productivity study plots in California, and in doing so illustrated the value of long-term studies, especially studies networked across the Nation.

Mark Coleman with the University of Idaho in Moscow, ID, gave an extensive review of the University's research on pyrolysis. He concluded that the science of pyrolysis requires thoughtful development and testing before it becomes operational. However, issues associated with char and char disposal will be a significant hurdle—or perhaps a significant opportunity as a bridge between carbon sequestration and biomass utilization.

Greg Jones with the Rocky Mountain Research Station in Missoula, MT, addressed concepts of greenhouse gas emissions. He noted that there is less greenhouse gas emission if biomass is processed for energy rather than if it is burned. He also suggested that that energy used to harvest, collect, and transport biomass is tiny compared to the energy lost when biomass is burned.

Tim Swedberg with the Joint Fire Science Program Office in Boise, ID, spoke about the development of integrated decision support systems for fuels treatments.

Henry Mcnab discussed a project in the southern Appalachians using shrubs to support overstory tree site index predictions. This project was inspired by field observations by an experienced professional, and serves as a testament to multiple applications of long-term research studies.

Andy Youngblood with the Pacific Northwest Research Station, La Grande, OR, examined the silvicultural suitability and practical application of ponderosa and lodgepole (*Pinus contorta* Douglas ex Louden) pines for biofuels. In doing

so, he suggested that such studies illustrate the ongoing value of the agency's network of Experimental Forests and Ranges into the 21st century.

John Shaw with the Rocky Mountain Research Station, Ogden, UT, explained current research opportunities and future applications of stand density index research using Forest Survey data, which further points to the value of that national data set in scientific inquiry.

Finally, Mike Ryan with the Rocky Mountain Research Station in Fort Collins, CO, provided a primer on carbon, and reminded the Workshop attendees that forest disturbance does not cause carbon loss unless the area remains unforested.

Discussion

The management actions associated with climate change will change the nature of silviculture in the 21st century. The scale of treatment needs threaten to overwhelm agency capacity, with escalating need for even the simplest ameliorative treatments, such as thinning in the face of declining numbers of personnel and declining budget capacity in the field. With the political overtones of climate change among the public and the ecological overtones implicit in decisions to promote species migration through silvicultural assistance, gridlock in project execution will likely further stifle widespread implementation of management activities conducted specifically for climate change.

Even if gridlock melted away, the agency's capacity to apply silvicultural treatments to forest stands nationwide is limited, given the extensive land base in roadless areas, in unsuitable condition for access or silvicultural operations, or in stands that cannot be economically managed. Some thought should be given to merging approaches for mitigating the effects of climate change on the Nation's forests and rangelands by combining stand-level silvicultural treatments with a much better understanding than we have today as to where on the landscape those treated stands should be located.

There is a need for interim silvicultural recommendations that managers can use to practice robust or resilient forestry in the face of changing climatic conditions. This can most effectively be accomplished through a collaborative partnership between our most creative and experienced practicing silviculturists in the field, and our best silviculture researchers and landscape ecologists in FS R&D and academia. That need exists at two levels—one level for professionals making silvicultural prescriptions consistent with stand dynamics and landscape ecology, and the other for field forestry and biological science technicians who are most often the personnel in the woods with the paint guns implementing the prescribed treatments. The work reported at this meeting by Nagel and others is an excellent step, and unfortunately at this time is limited to the federal agency foresters working for their silvicultural certification in the National Advanced Silviculture Program. There's a much broader need to provide this continuing education across the agency, and there's a concurrent need to carry this training beyond the green line to foresters working for forest industry, forest investment organizations, and non-industrial private forest landowners as well.

There seems to be an inherent dichotomy between management for carbon sequestration and management for biomass and bioenergy, where an intensive degree of utilization almost seems like the antithesis of sequestration. More thought should be given to connecting these apparently different ideas, especially in the context of policies that try to separate them. There are some silvicultural opportunities in this context if a stand could be partitioned into elements appropriate for biomass and elements appropriate for sequestration. Unfortunately, offset providers under a program such as the Chicago Climate Exchange typically

operate under a contract that involves all of the forested lands in the ownership. These contracts currently don't allow the owner to enroll some stands but not others, nor do they allow owners to include reserved portions of stands being harvested. This piecemeal approach is inconsistent with the concept of climate change adaptations at the landscape scale.

This all-or-nothing approach limits the ability of a landowner to be rewarded for forest practices that sequester carbon using simple tools such as partitioning some arbitrary portion of the stand into components appropriate for biomass or sequestration. For example, one of the stands on the Boise Basin EF had recently been thinned using the free selection method, with a residual stand containing about 60 ft^2/ac. If the long-term goal in this stand is to retain 60 ft^2/ac indefinitely, that portion of the stand is essentially serving the purpose of carbon sequestration, and the harvested component is serving as a biomass source. Much of the timber-available acreage in National Forest System would fall in this category. However, the retained trees would not qualify as an offset-providing stand for support under existing authorities. The concept of partitioning stands into sequestered residuals versus harvestable surplus is a practical way to combine sequestration and biomass production, but there's currently no provision for this mutualism in current cap and trade markets.

A number of the presentations in the meeting were prepared using FVS, and some of the results presented from these models portend dramatic changes in forest ecosystems of the Nation. It is important to develop the methodology using stand-level models such as FVS as well as landscape succession models to answer hypothetical questions about the long-term effects of silvicultural practices and ecological changes on forest ecosystems. A word of caution from this is that models are better interpreted in a relative sense than an absolute sense. One might ask whether broader efforts should be made to validate models such as FVS and landscape succession models with independent long-term data sets such as those available from FIA on forest growth, yield, and developmental dynamics—as well as data from long-term studies with repeated measurements over time, such as can be found on many of the agency's Experimental Forests and Ranges.

In a similar vein, program organizers should continue to include presentations from real data, especially long-term data, during future workshops. These papers often provide a field-based context for the discussions that occur. They also remind us that the development of tools such as FVS and landscape succession models depend on data collected in the field over time, and on high-quality spatial data from our Nation's forests. It should also remind us of the importance of retaining field-going FS R&D scientists who maintain existing long-term studies and install new ones. If our R&D capacity creates a new generation of research foresters who work only in FVS or other modeling applications rather than in field studies, our infrastructural capacity to build and maintain the field studies necessary to refine existing models and develop new ones will be compromised.

In this workshop, presenters from FS R&D and academia outnumbered those from Regional Offices and National Forests. This may have been due to administrative constraints on travel during the fiscal year in which the meeting was held. One of the great opportunities enabled by the National Silviculture Workshop is to hear the success and failures from the field perspective, and to have practicing silviculturists at Ranger Districts, Supervisor's Offices, and Regional Offices interact with R&D researchers and academics. Speaking as one of the researchers, this interaction clearly flows both ways, and those of us who inhabit the ivory tower learn as much if not more than our colleagues in the field from these interactions. Future meeting organizers are encouraged to continue, and perhaps even to slightly broaden, the opportunities for professionals from the field to give case study presentations during the technical sessions.

This was the first workshop where a deliberate effort was made to broaden participation so that University research scientists could attend. This was an unequivocal success at this meeting. The professors who attended, gave presentations, and participated in the field tour were a welcome addition to the structure of the meeting. In a forestry research environment where Universities and R&D both are losing capacity through erosion of budgets and staff, opportunities to expand participation of University research silviculturists at the workshop should continue. This will provide an additional venue for mutual interaction, and will continue to help University researchers understand and appreciate the silvicultural challenges and opportunities available on Federal forest lands. This also gives field silviculturists the opportunity to interact with academics as well as R&D researchers, with attendant benefits for the range of expertise they can tap when necessary. This concept also applies as the agency tackles the study and application of both mitigation and adaptation strategies for climate change.

The next workshop will be held in Region 8, with tentative plans to schedule the workshop in Tallahassee, Florida, in May 2011. The workshop will feature opportunities to discuss the ecology and silviculture of southern pines during the most pleasant time of year to visit the South. Here's hoping the meeting in Tallahassee can be as successful as was this current workshop in Boise.

References

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The content of this paper reflects the views of the authors, who are responsible for the facts and accuracy of the information presented herein.

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