Silviculture in Special Places:

Proceedings of the 2003 National Silviculture Workshop

September 8-11, 2003
Granby, Colorado

United States
Department
of Agriculture

Forest Service

Rocky Mountain
Research Station

Proceedings
RMRS-P-34

September 2004

Shepperd, Wayne D.; Eskew, Lane G., compilers. 2004. **Silviculture in special places: Proceedings of the National Silviculture Workshop; 2003 September 8-11; Granby, CO.** Proceedings RMRS-P-34. Fort Collins, CO: U.S. Department of Agriculture, Forest Service, Rocky Mountain Research Station. 255 p.

Abstract

This proceedings presents a compilation of 20 manuscripts and five posters summarizing results of research studies and management projects conducted throughout the United States in areas with special natural resource values. Topics include the restoration of various fire dependent forest ecosystems, studies of historical ecology, use of genetics in silviculture, development of old growth and late-successional prescriptions, documenting natural regeneration in burned areas, comparisons of cutting methods, coping with advancing blister rust, delineation of rare aspen forests, two-aged management in Appalachian hardwoods, forest soil productivity, managing a recreation river, and forest structure/burn severity relationships.

Key words: silviculture, forest management, genetics, forest restoration, prescribed fire, fuels treatments, cutting methods, white pine blister rust

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*Papers from workshop speakers are published here in the order in which presentations appeared on the workshop agenda.

Preface

As our agency enters the 21st century, we increasingly find ourselves engaged in the protection and management of forest ecosystems in "special" places for a complex mixture of resource values. Some of these areas are special because of where they are or what they contain, while others are special because of their history or because they are suitable for special uses. However, the one common feature of the special areas addressed in this document is that they are in forest ecosystems. Maintaining these special values within these forest ecosystems requires a non-standard approach to the practice of silviculture. A thorough knowledge of how these ecosystems function and skilled application of a variety of vegetation manipulation techniques are required to achieve and maintain the special attributes that we have or desire for these places. Whether these special places are recreation areas, critical wildlife habitat, urban-wildland interfaces, scenic vistas, or multi-use forests, they all may require some form of management intervention to retain their unique values. The reader is invited to enjoy the experiences shared by a special group of people who attended the 2003 U.S. Forest Service National Silviculture Workshop held in a "special place" in the beautiful Colorado Rockies and to share our experiences and the knowledge we gained about the practice of silviculture in special places everywhere.

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Acknowledgments

The organizing committee members of the 2003 National Silviculture Workshop deserves special recognition for their efforts in making the meeting and this publication a success:

Frank Burch, WO
Monty Montanoldo, WO
Clark Baldwin, WO
Bruce Short, R2
Jan Burke, R2
Wayne Shepperd, RMRS
Lance Asherin, RMRS
Rick Caissie, R2

Thank you to the Region 2 silviculturists who served as moderators and timekeepers:

Blaine Cook
Jim Allen
Christie Schneider
Philip Kemp
Robert Vermillion

A special thanks to District Ranger Craig McGuire and Project Leader Todd Mowrer and their staffs for hosting the event and associated field trips.

And finally, thanks to the Washington Office and the Rocky Mountain Research Station for their generous financial support of the workshop and this publication.

Past, Present, and Future Role of Silviculture in Forest Management

Russell T. Graham[1] and Theresa Benavidez Jain[1]

Silviculture

In general, silviculture can be defined as the art and science of controlling the establishment, growth, competition, health, and quality of forests and woodlands to meet the diverse needs and values of landowners and society on a sustainable basis (Helms 1998). This definition or variations of it have existed since the late 1800s. Gifford (1902), an Assistant Professor of Forestry at Cornell University in New York, used the term arboriculture to describe the growing of trees for any purpose and in any way whatever – singly, in groups, or in the form of forests. He went on to define silviculture as a part of the broader art of arboriculture. Schlich (1904), Professor of Forestry at the Royal Indian Engineering College, Coopers Hill, India, stated that "the culture of forests with the objective for which a particular forest is maintained depends on the will and pleasure of the owner, in so far as his freedom of action is not limited by rights of third persons or legal enactments." He went on to say "silviculture, in its narrowest sense, is understanding the formation, regeneration and tending of forests until they become ripe for the axe." Therefore, the beginning of silviculture in the United States was closely aligned with forest management, and the general theme of most silvicultural practices was to produce forest crops.

Silviculture and Timber Management Relations

As the foundations of silviculture were being developed in the late 1800s, the concept that forests should be reserved and managed for the good of society was also developing. Laws such as the Timber Culture Act of 1873 and the Timber and Stone Act of 1878 were passed allowing settlers on homesteads to switch from growing grain crops to trees as part of the residency requirements (Steen 1976). The acts authorized the sale of non-tillable public timberlands for personal use. By 1873 the American Association for the Advancement of Science (AAAS), through the leadership of Franklin Hough, a physician, began lobbying Congress to pass a resolution promoting the cultivation of timber and the preservation of forests. Hough continued his efforts to get a bill through Congress in 1874 and 1875 but was unsuccessful. He supported these efforts by studying and writing papers on forestry and distributing them through the AAAS. Congressman Dunnell from Minnesota championed the cause but all attempts to get the bill

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through the Public Lands Committee failed. In August of 1876, Dunnell made a motion to transfer the substance of the bill to the general appropriations bill authorizing the Department of Agriculture to appoint a man of "approved attainment" to report on forest supplies and conditions. With the passing of this law and through this parliamentary tactic began the long tradition of having the forestry agency in the Department of Agriculture with Hough becoming its first chief (Steen 1976). This was called the Department of Forestry with close ties to the American Forestry Association.

The majority of the information Hough used for his self-taught forestry education was based on European models of forestry, in particular forest management in Germany. This strong connection to German forestry was exemplified by the appointment of Bernhard Fernow as the third Chief of the Division of Forestry in 1886. (Nathaniel Egleston succeeded Hough as Chief in 1883 and served with uncertainty until replaced by Fernow.) Fernow started his forestry apprenticeship in the Prussian Forestry Department and also received advanced training in Prussia. He immigrated to the United States in 1876 and brought with him the German penchant for "slick and clean" forests regularly divided into blocks (Miller 1992).

Through fraud, timber companies used the Timber and Stone Act to acquire and harvest large quantities of timber on lands in the western United States. Some of the most blatant fraud occurred in northern California. As the price of timber rose, fraudulent practices increased causing agents in the Department of Interior to investigate thousands of fraud and trespass cases every year. But the practice continued to escalate and became a way of life in the western United States. In 1889, the American Forestry Association, with Fernow chairing the law committee, lobbied both Congress and the Administration for legislation creating reserved parcels of land and providing a commission to administer them. No action by either branch of government towards reserving forests occurred until Fernow and his associates, in 1891, convinced Interior Secretary Noble that it was his responsibility to protect the public domain. During this period a bill, The Creative Act, was being prepared in Congress to revise a series of land laws including the Timber and Stone Act. Noble was able to convince the conference committee at the eleventh hour to add Section 24 to this bill. This section authorized the President to create forest reserves and was not referred back to the originating committees for their consideration. Therefore, when the bill passed, section 24 became the law of the land by default. President Harrison wasted no time in using what became known as the Forest Reserve Act of 1891to create 15 forest reserves containing 13 million acres in the newly established western states. President Cleveland continued to add more acres but stopped until Congress provided a means to protect the reserves within the Department of Interior (Steen 1976).

Not only did Fernow and his associates influence forest legislation; they also framed the forestry education in the United States, controlled the early professional organizations (American Forestry Association), and produced most of the forestry publications (Forestry Quarterly). By 1897, 20 institutions, of which most were land grant colleges, offered some instruction in forestry with silviculture a part of the curriculum. In 1898, the New York State College of Forestry was organized and a year later the Pinchot family (a well-to-do upstate New York family, of which Gifford was a member and advocated conservation of Adirondack forests) endowed a forestry school at Yale (Ise 1920). Graduates of these schools formed the core of the Division of Forestry and later the Forest Service (Steen 1992).

Era of Gifford Pinchot

Figure 1—Gifford Pinchot succeeded Bernhard Fernow as Chief of the Division of Forestry in 1898. He graduated from Yale in 1889 and furthered his education in Europe, refining his views and philosophy of forestry.

Gifford Pinchot succeeded Fernow as Chief of the Division of Forestry in 1898. He had a tremendous impact on the forests of the United States both in their acquisition and their management (figure 1). He graduated from Yale in 1889, but he also studied formally in Europe and spent over a year touring and learning the forestry profession there. He returned to the United States and spurned Fernow's offers to become his assistant; instead, he went to work on the Biltmore Estate in western North Carolina to develop a forested estate worthy of Vanderbilt's wealth (Steen 1976). The Vanderbilt estate offered Pinchot the opportunity to put into practice the European systems he learned. This work allowed him to determine that forestry in North America could be a profitable venture, and helped solidify his views on forest management.

As the chief of the Division of Forestry, Pinchot, much like Fernow, mostly influenced forestry activities through publications and technical assistance to companies and private citizens. If Pinchot was to influence the management of the forest reserves, he had to work cooperatively with the Department of Interior because the forests were under its domain. By 1901 he was able to have the foresters in the Department of Agriculture make all technical decisions associated with the reserves and develop management plans while Interior personnel would patrol the reserves enforcing the land-use laws. In 1902 the Department of Interior issued the first manual on administration of the reserves outlining when grazing could occur in the reserves. But the bulk of the manual dealt with timber management. Even though other people were credited for drafting the text, most people credit Pinchot for the substance of the policies.

Theodore Roosevelt frequented upper New York State before he became Governor of New York and during this time he became acquainted with Pinchot. Roosevelt nominated Pinchot for membership in the Boone and Crockett Club, an elite hunter's club that Roosevelt helped to found. The two became best friends, even having wrestling and boxing matches and, after Roosevelt became President in 1901, they were frequent companions riding horses and playing tennis. So it was no surprise that after only three months in office Roosevelt told Congress that the forest reserves belonged not within the Department of Interior but in the Department of Agriculture, under Pinchot's Bureau of Forestry.

In addition to Pinchot, Roosevelt had strong views on how the forests of the United States should be managed and in March of 1903 he presented them to the Society of American Foresters. The essence of his views was captured as follows: "And now, first and foremost, you can never afford to forget for one moment what is the object of our forest policy. That object is not to preserve the forests because they are beautiful, though that is good in itself, nor because they are the refuges for the wild creatures of the wilderness, though that, too, is good in itself; but the primary object of our forest policy, as of the land policy of the United States, is the making of prosperous homes… Every other consideration is secondary" (Roosevelt 1905).

In 1905, again with considerable lobbying by the American Forestry Association, Pinchot's political savvy, some last minute political bargaining, and the argument that forests were crops, the forest reserves were transferred to the Department of Agriculture to be administered by the Bureau of Forestry. The Bureau of Forestry was then renamed the United States Forest Service. Two years later the reserves were renamed national forests, because the term reserve suggested they were to be held inviolate. They were not. Under Pinchot's vision, forests' use was not contrary to conservation, an important distinction from previous thought. When conflicting interests arose, the question would always be decided from the standpoint of the greatest good of the greatest number in the long run. During Pinchot's tenure as Chief, he gave a high priority to boundary survey, and men working alone on horseback often added up to 3 million acres per day per man to the national forests. For example, Pinchot and his Chief of Boundaries in one evening on a hotel room floor prepared 17 proclamations creating or adding to national forests in Arizona, California, New Mexico, Nevada, and Utah (Steen 1976).

During Pinchot's tenure, Yale was the foremost training ground for foresters joining the Forest Service. Building on the legacy Fernow initiated, the concept of forest management to produce timber crops was central to the education the schools offered. Therefore, the central theorem to the approach of producing timber crops was protecting forests from damaging animals, insects, and diseases and, most importantly, fire.

Timber Production and Forest Protection

By 1910 the forests of the United States were being utilized at a high rate to fuel the expanding economy. The Midwest was expanding rapidly and the forests of the West were ripe for providing raw materials. The western United States was also being settled and, as cities and towns were being developed, forest industries were quickly expanding to provide building materials locally to the cities and railroads while continuing to ship products to the Midwest. Western white pine and ponderosa pine were the primary species with Douglas-fir and western larch also of value; many other species were considered weeds and were often burned. Land clearing, railroads, and a nonchalant view of fires allowed fires to often burn freely throughout the Northern Rocky Mountains. In the spring of 1910, fires were ignited and continued to burn throughout the summer and, by August, 1,700 fires were burning throughout western Montana and northern Idaho. On August 20 and 21, dry Palouse winds blew causing these fires to erupt which resulted in over 3.1 million acres of often very valuable timberlands to burn. This loss created a sense of urgency to protect these valuable resources and to provide direction for the fledgling Forest Service by establishing a mission of protecting forests for human use (figure 2).

In 1864, because of the westward settlement movement, Congress conditionally granted the Northern Pacific Railroad Company nearly 40 million acres to aid in the construction and maintenance of a rail line from Lake Superior to the Puget Sound. The land was given as every other square mile in a checkerboard pattern in a 40-mile band through Wisconsin, Minnesota, and Oregon and an 80-mile band through North Dakota, Montana, Idaho, and Washington. These lands not only provided raw materials to the railroads in the western United States but also became important components of the timber industries in the region (Jensen and others 1995). With the combination of public and private lands producing raw materials along with the foundations of silviculture rooted in the German model, nearly all of the silvicultural methods and their supporting mensurational techniques being used were aimed at producing timber crops. The practice of silviculture was closely intertwined with timber management (Toumey 1916, Ise 1920).

Fernow (1916) expressly stated, “Silviculture, the production of wood crops, is pivot of the whole forestry business.” This close association of silviculture and timber management was evident even though Schlich (1904) and Gifford (1902) both indicated that forests, and the silvicultural practices used to maintain them, could be used for purposes other than timber production such as “protection and adornment.” The necessity to cultivate timber was being expressed by the amount of timber being consumed by the developing nation. And, for the United States to hold its position as a producer of timber or even ensure its future needs for forest products, a persistent effort to grow timber would be needed by the nation, states, and individuals. Public forests were to be managed by the Forest Service so they would ultimately attain their maximum production and retain it for all time (Toumey and Korstian 1947). This concept that wood supplies would diminish prevailed through the management plans and the policies affecting both private and public forests.

Contrary to western reserves, forests in the East were largely cut over and in private ownership or tax delinquent status. The Weeks Law of 1911 authorized the purchase of lands as national forests in the East, and by 1920 more than 2 million acres of land had been purchased (Steen 1976). The Clark-McNary Act of 1924 expanded the scope of the Weeks Law, and led to the establishment of agreements with states for purposes of fire protection on private lands. And finally, the McSweeney-McNary Act of 1928 laid the groundwork for a nationwide system of Forest Experiment Stations, which has evolved into the largest organization for the conduct of forestry research in the world.

Figure 2—Early wildfire prevention posters exemplified the urgency to protect forests from wildfires.

Intensive Forest Management

By the 1930s, with the available work force from the Civilian Conservation Corp (CCC), forests were being rapidly developed for human use, including recreation and water, but disease control, road building, and fire fighting activities were also undertaken. This workforce was cheap and, most importantly, enabled rigorous planting, cleaning, weedings, and thinnings to be accomplished, bringing intensive forest practices to many regions. The CCC also helped facilitate the large expansion of the research capabilities of the Forest Service. For example, a full 200-man CCC camp F-127 was established on the Priest River Experimental Forest and camp F-137 was allocated to the Deception Creek Experimental Forest, both in northern

Idaho (Graham 2004). During this period a wide range of experimental forests and ranges was established to provide information for intensively managing both public and private forests. These experimental areas were outdoor laboratories used for developing intensive silvicultural practices, fire danger rating systems, and insect and disease control strategies.

The CCC provided a work force for protecting forests from disease and fire. This work force pulled Ribes (the alternate host of white pine blister rust) on thousands of acres of public lands in the northern Rocky Mountains. In addition, they were readily available to fight fires throughout the United States. Both of these activities were key to bringing the national forests under management. Wildfire destroyed valuable timber resources, as did white pine blister rust. Because blister rust needed to be controlled on public lands to protect private lands from the disease, it made these practices of national significance. The legacy of this desire to protect the forests from insects, diseases and fire continues to impact forest development yet today (2004).

Projections of future wood consumption in the United States, along with estimates of wood production, indicated an increase in wood supply would be needed. This was the case during Pinchot's time and prevailed into the 1980s (USDA 1984). For example, in 1936 it was estimated that the United States used 48 billion board feet of timber but was only growing 32 billion board feet. The offered solution was to invest millions of dollars in acquiring additional areas as public forest, in fire protection, and in bringing denuded lands of the country into better condition for later crops (Toumey and Korstian 1947). The perception of a wood shortage in the United States was reinforced after World War II with the increased demand for home construction. The Forest Service was asked to meet this demand, especially by the timber industry. This was demonstrated by the passing of the Multiple Use and Sustained-Yield Act of 1960, which called for national forests to be used for recreation, watershed, and wildlife purposes and for harvest to be in balance with growth (Steen 1976). The view of a timber shortage continued as the annual net growth on commercial timberlands in 1984 was estimated at 21.7 billion cubic feet in the United States; but it was estimated that these lands could produce 32.8 billion cubic feet by 2030 (USDA 1984). Again, it was suggested that to meet the nation's growing demands for timber and timber products, large investments in silvicultural activities would be needed. Therefore, the management plans developed for the national forests throughout this period were generally timber management plans but often included a domestic livestock-grazing component, both critical elements of utilizing forests rather than preserving them. These management plans utilized concepts presented by Fernow in 1900 as the forests were divided into working circles, compartments, and sub-compartments. In each of these units timber resources were inventoried, timber growth estimated, and an allowable cut calculated to support a sustained yield of timber. Some of these plans went as far as to suggest that all lands within a working circle, both public and private, be regulated together to support the annual cut (USDA 1941).

During this period of expansion, 1910-1960, the Forest Service developed a tremendous work ethic and a "can do" attitude. Fires were vigorously suppressed and forest insect and disease epidemics were being addressed. Silvicultural practices and mensurational techniques to support these management plans rose to the challenge by developing planting, cleaning, thinning, fertilization, and harvesting methods to support high yield forestry (Baker 1934; Steen 1976; Smith and others 1997).

Forest Management Changes

Beginning in the 1960s and continuing in earnest in the 1970s, the public's perceptions and uses of the forests started to change. These changing views were supported by more and more knowledge that forests were more than crops to be grown and harvested (Spurr 1964). Forests provide an array of goods and services of which one of the most important is the protection and production of clean water. This fact was recognized by Theodore Roosevelt as one of the original reasons given for expanding the forest reserves (Gifford 1902). In the 1970s, these changing attitudes and beliefs of the role of forests in society were marked by the celebration of the first Earth Day in 1977. Also, this was a time in which significant laws were enacted such as the National Environmental Policy Act of 1969, the National Forest Management Act of 1976, and the Endangered Species Act of 1979 that impacted forest management. Individually and in combination these laws began to alter how the national forests were perceived and managed. In addition to these laws, air travel became more common during this era, which allowed the public to view forests from the air, disclosing the fragmented and artificial look that forests took on with the application of square harvest blocks and clearcutting used with high yield forestry (figure 3).

With these changing attitudes toward public forests and their use, silvicultural methods and concepts started to acknowledge other forest uses, in particular the production and maintenance of wildlife habitat. In 1981 the Society of American Foresters, in cooperation with the Wildlife Society,

Figure 3—View of clearcuts from the air showing the patchwork and fragmentation of forests.

published its monograph describing *Choices in Silviculture for American Forests* (Society of American Foresters 1981). Even though this text exemplified the benefits produced by forests including water production, wildlife habitat forage for livestock, aesthetic appeal, and recreation potential, the silvicultural systems described were very traditional and differed little from those described by Schlich in 1904. Similarly, *Silvicultural Systems for the Major Forest Types of the United States* (Burns 1983) approached silviculture in very traditional ways, producing traditional stand structures most often designed to produce timber products.

In 1988 guidelines were established for managing spotted owl habitat in the Pacific Northwest. These guidelines, and the listing of the spotted owl as threatened under the Endangered Species Act in 1990, changed the emphasis of forest management either directly or indirectly on nearly all lands administered by both the Forest Service and Bureau of Land Management (FEMAT 1993). Also during this time the prediction of timber shortfalls that had dictated forest management policies for decades was not materializing. From 1960 to 1985, the national forests met about 25 percent of America's softwood timber needs. This gave state and private stocks time to recover and it is estimated that 50 years from now, timber growing in the United States will be nearly double the levels in 1960 (Bosworth 2002).

Silviculture and Wildlife

Even though the conservation of spotted owl habitat was a novel forest management objective in many circles, the production and maintenance of wildlife habitat was not new to forestry. In addition to producing clean water, some of the original reasons for preserving and managing forests were the production of game animals for the aristocracies of Western Europe (Smith and others 1997). What became apparent in the desired forest conditions for wildlife was what remained was more important than what was removed in forest treatments. Instead of sustaining a flow of wood products from forests, the sustaining of forest processes, structures, and functions became more prominent as a reason to manage forests, even though much was not understood about these concepts and less was understood about how they could be sustained. From a silvicultural perspective a component of these concepts could be identified; that is stand and forest structures could be described as desirable for wildlife and possibly contain some other advantageous forest properties.

Thomas and others (1979) described successional stages of forests that played various roles in the life histories of wildlife species. These stages ranged from grass-forb to old growth and included composition, decadence, horizontal structure, vertical structure, and other elements important for wildlife. Oliver and Larson (1990) also described the development of forests using structural stages including stand initiation, stem exclusion, understory reinitiation, and old growth. Both of these classification systems concentrated on describing stands and forests and in particular what was left not what was being removed.

Reynolds and others (1992) used structural stage classifications to describe stand and forest habitat for the northern goshawk and its prey species for the forests of the southwestern United States. What were not included in the desired conditions for the goshawk were the preferred silvicultural methods to create and maintain these desired conditions. These desired

Figure 4—Ponderosa pine stand located in the southwestern United States illustrating the clumpy and irregular stand structure that is preferred goshawk habitat.

conditions were to be maintained over multiple spatial and temporal scales ranging form groups of trees to landscapes and over time periods exceeding 200 years (figure 4). "While superficially the recommendations by Reynolds and others 1992 were another example of narrow, single species focus, is in fact a coarse filter approach that includes a mosaic of age and structural classes to provide habitats and food chains for a broad spectrum of wildlife species including goshawk prey species… approximating the composition, structure, and landscape patterns existing in southwestern ponderosa pine forests before fundamental changes in natural disturbance regimes and forest structure"(Long and Smith 2000). The challenge for the art and science of silviculture was to use the knowledge gained over 100 years on treating forests to produce timber to use this to create and maintain desired conditions for goshawks and their prey. Some of the silvicultural concepts appropriate for goshawk habitat management include area regulation of desired conditions over large landscape units, free selection silvicultural systems (combining group and individual tree selection systems with reserve trees left in all structural stages), variable cleaning and weeding prescriptions, variable spacing in thinnings, coarse woody debris recruitment, and snag retention to name a few. This is far different from the "slick and clean" forestry advocated by Fernow in 1900.

Even though the public attitudes toward the value of forests and their management have changed, there continues to be a strong ethic "that the most important product of forest management is timber" resulting in timber management and silviculture being synonymous. Because the production and harvest of timber crops has been the primary objective of American silviculture for over 100 years, the association was inevitable. In addition, foresters felt comfortable with this objective and felt "good forestry" would result in strong, viable wildlife populations, clean water supplies, and ample recreational opportunities as a side benefit. Concerns about wildlife and aesthetics were reduced to constraints on timber management, such as the size and location of cutting areas and the minimum age of trees at the time of harvesting (Smith and others 1997). For the practitioners of silviculture or applied ecology to remain leaders in designing, prescribing, and implementing management systems, they need to be innovative, adaptable, open minded, and willing to partner with a range of other disciplines to sustain forests

Silviculture and Wildfire

Nowhere is this leadership and commitment of innovative silviculturists needed more than in designing forest management systems aimed at reducing the occurrence, intensity, and severity of wildfires (Graham 2003). Similar to creating and maintaining structures to produce wildlife habitats some of the same concepts apply to designing structures for affecting wildfire behavior and severity. In our desire to protect forests for human use, society has modified the structure, composition, and native processes occurring in many of our forests. Most evidence suggests, the dry forests dominated by ponderosa pine and Douglas-fir have undergone the most changes because of successful fire exclusion while the moist forests (western redcedar, western hemlock) and cold forests (lodgepole pine, Engelmann spruce, subalpine fire) were minimally impacted (Hann and others 1997). Like the methods used for producing wildlife habitat, what is left and its characteristics after treatment are important elements in designing stand and forest structures aimed at modifying wildfire behavior and severity.

Crown base height, number of fuel strata, surface fuels, fine fuels, coarse woody debris, hydrophobic soils, lower duff moisture, ladder fuels, crown bulk density, and fuel models are only some of the elements needed when designing vegetative treatments to modify the wildfire condition class of forests (Graham and others 1999; Scott and Reinhardt 2001; Robichaud and others 2000; Graham 2003). These elements are different than culmination of mean annual increment, normal stocking, yield capability, rotation age, net present value, rings per inch, Keen's tree classes, or site index that were common elements of many timber production silvicultural prescriptions (Smith and others 1997). However, the same basic understanding of climate, soil, forest development, silvics, succession, silvicultural methods (e.g., planting, tending, pruning, thinning), and so on used for the development of both timber and wildlife habitat prescriptions can be used to develop these critical fuel modification prescriptions. Most importantly, wildland fuels are composed of live and dead vegetation of which silviculture is the art and science of managing.

Change Is Often Difficult But Exciting

Silviculturists cannot be experts in all disciplines required for successful forest management. However, they need to have a basic understanding of these other disciplines. Not only is an understanding beneficial but also willingness and collaborative attitude are helpful when venturing into different and new management directions. Because of the long tradition of timber management and silvicultural systems associated with this management objective, it is easy to repackage the "tried and true" silvicultural methods and prescriptions into fuel management or wildlife emphasis prescriptions. For example, prescribe evenly spaced plantings, cleanings, and thinnings even though a clumpy or groupy nature of a forest may be desired. Similarly, through tradition, prescribe the removal of disease or insect susceptible trees even though they may be important elements of a functioning forest or desirable attributes for wildlife.

Nowhere on the landscape is innovation and imagination needed more from silviculturists than designing systems for managing stands within the urban interface. Most often people have a tremendous attachment to forests in these settings even though their very nature may threaten people's homes and lives if they burn (Kent and others 2003) (figure 5). Prescriptions in the urban interface usually necessitate the balancing of people's desires to live in a forest yet maintain conditions that reduce the risks of unwanted fire. Rarely will traditional silvicultural methods (e.g., seed tree, shelterwood) used for timber production produce and maintain the desired conditions in the urban interface.

Figure 5—In recent years management objectives aimed at reducing the intensity and severity of wildfires have become more common, especially in the urban interface.

The size of wildfires and the number of acres burned by wildfires has been increasing in recent years after declining for several decades (Agee 1993, Graham 2003). These areas (Bitterroot-Montana, Hayman-Colorado, Biscuit-Oregon, Rodeo-Chediski-Arizona) provide tremendous challenges for silviculturists in prescribing treatments to restore these forests. Many of these fires burned large areas destroying native seed sources, which makes planting of site-adapted seedlings challenging but imperative. The introduction of exotic plants (e.g., cheatgrass) can alter successional pathways and make the restoration of native vegetation uncertain. Similarly, because of uncharacteristically severe fires, soil properties can be altered to increase soil erosion and reduce site productivity, again increasing the challenges silviculturists face in addressing the conditions left after wildfires (Robichaud and others 2000). Depending on the type of forest burned, large amounts of standing and down woody material is often left after wildfires (Brown and others 2003, Graham 2003). In some circumstances this material has commercial value that can help pay for fire restoration efforts, but silvicultural systems need to be designed to ensure the integrity and long-term future of the forest. The above are only some of the issues in which the silviculture and fire disciplines must work collaboratively to address.

Silvicultural Legacy

Silviculturists can be extremely proud of what the discipline accomplished in the last 100 years. Through their leadership and innovation the timber famine projected for many decades never materialized. Within the Forest Service, silviculturists set the standard for continuing education and the application of science-based practices in land management, a standard which other disciplines try to emulate. Beginning with the aristocracies of Europe, the importance of forests in maintaining wildlife and water along with timber resources was recognized, and silviculturists such as Schlich (1904) provided silvicultural methods and principles applicable for meeting these management objectives. These same principles can be applied to present management objectives such as reducing the risk of severe and intense wildfires, or future unknown objectives. Most importantly, silviculturists need to be the champions of maintaining forest integrity and resiliency no matter the forest setting or the management objectives presented. No other discipline has the understanding, legacy, or long-term view necessary to design and prescribe forest management activities in the 21st century.

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A Collaborative Fire Hazard Reduction/ Ecosystem Restoration Stewardship Project in a Montana Mixed Ponderosa Pine/Douglas-Fir/ Western Larch Wildland Urban Interface

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Abstract—*Forest Service managers and researchers designed and evaluated alternative disturbance-based fire hazard reduction/ecosystem restoration treatments in a greatly altered low-elevation ponderosa pine/Douglas-fir/western larch wildland urban interface. Collaboratively planned improvement cutting and prescribed fire treatment alternatives were evaluated in simulations of disturbance processes and interactions with the partially restored wildland urban interface conditions. The SIMPPLLE modeling system was used to reconstruct historic landscape conditions across a broad range of fire regimes and to model future landscapes that reduce fire severity, restore wildlife habitats, reduce bark beetle severity; and disclose environmental effects.*

Introduction

The Frenchtown Face, on the Ninemile Ranger District of the Lolo National Forest, is a south to southwest facing landscape approximately 15 miles west of Missoula, Montana (figure 1). The 96,381 acre landscape is comprised of the Lolo National Forest (45 percent), private ownerships (27 percent), Plum Creek Timberlands (25 percent), and Montana Department of Natural Resources and Conservation land (3 percent). The landscape character is integral to the rural community settings of Frenchtown and Huson, located on the southern edge of the project boundary.

Figure 1—Frenchtown Face topography.

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Roughly one-third of the landscape is considered benchlands that gradually rise in elevation from the Clark Fork River to the toe of the steeper mountain slopes along the Ninemile Fault. The benchlands are characterized by open grassland, agricultural land, and/or residences within the forest that make up the wildland urban interface zone. The forested residence benchland areas consist mainly of ponderosa pine/Douglas-fir habitat types with inclusions of western larch.

In 1992 the Lolo National Forest implemented a landscape approach to ecosystem management: management for healthy and sustainable communities and landscapes, and management for sustainable human values, uses, and populations. Using this approach, 15 landscapes, or ecosystem management areas, of the Ninemile Ranger District were prioritized by restoration needs. The highest priority for restoration were landscapes containing the greatest amount of low-elevation warm forest habitat types characterized by low intensity, frequent fire regimes. Frenchtown Face became the fourth major project addressing this approach.

In March 2000, an additional landscape analysis highlighted the need to restore the landscape components of composition, structure, and function to near presettlement times. The new analysis pointed out the need for:

- fuel reductions in wildland urban interface and upland forests;
- improved forest health;
- reductions of insects and diseases from abnormally elevated risk levels;
- improved big game winter range;
- enhancement and recruitment of old growth forests; and
- meeting Lolo Forest Plan expectations in recreation and aesthetic scenery values.

Current Landscape Conditions

Restoration of ponderosa pine forests to landscapes resembling presettlement times has become a necessity due to the current upward density trends of small diameter trees along with higher fuel loading levels (Bonnicksen and Stone 1982; Chang 1996; Parker 1984; Parsons and DeBenedetti 1979). Fire suppression, historic grazing, timber harvesting, and climatic changes have all played a role in the upward trends of density and fuel loadings within the Frenchtown Face restoration project area (Arno and others 1997; Parsons and DeBenedetti 1979; Skinner and Chang 1996). The probability of high severity wildfire and deterioration of ecosystem integrity have increased on the landscape (Dahms and Deils 1997; Patton-Mallory 1997; Stephens 1998; Weatherspoon and Skinner 1996). This deterioration is similar to conditions reported in the Blue Mountains of Oregon and Washington (Everett 1993), the Columbia River Basin (Quigley and Cole 1997), and the Sierra Nevada Ecosystem Project (SNEP 1996, Weatherspoon and Skinner 1996). All of these have highlighted the need for large-scale, strategically located small tree thinning, fuel treatment, and use of prescribed fire (McIver and others 2001).

The dense, young ponderosa pine/Douglas-fir forests that occupy the low elevation areas of the Frenchtown Face are substantially different from historic ponderosa pine stands as a result of fire suppression. Several wildlife species are at risk as a result. The goals of the Frenchtown Face project include restoring habitat for those species. Wildlife species in the Frenchtown

Face area include those typical for the Northern Rockies. Species of special interest due to their sensitive, management indicator, or federally listed status include pileated woodpeckers, flammulated owls, northern goshawks, mule deer, elk, wolves, American martens, fishers, wolverines, and Canada lynx.

Along with wildlife habitat restoration, invasive weed mitigation is a major component of the project. Invasive weeds are abundant in much of the low elevation portions of the Frenchtown Face. Weeds can substantially reduce the forage productivity for wintering deer and elk (USDA 1999). Weeds have a competitive advantage over native plants and are shade-intolerant and disturbance-dependent, which complicates the restoration of frequent, fire-dependent forests.

Historic Landscape Conditions

The historic range of variability (HRV) encompasses a large temporal range that produced ecological conditions that were sustainable over a long time frame. The HRV attempts to describe the ecosystems prior to influences from European descendents. Human influences are considered a part of the natural condition. The HRV was developed from several sources: findings of the Interior Columbia Basin (USDA 1997); Fischer and Bradley (1987); Losensky (1993); a fire history study (Losensky 1989) within the analysis area; and SIMPPLLE simulations.

Two vegetation groupings used in this project are: (1) habitat type groups (HTG) as used in the Lolo Forest Plan (April 1987); and (2) fire groups (FG) (Fischer and Bradley 1987). Only the habitat types that comprise the warm, dry lower slopes are a focus of this project. These areas represent 61 percent of the project area.

Warm-Dry Forest Vegetation of Lower Slopes

Historical conditions perpetuated seral forests of ponderosa pine and western larch in association with Douglas-fir and, in some instances, lodgepole pine. The dry benchlands at low elevations during presettlement were typified by open grown stands of old growth ponderosa pine of large sawtimber size (Losensky 1993). Frequent low intensity fires kept litter and slash accumulations very low, brush species were less common than present day and more succulent, and Douglas-fir was a minor component of the forests. Fire thinned saplings, removed Douglas-fir thickets, and caused pitching of tree boles, which created long-standing snags. Stand replacement events were rare. Tree mortality was largely in the form of small pockets of windthrow, root disease, or bark beetle activity. These small openings were soon regenerated by ponderosa pine and Douglas-fir with ponderosa pine being favored by frequent fire.

Adjacent toe-slopes are characterized as warm-dry to warm-moist Douglas-fir, grand fir, ponderosa pine, and larch. Associated firegroups 4, 6, and 11 characterize these environments. A tendency toward overstocking and development of the dense understories increase the hazard of stand-replacement fires on these sites.

On the north sides of these ridges, it was not uncommon for Douglas-fir to dominate all stages of succession. Ponderosa pine, larch, and lodgepole pine are seral components whose abundance varies by habitat type phase. Figure 2 displays the current extent of the dominant cover types. Figure 3

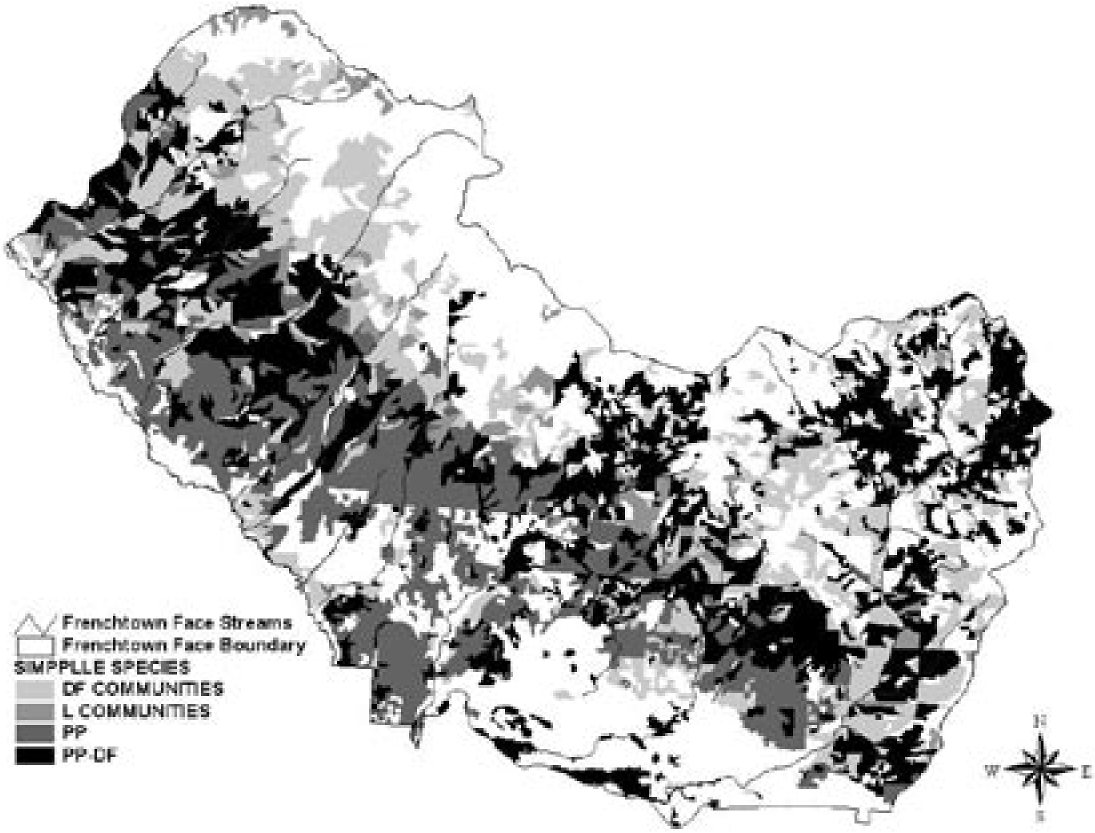

Figure 2—Current ponderosa pine, ponderosa pine/Douglas-fir, larch, and Douglas-fir cover types within Frenchtown Face.

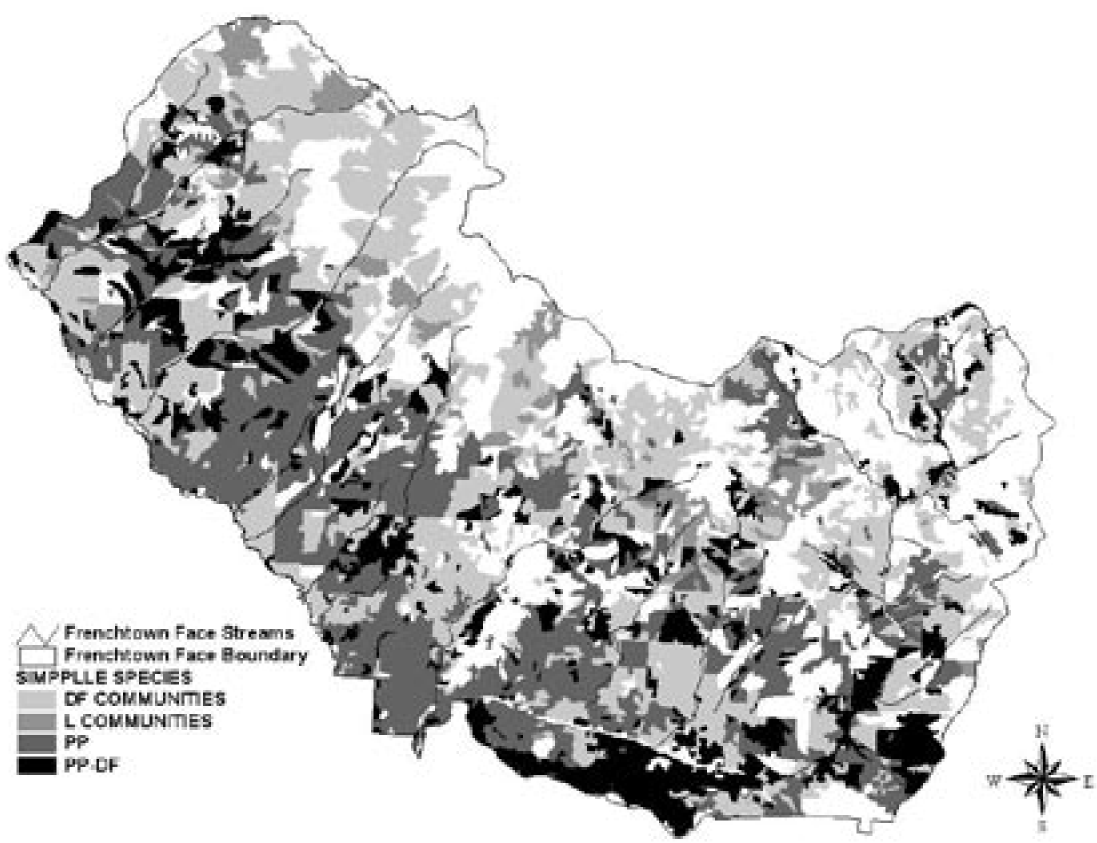

Figure 3—Historic representation of ponderosa pine, ponderosa pine/Douglas-fir, larch, and Douglas-fir cover types produced by SIMPPLLE simulations.

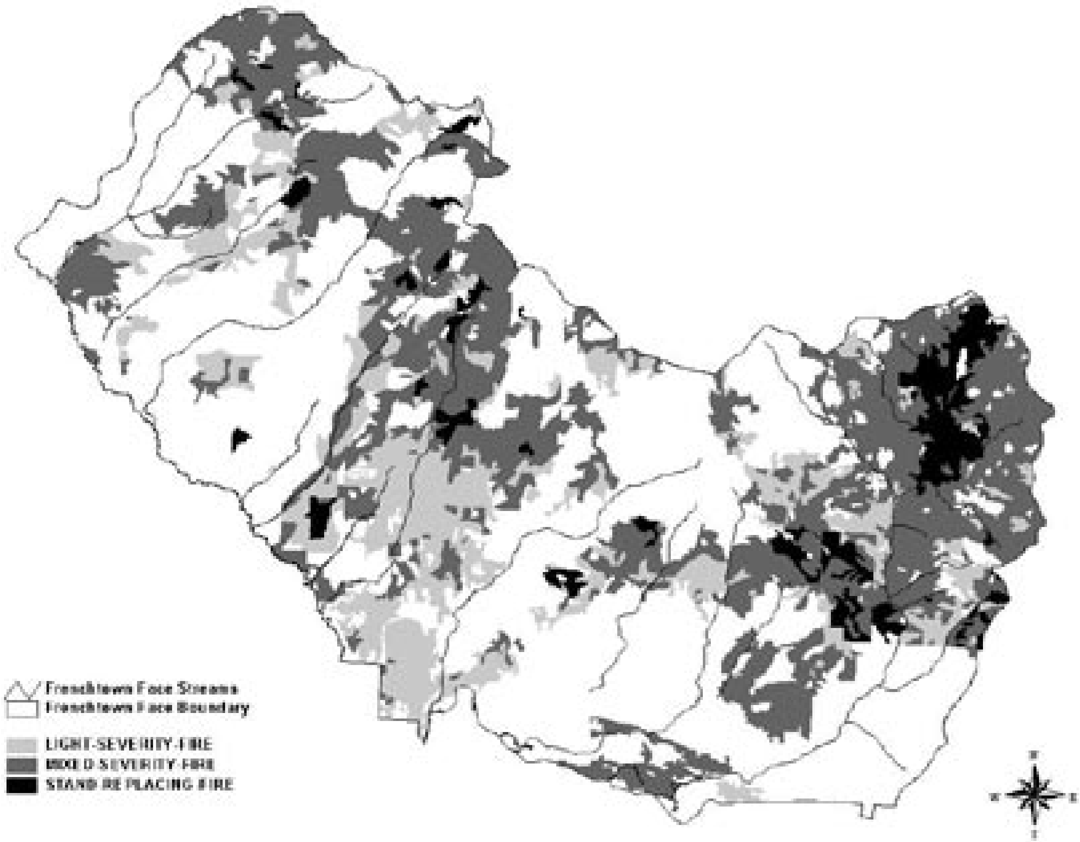

Figure 4—Simulated occurrence of light-severity, mixed-severity, and stand-replacing fire on a historic representation of the Frenchtown Face landscape for a one-decade period.

displays a representation of historic cover types produced by SIMPPLLE simulations. Stand-replacing fire hazard like the adjacent toe-slopes tendency increased on these sites due to dense understory vegetation. A wide array of forest structures and compositions within the natural fire regime are possible (figure 4). Stands tended to be evenly distributed over the various age classes with 30 percent less than 40 years of age and 35 percent old growth (Losensky 1993).

In firegroups 4 and 6, large diameter snags occurred at low densities (Ritter and others 2000) and provided nest habitat for pileated woodpeckers and flammulated owls (McClelland 1977, Wright 1996). The relatively open understories provided flammulated owls opportunities to forage using a combination of drop pouncing and hawk gleaning behavior on moths and grasshoppers (Wright 1996). Frequent, non-lethal wildfires repeatedly scarred ponderosa pines. This resulted in cumulative pitch build-up that made those trees very rot-resistant after they died, resulting in snags that stood for very long periods of time (Smith 1999). Low-to-moderate stocking and frequent non-lethal underburns resulted in a high forage productivity of understory shrubs and grasses, which provided forage for high populations of wintering mule deer and elk (Hillis and Applegate 1998). These open forests also provided excellent foraging habitat for northern goshawks (Clough 2000), although the stands were generally too open for nesting. The small percentage of old growth that remains has dense, continuous understories which preclude successful foraging by flammulated owls (Hillis and others 2002). While small diameter snags are abundant, they lack the high pitch content of trees that are exposed to frequent fires, and thus have little durability after death. Mule deer and elk have been largely replaced by white-tailed deer. There has been an increase in songbirds, such as vireos and Townsend's warblers, that occupy dense forests (Hutto and Young 1999).

Collaborative Process

A community-based purpose and need, and public-recommended proposed actions for the Frenchtown Face project, were formulated through a series of public meetings

These meetings formed the basis of the environmental analysis and formal public scoping process under NEPA (National Environmental Policy Act). An underling premise of this approach is that formal public participation in the development of a proposal will lead to a more efficient and less contentious environmental analysis and project decision.

Figure 5 represents the expanded NEPA sequence process including the steps taken in collaborating with the public.

Participation was fairly broad with a cross section of local residents, forest industry, State agencies, rural fire department, and media. Separate, concurrent meetings were held with local environmental group representatives who declined to attend public meetings. Public values were expressed as purpose and need statements by the interdisciplinary team and then validated by the public at subsequent meetings.

The public identified a need for coordinated block management of noxious weed treatments, environmental education in schools, historic site interpretation, increased communication through the formation of interest groups, and enforceable decisions, e.g., road closures. Environmental group participation resulted in a reduced magnitude, or area, of timber harvest restoration treatments, and the creation of three alternatives: 2, 3, and 4.

During formal scoping, the public and environmental groups responded with issues and concerns to the proposed action. The interdisciplinary team used these responses to formulate draft alternatives. The draft alternatives were then presented at public meetings for additional feedback and adjustment.

Restoration Treatments

Ecological sustainability requires the restoration of process as well as structure (Stephenson 1999, Arno 1996). Fire regimes and stand structures

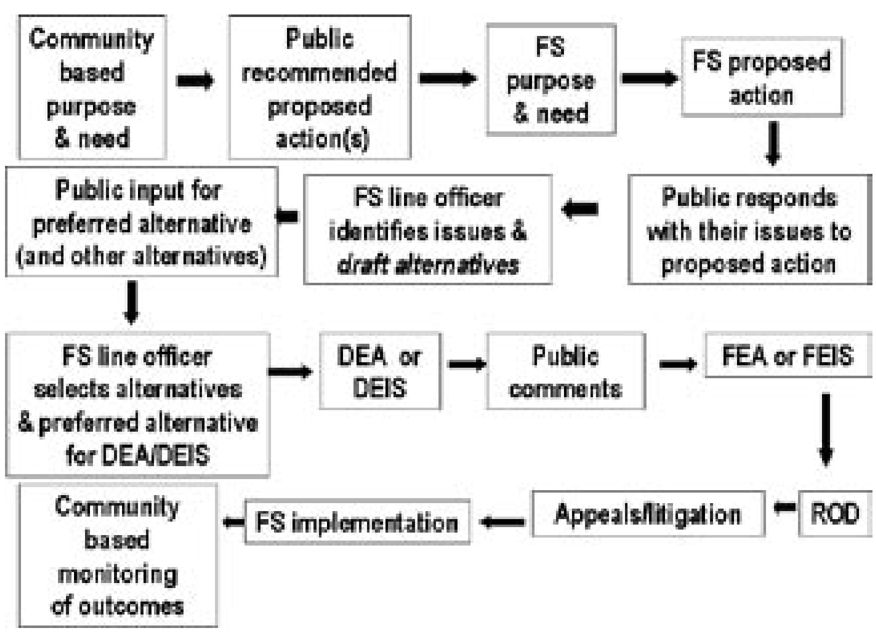

Figure 5—Frenchtown Face NEPA sequence.

interact and must be restored in an integrated way. Fire alone may be too imprecise or unsafe in many settings, so a combination of treatments may often be the safest and most certain restoration approach (Allen 2002). A recent wildland urban interface fuel reduction study (Scott 1998) conducted on the Ninemile Ranger District to compare thinning treatments found the most effective treatment was a thinning from below to a basal area of 76 ft^2/acre followed with prescribed fire (similar to the proposed action). And that periodic application of the treatment would lead to an open-structured forest of large trees with high aesthetic value.

Three recent restoration projects on the Ninemile Ranger District treat low-elevation ponderosa pine/Douglas-fir forests of frequent low intensity fire regime in a similar fashion as the Proposed Action, Alternative 2 (i.e., Starkhorse, Petty Rock, and Sawmill-Cyr). Single tree selection retaining a residual basal area of 30 to 60 ft^2/acre thinned stands from below, cutting excess understory trees and thinning excess crowns in the overstory to partially restore historic structure. The harvesting was followed with understory prescribed burning to partially restore historic ecological processes. The average harvest volume of these three projects was 3 MBF/acre with 49 percent of the volume coming from cut trees less than 12 inches DBH, 45 percent from cut trees 12 to 19 inches DBH, and 6 percent from trees over 19 inches DBH.

Through our collaborative process, a total of five alternatives were developed.

No Action - Alternative 1

Under the No Action alternative, no new actions would be implemented.

Proposed Action - Alternative 2

The Proposed Action provides for improvement cutting and underburning on gentle slopes under 35 percent in the warm-dry sites found on the benchlands; ecosystem maintenance burning on sites not feasible for improvement cutting or on steep slopes or high risk weed sites; decommissioning of roads; aerial and ground spraying of noxious weeds; and a host of recreation and interpretation activities. Reducing the stocking to a range of 70-100 BA would increase a stand's survivability of fire under normal burning conditions and provide greater growth and resistance to insect outbreaks. A recent wildland urban interface fuel reduction study (Scott 1998) conducted on the Ninemile Ranger District to compare thinning treatments found the most effective treatment was a thinning from below to a basal area of 76 ft^2/acre followed with prescribed fire (similar to the proposed action). And that periodic application of the treatment would lead to an open-structured forest of large trees of high aesthetic value. A stocking of 70-100 BA, however, was needed to avoid substantially increasing the risk of spreading noxious weeds. The 70-100 stocking level was a recognized compromise to meet mutually exclusive public needs.

Alternative 2, But With a 12-inch Diameter Limit - Alternative 3

This alternative places a 12-inch diameter cut limit on the improvement cutting of Alternative 2. This alternative was based on the Environmental Group collaboration and the aversion to cutting large diameter trees on national forestland. Approximately 78 percent of Alternative 2 timber harvest treatments would be feasible under a 12-inch DBH limitation. Feasible treatment locations have at least 3,000 board feet (MBF) (Barbour 2001 used 2.7MBF/acre) of excess stocking between 7 inches (minimum sawlog size) and 12 inches DBH (example: 38 cut trees per acre averaging 10 inches DBH represent 20 ft^2 of basal area and 3 MBF). Using stewardship contract revenues, additional area could be treated manually and/or mechanically to remove excess trees.

No Commercial Timber Harvest - Alternative 4

This alternative differs from Alternative 2 in that all commercial timber harvests are dropped. Prescribed fire is still used.

Modified Proposed Action - Alternative 5

This alternative builds on Alternative 2 by adding improvement cutting to high weed risk sites on gentle terrain and adding improvement cutting on steep slopes to enhance a portion of the existing old growth stands.

Table 1 compares the alternatives. "Improvement cutting" (IMP) consists of both thinning from below and crown thinning to remove excess stock of merchantable-sized trees (7 to 19 inches DBH) with a target residual basal area of 70 to 100 ft^2/acre. Shade-intolerant (seral) ponderosa pine and western larch trees are favored for retention though not to the exclusion of shade-tolerant Douglas-fir. "Mechanical" (MECH) is a combination of noncommercial understory fuel reduction treatments including slashing by hand using chainsaws followed by handpiling and burning of the handpiles where smoke from underburning would be unacceptable to the surrounding residences. "Underburning" (UB) is ecosystem maintenance burning following the improvement cutting or other silvicultural systems. A spring burn removing portions of the duff and litter, down fuels, understory Douglas-fir seedlings and saplings, and aboveground segments of associated understory flora. "Improvement cutting and group tree selection" (IMPGT) is group tree selection occurring on 10 percent of the area, in scattered small one-quarter-acre to 2-acre patches of seed tree or shelterwood-like cutting. "Slash and EMB" is noncommercial hand felling of excess understory (slashing) to augment fuel conditions for the subsequent ecosystem maintenance burn (EMB) or to simply ensure that unwanted excess understory seedlings and saplings are removed. "Thin" is commercial thinning of western larch stands that contain some Douglas-fir and ponderosa pine. Underburning is planned after these harvests. "Shelterwood (SW) with reserves" is proposed to replace heavily root disease infested Douglas-fir stands with planted non-host ponderosa pine.

Table 2 shows the restoration projects associated with the alternatives.

Table 1—Comparison of harvests and prescribed fire in alternatives.

Treatment	SIMPPLLE equivalent treatment	ALT 2	ALT 3	ALT 4	ALT 5
IMP+MECH	Ecosystem management thin & underburn	152	152		152
IMP+UB	Ecosystem management thin & underburn	2602	2382		337
ITS+UB	Ecosystem management thin & underburn				3242
IMPGT+UB	Ecosystem management thin & underburn	493			599
MECH	Ecosystem management thin & underburn	387	387	539	364
SW+UB+P	Shelterwood cut w/ reserves & plant	41			41
Slsh+EMB	Ecosystem management underburn	6829	7583	10104	5727
Thin	Ecosystem management thin & underburn	139	139		139
Total acres		**10643**	**10643**	**10643**	**10624**

Table 2—Frenchtown Face restoration projects associated with the alternatives and planned to be funded through stewardship projects or other appropriations.

	Proposed Action Alternative 2		Modified Proposed Action Alternative 5	
Stewardship funded activity	**Proposed**	**Likely funded through stewardship**	**Proposed activity**	**Likely funded through stewardship**
Road construction				
Long-term road	0 miles	0 miles	0 miles	0 miles
Short-term road	5.24 miles	5.24 miles	5.94 miles	5.94 miles
Road reconstruction	56.21 miles	56.21 miles	65.82 miles	65.82 miles
Road obliteration	17.68 miles	17.68 miles	22.91 miles	22.91 miles
Road decommissioning	76.8 miles	24.7 miles	114.7 miles	114.7 miles
BMP implementation	65.3 miles	43.77 miles	66.69 miles	66.69 miles
Culvert removal/replacement	19	2	19	15
Little McCormick Cr. stream restoration	0.5 miles	0.5 miles	0.5 miles	0.5 miles
Stony Cr. diversion restoration	0.5 miles	0 miles	0.5 miles	0 miles
Mule pasture/riparian fencing	0.25 miles	0 miles	0.25 miles	0 miles
Weed treatment	6100 acres	6100 acres	6100 acres	6100 acres
Recreation				
Mountain biking trail	0.25 miles	0 miles	0.25 mile	0 miles
Horse trail reconstruction	1.5 miles	0 miles	1.5 miles	0 miles
Dev. parking area	2	0	2	0
Parking area-update/improve	8	0	8	0
OHV trailheads	2	0	2	0
OHV trail	0.5 miles	0 miles	0.5 miles	0 miles
Education				
Signs	3	2	3	2
OHV curriculum	1	1	1	1
Student Monitoring Program-dev.	1	1	1	1

Comparison of Alternatives

Alternatives 3 and 4 were eliminated from the detailed study after a closer assessment. Attempting to prescribe-burn overly dense sawlog-sized live stands to meet the purpose and need is impractical without first removing "excess" trees (Allen 2002). Both Alternative 3 and 4 result in an accumulation of basal area over time (Barbour 2001) as trees 12 inches DBH and larger are never removed by timber harvest (Alt 3) and most trees over 5 inches DBH are never removed by prescribed fire (Alt 4). These alternatives create and maintain densely stocked stands of uniform-sized trees that have a high risk of bark beetle infestations (Barbour 2001) and fail to restore forest health or reduce the risk of stand replacement wildfires (Fiedler 2001). Sites with mechanical fuel treatment appear to have more dramatically reduced fire severity compared to sites with prescribed fire only. Forests with much lower density and larger trees have less continuous crown and ladder fuels, higher crowns off the ground, and thicker bark resulting in lower potential for crown fire initiation and propagation and for less severe fire effects (Pollet 1999).

The comparison of alternatives utilized simulations by SIMPPLLE.

The relatively small area treated under restoration timber harvests provides little distinction between alternatives (see table 1), including the No Action alternative, Alternative 1, on a landscape basis as reflected in SIMPPLLE simulations. There are no significant differences in simulated processes such as bark beetles, root disease, and fire, between alternatives at the landscape level. Figure 6 displays the level of fire that is simulated to occur with the alternatives and no treatment.

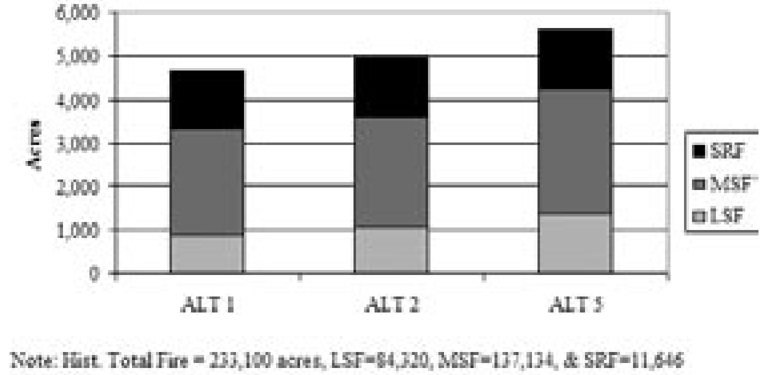

Figure 6—Acres of stand replacement fire (SRF), moderate severity fire (MSF), and low severity fire (LSF) simulated over 50 years by alternative.

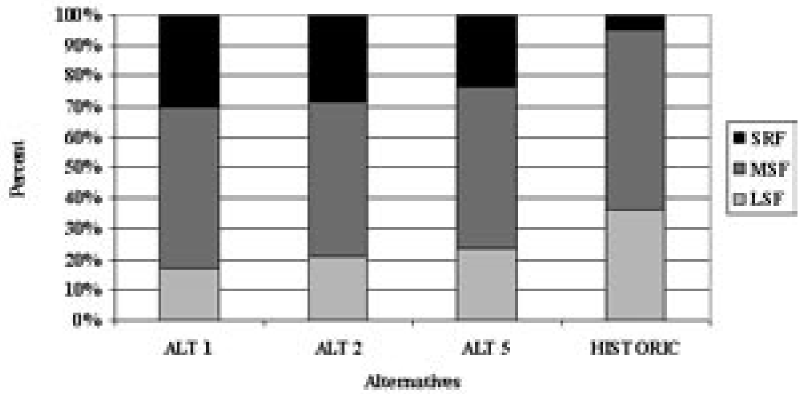

Figure 7—Distribution of stand replacement fire (SRF), moderate severity fire (MSF), and low severity fire (LSF) simulated over 50 years by alternative compared to the simulated historic condition.

Figure 7 displays the distribution of fire types between the alternatives and the historic representation created by SIMPPLLE. The two alternatives display very light gains toward the distribution modeled to be the historic representation.

Table 3 displays the area of restoration timber harvests by alternative.

Figures 8 and 9 display a slight shift in both ponderosa pine and ponderosa pine/Douglas-fir cover types toward the simulated historic conditions.

Table 3—Restoration timber harvest acres under each alternative shown as a percentage of: (1) warm, dry benchlands on national forest lands; (2) total national forest lands in the analysis area; and (3) the entire landscape across all ownerships.

Alternatives	Restoration timber harvest acres	Warm-dry benchlands National Forest lands	Total National Forest lands	Entire landscape
No Action	0	0%	0%	0%
ALT 2 – Proposed Action	3,405	11.7%	7.4%	3.5%
ALT 5 – Mod. Proposed Action	4,530	15.6%	9.8%	4.7%

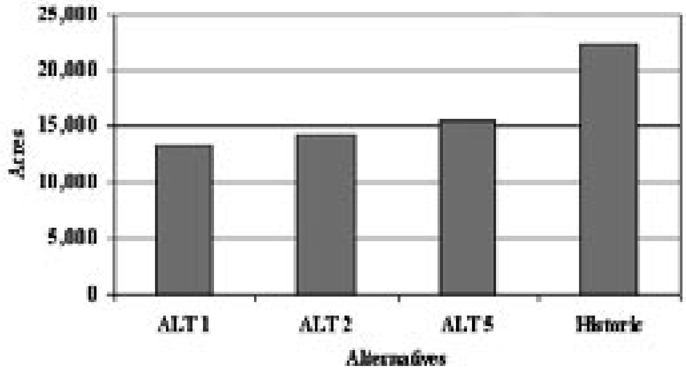

Figure 8—Post treatment ponderosa pine acreage in the entire Frenchtown Face landscape simulated over 50 years by alternative compared to the simulated historic condition.

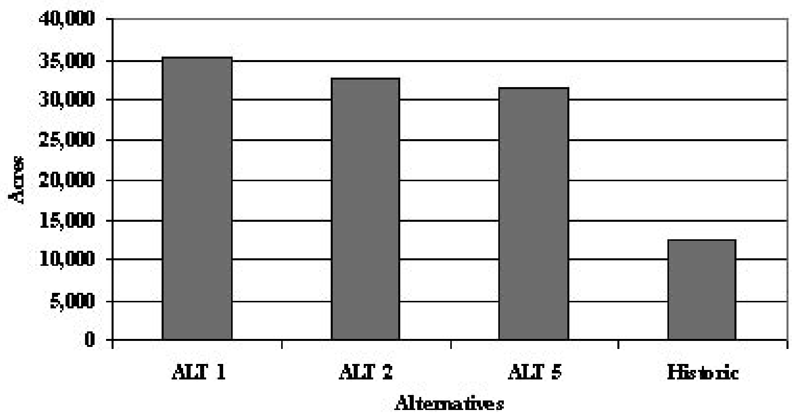

Figure 9—Post treatment ponderosa pine/Douglas-fir acreage in the entire Frenchtown Face landscape simulated over 50 years by alternative compared to the simulated historic condition.

However, very little change in density is made toward the historic condition with either alternative, as can be seen in figures 10 and 11.

Although major differences may not exist on a total landscape scale as a result of the alternatives, significant differences do exist between alternatives at very specific locations within the wildland urban interface in comparison with untreated conditions.

Since all of these alternatives treated a small portion of the total landscape, SIMPPLLE simulations were made increasing the magnitude of treatment by three-fold to help identify the level of treatments needed to have an impact on the total landscape.

A comparison of the simulated acres of fire spread from a single "locked-in" mixed severity fire was made between the original treatment acres and a tripled treatment acres. The simulations were made using average conditions with no extreme fire probability and no fire suppression. The tripled treatment acres had slightly fewer simulated fire acres. Tripling treatments and locking in a mixed-severity fire with extreme conditions, wind-driven, on the Frenchtown Face showed a dramatic difference in the amount of fire spread received from one locked in fire. Figure 12 represents the difference between tripling versus original acreage treated in Alternative 2, the proposed action, and Alternative 5, the modified proposed action.

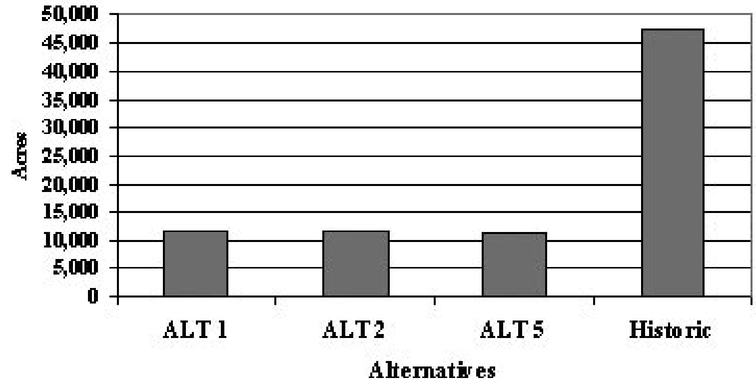

Figure 10—Post treatment acreage with 15-39% canopy coverage in the entire Frenchtown Face landscape simulated over 50 years by alternative compared to the simulated historic condition.

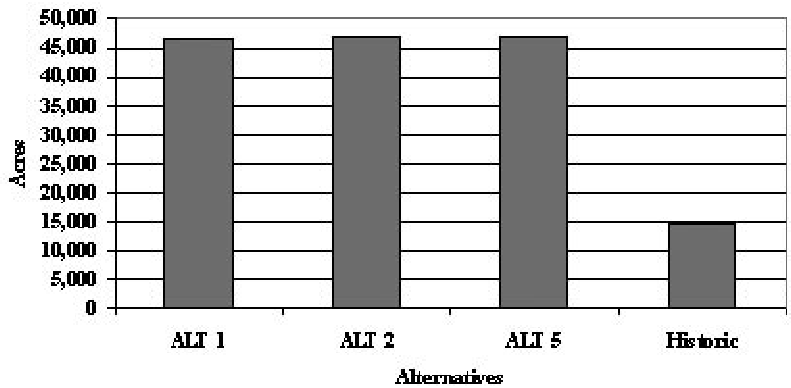

Figure 11—Post treatment acreage with 40-69% canopy coverage in the entire Frenchtown Face landscape simulated over 50 years by alternative compared to the simulated historic condition.

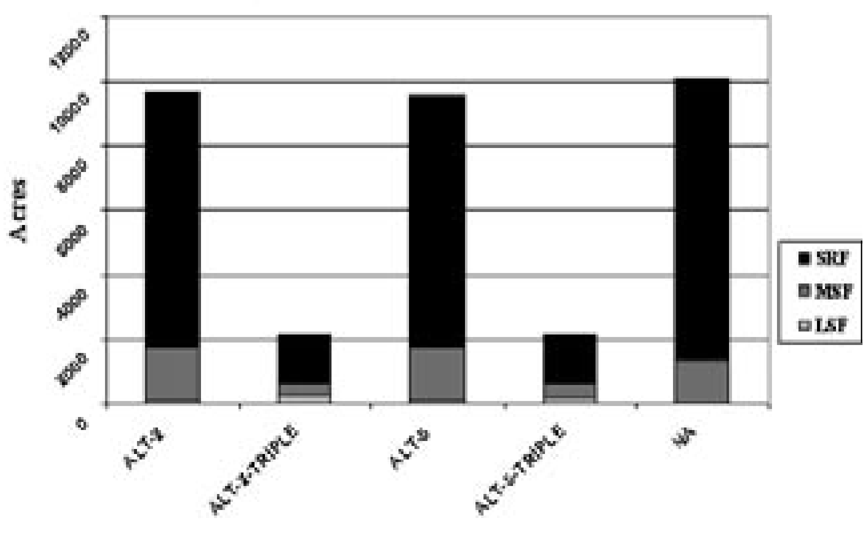

Figure 12—Comparison of simulated fire spread by fire severity for each alternative and the spread and severity of the same fire occurring when the treated area is tripled.

Project Status

The success of the collaborative process is not yet fully evident as the Draft Environmental Impact Statement (DEIS) and associated public comment period has not occurred. The Final Environmental Impact Statement (FEIS) and Record of Decision (ROD) are scheduled for completion and publication in May 2004. The level and content of public comment to the DEIS and subsequent appeals and litigation of the respective FEIS and ROD will provide the remaining evaluation of this collaborative process.

Comparisons from SIMPPLLE provide the agencies and the public excellent opportunities to discuss many questions. SIMPPLLE demonstrated that increasing the magnitude of treatment by three-fold would have increased the odds substantially that young and old growth stands would survive severe events. This helps to address questions as: (1) How much treatment is needed to substantially reduce the risk of stand-replacing fire? (2) For species-at-risk like flammulated owls, how much treatment across the landscape is needed to turn the habitat trend into a positive direction? (3) How much treatment is acceptable given the quantified risks of not treating those landscapes recognizing the inevitable consequences? SIMPPLLE provided landscape level and stand level significance in comparing the alternatives. The SIMPPLLE model provided an improved method of describing the range of historic variability across all ownerships.

The area feasible for restoration using commercial timber harvest (4,874 acres under Alternative 5) is typically a small percentage (18 percent of the landscape warm-dry type) under second growth forest conditions. SIMPPLLE provided landscape level and stand level significance in comparing the alternatives. The SIMPPLLE model provided an improved method of describing the range of historic variability across all ownerships. Since 1992, when the Lolo National Forest implemented a landscape approach to ecosystem management, just 4,365 acres of restoration timber harvests have been implemented on the Ninemile Ranger District. This represents just 2.7 percent of the warm-dry habitat type (163,339 acres) on the District. Alternative 5 essentially doubles the total area treated by restoration timber harvests, for a combined total 5.5 percent of the district's area. Presently, no other landscape scale restoration projects using timber harvests with prescribed fire are funded for analysis. A similar level of restoration accomplishment exists for using prescribed fire in these warm-dry habitat types where the district program struggles to complete approximately 2,000 acres of ecosystem maintenance burning annually, treating about 6 percent of the warm-dry type since 1992.

The public more readily accepts restoration projects involving timber harvest to enhance wildlife habitat than projects driven by commodity-extraction. In similar restoration projects, analysis has disclosed that treating a landscape with improvement cutting and underburning has protected and recruited old growth habitat, to the benefit of such species as flammulated owls and pileated woodpeckers. While the literature supports such findings (Hillis and others 2000), further quantification has been lacking. Using SIMPPLLE provides further quantification of the risk to survivability that any timber stand has for the long-term. For instance, SIMPPLLE demonstrated that Alternative 5 still carries substantial risk that much of the warm-dry portion of the landscape could lose young and old stands to stand-replacing fire during extreme wildfire conditions. SIMPPLLE also demonstrated that increasing the magnitude of treatment by three-fold would increase the odds substantially that young and old growth stands would survive extreme wildfire conditions. Such comparisons provide agencies and the public excellent opportunities to be involved in dialogue about issues such as: (1) what amount of treatment is needed to affect a positive wildlife habitat trend and (2) how do treatments compare given the quantified risks of taking no action.

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Silvicultural Activities in Pringle Falls Experimental Forest, Central Oregon

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Abstract—*Pringle Falls Experimental Forest has been a center for research in ponderosa pine forests east of the crest of the Cascade Range since 1931. Long-term research facilities, sites, and future research opportunities are currently at risk from stand-replacement wildfire because of changes in stand structure resulting from past fire exclusion. At the same time, many of the special values are increasingly at risk from recreational impacts and nearby urban development. We describe the special values associated with ponderosa pine (*Pinus ponderosa *Dougl. ex Laws.) stands in the Experimental Forest, we present our rationale for a series of treatments, and we discuss implementation of a set of silvicultural prescriptions designed to protect and enhance the special values of Pringle Falls Experimental Forest.*

Introduction

Experimental forests and ranges constitute a national network of outdoor laboratories designated by the U.S. Department of Agriculture, Forest Service for the express purpose of providing sites for research. They have a rich legacy of providing information to guide forest management activities. Experimental forests provide a unique research platform from which to address forest management questions at various scales and offer an important advantage for collaborative research. Pringle Falls Experimental Forest (hereafter Pringle Falls) (lat. 43°42' N, long. 121°37' W), within the Deschutes National Forest in central Oregon and 48 kilometers (30 miles) southwest of Bend, Oregon (figure 1), is a center for silviculture, forest management, and insect and disease research in ponderosa pine (*Pinus ponderosa* Dougl. ex Laws.) forests east of the Oregon Cascade Range. The 4477-hectare (11,055-acre) experimental forest is maintained by the Pacific Northwest Research Station, in cooperation with the Pacific Northwest Region and Deschutes National Forest, for research in ecosystem structure and function and demonstration of management techniques.

Pringle Falls is the oldest experimental forest and the site of some of the earliest forest management and silviculture research in the Pacific Northwest. Thornton T. Munger, first Director of the Pacific Northwest Research Station (then Experiment Station), and colleague and long-time friend of Gifford Pinchot, first Chief of the Forest Service, selected the site in 1914. It was formally established as a unit of the national network of experimental forests on May 20, 1931. Headquarters buildings were constructed between 1932 and 1934. Within Pringle Falls lies a two-unit Research Natural Area, established for non-manipulative research in 1936.

The eastside forests of Oregon are replete with special places ranging from mountain peaks shaped by fire and ice to rivers cutting through lava

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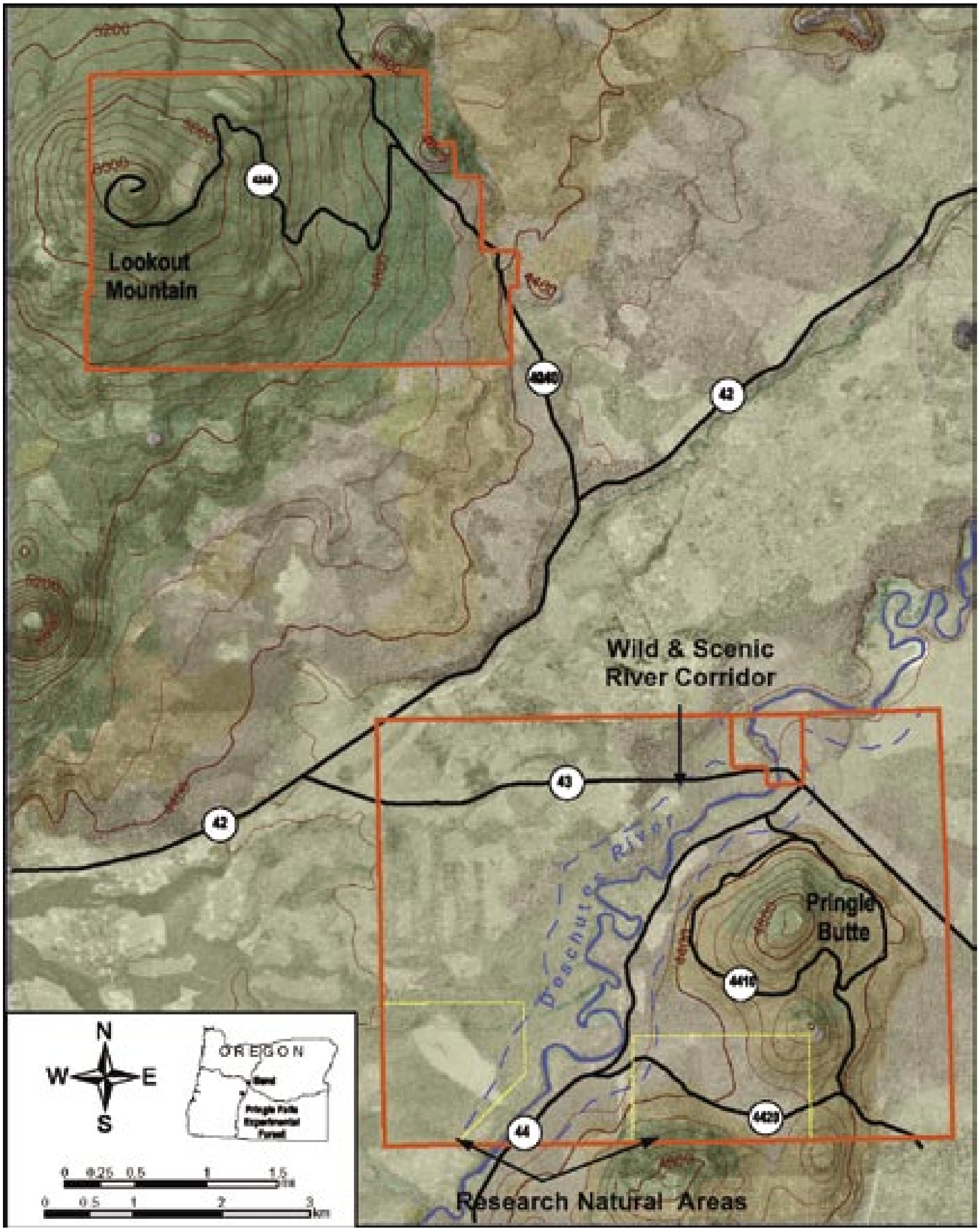

Figure 1—The Lookout Mountain and Pringle Butte Units of Pringle Falls Experimental Forest, located on the Deschutes National Forest southwest of Bend, Oregon. Also indicated are units of the Research Natural Area and a Wild and Scenic River corridor along the Deschutes River.

tubes and basalt beds. The ponderosa pine forests within Pringle Falls Experimental Forest, however, represent special places with unique values: their location within the context of environmental and historical settings, the resource outputs they have provided and the resource values they continue to provide, and the opportunities they afford to address current and future management issues of wildland resource management. In this paper, we (1) describe the special values associated with ponderosa pine stands in Pringle Falls, (2) present our rationale for developing a set of silvicultural prescriptions to protect and enhance these special values, and (3) discuss some unique features involved with implementing the silvicultural prescriptions within the experimental forest.

Special Values of Ponderosa Pine Stands in Pringle Falls Experimental Forest

We base our assignment of special value of ponderosa pine stands within Pringle Falls Experimental Forest on four points. First, ponderosa pine stands in Pringle Falls have special value because they represent a disproportionate amount of remnant eastside old-growth ponderosa pine forests. Second, the administration site has special value because the ponderosa pine stand encompasses historic buildings that provide a link to the establishment and initial functioning of Pringle Falls. Third, ponderosa pine stands on Pringle Butte have special value because they are the sites of many historical and ongoing long-term studies. Finally, ponderosa pine stands surrounding Pringle Butte have special value because they uniquely provide opportunities for new landscape-scale research.

Special Value of Remnant Old-Growth Stands

Pringle Falls Experimental Forest exists as two separate but closely spaced units, each named for the dominant volcanic feature contained within the unit. Pringle Butte, the oldest known geologic formation in the Pringle Butte Unit, is a 5-million-year-old shield volcano rising to 1530 meters (5020 feet). Lookout Mountain, the highest point in the Experimental Forest at 1900 meters (6215 feet) in elevation, is a 300,000-year-old shield volcano. Both these features extend above a generally flat or gently rolling ancient lake basin with average elevation of 1280 meters (4200 feet) that is dotted with small volcanic peaks and cinder cones. Both units are characteristic of low- and mid-elevation portions of the High Cascades physiographic province (Franklin and Dyrness 1973). The Deschutes River, designated a Wild and Scenic River because of outstanding scenic and recreation values, flows northeasterly through the Pringle Butte Unit.

Soils in Pringle Falls are dominated by 0.5 to 2 meters (1.5 to 6 feet) of 6600-year-old aerially deposited pumice and ash from Mt. Mazama (now Crater Lake). More recent deposits are additional ash, pumice, and cinders from surrounding volcanic cones and sand and silt sediments of the La Pine basin, overlain with sands and gravels deposited by glacial outwash from the Cascade Range. Soils derived from Mt. Mazama pumice and ash have only a thin weathered surface layer. Most of the soil profile is undeveloped, with low organic matter content, low nitrogen, sulfur, and phosphorus content, and high porosity. Daytime to nighttime temperature variation within the soil profile can be extreme.

The climate is continental, modified by proximity of the Cascade Range to the west and the Great Basin desert to the east. Most precipitation occurs as snowfall. Annual precipitation averages 600 millimeters (24 inches) on Pringle Butte and over 1000 millimeters (40 inches) on Lookout Mountain. Daytime high temperatures in the summer range between 21 and 32 degrees Celsius (70 and 90 degrees Fahrenheit). Summer nights are cool and frosts can occur throughout the growing season.

Forest communities within Pringle Falls are representative of low- and mid-elevation regional landscapes and contain outstanding examples of undisturbed and managed ponderosa pine, lodgepole pine (*Pinus contorta* Dougl. ex Loud.) and higher elevation mixed conifer forests common throughout central and south-central Oregon. Ecological site differences such as aspect and elevation, and past disturbance events, especially fires, insects, and diseases, and more recent timber harvesting, have created a mosaic of rich biological diversity. Ponderosa pine is the dominant conifer throughout most of Pringle Falls. Shrub layers associated with ponderosa pine include antelope bitterbrush (*Purshia tridentata* (Pursh) DC.), snowbrush ceanothus (*Ceanothus velutinus* Dougl. ex Hook.), greenleaf manzanita (*Arctostaphylos patula* Greene), giant chinquapin (*Chrysolepis chrysophylla* (Dougl. ex Hook.) Hjelmqvist), and pinemat manzanita (*Arctostaphylos nevadensis* Gray).

Ponderosa pine forests east of the crest of the Cascade Range in the Pacific Northwest, including those in Pringle Falls, have undergone dramatic physiognomic changes in the last 100 years. Early settlers and surveyors at the beginning of the 20th century passed through open forests of ponderosa pine with widely spaced trees, few if any down logs, and little litter and woody undergrowth (Bonnickson 2000; Languille and others 1903; Wickman 1992). Witness trees established during the late 1800s in central Oregon were predominantly ponderosa pine with diameters that exceeded 50 centimeters (20 inches) (Perry and others 1995). The stem pattern of these eastside forests was a seemingly uniform parkland of widely spaced medium to large and old trees and continuous herbaceous undergrowth (Agee 1994). Historical fire regimes in these forests consisted of very frequent to frequent, low-intensity fires that burned some or most forest floor plants, consumed litter, and killed primarily small trees (Agee 1993). A fire regime of low-intensity burns, coupled with infrequent large and more intense fires, was common prior to the advent of modern fire suppression efforts. Estimated mean fire return interval was 4 to 11 years within Pringle Falls (Bork 1984; Morrow 1986).

Current amounts of eastside old-growth ponderosa pine forest are estimated to range from 3 to 15 percent of pre-settlement levels (Beardsley and others 1999; Bolsinger and Waddell 1993; Everett and others 1994; Hann and others 1997; Perry and others 1995). Decline in the overall extent of eastside old-growth ponderosa pine forest can be attributed to changes in natural disturbance regimes resulting from active management programs for fire suppression, livestock grazing, selective logging of old fire-resistant trees for timber and insect control, and extensive road building (Bergoffen 1976; Johnson and others 1994; Oliver and others 1994). Pringle Falls is one of the few remaining places where old-growth ponderosa pine forests endure; about 2145 hectares (5300 acres) of old-growth ponderosa pine remain within the experimental forest. Individual dominant ponderosa pine in these stands range from 250 to 620 years in age (data on file, PNW Research Station, LaGrande, Oregon). With effective fire exclusion, understory tree density in these remaining ponderosa pine stands has increased, however, and these stands often contain more fire-intolerant species such as lodgepole pine, Douglas-fir, and grand fir. With the exception of stands recently treated, old-growth ponderosa pine stands are multilayered, contain a variety of size classes, and are greatly overstocked with small-diameter stems. The smaller, younger trees compete for site resources with residual old-growth trees and often lead to mortality of the older and larger diameter trees. Multilayered stands also contain increased number of fuel ladders and greater ground fuels. Consequently, the old-growth stands are at greater risk of stand-replacement wildfire.

Thus, Pringle Falls is a special place because of the disproportionate amount of old-growth ponderosa pine forests it contains.

Special Value of the Administration Site

The administration site at Pringle Falls Experimental Forest consists of 10 hectares (25 acres) on the north bank of the Deschutes River in the Pringle Butte Unit. Original headquarters buildings consist of a three-story administration building and a single-story cottage constructed between 1932 and 1934 by Works Progress Administration (WPA) craftsman. These buildings are excellent examples of the period architecture and rustic rock, log, and frame construction. Later, an additional two-story dormitory, a garage/shop complex, and various outbuildings were added. This infrastructure provides seasonal living and working conditions for about 20 people.

The ponderosa pine stand surrounding the buildings at the administration site has been continuously protected from fire since the early 1930s and, like other ponderosa pine stands that have had fire exclusion, has undergone dramatic changes in structure and composition. Near the turn of the 20th century, 40 to 70 large pines per hectare (16 to 28 per acre) comprised the overstory, with only a few stems in the understory. With fire exclusion, the number of small diameter ponderosa and lodgepole pine saplings increased to a stand density of over 2000 trees per hectare (810 trees per acre). Contiguous fuels represented by decadent antelope bitterbrush, ponderosa pine needle drape, and abundant ladder fuels also increased dramatically. These conditions represent a high risk for stand replacing wildfire, with the first fire likely a high intensity crown fire that would be difficult to control and would likely endanger crews living and working in the headquarters buildings.

Thus, the administration site has special value because the ponderosa pine stand encompasses historical buildings that provide a link to the establishment and initial functioning of Pringle Falls and represent fine craftsmanship from the WPA era.

Special Value of Research Sites on Pringle Butte

Pringle Falls Experimental Forest has a rich history of serving as a diverse natural laboratory used by university and federal scientists for field research. Descriptions of past work from establishment through the early 1990s have been chronicled in an annotated bibliography (Youngblood 1995). Some of the earliest forestry research in central and eastern Oregon occurred within Pringle Falls. A large number of these early studies were located on or immediately adjacent to Pringle Butte (figure 2). The earliest known published work was a rating system for determining the susceptibility of ponderosa pine trees to western pine beetle attack by F. Paul Keen (1936). Keen later documented the age-class distribution in several stands on Pringle Butte, including stands with stems having establishment dates of 1330 A.D. (Keen 1940). One of the earliest silvicultural studies was a test of cutting methods with different intensities of selection, initiated in 1937 along the northeast slope of Pringle Butte (Kolbe and McKay 1939). Later, Edwin L. Mowat described the stand structure and analyzed periodic growth measurements for suppressed ponderosa pine seedlings that were released from a lodgepole pine canopy (Mowat 1950). And at the same time, James Sowder conducted one of the earliest studies combining the objective of sanitation and salvage of ponderosa pine, judged to be highly susceptible to insect attack, with the objective of fuel and fire hazard reduction (Sowder 1951). Contributions such as these added greatly to management of ponderosa pine forests throughout eastside forests

Figure 2—Historical and ongoing research sites, either mapped portions of stands (shown in yellow) or mapped points (shown in green) in the Pringle Butte Unit, Pringle Falls Experimental Forest.

of Oregon and Washington at a time when vast segments of these forests were being harvested to meet society's increasing demands for lumber products.

During the next several decades, studies that were established on Pringle Butte concentrated on determining the competitive effect of shrubs growing with ponderosa pine (Barrett 1965; Dahms 1961); the soil thermal properties, surface temperatures, and seed bed characteristics required for lodgepole and ponderosa pine regeneration from natural seedfall (Barrett 1966; Dahms and Barrett 1975); the biology of dwarf mistletoe in ponderosa pine, its spread, and subsequent damage in understory pine (Roth 1953, 1971; Roth and Barrett 1985); and the effect of underburning on dwarf mistletoe in ponderosa pine (Koonce and Roth 1980). During this time, logging methods that ensured survival of existing seedlings and saplings were developed, thus reducing future reforestation efforts and costs (Barrett 1960). Long-term or permanent research plots were established to study the response of ponderosa pine to fertilization (Cochran 1977), and the release and subsequent growth of ponderosa pine at various tree densities (Barrett 1982; Dahms 1960; Oren and others 1987). Periodic evaluation of these stands added to our understanding of structural changes occurring in natural and managed stands. Also, the frequency, intensity, and spatial patterns of wildfire in old-growth ponderosa pine stands near the top of Pringle Butte were examined (Bork 1984; Mazany and Thompson 1983; Morrow 1986).

Work extending through the 1990s emphasized the relationship between ponderosa pine vigor and mountain pine beetle attacks (Larsson and others 1983); the effect of fire on root decay in ponderosa pine and the occurrence of fungal microflora on burned and unburned sites (Reaves and others

1990); and the cyclic population dynamics of Pandora moth (*Coloradia pandora* Blake), an important defoliator of ponderosa pine (Speer and others 2001). Although much of the long-term research in applied forestry has continued over the years, such as identification of optimal growing regimes for planted pine (Cochran and others 1991), other topics have increased in importance. Current research on Pringle Butte is designed to increase our understanding of the processes that regulate or influence the structure, composition, and pattern of forests and that are critical for the maintenance of diverse, healthy, productive, and sustainable forest ecosystems.

Thus, the Pringle Butte portion of Pringle Falls has special value because of the concentration and legacy of historical and ongoing research sites.

Special Value of New Research Opportunities

Pringle Falls Experimental Forest was established and continues to be managed *a priori* for research and demonstration. One example of research and demonstration identified within the Pacific Northwest Research Station's strategic research planning process is the need to evaluate practices and strategies to better manage risks within the wildland/urban interface. A critical component in establishing a mutually acceptable active management strategy in the area where residential development and wildlands share boundaries is the ability to assess risk of wildland fire moving from the surrounding forest into urban areas in combination with the ability to assess risk of fire originating from the urban setting moving out into the adjacent wildlands. Additional components include the needs to develop and test operational practices and techniques and to evaluate strategies for reducing fuels to manage risks within the wildland/urban interface. Research is needed to assess how silvicultural treatments affect fire risk, stand structure, wildlife habitat, and the risk of other disturbances such as insect outbreaks and invasion by nonnative plants, as well as how treatments influence the social values held by forest users and conditions of local communities. Evaluating management choices at the wildland/urban interface is limited by both inadequate technical knowledge of the effects of treatments, such as prescribed fire or surrogates such as thinning, mowing, or crushing, and also by public resistance to these treatments or to perceived resulting conditions. There also are economic tradeoffs associated with short- and long-term solutions to fire risk reduction activities on both public and private lands, which include initial costs of treatment, employment opportunities, and their attendant impacts on economic well-being, as well as the potential repeated treatment needs and property value considerations. There is a need to identify factors that influence acceptability of wildland fuel reduction strategies and underlying decision-making processes. This includes improved understanding of public knowledge, preferences, and understanding of tradeoffs and opportunities for mutual gains, and also understanding of effectiveness of alternative approaches to enhance public understanding and knowledge of wildland management.

Pringle Falls represents a unique place to conduct such research because of its proximity to the wildland/urban interface and the high number of forest visitors passing through, especially those using the paved highway to the higher Cascade lakes or boating on the Deschutes River. Pringle Falls could play a pivotal role in focusing attention from various research disciplines and resource managers on a set of operational methods for fuels reduction in the ponderosa pine stands surrounding Pringle Butte, on the role various partnerships may play across multiple ownerships including the wildland/urban interface, and on the environmental and social consequences of the various methods. Thus, the ponderosa pine stands surrounding Pringle Butte at Pringle Falls have special value because they uniquely provide opportunities for new landscape-scale research.

Silvicultural Prescriptions for Special Places: The Dilman Project

Discussions among Research Station scientists and personnel on the Bend/Fort Rock District of the Deschutes National Forest resulted in agreement on the special values of ponderosa pine stands in Pringle Falls Experimental Forest, a set of risks associated with each special value, and the need to develop and apply silvicultural prescriptions to maintain and protect the values. These values and their needs include the following:

- Pringle Falls has special value because of the disproportionate amount of old-growth ponderosa pine forest it contains. Silvicultural prescriptions are needed to restore the frequency of low-intensity disturbances and thus the resulting periods of stability in these stands. Prescriptions are needed to protect these remnant old-growth stands and individual trees from stand-replacement disturbances.
- The administration site at Pringle Falls has special value because the ponderosa pine stand encompasses historical buildings that provide a link to the establishment and initial functioning of the experimental forest and represent fine craftsmanship from the WPA era. Because the site continues to be used in support of ongoing research, there is a need to provide a safe working environment for occupants. Silvicultural prescriptions are needed to reduce the fuels and reduce the risk of stand-replacement fire within the administration site in order to protect the historical buildings and protect the lives of those using the buildings.
- Pringle Butte in Pringle Falls has special value because of the concentration and legacy of historical and ongoing research sites. This portion of the experimental forest is bordered on three sides by urban development or dense recreation sites. Recently, fires started in the wildland/urban interface or within recreation sites along the Deschutes River threatened ponderosa pine stands and research sites contained within them on Pringle Butte. Silvicultural prescriptions are needed to reduce the risk of stand replacement fires entering this area from surrounding areas in order to protect historical and ongoing research sites.
- Ponderosa pine stands surrounding Pringle Butte have special value because they uniquely provide opportunities for new landscape-scale research. One fundamental constraint to new research addressing fire risk within the wildland/urban interface, however, is that the once relatively homogeneous landscape surrounding Pringle Butte is fragmented into small stands as a result of past research and management actions. The current landscape provides little opportunity for large-scale operational studies with sufficient treatment replication. Silvicultural prescriptions are needed to modify existing stand structures and consolidate fragmented stands into larger blocks with similar structures such that large-scale operational fuels reduction practices and strategies to better manage risks within the wildland/urban interface may be evaluated.

Prescription Development

Consideration and analysis of silvicultural prescriptions for special places within Pringle Falls occurred within the context of the Dilman Environmental Assessment (EA) and the resulting Dilman project. Under National Environmental Policy Act guidelines, scoping for the Dilman Environmental Assessment began in July 1999. In addition to vegetation management, the environmental assessment addressed road closures and recreation management activities within Pringle Falls. It was completed in December 2001.The environmental assessment is tiered to several layers of management direction that guided the development of project alternatives and prescription development. Existing management direction was provided by:

- Deschutes Land and Resource Management Plan (DLRMP), 1990, and the Regional Forester's Forest Plan Amendment #2, 1995. Within the DLRMP, Pringle Falls is identified as a single management area with specific standards and guidelines for managing resource values. The Regional Forester's Plan Amendment, known as "Eastside Screens," established a policy restricting harvest of trees greater than 53 centimeters (21 inches) in diameter at breast height (DBH). The DLRMP also provided direction for visual concerns along major paved travel-ways.
- Inland Native Fish, 1995. Interim Riparian Habitat Conservation Area applied to the Deschutes River. This policy established a buffer on either side of the river extending outward from the edge of the active stream channel 91 meters (300 feet), within which timber harvesting, including fuelwood cutting, is prohibited except where silvicultural practices are needed to attain desired vegetation characteristics to meet riparian management objectives. Under the Dilman Environmental Assessment, silvicultural practices were carefully considered to avoid adverse effects on inland native fish and included the use of horses for skidding and locating treatment boundaries away from the slope break to prevent sediment from entering the river.
- The Upper Deschutes Wild and Scenic River Final Environmental Impact Statement, 1996. Superimposed on Pringle Falls is a 1.6-kilometer-wide (1-mile) corridor, centered on the Deschutes River, within which management direction emphasizes protection of the outstanding scenic and recreation values.

The Dilman Project focused on the following four objectives:

- Implement the Upper Deschutes Wild and Scenic River Environmental Impact Statement for segment 2 of the river to meet stated goals for protection and enhancement of outstanding scenic and recreation values, and thus protect the special values of old-growth ponderosa pine stands and individual trees (special value 1).
- Provide defensible space along wildland/urban interfaces, especially the administration site, private in-holdings on the north side of the Pringle Butte unit, and the urban interface immediately to the east of Pringle Falls (special value 2).
- Provide defensible space along major travel corridors that access National Forest land, especially roads on either side of the Deschutes River that serve as key access corridors through Pringle Falls, thus protecting the special values of historical and long-term research sites concentrated on Pringle Butte (special value 3).
- Enhance the special values of existing old-growth ponderosa pine stands and create options for future research within Pringle Falls by concentrating

treatments around the base of Pringle Butte to restore old-growth conditions and processes (special value 4).

A mix of vegetative treatments was developed that addressed these objectives. Because many of the stands were multistructured, contained a variety of size classes, and differed greatly in composition, density, and past history, various treatments with multiple entries were determined necessary to achieve desired conditions. Although the Dilman Project specifically addresses about 746 hectares (1844 acres) within Pringle Falls, it also addresses stands with similar needs and treatments in proximity to the experimental forest. Within Pringle Falls, about 471 hectares (1164 acres) will receive commercial thinning, about 85 hectares (185 acres) will receive noncommercial thinning, and about 200 hectares (495 acres) will have fuel reduction from prescribed broadcast burning, hand piling and burning, mowing, mechanical mastication, or some combination of fuel treatments. In the old-growth ponderosa pine stands surrounding Pringle Butte, commercial thinning of 13- to 53-centimeter (5- to 21-inch) DBH ponderosa pine, at roughly 6- by 6-meter (20- by 20-foot) spacing with retention of natural clumping, will reduce stand density, disaggregate fuel continuity, and decrease fuel ladders. No ponderosa pines over 53 centimeters (21 inches) DBH will be removed. All large lodgepole pines, however, will be removed, further reducing stand density and fuel loadings. Along the river corridor, all lodgepole pines greater than 8 centimeters (3 inches) DBH will be commercially thinned, with a subsequent noncommercial thin at 3.3-meter (11-foot) spacing to follow. To begin developing open stand structures along highly visible paved travel ways, ponderosa pines in the 13- to 53-centimeter (5- to 21-inch) DBH class will be commercially thinned, again retaining all existing natural clumping, and all lodgepole pines will be removed through a series of commercial and noncommercial thinnings. Noncommercial thinning of ponderosa and lodgepole pines will occur in areas of previous harvest activity such as old clearcuts and fire and beetle-kill salvage areas to reduce fuels and accelerate residual tree growth. Finally, fuel reduction in areas of previous harvest, in areas of commercial or noncommercial thinning, or as an activity by itself, may include slashing, hand piling, mowing, mechanical mastication, or prescribed burning either singly or in combination with other fuel reduction methods. When fully applied, the silvicultural prescriptions collectively represent a significant amount of vegetation management treatments within Pringle Falls (figure 3) while addressing the four identified special values.

Prescription Projection

We used the southeast Oregon variant of Forest Vegetation Simulator (FVS), coupled with the Stand Visualization System (SVS) to model and project the effects of our prescriptions into the future. For each of the stands modeled, the following assumptions applied:

- Marking prescriptions are fully met 90 percent of the time.
- The shrub layer is treated concurrently with activity fuel reduction treatments.
- Natural regeneration of ponderosa and lodgepole pines occurs on a frequent but nonuniform basis; only ponderosa pine establishment and growth was modeled.
- A single noncommercial thinning after commercial thinning is more realistic than frequent noncommercial thinnings given budget constraints;

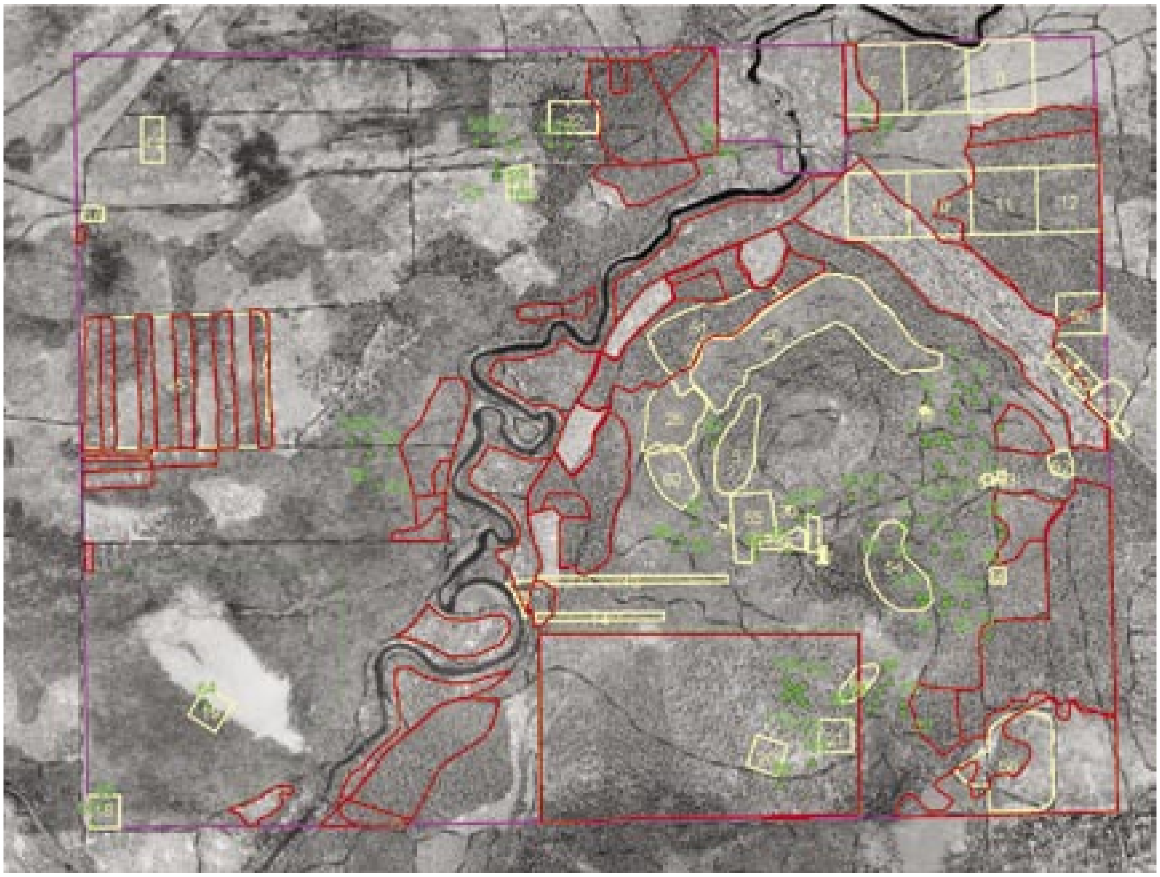

Figure 3—Commercial thinning, noncommercial thinning, and fuel reduction units of the Dilman project (shown in red), overlapped with historical and ongoing research sites (shown in yellow or green) in the Pringle Butte Unit, Pringle Falls Experimental Forest.

this suggests that prescribed fire in subsequent years may serve as a surrogate for additional noncommercial thinning.

- Prescribed fire will be applied after each initial commercial thinning treatment, 15 years after the initial treatment and then conservatively, at 20-year intervals through the remaining 100-year timeframe.

Model outputs were initially configured for 5-year cycles to more accurately account for initial prescribed fire and initiation of natural regeneration of ponderosa pine.

As one example of prescription application and projection, we present graphical representations of structural changes in stand 904. In 2000, this stand consisted of about 2000 ponderosa pine trees per hectare (806 per acre) and 900 lodgepole pines per hectare (363 per acre) (figure 4). Although most of the basal area was in large-diameter ponderosa pine, most of the density was in small-diameter ponderosa and lodgepole pine. Less than 15 ponderosa pines per hectare (6 per acre) were greater than 53 centimeters (21 inches) DBH. Total basal area was about 24 square meters per hectare (106 square feet per acre). Commercial thinning is projected to reduce the density to about 1790 ponderosa pine trees per hectare (726 per acre) and eliminate most of the standing lodgepole pine (table 1). The first prescribed fire is scheduled as a broadcast underburn in 2005; this treatment is projected to eliminate all lodgepole pines that escaped the initial

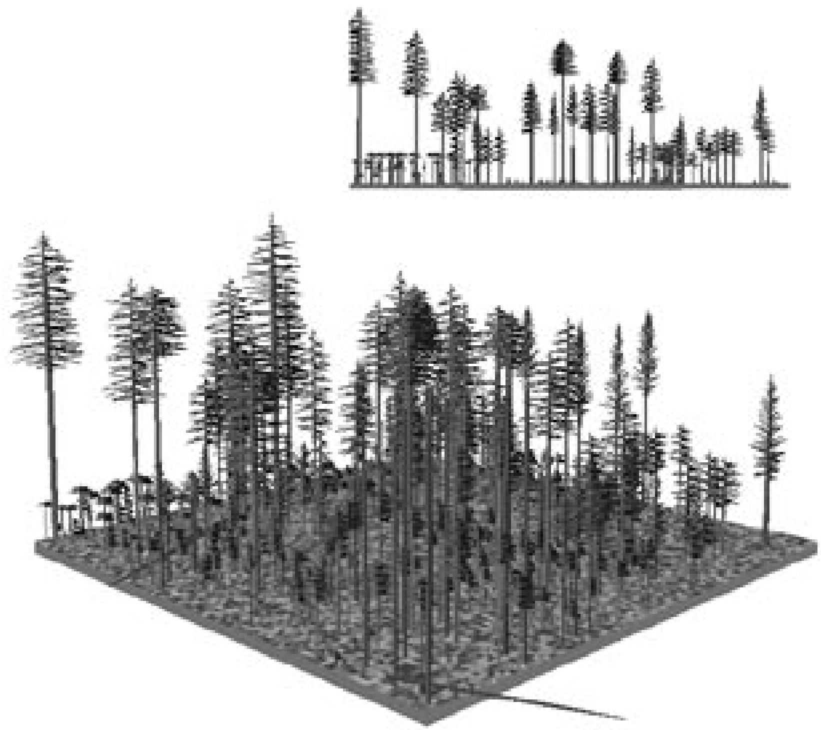

Figure 4—Graphical representation of initial structural conditions in stand 904, designated for commercial thinning and broadcast burning as part of the Dilman project in Pringle Falls Experimental Forest.

Table 1—Change in structural features for stand 904 projected with FVS/SVS.

Year	Density	Percent of maximum stand density index	Quadratic mean diameter	Basal area	Treatment and target
	Trees/hectare (trees/acre)		Centimeters (inches)	Square meters (square feet)	Trees/ha (trees/acre)
2000	2877 (1165)	30	11.7 (4.6)	30.9 (134.4)	Thin,1793 (726)
2005	1793 (726)	23	11.7 (4.6)	19.2 (83.8)	Burn, 321 (130)
2020	442 (179)	24	23.9 (9.4)	19.0 (82.6)	Burn, 193 (78)
2040	230 (93)	25	33.8 (13.3)	20.6 (89.7)	Burn, 131 (53)
2060	175 (71)	26	39.9 (15.7)	21.9 (95.4)	Burn, 106 (43)
2080	151 (61)	28	43.9 (17.3)	22.9 (99.6)	Burn, 94 (38)

thinning and to reduce the stand density to about 321 ponderosa pines per hectare (131 per acre) (figure 5). Subsequent underburnings are scheduled at 20-year intervals to control establishment of lodgepole pine, to gradually reduce density, and to restrain down woody fuels and live shrubs. After five underburns, overall density is projected to be about 94 trees per hectare (38 trees per acre), consisting almost entirely of large-diameter, widely spaced old-growth ponderosa pine (figure 6).

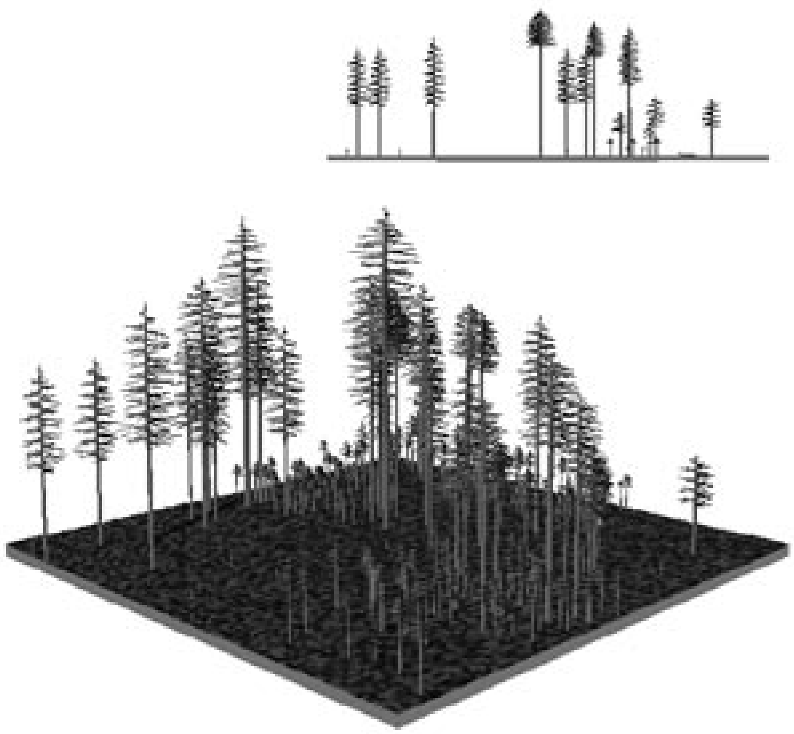

Figure 5—Graphical representation of structural conditions in stand 904 projected for the year 2005, after commercial thinning and initial broadcast burning as part of the Dilman project in Pringle Falls Experimental Forest.

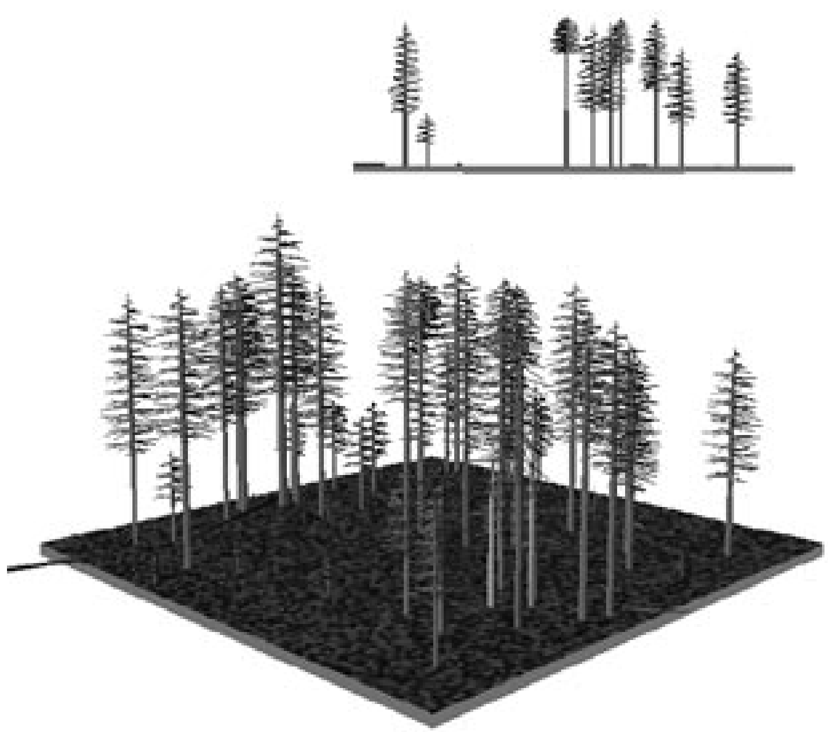

Figure 6—Graphical representation of structural conditions in stand 904 projected for the year 2080, after the initial commercial thinning and broadcast burning followed by four broadcast burnings, as part of the Dilman project in Pringle Falls Experimental Forest.

Implementation of Prescriptions for Special Places

Treatment implementation in Pringle Falls began with the administration site. While initially considered within the Dilman Environmental Assessment, a categorical exclusion was written to assess treatment on the 10 hectares (25 acres) of the administration site separately from the rest of the Dilman project in order to take advantage of PNW Station facilities maintenance funds. The categorical exclusion was based on Forest Service Policy and Procedure Handbook 1909.15 that provides for routine operation and maintenance of administration sites and was signed in February of 2000. Treatment of the administration site consisted of a single commercial timber sale of 30,000 board feet, hand thinning of trees around the buildings, pruning of dwarf mistletoe-infested limbs, hazard tree removal, hand-piling and burning of slash, and broadcast burning of the shrub layer. Work in the administration site began in March 2000 and was completed in October 2001. This stand now contains about 25 large pines per hectare (10 per acre) with DBH greater than 53.3 centimeters (21 inches). Total stand density is about 247 trees per hectare (100 per acre), basal area is 22.1 square meters per hectare (96.4 square feet per acre), and stand density index is 40 percent of maximum (figure 7).

Outside of the administration site, four separate types of timber sale contracts (2400-1, 2400-3, 2400-4, 2400-6), service contracts, purchase

Figure 7—Pringle Falls Experimental Forest administration area after thinning and underburning for fuels reduction.

orders, and force account crews are being used to implement the silvicultural prescriptions within the Dilman project. Work began in the spring of 2002. All activities associated with the first entry are planned for completion by spring 2009. High fuels areas within the wildland/urban interface, administrative sites, campgrounds, and major access roads represent the highest priorities and will be completed first.

Operational Considerations

To address concerns for fisheries during the treatments along the Deschutes River within Pringle Falls, thinning within 91 meters (300 feet) of the bank involves a combination of hand crews, including those from local Youth Conservation Corps, Oregon State Department of Correction, Deschutes County Corrections, force account crews from the Deschutes National Forest, and troubled youth programs, in addition to service contracts and a contract for horse logging.

Selection of horse logging as a means for skidding of logs within the designated buffer along the Deschutes River was based, in part, on a desire to prevent opening or creating additional areas for nondesignated, dispersed recreational camping. Conventional ground-based skidding operations require a network of designated skid roads. On the flat benches adjacent to the Deschutes River, these designated skid roads are slow to revegetate with trees and shrubs and quickly become permanent access roads to desirable dispersed recreation campsites along the river. In addition, horse logging has been used over the last decade in campgrounds and other recreational sites on the Deschutes National Forest to remove hazard trees and has been well received by the public and public interest groups as an environmentally sensitive means of treatment. We anticipated greater public support for this part of the Dilman project if horse logging was featured. Material removed during horse logging consisted of lodgepole pine greater than 15 centimeters (6 inches) in diameter under a 2400-3 contract. This diameter limit was set in part because of the potential for the combination of products removed to generate positive net revenue for the operator, and the feasibility of hand piling the residual stems. Local experience has shown that the upper limit in piece size for most hand crews is about 23 centimeters (9 inches) DBH. Production rates for horse logging on the gentle terrain average about 2,000 to 4,000 board feet per day depending on factors such as skidding distance, average volume per log, volume per acre, and stand density. Brush disposal deposits collected from the purchaser are used to fund piling of slash by hand crews.

One large sale designed to remove about 1.04 million cubic feet (about 5.4 million board feet) from Pringle Falls was sold under a 2400-6T contract (tree measurement contract) with minimum specifications for merchantable trees of 8 centimeters (3 inches) DBH for lodgepole and ponderosa pine. Again, harvesting is restricted to trees less than 53 centimeters (21 inches) DBH. Final bid price was $35.75 per hundred cubic feet for lodgepole pine and $118.58 per hundred cubic feet for ponderosa pine. To minimize damage on residual vegetation and soils, the contract requires a boom-mounted saw/shear with a reach of at least 5 meters (17 feet) from the equipment center point. Other requirements include the use of designated skid roads at 30.5- meter (100-foot) average spacing and moving logging equipment over snow and frozen ground. Brush disposal deposits and whole-tree yarding ensure adequate treatment of slash. Minimal residual stand damage associated with whole-tree yarding is anticipated because of the flat topography across harvest units, which minimizes the need for sharp turns during skidding, and the relatively short tree lengths being harvested.

Post-sale service contracts for noncommercial thinning and removal of damaged and unsuitable trees are scheduled in all commercial timber sale units. Service contracts will use a variety of funding sources including hazardous fuels, Knutsen-Vandenberg Act (KV), and appropriated timber stand improvement thinning. Service contracts and purchase orders also will be used to conduct noncommercial thinning in stands that have little or no commercial size material.

Force account crews will be used to conduct prescribed burns, prune, dispose of brush, establish road closures, and rehabilitate dispersed recreation sites and designated campsites within Pringle Falls.

Summary

Pringle Falls Experimental Forest has been a center for research in ponderosa pine forests east of the crest of the Cascade Range since 1931. Long-term research facilities, sites, and future research opportunities are currently at risk from stand-replacement wildfire because of changes in stand structure resulting from past fire exclusion, especially the dramatic changes in tree density and establishment of lodgepole pine under ponderosa pine. We identified four conditions or locations within Pringle Falls that represent special values that are increasingly at risk from fire because of the structural changes either within the stands themselves or in adjacent stands that receive recreational impacts or urban development. Our set of silvicultural prescriptions is designed to protect and enhance these four special values. Implementation of the silvicultural prescriptions involves innovative use of existing contracting and workforces. To date, results have been well received by the visiting public.

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Enhancing Moist Forest Restoration Opportunities in Riparian Systems

Theresa Benavidez Jain[1] and Russell T. Graham[1]

Abstract—*In northern Rocky Mountain moist forests, riparian systems contain many attributes that create unique biophysical conditions that alter disturbances and microenvironments; thus creating distinct forest structures, species composition, and management challenges. For example, browsing, limited opening size, competition from surrounding ground vegetation, high soil moisture, and cold air drainage challenge the application of any silvicultural method, but if these aspects are considered prior to applying restoration efforts, they can also facilitate a successful result. This paper discusses a series of silvicultural tools that can be used in riparian restoration, including integrating knowledge on competitive thresholds for western white pine (*Pinus monticola*) (occupancy, competitive advantage, and free-to-grow status), maintaining overstory canopy for modifying cold air drainage, and using coarse woody debris and other vegetation to decrease browsing damage while minimizing sedimentation input and soil compaction. Although applying an integrated silvicultural system is critical in any restoration project, non-technical expertise concentrating on the interactions among people during project implementation is needed to achieve successful restoration results.*

Introduction

In northern Rocky Mountain moist forests, riparian systems contain many attributes different from upland forests. These systems are characterized as areas where vegetation and physical components (soils, topography) contribute directly to a stream or lake's physical and biological characteristics (i.e., shading, stream fauna habitat) (Swanson and Franklin 1992). Depending on the stream type, the associated riparian areas contain diverse environmental conditions that affect the composition, regeneration, establishment, and growth of plants. Herbivory, competition, microsite topography, floods, erosion, abrasion, drought, frost, and variable nutrition directly affect these plants. Riparian plants also are indirectly affected by landscape components including topography, geomorphology, stream shape, soil type, water quality, elevation, climate, and surrounding upland vegetation (Odum 1971). Fire, ice, windstorms, and insect infestations, although less common, can directly or indirectly influence riparian systems (Agee 1988, Naiman and Dècamps 1997).

Plants that colonize and grow in riparian areas have evolved to adapt to these diverse environments and disturbances by invading, enduring, or resisting these conditions (Agee 1988, Naiman and Dècamps 1997). Therefore, it is important to understand both the riparian environment and a plant's adaptations and life history prior to applying silvicultural methods for restoring these systems. The objective of this paper is to discuss the role of

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silviculture in riparian restoration. Topic areas include the ecological aspects of the northern Rocky Mountain moist forest riparian environment, which can affect silvicultural applications, the applicability of integrating multiple spatial and temporal scales into the silvicultural system or method, and providing silvicultural tools useful in riparian restoration.

Riparian Environment

Three classes of perennial streams occur in moist forest settings: riffle-pool, cascade-pool, and meandering glide (Rabe and others 1994, Savage and Rabe 1979). Riffle-pool streams have moderate gradients and contain riffles (shallow, turbulent flow over rock) alternating with smooth-flowing glides or deep, quiet pools (figure 1). These occur in valleys with narrow flood plains. Shrubs, grasses, and sedges are the primary riparian vegetation on the flood plains, and trees occupy settings above these plains (Savage and Rabe 1979). Since they contain a diverse aquatic insect community and favorable fish habitat, these streams are often fish bearing. Cascade-pool streams have torrential flows over large rocks and logs; these streams dissect steep slopes and have narrow riparian zones (figure 2). Logs are an important component in cascade-pool streams and are largely responsible for creating the cascades. Bedrock is usually exposed in the channel and heavy shading from trees is common. Fish rarely occur in these streams because the cascades create barriers during low water flow, water velocity is too high during spring runoff, and resting areas for fish are less abundant (Savage and Rabe 1979).

Figure 1—Canyon Creek at Priest River Experimental Forest in northern Idaho is an example of a riffle-pool stream. These streams are often fish bearing and contain a diversity of aquatic insects.

Figure 2—Benton Creek at Priest River Experimental Forest in northern Idaho is an example of a cascade-pool stream type. This is characterized as containing very narrow riparian areas often with trees and logs in the stream and along the riparian area.

Riffle-pool and cascade-pool streams in northern Rocky Mountain moist forests are characterized by two habitat types: western redcedar (*Thuja plicata*)/devil's club (*Oplopanax horridum*) and western hemlock (*Tsuga heterophylla*)/wild ginger (*Asarum cadatum*) (Cooper and others 1991). Dominant tree species include western redcedar and western hemlock, but Engelmann spruce (*Picea engelmannii*), western white pine (*Pinus monticola*), western larch (*Larix occidentalis*), and grand fir (*Abies grandis*) can also occur. Soils are quartzite and alluvial mixtures of metasediments, siltite, ash, and mica schist. These soils have fairly coarse textures (gravelly loamy sands to sandy loams) with up to 40 to 50 percent gravel content. The riparian areas contain deep forest floors and no bare soils or rock (Cooper and others 1991).

Meandering glide streams contain many curves and meander along a shallow gradient (approximately 1 percent) (Savage and Rabe 1979). These streams have riparian areas that support significant wetland communities maintained by high water tables and are frequently flooded (figure 3). The stream biota is adapted to soft substrate, slow water velocities, and sometimes-low oxygen saturation. These conditions often favor only plant species (sedges, grasses, and forbs) adapted to these conditions. Although many tree species do not grow in the wetland surrounding the stream or lake, trees may grow along the edge (poorly drained areas often occur between, but are not limited to, permanent open water zones and uplands) (Rabe and Bursik 1991, Tiner 1999). For example, some of these areas were once occupied by large old western redcedar forming shady groves (figure 4).

Figure 3—Meandering glide streams have low gradients and considerable sinuosity. They support wetland ecosystems consisting of grasses, sedges, and shrubs. This is the North Fork of the Clearwater River in northern Idaho.

Figure 4—This picture shows a western redcedar riparian grove along Cedar Creek above the North Fork of the Clearwater in northern Idaho. These western redcedar are approximately 400+ years, and the understory consists of dense herbaceous cover.

Why Restore Riparian Areas?

There are many physical, biological, and social reasons for managing or restoring riparian areas. Streams and associated riparian areas influence hydrologic characteristics (Naiman and Dècamps 1997, Windell and others 1986). Depending on the soil type and permeability, they alter biogeochemistry, ground water discharge and recharge, erosion control, water purification, and flood control; moderate air temperatures; contribute water vapor to the atmosphere; and produce gasses from biomass decomposition and nutrient cycling (Windell and others 1986). Biologically, they provide habitat and corridors for a wide range of wildlife species and the vegetation, soils, and micro-topographical environments favor insect populations (a requirement for maintaining fisheries). Socially, they are prime areas for recreational use such as providing spiritual, physical, aesthetic, and recreation values (Windell and others 1986). In addition, they can also be quite valuable for timber production (Berg 1994, Newton and others 1996).

Historically, riparian areas (particularly, meandering glide and riffle-pool stream types) were frequently the first places developed by European immigrants because the floodplains provided excellent farmlands. Trees (narrowleaf cottonwood [*Populus angustifolia*], western redcedar, western hemlock, and western white pine) were used for firewood, timber, house building material, or for a combination of uses (Windell and others 1986). In the moist forests, channelization often occurred in streams and rivers, thus decreasing sinuosity (Hann and others 1997, Windell and others 1986). Excessive cattle grazing damages vegetation, increases soil compaction and erosion, introduces exotic plants, and degrades water quality with fecal contamination (Dobkin and others 1998). Because riparian zones are highly valued for a variety of purposes and represent a limited fraction of the landscape, and because past use has led to degraded conditions (Windell and others 1986), riparian restoration has become an increasingly important issue.

Restoration of Moist Forest Riparian Ecosystems

Knowing where to begin is the first step in any restoration effort. Landscape attributes can provide a biophysical template for setting restoration priorities. Some have suggested a step-down process from broad to fine scales for planning restoration activities (Jensen and Greene 1991, Naiman and others 1993). For example, in the Coeur d'Alene River Basin of the northern Rocky Mountains, Jain and others (2002) used multiple spatial scales combined with historical pattern of western white pine abundance to define possible restoration priorities. They determined western white pine was most abundant and most productive in places where subsurface flow of water and water retention occurred in areas found on slopes highly dissected by streams, slopes adjacent to streams, toeslopes, benches, or wide stream bottom riparian areas. Camp and others (1997) identified fire refugia based on physical landscape attributes occurring at multiple spatial scales in the eastern Cascades. They too found these protected areas occurred near or adjacent to riparian areas. Jensen and Greene (1991) used a hierarchical approach to describe and map riparian areas. They used this information to identify location, extent, and diversity of riparian areas, evaluate existing condition, and identify reasonable desired future conditions for management. Because the approach was hierarchical in nature, broad scales provided context for fine scale prioritization; and the approach identified relative uniqueness of stream and riparian areas, current condition relative to other riparian areas, and whether a particular future desired condition was possible (Jensen and Greene 1991). Using a multiple scale approach at least by linking the entire watershed to site-specific treatments of riparian areas is one key area that has proven effective in restoration projects (Cannin 1991).

Temporally, understanding the past history relative to the current condition can help identify the time frame needed to attain a future desired goal in restoration efforts. Moreover, time can provide an indication of what might be a feasible desired future condition. For example, if old growth western redcedar once occupied the site, but was harvested in the early part of the 20th century, a possible desired future condition is to restore this area to a western redcedar grove. However, the conditions for regeneration may be vastly different today when compared to 400 years ago, when the original western redcedar regenerated. Furthermore, over time, intermittent disturbances probably encouraged the development of the grove. The climate, stream morphology, and other physical and biological aspects may also be quite different today when compared to historical conditions. Chambers and others (1998) discovered that riparian areas in Nevada could not be restored to conditions that existed prior to the past 150-200 years. They determined climate change and stream incision from recent floods prevented these riparian areas from attaining characteristic forest compositions and structures of the past. If similar changes occurred in riparian areas that once held large old western redcedar, it may not be appropriate to plant western redcedar (late-seral species) in hopes of obtaining a historical condition. This may be particularly true if current plant communities reflect an early successional stage. In this case, early-successional tree species (western white pine, lodgepole pine, Engelmann spruce) may be more appropriate with future western redcedar reintroductions possible underneath an established canopy. Therefore, the time frame to achieve the desired condition may take multiple centuries rather than one or two centuries.

Two silvicultural objectives often applied in riparian restoration include establishing desirable high cover (>12 m in height) or improving forest ground cover (<3.5 m in height). Large conifers play important roles in riparian and stream sustainability (such as wood input, wildlife habitat, and long-term nutrition); hence, maintaining or regenerating conifers is often a goal in riparian restoration (Newton and others 1996). Meandering glide or riffle-pool streams occur in valley bottoms and have been most likely harvested in the past or have roads along the streams. Therefore, the following discussion will be most applicable in these stream types but may also be applied to other types (i.e., lakes, small springs) of riparian restoration.

Restoration activities associated with silvicultural systems occurring along cascade-pool streams will be most similar to upland regeneration techniques. Minimum competition from shrubs and grasses will tend to occur in these settings, since the dominant vegetation is often composed of trees. Browsing damage can occur from deer (*Odocoileus* spp.) and elk (*Cervus elaphus nelsoni*) but will be similar to damage occurring in the upland forest. Regeneration in riffle-pool stream riparian areas will have some competition from grass and shrubs, but if regeneration occurs far enough from the stream, competition may be minimized. However, browsing may impact regeneration efforts, since riffle-pool streams attract both ungulates and small animals. In the meandering-glide streams, a silvicultural system will need to address competition, high water tables, browsing, and sedimentation from flooding.

Restoration techniques that include enhancing current forest structure or composition may include cleanings, weedings, and thinnings. Historically, these treatments were associated with altering tree structure and composition. Silvicultural methods could be applied to encourage sprouting in deciduous trees such as narrowleaf cottonwood (*Populus angustifolia*), paper birch (*Betula papyrifera*), and aspen (*Populus trimuloides*). Silvicultural treatments can also be used to develop desired shrub communities. For instance, coppicing can be applied to favor large shrubs like alder (*Alnus* spp.), willow (*Salix* spp.), or Rocky Mountain maple (*Acer glabrum*). However, care must be taken as not to have adverse outcomes such as introducing exotic plants, compacting or displacing soil, or losing excessive surface organic matter.

Establishing Trees

To meet many restoration goals, species presence may be as important as ensuring tree numbers. In many restoration efforts, regeneration and establishment of conifers is difficult and often fails without some type of disturbance (Newton and others 1996). However, these treatments must minimize erosion, avoid harmful levels of water contamination by silt or herbicides, and maintain adequate stream cover (Newton and others 1996). In riparian settings prone to aggressive colonization by ground level vegetation, large planting stock (3-0 or greater) is preferred no matter what species or combination of species is selected. Grasses, forbs, and sedges not only compete for nutrients and light but they can also mechanically injure trees and attract trampling animals. Moreover, overstory competition (trees, shrubs) should be irregularly spaced to maximize sunfleck duration and decrease sunfleck density (Jain 2001). Large seedlings are more resilient to damage from browsing or other animal damage and once established can compete more readily with other plants (Cafferata 1992; Giusti and others 1992; Graham and others 1992; Marsh and Steele 1992; Newton 2002; Rochelle 1992). However, the planting of large seedlings requires additional care and handling to ensure they have proper root to shoot ratios, are not bent or twisted (j-rooted) when planted, and have good root to soil contact.

Species Preference

In moist forests, suggested species include western white pine, lodgepole pine, western redcedar, western hemlock, and Engelmann spruce. In stream reaches that tend to pool cold air creating frost pockets, lodgepole pine, western white pine, and Engelmann spruce are the favored species, because of their tolerance to frost when dormant (Minore 1979). In settings with high forest cover and minimal competition from ground level vegetation, western redcedar, western hemlock and/or western white pine may be more applicable.

Although western white pine is an early to mid-successional species, it is well suited to growing in many riparian settings since it can tolerate a range of growing conditions and endemic diseases in northern Rocky Mountains moist forests. The species is well suited for planting in small openings within riparian systems and its growth is predicated on the size of opening or gap in which it is located. Jain and others (2002) determined openings within riparian areas might only need to be 0.25 ha in size for western white pine to achieve competitive advantage and 0.5 ha in size to achieve free-to-grow status (i.e., when a seedling or small tree is free from competition from other plants) (Helms 1998).

Controlling Competition

In many riparian areas, successful conifer establishment and growth is dependent upon the ability to control competing vegetation. Most often, overtopping of seedlings needs to be minimized until they become established and are able to obtain free-to-grow status (figure 5). In riparian areas, grasses tend to be tall (sometimes 2 meters) and there is often a high density of shrubs and various herbaceous plants (figure 6). Moreover, when overtopping grass or forbs die or collapse, seedlings can be crushed and/or covered by the grass (especially under snow). Therefore, competing vegetation control needs to extend beyond the immediate planting area (possibly up to a 2 meter radius around a tree) (figure 7). The preferred method for controlling competition depends on cost, impacts, method efficacy, and personal safety when applied (Newton and others 1996), but it can include mechanical or chemical treatments.

Mechanically removing vegetation can elevate sediment input, increase soil compaction, and may be difficult to apply to small areas (Harvey and others 1989). Furthermore, results may be short lived. Mechanical applications often favor sprouting of shrubs, and forbs and grasses may colonize areas before a tree becomes established and achieves free-to-grow status (Miller 1986). Applying a second mechanical treatment risks injuring or destroying planted seedlings. Mechanical treatments may also present risks for exotic plant invasion, since mineral soil exposure is an ideal seedbed for many plant species (Haig and others 1941). Manually cutting and removing competition

Figure 5—In riparian restoration, grass and shrubs are considered part of the canopy when establishing new seedlings. Under tall grass, this canopy opening is approximately 60 percent and would not achieve free-to-grow status for western white pine.

Figure 6—Riparian areas tend to have high concentrations of grass, forbs, and shrubs that are usually quite tall. In this picture, the grass is at least 1 m tall. If the objective is to establish conifers, some site preparation and competition control is required.

Figure 7—Western white pine seedling with competition removed in planting spot that was not large enough to avoid being crushed by surrounding grass and other vegetation.

minimizes compaction and sediment input, but it is extremely labor intensive and may require several treatments per year (Newton and others 1996).

Another option is to use herbicides to control competition. Spot herbicide application has several advantages over broadcast application (Boyd 1986). First, it is less costly because there is less chemical used per unit area. Second, spot application is usually more environmentally acceptable and desirable over broadcast application, because small areas are treated and application is possible under a wider variety of weather conditions. Finally, this treatment provides a diversity of habitats that may benefit wildlife and prevent the concentration of animals that could physically damage trees. If the herbicide is applied conservatively and the appropriate herbicide (glyphyosate, imazapyr, metsulfuron, and/or triclopyr) is used, this method can provide systemic and non-systemic herb and shrub control with no water contamination (Newton and others 1996, Newton 2002). Disadvantages include greater labor costs compared to broadcast application, more hazardous to workers because they most likely will be applying it by hand or intimately working with the herbicide, and if used in site preparation, spots may be difficult to locate at planting time so flags may need to be placed in applied areas (Boyd 1986). Herbicides used for shrub control in forests include 2,4-D, glyphosate, imazapyr, picloram, or triclopyr. To control herbaceous plants (grasses and forbs), Atrizine, 2,4-D, sulfometuron, and hexazinone are suggested (Newton 2002). However, specific time of application and effectiveness of herbicide to affect targeted vegetation varies. Specific details on application and target species are available through the Pacific Northwest Experiment Station Weed Management Handbook (Newton 2002).

If one cannot treat competition either mechanically or chemically, the only viable option is planting Engelmann spruce, since it has a stiff enough stem to avoid crushing or bending under grasses or other vegetation (Robert Hassoldt, personnel communication) (figure 8). Additionally, there is some evidence that spruce may grow relatively well in places with moderate amounts of competition. For example, white spruce (*Picea glauca*) has been shown to perform similarly or better in places with low and medium shrub densities when compared to areas with no shrubs. White spruce growth was only affected in places with high shrub densities (Posner and Jordan 2002).

Figure 8—Engelmann spruce seedling planted with no competition control.

Browsing

A variety of animals (insects, rodents, omnivores, ungulates, and livestock) may eat or damage tree seedlings. Livestock and wildlife damage can occur from browsing, trampling, and rubbing, and most western tree species are susceptible. Wildlife species including, but not limited to, beaver (*Castor canadensis*), porcupines (*Erethizon dorsatum*), lagomorphs (*Lepus* spp. and *Sylvilagus* spp.), black bear (*Ursus americanus*), deer, and elk can damage seedlings. Riparian areas attract a wide range of these wildlife and livestock, making animal conflicts an issue in many restoration efforts. Hence, the potential for browse damage should be thoroughly evaluated prior to implementing the silvicultural system (Knapp and Brodie 1992, Nolte 2003a). Nolte (2003a) suggested using a five step process: (1) assess the severity and potential damage if no action is taken, (2) evaluate the feasibility of alleviating the problem, (3) develop a strategy prior to browse damage prevention measures, (4) implement program, and (5) monitor consequences.

It may also be wise to evaluate potential browse impacts at multiple spatial scales, to help identify how a riparian area contributes to the overall wildlife habitat matrix (McComb 1992). The size of the area to evaluate will depend on the species of interest (figure 9). If the species is beaver, then an evaluation of riparian attributes will be sufficient; however, if deer or elk are the species of interest then a landscape (watershed) perspective may be more appropriate. If the riparian area to be restored is the only source of water or has unique habitat attributes favoring a particular species, then it may receive abundant use. Under these circumstances seedlings may require considerable protection or else damage can be severe enough to reevaluate restoration objectives.

A variety of preventive and remedial techniques have been tested, with mixed results. These have included providing alternative food source or planting unpalatable trees species, silviculturally modifying habitat to disfavor specific browsing species, physically or chemically protecting tree seedlings, frightening browsers away, or trapping or killing browsing threats. Unfortunately, there is not one method that solves all browsing problems. The preferred approach will depend on assessment results and the most effective treatment may require integrating several methods.

Sometimes, providing a preferred food source decreases the probability of trees being browsed (Nolte 2003b). This method, in theory, provides benefits like the maintenance of plant diversity and water quality, and can be relatively cost-effective compared to fencing or other types of plant protection. But extensive evaluation of methods is limited and results are highly variable (Cafferata 1992; Giusti and others 1992; Graham and others 1992; Marsh and Steele 1992; Newton 2002; Rochelle 1992). With spot application of herbicides, fewer food sources are eliminated, which may potentially diminish browsing problems. Another technique is to plant tree species that are tolerant to or less susceptible of being browsed (Black and Lawrence 1992). Unfortunately, in the moist forests, western redcedar (which is very palatable)

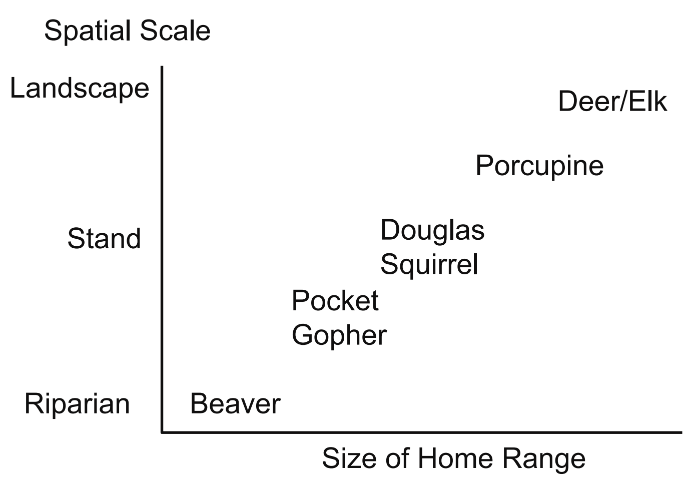

Figure 9—Wildlife habitats occur at different spatial scales (McComb 1992). Therefore, riparian restoration efforts should consider multiple spatial scales when evaluating potential animal damage.

is the preferred species used in riparian restoration; therefore seedling damage from wildlife can be prohibitive to its establishment. Many recommend large planting stock because it typically is less vulnerable to animal damage (Cafferata 1992; Giusti and others 1992; Graham and others 1992; Marsh and Steele 1992; Newton 2002; Rochelle 1992).

Physical protection of seedlings with polypropylene mesh tubes is an option and appears to be successful in some cases (Black and Lawrence 1992). Fencing riparian areas to keep livestock out can be effective, but expense limits its use (Nolte 2003c). Other forms of physical deterrents might be possible. Graham and others (1992) noted that when coarse woody debris (>7.5 cm in diameter) was greater than 50 Mg/ha before livestock utilization fell below 10 percent. These are well within the recommended amounts (37 to 74 Mg/Ha) necessary for maintaining long-term soil productivity (Graham and others 1994). In some cases, minimizing disturbance avoids creating habitat that may increase pocket gopher (*Thomomys talpoides*) populations (Marsh and Steele 1992).

In riparian restoration, application of chemical repellents or poisons may not be an acceptable option unless the browsing problem is severe and positive results are substantial. First, water quality issues should be investigated before using any repellents or poisons. In some cases repellents have had inconsistent results, making chemical treatment an impractical option (Nolte 2003d). Moreover, competition and browsing issues are often interdependent. In these situations, herbicide application for competition control may take precedence over the use of chemical repellents. Removing, killing, trapping, or frightening the animal may be valid options. Studies have shown that controlling pocket gopher populations with strychnine baiting poses relatively little risk to non-target species (Arjo 2003). But the effects of removal may be short-lived since a replacement mammal usually occupies the vacant habitat, necessitating the continuous application of treatments. This option may also prove socially unacceptable (Schmidt and others 1992). Frightening devices are usually ineffective in deterring ungulates; however, other methods are currently under evaluation and testing (Nolte 2003c).

Biological methods may be useful for decreasing populations of unwanted browsers. For example, a recent study considered the interaction between weasels (*Mustela* spp.) and pocket gophers (Arjo 2003). In this study, 80 percent of the weasels killed and consumed healthy pocket gophers. All weasels ate strychnine-baited gopher carcasses 72 hours after gophers died, but no weasels died from secondary poisoning.

Successful Restoration Requires More Than Technical Expertise

Riparian restoration can be enhanced and successful only when treatments are integrated into a silvicultural system. However, the application of a silvicultural system by itself will not lead to a successful restoration project; other aspects also need ample consideration. Cannin (1991) summarized attributes characteristic of successful riparian restoration projects. Many were not technical application of treatments but rather the interaction of people in conducting the project. Strong leadership from a few designated people was critical, as was a political environment that promoted creativity, financial support, and effective implementation.

A multiple scale approach when planning projects is essential to recognize riparian zones as a part of the landscape rather than treating them as isolated areas. Pretreatment evaluation and surveys that clarify goals at the beginning allow participants to develop effective solutions to address problems. Post-treatment monitoring to evaluate success (or failure) allows for adaptive management. Increased public awareness through demonstration projects and proper land use practices should positively influence human behavior toward respecting sensitive riparian areas. Community involvement in project implementation is critical as is close communication between agencies, local governments, and landowners. In conclusion, it takes both technical and social expertise to implement a successful restoration project with ingenuity and imagination.

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Silvics and Silviculture in the Southwestern Pinyon-Juniper Woodlands

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***Abstract**—Southwestern pinyon-juniper and juniper woodlands cover large areas of the western United States. The woodlands have been viewed as places of beauty and sources of valuable resource products or as weed-dominated landscapes that hinder the production of forage for livestock. They are special places because of the emotions and controversies that encircle their management. Silvicultural methods can be employed on better sites to meet multiresource objectives and to maintain the health and sustainability of the woodlands. Silviculture must be based on an understanding of the silvics of the woodlands and their major species. Single-tree selection and diameter-limit prescriptions are being evaluated in central Arizona. Silvopastoral prescriptions that can maintain the tree component and provide for increased forage production and improved wildlife habitat are being tested in New Mexico.*

Introduction

Why are pinyon-juniper and juniper woodlands special places? Is it because they are uncommon? There are more than 47 million acres of coniferous woodlands within the western states and they are important landscape components in seven states (Evans 1988). The woodlands are divided into the Great Basin and southwestern woodlands. Pinyon-juniper woodlands cover approximately 7.7 million acres and associated juniper woodlands cover an additional 3.1 million acres in Arizona. Together the two woodland types, which will be considered together for this paper, comprise 56 percent of the forestland within the state (O'Brien 2002). The woodland types also are important in New Mexico, where they cover about 56% of the forestland or 8.5 million acres (Van Hooser et al. 1993). The woodlands are special places because of the emotions and controversies that their management generates among the diverse human populations of the Western United States. Some people view them as areas of natural beauty, an integral part of many southwestern national parks such as the Grand Canyon, Mesa Verde, or Zion. The woodlands are important to the cultural traditions and activities of the region's American Indian and Spanish people, some of whom depend on the woodlands for fuelwood, timber, and pinyon nuts; for habitats for game and species with ceremonial importance; and for medicinal crops and for grazing livestock. They provide a source of employment in areas where jobs are often scarce. The woodlands provide important watershed cover and are of increasing importance for recreation by the region's growing urban populations. On the other side are some interests who hold that the trees are weeds that are invading natural grasslands and that the best management is to remove them so that more forage can be produced for livestock. There are, of course, large ranges of opinions that fall between the two extremes.

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Pinyon-juniper woodlands produce a large variety of natural resource products and amenities. Silvicultural prescriptions can be used to sustain productivity of these lands for multiple resources and to maintain stand health. Silvicultural activities have the best chance of ecological and economic success on better sites. Approximately 88 percent of the pinyon-juniper and juniper woodlands have been classified as "high-site" indicating that they have the potential for growing wood products on a sustainable basis (Conner and others 1990). Silvicultural prescriptions can be formulated to enhance other resources such as wildlife habitat or forage for livestock.

The objectives of this paper are to review the silvics of southwestern pinyon-juniper woodlands and their component species and the relative merits of silvicultural options that have been applied to or proposed for woodland stands. The paper will then describe preliminary results from two ongoing silvicultural case studies. The first study involves an evaluation of single-tree selection and diameter-limit prescriptions in northern Arizona. Silvicultural prescriptions often are prepared to primarily benefit other resources such as wildlife habitat or forage production where the impacts on the tree products are secondary. The second case study describes three silvopastoral treatments that recently have been completed in central New Mexico to demonstrate that tree management and forage or wildlife habitat management can be compatible.

Pinyon-Juniper Woodland Silvics For Silviculturists

Tree and Stand Characteristics

The southwestern pinyon-juniper woodlands vary in species composition, density, and physiographic characteristics. At least 70 plant associations have been recognized in Arizona and New Mexico (Moir and Carleton 1987). Colorado pinyon (*Pinus edulis*), which has two-needles on a fascicle, is the most common pine within the type. Pinyon grows to between 9 and 35 ft in height and 5 to 18 inches in diameter. These slow growing trees may attain ages of 300 years or older on some sites in Arizona and New Mexico. Stands may contain one or several species of junipers; the four main species are oneseed juniper (*Juniperus monosperma*), Utah juniper (*J. osteosperma*), alligator juniper (*J. deppeana*), and Rocky Mountain juniper (*J. scopulorum*). Junipers usually are less than 40 ft tall. They can attain great age, but it is difficult to age most juniper trees because of the large number of false or missing rings. The floristic diversity in the woodlands is reflected in their herbaceous components rather than in the tree cover. While the total understory biomass may be small, the total number of species associated with the widely distributed woodlands is large (Gottfried and others 1995).

Most natural stands have an uneven-aged structure. In Arizona, total tree volume per acre averages 698 ft^3 in the pinyon-juniper type; net annual volume growth averages 6.4 ft^3/yr (Conner and others 1990). Clary (1987) reported that herbaceous understory plant biomass ranged from 78 to 1,042 lb/acre on seven sites in Arizona and New Mexico.

Autecology

Pinyon trees are generally monoecious, although dioeciousness may occur in trees stressed by drought or insect attack. The pine produces a relatively large, wingless seed, which weighs less than 0.01 oz (Ronco 1990). Seed crops are usually produced every four to seven years, depending on weather and site conditions. Cones mature in three growing seasons and seeds are released in mid-September and October. About 300 lbs of seed are produced on an acre in a good year (Ronco 1990). Seeds have a high nutritional value and are important food for wildlife. They are harvested by local human populations for personal consumption or sold commercially. Seed and cone insects sometimes reduce the amount of seed available for regeneration, wildlife, or human consumption.

Some of the southwestern junipers are monoecious, such as Utah juniper, and some are predominately dioecious, such as oneseed juniper (Johnsen and Alexander 1974). Seed-bearing age varies by species and by climatic conditions during seed development. Juniper "berries" contain one to four seeds depending on the species. The flowers of most southwestern junipers develop in the spring and the fruit ripens in the fall; some species require two years for the seeds to mature. Alligator juniper is the only major woodland tree that has the ability to regenerate vegetatively when the main trunk is injured.

The heavy mature seeds generally fall to the ground under or next to the tree crowns. Birds and small mammals are important for the wide dispersal of both pinyon and juniper seeds. Balda (1987) reported that four species of corvid birds, scrub jays (*Aphelocoma coerulescens*), pinyon jays (*Gymnorhinus cyanocephalus*), Clark's nutcracker (*Nucifraga columbiana*), and Steller's jay (*Cyanocitta stelleri*), are responsible for caching thousands of pinyon seeds during a year with a large crop. The birds cache the seeds in the ground and return in the spring to feed on the buried seeds. Seed that escapes the birds and rodents provide a main source for tree regeneration. Several birds, such as Townsend's solitaire (*Myadestes townsendi*), are important dispersal agents for juniper seed. Germination of oneseed juniper is enhanced after passing through a bird's digestive tract (Johnsen 1962).

Pinyon will germinate in the spring, but if conditions are not suitable, germination will be delayed until the summer monsoon period (Gottfried and others 1995). Most juniper seed germinates in the spring, but germination may be delayed because of embryo dormancy, chemical inhibitors, or impermeable seed coats. Juniper germination is generally less than 50 percent while pinyon germination is between 83 and 96 percent. Both trees are shade-intolerant, but germination and establishment is greater in the protection of mature trees, shrubs, and logging debris. Large cohorts of seedlings in the Southwest have been linked to the combination of bumper seed crops and favorable climatic conditions during the initial germination and establishment period. Seedling growth is slow, with root growth exceeding top growth in the early years. Growth of older trees of both genera also is relatively slow; a pinyon may grow to 12 inches in diameter in 150 years on a good site (Ronco 1990). However, pinyon grows at twice the rate of junipers (Conner and others 1990).

Synecology

Pinyon-juniper woodlands occupy the warmest tree-dominated zone in the southwestern United States. They are found from about 4,500 to 7,500 ft in elevation where annual precipitation ranges from 12 to 22 inches. Precipitation is influenced by geography, topography, and elevation. Differences in species composition have been related to the proportion of winter and summer precipitation (Springfield 1976). Woodlands are found on soils derived from a variety of parent materials.

The woodlands grade into juniper savannas, grasslands, oak woodlands, and brush-dominated vegetation zones on drier sites and into ponderosa pine (*P. ponderosa*) forests on moister, higher elevation sites. Junipers, which are more drought-tolerant, dominate on drier sites but the proportion of pinyon increases with increased elevation and available water. The upper and lower ecotones have shifted over time because of wildfires and decade-level climate fluctuations. The extended drought of the 1950s caused extensive mortality of all woodland tree species and caused shifts in ecotonal areas throughout the region. The woodlands increased at higher elevations replacing ponderosa pine stands, and grasslands or shrub ecosystems became more common at the lower elevations. Several insects, diseases, and parasites attack the trees, and insect infestations during drought cycles can result in high mortality over relatively large areas. Outbreaks of a bark engraver beetle, the pinyon ips (*Ips confusus*), during the current period of drought, are causing heavy pinyon mortality in the Southwest and southwestern Colorado (USDA Forest Service 2004). The juniper bark beetle (*Phloeosinus christatus*) is contributing to the mortality of drought-stressed junipers in areas throughout the Southwest. Pinyon dwarf mistletoe (*Arceuthobium divaricatum*) is an important parasite that causes locally severe damage and mortality. True mistletoes (*Phorodendron* spp.) are common on junipers but generally do not cause heavy damage. Fire was the most common natural disturbance prior to the introduction of livestock by European-American settlers. Fires were uncommon in the recent past because of the loss of a healthy and continuous herbaceous understory that could carry fire through the stands. Fire exclusion has been linked to increases in tree stand densities in the forests, woodlands, and savannas of the Southwest. However, pinyon-juniper woodlands will burn under severe conditions, and one of the impacts of the recent drought and associated insect mortality has been an increase in the intensity and frequency of wildfires within the woodlands. Successional stages in the woodlands usually contain the same species but in differing densities and dominance (Evans 1988). Junipers are the first tree species to return to a site after a fire or other disturbance but are often followed and replaced by pinyon.

Silviculture For Multiple Resources

There was a shift in attitudes toward pinyon-juniper woodlands after the oil crisis of the mid-1970s when the demands for firewood increased dramatically throughout the Southwest. Managers began to consider woodland management that would sustain healthy stands that could be managed for multiple resources. However, not all sites can produce the full range of resource benefits, and this must be considered in land management planning. Silviculture has the best potential for success on the most productive sites that can sustain the production of tree products based on soil properties, slope, and the presence of regeneration (Van Hooser and others 1993). Most pinyon-juniper woodlands in the Southwest have been classified as high-sites. There is a renewed interest in silvicultural systems and methods for the woodlands, especially on the more productive high-sites.

A number of silvicultural regeneration methods can be prescribed for pinyon-juniper woodlands (Bassett 1987), depending on the land manager's desired biological and economic objectives. Single-tree selection has a number of advantages since it favors natural regeneration of the main tree species, protects the site from wind and water erosion, can maximize vertical diversity important for wildlife, is easier to manipulate composition, and is esthetically pleasing (Bassett 1987). There are disadvantages since it is more difficult to plan and administer wood sales, residual trees can be damaged, horizontal diversity may be reduced over large areas, prescribed burning is not possible, and dwarf mistletoe control is difficult.

Other prescriptions, such as two-step or three-step shelterwood and group selection, are used in the Southwest. Clearcutting, which is the easiest prescription to plan and administer, is discouraged unless the objective is to increase forage and browse for livestock and wildlife, or to control dwarf mistletoe. Clearcuts are difficult to regenerate because of poor seed dispersal, except where alligator juniper, which sprouts, is a major stand component. Clearcuts are the least esthetically pleasing. However, the harvesting of narrow stripes of woodland or small openings is beneficial for deer (*Odocoileus* spp.) and elk (*Cervus elaphus*) because large homogeneous landscapes are broken up, providing food and adjacent hiding-thermal cover. While some private landowners may continue to remove the tree cover, many have recognized the values for their lands and livestock operations of creating mosaics of openings and woodlands, or of attempting to create savannas by retaining larger trees or groups of trees. Artificial regeneration of woodland species is not common because of the high expense but is used to reclaim mining sites and to restore vegetation around recreational areas following wildfires. However, artificial regeneration may be necessary if pinyon is to be restored in drought and insect-impacted woodlands. One treatment will not fit all situations and several may be valid within a landscape. New ecological knowledge and management techniques will contribute to future activities within the southwestern pinyon-juniper woodlands.

A Silviculture Experiment

The Rocky Mountain Research Station, in cooperation with the Black Mesa Ranger District of the Apache-Sitgreaves National Forests, Arizona, has completed the field phase of a study of several woodland silvicultural treatments, including single-tree selection and diameter-limit prescriptions, compared to changes in unharvested control plots. The diameter-limit prescription also could be characterized as the removal harvest of a one-cut shelterwood or an overstory removal, except that an upper diameter for residual trees was specified. The prescriptions were selected because they were being conducted by the District or were being considered for future management. The objectives of the treatments were to evaluate the effects of treatment on overstory characteristics and tree regeneration and to demonstrate the feasibility of these prescriptions for woodland management. A case study will be reported based on results from one of the single-tree selection plots and from one of the diameter-limit plots. Prescription planning was coordinated with the forest managers who administered the treatments as commercial fuelwood sales. Treatments had to be practical, considering the constraints of time and money, to be accepted by managers and fuelwood contractors.

The Study Area

The long-term study is located on the Black Mesa Ranger District of the Apache-Sitgreaves National Forests. The study site is 7 miles northeast of the town of Heber, which is approximately 110 mi northeast of Phoenix. Topography on the study site is relatively flat. Ephemeral stream channels that drain the area were not included in the study plots to reduce variability. Elevation is approximately 6,600 to 6,800 ft. Precipitation occurs during two seasons. Winter precipitation, usually snow, is produced by frontal storms that originate in the Pacific Ocean while summer monsoon precipitation occurs as convectional rains from moisture from the Gulf of Mexico. Winter storms produce about 55 percent of the average annual precipitation (with standard deviation) of 19.0 ± 3.3 inches, as measured at the Ranger District office from 1981 through 2001. Precipitation for the 12-year study period was 18.5 ± 4.2 inches. The soils are derived from undivided Cretaceous sedimentary rocks, mostly limestone, shale, and sandstone; most are classified as Lithic Ustochrepts or Udic Haplustalfs and have fine loams in the surface horizon (Laing and others 1987).

The woodland in the study area consisted of Colorado pinyon, oneseed juniper sites, alligator juniper, and occasional ponderosa pine. Pinyon is the most common tree species. Stand conditions in the general area had an average basal area of 101 ± 23.5 ft^2/acre and average canopy cover of about 40 percent (Laing and others 1987). The primary plant association is *Pinus edulis/Bouteloua gracilis* (USDA Forest Service 1997), which is one of the most common associations in Arizona and New Mexico. Cattle grazed the area during part of the study period, but use was minimal. Local residents had removed some large trees over the years prior to the study.

The preliminary results reported here for the single-tree selection and diameter-limit silvicultural treatment are from one replication (block) of a larger study. The prescriptions were applied to 10-acre plots. Each treatment plot contained 12 permanent circular 0.20-acre inventory plots. The treatments were randomly assigned among the four plots in the block, and inventory plots were located using a stratified random design. Measurements included species, diameter or equivalent diameter at root collar (d.r.c. or e.d.r.c.), height, disease or insect damage, crown characteristics, and tree defects or utilization. Equivalent diameter is necessary because most of the oneseed junipers are multi-stemmed with branching occurring at or near ground level. Tree seedlings were located within each inventory plot and pinned and numbered for re-identification. The blocks were measured in 1989; prior to treatment; in 1993; after harvesting; and in 2000. Changes in small mammal populations, understory responses, and soil-plant nutrient dynamics associated with the treatments were studied in some of the silvicultural treatment blocks (Kruse 1999, Kruse and Perry 1995).

Treatment Design and Administration

Single-Tree Selection

The single-tree selection prescription was based on the 1989 pre-treatment inventory that measured a total of 456 trees/acre and 150 ft^2/acre. The general objective was to sustain the production of tree products while maintaining the stand's uneven-aged structure, provide micro-sites for tree regeneration, improve stand health, maintain hiding and thermal cover for wildlife, and produce an aesthetically acceptable landscape. The immediate objective was to reduce the basal area of trees greater than 4 inches in diameter by about 60 percent while maintaining the existing structure. The desired maximum diameter for crop trees was 13 to 14 inches; however, some larger junipers were retained for wildlife and aesthetic considerations.

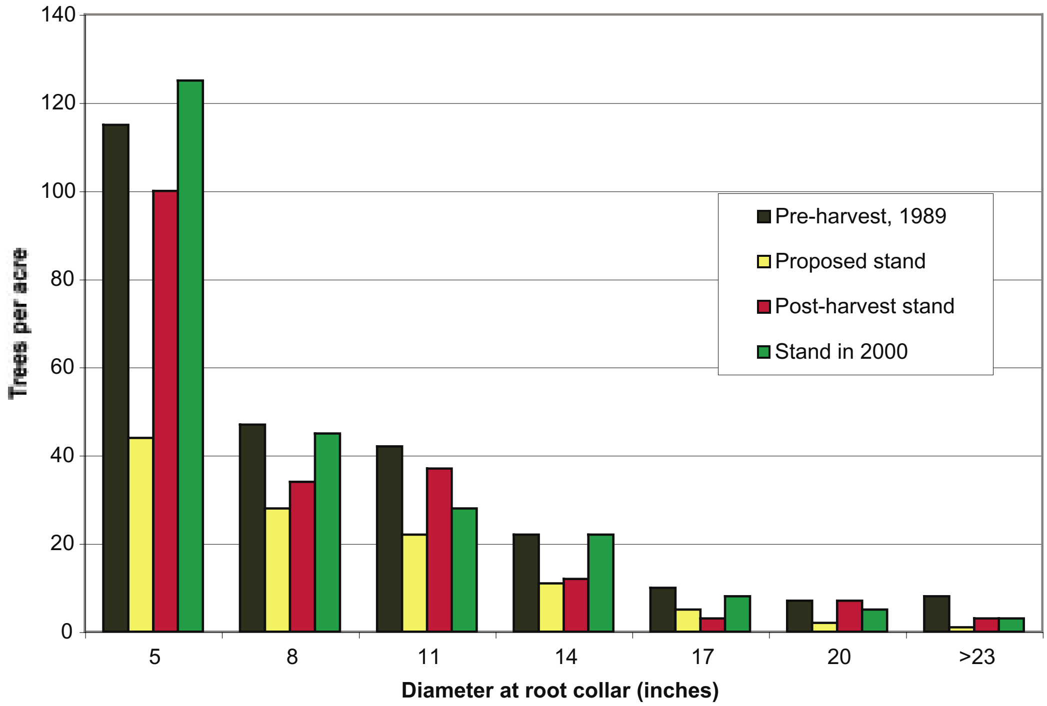

Figure 1—Initial, proposed, and post-harvest stand conditions in 1993 and 2000 for the single-tree selection block. The graph shows the changes related to the treatment and to growth and mortality among the trees. Diameter is measured at the root collar (d.r.c).

These large trees were considered when the inverse-J diameter distribution curve was defined. Regulation was directed to trees that were equal to or greater than 4 inch d.r.c., about 95 percent of the total basal area, because smaller trees do not have an economic value and it would be difficult to justify the tree marking costs to achieve the desired diameter distribution in these smaller trees. One objective was to keep the existing distribution of species in the stand. The desired number of trees in each diameter class was calculated using a "*q*-value" of 1.25 (figure 1), and a basal area target of 60 ft^2/acre. The *q* is the ratio of the geometric series that defines the number of trees in each successive diameter class (Husch and others 1972).

The Ranger District marked the residual trees within the harvesting block. The crew consisted of three people: a tally keeper and two measurers/markers. The crew was supplied with the desired stand structure and noted residual trees as they were measured and marked. Leave trees exhibited good vigor, had a potential for seed production, and were free of insect or disease problems. Higher basal areas were allowed in part of the area to keep high-quality trees. The guides also specified that cutting should not create new or enlarged openings of more than 0.25 acre. Markers used a 10 BAF wedge to maintain an average basal area of 60 ft^2/acre, and they were within 0.9 ft^2/acre of the target.

Diameter-Limit Prescription

The diameter-limit prescription was applied to another 10-acre plot. The stand on an average acre in the block had 438 trees and 142 ft^2 of basal area. The prescription called for the removal of all trees equal to or greater than 7 inches in diameter and the protection of remaining trees and regeneration classes. This prescription was similar to one of the common practices in the area, but one that has not previously been carefully evaluated. The logging debris was not burned on one block. Burning is a common practice but currently is questioned because of damage to residual trees, and because high-intensity fires can have a negative effect on soil nutrient dynamics (Teidemann 1987). Retaining the debris provides protected regeneration sites for trees and herbaceous plants, slows surface runoff and sedimentation, provides shelter for small mammals, and in some rural areas, is an important source for firewood (Gottfried and others 1995).

Results and Discussion

Single-Tree Selection

The block was harvested in December 1992; approximately 225 ft^3/acre were removed. Although the diameter distribution for larger trees was achieved (figure 1), stand density goals were not achieved because of the reluctance of the harvesters to cut smaller diameter trees. The post-harvest *q*-value met the goal of 1.20 but the harvesting did not achieve the basal area reduction goal for trees equal to or greater than 4 inches in diameter; only 36 percent of the stand basal area was removed leaving about 90 ft^2/ac. The graph shows the post-harvest and the present stand, including movement of trees among the diameter classes. One solution in the future is to give greater consideration to market preferences; it may be more realistic to regulate trees in the 7-inch and larger classes than to include the smaller sizes of trees. However, the impacts of dense groups of small trees on residual tree and stand growth still need to be determined. Approximately 678 trees/acre in the regeneration classes (85 percent) survived the harvest. The treatment did achieve the overall goals of retaining tree productivity, wildlife habitats, and of aesthetics. While an economic analysis was not part of the study, Ranger District personnel felt that they would recover the additional administrative costs from the amount received from a logging contractor for the wood. The effects of treatment on individual residual tree growth relative to growth on similar sized trees in the control block will be analyzed, as will the impacts of treatment on tree regeneration. However, the number of trees/acre increased in many size classes from 1993 through 2000, indicating increased growth of residual trees (figure 1).

Diameter-Limit Prescription

The diameter-limit harvest, without debris burning, removed 112 ft^2 /acre of basal area or 79 percent of the initial overstory cover, retaining 30 ft^2/acre, and removed 37 percent of the trees per acre, leaving 275 trees/ acre. The harvest removed about 375 ft^3 /acre of volume. Approximately 89 percent of the tree seedlings survived harvesting (515 trees/acre). Stand density in the diameter-limit block was similar in 1993 and 2000.

Some of the reductions in both blocks can be attributed to attacks and mortality by ips. The infestation that Wilson and Tkacz (1992) described occurred a short distance to the north of the study area. A 1993 inventory of herbaceous vegetation in harvested and un-harvested blocks indicated that harvesting increased the production of blue grama (*Bouteloua gracilis*) (the primary understory species), perennial forbs, and total herbaceous cover (Kruse and Perry 1995). Total production, for example, was 172 lb/acre in the treated blocks and 70 lb/acre in the un-harvested blocks.

Silvopastoral Prescriptions

The lack of commercial markets for alternative, higher-value juniper wood products limits management practices (Ffolliott and others 1999). In February 1999, the USDA Forest Service's Forest Products Laboratory and Rocky Mountain Research Station received a CROPS (Creative Opportunities) grant for the restoration demonstrations and workshops for management of pinyon-juniper savannas in New Mexico. The grant is part of an effort to develop new products and markets for the juniper resource that could improve the economics of treating these woodlands, not only for range restoration but also for more intensive management for sustainable tree products. Ongoing research projects at the Forest Products Laboratory in Madison, Wisconsin, have demonstrated the potential of value-added products from the wood and fiber of oneseed juniper. P & M Signs, Inc. in Mountainair, New Mexico, is using extrusion and injection molding technologies in the manufacture of sign panels and sign posts. The use of wood chips and fiber would increase the economic potential of woodlands dominated by smaller trees that are difficult to harvest for traditional products. The proposed manufacturing facility could influence management on a large part of the 252,402 acres of woodlands in Torrance County with its net volume of about 102,579,000 ft^3 (Van Hooser and others 1993). The facility would have a positive effect on employment and the general economy of Mountainair and Torrance County and adjacent areas.

Approximately 61 percent of the woodland area and 57 percent of the woodland volume are on private land in Torrance County. The goal of the project is to demonstrate to the landowners several ecosystem restoration prescriptions with the potential for economic wood and range products recovery while resulting in sustainable management. The plan is to use different techniques on three areas and to compare results with an adjacent untreated control site. The activity has resulted in two field workshops to provide participants with overviews of restoration approaches and in an evaluation of the economics of restoration including the value of products compared to the cost of treatments. Although the prescriptions were designed to integrate range and tree production objectives, the prescriptions could also be useful for treatments in pinyon-juniper dominated wildland-urban-interface areas.

The Demonstration Site

The demonstration was conducted within an area on the Greene Ranch in the Estancia Basin of Torrance County, New Mexico. New Mexico State University is studying the economics of the value of wood products relative to treatment costs in the same general area. It has six 1-acre plots that have been harvested by mechanized equipment (Bobcat) or by chainsaws (Maynard and others, unpublished report). Stand densities were reduced to 5-10 ft^2 /acre.

The site contains sandy soils that are 5 to 6 ft deep, and are representative of a band of soil that extends across the county. It is within a mile of the Gran Quivera Unit of the Salinas Missions National Monument and US Highway 54. The site is unique in the number of huge oneseed junipers that it supports; many have straight trunks with large diameters at breast height. This area is considered old-growth by local ecologists. The larger trees may date from the period when Gran Quivira was abandoned in the 1670s.

There is little surface erosion on the site that can be related to water movement probably because of high infiltration capacity of the sands. The area is grazed in winter but has a good cover of grasses, including blue grama (*B. gracilis*), side-oats grama (*B. curtipendula*), and sand bluestem (*Andropogon hallii*). Most grass is under the protection of larger junipers and there is less in interspaces. Larger natural openings within the area have a good cover of grass. This site is reserved for winter grazing partially because the tree cover provides thermal cover for the cattle. Average annual precipitation at the Gran Quivira National Monument was 15.4 inches between 1938 and 2001; most of the precipitation occurs during the summer.

Monitoring and Marking

The site was divided into four 20.3-acre treatment blocks, and a tree inventory was conducted in each block prior to marking the residual trees or designating prescriptions. Since the hope was to make this practical for ranchers and small acreage landowners, it was decided to arbitrarily limit sampling to 10 randomly located, permanent 0.20-acre fixed plots within each block. It later was apparent that either larger plots or more numerous plots would have given us a better idea of stand conditions because of the variability in each plot. Often 30 percent of the plots were non-stocked and others contained more than 32 trees/plot. The crew measured species and d.r.c. or e.d.r.c.; on some plots, total height was measured so that volumes could be determined. However, the permanent plots will be measured to provide an indication of post-treatment growth. Harvested trees are utilized for firewood, fenceposts, latillas, and vigas. Range resources were sampled on four transects in each block using a double sampling procedure (Maynard, J. personal correspondence, 2002). The average forage for each plot was: Block I with 260.2 lb/acre; Block II with 373.3 lb/acre; Block III with 585.9 lb/acre; and Block IV with 589.2 lb/acre.

All residual trees were marked within the blocks to be harvested. The goal was to maintain a relatively uniform crown cover within the limitations of the existing stand; however, groups of trees were retained along water courses and to maintain wildlife cover. Trees that had signs of wildlife activity, such as bird or rodent nests, were retained. Diameters were measured on all residual trees. The crew consisted of three people: two diameter measurers and one person who calculated and recorded the e.d.r.c. values. Leave trees were flagged in all directions around the tree.

The Prescriptions and Results

The specific prescriptions were designed to be general enough to be applied to juniper woodlands in a variety of different sites. The four treatments included a multiresource production block, a “savannarization” cut, a strip cut for wildlife, and an untreated control block.

The blocks were marked and harvested for firewood during the summer of 2002. A Bobcat equipped with a shear was used to fell trees in the savannarization and strip cut blocks. The trees were bucked for transportation and sale. The sustained multi-resource production block was harvested by chainsaw because there were concerns that the Bobcat would cause excessive damage to residual trees. At this time, not all of the wood has been removed from the site, so only the results of the harvesting can be reported. An evaluation of the impacts on forage production will wait until the wood is removed; however, the rancher recently has noticed more cattle and deer use in the treated blocks.

Sustained Multi-Resource Production

The prescription for the first treatment block (Block I) was designed to increase the herbaceous cover but still retain sufficient trees of all size classes in order to sustain the tree production option on these productive sites. The denser stand could have wildlife benefits for some small mammal and bird species. The prescription was designed to remove approximately 50 percent of the initial basal area but retain the variety of size classes present on the site. However, at least 65 percent of the crown cover should be left. The objective was not to force the residual stand into either an even-aged or uneven-aged structure, although the final result (figure 2) was a relatively all-aged stand. The marking favored healthy trees of all size classes in an attempt to retain younger trees to replace natural losses or additional harvesting. (Slash can be chipped for fiber as long as it can be done without damaging the residual trees.) Pinyon, which is a minor component of the block, and some snags were retained and protected for wildlife. This block contains some channels and signs of erosion, and slash was left in the channels to slow water movement and to trap soil. Groups of trees were retained for wildlife or for esthetic considerations. The final tally indicated that the residual stand contained 30 trees/acre and 29.4 ft^2/acre of basal area. The residual volume was estimated at 2.9 cds/acre. Preliminary estimates are that about 7 to 10 cds/acre were harvested but a final tally had not been conducted. Measurements of the inventory plots indicate that the residual basal area is 38 percent and the density is about 21 percent, respectively, of the original amounts.

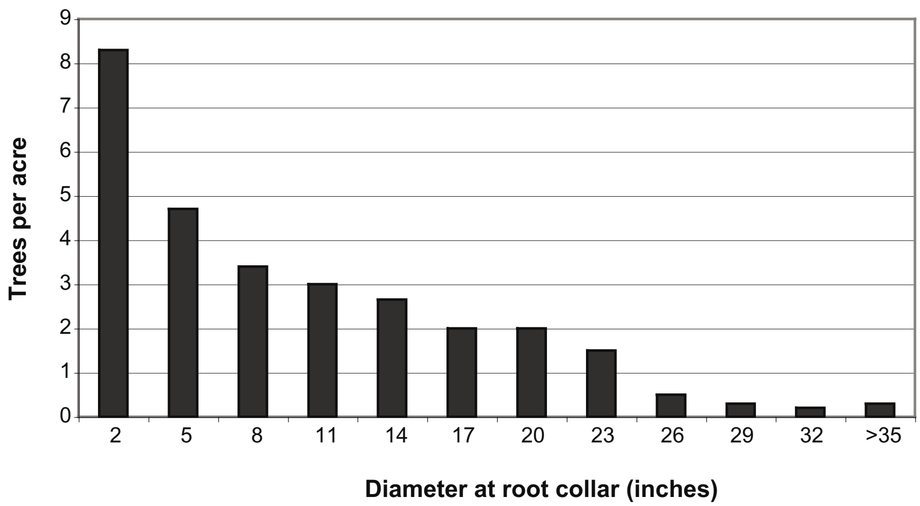

Figure 2—Residual stand on the multiresource silvopastoral treatment in New Mexico . The residual stand is uneven-aged and has a "q-value" of 1.08. Some of the largest trees are about 50 inches in d.r.c.

Savannarization

The second block (Block II) was treated according to a savannarization prescription. The objective is to restore the range value of the landscape by returning it to the savanna condition that probably existed prior to European settlement. However, no one knows exactly what conditions existed during the period, so managers must select an option. One option of leaving six trees/acre had already been applied to an experimental site near the Abo Unit of the Salinas Missions National Monument in the Cibola National

Forest (Brockway and others 2002), and it would not be that useful to reproduce it here. The selected prescription on the demonstration site was designed to leave a larger number of individual large trees or groups of large, medium, and small trees throughout the 20-acre block. The distribution of trees would not be uniform and would consider scenic views. The selected option was to leave between 15 and 25 large trees or groups of smaller trees per acre. Some areas would have no trees and others had more than 25 trees. One recommendation is that large trees should be retained on 40 to 60 percent of the area (USDA Forest Service 1993). The larger slash elements would be chipped and smaller material would be lopped and left for soil cover and regeneration protection. Some snags were retained and protected but were not counted as part of the residual stand.

The final mark indicated that 14 trees/acre in a variety of size classes had been retained on the savanna block (figure 3); this was 34 percent of the amount indicated by the pre-harvest inventory. The residual basal area was 26.3 ft^2/acre and the residual trees contained about 1.2 cds/acre.

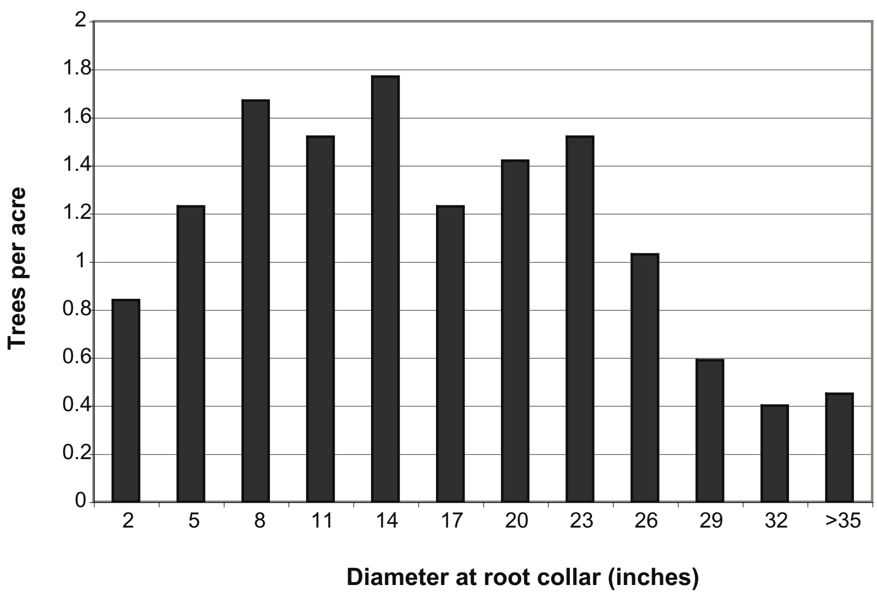

Figure 3—Residual stand for the savannaization silvopastoral treatment. The harvest left about 14 trees/acre. Approximately 1.2 cds/acre remain in the largest size classes.

Strip Cut

Research and observations throughout the West have indicated that wildlife do not move into openings that are too large, even when sufficient forage or browse is available. Animals tend to remain near the edges to take advantage of hiding cover. The general recommendation is that openings be limited to about 600 ft in width (Gottfried and Severson 1994) and that "leave areas" that border the strip be at least 200 and 330 ft wide (USDA Forest Service 1993, Gottfried and Severson 1994). The leave areas can be harvested but there should be sufficient residual density so that the animals will not be able to see through the stand to other nearby openings. Very open stands are treated as extensions of the opening and lose their value as hiding and thermal cover.

The final prescription for Block IV was to harvest a strip of 500 to 600 ft in width to run through the block and to cover about 12 acre. The strip was to have "feathered" edges and not be a regular rectangle and be oriented perpendicular to the direction of the prevailing winds to minimize soil erosion because of wind action on the sandy soil. The border strips were to be harvested to reduce stand density by 20 percent but had to retain a mix of size classes. The actual width of the border was closer to 300 ft since areas immediately outside of the plot were included. Some trees or small groups of trees were retained in the strip to break up and raise the wind flow. Unmerchantable slash was to be lopped and left on the ground to keep the wind above the soil surface and to provide protected regeneration sites for herbaceous generation. An administrative study on an alligator juniper (*J. deppeana*) site in central Arizona estimated that forage production increased to 809 lb/acre in openings where harvesting slash had been treated and to 1,366 lb/acre under slash (Soeth and Gottfried 2000). Larger slash could be chipped for the P & M plant or left on the site. Some snags in the strip and borders were to be retained and protected for wildlife. Critical nesting or birthing sites were to be identified and the plan altered accordingly.

The actual harvesting created a 13.1 acre strip in the middle of the treatment block; the base was 556 ft wide. The edges were feathered and 3.9 trees/acre were left in the strip to provide additional hiding or thermal cover. In addition, an average of 2.9 trees/acre were harvested in the border strips; this accounted for 14 percent of the strip basal area.

Control

The fourth block (Block III) was not treated and will be monitored to compare with the three treated units. The control is particularly important for herbaceous production and wildlife comparisons. It is anticipated that stand differences will not be great over the demonstration period.

Conclusions

There is a growing recognition that the southwestern pinyon-juniper woodlands are valuable and should be managed for multiple resources. Silviculture, based on a sound knowledge of silvics, provides a tool for multiple resource management. Several silvicultural systems and methods are applicable to the southwestern pinyon-juniper and juniper woodlands, but the prescription must be matched to stand and site characteristics and to the landowner's objectives.

Most woodland silvicultural prescriptions have been developed through adaptive management procedures often with little post-treatment evaluation. A case study was initiated in Arizona to evaluate several prescriptions with the objectives of providing managers with information that could be used in evaluating and planning treatments. The results indicate that single-tree selection is feasible for high-quality sites. The selection treatment met the objectives of sustaining tree production and maintaining habitat for woodland dependent species but full regulation and targeted density reductions are difficult because of the lack of demand for small diameter wood products. However, attitudes should change if markets develop for pinyon and juniper fiber. The stand continues to be esthetically pleasing and can sustain future entries on a relatively short cycle. It appears that residual trees are growing but it is not yet known if post-harvest growth exceeds normal growth in non-treated stands. The need for growth and survival information for the advance regeneration and new regeneration is important to the question of long-term sustainability. The more dramatic diameter-limit prescription reduced stand densities but accelerated growth of residual trees, and the survival of most of the advance regeneration should allow a more rapid return to productivity for tree products relative to more severe stand reductions. However, the diameter-limit area has been removed from general tree product production for a long period. The observed increases in herbage production should benefit livestock and some wildlife species.

The three silvopastoral treatments in New Mexico should show that tree production is compatible with forage production for livestock and wildlife. However, it is too early to make this assessment until additional range inventories can be conducted. The characteristics of the residual stands will provide hiding and thermal cover for animals and are esthetically more pleasing to most observers than cleared areas. The trees also provide a financial reserve for the ranches. The trees continue to grow and add volume. In some years, ranchers may earn more from selling firewood and vigas than from calf crops. Silvopastoral treatments are a viable option to tree eradication programs and also are applicable for treating woodlands in wildland-urban-interface areas.

The pinyon-juniper woodlands are important to many of our constituents—they are special places. Even our urban neighbors are becoming aware and concerned about the woodlands and lower ponderosa pine forests as drought, fires, and insects take their toll. The current natural onslaught is creating challenges to foresters and other land managers. What are we going to do with the areas that have suffered extreme mortality? Do we take an active approach to rehabilitation or do we allow nature to take its course? The loss of large areas of woodlands will put addition pressures on the remaining lands; not just by humans but also by wildlife that depend on the woodlands for all or part of their habitat requirements. It is my opinion that silviculture will become more important in the pinyon-juniper woodlands as we try to manage them for sustain and improved health and productivity. We have seen that there are a large number of silvicultural options that are appropriate to the woodlands and are available to us. New or modified prescriptions will be developed to fit the variety of stand and site conditions and management objectives. New scientific knowledge will contribute to future silvicultural prescriptions and management activities. The pinyon-juniper woodlands are worthy of our attention—and they are special places.

Acknowledgments

The assistance of H. C. Sanchez and J. H. Yazzie of the Rocky Mountain Research Station, Flagstaff, Arizona, and statistical assistance of R. M. King of the Rocky Mountain Research Station, Fort Collins, Colorado, are greatly appreciated. The study could not be conducted without the support and active collaboration of the personnel of the Black Mesa Ranger District, Apache-Sitgreaves National Forests, Arizona.

The New Mexico Demonstration Study would not have been possible without the support of the CROPS Grant Program. The contributions of J. Youngquist of the Forest Products Laboratory, H. Shanks of the Central Mountains R.C. & D., ranchers Brian and Lynn Greene, who gave permission to use the site and cooperated with the treatments, and the New Mexico Land's Department who administers the site are acknowledged. The field assistance of J. H. Yazzie and P. Niv is appreciated.

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A Prescription for Old-Growth-Like Characteristics in Southern Pines

Don C. Bragg[1]

Abstract—*Recent interest in adding old-growth reserves conflicts with a projected increase in the demand for forest commodities. However, managing for old-growth-like characteristics may permit timber production from stands designed to be similar to primeval forests. A silvicultural strategy based on presettlement forest conditions is being tested on 120 ac of mature loblolly (*Pinus taeda*) and shortleaf (*Pinus echinata*) pine on the Crossett Experimental Forest in Arkansas. Reference conditions from historical photographs, surveyor notes, old explorer journals, early research papers, and technical reports guided the prescription's design. A combination of harvesting, prescribed burns, and competition control should gradually produce structure similar to pine-dominated presettlement forests. Timber yield and natural attributes will be monitored and compared to traditional silvicultural practices to develop flexible prescriptions that can be modified later, if appropriate.*

Introduction

Old-growth forests have garnered considerable attention in public land management because they are often associated with higher levels of biodiversity, ecological complexity, aesthetics, and unique recreational opportunities. However, the desire for additional old-growth preserves conflicts with a projected increase in the demand for wood products. For instance, by 2040 the need for softwood fiber in the southeastern United States is forecast to increase more than 50 percent over current levels (Prestemon and Abt 2002). This level of consumption does not favor new areas being made available for unmanaged old forests, especially when almost 90 percent of the timberland in the southeast is privately or industrially owned (Wear and Greis 2002).

Public land managers are under increasing pressure to reduce their commercial timber production and alter their harvesting methodologies (Murphy and others 1993). In part, this is a response to widespread displeasure with clearcutting and monoculture plantations and a perceived timber bias in public land management. It also reflects a growing interest in matching anthropogenic disturbances with natural disturbance regimes (Aber and others 2000; Palik and others 2002; Seymour and others 2002). Furthermore, our value systems have shifted to include non-timber attributes like biodiversity, aesthetics, and water quality that may be compromised under intensive, short rotation monocultures.

We are just becoming aware of many of the complex patterns and processes involved in the formation and maintenance of old-growth (Aber and others 2000; Franklin and others 2002). Silviculture for old-growth-like characteristics permits harvesting in mature stands structured to better

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Figure 1—A perspective of contemporary forest conditions typical of Compartments 1, 2, and 12. Photo by D.C. Bragg in 2003.

resemble primeval forests. Note that artificially creating old-growth-like environments does *not* result in conditions identical to those from unaltered natural events. However, many old forest attributes can be encouraged in managed landscapes (Deal and others 2002; Guldin 1991; Morton and others 1991). For example, Lennartz and Lancia (1989) proposed the use of "creative" silviculture to enhance second-growth habitat for red-cockaded woodpeckers (*Picoides borealis*) by retaining larger trees and reducing midstory density.

A strategy for managing for old-growth-like conditions is being implemented on the Crossett Experimental Forest (CEF) in southern Arkansas. One hundred and twenty acres of mature loblolly (*Pinus taeda*) and shortleaf (*Pinus echinata*) pine will be transformed from an even-aged, relatively homogeneous stand (figure 1) into a multi-aged complex using a combination of group selection, competition control, and ecosystem management principles. This paper will outline the basic principles of a managing for old-growth-like upland pine forests, including the monitoring of project progress to determine the success of the effort.

Methods

Study Area

The study area is located in the Upper West Gulf Coastal Plain of southern Arkansas, on three 40 ac parcels on the CEF (figure 2). Compartments 1, 2, and 12 are relatively level, with slopes less than 3 percent. The soils adjacent to the drainage are Arkabutla silt loams, midslopes (comprising most of the area) are Bude silt loams, and Providence silt loams cap the low ridgetops (Gill and others 1979). A small, ephemeral stream runs down the west side of the study area. The CEF receives about 54 inches of precipitation annually, with average winter and summer temperatures of 47°F and 80°F, respectively (Gill and others 1979).

Currently, Compartments 1, 2, and 12 are dominated by loblolly pine, with a lesser component of shortleaf pine and hardwoods (table 1, figure 3). The woody understory consists primarily of hardwoods like sweetgum

Figure 2—Location of the study compartments and the Crossett Experimental Forest in Arkansas.

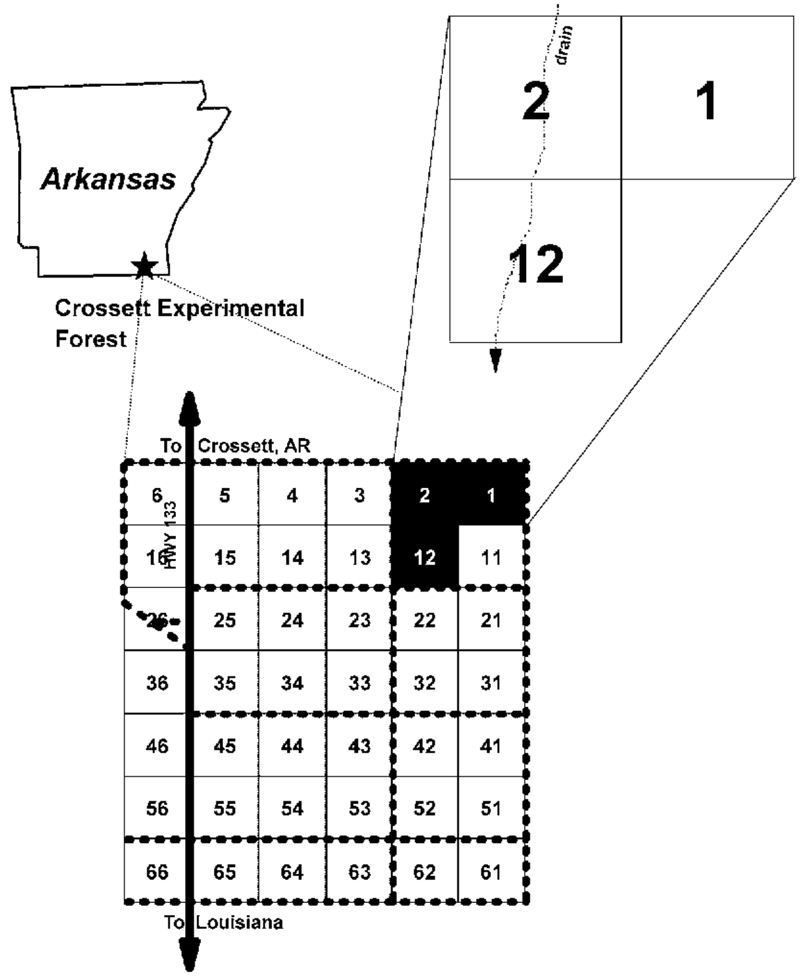

Table 1—Merchantable (DBH >3.5 inches) species composition of Compartments 2 and 12 only, sampled in the summer of 2000.

Species	Min. DBH	Mean DBH	Max. DBH	Live trees	Basal area (ft²)
	----------------	inches	----------------	--------- per acre	---------
loblolly pine (*Pinus taeda*)	3.9	15.2	28.5	40.49	57.69
shortleaf pine (*Pinus echinata*)	4.4	17.5	26.1	11.34	20.80
water oak (*Quercus nigra*)	4.1	8.6	16.2	7.56	3.48
sweetgum (*Liquidambar styraciflua*)	4.0	6.8	21.7	10.80	3.31
white oak (*Quercus alba*)	3.6	7.5	14.9	8.10	2.85
winged elm (*Ulmus alata*)	3.8	5.6	9.7	9.18	1.73
cherrybark oak (*Quercus pagoda*)	4.3	6.7	8.7	2.97	0.76
red maple (*Acer rubrum*)	3.8	6.3	10.0	3.24	0.75
flowering dogwood (*Cornus florida*)	3.8	4.8	6.3	3.51	0.45
southern red oak (*Quercus falcata*)	5.8	7.9	13.6	1.08	0.43
blackgum (*Nyssa sylvatica*)	4.0	6.3	8.3	1.35	0.32
black cherry (*Prunus serotina*)	3.8	6.5	8.3	1.08	0.27
post oak (*Quercus stellata*)	4.4	5.5	6.8	0.81	0.14
green ash (*Fraxinus pennsylvanica*)	3.9	4.7	5.5	1.08	0.13
red mulberry (*Morus rubra*)	4.6	5.2	5.6	0.81	0.12
blackjack oak (*Quercus marilandica*)	9.0	9.0	9.0	0.27	0.12
sassafras (*Sassafras albidum*)	5.3	6.4	7.4	0.54	0.12
American holly (*Ilex opaca*)	3.8	4.8	6.7	0.81	0.11
black oak (*Quercus velutina*)	7.7	7.7	7.7	0.27	0.09
eastern redcedar (*Juniperus virginiana*)	6.4	6.4	6.4	0.27	0.06
mockernut hickory (*Carya tomentosa*)	3.6	4.4	5.2	0.54	0.06
willow oak (*Quercus phellos*)	3.9	3.9	3.9	0.27	0.02
			TOTALS:	**106.37**	**93.81**

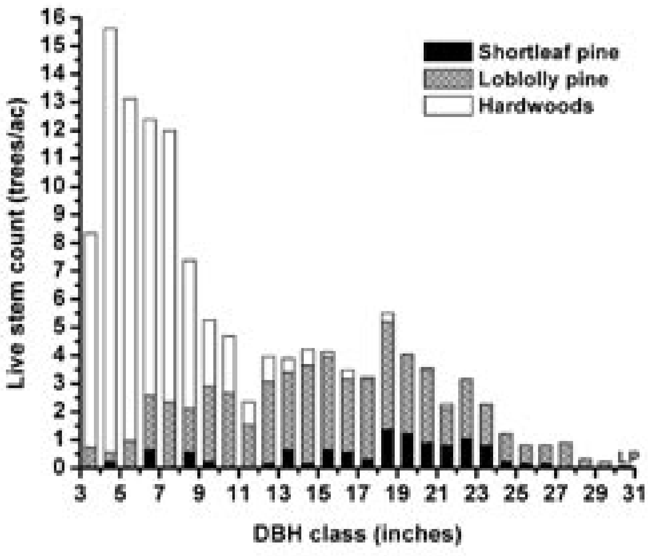

Figure 3—The 2000 inventory of Compartments 2 and 12, featuring hardwoods and pines by size class.

(*Liquidambar styraciflua*), red maple (*Acer rubrum*), and white oak (*Quercus alba*), with a large component of briars (for example, *Rubus* spp., *Smilax* spp.), vines (for example, *Vitis* spp., *Toxicodendron radicans*), and shrubs (for example, *Callicarpa americana*). There is virtually no pine in the understory and very little midcanopy except for some scattered hardwoods.

Reference Condition Acquisition

Reference conditions for presettlement pine forests in the Upper West Gulf Coastal Plain were developed from a number of historical sources, including photographs, surveyor notes, old explorer journals, and early research and technical reports. This work helped guide the old-growth-like prescription's design and implementation by quantifying key attributes of stand structure and composition to use as silvicultural targets (table 2).

Table 2—Proposed reference targets for restoring old-growth-like upland pine stands on the Crossett Experimental Forest in southeastern Arkansas.

Attribute	Reference target	Implementation strategy
Species composition	50 to 60 percent loblolly 35 to 45 percent shortleaf up to 10 percent hardwoods	Preferentially cut loblolly and hardwoods
Basal area	50 to 70 ft^2/ac	Group selection and periodic thinnings
Maximum tree DBH/age	unlimited	Avoid cutting trees >25 inches DBH
Number of big trees	5 to 15 pines >30 inches/ac	Cut no pines >30 inches DBH
Reserved timber volume	5000 to 10,000 board feet/ac	Volume reserved solely in "keepers"
Spatial pattern	patchy	Group selection with reserves
Under/midstory	open	Fire and herbicide
Red heart	10 to 50 percent cull in retained trees	Old trees, fungal inoculation(?)
Large woody debris	5 to 10 snags/ac 285 to 715 ft^3/ac	No salvage, girdling, hot fires

Historical information used to define reference conditions was preferred to contemporary studies of old-growth pine remnants (for example, Cain and Shelton 1994; Fountain and Sweeney 1987; and Murphy and Nowacki 1997) because of the impact of decades of fire exclusion, exotic species introduction, and other alterations to the original structural, compositional, and functional behavior of these reserves. For instance, evidence suggests that presettlement upland forests had a much greater proportion of shortleaf pine than modern examples (Bragg 2002). Other traits common to the presettlement pine forests of southern Arkansas included open, relatively poorly stocked stands with an abundance of grasses and forbs and fewer woody stems and vines, substantially higher levels of old, very large, and frequently decadent canopy pines, and sporadic but locally considerable volumes of coarse woody debris (Bragg 2002, 2003).

Treatment Implementation

Using a combination of harvesting, competition control, and adaptive management, the study compartments will be gradually converted into a stand similar in composition, structure, and dynamics to the pine-dominated presettlement forests once common to the region. Adaptability is key to this silvicultural prescription: there is no absolute, immutable recipe for producing an old-growth-like forest. For instance, we believe that we must incorporate the ability to adjust, modify, or even redesign some aspects of its implementation if suggested by monitoring.

Flexibility…flexibility…flexibility. Adaptive management strategies based on effective monitoring and the response to unforeseen change will help achieve the desired prescription. Given the duration of this effort, it is inevitable that unanticipated events (droughts, excessive rain, beetle outbreaks, windthrow, wildfire) will complicate the restoration. Such disturbances are not necessarily a problem unless the merchantable timber is completely lost. After all, presettlement forests were characterized by their large volume of dying and dead trees (biological legacies that contributed to ecosystem complexity). However, since a major objective of this prescription is to produce some timber products, limited salvage or preemptive thinning to avoid catastrophic loss may be required. Delays in harvesting due to bad weather, unexpected slow growth, or weak timber markets are also inconvenient but not crippling.

In this region, an operable cut typically contains 1500 to 2000 board feet (Doyle log rule)/ac in sawtimber, with pulpwood usually supplementing the sawlog yield. If a typical stand grows 300 to 400 board feet per acre per year, this results in 5-year cutting cycles. Longer cutting cycles than traditionally applied (for instance, 10-year versus 5-year) will probably be needed to provide the desired structural and compositional control. Prolonged cutting cycles are important because it may take longer to grow sapling- and pole-sized pine to sufficiently large size under these conditions, particularly when fewer large pines are available for harvesting. Extended harvest return intervals should also help avoid unnecessary logging damage, especially to the smallest merchantable size classes.

Anticipated Competition Control

Long treatment cycles should also allow most of the advanced pine regeneration to survive periodic controlled burns (Cain 1993). Prescribed fire will be an important component of this study for several reasons. First, it consumes the litter and duff and improves pine establishment. Second,

Figure 4—A "decadent" 54-inch DBH loblolly pine from Ashley County (circa 1937). If sound, this tree would have scaled 7,000 board feet (International rule), but note the prominent cankers and scars. Photo #350916 in CEF (USFS) archives.

it provides some degree of competition control. As an example, efforts to control woody vines may especially benefit from the return of fire. Third, controlled burning encourages the return of native fire-dependent grasses, forbs, and shrubs that have largely disappeared under traditional forest management. Finally, the fire-related wounding and subsequent decay of large trees helps to reintroduce decadence absent in most managed stands (figure 4). This is significant because punky, hollow, or dead trees provide critical habitat for cavity-dependent species.

However, there is only so much that can be achieved with controlled fire in the fragmented forests of southern Arkansas. Issues of smoke management, liability, and the ecological timing of the burns represent major challenges. In addition, excessive burning can drastically understock pine stands and introduce too much decay (Bruner 1930), reducing the potential of timber harvesting to support the overall restoration effort. Hence, some chemical competition control will almost certainly be needed to achieve management objectives. Hardwood and woody shrub rootstocks are often so well established that most controlled burns do little more than topkill. Given their ability to resprout, these competitors have a distinct edge over seed-origin pines. Experience has shown that an appropriate mixture and timing of herbicides and controlled burning can effectively reduce hardwood and brush competition (Cain 1993; Zedaker 2000).

Hardwood Management

As can be seen in figure 3, many small hardwoods are found in the study compartments. Hardwoods were a minor component of the presettlement pine forests of the region, usually constituting less than 25 percent of the stand (Bragg 2002; Chapman 1913; Reynolds 1980). The hardwood-filled drain in Compartments 2 and 12 will be treated as a riparian management zone, with very few of the hardwoods removed. Small hardwoods in the upland forest zone are available for cutting, girdling, or spraying, but a handful of large oak will be retained to preserve some mast.

Monitoring

As mentioned in the preceding paragraphs, monitoring is an important component of any long-term strategy because it tells the manager if the treatment has been implemented as designed, and if not, what needs modification. Close supervision should also help alert the manager to growing pest or competition problems that may require unscheduled intermediate treatment(s). Furthermore, monitoring can facilitate the prediction of future conditions.

If overall structural or compositional targets are not consistently reached, either the targets or the thinning strategies must change to meet the desired objectives. However, we must manage for a historical range of a suite of acceptable stand features, not a narrowly defined and singular density or compositional target (Trombulak 1996). Irregularity and heterogeneity are key attributes of old-growth forests. For this reason, table 2 identifies ranges of a number of characteristics expected in old-growth stands. As an example, the spatial distribution of stems in the presettlement forests of southern Arkansas (figure 5) lacked the consistency of most managed forests (Bragg 2002; Chapman 1912). Hence, some locations would match the "average" stocking range, while others are denser or more open.

Timber production and natural attributes will be compared to traditional alternatives (intensive timber yield and no harvest reserve) to help identify the economic trade-offs of managing for old-growth-like attributes. For example, it is expected that the relatively understocked, lightly cut old-growth-like prescription will produce noticeably less fiber. Other non-timber attributes will also be tracked to more fully evaluate the success of the system.

Figure 5—Example of a possible spatial pattern of the original pine stands of southern Arkansas circa 1910 (adapted from Bragg (2002) and Chapman (1912)). The dark areas represent individuals or small clusters of old, declining pine, the stippled area is large, vigorous pine, and the white areas are young timber.

Anticipated Results

Compartments 2 and 12 have been marked using thinning from below to reduce stand basal area to approximately 65 to 70 ft²/ac. No trees greater than 21 inches in DBH were selected, and all shortleaf pine will be spared in this harvest. Compartment 1 was only recently added to this study, and since it was harvested about 3 years ago, it was not remarked for treatment. Structurally and compositionally, Compartment 1 differs little from 2 and 12, although it will be a few cutting cycles before the compartments are fully integrated. The desired stand structure (encapsulated in figure 6) is the critical result, not the starting condition or developmental path of any given compartment. Using group selection and thinning from below, the first treatment cut (scheduled for 2008) will reduce average stand density to approximately 60 ft²/ac.

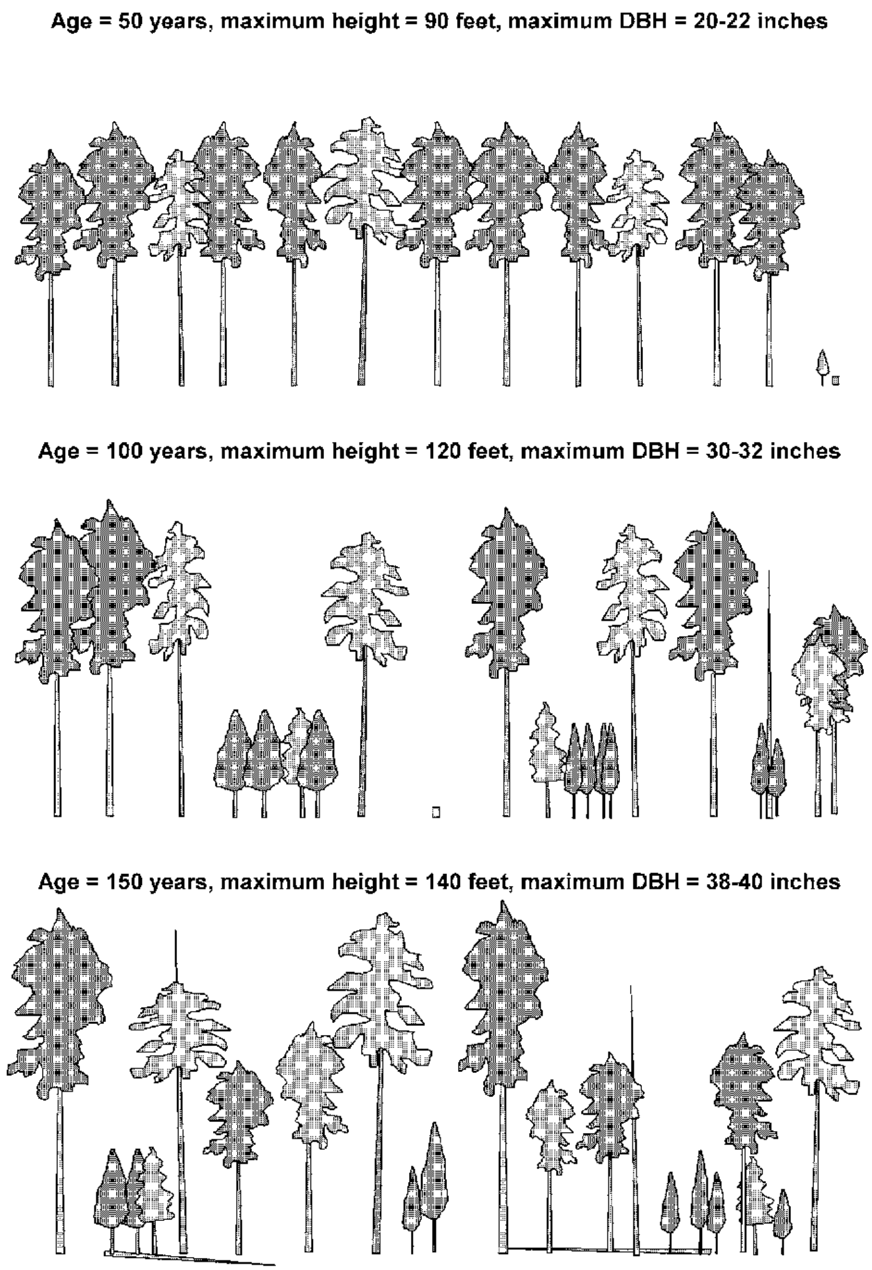

Figure 6—Temporal sequence showing the idealized developmental trajectory of the old-growth-like study compartments. Starting from a relatively even-aged, 50-year-old stand, repeated harvests and natural mortality gradually open the stand, which by year 100 has numerous regenerating gaps, and contains a relatively small number of large "keeper" pines with multiple patches of varying age by year 150.

Keepers, Groomers, Thinners, and Regeneration

Rather than following a predetermined and fixed rotation length (for example, 150 years in loblolly/shortleaf pine), certain individual trees ("keepers") will be identified and permanently excluded from regular timber marking. This will allow them to reach their biological lifespan, perish of natural causes, and eventually fall to the earth and decompose (unless specifically identified as an unacceptable hazard). Keepers will exceed 30 inches DBH and may range from poorly formed culls to prime crop trees. Keepers may be found individually or in clumps, but rarely in patches larger than a fraction of an acre. The residual volume held in keepers will eventually range from 5000 to 10000 board feet (Doyle)/ac, comparable to presettlement forests (Bragg 2002) (a single 30 inch DBH pine contributes about 5 ft^2 of basal area and scales approximately 1400 board feet).

Pines from 20 to 30 inches DBH will be treated as "groomers" in which the most "eligible" individuals are destined for a future as keepers. Groomers will be continually evaluated to ensure they contribute to long-term stand goals. Groomers that show promise as long-term keepers will be preferentially retained, while others will be harvested as appropriate. Most large groomers that perish will also be left to supplement the large dead wood pool.

The merchantable-sized pines less than 20 inches DBH are called "thinners." Thinners may range from saplings barely making pulpwood (3.6 inches DBH) to prime sawtimber 17 to 20 inches DBH During the years a pine grows from saplings to sawlogs, good forestry practices should be encouraged. Hence, cut the worst to favor the best. Remove the poles bent by glaze accumulation or afflicted with fusiform rust (*Cronartium fusiforme*). Aggressively thin the maturing groups to encourage rapid growth, but protect the residual stand from unnecessary logging damage by encouraging the loggers to leave tops and defective logs in the woods.

Regeneration will be achieved via variable-sized group selection openings. These gaps should range from 0.25 to 1 ac, often with keepers scattered amongst them. Since loblolly and shortleaf pine are shade intolerant species, most gaps will cover at least 0.5 ac. Once established, it is critical that the regeneration be protected to ensure that the gap maintains adequate stocking.

Treatment Timeframe

The objective of this effort is to gradually convert a relatively even-aged, mature forest into a patchy mixture of immature, mature, and old timber similar to the presettlement upland pine forests of southeastern Arkansas (figure 7). Thinnings may vary depending on access, markets, and growth. Competition control treatments will be scheduled to ensure that pine regeneration benefits the most from overstory and understory release. However, an extended period between seedbed preparation and controlled burns is needed so that enough pine saplings get large enough to survive the fire. Chapman (1952) recommended an 8- to 10-year burn interval for loblolly pine-dominated ecosystems.

Figure 8 provides a framework for treatment applications, including the long-term application of group selection with reserves. Since one of the goals of this prescription is to maintain an average stand basal area of 50 to 70 ft^2/ac, this means that locally some areas will average less than 30 ft^2/ac, while others will exceed 90 ft^2/ac. If a well-stocked stand on the CEF adds 3 ft^2/ac of basal area annually (Baker and others 1996), then it is capable of growing 30 ft^2/ac in 10 years. This longer cutting cycle should allow for low

Figure 7—An image of presettlement pine forest stand structure in southern Arkansas. Photo by Russ Reynolds, circa 1935.

YR	ACTION
0	Harvest treatment
1	Herbicide/burn
2	
3	
4	Regeneration check
5	Mid-rotation thin(?)
6	
7	
8	Controlled burn
9	Pre-harvest cruise

Figure 8—A possible schedule of treatment actions under regulated management for old-growth-like stand characteristics in loblolly and shortleaf pine-dominated stands in southern Arkansas.

density areas to recover to the desired level. The denser areas will be more heavily cut when conditions are suitable. Mid-cycle thinnings may occur if an operational harvest volume is available.

Conclusions

A prescription that focuses on old-growth-like forests requires a dedicated, flexible, and long-term commitment to the treatment. Unlike many other timber operations, the desired outcome of this strategy may not become apparent for many years. Close monitoring of key stand attributes like species composition or big tree numbers is vital to help adjust treatments over time, with the sustainable achievement of structural and composition complexity (figure 6) being the true measure of success.

In principle, managing for old-growth-like characteristics appears to be a workable compromise between sustainable timber yield and functional old forests. This type of silviculture may seem inefficient, but since the primary objective is old-growth-like structure rather than commodity production, some irregularity is desirable. The effort involved in this project, the extension of the rotation period, and the reduction in timber yield are not likely to make this strategy widespread on the Gulf Coastal Plain, but when implemented, many other non-timber benefits should be realized.

Acknowledgments

The author would like to thank O.H. "Doogie" Darling, Jim Guldin, Ernest Lovett, Mike Shelton, Bruce Walsh, and many others for making this research possible. Special thanks go to the Crossett Public Library for access to their historical archives and photographs. Robert Deal and Michael Shelton provided reviews of an earlier draft of this paper.

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Spatial Distribution of Ponderosa Pine Seedlings Along Environmental Gradients Within Burned Areas in the Black Hills, South Dakota

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Abstract—*In 2000, the Jasper fire in the Black Hills, SD, created a mosaic of burned and unburned patches of different sizes within the contiguous ponderosa pine forest. To study the spatial regeneration of ponderosa pine seedlings and the ecological gradients existing between burned and unburned areas two years after fire, we used a transect approach. We demonstrated that seedling establishment was prolific within the peripheral part of the burned areas due to the presence of seed sources close by, the seedbed conditions, and the relatively low competitive pressure. This transect study provides information to consider when managing forests after fire.*

Introduction

Regeneration of *Pinus ponderosa* (ponderosa pine) has been previously described as prolific in the Black Hills of Wyoming and South Dakota (Shepperd and Battaglia 2002) compared to the common pattern encountered in the Rocky Mountain and Southwest areas. This prolific regeneration is due to local climate conditions and more particularly to abundant precipitation occurring during spring and early summer. However, the unprecedented size and severity of the fire that occurred in 2000 in the Black Hills (Jasper fire) may affect the future establishment of ponderosa pine seedlings in this area. Ponderosa pine has long been considered a species very well adapted to fire (Vlamis and others 1955; Weaver 1967, 1974) because of its high tolerance to heat and high regeneration abilities after fire. Dispersal distance of ponderosa pine seeds is considered to extend to approximately 75 m to 100 m from the seed trees (Barrett 1966), with a minor influence of secondary dispersion factors on dispersal distance (Vander Wall 1997). Oliver and Ryker (1990) reported 8 percent seedfall at 120 m from seed source in central Oregon. Seed dispersal distance is not the only factor affecting ponderosa pine seedling distribution. Post-fire environmental conditions may also affect seedling establishment. Ponderosa pine has propensity for developing seedlings on mineral soil seedbeds, such as after severe fire (Harrington and Kelsey 1979). The seedlings benefit from the ash-covered soils, whereas unburned compacted litter would allow little moisture for seedling survival (Biswell 1973, Harrington and Kelsey 1979). The reduction of litter by fire favors ponderosa pine seedlings by reducing shade and mechanical obstructions to seedling emergence (Schultz and Biswell 1959). Finally, water and nutrient supply can limit the establishment of ponderosa pine after fire, in conjunction with the competition pressure that herbaceous species can exert on seedlings (Elliott and White 1987, Larson and Shubert 1969). Consequently, areas subject to high herbaceous recovery after the Jasper fire may obstruct ponderosa pine seedling establishment.

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We set out to determine the favorable and unfavorable environmental conditions for seedling establishment within the Jasper fire. We estimated the spatial distribution of seedlings by using transects running from unburned areas to the center of burned areas. This paper discusses the benefits of using a transect-based approach for defining areas where rehabilitation operations can be useful and other areas where they could threaten seedling regeneration.

Study Area

The study area is located in the south-central Black Hills (South Dakota) on the western section of the limestone plateau on the Black Hills National Forest. The elevation varies from 1800 m to 2200 m. The southern Black Hills is characterized by a continental climate, with average annual precipitation around 400 mm (Driscoll and others 2000), mostly occurring from April to July. Mean annual temperature is around 9 °C, with a mean annual high of 17.2 °C and mean annual low of 2.1 °C (Shepperd and Battaglia 2002). The vegetation is mainly ponderosa pine forests, with *Populus tremuloides* (aspen), *Quercus macrocarpa* (bur oak), and *Picea glauca* (white spruce) groves. Pine regeneration in this area, under natural conditions or silvicultural treatments, is among the most successful in the western United States (Shepperd and Battaglia 2002) and most of the germination occurs in early summer, during the months of May and June when precipitation is abundant and temperature starts increasing. The Jasper fire burned 33,000 ha of ponderosa pine forests in August and September 2000 in the Black Hills of South Dakota, and it was the largest fire recorded in the history of the Black Hills. It created a variety of competitive environments. Our study took place within this area two years after the fire, from May to October 2002.

Methodology

We established 20 transects within 10 burned patches of different sizes (two transects per patch). To allow a regular and dense sampling pressure, we prioritized the utilization of a high number of small plots. To enhance the effects of distance on seedling establishment and environmental variables, we chose rectangular plots having their longest side (6 m) perpendicular to the line of the transect and the smallest side (2 m) parallel to the transect. The distance between each plot was chosen to be twice the width of the plots to favor independence between plots. Transects began 36 m within the unburned areas (-36 m) and extended to the center of the burned areas (distance depending on the size of the burned patch). The unburned edge corresponded to 0 m. The shortest transect was 66 m long (from -36 m to +30 m) and the longest was 288 m (-36 m to +252 m).

Within the plots, we counted the new 2002 germinated ponderosa pine seedlings. We recorded the ground and vegetation conditions, the floristic richness, the topographic conditions, and other tree characteristics, at 2 x 6 m scale (total of 24 variables). The 24 variables measured in each 2 x 6 m plot are summarized in table 1 and all the variables are described thereafter.

Table 1—Environmental variables taken into account to explain seedling distribution.

Ground covers	BGR	% cover burned ground
	BL	% cover burned litter
	BSH	% cover burned shrubs
	CRY	% cover mosses + lichens
	GRV	% cover gravel
	RCK	% cover rocks
	UGR	% cover unburned ground
	SL	% cover scorched litter on burned ground
	UL	% cover unburned litter on unburned ground
	VEG	% cover understory vegetation
	WDB	% cover woody debris
Topography	ASP	Aspect
	SLP	Slope
Vegetation covers	BTRA	% cover dead trees <10 m
	BTRB	% cover dead trees >10 m
	UTRA	% cover alive trees <10 m
	UTRB	% cover alive trees >10 m
	C	% cover shrubs
	D	% cover stratum 0.1 - 0.5 m
	E	% cover stratum <0.1 m
	TOTC	% cover whole vegetation
Other variables	MeanDm	Mean tree DBH
	MeanHg	Mean tree height
	BasArea	Basal area
	Moist	Soil moisture
	Temp	Soil temperature
	Richn	Floristic richness
	Tree #	Tree density

Ground Conditions Per Plot

1. Within the 2 x 6 m plots, ground cover appearance was measured using estimated percentages of: unburned ground (UGR); unburned needle litter (ULUG); scorched needle litter on burned ground (ULBG); burned ground (BGR); burned needle litter (BLi); Cryptogam, i.e., bryophytes in this study (Cry); rock (RCK); gravel (GRV); woody debris (WDB); and the total cover of understory vegetation (E). The total cover of these variables equaled 100 percent of the ground cover seen from above and excluding the overstory canopy.
2. Vegetation conditions per plot: The percent cover and burn status of vegetation per stratum (0-10 cm high = stratum E; 10-50 cm high = stratum D; 50-200 cm high = stratum C; burned shrubs 50-200 cm high = stratum BSH; "unburned" trees <10 m = stratum UTRB; "burned" trees <10 m = BTRB; "unburned" trees >10 m = UTRA; "burned" trees >10 m = BTRA) and understory + overstory vegetation (TOTC) on the ground were also recorded. The "burned" or "unburned" status of the trees was established in relation to the percentage of the needle canopy burned (>50 percent / <50 percent). The total cover of these different strata may be greater than 100 percent.
3. Topography: Slope (SLP) and aspect (ASP) were recorded for each plot.

4. Other variables per plot: The number of trees (>3 m tall and with DBH >7 cm) (Tree#) was counted and their diameter and height were measured and means were calculated for each plot. The floristic richness, defined as the number of species in each plot (Richn), was measured for information about plant diversity.

Statistical Analyses

The distribution of the ponderosa pine seedlings along the transects was analyzed using non-linear regressions. Relationships were explored between average seedling density in June 2002 and distance from the unburned areas. Nineteen out of 20 transects were used for the analyses, from -36 m up to 150 m; plots located farther than 150 m were used only as observations due to low sample size. One transect was not used for analysis because of the high number of seedlings in some of the plots (up to 115 seedlings per plot at -12 m), corresponding to 55 times the average number of seedlings in any other plot.

To test the influence of environmental variables on seedling emergence in June, a PLS (Partial Least Squares) regression was performed between the 28 environmental variables, used as explanatory variables (table 1), and the number of seedlings per plot used as the dependent variable. This analysis was conducted on the 406 individual plots located along the 19 transects at up to 150 m from the edge. The PLS provides for each explanatory variable a regression coefficient, either negative or positive, with the dependent variable, and thus a hierarchy of the influence of the explanatory variables on the dependent variable.

The regressions were created using SPSS (SPSS Inc., Chicago IL) and Statistica (Statsoft France 1997); the plots within transects were examined using SAS Proc Mixed (SAS Institute Inc., Cary NC) and found to be relatively uncorrelated (serial $\gamma = 0.19$). The PLS were performed using ADE-4 (Thioulouse and others 1997).

Results

Average seedling density in June across the 19 transects varied with the distance from the edge (figure 1). This transect approach allowed us to determine different distances from the unburned areas where a change occurred in the distribution of the seedlings: (1) from -36 to -6 m, the distribution was regular and the average seedling density was about 350 seedlings per ha; (2) from 0 to 24 m, the seedling density increased to an average of 660 seedlings per ha and reached up to 1125 seedlings per ha at 12 m; (3) from 30 to 60 m, the seedling density decreased drastically with an average density of 110 seedlings per ha; and (4) after 60 m from the unburned areas, the seedling density averaged 40 seedlings per ha. In addition, some seedlings were encountered at up to 180 m from the unburned areas.

To explain the irregular distribution of the seedling density, the quantifiable environmental conditions were studied as explanatory variables of the number of seedling per hectare. The PLS regression performed between the 28 variables and the seedling densities per ha (figure 2) demonstrated that the seedling densities were explained by one combination of the different variables (first component significant $p = 0.04$) and were positively

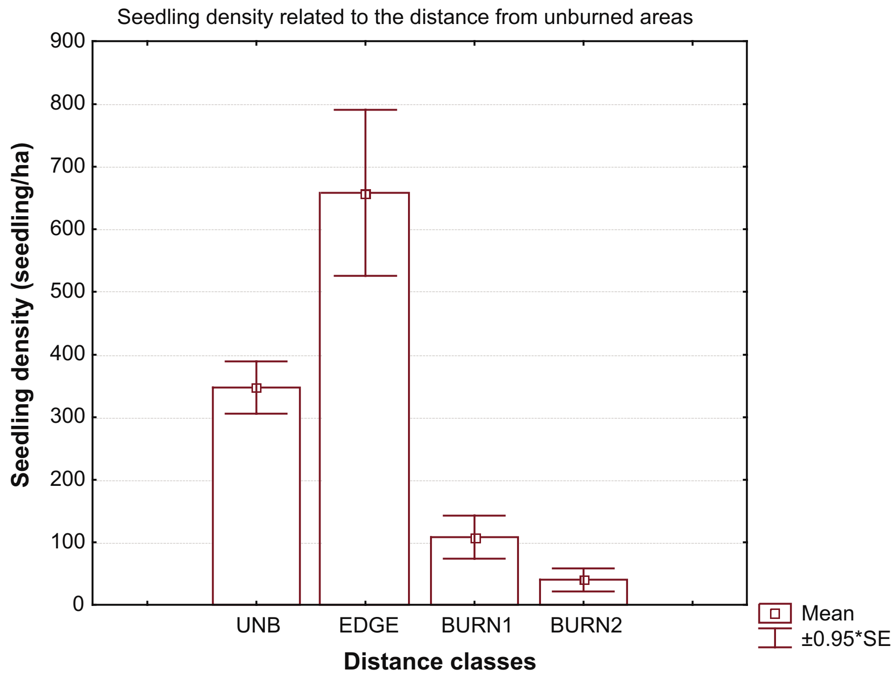

Figure1—Ponderosa pine seedling densities per distance classes: UNB = -36 m to -6 m; EDGE = 0 m to 24 m; BURN1 = 30 m to 60 m; BURN 2 = 66 m to 150 m.

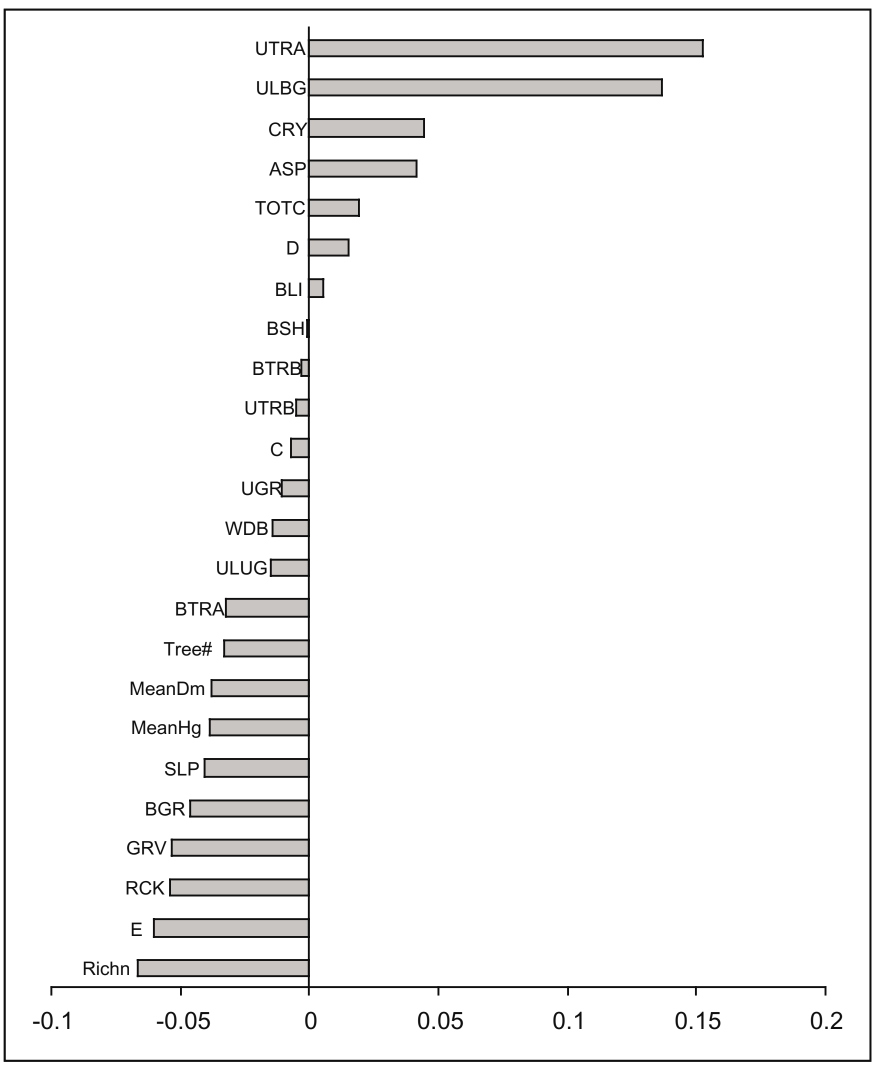

Figure 2—Graphic representation of the regression coefficients of environmental variables on seedling density; Partial Least Squares regression, 1st component. UTRA: Unburned trees <10 m; ULBG: Scorched litter cover; Cry: Cryptogamous cover; ASP: Aspect; TOTC: Total understory cover; D: Small unburned shrubs; BLi: Burned litter; BSH: Burned shrubs; BTRB: Burned trees >10 m; UTRB: Unburned trees >10 m; C: High unburned shrubs; UGR: Unburned ground; WDB: Woody debri cover; ULUG: Unburned litter; BTRA: Burned trees <10 m; Tree#: Tree number; Mean Dm: Mean tree diameter; Mean Hg: Mean tree height; SLP: Slope; BGR: Burned ground; GRV: Gravel cover; RCK: rock cover; E: Herbaceous cover; Richness: Floristic richness.

correlated to the abundance of live trees >10 m high (seed sources) and to the abundance of scorched litter on top of burned ground, and negatively correlated to the high floristic richness and to high covers of vegetation <10 cm high (indicating high competition pressure). Figure 2 shows the regression coefficient of each variable with the seedling density.

Discussion

The results of our study show that seedlings regenerated very well close to the edge between burned and unburned areas. Seedling establishment was explained by the presence of seed sources close by and by the burned soil conditions characteristic of moderately burned areas where scorched needles are still present in the trees after the fire. Seedlings also established better in areas where the competitive pressure from other species was lowest. Burned ground alone didn't appear to be a favorable environment for seedling establishment, whereas scorched litter on top of burned soil was one of the factors explaining seedling distribution. The area along the edge between the burned and unburned areas and extending around 30 m into the burned area constitutes the best condition, because of the short distance of seed dispersal, favorable seedbed conditions, presence of scorched trees, and lower competition pressure. It is possible that the presence of scorched trees also provided shade to the seedlings, which are moderately shade tolerant.

Thanks to the quantification of environmental variables along the transects, we can explain the spatial distribution of the seedlings: (1) from -36 to -6 m, seed sources are abundant but soil conditions are not optimal and competition pressure is high; (2) from 0 to 24 m, the seed sources are close enough to provide seeds, canopy and soil conditions are optimal, and competition pressure is lowered; (3) from 30 to 60 m, foreseeable diminution of seed arrival increased by the unfavorable seedbed conditions; (4) 66 m and farther, the seed dispersal is probably responsible for the decrease in seedling establishment compared to the area included between 30 and 60 m from the edge, the other conditions staying pretty similar between those two areas.

This study allowed us to give a spatial dimension to the regeneration success of ponderosa pine after fire in the Black Hills. According to our results, areas located farther than 30 m from the unburned areas, even if they still receive good quantity of seeds from the seed sources, should constitute an issue for seedling regeneration. This means that we can expect slower natural forest recovery in burned patches larger than 60 m of diameter and that artificial regeneration may be needed in those areas. Seedling establishment in the area included between the edge and 30 m should be very good and no salvage operations are necessary to favor regeneration in such areas or in small burned patches in order to preserve the re-establishment of ponderosa pine in burned areas.

Figures 3 a,b,c show the distribution of this area of good seedling establishment within three different burned patches (small patch, large patch, large patch after salvage). The small patch has a high percentage of good conditions for regeneration. The large patch without salvage has a small percentage of good conditions for regeneration. Using the same definition of good conditions (area located close to the edge, close to seed sources, with abundant scorched litter on top of burned ground and lower floristic richness and cover), the large patch after salvage does not present good conditions for regeneration, because the scorched trees have been cut down and

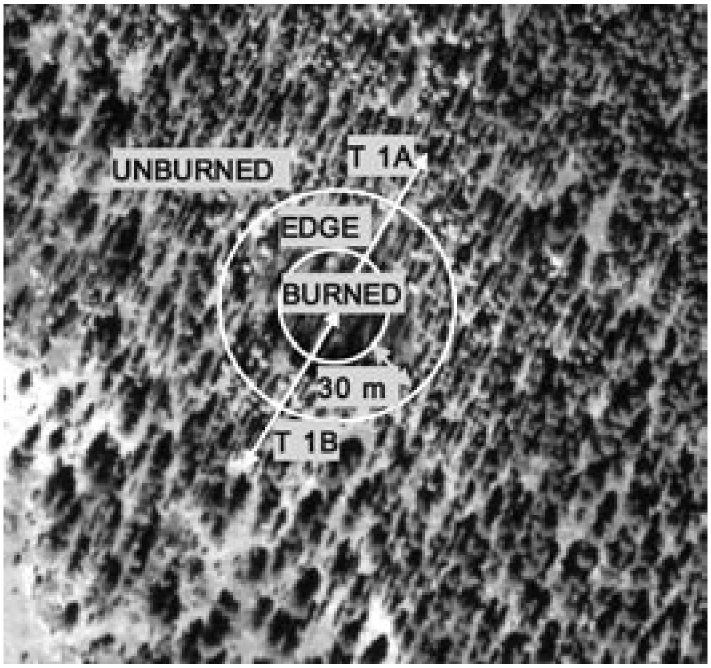

Figure 3a—Small burned patch and representation of burned, unburned, and edge conditions.

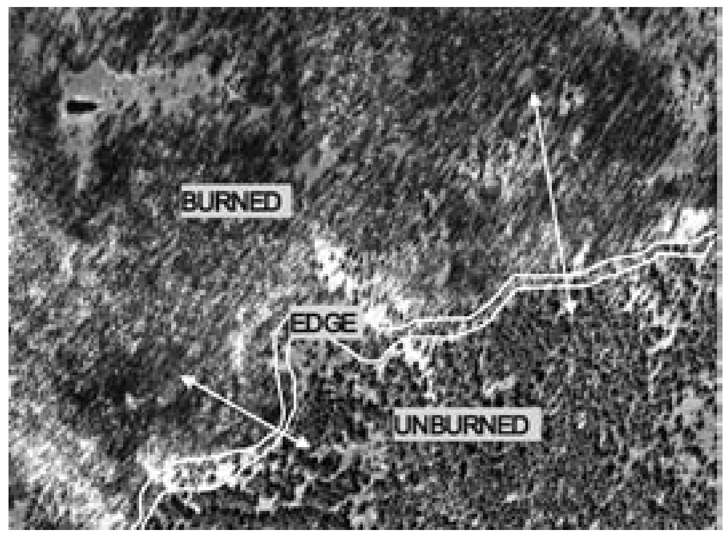

Figure 3b—Large burned patch and representation of burned, unburned, and edge conditions.

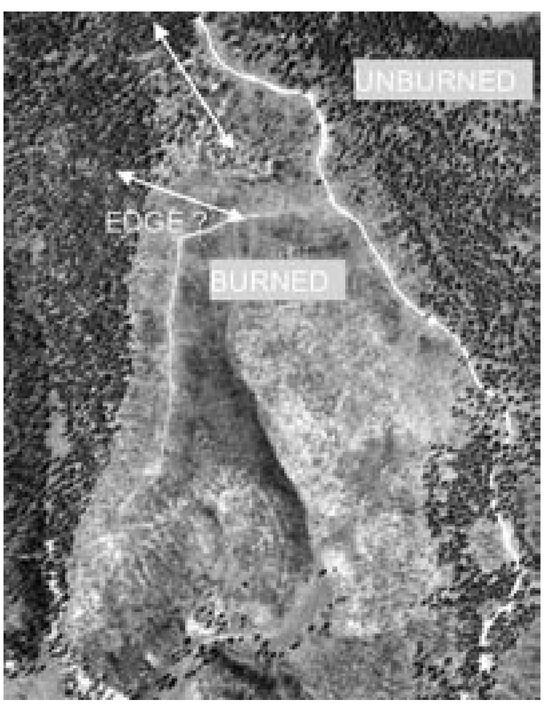

Figure 3c—Large burned patch after salvage and representation of burned, unburned, and edge conditions.

taken away. The preservation of a strip of scorched trees providing good soil conditions and moderate shade for seedlings at the beginning of the burned area would have preserved those good seedling establishment conditions.

Most of the previous post-fire vegetation assessments aimed to look at the influence of one component of the fire regime on plant regeneration. Fire intensity, fire frequency, and fire season have often been the main subjects of interest (Bond and van Wilgen 1996). However, some authors are now interested in showing the effects of spatial characteristics of the disturbance on post-fire regeneration (Turner and others 1997, Bonnet and others 2002). The effect of the size of the burned patch on ecological gradients can be assessed by the use of transects (Whittaker 1960). The originality of our study was to use a transect approach in order to quantify and explain the regeneration as a function of the distance from the edge. This approach used the multiple components of the post-fire environment to explain the spatial patterns of the regeneration. It does not assume *a priori* the role of fire severity but instead decomposes the environmental conditions into basic components and quantifies the role of each of them to explain the regeneration patterns. The approach we used in this paper to determine the conditions of regeneration for ponderosa pine and their spatial distribution within burned areas can provide information for foresters in making decisions related to post-fire logging.

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The Cutting Methods Demonstration Study at Fraser Experimental Forest

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Abstract*—Efficiency of harvest, overstory regrowth, and regeneration establishment are compared among 12 even- and uneven-aged regeneration cuttings installed in 1984 in subalpine Engelmann spruce and lodgepole pine forests on the Fraser Experimental Forest in central Colorado. Individual tree selection and overstory removal prescriptions were much less efficient to log than two- or three-step shelterwood cuts, clearcuts, or group selection cuts. Considerable overstory growth has occurred in all treatments, but most noticeably in the spruce-fir shelterwood and lodgepole pine uneven-aged treatments. Abundant regeneration was present in all treatments, but subalpine fir dominated in partial harvest treatments, illustrating the need for precommercial thinning in subalpine forests.*

Background

The complexities of finding ways to manage subalpine forests to satisfy the variety of challenging resource issues in special places requires a full suite of silviculture tools and techniques. A study was established on the Fraser Experimental Forest in 1983 to provide on-the-ground examples of all even- and uneven-aged cutting methods applicable to spruce-fir (Engelmann spruce [*Picea engelmanii*] and subalpine fir [*Abies lasiocarpa*]) and lodgepole pine (*Pinus contorta*) forests in the central Rocky Mountains. It also served as a case study to compare the relative efficiency of harvesting these silviculture prescriptions under controlled conditions and continues to provide valuable information on subsequent tree survival, growth, and regeneration.

Twelve plots were established in mature spruce-fir and lodgepole pine stands (six plots in each forest type) on the Fraser Experimental Forest in Central Colorado. All plots were located on slopes less than 15 percent that adjoined existing roads, and all but one were one acre in size (because of existing stand structure, the lodgepole pine group selection plot was 4 acres in size). Silvicultural prescriptions based on recommended practice (Alexander 1987, Shepperd and Alexander 1983) were developed for each plot based on existing stand conditions and included both even-aged and uneven-aged management goals. Pre-harvest stand structure varied from plot to plot and was strongly influenced by settlement-era logging that occurred in the early 20th century. Overstory trees left after the initial harvests were generally from 12 to 20+ inches DBH and from 120-300+ years of age prior to this study. Some plots were relatively undisturbed by the earlier harvests and others contained understories of smaller trees that had been initiated by the earlier harvests.

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Treatments

Even-aged treatments were clearcutting (CC), the first entry of a two- and three-step shelterwood (SW2 and SW3), and overstory removal (OR). The latter treatment consisted of the removal of a mature overstory growing over a fully stocked sapling and pole-sized understory. Uneven-aged harvests were individual tree and group selection. Each treatment was duplicated in spruce-fir and lodgepole pine forest types. The BDQ method (Alexander and Edminster 1977) was used to regulate stocking in the uneven-aged prescriptions. A Q factor (the ratio of stem/acre in one DBH class to that of the next smaller class) of 1.3 (using 1-inch DBH classes) was used in spruce-fir with DBH classes ranging from 4 to 24 inches, the maximum retained diameter. Residual basal area targets after cutting were 80 ft^2/acre in the uneven-aged treatments. The same Q factor BDQ prescription was chosen for the uneven-aged lodgepole pine treatments, except the upper DBH limit was 18 inches. These uneven-aged configurations were selected because they (1) closely matched the distribution of growing stock on the sites, (2) removed sufficient growing stock to expect new regeneration, and (3) allowed the residual forest to remain at low risk to windthrow.

All areas were logged during one season by the same contract logger using chainsaw felling and bucking and a rubber tired cable skidder to remove and deck logs. The logger was free to log the plots as he saw fit but had to remove all marked stems, use designated skid trails, and not damage residual stems. Researchers observed all logging and kept detailed notes on the amount of time spent in felling, skidding, decking, and cleanup of each harvest unit. Every log was scaled and the number of pieces removed in each skidder turn was recorded.

Logging efficiencies for each one-acre harvest treatment unit were estimated by calculating the cubic feet of wood removed divided by the total hours of effort required by all members of the logging crew to complete the unit, resulting in a person-hour/ft^3 production rate. Data for the four-acre lodgepole pine group selection unit were adjusted to a per-acre basis.

Results

Post-Harvest

Specific pre- and post-treatment stocking data are listed in table 1. The seed cuts removed 60 percent and 65 percent of the pretreatment growing stock from the spruce-fir and lodgepole two-step shelterwood plots, respectively. The preparatory three-step shelterwood cuts removed 40 percent and 30 percent of the stocking from the spruce-fir and lodgepole pine plots, respectively. All of the growing stock above 3.5 inches DBH was removed from the overstory removal and clearcut treatments. The individual tree and group selection treatments removed 28 to 55 percent of the pretreatment growing stock distributed as closely as possible to the target diameter class stocking curves. All visible pre-existing seedlings and saplings were removed from shelterwood and clearcut treatments to facilitate observation of the regeneration response to these treatments. Some windthrow occurred in both the spruce and lodgepole shelterwood treatments, as might be expected (Alexander 1987, Shepperd and Alexander 1983).

Table 1—Conditions existing pre- and post-treatment and in 2003 by forest type and harvest treatment: overstory basal area per acre (BA); trees per acre (TPA); and quadratic average stand diameter (DQ). Treatment codes: SW3 = three-step shelterwood. SW2 = two-step shelterwood. ITS = individual tree selection. CC = clearcut. OR = overstory removal. GS = group selection.

Engelmann Spruce – Subalpine Fir

Treat-ment	Pre BA	Pre TPA	Pre DQ	BA cut	% BA cut	TPA cut	Post BA	Post TPA	Post DQ	2003 BA	2003 TPA	2003 DQ	BA change
SW3	205	229	12.8	82	40	85	102	78	15.4	77.1	46	17.5	► -24.1
SW2	220	160	15.8	132	60	185	88	65	11.8	97.6	57	17.7	► 9.6
ITS	176	285	10.6	97	55	229	* 79	199	8.5	136.2	347	8.5	57.2
CC	136	200	11.2	136	100	241	0	0	0	0	0	0	0
OR	160	341	9.3	160	100	426	+ 0	0	0	40.4	268	5.3	40.4
GS	165	223	11.6	69	31	53	95.7	155	10.6	124.6	199	7.7	28.9

Lodgepole Pine

Treat-ment	Pre BA	Pre TPA	Pre DQ	BA cut	% BA cut	TPA cut	Post BA	Post TPA	Post DQ	2003 BA	2003 TPA	2003 DQ	BA change
SW3	156	232	11.1	47	30	151	110.6	147	11.7	94.6	112	12.4	► -16.0
SW2	220	285	11.9	143	65	95	75.3	100	11.8	68.8	74	13.0	► -1.3
ITS	145	481	7.4	65	45	86	79.0	252	7.6	136.2	432	7.2	57.2
CC	180	241	11.7	180	100	200	0	0	0	0	0	0	0
OR	98	426	7.1	97	100	341	0	0	0	4.4	32	5.0	4.4
GS	100	334	7.4	28	28	68	67.5	281	6.7	108.4	364	7.3	40.9

* Marked to 80 BA, 5 stems lost due to logging.
+ Understory not included.
► Reduced due to windthrow losses.

Efficiency of Harvest

Logging activities are presented on a person-hour and cubic volume basis in table 2. Harvest efficiency expressed in terms of person-hours/ft^3 is compared by prescription and species in figure 1. Production rates did not vary substantially, except that individual tree selection and overstory removal prescriptions were much less efficient to log, due to the necessity of avoiding existing seedlings and saplings in these treatments. Pieces of all sizes were removed from all plots (table 2), reflecting the irregular structure of the pre-existing forest. However, the larger piece sizes in group selection treatments was a consequence of harvesting predominately mature groups of trees in those stands.

Conditions in 2003

Nineteen years after the 1984 harvests, all of the 12 treatment plots are fully stocked and remain in healthy condition. Considerable overstory basal area growth has occurred in group and individual-tree selection treatments, but windthrow losses have affected growth in shelterwood treatments (table 1). However, Scribner volumes (board feet/acre) increased most noticeably in the spruce-fir shelterwood (figure 2) and in lodgepole pine uneven-aged treatments (figure 3). Scribner volumes in the latter treatments have nearly grown back to pre-harvest levels (figure 3). Current diameter distributions of all uneven-aged treatments (figures 4, 5) show stocking surpluses are greatest in the smaller diameter classes. Sufficient surpluses exist in larger diameter classes in both spruce-fir and lodgepole individual tree selection treatments (figure 5) to currently support another commercial entry cycle (figure 3).

All treatment plots are currently stocked with regeneration well in excess of regional guidelines (USDA Forest Service, Rocky Mountain Region

Table 2—Comparison of harvest activities by silviculture treatment.

Engelmann Spruce

Treat-ment	Felling and bucking Hours	ft³/hour	Skidding Hours	No. turns	Logs/turn	ft³/piece	ft³/hour	Cleanup hours	Other *	Total hours	ft³/hour	Total ft³
SW3	51.2	178.9	34.0	86.1	4.3	25.0	269.8	24.5	0.0	109.7	83.5	9161.0
SW2	37.1	148.2	21.6	56.1	4.4	22.2	254.5	12.8	0.5	72.0	76.4	5498.0
ITS	45.4	91.5	84.7	52.1	4.0	19.9	49.1	27.0	0.0	157.1	26.4	4154.0
CC	22.3	311.4	14.6	56.1	4.4	28.4	474.9	37.5	0.0	74.4	93.3	6939.0
OR	79.7	57.3	59.1	91.1	3.9	12.7	77.3	9.5	0.0	148.3	30.8	4570.0
GS	21.2	290.3	20.2	44.0	3.8	37.0	304.4	22.8	0.8	64.9	94.6	6142.0
							Engelmann spruce averages				58.2	36464.0

Lodgepole Pine

Treat-ment	Felling and bucking Hours	ft³/hour	Skidding Hours	No. turns	Logs/turn	ft³/piece	ft³/hour	Cleanup hours	Other *	Total hours	ft³/hour	Total ft³
SW2	25.5	198.3	32.7	74.0	5.0	13.7	154.5	4.0	0.6	62.8	80.5	5056.0
SW3	19.8	235.1	24.0	56.0	4.8	17.2	194.2	5.0	0.9	49.7	93.7	4657.0
ITS	25.3	96.6	13.8	31.0	4.6	17.4	177.0	17.0	0.0	56.2	43.6	2448.0
CC	36.0	217.1	28.7	73.1	4.8	22.3	272.3	6.8	0.0	70.7	110.6	7817.0
OR	32.2	68.3	58.5	32.0	5.1	13.5	37.6	6.0	0.0	96.7	22.7	2197.0
GS	24.8	238.3	28.6	49.1	4.9	24.4	206.0	12.0	1.1	66.5	88.7	5897.0
							Lodgepole pine averages				69.7	28072.0

* Includes repair, skid trail construction, etc.

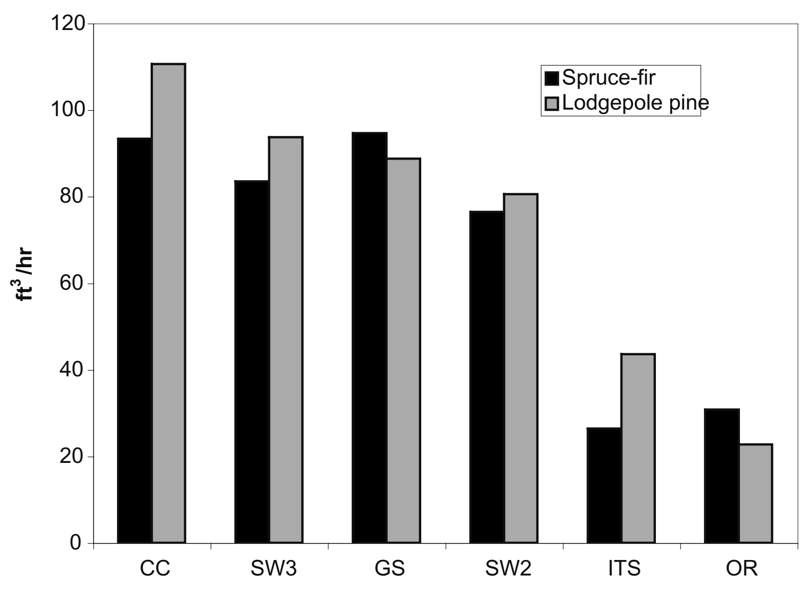


Figure 1—Harvest production rates in cubic feet per person-hour by forest type for clearcut (CC), three-step shelterwood (SW3), group selection (GS), two-step shelterwood (SW2), individual tree selection (ITS), and overstory removal (OR) cutting methods.

Reforestation Handbook) for spruce-fir and lodgepole pine forests (table 3). The abundance of small seedlings indicates that new trees are still being recruited into these areas. The predominance of subalpine fir in all spruce-fir treatments is also noteworthy. Fir will obviously dominate the future stocking in all of the spruce-fir treatment plots, but especially so in the individual tree selection and three-step shelterwood treatments.

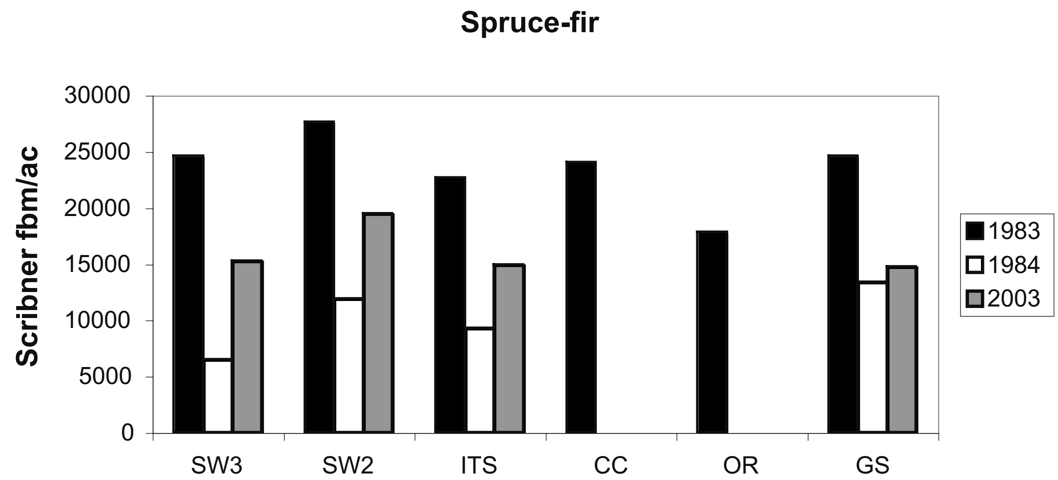


Figure 2—Pre-harvest, post-harvest, and 2003 board foot Scribner volumes in spruce-fir units by silviculture treatment.

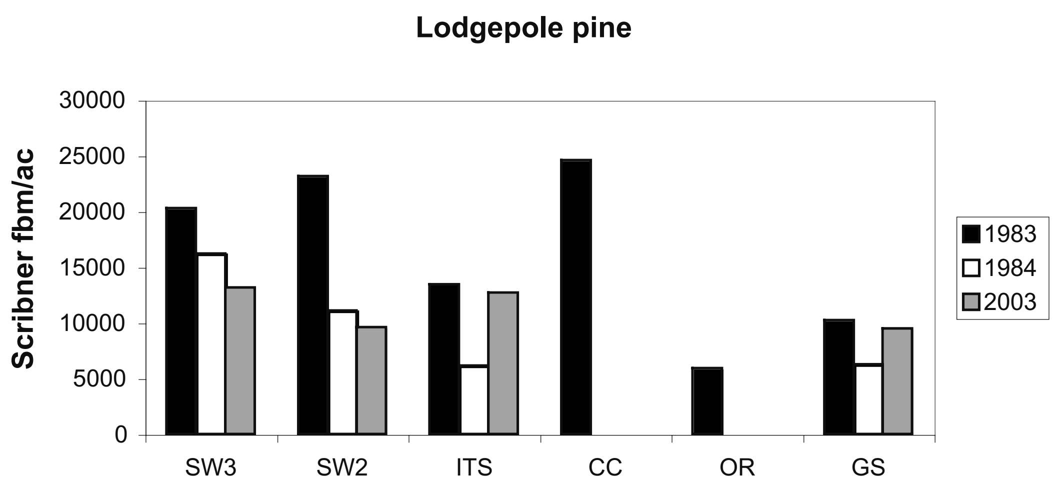


Figure 3—Pre-harvest, post-harvest, and 2003 board foot Scribner volumes in lodgepole pine units by silviculture treatment.

Discussion

These demonstration areas on the Fraser Experimental Forest provide an opportunity to view and compare a variety of management options in one readily accessible location. All of these treatments have been successful in that they resulted in the establishment of new regeneration, successfully prepared the forest for subsequent entries without excessive windthrow, or have demonstrated the applicability of both even-aged and especially uneven-aged regeneration methods in both spruce-fir and lodgepole pine forest types.

The production data collected during the 1984 harvests clearly show that conventional handfelling and tractor skidding was possible in all treatments. Although the overstory removals in the overstory removal treatments were least efficient, performing these operations while regeneration was still in the seedling stage would lessen damage from equipment and tree falling and substantially increase harvest efficiency. Using new mechanical harvesting equipment available since this study was harvested would probably increase

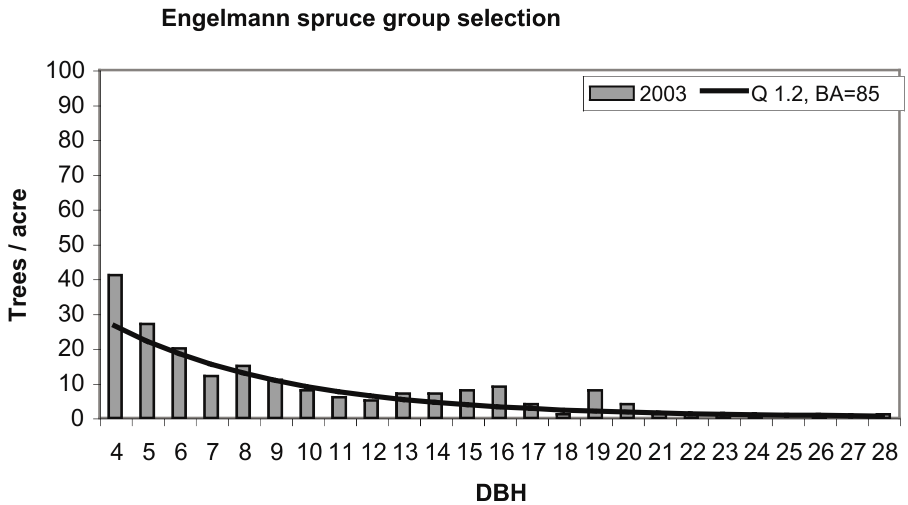
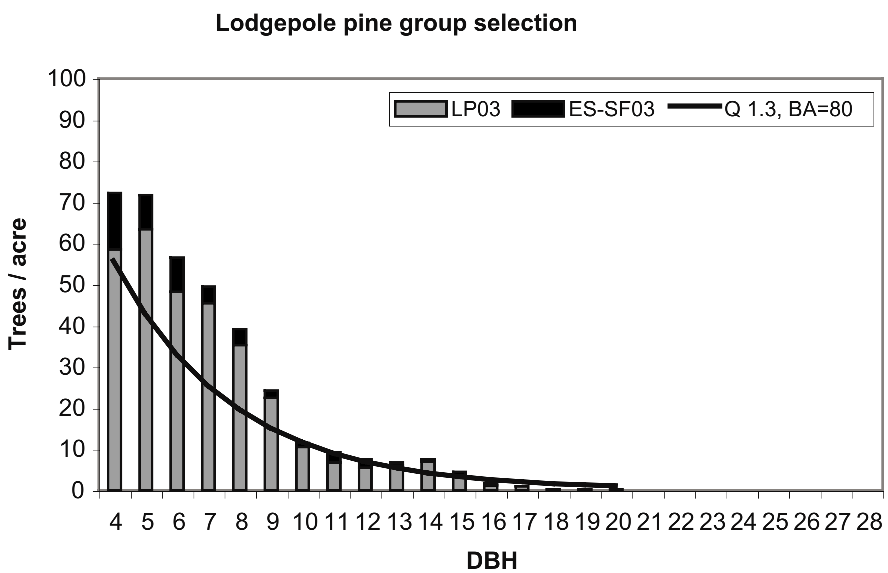


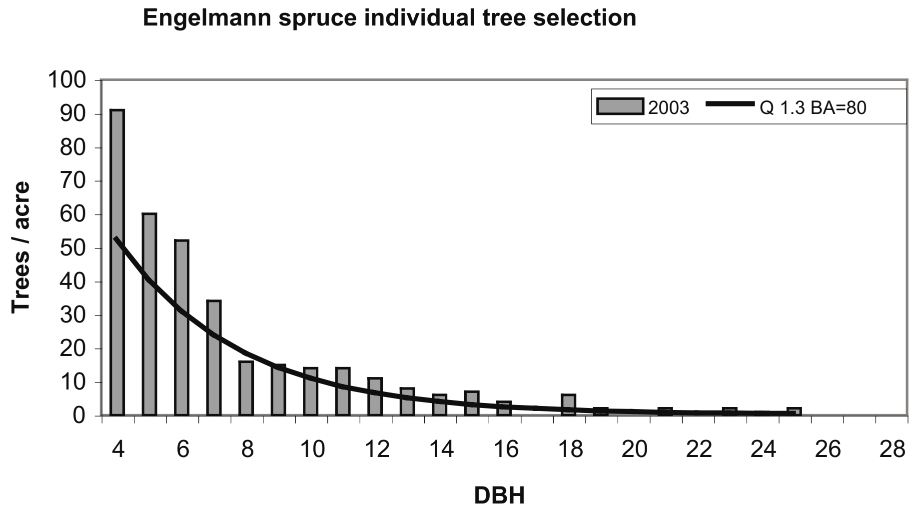
Figure 4—Comparison of 2003 stocking by diameter class with target stocking guidelines for spruce-fir and lodgepole pine group selection treatments. The LP03 and ES-SF03 labels indicate 2003 stocking of lodgepole pine, spruce, and fir in the lodgepole group selection treatment.



the harvest efficiency in all treatments, but the ranking of treatments would likely not change, since it was closely related to the number of pieces that needed to be moved.

Both the appearance and the current stocking of these plots indicate that additional silviculture activities are needed in all of these treatments. Sufficient growth and regeneration have occurred in the two-step shelterwood plots that overstory removals could now be done. Adequate wind firmness and advanced regeneration exist in the three-step shelterwood treatments so that seed cuts could now be completed. Stocking has increased in the uneven-aged treatments to the point that growing stock should once again be reduced to desired levels by another cutting cycle harvest.

A more critical concern with regard to the future composition of these forests would be the need to precommercially thin excessive subalpine fir stocking from both the spruce-fir and lodgepole pine plots. Subalpine fir comprises the majority of seedling and sapling stocking in the spruce-fir individual tree selection, group selection, overstory removal, and three-step shelterwood treatments. Subalpine fir also accounts for over 30 percent of the regeneration in the lodgepole pine individual tree selection and over-



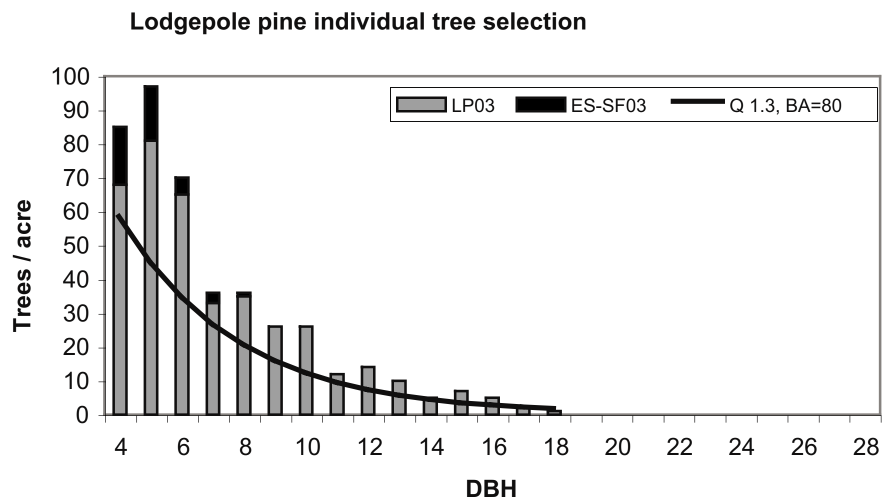


Figure 5—Comparison of 2003 stocking by diameter class with target stocking guidelines for spruce-fir and lodgepole pine individual tree selection treatments. The LP03 and ES-SF03 labels indicate 2003 stocking of lodgepole pine, spruce, and fir in the lodgepole individual-tree selection treatment.

story removal treatments. Subalpine fir is much more shade tolerant than either spruce or lodgepole pine and can persist very well under these other species. As the fir trees grow larger, the dense shade cast by their full crowns will strongly inhibit the ability of lodgepole pine and spruce to successfully establish. The result will be a type-shift to fir dominated forests rather than the spruce and lodgepole forests that we intended to regenerate.

In many ways these demonstration plots represent conditions and trends that are present in many special places throughout the central and southern Rocky Mountains. This study clearly illustrates that forest conditions are not static but dynamically change over time in response to disturbance. If that disturbance is a program of silviculture treatments intended to improve the condition of the forest, we are obligated to continue management to either ultimately replace the forest or maintain it in a desired condition. The natural advantage that shade-tolerant subalpine fir has in establishing under overstory canopies ensures that management goals for any partial cutting silviculture treatment in subalpine forest types cannot be accomplished unless precommercial thinning is an integral part of the silvicultural system.

Table 3—Number of seedlings and saplings (<3.5 inches DBH) per acre in 2003 by harvest treatment, species, and size class.

Engelmann Spruce

	Tree species				**Size class**				
Treatment	Engelmann spruce	Lodgepole pine	Subalpine fir	Total	Less than 1.5 ft.	1.5 to 4.5 ft ht.	4.5 ft. to 1 in. DBH	1 to 3.5 in. DBH	Percent stocking
OR	780	180	2010	2970	750	990	750	480	100
CC	720	240	480	1440	540	240	360	300	80
SW2	1230	150	720	2100	990	540	450	120	100
SW3	330	0	930	1260	480	360	300	120	80
ITS	2053	0	3130	5183	1440	2340	900	503	100
GS[1]	429	0	2210	2639	510	900	720	509	90
Average	924	95	1580	2599	785	895	580	339	92

Lodgepole Pine

	Tree species				**Size class**				
Treatment	Engelmann spruce	Lodgepole pine	Subalpine fir	Total	Less than 1.5 ft.	1.5 to 4.5 ft ht.	4.5 ft. to 1 in. DBH	1 to 3.5 in. DBH	Percent stocking
OR	80	550	550	1180	250	150	280	500	100
CC	30	2520	120	2670	180	420	780	1290	80
SW2	150	3300	630	4080	2370	900	570	240	100
SW3	150	3660	840	4650	3090	1050	390	120	90
ITS	307	1538	1061	2906	1650	570	240	446	100
GS[1]	334	2733	429	3496	1350	870	510	766	95
Average	175	2384	605	3164	1482	660	462	560	94

[1] Group selection (GS) treatments include regeneration in group openings and intervening forest.

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Silviculture in Cooperation With Hunters: The Kinzua Quality Deer Cooperative

Scott Reitz[1], Andrea Hille[1], and Susan Stout[2]

Abstract*—The long history of deer overabundance in Pennsylvania is associated with very high reforestation costs and substantial threats to diversity and sustainability. In response to this legacy, several landowners and agency personnel formed the Kinzua Quality Deer Cooperative (KQDC) in partnership with the Sand County Foundation. This Cooperative focuses on about 74,000 acres in the northeast corner of the Allegheny National Forest (ANF), the setting for the Sugar Run Project under planning by the ANF at present. The goals of the KQDC are to develop a quality deer herd in quality habitat through cooperation with local sportsmen and sportswomen. In this paper, we discuss the actions proposed in the Sugar Run project to use improved hunter access and hunter success as silvicultural tools, given a definition of silviculture as "controlling the establishment, growth, competition, health, and quality of forests." These include the scheduling of regeneration activities to provide a stable level of forage production, increases in road quality, layout and development of skid trails as hunter access trails, creation of viewing pull-outs to stimulate hunter interest, and development of a demonstration of the use of silviculture and the interaction of deer and silviculture in shaping habitat.*

Introduction

In his 1996 textbook, Ralph Nyland (1996) defines silviculture as "establishing and maintaining communities of trees and other vegetation that have value to people." The Society of American Foresters (1998) provides a similar definition, used by Russ Graham in this proceedings, saying that silviculture is "the art and science of controlling the establishment, growth, competition, health, and quality of forests and woodlands" for an ever-widening array of management objectives. Thus, practicing silviculture grows more diverse and complex with every new understanding that we develop of the growth and establishment of forests. In this paper we tell the story of increasing cooperation with hunters as a silvicultural tool in order to achieve objectives of forest regeneration and renewal—establishing and growing diverse communities of trees and other vegetation—in one corner of the Allegheny National Forest (ANF) where white-tailed deer (*Odocoileus virginianus*) have been overabundant for more than 70 years. The area in question is about 74,000 acres in the northeast corner of the National Forest, a landscape owned by a municipal watershed, a timber investment management organization, two different timber companies, and the American people. These landowners and managers have been cooperating for decades to change deer management in Pennsylvania and are now cooperating to engage hunters to achieve healthy deer in a healthy habitat. The name of both the shared landscape and the cooperative efforts to

[1] USDA Forest Service, Allegheny National Forest, Bradford Ranger District, Bradford, PA.

[2] USDA Forest Service, Northeastern Forest Experiment Station Laboratory, Irvine, PA.

restore both habitat and herd quality is the Kinzua Quality Deer Cooperative (KQDC) (figure 1). Within the area, managers on the ANF have been cooperating with the public in planning a project for the Sugar Run Analysis Area, the only management project likely to occur on National Forest land within the KQDC during the current decade.

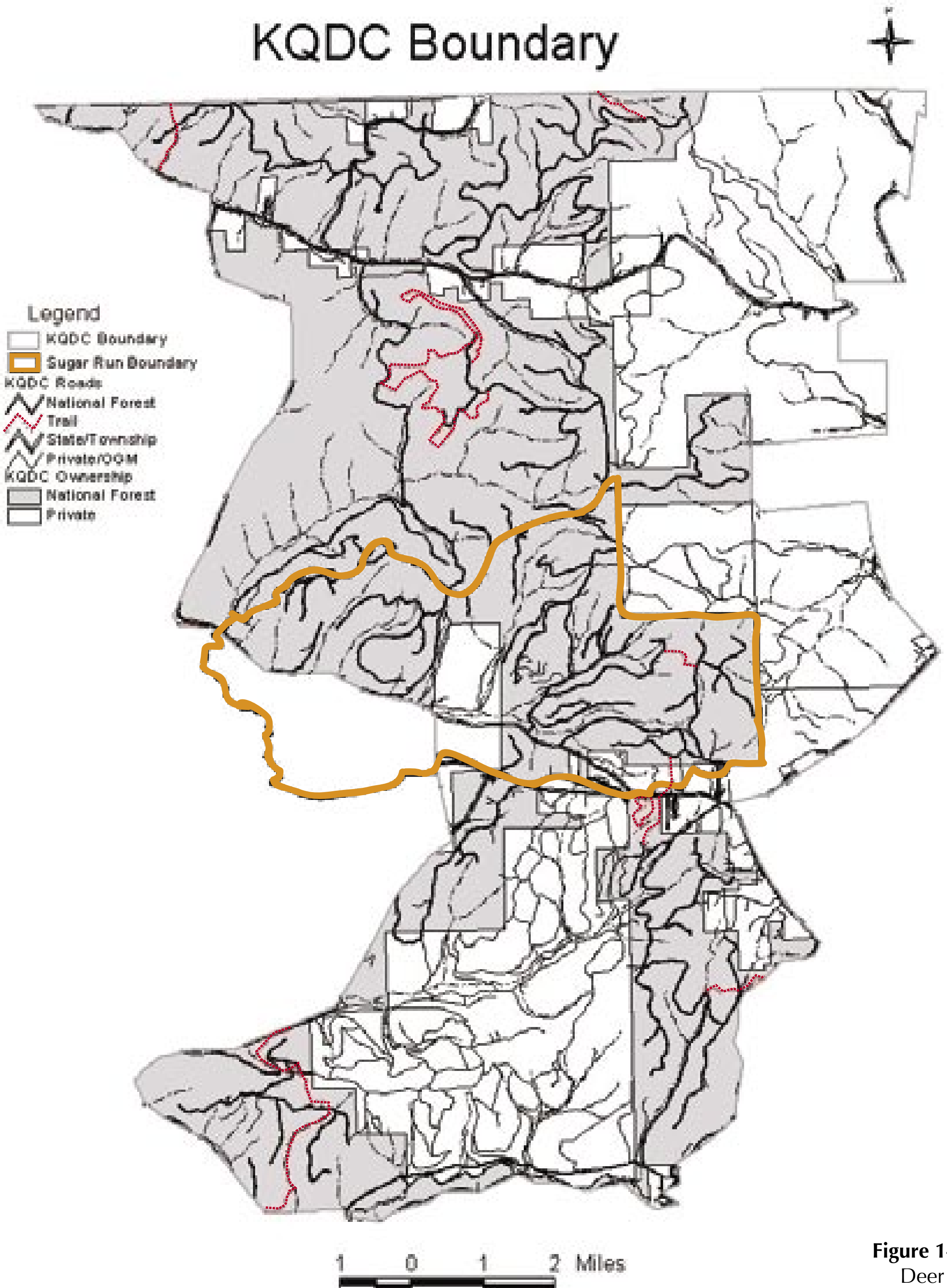

Figure 1—A map of the Kinzua Quality Deer Cooperative. The Sugar Run Project Area is outlined near the narrowest portion of the KQDC.

The activities described in this paper are designed to change forest regeneration and renewal at the landscape level. The intent is to alter responses to more familiar stand-level silvicultural activities by changing the context within which they occur. Those who believe that silviculture occurs strictly at the single stand level may question whether activities like cooperating with hunters to reduce deer impact are silviculture. Surely if deer impact is reduced in one stand, it is also reduced in adjacent stands. Those people who believe that all silviculture occurs strictly at the stand level suggest that these activities should more accurately be described as "forest management" rather than silviculture. Similarly, those who believe that silvicultural activities lend themselves to precise quantification are likely to be disappointed at the imprecision with which hunters change silvicultural outcomes. Our contention is that the intent of these activities is to change stand-level responses related to the "establishment, growth, competition, health, and quality of forests and woodlands," and that they are, therefore, "silviculture." We hope that this paper will advance the discussion of whether and which activities that occur at scales larger than the stand are still appropriately characterized as silviculture.

History of Deer Impacts on the Allegheny Plateau

The Northern Unglaciated Allegheny Plateau Section (Keys and others 1995) of Pennsylvania has long been notorious for the heavy deer impacts borne by its plant and wildlife communities (Hough 1965; Leopold 1943; Redding 1995; Marquis 1981; Marquis and Brenneman 1981; Tilghman 1989; deCalesta 1994; Rooney 1997; Horsley and others 2003). Estimates of deer densities during the period when the region was occupied only by Native Americans range from 8-15 deer per square mile (McCabe and McCabe 1997). As European settlers moved into the region, hunting pressure, including hunting for urban markets, increased substantially. After near extirpation of the Pennsylvania herd in the late 19th Century, the Pennsylvania Game Commission (PGC) was established in 1895 in large part to protect what was perceived as a precious and scarce resource--the white-tailed deer. With strict regulation of hunting seasons, early prohibitions against harvesting does, and small-scale reintroductions from Michigan and Virginia coinciding with the creation, statewide, of nearly perfect deer habitat through heavy forest harvesting[3], herd size sky-rocketed. By the early 1920s, farmers sought relief from overabundant deer in some parts of the state, and against protests by hunters, doe seasons were launched in selected agricultural counties. By the late 1920s, foresters, too, were noticing the negative consequences of local overabundance, including on the territory of the newly created ANF, and the first statewide doe harvest was scheduled in 1928. The idea was met with stiff opposition from hunting clubs, local newspapers, and politically active hunters, but went forward (Kosack 1995). By 1943, after a visit to Pennsylvania, Aldo Leopold (1943) warned of an impending crisis in Pennsylvania deer management. Through the intervening years, there have been periodic reductions in average deer density. These usually occurred when a PGC policy change to reduce herd density coincided with two bad winters in a row, as in the early 1940s and the late 1970s (figure 2). Often these reductions have resulted in politically effective backlash from hunters and sportsmen, and initiatives to control

[3] Marquis (1975) describes the harvests that occurred at this time: "Between 1890 and 1920, the virgin and partially cut forests were almost completely clearcut in what must have been the highest degree of forest utilization that the world has ever seen in any commercial lumbering area."

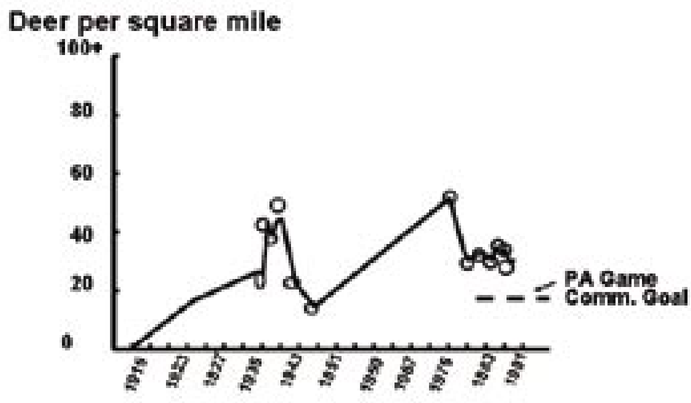

Figure 2—Deer densities through the 20th century in the four-county area of the Allegheny National Forest (from Redding 1995). Circles represent actual data points from studies conducted on the ANF. Bad winters occurred in sequence in the early 1940s and in the late 1970s. The PGC initiated habitat-based population goals in 1979.

overabundance have failed. In general, until recent years, spokespeople for hunting organizations have sought to have deer managed for maximum huntable numbers, while foresters and farmers have lobbied for reductions in herd numbers. Both groups turned primarily to the semi-independent PGC as the arbiter of this profound and long-standing dispute. Data collected over the years in the Northern Unglaciated Allegheny Plateau Section suggest that the deer herd has not been at levels now established by the Game Commission as compatible with multiple-use management (18-21 deer per square mile) in this region for most of the last 60 years (Redding 1995).

Ecological Consequences of Deer Abundance

The ecological consequences of this overabundance are many. Detailed research concerning the impact of white-tailed deer on forest resources has also been a hallmark of this region. Early researchers noticed the loss of shrubs, especially the once-common hobblebush (*Viburnum alnifolium*) (Hough 1965, Kosack 1995) from both old-growth and second-growth forests. Marquis (1981) studied the many factors that would influence the outcome of regeneration harvest treatments on the ANF. Deer browsing explained 87 percent of the regeneration failures that occurred in the study, and the presence of abundant advance regeneration, even very small seedlings, was the single factor that best predicted which areas were likely to succeed (Marquis 1981). Later studies showed that the dynamics of vegetation development differed sharply at different deer densities. Species diversity, height growth, and stocking of trees and raspberry bushes decreased as deer density increased; stocking with ferns and grasses increased. So did the dominance of a single tree species relatively less preferred by deer: black cherry (*Prunus serotina*) (Tilghman 1989, Horsley and others 2003). Forest structure varied with deer density as well; in thinned stands in the lowest deer density enclosures, a midstory formed that housed a community of birds absent at higher deer densities (deCalesta 1994). Finally, the study also showed that the *impact* of deer on vegetation in managed forests was a joint function of their density and the forage available in the landscape surrounding a management area (figure 3) (Marquis and others 1992, deCalesta and Stout 1997). One implication of this study is that there is no universally "right" number of deer.

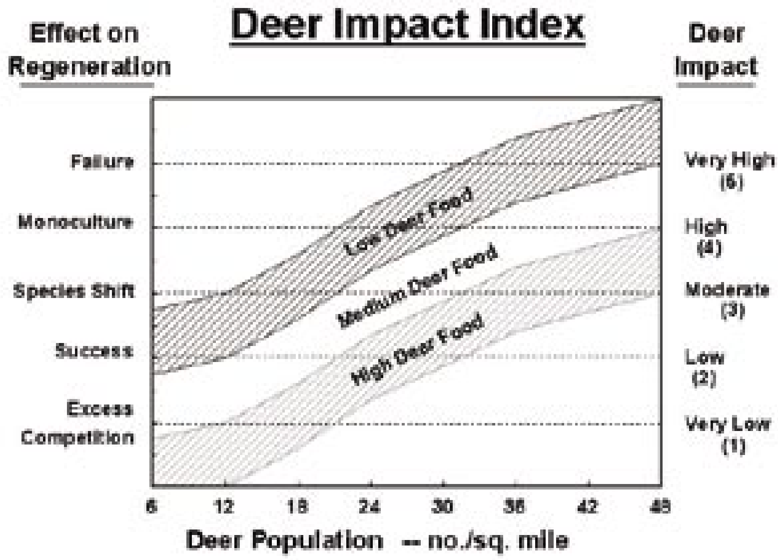

Figure 3—Conceptual framework showing that the *impact* of deer on the outcome of silvicultural regeneration harvests is a joint function of the density of deer and the amount of forage found on the surrounding landscape (from Marquis and others 1992).

The cumulative effects of such pressures on vegetation dynamics, sustained over 60 years, can be seen everywhere in the landscape. When a 1985 tornado blew over 800 acres of remnant old growth in the Tionesta Scenic and Research Natural Areas, advance regeneration was dominated by the browse-resilient American beech (*Fagus grandifolia)* and striped maple (*Acer pennsylvanicum*). The moderately preferred birches (*Betula lenta* and *B. alleghaniensis*) blew in, established on the exposed mineral soil, and became the most numerous seedlings; eastern hemlock (*Tsuga canadensis*), a preferred winter food, benefited from the exposed mineral soil, but only those seedlings that became established near the top of tip-up mounds, where deer could not reach them, persisted (Peterson and Pickett 1995, Long and others 1998). A 1991 study of the most intensive timber management zone on the ANF showed that 46 percent of that area had interfering levels of fern in the understory. The same survey showed that black cherry seedlings were the most common tree seedling, representing 47 percent of all seedlings measured in the survey (Allegheny National Forest 1995), even though black cherry represented only 28 percent of the overstory. The statewide 1989 Forest Inventory and Analysis survey of Pennsylvania forests found that more than 30 percent of analyzed plots statewide had fern cover at or above the level that interferes with the establishment and growth of seedlings, while only 4.2 percent of the analyzed samples had sufficient tree seedlings to ensure reforestation after a disturbance at high deer density (McWilliams and others 1995). The first vegetation survey in the KQDC area, conducted during the summer of 2001, revealed that 43 percent of the sampled plots had interfering levels of fern, 71 percent had beech or striped maple taller than other understory plants, and 88 percent of the understory sample plots had interfering levels of beech, striped maple, fern, or both (unpublished data on file at the Forestry Sciences Laboratory, Irvine, PA).

History of Deer Hunting in the KQDC Area and Beyond

For the last 40 years, Pennsylvania deer hunting seasons have included a two-week antlered deer season, followed by a three-day antlerless season.

Up through the mid 1980s, many hunters utilizing lands that are now included in the KQDC came from areas where the deer population was low, particularly western Ohio and southwest and central Pennsylvania. Because of the distance many hunters traveled, they often hunted for long periods of time and stayed for a week or more in local hunting camps, as well as large tent camps on the National Forest. Many hunters stayed for two weeks and hunted through the antlerless season. In the last 10 to 15 years, deer numbers have increased in other parts of the state and the east, and many hunters who formerly hunted on the KQDC now hunt in other areas, closer to home. Cultural changes—dual career couples, loss of jobs with extended vacation benefits—have also had an impact. Many out of area hunters only stay for the first two days of deer season, and based on roadside vehicle counts (unpublished data on file at the Bradford Ranger District, ANF), hunter use has declined by 50 percent in the KQDC area since 1993.

Although the overall trend in deer abundance in many parts of the state throughout the 20th century was up, there were localized reductions in deer abundance on the Allegheny Plateau after 1980. Antlerless permits were periodically increased, and where hunter access was good, this led to reductions. The continued use of a three-day antlerless season coming after the two-week buck season limited the effectiveness of the increased antlerless permits. Where access was poor, however, decreased hunter use was observed. This situation was confounded on the Allegheny National Forest by reductions in timber harvesting as a result of appeals and litigation, with an associated reduction in forage supply.

In an effort to increase the doe harvest, encourage hunter use, balance the herd's sex ratio and increase overall hunter success, the PGC has been issuing large numbers of antlerless permits, added early season rifle hunts for junior and senior hunters, and, starting in 2001, changed from the separate antlered and antlerless seasons to a 2 week concurrent rifle hunt, during which either sex can be harvested.

During the same period, many forest landowners in northwestern Pennsylvania became aware of the work of the Sand County Foundation (SCF), a Madison, Wisconsin, foundation dedicated to promotion of Aldo Leopold's land ethic and focused on issues on which Leopold had worked during his career (http://www.sandcounty.org/). One SCF program is Quality Hunting Ecology. The philosophy underlying Quality Hunting Ecology is that management of deer herds and deer habitats must be coordinated to ensure the long-term health of both. The landowners who initiated the KQDC effort, in partnership with SCF, recognized that collaborative efforts to interest, engage, and increase the effectiveness of hunters through a Quality Hunting Ecology program in the KQDC area could be a valuable tool for restoring these forests.

The average deer density for the KQDC is 28.3 deer per square mile. The deer impact study described above (Tilghman 1989; deCalesta 1994; Horsley and others 2003) suggested that in managed forests like KQDC, a deer density of about 18 deer per square mile would be compatible with management objectives to sustain diverse mixed forests. For the fall 2003 hunting seasons, the PGC launched a Deer Management Assistance Program for landowners and public lands whose property is open to public hunting. Through this program, landowners can make additional antlerless licenses available to hunters for use on properties with an approved Deer Management Plan, and the KQDC Leadership Team requested, received, and has distributed coupons for 5,000 additional antlerless licenses. With decreasing numbers of hunters, legacies of overabundant deer such as high fern cover across the landscape, and deer numbers about 50 percent above those compatible with diverse regeneration of Allegheny Plateau forests, the efforts of the KQDC Leadership Team and the silvicultural efforts of forest managers across the area are an important complement to additional licenses in the effort to promote healthy deer and healthy habitat.

The Landscape of the Kinzua Quality Deer Cooperative

The 74,350 acres of the KQDC are special not because they are different from the surrounding landscape, but because they are very similar. Lessons learned about hunter involvement, about silvicultural strategies that increase hunter access and success, and about forest restoration, can be applied to the larger landscape.

The landowners and managers of the KQDC share a commitment to sustainable management of this forest, although the management emphases vary. The National Forest lands within the KQDC fall primarily into two management zones. One emphasizes production of high-value sawtimber, management for compatible wildlife species including deer, and dispersed, roaded recreational opportunities like hunting and scenic driving. The other zone features management for species that prefer primarily high forest cover, like turkey and bear, and mature forest conditions. The KQDC and surrounding area provides habitat for threatened, endangered, and sensitive species including the Indiana bat and bald eagle. An unroaded 10,000 acre Congressionally designated National Recreation Area is adjacent to the northern half of the KQDC. Three of the private landowners have a major focus on production of high-value sawtimber products, and two of these have achieved third party verified certification, through the Forest Stewardship Council and the Sustainable Forestry Initiative. The municipal watershed is managed by a consultant that is also third-party certified, and its management objectives include a primary focus on watershed protection, with a secondary emphasis on income from timber production.

All KQDC landowners and land managers are interested in improving the balance of age classes on their respective ownerships, and they are actively working to regenerate diverse forests within the KQDC area. All use a variety of expensive techniques to overcome the impacts of overabundant deer in reaching management objectives. These frequently include broadcast herbicide treatment of interfering fern, beech and striped maple, erection and maintenance for 3 to 10 years of 8-foot woven wire fencing in conjunction with either shelterwood seed cuts or removal cuts, and sometimes broadcast application of nitrogen and phosphorous fertilizer to speed the growth of seedlings out of the reach of deer. All have found that even-age silvicultural systems are the only ones sustainable in the face of high deer herds – the slower growth of seedlings in uneven-aged systems dooms them to failure (Marquis and Gearhart 1983). Per acre expenditures to achieve successful regeneration using these techniques can easily run to $800. These landowners have tried a variety of approaches for encouraging hunter use and success on their KQDC and other lands, ranging from encouraging public hunting and open access with road plowing and other services through lease-hunting arrangements that require the lessees to harvest specific numbers of does in order to retain the lease. All are eager to reduce the expense associated with successful regeneration of diverse species.

The Sugar Run Project

At present, the ANF is cooperating with its publics to plan projects for the 11,604-acre Sugar Run Management Area, the only projects likely to occur on National Forest land within the KQDC area during the current decade. As part of the Sugar Run Project analysis and public involvement, a special mailing soliciting input and comments was sent to hunters who have participated in KQDC activities, and ANF managers have also given special thought to silvicultural and other strategies to ease hunter access and improve hunter success as part of the Sugar Run Project.

The Sugar Run Project Area is broadly similar to the entire KQDC landscape, and as this paper focuses on the silvicultural and management strategies to be applied to improve hunter access and success within this project area, we will present details about its landscape characteristics. The project area is covered by second growth forests that originated after very heavy timber harvesting at the turn of the 19th century. Four major forest types dominate the project area—Allegheny hardwoods, mixed upland hardwoods, northern hardwoods, and red maple—which together occupy 89 percent of the project area. Tree species commonly found in the project area include black cherry, white ash (*Fraxinus americana*), tulip poplar (*Liriodendron tulipifera*), red and sugar maple (*Acer rubrum* and *saccharum*), black birch, American beech, oaks (*Quercus* spp.), and hemlock. Table 1 displays the vegetation types and age-class distribution of National Forest System lands currently within the Sugar Run project area.

Almost all (98 percent) of the project area consists of forest cover. Permanent openings, including pipelines, roads, and openings for wells, make up about 2 percent of the National Forest area and generally consist of lowland shrubs, upland shrubs, sparsely stocked riparian bottoms, or ferns and grasses. Roughly 78 percent of the Sugar Run project contains stands that are 51 to 110 years old. Stands that have been recently regenerated, between 0 to 10 years old, account for 3 percent of the National Forest land in the project area.

Stands in the Sugar Run project area have experienced a variety of forest health challenges in recent decades. The project area was defoliated by insects, on average, two to three times between 1984 and 1998. Portions of the project area were defoliated as many as 5 times during this same time period (Morin and others 2001). Insects include both natives and exotics. Outbreaks of cherry scallopshell moth, elm spanworm, forest tent caterpillar, oak leaftier, gypsy moth, and beech bark disease have all occurred on the ANF and have affected the project area. There have also been six years

Table 1—Distribution of forest types and age classes within the Sugar Run Project Area.

	Age class (acres)						
Forest type	**0-10**	**11-20**	**21-50**	**51-110**	**111+**	**Total acres**	**% of total USFS ownership**
Other		19	95	963	13	1090	10
Northern hardwoods	52	6	161	2,839	434	3,492	30
Allegheny hardwoods	363	560	305	2,173	123	3,524	30
Red maple				1,118	28	1,146	10
Mixed upland hardwoods		11	161	2,106	74	2,352	20
Total FS lands	415	596	722	9,199	672	11,604	100
% of total USFS land	3	5	6	78	6		

with drought conditions for a portion of the year during this time period. Disturbances such as defoliation episodes, particularly when concurrent with droughts, may cause mortality that otherwise would not be expected (Morin and others 2001). Recent forest health monitoring completed across the ANF indicated that among the shade-tolerant species, 18.2 percent of the standing sugar maple basal area and 7.3 percent of the beech basal area are dead. Of particular concern is the fact that nearly half of the large beech trees (greater than 20 inches diameter) measured were dead, most likely due to the impacts of beech bark disease complex. Among the more shade-intolerant or shade mid-tolerant species, black cherry was found to have 6 percent, and red maple 7.1 percent, of the standing basal area dead (Morin and others 2001). Among the five most abundant tree species on the ANF, dead trees are proportionally greatest for sugar maple (Morin and others 2001).

In the absence of deer overabundance, this mortality would stimulate the development of diverse advance regeneration, but in the Sugar Run Project Area, it has instead stimulated the development of dense layers of understory plants less preferred by deer or resilient to deer browsing. Approximately 72 percent of the stands considered for treatment, and 75 percent of forested stands in the project area as a whole have interfering understory vegetation of some type.

Deer in the Sugar Run Project Area

Deer are a landscape level species whose distribution is affected by the availability of forage, thermal and hiding cover conditions and seasonal mast availability. As a result, deer use and density varies spatially and seasonally across the project area. In order to better characterize and assess deer and deer related impacts, the project area was broken down into three sub-analysis areas (figure 4). Since deer numbers and impacts are largely determined by hunting and forage availability, existing deer habitat is addressed by looking at a combination of deer density, estimated forage production, and hunter access.

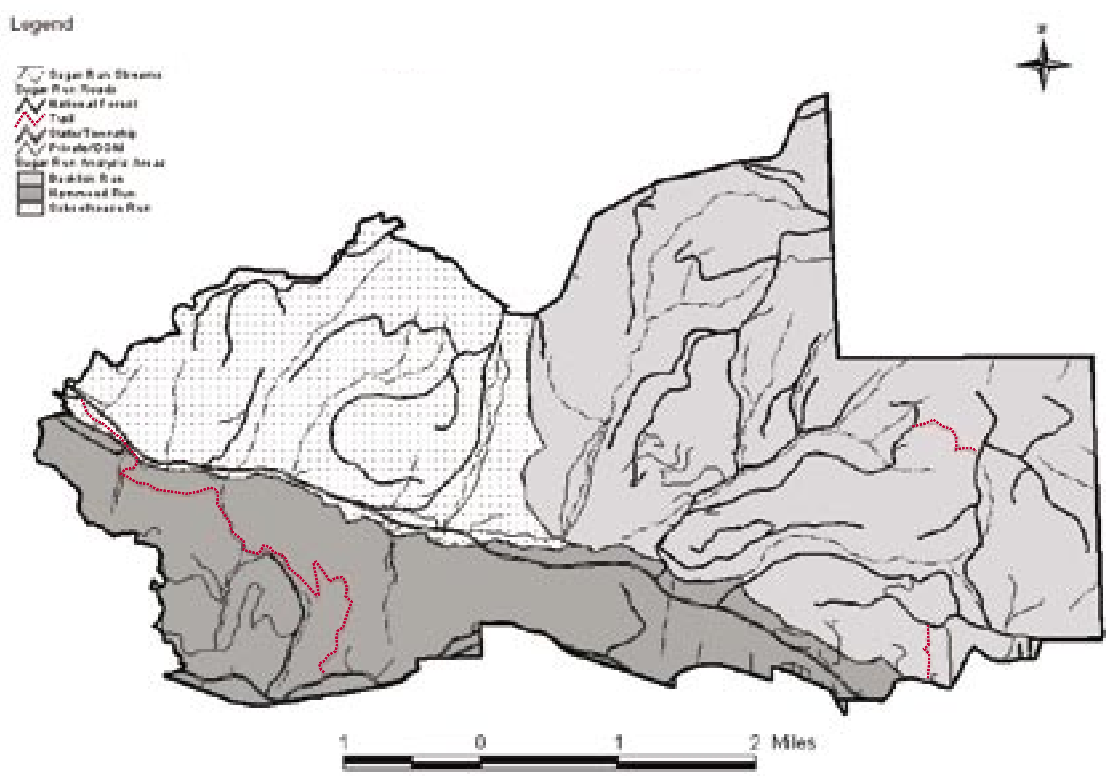

Figure 4—The Sugar Run Project Area and its sub-analysis areas.

Table 2—Winter deer densities in Sugar Run Project Area analysis units. Each estimate is based on three early spring pellet group counts, each conducted over five mile-long transects within a randomly selected mile-square unit.

Analysis area	2002 winter deer density (deer/mi^2)	2003 winter deer density (deer/mi^2)
Bucklick Run (6173 NF acres)	Average = 28.7 Range = 13.3-37.5	Average = 23.3 Range= 12.5-37.9
Schoolhouse Run (2702 NF acres)	Average = 20.1 Range = 13.0-32.4	Average = 34.5 Range = 30.8-41.7

Deer Density

An estimate of deer density was collected from 24 transects across the KQDC in 2002 and 2003. The PGC deer density goal for McKean Co is 20 deer per square mile but the northern half of the KQDC (including the Sugar Run project areas) had winter deer densities of 23.9 and 25.1 deer per square mile respectively in 2001 and 2002, including several "hot spots" ranging from 33 to 41 deer per square mile. Six of the KQDC transects were located in the Sugar Run project area, including three in the Schoolhouse Analysis Area and three in the Bucklick Analysis Area (table 2). Between 2002 and 2003, deer densities in the Schoolhouse area have increased by 70 percent and decreased by 12 percent in the Bucklick Area. During both years, both areas had hot spots, or areas of deer density in excess of 30 deer per square mile. While no deer density measurements were taken in the Hammond Run Area, the lack of tree regeneration, combined with browsing of beech and striped maple indicate high deer density there.

Road Density and Access

The Kinzua Quality Deer Cooperative lands are divided into a north and south zone by a state highway. The northern zone (about 52,000 acres) is about 70 percent National Forest System lands and includes the Sugar Run Project Area. Total road density is 2.5 miles per square mile, including 2.8 miles per square mile on private land and 1.7 miles per square mile on National Forest System lands. Compared to the southern zone, the northern zone has better access from State Highways and paved roads, is larger, and has lower overall road densities on both private and National Forest System lands. Also, 66 percent of the Forest System roads are built to a lower standard and are not open to the public during the late fall/early winter deer seasons (October 1 to January 25). Over 50 percent of National Forest System lands within the north zone are more than ¼ mile from an open road; present road management is not considered adequate to effectively disperse deer hunters.

Forage Availability

Marquis (1987) suggested an index of relative forage availability based on the proportion of forested area in a few broad forage production classes. Marquis (1987) suggested that seedling stands (those 0 to 10 years of age) should be assigned an index value of 10, representing an average forage production of 1000 pounds per acre; thinned stands of older classes (> 50 years) should be assigned an index value of 2.25, representing an average forage production of 225 pounds per acre; and unthinned mature stands should be assigned an index value of 1, representing an average forage production of

Table 3—Recent past, present, and projected future carrying capacity and proportion remote area. Carrying capacity is calculated according to Marquis (1987) as an index of forage production based on stand age class.

	Index of carrying capacity				% remote area	
Analysis area	**1993**	**2003**	**2013**	**2018**	**2003**	**2005**
Bucklick	12,570	7274 (-43%)[1]	8841 (+22%)[2]	7018 (-4%)[2]	42%	23%
Schoolhouse	3,728	3989 (+7%)[1]	5483 (+15%)[2]	3884 (-3%)[2]	84%	39%
Hammond	2,747	2930 (+7%)[1]	3517 (+20%)[2]	3144 (+7%)[2]	52%	26%

[1] % change from 1993
[2] % change from 2003

100 pounds per acre. Sapling and pole stands (ages 10 to 49) are undergoing stem exclusion and produce little forage so are assigned index values of 0 as are fenced stands. Using these indices and site-specific estimates of the past, current, and future age and treatment class within the project area, we can estimate changes in relative forage availability (table 3).

By looking at changes in forage availability over time, this information can also be used to help predict potential deer impacts to understory vegetation within the project area. Table 3 displays past and present forage production resulting from silvicultural activities within each of the sub-analysis areas, as well as reductions in remote areas (>.25 miles from open road) that will result from proposed road management changes.

While available forage in the Schoolhouse and Hammond Run areas have not changed signficantly in the last decade, available forage in the Bucklick area has decreased 40 to 50 percent during the last two decades as a result of fencing new regeneration treatments and growth of previous regeneration units. Considering the present deer density, the reduction in available forage within the Bucklick area, and documented deer impacts, management recommendations included maintaining or improving available deer forage within all three analysis areas, as well as providing strategies to more effectively manage hunters and improve hunter success. This combination of increased hunting success and increased landscape forage should reduce deer impact.

Proposed Actions in the Sugar Run Project

The treatments listed here may raise some eyebrows in a proceedings focused on silviculture. The focus provided by the KQDC project helped planners for the Sugar Run project recognize that many activities not traditionally considered to fall within the "silviculture" toolkit are, in fact, essential to "controlling the establishment, growth, competition, health, and quality of forests" (Society of American Foresters 1998) in this area so heavily affected by deer overabundance. If these activities succeed in reducing deer impact across the project area, they will change the outcome of other silvicultural activities stand-by-stand.

Road Management Changes

Road management changes proposed include (1) opening an additional 8.5 miles of existing Forest System road in the Bucklick and Schoolhouse

Analysis Areas and (2) construction of approximately 1 mile of new road into the Hammond Run Area. These activities will reduce the amount of National Forest System land greater than ¼ mile from a road open to hunters. Changes by Analysis Area are shown in table 3.

Silvicultural Prescriptions

The concept of deer impact as a joint function of forage availability and deer density (figure 3) suggests that both increases in forage availability and decreases in deer density will result in reductions in deer impact. That is, silvicultural treatments that stimulate advance seedling growth simultaneously increase forage on the landscape and provide advance growth for future regeneration treatments. Because increased forage availability has been shown to increase recruitment into the deer herd, it is possible to initiate a vicious cycle. A key assumption of the KQDC leadership team is that forage production increases will be accompanied by increases in hunting pressure and success, resulting in an accelerated reduction in deer impact and development of desirable vegetative communities.

Silvicultural treatments proposed in the Sugar Run project area emphasize even-aged silvicultural systems, a continuing supply of early-successional habitat, and hunter access for the project area to meet KQDC goals. Shelterwood regeneration harvests have several advantages during the transitional effort to increase hunter success while reducing deer impact. Stands that have received the seed cut of a shelterwood sequence have good visibility for hunters, as well as skid trails available to ease hunter movement. Use of shelterwood sequences allows managers to spread forage production over an extended period, using the high forage production capacity of stands that have received removal cuts to reduce deer pressure on stands that are in the seed cut stage of the sequence. Initially, new even-aged regeneration treatments are proposed in the northwestern portion of the project area. Others would be delayed in the eastern portion of the project area to provide a more continuous supply of seedling habitat over a longer time. The delayed shelterwood seed cut treatments would occur when second entry shelterwood removal cuts occur. The second entry shelterwood removal cuts would increase available forage throughout the area, thus reducing deer impacts on the delayed shelterwood treatments, enhancing diverse seedling and herbaceous vegetation development. Reforestation activities needed to ensure the successful establishment of seedlings in both even-aged and uneven-aged treatment areas are also proposed. In addition to the even-aged regeneration treatments, some intermediate thinning treatments, uneven-aged management, non-commercial thinning, and oak and conifer release are proposed.

One contrast between the silvicultural treatments proposed for the KQDC area and silvicultural treatments typically proposed in ANF projects with similar management objectives is restrained use of fencing. While fencing can eliminate deer impact in stands that are directly protected, fencing in large proportions of a forest region has the unintended consequence of increasing deer impact on the unfenced portion of the forest, by reducing the effective, high-forage-producing-area through which a deer can search for forage within its home range. Within the KQDC, managers will try to use well-timed seed and removal cuts with increased hunter access and success to reduce deer impact across the study area.

Under the alternative discussed here, 25 acres would be converted to permanent non-forested openings. Later, when deer impact has been reduced,

Table 4—Acres proposed for regeneration.

	Treatment areas (acres)[1]
Previously initiated even-aged regeneration sequence	220
New even-aged regeneration initiated	312
Total acres proposed for even-aged regeneration	532
Previously initiated uneven-aged regeneration	61
New uneven-aged regeneration initiated	32
Total acres proposed for uneven-aged regeneration	93
Total acres proposed for regeneration	625

[1] Does not include 5 acres proposed for regeneration as part of KQDC Demonstration Area.

regeneration systems such as single-tree and group selection that have been shown to fail under high deer impact may be able to succeed.

Table 4 displays the acres of regeneration proposed and the amount of these treatments that consist of followup on previously initiated regeneration sequences. Changes in forage availabilty and remote area resulting from proposed silvicultural activities are displayed in table 3.

Footbridge Across Sugar Run

State Route 321N forms the northern boundary of the Hammond Run area and serves as a primary hunter access. Sugar Run, a large stream that parallels SR 321, presently restricts hunter access into this analysis area. In order to provide better hunter access from the north, a footbridge across Sugar Run and associated hunter parking lot on SR 321 are proposed. This is expected to facilitate hunter access into many of the lower slopes in the central portion of the analysis area and complement existing crossings to the east and west.

Provide Hunter Access Trails

While SR 59 provides good vehicle access along the southern boundary of the Hammond Run analysis area, there is a nearly impenetrable wall of mountain laurel along SR 59 that makes access by foot very difficult. Additionally, experience on the ANF has shown that hunters will walk farther if they have an old road or trail to follow. As a result, all skid trails from proposed thinnings in Hammond Run will be seeded and laid out in a manner to facilitate hunter movement through the laurel and to provide better access onto the plateau tops north of SR 59.

Develop Openings Along Open Roads for Hunters to View Deer

A total of 25 acres of savannah and opening construction are proposed on nine sites across the project area. All of these areas will provide cool season grasses and legumes and seasonal forage for deer. While six of these sites will be constructed away from existing roads, three acres on two sites will be constructed along open Forest Roads that provide primary access into the Bucklick Area. These sites are being constructed close to roads in order to provide hunters with an opportunity to view deer that are attracted to these openings. Local experience has shown that hunters are more likely to use or hunt in an area where they have seen deer (John Dzemyan, PGC, personal communication). Since many hunters will drive prospective areas throughout the summer and fall in an effort to locate an area that contains deer, these openings are expected to attract deer that currently use the area, so that hunters can view them. This validation that deer are in the area is expected to increase hunter use within the Bucklick Area

Hunter and Harvest Map

Many non-local hunters don't scout the area prior to season and frequently ask about potential areas to hunt as well as local information on deer densities and access. For the last 10 years, the ANF has made a hunter map available that identifies roads, areas of recent timber harvest, campgrounds, and areas of higher deer density.

In order to make hunters more effective at harvesting deer, the KQDC is preparing a deer hunter map. While the present forest map provides general information about the area, the KQDC map will provide very site-specific information that will aid hunters. Information provided on the map will include specific deer density estimates, roads open to hunting in the area and the location of foot access trails, the locations of seasonal food sources such as apple orchards and areas with oak and hickory, openings, and the location of fences (often hunted by muzzleloaders). In addition to the hunting map, a deer harvest map will also be made available that provides the exact locations in which deer were harvested on the KQDC in the past. Like the openings that permit hunters to view deer, a harvest map serves as a "validation" that deer are in the area and can significantly help to generate interest in the area.

Demonstration Area

KQDC was identified as a priority interpretive site in the Master Interpretive Plan recently completed for the ANF[4]. The KQDC was recognized as including federal and private industrial landowners working together to implement a comprehensive management program to improve deer quality, hunter satisfaction, forest ecosystem health, and deer habitat through quality hunting ecology. The KQDC Interpretive site focuses on the hunter and public education role in effective deer management. The overall theme of the KQDC site is "Lands That Everybody Wants—Managing Multiple Uses" and focuses on the topic of deer herd management in relationship to sustainable forest ecosystems. The audience is the general public and hunters. The objectives of this interpretive site are for visitors to:

- Understand the connection between maintaining healthy deer populations and native plant and animal diversity.
- Gain an appreciation of the importance of special hunting regulations to regulate deer herds.
- Understand that managing the deer population is important to meeting ANF stewardship and management objectives.

The Demonstration Area provides easy public access off a state highway, and is located directly across from the Bradford Ranger District. Activities to demonstrate both even-aged (three acres) and uneven-aged (two acres) management including various combinations of associated reforestation treatments are proposed. Specific features associated with the KQDC Demonstration Area include a trail, a parking area/bus turn-around, a brochure, an interpretive kiosk, and signs to identify nine alternative treatments applied to one-acre plots that show the interaction of deer impact and silviculture in sustaining quality deer habitat. The treatments include single-tree selection

[4]USDA Forest Service, Rocky Mountain Region, Center for Design & Interpretation. 2001. Master Interpretive Plan for the Allegheny National Forest. Available from the Allegheny National Forest.

cuts, shelterwood seed cuts, shelterwood removal cuts, and areas with no cutting. Associated treatments include site preparation, herbicide application, fertilization, shelterwood removal cuts, and, in most cases, a contrast between a fenced and an unfenced example of each treatment.

Summary and Management Implications

The success of these silvicultural initiatives in the Sugar Run Project Area will only be known after they've been implemented, and still only if the PGC sustains its current direction of facilitating landowner/hunter coalitions to develop healthy local deer herds in healthy local habitats. But while we wait for the overall outcomes of the KQDC project from an ecological perspective, there are some lessons for silviculture.

First, partnerships represent a good stimulus for creative interdisciplinary thinking about silviculture. In the KQDC project, we benefit from landowner and interagency cooperation at the project level, and from interdisciplinary and land manager-hunter cooperation at the planning and implementation level. Many nontraditional tools, from skid trails through laurel to the concept of creating a "sustained" supply of forage producing condition on the landscape, have emerged from these partnerships.

Second, managing deer impact on silvicultural outcomes is complex and includes a number of tradeoffs, some of which are poorly understood. Fencing, a frequently used silvicultural tool for managing deer impact, gives relatively precise control, but has negative consequences for the condition of the unmanaged forest. A combination of timing the availability of high-forage producing stands with increasing hunter pressure has fewer negative consequences on the landscape but provides much less control to the silviculturist. The reduced costs of using timed forage production and hunting, compared to fencing, to achieve regeneration objectives is an important benefit of this approach, but the risk of failure—hunters aren't interested in the area, PGC policies change—are high.

On balance, we believe that the opportunities to try "new silviculture" and to demonstrate the interaction of deer impact and silviculture to the public are important benefits of the Kinzua Quality Deer Cooperative--a special place.

Acknowledgments

The authors acknowledge the contributions of other people and agencies working on the Kinzua Quality Deer Cooperative and the contributions of other members of the Sugar Run Interdisciplinary Team.

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Restoring the Longleaf Pine Ecosystem: The Role of Container Seedling Technology

James P. Barnett[1]

Abstract*—Longleaf pine (*Pinus palustris *Mill.) ecosystems once occupied 90 million acres in the southern United States' coastal plain. Restoration of longleaf pine ecosystems has been difficult because reestablishment of the species by either natural or artificial means has been problematic. The application of container seedling technology to longleaf pine markedly improves reforestation success. It allows nursery managers and silviculturists to more effectively deal with the inherent seed and seedling characteristics that make longleaf establishment so difficult. Improved seed collecting and processing and seedling cultural techniques have resulted in planting stock that can be successfully established in the field. As a result, a 10-fold increase in longleaf pine seedling production has occurred in the last few years to meet restoration needs across the South.*

Introduction

Restoration of the longleaf pine (*Pinus palustris* Mill.) ecosystem in the southern United States is receiving a great deal of attention (Landers and others 1995, Noss 1989). Longleaf pine ecosystems once occupied over 90 million acres from southern Virginia to central Florida and west to eastern Texas (Frost 1993). These fire-dependent ecosystems dominated a wide array of sites within the region. Today, less than 4 million acres remain (Kelly and Bechtold 1990), with much of this in an unhealthy state. This extensive ecosystem that once contained tremendous timber resources, wide ecological diversity, and essential habitat for many unique plant and animal communities has nearly vanished. Restoration of the ecosystem is an objective on most southern coastal plain National Forests. It, too, is a desired condition on other federal and state lands, as well as on lands of many private landowners. However, until recently, regeneration of the longleaf pine ecosystem has been problematic due to lack of an adequate seed supply, reduction in use of fire that controls competition, poor establishment success with bareroot seedling stock, and seedlings that require several years to initiate height growth.

The success in restoration of the ecosystem is largely the result of recent improvements in the technology to artificially regenerate longleaf pine. This improved technology is based on a better understanding of the ecology of this species. The objectives of this paper are to describe the nature of the longleaf pine ecosystem and its ecological and economic values, and to review the development of the container seedling technology that facilitates the restoration of the species.

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The Longleaf Pine Ecosystem

The natural range of longleaf pine covers most of the Atlantic and Gulf Coastal Plains with extensions into in the Piedmont and mountains of north Alabama and northwest Georgia. The species occurs on a wide variety of sites, from wet, poorly drained flatwoods near the coast to dry, rocky mountain ridges (Boyer 1990). It is a long-lived tree, potentially reaching an age of several hundred years; but longleaf pine forests are often exposed to catastrophic hazards such as tropical storms and to continuing attrition from lightning strikes that cause tree mortality and shorten possible rotation ages (Landers and others 1995).

Longleaf pine is a very intolerant pioneer species and the seedlings go through a stemless grass stage that is usually broken when the young stand and grass is burned in a low intensity ground fire. If competition is severe, they may remain in this grass stage for years. The ecosystem is distinguished by open, park-like stands with a grassy understory, which are composed of even-aged and multi-aged mosaics of forests, woodlands, and savannas, with a diverse groundcover dominated by bunch grasses and usually free of understory hardwoods and brush (Landers and others 1995). The diversity of understory plants per unit of area places longleaf pine ecosystems among the most species-rich plant communities outside the tropics (Peet and Allard 1993). Although the forests are known for persistence and diversity, they often occur on infertile soils. The ecological persistence of these areas is a product of long-term interactions among climate, fire, and traits of the key plants.

Fire was an essential component of the original longleaf pine ecosystems. Longleaf pine and bunch grasses (e.g., wiregrass and certain bluestems) possess traits that facilitate the ignition and spread of fire during the humid growing seasons (Landers 1991). Frequent fire was largely responsible for the competitive success of longleaf pine and its associated grasses. These keystone species exhibit pronounced fire tolerance, longevity, and nutrient-water retention that reinforce their dominance and restrict the scale of vegetation change following disturbance. Fires that were ignited by Native Americans or that resulted from thunderstorms with frequent lightning prevailed over the region. Many of these fires occurred during the growing season and largely prevented species native to other habitats from encroaching into the pine barrens. The chronic fire regime also maintained the soil structure and poor nutrient dynamics to which longleaf pine is adapted (McKee 1982). These fire effects tended to make longleaf pine sites more favorable to resident species than those indigenous to more nutrient-rich habitats.

Decline of the Longleaf Pine Ecosystem

The depletion of the longleaf ecosystem resulted from its many desirable attributes that have caused it to be exploited since the settlement of the nation by Europeans (Croker 1979). However, it was the event of railroad harvesting in the late 1800s and early 1900s that provided access to and depleted the vast remaining longleaf timberland. Cutting proceeded from the Atlantic states west through the Gulf Coast Region with increasing intensity of use with time. Longleaf pine logging reached a peak in 1907, when an estimated 13 billion board feet were cut (Wahlenberg 1946). The longleaf pine ecosystem now occupies only a small part (less than 5 percent) of its original area. This habitat reduction is the reason for the precarious state of at least 191 taxa of vascular plants (Hardin and White 1989, Walker 1993) and key wildlife species such as the red-cockaded woodpecker, gopher tortoise, and southern fox squirrel (Landers and others 1995).

Regeneration of longleaf pine was limited because of a combination of circumstances. The completeness of the harvest left little seed source for natural regeneration and much of the harvested land was cleared for cropland or pasture. Longleaf pine does not successfully invade open land in competition with more aggressive pine or grass species. Regeneration sometimes succeeded the removed old-growth when periodic fires provided a seedbed and controlled woody competition, and when wild hogs did not reach a density high enough to destroy established seedlings (Wahlenberg 1946). The disruption of natural fire regimes, resulting in part from forest fire protection policies implemented during the 1920s, allowed invasion of longleaf sites by hardwoods and more aggressive pine species. Regeneration, both naturally and artificially, is more difficult than for any other southern pine due to the delay in stem elongation (the grass stage) that is a genetic trait of the species. Also, survival of planted bareroot nursery stock is generally poor and established seedlings in the grass stage are very sensitive to competition.

Restoring the Ecosystem

A key to restoration of the longleaf pine ecosystem is to ensure that society understands the value of the species and its habitat. Without economic benefits, long-term conservation projects usually do not succeed (Oliver 1992). Longleaf pine forests have high economic value due to the quality of solid-wood products produced. Harvesting or forest management need not be eliminated or restricted to restore and maintain longleaf pine ecosystems, as evidenced by the fact that logging at the turn of the century apparently had little effect on groundcover diversity (Noss 1989). Restrictions on harvest would be a disincentive to many landowners and could result in the elimination of much of the remaining longleaf pine on private lands.

Restoration of the longleaf pine ecosystem is achievable since pockets of longleaf pine occur across much of its former range. It should be feasible to gradually expand longleaf pine acreage through education, research, and commitment on the part of resource managers. Restoration is now a goal on much of the public land in the southern United States, where longleaf pine remains as a component of the forest. In fact, much of the current acreage of the ecosystem occurs on public lands.

A number of interacting factors will determine whether the restoration of the longleaf pine ecosystem can be achieved. These include the capability to successfully regenerate longleaf pine on its native sites, to use fire to enhance establishment and management of both the overstory and understory species, to educate the public and resource managers on the value and technology of restoration, and to evaluate restoration success.

Restoration Technology

Utilization of the trees in the original forest was so complete that inadequate numbers of seed trees remained to naturally regenerate many of the harvested stands. So, artificial regeneration must be used to restore longleaf on many of the appropriate sites where it originally grew. Until recently, regeneration success from planting was generally unacceptable due to problems related to severe competing vegetation, delayed stem elongation, and poor storability of bareroot seedlings. We now have the knowledge and technology to reestablish longleaf pine by planting bareroot stock. The keys to successful establishment are: well-prepared, competition-free sites; healthy, top-quality, fresh planting stock; meticulous care of stock from lifting to planting; precision planting; and proper post-planting care (Barnett 1992). Failure to implement a single step in the regeneration chain of events can doom a plantation. Bareroot stock is very sensitive to damage during lifting, handling and planting, increasing the chance of poor performance when planted in the field.

Merits of Container Technology

Planting of container stock is now accepted as the most successful method of regenerating longleaf pine (Barnett and McGilvray 1997). Production of longleaf container seedlings has increased from about 15 million to 85 million annually in the last five years. This improved survival and growth is generally attributed to root systems that remain intact during lifting while roots of bareroot plants are severely damaged. Thus, container seedlings experience a significantly shorter period of transplant shock or adjustment than bareroot stock. However, using container stock does not eliminate the critical need for controlling competition during the first growing season after planting.

The establishment of a facility to produce contain seedlings is simple, and can be done at a fraction of the cost of establishing a bareroot nursery. A container nursery facility does not require any special attention to soil type or soil health, since seedlings are grown in potting medium in containers rather than in the soil were the nursery is located. Two elements of physical infrastructure are needed for the container nursery—an adequate supply of water for timely irrigation of seedlings, and some means to deploy netting over the seedlings as germination occurs to minimize predation of seeds from birds (Barnett and McGilvray 1997, Barnett and others 2002).

Essentials for Container Production

Availability of high quality seeds currently limits production of longleaf pine container seedlings. Recently guidelines for producing quality longleaf pine seeds have been published that improve both germination and yield of plantable stock (Barnett and McGilvray 2002a). An important aspect of this increased performance is the reduction of seed coat pathogens that negatively affect germination and seedling establishment (Barnett and McGilvray 2002b).

Prompt and uniform germination is important in seedling production. Once germination has occurred, it is critical to follow established protocols for growing the seedling crop (Barnett and McGilvray 1997). Normally crops are sown in April and are ready to be outplanted in October or November—whenever soil moisture will allow.

Determining Stock Quality

Until recently, broadly recognized standard specifications for container longleaf seedlings were lacking. Establishing standards based on research has been difficult because “substandard” container stock will survive in

Figure 1—Interim specifications for longleaf pine container seedling

Characteristics	Preferred	Not Acceptable
Needles		
Length if not top clipped	8 to 12 inches	<4 inches
Length if top clipped	6 to 10 inches	<4 inches
Fascicles	Many present	None present
Color	Medium to dark green	Yellow or brown
Roots		
Root collar diameter[a]	≥¼ inch	<3/16 inch
Color	Light brown with white tips	Black (diseased)
Mycorrhizae	Present (the more the better)	
Evidence of disease	None present	Any present
Root spiraling	None present	Any noticeable amount
Buds		
Present	Present on 90 percent of crop	
Color	Green to brown	Yellow or chlorotic
Container size (per plant)		
Diameter	≥1.5 inches	<1 inch
Length	≥4.5 inches	<3.5 inches
Volume	≥6 cubic inches	<5.5 cubic inches

Other important attributes
Firmness: Plug stays intact when extracted and during handling; no loss of potting medium.
Moisture: Root plug is always moist, never dry.
Pests: No competing weeds or insects are present.
Sonderegger: Buyer specifies whether to cull Sonderegger seedlings.

[a] At base of needles

years when rainfall is abundant, but early development will be behind that of a good seedling. Developing seedling grades requires outplanting and performance evaluation of seedlings with a range of physiological and morphological characteristics—tested over a number of years and over a range of site conditions. Lacking the resources to conduct the research to establish standards, the Longleaf Alliance and two USDA Forest Service units—the Cooperative Forestry group in Atlanta and the Southern Research Station silviculture unit in Pineville—decided to seek agreement among producers and users on acceptable seedling criteria.

A canvassing of those who produce and use longleaf pine container stock for their recommendations revealed that available information was insufficient to develop three different seedling grades similar to those for bareroot stock (Wakeley 1954). So, we decided to develop only two, "preferred" and "non acceptable." These guidelines (figure 1) are now used across the South (Barnett and others 2002).

Application of Container Technology

Because of difficulty in obtaining consistent field survival with bareroot stock, about 85 percent of longleaf pines now planted are container grown seedlings. The ability to plant container stock early in the fall also improves field survival and early height growth on many sites. The ability of fall planted seedlings to establish a root system during the winter months usually results in earlier initiation of height grown.

The Role of Fire

Fire is an essential component of the restoration and management of the longleaf pine ecosystem. Long-term studies show that the frequent use of fire hastens initiation of height growth by reducing undesirable competing vegetation and foliage that is infected with brown-spot needle blight (*Mycosphaerella dearnessii* Barr.) (Siggers 1934). Longleaf pine seedlings that are in grass stage are resistant to injury by fire because the buds are protected by a rosette of needles (Walker and Wiatt 1966). Prescribed fire also stimulates growth and development of species that are an essential component of the understory. Seasonal burning studies show that late spring burns are much more effective in the restoration process than the typical winter burns that are usually favored by other pine species because they are hotter and more effective in reducing competing woody vegetation (Grelen 1978, Haywood and others 2001). Fire is an important element in establishing the species and is a critical component for achieving and maintaining the biologically diverse understory that is characteristic of the ecosystem.

Education and Commitment

Education of the public regarding the current status of the longleaf pine ecosystem, its potential economic value, its outstanding biodiversity, and the role of fire in maintaining the system is an initial step in securing support for restoration (Landers and others 1995). A primary need in this process is to promote the use of fire as an ecological force necessary to maintaining this fire-dependent ecosystem. Frequent prescribed burning, including use of growing-season fires where appropriate, promotes the diversity and stability of these communities (Noss 1989). Many private landowners are concerned about the environment and will support restoration, if through the process they generate income from their land. Longleaf pine can be managed in an ecologically sensitive manner that generates income satisfactory to interest a landowner in restoration (Landers and others 1990).

Determining Success

One way to measure the success of the restoration process is to determine through periodic forest surveys if the area in the longleaf pine type increases. Another method is to determine if the production of longleaf pine nursery stock increases in relation to the other southern pines. Some would question whether an increase in area of longleaf pine plantations equates to an increase in ecosystem restoration. Certainly it takes more than planting trees to restore the ecosystem, but it is the critical first step. Recent research indicates that the productivity of an ecosystem is controlled to an overwhelming extent by the functional characteristics of the dominant plants (Grime 1997). So, with reestablishment and appropriate management, including the appropriate use of fire, restoration processes that include development of the typical diverse understory vegetation will begin.

Conclusions

Restoration of the longleaf pine ecosystem is facilitated by planting seedlings produced in containers. Planting of longleaf is required on most sites needing restoration because sites normally are being converted from agricultural crops or other pine species. Quality container stock survives better than bareroot stock on typical longleaf pine sites and the length of time seedlings stay in the grass stage is reduced. Restoration requires more than just planting of longleaf seedlings. Survival and initiation of height growth requires the control of competition by fire or other means and careful control of the planting "chain of events." Container stock is not as sensitive to handling problems, but still needs good site preparation, care during handling, and precision planting. Fire is an important component needed to establish and maintain the longleaf ecosystem.

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Restoration and Management of Eastern White Pine Within High Blister Rust Hazard Zones in the Lake States

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Abstract*—In areas considered high hazard for blister rust in the northern Lake States, six white pine plantings were established between 1989 and 1999 to: (1) evaluate the impacts of blister rust, white pine weevil, browsing, and competition stress on tree growth and survival, and (2) evaluate the effectiveness of genetic and silvicultural strategies to minimize damage. The effectiveness of a genetic approach is being examined by evaluating seedlings from selected rust-free source trees (seed orchard collected seed) vs. non-selected nursery seedlings (field collected seed). The silvicultural approach is being examined by comparing pruned vs. unpruned trees, and shelterwood vs. clearcut treatments. Early results based on data collected through 2003 are summarized in this paper. Deer and hare browsing have caused widespread mortality at two sites. Competition has been intense on many of the sites, especially in the open-grown (clearcut) treatments, severely affecting growth and survival. The incidence of tree mortality caused by blister rust has been relatively minor, with Armillaria root disease killing more trees than blister rust. Results are too preliminary to fully evaluate the long-term effectiveness of pruning, shelterwood treatment, or planting selected stock in reducing blister rust incidence.*

Introduction

Prior to the mid-1800s, eastern white pine, *Pinus strobus* L., played a dominant role in many of the forest ecosystems in the Lake States of Michigan, Minnesota and Wisconsin. Gevorkiantz (1930) estimated that Wisconsin alone had over 7.3 million ha that included a significant white pine component. Logging of the Lake States pinery occurred throughout the mid and late 1800s, almost eliminating the mature white pine resource. Between 1850 and 1930, over 104 billion board feet of white pine lumber was removed from northern Wisconsin (Gevorkiantz 1930). Widespread harvesting created conditions conducive to destructive fires that killed white pine regeneration and many of the remaining large white pines. Thus, future seed sources were removed from many areas. By the early 1900s the white pine resource was significantly reduced from its status 100 years earlier. Recovery has been very slow. Forest Inventory and Analysis (FIA) survey data from Michigan (1980), Minnesota (1990) and Wisconsin (1983) revealed the area of timberland in the white pine type to be only 203,564 ha (Spencer and others 1992).

Damaging Agents

There are a number of reasons why there has been limited success in efforts to restore eastern white pine. These include the introduction of the blister rust fungus *Cronartium ribicola* J.C. Fischer ex. Rabenh., a lack of

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seed trees in many locations, outbreaks of white pine weevil *Pissodes strobi* Peck, and high white-tailed deer *Odocoileus virginianus* Zimmermann populations. These limitations have given white pine a reputation as a difficult species to manage with some forest managers (Marty 1986, Jones 1992), resulting in reduced planting and decreasing the likelihood of a significant recovery of the species.

Blister Rust Hazard Zones

White pine blister rust was first detected in Wisconsin in 1913, Minnesota in 1914, and Michigan in 1917 (Benedict 1981). The fungus produces spore stages on white pine and its alternate host, species of *Ribes.* The Lake States region's cool moist weather patterns in the late summer and fall, prevalence of many lakes and wetlands, and abundant *Ribes* populations created ideal conditions for blister rust.

Climatic blister rust hazard zone maps for the region were developed in the 1960s (Van Arsdel 1961a, 1964). Much of the northern Lake States region was in zones 3 and 4, indicating moderate to high hazard of blister rust incidence (figure 1).

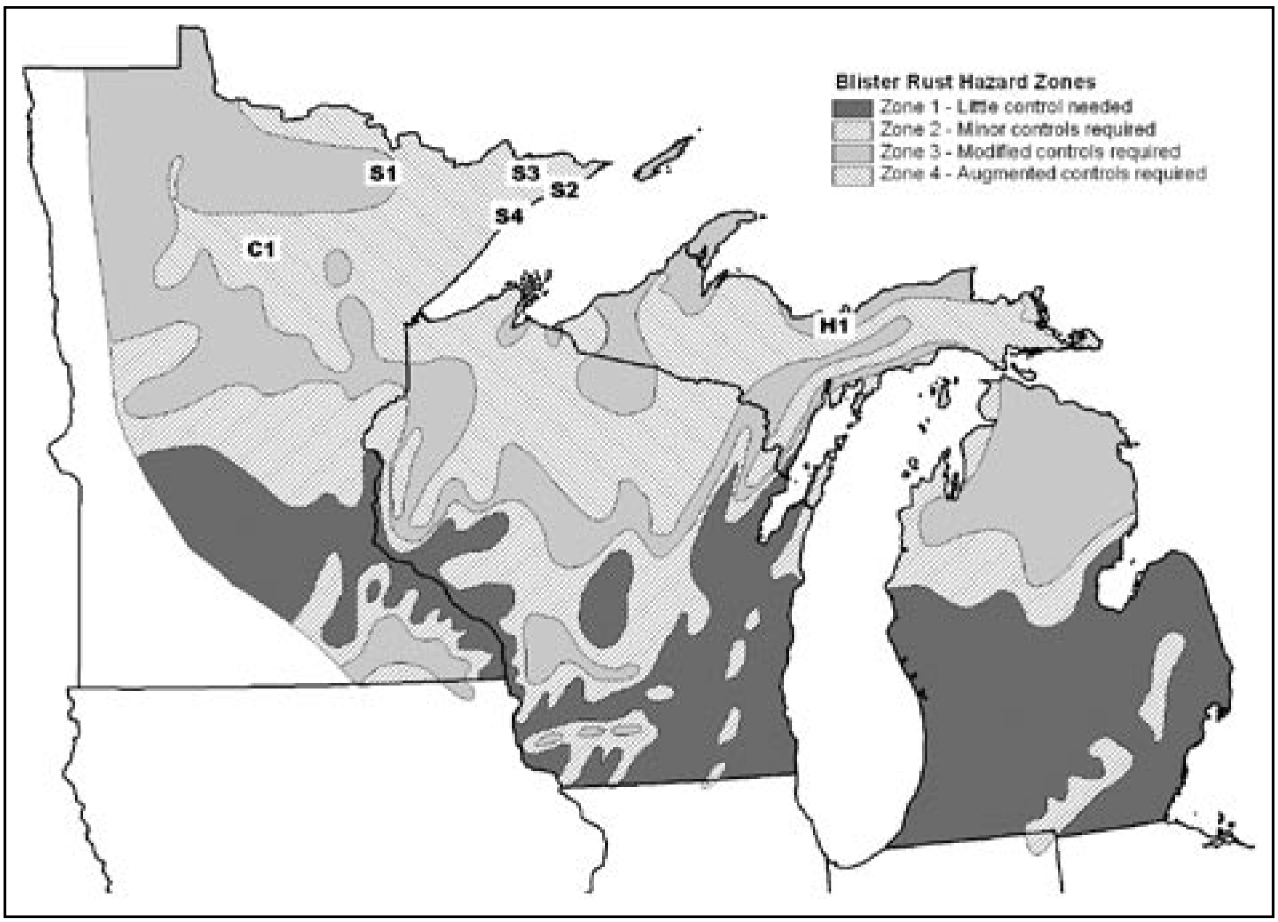

Figure 1—Climatic hazard zones of blister rust infection potential. Van Arsdel (1964) described the zones as follows: Zones 1 and 2—enough pines survive to give a commercial stand without controls. Zones 3 and 4—recommended controls included maintaining an overstory of thin-crowned species over young trees, pruning, avoiding small openings, and maintaining a closed white pine canopy when white pine is open-grown. In addition, in blister rust hazard zone 4, Van Arsdel recommended planting rust-resistant seedlings. Research/demonstration sites are H1 on the Hiawatha National Forest, C1 on the Chippewa National Forest, and S1-S4 on the Superior National Forest.

In the Lake States, rust incidence varies greatly across each zone, dependent upon local topography and vegetation. It is not uncommon for trees in areas within the highest hazard zones to have a low rust incidence. Substantial local variation was recognized at the time of the development of the broad climatic hazard zone maps (Van Arsdel 1961b, 1972). Several surveys have confirmed that the incidence of rust varies in the high hazard zones. Robbins and others (1988) surveyed stands in the Upper Peninsula of Michigan and reported that only 1.5 percent of the sampled trees were diseased. In northern Wisconsin rust incidence in pole-sized stands was only 7.2 percent, and varied from 0 to 28.6 percent (Dahir and Cummings Carlson 2001).

Refinement of the climatic hazard maps to identify areas of low rust incidence may provide more opportunities for successful white pine restoration. White and others (2002) created a high-resolution map for a portion of northern Minnesota using geographical information system (GIS) techniques. Their map illustrated that there were significant acreages of "low hazard" in areas previously identified as zone 4, the area of highest risk for blister rust.

Despite the constraints in managing white pine mentioned earlier, there is growing interest in the restoration of white pine in the Lake States (Stine and Baughman 1992). However, refinements of existing management recommendations for establishing white pine across a regional landscape are needed to identify areas where restoration is likely to be successful.

To address this need, we established six white pine plantings between 1989 and 1999 in the northern Lake States to evaluate the impacts of blister rust, white pine weevil, browsing, and competition on tree survival and growth and to compare silvicultural and genetic strategies to minimize damage. These plantings are managed by USDA Forest Service Ranger Districts, following local management guidelines. In this paper, we present and discuss early results on survival and growth of selected and non-selected seedling stock in relation to browsing, blister rust, competition and other mortality agents.

Material and Methods

Seedling Stock

Seedlings of the "selected" source trees were grown at the USDA Forest Service Toumey Nursery in Michigan. Seed for these trees was collected at the USDA Forest Service Oconto River Seed Orchard (ORSO) in Wisconsin. ORSO trees originated from source trees selected in the 1960s for good tree form and freedom from blister rust in stands with a high rust incidence. At one site, Superior 1, trees were derived from an additional selected seed source originally obtained by the Minnesota Quetico-Superior Wilderness Research Center (WRC) and grown by a private greenhouse to provide containerized planting stock. The non-selected seedlings were obtained from a variety of field collected seed sources, most of unknown parentage.

Treatments

Various treatments were imposed on replicated, randomized 25-tree plots. Trees were planted at 3 x 3 m spacing. Treatments were: (1) nursery grown seedlings from selected rust-free source trees vs. nursery grown non-selected seedlings, (2) pruned vs. unpruned trees, and (3) trees planted under a shelterwood vs. trees planted in clearcuts. All of the study sites included selected vs. non-selected seedlings. Not all of the sites included the shelterwood vs. clearcut treatments and trees have not yet been pruned at some of the locations. Pruning was planned to be initiated five years after planting, depending upon growth rates.

Data Collection and Analyses

Tree survival, tree size, incidence of browse, blister rust, white pine weevil, and other damaging agents were recorded annually through 2003. Tree size was measured using tree height for the initial 10 years and diameter at breast height thereafter. Tree size was analyzed using ANOVA and pair wise comparisons were made using Tukey's HSD comparisons. Logistic regression was used to compare tree survival, the prevalence of blister rust, Armillaria root rot, and white pine weevil attacks among planting stock types. Backward stepwise regression and the software Arc (Cook and Weisberg 1999) were used to test variables in the analyses.

Study Site Descriptions

All of the plantings are within blister rust hazard zones 3 and 4, the two highest climatic hazard zones proposed by Van Arsdel (1964). Sites are located on USDA Forest Service National Forest lands and are all within the Laurentian mixed forest. Each site is described in detail below and study locations are shown in figure 1. Sites are placed into Sections as defined by the National Hierarchical Framework of Ecological Units (McNab and Avers 1994).

Hiawatha 1 (H1), established in 1989, was located on the Munising Ranger District, Hiawatha National Forest, within Section 212H, Northern Great Lakes. Northern hardwoods occupied the site prior to plot establishment. Treatments included planting trees under a shelterwood vs. planting in a clearcut, selected vs. non-selected seedlings, and pruned trees vs. unpruned trees. Non-selected trees came from seed collected locally on the Hiawatha National Forest. A total of 1,200 trees were planted in 48 plots across six treatment blocks. Pruning was initiated in 1994.

Chippewa 1 (C1) was located on the Cass Lake Ranger District, Chippewa National Forest, within Section 212N, Northern Minnesota Drift and Lake Plains. Plots were established in 1998 within a series of small (0.04 to 0.27 ha) harvest units resulting in a series of small clearcuts. The local forest type was a mix of aspen *Populus tremuloides* Michx., paper birch *Betula papyrifera* Marsh., red pine *P. resinosa* Ait., jack pine *P. banksiana* Lamb. and a few scattered large white pine. Treatments included selected vs. non-selected seedlings, and pruned vs. unpruned trees. There was no shelterwood treatment on this site. Non-selected seedlings came from the State of Minnesota, Willow River Nursery. A total of 600 trees were planted in six replicated blocks that included the four treatments. Pruning was initiated in 2001.

Superior 1 (S1) was located on the LaCroix Ranger District, Superior National Forest, within Section 212L, Northern Superior Uplands. The clearcut site was occupied by a two-year-old aspen stand when planted in spring 1997. Treatments included two selected seedling sources (ORSO and WRC) vs. non-selected seedlings, and pruned vs. unpruned trees. Non-selected trees came from the State of Minnesota, Willow River Nursery. There was no shelterwood treatment on this site. A total of 900 trees were planted in six replicated blocks that included the six treatments. Pruning has not been initiated.

Superior 2 (S2) was located on the Gunflint Ranger District, Superior National Forest, within Section 212L, Northern Superior Uplands. This site was occupied by a mature paper birch stand with harvest treatments completed in 1996 and planting in the spring of 1997. Treatments included planting trees under a shelterwood vs. planting in a clearcut, and selected vs. non-selected seedlings. The shelterwood was established as narrow strips (15.2 m) cut through a mature paper birch stand that was adjacent to the clearcut unit. Non-selected trees came from the State of Minnesota, Willow River Nursery. A total of 1,200 trees were planted in 12 replicated blocks that included the four treatments. This site was replanted in 1998 because of heavy browse damage. Pruning has not been initiated.

Superior 3 (S3) was located on the Gunflint Ranger District, Superior National Forest, within Section 212L, Northern Superior Uplands. The stand was mature red and white pine. A seed tree harvest was completed in 1998 and the site was planted in 1999. Treatments included selected vs. non-selected seedlings, and pruned vs. unpruned trees. No shelterwood or clearcut treatments were applied. Non-selected trees came from the State of Minnesota, Willow River Nursery. A total of 600 trees were planted in six replicated blocks that included the four treatments. Pruning was initiated in 2003.

Superior 4 (S4) was located on the Tofte Ranger District, Superior National Forest, within Section 212L, Northern Superior Uplands. The stand was mature red and white pine. A seed tree harvest was completed in 1998 and the site was planted in 1999. Treatments included selected vs. non-selected seedlings, and pruned vs. unpruned trees. No shelterwood or clearcut treatments were applied. Non-selected trees came from the State of Minnesota, Willow River Nursery. A total of 600 trees were planted in six replicated blocks that included the four treatments. Pruning has not been initiated.

Mechanical release of seedlings from competing vegetation has been done on the plots on an as-needed basis. Control of animal browsing was done on sites C1 and S2 using paper terminal bud caps. The chemical animal deterrent (Plantskydd®) was used at C1.

Results

Hiawatha 1

In 2003 survival of non-selected and selected trees was similar, 55 and 56 percent respectively ($p = 0.91$). Survival was lower in the clearcut treatment compared with trees in the shelterwood, 51 vs. 63 percent ($p < 0.01$). Much of the early mortality was caused by Armillaria root disease and unknown causes.

Blister rust incidence was less on the selected trees (3 percent) compared with non-selected trees (7 percent) ($p = 0.03$). Blister rust incidence on all trees was greater in the shelterwood treatment (7 percent) compared with the clearcut treatment (2 percent) ($p < 0.01$).

The incidence of white pine weevil attack on surviving trees was greater in the clearcut treatment (56 percent) than in the shelterwood treatment (41 percent) ($p < 0.01$).

Tree diameter was greater in the clearcut than in the shelterwood, 7.7 cm (SE 0.144) vs. 4.2 cm (SE 0.134) ($p < 0.01$). In the clearcut plots, competition from sprouting hardwood stumps has affected growth of some trees.

Deer or hare browse has not occurred on this site. Diameter of selected trees (6.5 cm) (SE 0.163) was greater than non-selected trees (5.2 cm) (SE 0.165) ($p < 0.01$). The diameter of pruned trees (5.6 cm) (SE 0.170) was less than unpruned trees (6.1 cm) (SE 0.164) ($p = 0.03$).

Chippewa 1

In 2003 there was no significant difference in tree survival between non-selected (66 percent) and selected trees (69 percent) ($p = 0.43$). Selected trees were taller (1.3 m) (SE 0.035) than non-selected trees (1.2 m) (SE 0.034) ($p = 0.01$).

Damaging agents have had minor impacts thus far. Armillaria root disease killed 27 trees between 2000 and 2003. This disease has probably killed additional trees, but this could not be confirmed. In 2003, blister rust incidence was still low: three trees were killed and nine additional trees were diseased. Blister rust incidence was similar on selected trees (three diseased, two killed) and on non-selected trees (six diseased, one killed). Browse damage on trees at this site has been minor. Competition from woody and herbaceous vegetation, overtopping the young pine, has been severe.

Superior 1

Browse damage and competition were major contributing factors to tree mortality on this site. In 2003, overall survival was 56 percent. Survival was greater in non-selected trees (66 percent) than in selected ORSO (53 percent) and selected WRC (49 percent) trees ($p < 0.01$). Deer and snowshoe hares have caused extensive browse damage. In 2002 more than 85 percent of the trees had been browsed to near ground level. Competition stress from aspen suckers along with other woody and herbaceous growth has been heavy.

After six growing seasons, mean tree height was less than 1.0 m, largely due to browsing and competition. The non-selected trees were taller (0.7 m) (SE 0.021) than either the selected WRC (0.5 m) (SE 0.025) or selected ORSO (0.5 m) (SE 0.024) trees ($p < 0.01$). There have been no weevil attacks.

There were no significant differences in blister rust incidence among selected and non-selected trees ($p = 0.91$). A total of 21 trees (six killed) have blister rust cankers: eight non-selected trees, six selected ORSO trees, and seven selected WRC trees.

Superior 2

In 2002, tree survival of the non-selected and selected seedlings was 29 and 24 percent respectively. This site was replanted in 1998 after the original 1997 planting failed because of extensive browsing and competition stress. In 2002, over 97 percent of the surviving trees were browsed. This was despite the use of paper bud caps used as browse protection. Further, the use of paper bud caps on the small trees caused trees to bend and become deformed under snow.

Superior 3

In 2003, overall tree survival was 80 percent. Survival between non-selected and selected trees was 78 percent and 83 percent, respectively ($p = 0.22$). In 2003, non-selected trees (1.1 m) (SE 0.023) were taller than selected trees (1.0 m) (SE 0.023) ($p < 0.01$).

Damage to trees by disease and herbivory has been minimal thus far. Armillaria root disease was confirmed on 15 trees that died between 2001 and 2003. Blister rust incidence has been relatively low; 14 trees have been killed (eight selected, six non-selected) by blister rust through 2003 with 20 additional diseased trees seven selected, 13 non-selected). Competition from woody and herbaceous vegetation has been minimal.

Superior 4

In 2003, overall tree survival was 56 percent. Recent tree mortality was caused by equipment used for salvage logging in the area in spring 2003. Because tree injury was extensive in four plots, these plots were removed from the data analyses. In the remaining plots survival was 67 percent. Differences in survival between non-selected and selected trees were minimal, 66 percent and 69 percent respectively ($p = 0.42$). Mean tree heights for non-selected trees were (1.1 m) (SE 0.027) and selected trees (1.0 m) (SE 0.014) ($p= 0.13$).

Pest incidence and severity has been low. Armillaria root disease was confirmed on 14 trees that died between 2001 and 2003. Blister rust has affected only six selected trees (two dead) and one non-selected tree. The incidence of browse damage has been low; however, competition from woody and herbaceous vegetation overtopping the young pine has been severe.

Combined Results Across Sites

Overall, tree survival of the non-selected trees (67 percent) was greater than the ORSO selected trees (62 percent) ($p = 0.01$). More non-selected trees (4.1 percent) were infected by *C. ribicola* than ORSO trees (2.6 percent) ($p = 0.04$). However, there was no significant difference in the number of non-selected selected trees (1.1 percent) killed by blister rust compared to the ORSO trees (1.4 percent) ($p = 0.50$).

Discussion

Deer and hare browsing and competition stress have been strong contributors to tree mortality on several of the sites, especially Superior 1 and 2. Many trees were browsed to the ground line. Trees on other sites, e.g., Superior 3 and 4, had little browsing damage. In a recent Minnesota study, Krueger and Puettmann (in press) concluded that herbivory was more likely to cause plantation failure than insect or disease incidence. White pine is a winter food source (January through March) for white-tailed deer (Rogers and others 1981). The intensity of browse on white pine occurring at the local level can be influenced by over-wintering deer populations and the local availability of more preferred food sources (Hamerstrom and Blake 1939). High incidence of feeding damage can occur in one area while in other areas little damage may occur. In addition to deer and snowshoe hares, moose also browse white pine (Pastor 1992).

The use of paper bud caps for protection against deer browsing has been widely recommended in Minnesota. However, in our experience, the bud caps themselves may be an impediment to the growth of young trees. The paper bud caps restricted terminal growth on many of the smaller seedlings (<10 cm tall) and often distorted the terminals while under snow cover. Bud caps should probably not be used until trees reach a height of 50 cm.

Though woody and herbaceous competition has not been quantitatively measured, it is evident that growth and survival of pine on several sites has been severely affected by heavy cover. This has been especially evident in the clearcut treatments at the Superior 1 and 2 and Chippewa 1 sites and on the nutrient rich, mesic sites.

Mechanical removal of competing woody vegetation was done at the Superior 1, Chippewa 1, and Hiawatha 1 sites. However, within a year resprouting, young pine was again overwhelmed by woody vegetation. Grass and other herbaceous vegetation often smothered young pines after heavy snowfalls.

Van Arsdel (1964) recommended planting rust resistant seedlings in hazard zone 4 as the most effective control measure for that zone. Early data from Hiawatha 1 (Ostry 2000) indicated that the ORSO selected stock had significantly lower blister rust incidence than non-selected stock. Although these data reported here are preliminary, this trend has continued on the Hiawatha 1 site; but across all sites no significant difference was detected in the number of trees killed among selected and non-selected trees.

Most fatal blister rust cankers occur in the lower portions of trees, thus pruning of lower branches can significantly reduce the likelihood of lethal cankers (Weber 1964; Lehrer 1982; Hunt 1991). Van Arsdel (1964) recommended that pruning be initiated as early as two years after planting. At that time it also may be advisable to remove needles on the main stem because we have documented that stem cankers on the lower stem can originate from these infected needles.

An existing overstory can reduce blister rust incidence by promoting dew formation in the upper canopy (Van Arsdel 1961b) rather than on the seedlings growing in the understory. Microclimatic conditions are crucial in the movement of fungal spores and infection of pines (Van Arsdel 1967). However, at the Hiawatha 1 site, infection of trees has occurred despite the shelterwood treatment. These trees were closest to a local population of *Ribes*. This supports the recommendation to eliminate *Ribes* in proximity to pine. On this site the presence of *Ribes* adjacent to trees in the shelterwood treatment is apparently making the overstory protection ineffective.

There was an intensive effort to eradicate *Ribes* plants in many forested areas beginning in the early 1900s. This effort lasted over 50 years and represented the largest tree disease control program ever undertaken in the United States (Benedict 1981). The program was eventually largely abandoned and viewed as not effective in many parts of the country. However, it was deemed partially effective in the East (Benedict 1981, Maloy 1997) and has been shown to be effective in local areas (Van Arsdel 1972, Stewart 1957). Ostrofsky and others (1988) reported that in Maine, where *Ribes* eradication efforts had been ongoing for over 70 years, incidence of blister rust was reduced by over 50 percent in treated areas.

White pine weevil attacks do not kill trees but can lead to stem deformity and significant value loss in wood products (Brace 1971). To date, trees have been damaged by white pine weevil only at the Hiawatha 1 site, the oldest planting. The weevil attacks were more prevalent on trees in the clearcut treatment. Weevil populations are expected to increase at all sites as the trees reach 10-20 years of age. White pine weevil preferentially attacks the largest, most vigorous trees in a stand (Kreibel 1954), and open-grown white pine are more heavily attacked by this weevil than trees grown in shade (Graham 1918). Management recommendations for reducing weevil impacts have largely been based on growing trees under an existing canopy. However, shade reduces growth rates, especially diameter growth (Borman 1965). Pubanz and others (1999) observed that in 30-80 year-old open-grown plantations of white pine in Wisconsin, sufficient numbers of "crop-trees" did develop even though weevil attacks were prevalent. They concluded that weevil impacts were often over-stated and recommended that management strategies stress pruning, maintaining unsuppressed crown position (open-grown), and full stocking. Our study should allow further evaluation of the differences in the incidence of weevil attacks on trees growing in the clearcuts vs. shelterwood and pruned vs. unpruned treatments.

Management Implications

While white pine blister rust is a serious disease, it is only one of several damaging agents affecting early tree survival and growth. Factors such as woody and herbaceous competition, browsing, and Armillaria root disease may be more significant threats to the survival of young white pine.

Local variation in rust incidence within the northern Lake States is recognized. Potential planting sites should be evaluated based on local microclimatic and topography risk factors previously described by Van Arsdel (1961b). Rust incidence is influenced by proximity of *Ribes* to white pine. Therefore, managers should avoid planting in areas that harbor abundant *Ribes* populations. Local eradication of *Ribes* in blister rust hazard zones 3 and 4 has been shown to be effective in reducing the incidence of blister rust and may be a viable management strategy.

Although pruning has been shown to be effective in reducing the incidence of blister rust (Hunt 1991; Lehrer 1982; Weber 1964), early results from this study cannot yet validate those results.

Prior to planting, managers should evaluate local herbivore populations, especially in the winter and early spring. Potential planting sites with high deer or hare populations should be avoided unless effective browse protection is provided. Slow growth rates and browse damage to trees on many of our sites have delayed tree pruning and may eventually increase their risk to infection by blister rust.

Krueger and Puettmann (in press) concluded that competition, especially from woody plants, can severely impact white pine establishment. Our results support those conclusions. Without adequate weed control, planting failures are likely, especially in clearcut areas and on mesic, nutrient rich sites.

Acknowledgments

We thank Mike Lanasa, Alan Saberniak, Dave Upgren, Phil Kinney, Linda Haugen, John Murphy, Dave Worel, Don Potter, and Don Peterson for their assistance in the establishment and maintenance of the study plots.

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Developing Proactive Management Options to Sustain Bristlecone and Limber Pine Ecosystems in the Presence of a Non-Native Pathogen

A. W. Schoettle[1]

***Abstract**—Limber pine and Rocky Mountain bristlecone pine are currently threatened by the non-native pathogen white pine blister rust (WPBR). Limber pine is experiencing mortality in the Northern Rocky Mountains and the infection front continues to move southward. The first report of WPBR on Rocky Mountain bristlecone pine was made in 2003 (Blodgett and Sullivan 2004), at a site that is more than 220 miles away from the former infection front. No mortality has been observed in this recently infected area but the species is highly susceptible. There are no ecological reasons to suspect that WPBR on bristlecone and the southern distribution of limber pine will not expand over time. Learning from experiences in impacted ecosystems will facilitate the development of proactive measures to mitigate impacts in these southern populations in the future. If no action is taken, and the pathogen takes its course, we risk losses of aesthetic landscapes; impacts to ecosystem boundaries, successional pathways, and watershed processes; and shifts from forested to treeless sites at some landscape positions. This paper introduces an interdisciplinary approach to developing proactive management options for limber and bristlecone pines in the southern Rocky Mountains. Managers, researchers, operational professionals and interested public groups will have to work together and share their knowledge and perspectives to sustain these ecosystems for future generations.*

Introduction

Limber pine (*Pinus flexilis* James) and Rocky Mountain bristlecone (*Pinus aristata* Engelm.) are white pines (subgenus *Strobus*) yet limber pine is in section *Strobus*, subsection *Strobi* and Rocky Mountain bristlecone pine is in subgenus *Parrya*, subsection *Balfournianae* (Lanner 1990). They both have 5-needles per fascicle. Limber pine and bristlecone pines can grow as erect trees, clusters of erect trees and as wind-sculpted wedge-shaped shrubs (krummholz). Limber pine has a very broad elevational distribution ranging from the grassland treeline to the alpine treeline as well as a broad latitudinal distribution from Canada southward into New Mexico (Schoettle and Rochelle 2000). Bristlecone has a narrower distribution, primarily occupying higher elevation sites in central and southern Colorado with a small distribution into North New Mexico and a peripheral population in the San Francisco Peaks of Arizona. Their often bushy growth form and slow growth rate combined with the inaccessibility of the rocky sites that they dominate make them poor timber species and ones that have long been overlooked by the forestry community. The most basic ecological information has not been quantified for these species (Schoettle 2004).

Limber pine and bristlecone pine are currently threatened by the non-native pathogen white pine blister rust (*Cronartium ribicola* J. C. Fisch.). The impact of white pine blister rust on commercial North American white

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pines has been a focus of attention since its introduction from Europe in the early 1900s. In the mid-1980s, the focus expanded to impacts of the disease to the non-commercial whitebark pine (*Pinus albicaulis* Engelm.) as forest practices shifted toward management of ecosystems. White pine blister rust's threat to whitebark pine and the resultant impacts to the habitat of the endangered grizzly bear (*Ursus arctos horribilis*) have brought whitebark pine ecosystems into view by the management and research community (e.g., Schmidt and McDonald 1990, Tomback and others 2001). Limber pine has been infected in the Northern Rocky Mountains for decades and the infection front continues to move southward; infections in Colorado were found in 1998 (Johnson and Jacobi 2000). White pine blister rust was first reported on Rocky Mountain bristlecone pine in 2003 (Blodgett and Sullivan 2004). This new infection site supports infected limber pine and bristlecone pine and appears disjunct from the more continuous infection front more than 200 miles to the north. The disease appears to have jumped over a near continuous corridor of limber pine from the infection front to the bristlecone/limber pine forests (figure 1). Introduction of the rust into the southern ecosystems may have occurred from infected nursery stock planted in the growing communities of the urban-wildland interface or long-distance transport of rust spores from California or other infected areas.

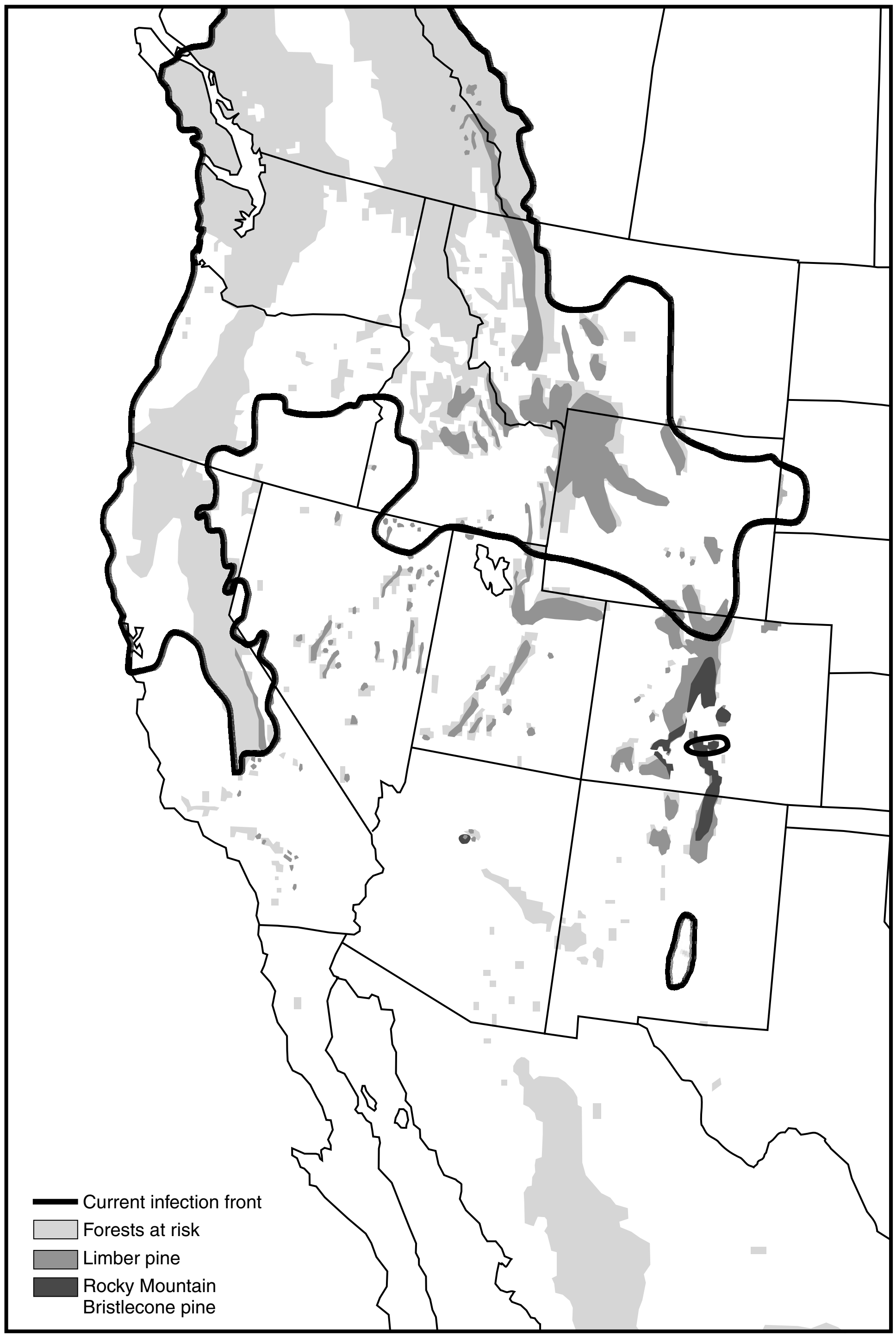

Figure 1—A map of the distribution of forest at risk in western North America for white pine blister rust impacts (light shading) and the distribution of limber pine (medium shading) and Rocky Mountain bristlecone pine (dark shading). Forests at risk are all those containing the 5-needle pine species that are susceptible to the rust. The dark line denotes the current white pine blister rust infection front; not all stands are infected within the lined areas but the rust has been documented on the pines in those areas.

Screening studies reveal that all of the North American 5-needle white pines are highly susceptible to the rust. It is estimated that less than 5 percent of the population of each species has any genetic resistance to the rust. Heavily infected areas have experienced almost complete mortality of the white pine component of the forest and the replacement forest communities are more prone to epidemics of native pests and pathogens. Despite significant efforts to contain the pathogen after its introduction in the early 1900s, this fungus continues to spread. Blister rust is now a permanent resident of North America affecting even the high elevation and drier forest ecosystems once thought to escape infection.

The early studies of rust susceptibility for RM bristlecone pine are confounded by taxonomic confusion associated with the species. The bristlecone pines throughout the western United States were thought to be one species (*P. aristata*) until 1970 when Bailey (1970) distinguished the populations into two species. The populations in Colorado, northern New Mexico and the isolated population in Arizona were designated Rocky Mountain bristlecone pine and retained the name of *P. aristata* and the populations in Utah, Nevada and California were called Great Basin bristlecone pine and newly named *P. longaeva*. The results from the rust-screening studies of Hoff and others (1980) and Bingham (1972) are confounded by the combining of seed collections from the two bristlecone pine species. Only the Childs and Bedwell (1948) study explicitly sampled a Colorado population (RM bristlecone pine) and shows that while this species is susceptible to the rust, it appears to have slighter greater resistance than western white pine or sugar pine. Although these early studies were conducted with bulk seed lots, they suggest that sustaining bristlecone pine forests through management may be more successful than other species.

Taking advantage of learning from experiences in impacted ecosystems and using the time to develop and instigate proactive measures to help prepare the bristlecone pine and southern limber pine ecosystems for the pathogen provide the opportunity to attempt to mitigate impacts in the future. This paper will discuss the reasons for developing information and management options now, even in these early stages of pathogen infection, and introduce an interdisciplinary approach to developing a proactive management strategy for limber pine and Rocky Mountain bristlecone pine in the central Rocky Mountains.

The White Pine Blister Rust Threat: The Situation

White pine blister rust has a complex life cycle that requires two obligate hosts: the 5-needle white pine and the currant or gooseberry species (*Ribes* ssp.) (figure 2). Infection of the *Ribes* occurs in the spring through wind transport of aeciospores released from cankers on the pines. Several spore stages are completed on the *Ribes* leaves until finally the basidiospores are released in late summer or early fall. The fungus is confined to the leaves of the *Ribes* plants where it completes it life stages. *Ribes* are deciduous and shed their leaves and the fungal infection each year. As a result the fungus does not cause mortality of the *Ribes* plant. In contrast, once infected the fungus is persistent, perennial, and invasive within the pine. The white pine blister rust basidiospores enter pine needles through stomatal openings and the fungus grows into the twig (McDonald and Hoff 2001). Aecia (which

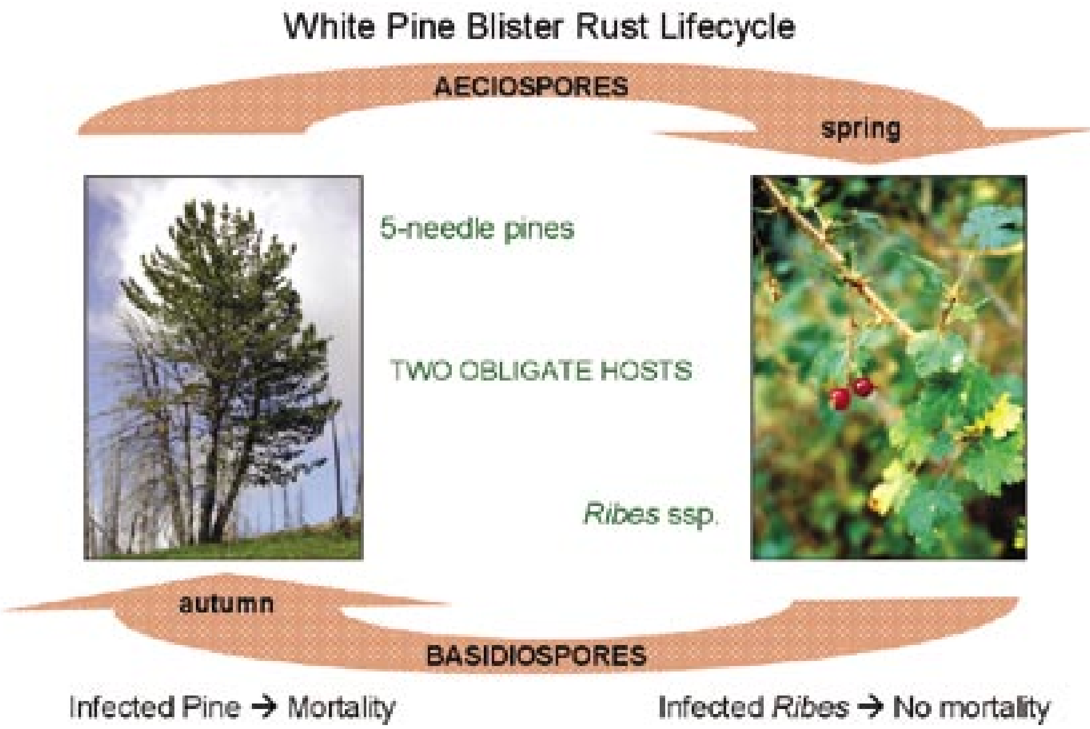

Figure 2—Simplified schematic of the life cycle of white pine blister rust.

release the aeciospores) erupt through the bark of the twig and form the tell-tail canker. The fungus continues to grow into the branch and ultimately the main stem of the tree. Cankers girdle the infected branch or stem killing the distal tissue. Branch cankers often will not kill the tree until the reduction in leaf area is so great that the tree cannot survive or the canker grows to affect the main stem. The contribution of rust-caused branch mortality to reduced cone production and an increase in sensitivity of the tree to other stresses such as drought, competition, and bark beetle attacks deserves research attention to fully assess the impacts of the disease. Cankers on the main stem of a tree cause top-kill and will usually kill the individual. White pine blister rust exerts strong selective pressure at the seedling-sapling stage and can cause high rates of seedling mortality within several years of infection. Very old trees that have significant partial cambial dieback, such that all of the tree's surviving foliage is supported on a few branches, may be rapidly killed by white pine blister rust once infected (Schoettle 2004).

White pine blister rust has its own set of environmental constraints as influenced by the tolerances of its biology as well as the distribution of its two hosts, the five-needle white pines and *Ribes* ssp. The degree of overlap between the rust's potential habitat with that of limber pine and bristlecone pine's distributions has not been fully defined. While the selective pressure exerted by the rust on these five-needle pines will not be uniform across their distribution, existing information on *Ribes* distributions suggests that it may be extensive; three-fourths of the limber pine sites sampled along the elevation gradient of Colorado's Front Range contained *Ribes* ssp. (8 of 12 stands; Schoettle and Rochelle 2000) and more than half of the bristlecone pine sites evaluated by Ranne and others (1997) contained *Ribes* ssp. (27 of 50 stands). Long-range transport of rust spores may be possible (Mielke 1943) but may not be necessary for the rust to spread through these ecosystems. The suitability of different *Ribes* species to host the rust varies (Van Arsdel and others 1998) but unfortunately, those species that support good rust spore production are present throughout the range of both bristlecone and limber pine (Kearns and others, in press).

Consequences of Non-intervention

Bristlecone and limber pine ecosystems are unique and valued. The effects of blister rust-caused mortality in these systems will be greater than the loss of the individual trees. In addition to their ecological roles, which will be discussed below, these species are appreciated by people for their artistic forms and extreme longevity (e.g., Cohen 1998). Bristlecone pine and limber pine are often used as symbols of perseverance and tolerance. In central Colorado, over 100,000 people a year pay an entrance fee to visit an ancient bristlecone pine forest in a Research National Area. Also, because these species occupy ridge tops they are often the species that surround forest visitors as they enjoy the mountain vistas at their hike's destination. The loss of these species to a non-native pathogen would be a national loss.

Ecologically, bristlecone and limber pines species play critical roles in maintaining the resilience and integrity of many Rocky Mountain ecosystems. Wildlife relies on these species for food. Limber pine has large wingless (or near wingless) seeds and has a mutualistic relationship with corvid species (e.g., Clark's nutcracker, *Nucifraga columbiana* Wilson) such that the corvids feed on the seed and serve to disperse the seeds (Lanner and Vander Wall 1980, Tomback and Kramer 1980). As for whitebark pine, seeds of limber pine are also an important food source for black and grizzly bears (*Ursus* ssp.; Kendell 1983, McCutchen 1996), red squirrels (*Tamaisciurus hudsonicus*; Hutchins and Lanner 1982) and other small rodents. For the grizzly bear it is known that during years of low whitebark pine seed production the fecundity of bears is reduced and they depend more heavily on limber pine seed to fulfill their nutritional needs. Low limber pine seed production likely affects squirrel populations and the carnivore species that depend on them, including possibly the Canada lynx (*Lynx canadensis*).

Bristlecone and limber pines have a suite of structural and physiological traits that enable them to be very stress tolerant and occupy sites that other species cannot (Schoettle 2004). Mortality caused by the rust in these harsh sites will transition these forested sites to treeless areas affecting slope stability, snow retention and watershed hydrology. While these rocky ridges are the most obvious habitat occupied by bristlecone and limber pine, scattered occurrence of these species throughout the high-elevation forested region of the Colorado is typical (Schoettle 2004). On these more mesic sites, limber pine's early post-disturbance dominance succeeds over time to other conifer species (Rebertus and others 1991). Limber pine acts as a nurse tree, mitigating the harsh open environment after disturbances and facilitating the establishment of Engelmann spruce and subalpine fir in the subalpine (Rebertus and others 1991, Donnegan and Rebertus 1999) and of Douglas-fir at the lower treeline (Baumeister 2002). Engelmann spruce and subalpine fir are able to become established in the lee of bristlecone pine at the alpine treeline at elevations where they cannot become established alone (personal observations). The loss of limber and bristlecone pines in these more mesic areas would alter successional trajectories and future forest composition.

The ecological trade-off of the traits that confer stress tolerance is slow growth and poor competitive ability. These species have long tree and leaf lifespans (see Schoettle 1994). Both limber and bristlecone pine have delayed reproduction such that it takes more than 50 years for a seedling to mature to become cone-bearing. As a result, after a disturbance, there is a long lag before the reforested site is ecologically functional with respect to seed production and the species and processes that depend on them. This posed a compelling reason for attempting to establish rust-resistant seedlings of these species as soon as possible to minimize this lag period and accelerate the natural production and dispersal of rust-resistant seeds.

The effects of white pine blister rust on five-needle pines will interact with the changing fire regimes in the Rocky Mountains. As fire regimes get more frequent and unpredictable due to past fire suppression and forest practices, large wildfires may jeopardize the usually less-flammable five-needle pine ecosystems on dry sites. In addition, branch and tree mortality caused by white pine blister rust may contribute to fuel loading in white pine stands, increasing the susceptibility of these stands to sustain and be consumed by fire. In the event of larger fires, especially those covering a larger area than cannot be seeded effectively by wind dispersal mechanisms, the loss of bird-dispersed pines as colonizers may be especially pronounced.

In summary, these species and their ecosystems provide aesthetic and spiritual experiences for forest visitors and diet and habitat for wildlife. In addition, they have unique structural and physiological traits that lead to unique ecological functions on the landscape including post-fire recovery and facilitating succession. These species are, however, very slow growing and the ecosystems are slow to recover after disturbance. The mortality and reduced cone production caused by the non-native pathogen white pine blister rust will further slow the post-disturbance recovery of these ecosystems. Observations of effects caused by this pathogen in the northern Rockies shows that the impacts can be devastating and far reaching (Tomback and others 2001). Learning from experiences in other ecosystems and initiating proactive measures provides the opportunity to help sustain these ecosystems during their pending persistent assault by white pine blister rust. There is no ecological reason to suspect that WPBR won't continue to spread through bristlecone and limber pine ecosystems in the southern Rocky Mountains. If no action is taken, and the pathogen takes its course, we risk losses of aesthetic landscapes, impacts to ecosystem boundaries and successional pathways, and shifts from forested to treeless sites at some landscape positions causing changes in slope stability and watershed hydrology.

Developing Management Options

The Need for an Interdisciplinary Approach

Development of strategies for sustaining limber pine and bristlecone pine ecosystems in the presence of the non-native pathogen requires an interdisciplinary approach (figure 3). Information is very limited for these non-timber species and ecosystems. Information and integration is needed in the areas of (1) pathology, including etiology and epidemiology, (2) genetics of both hosts and the genetics of resistance mechanisms, and (3) ecology of both hosts and the fungus as well as the interactive effects of the disease on ecosystem function. The integration of existing information and the gathering of new information in each of these areas will help the development of management options to sustain white pine ecosystem function and maintain the species' existing distribution. The goal of this effort is to provide

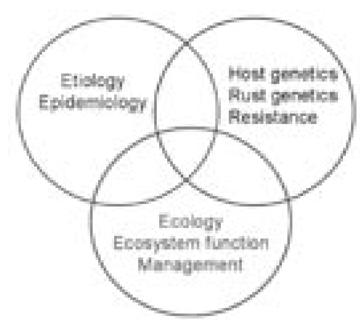

Figure 3—The need for an interdisciplinary approach in developing management options to sustain white pine ecosystem function in the presence of the non-native pathogen white pine blister rust.

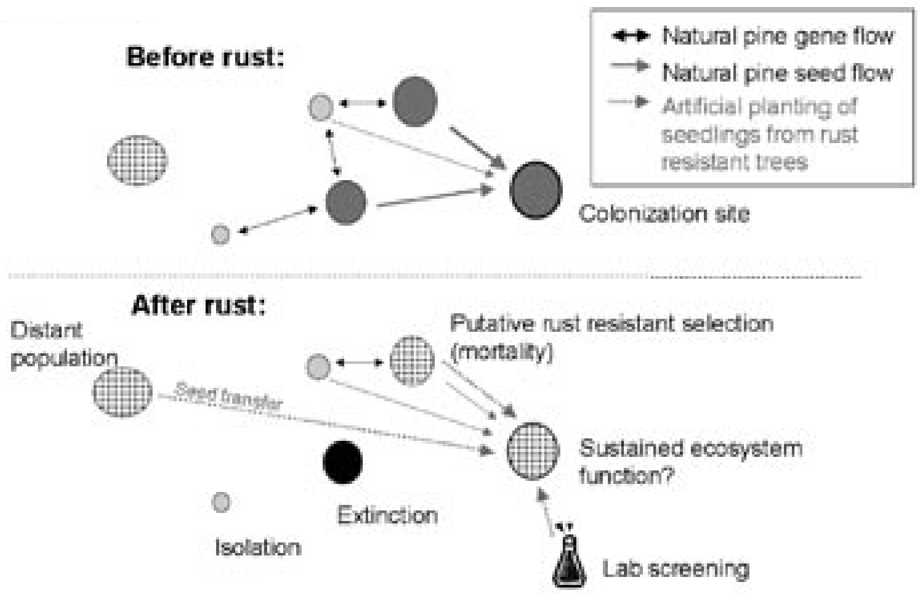

Figure 4—Schematic of potential effects of white pine blister rust on limber pine and bristlecone pine populations. The rust may cause extinction of some stands and isolation of others. After the rust has impacted an area, creating a colonization site may promote establishment of rust-resistant individuals if seeds are available. Artificial seed transfer, via outplanting of rust resistant seedlings, may accelerate the establishment of rust-resistant seedlings in the colonization site. Creating small regeneration sites before the rust arrives will result in young stands on the landscape for efficient rust-resistance selection upon invasion.

managers with the ability to (1) create regeneration opportunities for limber and bristlecone pines, (2) accelerate the establishment of rust resistant individuals, (3) prioritize stands for management intervention and assess some management options for those stands, and (4) sustain the functioning and resilience of these forest ecosystems for the future.

The goal is to accelerate the establishment of white pine blister rust resistant genotypes of limber pine and bristlecone pine across the landscape. Exploring the use of natural processes in addition to tree-planting approaches deserves attention with these species (figure 4). Protecting seed source stands and creating nearby regeneration opportunities to provide for rapid selection of rust-resistant genotypes in the presence of the rust may be an option. The susceptibility of trees to rust infection is not constant with age; young susceptible trees are killed rapidly by the rust while some older trees appear to develop resistance over time (ontogenetic resistance). Ontogenetic resistance can be significant in sugar pine yet the degree this occurs in bristlecone and limber pine is not known. If present, the older stand may be less impacted by the rust yet the progeny of those ontogenetically resistant trees are susceptible to infection by the rust. Therefore in the event of a disturbance such as a fire these older stands may not ensure the recovery of the area in the presence of the pathogen. Because ontogenetic resistance does not contribute to true genetic rust resistance it may serve to retain trees on the landscape for a generation but does not ensure future landscape sustainability over longer time scales. As a result, to accelerate the establishment of rust-resistant seedlings, it will be important to provide an opportunity in portions of the landscape for rust-resistant selection of reproductive seed trees and regeneration of their progeny. Creating a mosaic of mixed age classes and regeneration opportunities across the landscape before the pathogen is present may retain bristlecone and limber pine attributes in the area while rust-resistant selection occurs rapidly in the young stands and slowly in the older stands. This approach may sustain the present and future resiliency of the ecosystem in the presence of the pathogen.

Alternatively, selecting rust resistant genotypes through research studies and outplanting is another approach. Identifying resistant individuals can be done, as has been done for other white pines, by field assessment in areas already challenged by white pine blister rust or by screening seedlings in nursery trials with artificial inoculations. This more active management approach also requires sufficient knowledge to generate seed transfer guidelines to avoid outplanting resistant individuals that are maladapted to the site. This process will take time. A combination of approaches may be useful: prepare the landscape before infection by creating a diverse age class structure; promote natural regeneration from resistant trees after infection; and augment, if needed, with artificial regeneration of selected genotypes.

Proposed Strategy

Achieving the integrated interdisciplinary approach to sustain white pine ecosystems requires the cooperation of diverse partners and expertise. Developing the necessary knowledge to create regeneration opportunities to accelerate the selection for and establishment of a rust-resistant population will require information on the colonization dynamics of both the pines and *Ribes*, the geographic pattern of local adaptation of both hosts and the pathogen, and the identification of rust-resistance mechanisms and their distributions in the pine populations (figure 5).

Programs within Region 2 Forest Health Management, Colorado State University, and Rocky Mountain Research Station have begun to tackle this problem. Ongoing studies in the area of geographic patterns of local adaptation suggest that local differentiation among bristlecone pine populations is sufficient to warrant the definition of seed transfer zones. Studies have also begun to screen bristlecone for rust-resistance to identify possible resistant individuals and assess the possibility of differential distribution of resistance among populations. Extensive monitoring of rust infection, meteorological conditions, and host distributions are being used to generate rust hazard models for southern Wyoming and Colorado. Studies of the regeneration dynamics of bristlecone and limber pine show that they establish well after

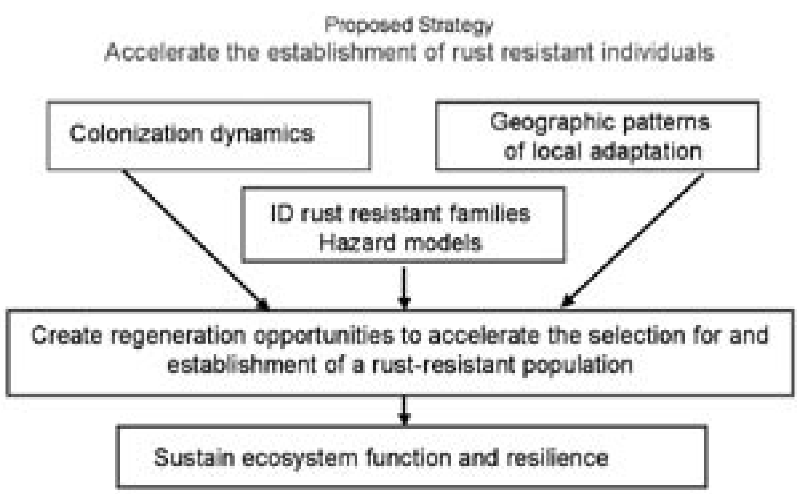

Figure 5—Schematic of a strategy to develop management options to sustain white pine ecosystem function in the presence of the non-native pathogen white pine blister rust.

fire and are able to colonize the interior of large burned areas. However, *Ribes* densities are greatly increased after fire (Schoettle 2003), elevating the risk of rust in the area. Therefore fire can be used to generate colonization sites for bristlecone and limber pines but prior to its use in an area one should consider whether *Ribes* is also likely to proliferate.

Ongoing Needs

The program to develop restoration options for high-elevation white pines will take time and time is running short. Gathering, integrating, and synthesizing information is critical for the development of management options in a timely manner to help sustain bristlecone and limber pine ecosystem function and resilience. Increasing awareness of the threat to these valued ecosystems will stimulate work to fill the information gaps. In addition, education to encourage recognition of the hosts and the symptoms of the disease will facilitate efforts to learn the extent of the disease and to restrict the transplanting of infected horticultural stock. Other ecosystems that have been affected by the non-native rust for longer periods offer learning opportunities. While information may not be directly transferable among ecosystems, insights from past experiences in other systems regarding what restoration treatments might work in bristlecone and limber pine ecosystems may be valuable. Information from the uninfected ecosystems can provide baselines to help managers in infected areas better assess the effectiveness of restoration treatments in their areas. Finally, managers, researchers, operational professionals and interested public groups must work together and share their knowledge and perspectives to develop and implement effective management options to sustain and restore these ecosystems for future generations.

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Long-Term Sustainability in Davis Late Successional Reserve

Jim Stone[1] and Joan Kittrell[2]

***Abstract**—A pilot watershed analysis along with the Davis Late Successional Reserve analysis in the range of the northern spotted owl identified potential loss of suitable habitat, reduction in the numbers of large trees, and lack of replacements for large trees. A variety of silvicultural thinning techniques has been used to address these important issues and to move toward a sustainable mix of stand conditions on the landscape.*

Introduction

In 1993 President Clinton ordered that a resolution be found to the gridlock between the needs of wildlife associated with late- and old-structured stands (LOS) and the need to thin timbered stands in the Pacific Northwest and northern California areas. He put together a commission, which met in April 1993, to reach consensus on how to conserve the wildlife species that require Late and Old Structure (LOS) within the range of the northern spotted owl. It also purposed to provide a steady flow of timber products from the same landscapes. This resulted in the Northwest Forest Plan (NWFP), or the President's plan as it is commonly referred to, signed April 13, 1994.

The NWFP applied to the whole range of the northern spotted owl. The bulk of the range of the northern spotted owl is west of the crest of the Cascade Mountains in California, Oregon, and Washington. Portions of the range extend to the east side of the Cascades. In the Pacific Northwest, this area is commonly referred to as the east side.

As a result of the decision from the NWFP, a network of late successional reserves (LSR) was identified to provide for old growth dependent species. Davis LSR is one of the east side LSRs nestled mostly in Odell Creek and Moore Creek subwatersheds (figure 1). Along with accompanying standards and guidelines, watershed level analysis at the sixth level watersheds was initiated in certain key areas called pilot areas.

One of the 15 key areas selected as a pilot watershed to assess was the Odell subwatershed on the Crescent Ranger District. This subwatershed grouped the Moore Creek, the Odell Lake, and the Odell Creek subwatersheds for a total assessment area of 74,933 acres. The Odell Pilot Watershed Assessment identified key processes and flows in and through the area in the physical, biological, and social realms. Vegetative condition was common to most of the processes and flows in the subwatershed. From the Odell Pilot analysis, certain items were chosen as key items of concern. One of the key items of concern was the loss of large trees and the lack of replacements for the large trees in the area. Most of the mixed conifer stands currently have overstories of fire-resistant species, primarily ponderosa pine and Douglas-fir.

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[2] Crescent Ranger District, Crescent, OR.

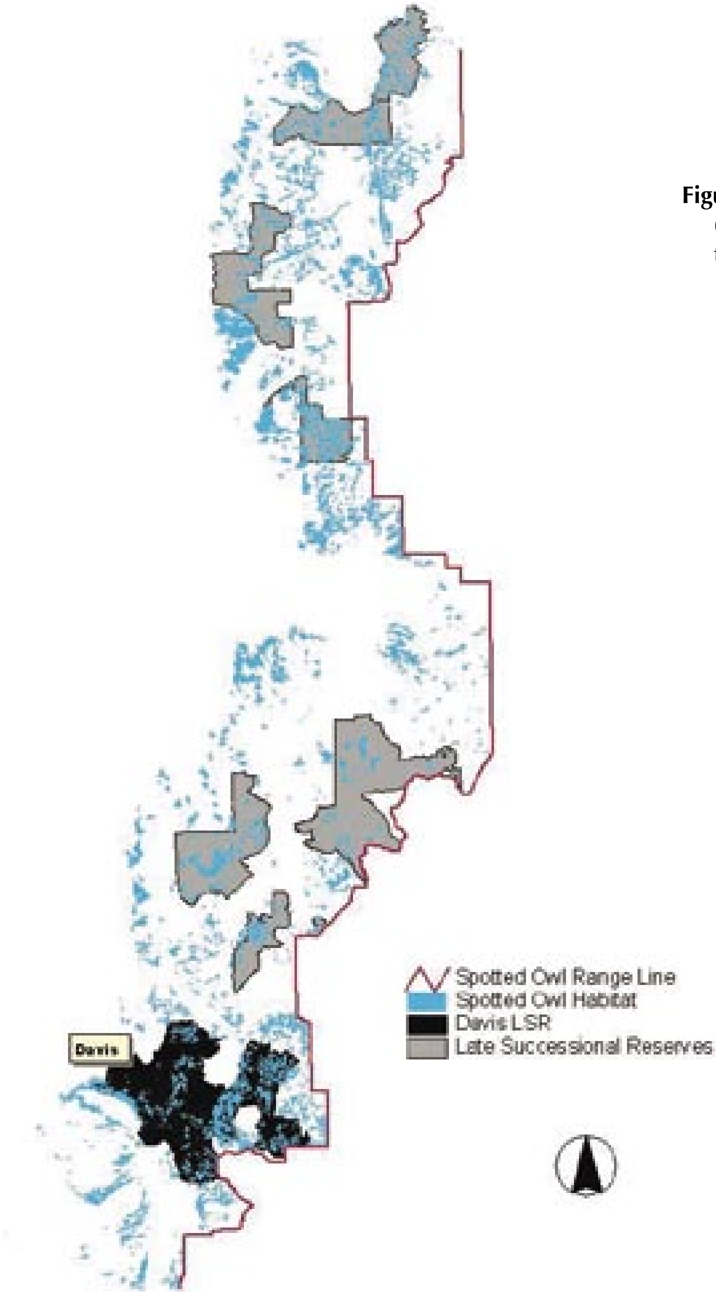

Figure 1—Distribution of Late Successional Reserves (LSRs) and nesting, roosting, and foraging habitat on the Deschutes National Forest.

Understories are comprised mostly of the true fir species and lodgepole pine, so dense that ponderosa pine and Douglas-fir regeneration are few to non-existent and very poor in form, unable to respond to release.

Teamwork

As the issues became clear, time was taken for members of the interdisciplinary team (IDT) to educate each other on the issues, concerns, and desires of their discipline. This was called team teaching. Being able to understand and speak each other's "language" proved to be very effective in communicating with other agencies, administrators, and the general public. Each team member could clearly describe the conditions, processes, flows, and needs of any of the other disciplines. It was common on "show-me" trips to see the wildlife biologist explain the need for silvicultural treatments, the silviculturist to explain the need to retain or provide a particular habitat structure, and/or a fuels specialist to explain the need for certain snags or down woody material for particular wildlife needs.

The cohesiveness of the group and the trust built through this process carried over to implementation of the project. This was key in the success of the project and providing continuity when integrating new people into the project, as all but one of the team members moved away.

Special Place

The 48,890-acre Davis Late Successional Reserve (LSR) (figure 2), one of the largest of the LSRs in the system, was analyzed for its ability to continue functioning as an ecosystem dominated by late and old seral structural conditions. The LSR lies mostly within the Odell Pilot subwatershed, spans seven buttes, and skirts the shores of Davis Lake. It includes lodgepole pine in the flat basins, ponderosa pine at the base of the buttes, mixed conifer stands on the midslopes of the buttes, and mountain hemlock at the tops. Snags and down wood are common from endemic levels of insects, disease, and competition-induced mortality. The elements of the LSR that make it suitable for the northern spotted owl and bald eagles are the multi-storied mixed conifer stands.

The lodgepole and ponderosa pine stands are not suitable as habitat for the northern spotted owl, but each has groups of species associated with its old-structured stands. Stands are, however, used for dispersal by the owls, since the mixed conifer stands are bisected by the other stands across the landscape.

Other animals including deer, elk, coyotes, neotropical migratory birds, owls, woodpeckers, and hawks take advantage of the diversity of habitats.

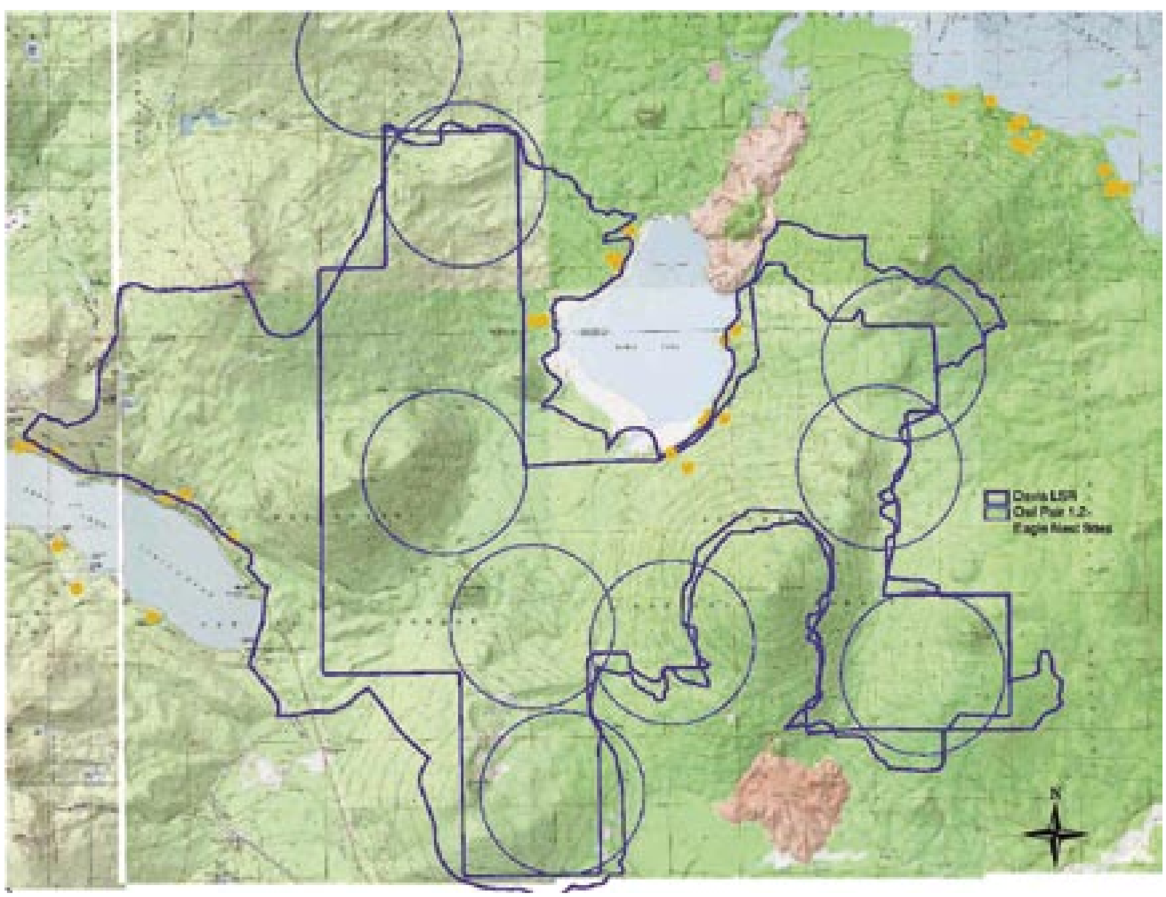

Figure 2—Davis Late Successional Reserve with owl home ranges and eagle use areas.

Home Sweet Home

Davis LSR owl habitat is important to maintain over time. Like other LSRs on the east side, Davis LSR is on the edge of the range of the northern spotted owl. The owls resident to this area are considered important to the genetic base because of their "pioneering" nature; i.e., they go where no owl has gone before and *survived*. This is considered to be an important trait for the survival of the species.

Spotted owls generally require mature or old-growth coniferous forest with complex structure including multiple canopy layers, large green trees and snags, heavy canopy habitat, and coarse woody material on the forest floor. A wide variety of forest types are utilized. Nesting, roosting, and foraging (NRF) habitat for the northern spotted owl on the Deschutes NF includes stands of mixed conifer, ponderosa pine with white fir understory, and mountain hemlock with subalpine fir. Suitable nest sites are generally in cavities in the boles of either dead or live trees of at least 30 inches DBH. Platform nests such as abandoned raptor nests, broken treetops, mistletoe brooms, and squirrel nests are used on rare occasions. Relatively heavy canopy habitat with a semi-open understory is essential for effective hunting and secure movement. Habitat conditions that support good populations of northern flying squirrels (*Glaucomys sabrinus*), western red-backed voles (*Clethrionomys californicus*), and other nocturnal or crepuscular small mammals, birds, and insects are essential to supporting spotted owls.

Edge effects from large forest openings may adversely impact the micro-habitat conditions necessary for suitable owl habitat as well as contribute to increasing the risk to spotted owls imposed by predators or to competition from the barred owl (*Strix varia*). Spotted owls will use younger, managed forests that have key habitat components available. These younger forests provide dispersal habitat for owls and foraging habitat if near nesting or roosting areas.

Hence, complexity of stand structure plays a very important role as habitat for the northern spotted owl. This is also a factor for the bald eagle habitat in the LSR. The eagles use the dense, multi-storied stands for winter roosting. The big, open-grown ponderosa pine and Douglas-fir trees with forks in the tops are the preferred sites for nest building. The understory development since fire exclusion has created serious competition and ladder fuel conditions.

Danger in Paradise

Such complex stand structure in east side forests is a condition that is transitory, at best. The historic condition of these stands seems to clearly have been mostly open park-like stands dominated by fire-resistant species such as the pines and Douglas-fir. Ponderosa pine and, to a lesser extent, Douglas-fir, dominate and are the oldest of the overstory trees in the LSR. Fire frequencies ranged from five to 30 years on most of the ponderosa and mixed conifer stands. Most all of the understory trees have come in since the time of fire suppression, about the end of the 18th century. Records of early recreationists and settlers in the area indicate they would often fight fires that were a threat to their campsites and homes (personal communication, Leroy Steece, Archeology Tech).

Since that time, fire intolerant species (lodgepole pine and true firs) have developed in the understories at such densities that competition-associated mortality of the overstory trees is increasing rapidly at a rate that exceeds replacement of those trees. Personal observations of old and newer stumps in the area indicate that the first 8-15 decades of the overstory trees were characterized by rapid growth, indicating somewhat open-grown conditions. These growth rates certainly exceed those of any overstory replacement trees presently growing in untreated stands. Thus, conditions favorable for species such as the northern spotted owl and its prey were probably not common on the east slopes of the Cascades in much of Central Oregon until the last half century or less. Now, with those conditions common on the landscape, the concern for large tree retention and for stand replacement fire events is high.

Building on the foundation of the Odell Pilot WA, one of the key concerns in the analysis of the LSR was the long-term sustainability of the late-successional stands. Multi-storied stands of the early seral ponderosa pine and Douglas-fir overstories with dense true fir and lodgepole pine understories are highly susceptible to insects, disease and/or fire events at stand replacement intensities. These stands developed because of fire suppression in the area for the last century. This, combined with limited management activities to control densities, has resulted in vegetative structural conditions that probably never previously occurred at this level on the landscape.

Risk Reduction or Habitat on a Timeline

The 7 Buttes and 7 Buttes Return analyses were initiated to try to address the vegetative condition concerns raised in the Odell Pilot and Davis LSR assessments. The purpose and need focused on the key vegetative issues and how to approach them from a silvicultural perspective. A variety of silvicultural approaches were proposed and planned for the landscape. They included:

1. Regeneration harvest (HSH), 32 acres, leaving an overstory in a shelterwood density.
2. Salvage (HSV) or (HSL-SV) of lodgepole pine, 1149 acres, removal of excess dead from stands and/or selective removal of lodgepole understory.
3. Light thinning (HTH9M or S) would reduce the post-harvest basal area to about 90 percent of the upper management zone (UMZ) and maintain a multi-storied stand on 7509 acres or a single-storied stand on 613 acres. These treatments usually include precommercial thinning (PCT) to reach target basal area. The ponderosa pine plant association groups or mixed conifer dry plant association groups with ponderosa pine as the dominant species are targeted for single-storied stands.
4. Moderate thinning (HTH5) would reduce the basal area to within 50 percent to 80 percent of the UMZ and maintain a multi-storied stand on 226 acres and single-storied stand on 163 acres. These treatments usually include precommercial thinning to reach target basal area. The ponderosa pine plant association groups or mixed conifer dry plant association groups with ponderosa pine as the dominant species are targeted for single-storied stands.
5. Single-tree selection (HSL) in three lodgepole pine stands (272 acres), to promote development of larger trees with full crowns.
6. Precommercial thinning on 10,000 acres.

The intent of treatments was to "set the clock back" on the successional processes going on in the area. This was to be done by retaining most or all of the large trees while thinning out the later successional understory species and the lodgepole pine. The goal was to get the stands to 90 percent of upper management zone (UMZ) or less, as discussed in Cochran's paper (1992). The strategy was to have a commercial entry to generate funds that could at least partially offset the costs of thinning the noncommercial components of the stands. Generally speaking, in the mixed conifer stands, if precommercial thinning was not completed, the stands were not effectively

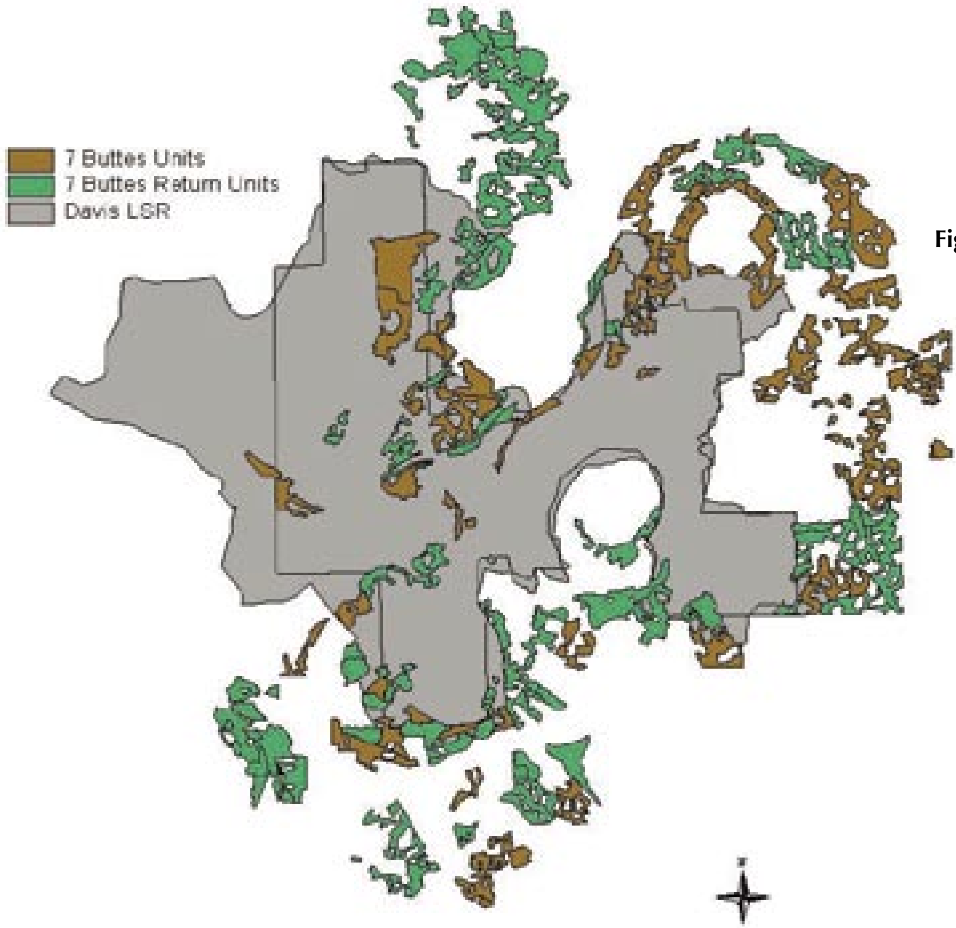

Figure 3—7 Buttes and 7 Buttes Return vegetative treatment units. The shaded area is the Davis LSR.

treated to the stated objectives of increased resistance to insects, disease, and fire and of large tree retention on the landscape.

Most of the stands proposed for treatment with timber sales were considered suitable as nesting, roosting, and foraging habitat, or NRF. Initial evaluations of NRF did not include trees less than 8 inches DBH as part of the habitat structure. However, as precommercial-thinning projects began to emerge, the importance of this component surfaced causing serious concern among biologists. Precommercial thinning to a level appropriate for long-term sustainability of the large tree components clearly removed the stands from NRF status. Any lesser treatment was not considered adequate to meet the long-term goals for the stands. This dilemma prompted field reviews and close coordination with USFWS in order to develop some possible treatment approaches that could allow both effective treatment and retention of NRF at acceptable levels on the landscape. The bottom line is that NRF is not a sustainable condition in these stands.

Nuts and Bolts

The silvicultural treatment objectives within the LSR are based on habitat conditions rather than the traditional production and yield. They include:

- Increase resistance to insects, disease, and wildfire stand replacement events.
- Maintain and/or enhance large tree retention and growth.
- Utilize down dead material in excess of that needed to meet wildlife guidelines.

Treatment specifics to meet these guidelines include reduction of density to below UMZ using basal area as a surrogate for leaf area estimations. Using stand examination information, the stand density index (SDI) was calculated for each of the stands as outlined by Cochran (1992). The "allocations" of SDI varied between even-aged and uneven-aged stands and were averaged among 10-inch diameter classes for uneven-aged stands. In the even-aged stands, overstory replacement trees were allowed for by stratifying the stands to precommercial size, less than 21 inches and greater than 21 inches trees.

Marking instructions were simplified based on site visits by the silviculturist and interpretation of the SDI information. On many of the stands, both the silviculturist and the wildlife biologist worked with the marking crew to help them apply treatment objectives to their marking. The silviculturist completed random monitoring of the marking and the results were shared with the marking crew in a timely manner.

Harvest activity yielded an average of 6.3 CCF per acre (3.25 MBF per acre) from an average of 35 trees per acre removed from the mixed conifer stands. Most of the volume removed came from the true firs (79 percent) and lodgepole pine (10 percent), with incidental volume from ponderosa pine and Douglas-fir. An average one-third of the basal area per acre was removed and about 26 percent of the merchantable stems was removed. Most stands had from 1000 to 3000 sub-merchantable stems per acre. Precommercial thinning dropped those numbers to 190-270 stems per acre or less.

A variety of precommercial treatments were proposed to try to address the need to keep multi-storied stands. Guidelines developed for NRF habitat in the Davis LSR include 274 trees per acre (12.5-foot spacing) of seedlings, saplings, and small poles. This is a density that makes treating the fuels very difficult. Since these stands are historically fire associated, the goal is to reintroduce fire wherever it is appropriate, based on landscape arrangement of treatments and vegetative conditions. But fuel loadings are so high at present that an intermediate treatment is needed prior to use of prescribed fire. So most of the stands are machine or hand piled and the piles burned with this harvest entry. Then, with the next harvest entry, enough of the true fir and lodgepole pine will be removed that fire can successfully be reintroduced with underburning techniques.

Precommercially thinning to 274 trees per acre (TPA) as a guideline, then, was recognized as a serious concern and alternatives were sought to mitigate the problems. A variety of treatment alternatives were developed, recognizing each alternative would have a different effect on a given site, but would bring the desired conditions across the landscape. In summary, these treatments are:

Prescription 1—Maintain the existing specifications as stated (274 TPA) but use an average 13-foot spacing between leave trees with an average 260 TPA of seedling/sapling and pole size trees. This would leave the approximate numbers of trees but would still include many trees incapable of responding to release in some stands.

Prescription 2—Leave an average 260 TPA of the healthiest understory regardless of size or species and that are likely to respond to thinning. Spacing would not be a very significant factor with this entry, as long as at least 260 trees remained on site.

Prescription 3—Reduce the 274 TPA to approximately 190 TPA averaging 15-foot spacing to accommodate grapple piling machinery to treat slash accumulations, leaving the healthiest trees possible while meeting the spacing requirements.

Prescription 4—Leave portions of units unthinned and portions thinned with only healthy trees remaining and with a minimum spacing specified.

Prescription 5—Thin leaving only healthy trees that meet standards known to effectively respond to thinning with increased growth and vigor. This differs from option 5 in that no unthinned portions of the units would be left.

A combination of these treatments were applied on two of the sale areas and reviewed by the ID team and US Fish and Wildlife Service personnel and the PAC (Provincial Advisory Committee). It was agreed that using a mix of these treatments across the landscape was probably the best way to mitigate the needs to maintain the desired conditions on the landscape. This would provide a mix of treated and untreated fuels in stands arranged in such a way that different conditions were scattered across the landscape with no single condition contiguous across a large area. This helps meet fuels objectives for suppressing wildfires and meets wildlife objectives of a diversity of structures across the landscape.

Sales successfully moved forward with the various treatments. A second analysis, 7 Buttes Return, was completed to treat additional stands at risk using the lessons learned from the original 7 Buttes. Once all sales and post sale work was completed, 18,000 acres or about 40 percent of the LSR would have fuels reduced, large trees with improved health, and replacements well on their way.

Epilogue

On June 28, 2003, a fire started in the LSR and quickly grew to about 1200 acres by nightfall. The next day, weather conditions were such that as the wind picked up, the fire grew to almost 20,000 acres in about four hours. This resulted in a stand replacement event on about 24 percent of the LSR. Fire conditions were such that the fire spread as a wind-driven plume-dominated event with a plume reaching over 20,000 feet high and spotting ½ to 1 mile ahead of the leading edge. The intensity of the fire was such that very severe fire whorls leveled much of the interior of the fire, burned overstory trees to the point that few or no limbs are evident, and totally consumed all dead material, standing and down.

Interestingly, some of the units that were treated were used successfully as burnout areas effectively stopping the fire. It is not clear, with the weather conditions being what they were, if the outcome would have been any different were more of the area treated as prescribed. In this case, the outcome probably would have been very similar, since all proposed treatment areas were located down wind from where the fire developed its head of steam. However, had the fuels been treated at the ignition source (a very large area of down dead lodgepole pine) the initial attack might have been effective and the large fire may not have happened. The lodgepole pine area at the ignition source was slated for fuels treatment sometime in the next two to three years.

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Development of a Protocol for the Ecological Assessment of a Special Species

David Burton[1]

Abstract—*Developing consistent inventory and assessment protocols is important to people working on aspen issues in California and Nevada. Efforts have focused on identifying key indicators of ecological condition within aspen stands. The protocols have incorporated a range of factors that create or affect those indicators. Resulting ecological assessments conducted through the protocols describe stand structure, indicate unique stand management conditions, and record factors that might be putting stands at risk. Protocols for determining ecological condition were developed and field-tested by review groups in 2002. To date, units from seven state and federal agencies have collected data using the same protocols and field form.*

Overview

There has been heightened interest in aspen issues in the Far West as governmental mandates like the Sierra Nevada Forest Plan Amendment (U.S. Forest Service 2001) have increasingly directed land managers to protect the biodiversity of native flora and fauna. Many sub-units of management agencies have been inventorying, assessing, and treating aspen stands throughout their range, but prior to 2002 these efforts were not coordinated (U.S. Forest Service, Region 5, Aspen Survey Summary, May 2002).

The Aspen Delineation Project—a collaborative effort of the U.S. Forest Service's Pacific Southwest Region, California Department of Fish and Game, and the California Office of Bureau of Land Management—has began to develop a consistent approach for collecting data about ecological condition of aspen stands on agency lands. During workshops and field seminars co-sponsored by the Aspen Delineation Project, resource managers from multiple agencies and scientific disciplines identified the importance of developing and integrating consistent inventory, assessment, monitoring, and research protocols. The overriding objectives of this package of protocols were threefold: (1) to improve the scientific basis of management decisions, (2) to share data more easily within and between agencies, and (3) to produce more consistent evaluation of management practices, thus resulting in more effective adaptive management.

Because there was no consistency in how agencies were collecting data on location and condition of aspen stands, agency biologists and ecologists decided that the Aspen Delineation Project should initially develop a standardized protocol for (1) reconnaissance of the location of aspen stands and (2) assessment of a stand's ecological condition. Participants working on this project recognized that they would have to work fast due to time and budgeting constraints. It was agreed that the information gained using this protocol could lay the groundwork for establishing specific management

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objectives later on and be used as the foundation for decisions regarding more quantitative inventory or monitoring efforts.

A protocol and a data entry program were developed and field-tested by review groups during the summer of 2002. To date, sub-units from seven state and federal agencies—Forest Service, BLM, California F&G, California State Parks, California Tahoe Conservancy, Nevada State Parks, and Tahoe Regional Planning Agency—have collected data using the same protocol, field form, and data entry program.

Protocol for Establishing the Ecological Condition of Aspen Stands

Because the desired condition for aspen habitats across the Western landscape has been defined as fully functional, structurally diverse aspen stands (U.S. Forest Service 1994, U.S. Forest Service 2001), the principal objective of this protocol is to determine whether the age class, structural diversity, composition, and cover of individual aspen stands are within the desired range of natural variability for the vegetative community.

The range of naturally viable, fully functional aspen ecosystems does not express itself in any single simple form. Research has shown that aspen naturally expresses itself through the relationships between its reproduction processes; clones generating suckers off lateral roots; responses to natural environmental events; shade intolerance; growth hormonal balance mechanism; and disturbance dependency (Schier and others 1985).

However, there are a number of historic and current land management factors that have affected the natural viability of Western aspen habitats (Bartos 2001). The protocols focus on three thoroughly researched processes that have been shown to affect the natural viability of aspen: (1) effects of increased conifer encroachment caused by prolonged fire suppression, (2) failure of successful regeneration stemming from poor livestock management or unnatural wildlife stocking numbers, and (3) combinations of these factors. In order to see how these historic and current factors may have affected the current viability of individual stands, the protocol focuses on three key indicators of stand condition: overall stand structure, unique stand management issues, and a ranking of factors that might be putting stands at risk.

Stand Structure

The first key indicator in analyzing a stand's ecological condition is delineating its structure. The protocol's foundation is the use of a definition of "stand" (California Native Plant Society 2001) that calls for separating aspen units into units that are compositionally and structurally similar, i.e., if the mix of all species within the stand has structural and compositional integrity, it meets this definition (figure 1). This delineation of structure should establish not only the presence and condition of aspen age cohorts but also establish the relationship of those cohorts to the two most significant factors affecting stand viability: conifer encroachment and effects of browsing by wild and domestic ungulates.

The protocol divides the stand structure into three classes: overstory canopy cover (aspens and conifers >8 inches DBH), mid-level canopy cover (aspens and conifers 1 inch DBH to 8 inches DBH), and understory (aspens and conifers <1 inch DBH). The standard of 8 inches DBH was selected as

Figure 1—Structurally diverse stand with multiple age cohorts.

a candidate for the separation of overstory and mid-level classes after field-testing established that stems with 8 inches DBH were consistently shorter than the average height of the range of all larger DBH aged cohorts in the overstory. The first two classes—overstory and midlevel—focus on the presence or absence of aspen and conifer species and quantify their percentage presence and ratio to each other. The third class—understory—focuses on aspen regeneration and factors affecting its success. The protocol proceeds from class to class establishing a picture of the current structure of a stand and factors contributing to its functional condition.

The first step of the protocol is to identify what the actual percent of canopy cover is in the overstory and calls for five classes into which that percentage can be placed: 100-85 percent, 84-51 percent, 50-16 percent, 15-1 percent, and 0 percent. Then, the ratio of conifer:aspen is established within the canopy cover that does exist (figures 2, 3, 4). The classes for conifer: aspen ratio are 10:0, 8:2, 6:4, 5:5, 4:6, 2:8, and 0:10. The ratio establishes the dominance or co-dominance relationships between aspen and conifer and provides both an indication of current condition of the overstory canopy and hints of potential future conditions.

Next, after the actual overstory canopy cover is determined, the protocol calls for establishing the presence and condition of the actual mid-level canopy cover. As in the overstory, the protocol calls for establishing the presence or absence of a mid-level canopy cover (1 inch DBH to 8 inches DBH). If present, the actual percent of canopy cover in the mid-level canopy and the ratio of conifer:aspen in the mid-level canopy are established using the same size classes that were used in evaluating the overstory.

Finally, in the third class—the understory(<1 inch DBH)—the protocol calls for establishing the presence or absence of regeneration and identifying the presence or absence of browsing. Greater than 500 stems per acre is the class size used for signifying the presence of a distinctive regeneration stem aspen class. Five hundred stems per acre are used because it is believed that fewer stems per acre will not provide enough stems to establish a fully stocked stand (Mueggler 1989).

The protocol then calls for indicating the presence of significant browse on current year's terminal growth. Browse of 20 percent of current terminal

Figure 2—100-85 percent canopy cover and a ratio of conifer:aspen of 0:10. No other aspen or conifer age cohorts present.

Figure 3—85-51 percent overstory canopy cover and a conifer:aspen ratio of 0:10 present. A midlevel canopy cover of 100-85 percent with a ratio of conifer:aspen of 10:0.

leader growth was selected because it is the current browse standard for woody stem vegetation found in the SNFPA (USDA Forest Service 2001) as well as in the standard used by the Bureau of Land Management in California (BLM—California State Office 2002).

A final note regarding stand structure: the protocol has an indicator for establishing the presence of a "decadent stand." Decadent stands are qualitatively described in this protocol as stands that have more dead or down aspen stems than living stems.

Management Issues

The second key indicator of a stand's ecological condition is delineation of any special stand management issues that provide clues to factors playing a role in the current status of the stand, a potential role in the future condition of the stand, or a role in potential stand management options. Included in the management issues are:

- location of stands within or adjacent to biologically sensitive habitats like meadows, riparian corridors, or springs;
- location of stands within or adjacent to geological refugia—environments conducive to stand

Figure 4—Two age cohorts with an overstory canopy cover of 85-51 percent and a ratio of conifer:aspen of 2:8 with an understory age cohort of >500 stems per acre.

continuation because of protection from browsing pressures or conifer encroachment or because of location on a site with particular geological characteristics such as lava flow, talus slope, rock outcrop, or moraine material;

- indication of significant (>20 percent) insect or disease damage;
- presence of significant human impacts such as regulated or non-regulated camping or structures;
- indication of signs of recent wildfires within or adjacent to the stand;
- indication of prior management treatment such as conifer removal within a stand or prescribed burns;
- indication of the presence or historic effect of beaver populations.

Risk Loss Analysis

The third key step in indicating a stand's ecological condition is evaluating combinations of factors observed in a stand and rating its potential for being lost. This procedure, which appears in the protocol as "Stand Loss Risk Factor," was initially set forth in a prioritized key for risk factors developed by Bob Campbell, Fish Lake NF, and Dale Bartos, RMRS (2001). The process, as it appears in the protocol, is the result of adaptation of the Campbell and Bartos work by the Eagle Lake Ranger District, Lassen National Forest and extensive field-testing of the end product.

Five classes of risk loss are found in the protocol:

Figure 5—Risk to stand is "Highest." This stand is being lost from above and not being replaced from below.

Highest: The clone is being lost from above and is not being replaced from below (figure 5).

- Conifer crowns have overtopped the aspen crowns, (primary risk factor), and conifer species comprise at least half the canopy (primary risk factor), *and*
- Regeneration is absent or unsuccessful because of excessive browsing or shading from conifer encroachment (primary risk factor).

High: The clone is being lost from above or is not being replaced from below (figure 6).

Moderate: One or more risk factors are present, but the clone is not in immediate danger. This class may include one or more of the factors below:

- conifer closure >25 percent, but <50 percent (if >50 percent, rating is High or Highest)
- aspen cover <40 percent
- dominant aspen are decadent
- aspen regeneration 5-15 feet tall is <500 stems per acre
- regeneration is excessively shaded by conifers
- browsing is limiting extent and numbers of successful (>5 feet tall) regeneration

Figure 6—Risk to stand is "High." This stand is being lost from below through conifer encroachment.

Figure 7—Risk to stand is "Low." Multiple age cohorts are present and few risk factors (<15 percent) exist.

Low: The clone is essentially healthy, containing mature trees and/or healthy and vigorous regeneration; there are no obvious signs that the clone has receded; and <15 percent of the clone is affected by risk factors (figure 7).

None: None of the above risk factors are present. Mature trees are vigorous, and regeneration 5-15 feet tall ≥500 stems per acre.

Support Material

To date, the Aspen Delineation Project has developed three support programs for facilitating effective and efficient use of the protocol:

- A Data Entry Program to simplify the process of moving hard copy (field form) data into GIS shape files as well as providing a vehicle for merging interagency aspen data into some centralized data source.
- A field crew training program to help crews understand and interpret goals and objectives of the assessment process and develop consistent interpretations of stand conditions.
- A CD containing PDF copies of the protocol and field form, a copy of the Data Entry Program, and an interactive guide to the protocol.

Copies of the CD or details about the protocol and the Aspen Delineation Project's field crew training program can be obtained by contacting peregrines@prodigy.net or the Aspen Delineation Project, P.O. Box 348, Penryn, CA 95663.

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Composition and Development of Reproduction in Two-Age Appalachian Hardwood Stands: 20-Year Results

Gary W. Miller[1], James N. Kochenderfer[2], and Desta Fekedulegn[3]

Abstract— *In the early 1980s, silviculturists with the Northeastern Research Station and Monongahela National Forest envisioned that managing some Appalachian hardwood stands to promote two-age structures would be part of an effective strategy for managing multi-use forests. Two-age stands provided the light and seedbed conditions necessary for regenerating numerous desirable hardwood species and maintaining species diversity. The residual overstory trees also served to maintain vertical structure, mast production, wildlife habitat, and esthetic quality in visually sensitive areas. This paper examines the composition and development of approximately 20-year-old reproduction growing beneath 100-year-old residual overstory trees in two-age central Appalachian hardwood stands. After 20 years, the crowns of the residual overstory trees had expanded by nearly 80 percent and collectively covered almost half of the stand area. Desirable shade-intolerant species such as black cherry, northern red oak, and yellow-poplar in the new age class had remained competitive only in areas located between the crowns of the residual overstory trees. The areas located beneath the residual overstory trees were occupied by tolerant species such as sugar maple, red maple, and American beech. Data indicated that the residual overstory trees had a strong influence on the development and long-term species composition of reproduction. Practical alternatives for planning and implementing two-age management systems are discussed.*

Introduction

Clearcut harvesting became a common silvicultural practice in eastern hardwoods nearly 40 years ago for several reasons. Some were based on the need to facilitate the management of large forested areas, while others were based on the need to promote the biological conditions necessary for regenerating a wide range of hardwood species. For management objectives, clearcut harvesting resulted in new stands with even-aged structures, which allowed forest planners to manage mosaics of stands and balance the age distribution of large industrial and public forests, thus sustaining the long-term yield of a variety of woodland benefits. In addition, simultaneous removal of all merchantable stems promoted harvesting and marketing economies of scale that were advantageous to both buyer and seller of wood products. For biological objectives, the open conditions following removal of the overstory created the sunlight and seedbed conditions that led to prolific regeneration of the full range of native tree species from stored seed, advanced seedlings, and sprouts. A diverse mix of highly valuable hardwood species for both wildlife and timber developed rapidly as young trees responded to excess growing space freed by the harvest operations. Silviculturists understood that disturbance patterns greatly affect the regeneration processes among tree species. Scientific evidence had shown that repeated partial harvests

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promoted uneven-age stand structures, undesirable shifts in species composition toward shade-tolerant species, and over time, a general loss of tree species diversity (Trimble 1973). Alternatively, clearcut harvesting produced new even-age stands that contained numerous tree species, regardless of shade tolerance. This response allowed forest managers to achieve the critical management objective of replacing harvested trees with young vigorous trees of the same species, thus maintaining species diversity. In the 1960s, clearcut harvesting became a common harvesting practice because it was an effective way to meet multiple management objectives within the biological constraints of the hardwood forest.

Along the way, public opposition to the appearance of forest stands immediately after clearcut harvests led forest managers and scientists to seek an alternative practice that was equally effective in sustaining desirable species composition but less offensive to the eye. The cooperative effort between the Monongahela National Forest and the Northeastern Research Station to study the effects of two-age stand structures began in the 1970s and continues today (Miller and others 1997b).

From 1979 to 1984, a form of deferment cutting (Smith and others 1989) was applied in mature, second-growth Appalachian hardwood forests to create experimental stands with two-aged structures. Deferment cutting (also called clearcutting-with-reserves, shelterwood-with-reserves, or irregular shelterwood) entails leaving 10 to 20 residual overstory trees per acre and cutting all other stems ≥1.0-inch diameter at breast height (DBH) (Smith and Miller 1991a). The open conditions following deferment cutting are similar to those exhibited after clearcut harvesting, thus providing adequate sunlight and seedbed conditions for numerous hardwood species to regenerate naturally (Johnson and others 1998). Several previous reports on the experimental two-age stands documented the initial response of the residual overstory trees (Smith and Miller 1991b; Smith and others 1994; Miller and others 1995; Miller 1996), the early development of reproduction (Miller and Schuler 1995; Miller and others 1997a), and the impact of vertical structure on insect and songbird habitat (Duguay and others 2000).

Many years have elapsed since the experimental stands were created, and there is now an opportunity to further evaluate the interaction of residual overstory trees and reproduction in two-age stands. As these stands continue to develop, several key silvicultural questions remain. How fast does the stand area covered by the crowns of the residual overstory trees expand over time? How do residual overstory trees influence the development of reproduction? How can silvicultural prescriptions applied to promote two-age stands be improved? Although the results presented here are relatively limited in scope, they shed light on important issues that silviculturists need to consider when applying two-age management systems in hardwood forests.

Study Areas

The four study areas are located on the Monongahela National Forest in north-central West Virginia. All stands were on northern red oak site index 70 to 80 (base age 50 years). The average soil depth exceeds 3 feet, and annual precipitation averages 59 inches distributed throughout the year. The soil series for each stand were described as follows: Riffle Creek (Berks channery silt loam), Fish Trough (Calvin channery silt loam), Red

House (Dekalb stony loam), and Shavers Fork (Berks channery silt loam), all characteristic of loamy-skeletal, mixed, mesic Typic Dystrochrepts (USDA Soil Conservation Service 1967; 1982).

When the study began, the study areas were 75- to 80-year-old, second-growth Appalachian hardwood stands that became established after heavy commercial harvests that occurred in the early 1900s. Periodic fire was common throughout the region from 1910 to 1930 as the stands became established. Chestnut blight was also a common disturbance during the 1930s, but no other major disturbances occurred until deferment cutting was applied in 1979 to 1984 to begin studying the two-aged management system. Stand area ranged from 10 to 15 acres.

Methods

In each stand, deferment cutting (Smith and others 1989) was applied to regenerate a new age class of trees and promote the development of a two-age stand structure. The boles of merchantable cut-trees were removed, while their tops and all non-merchantable cut-trees were left on the site. Residual trees were selected with preference for codominant, high-quality northern red oak (*Quercus rubra*) followed by yellow-poplar (*Liriodendron tulipifera*) and black cherry (*Prunus serotina*). Marking guidelines also called for leaving 15 to 20 overstory trees/acre with residual basal area less than 30 ft^2/acre, thus promoting suitable light conditions for vigorous natural regeneration of a diverse mix of hardwood species. The residual overstory trees were tagged for long-term monitoring. DBH, total height, and crown radius of residual overstory trees were measured immediately after the logging operations were completed.

Reproduction data were obtained before logging and at 5-year intervals after logging from 170 permanent sample points located along systematic grids throughout the four study areas. Small reproduction, defined as woody stems ≥12 inches tall and <1.0 inch DBH, was tallied on 0.001-acre circular plots. Large reproduction, defined as woody stems ≥1.0 inch DBH, was tallied on 0.01-acre circular plots using the same center points. These plots provided a general inventory of stem counts of reproduction over time, but they did not indicate how residual overstory trees affected the growth and distribution of reproduction within the stand.

In 2001, data were collected on randomly selected northern red oaks in the residual overstory and on the reproduction growing adjacent to or under their crowns. Measurements on the overstory trees included DBH, total height, and crown radius along the 0°, 90°, 180°, and 270° azimuths. Reproduction was tallied in four transects that radiated from the bole of the residual overstory tree oriented along the same four azimuths. The transect length was equal to the crown radius plus 15 feet along each azimuth, and the transect width was defined as the area within 5 feet of the centerline, for a total width of 10 feet (figure 1). Measurements on the reproduction included species, DBH, total height, crown class, and distance to the residual overstory tree bole for each stem found within a transect. Crown class of the reproduction was based on the status of each stem relative to neighboring stems of the same age, without regard for the residual overstory trees. Crown classes were defined by Smith (1986). The residual overstory tree crown radius minus the distance between a stem and the residual overstory tree bole defined the relative position of each stem tallied. Note

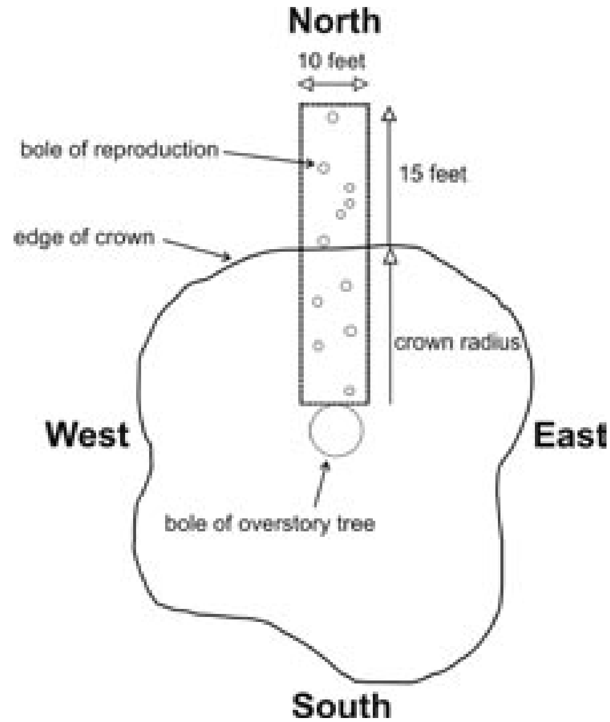

Figure 1—Schematic of transects used to collect reproduction data under and adjacent to residual overstory trees.

that trees with relative position <0 were located under the residual overstory tree crown, while trees with relative position >0 were located adjacent to or beyond the edge of the residual overstory tree crown. Relative position was used as a predictor in graphical and regression analyses of DBH and total height of reproduction.

The data were analyzed to determine if DBH, total height, basal area per acre, and species composition of the reproduction were related to proximity to the residual overstory trees. To conduct the analysis, reproduction was classified by two location groups and three species groups. Stems located within the overstory tree crown radius were classified as under, while stems located beyond the overstory tree crown radius were classified as adjacent. In addition, species groups were defined by shade tolerance class as defined by Trimble (1975). Intolerant stems included yellow-poplar, black cherry, and fire cherry (*Prunus pensylvanica*); mid-tolerant stems included the oaks (*Quercus* spp.), hickories (*Carya* spp.), white ash (*Fraxinus americana*), cucumbertree (*Magnolia acuminata*), and sweet birch (*Betula lenta*); tolerant stems included American beech (*Fagus grandifolia*), sugar maple (*Acer saccharum*), red maple (*Acer rubrum*), and basswood (*Tilia americana*).

Statistical Analyses

Regression analyses using the GLM procedure (SAS 2000) were used to relate DBH and total height of codominant reproduction to the position of each tree relative to the residual overstory tree crown. The assumption of common slope among the four stands was tested using analysis of covariance, which related DBH and height to stand, position, and their interaction. Violation of normality and homogeneity of variance assumptions required the use of log transformed values.

The basal area per acre of reproduction tallied within a transect by tolerance class (intolerant, mid-tolerant, and tolerant) and location (under versus adjacent) relative to a residual overstory tree was compared using a multiple factor, repeated measures analysis of variance modeled in the MIXED procedure (SAS 2000). Class variables included stand, cardinal direction, location, and tolerance class, where the latter three variables were repeated measures on a given subject residual overstory tree. Preliminary analyses of the covariance matrix revealed that the compound symmetry assumption was not feasible, so analyses were performed assuming a heterogeneous autoregressive structure. Log transformed values were also used in this analysis to account for violation of normality and homogeneity of variance assumptions.

Results

Pre-harvest basal area averaged 138 ft^2/acre including poletimber and sawtimber, with an additional 5 to 10 ft^2/acre in saplings (table 1). The initial overstory included over 80 sawtimber-sized trees/acre with an average DBH of 15.8 inches. In general, yellow-poplar and northern red oak occupied most of the initial overstory, although black cherry, white oak (*Quercus alba*), sugar maple, and American beech also occupied significant proportions of the overstory in individual stands (table 2). Sugar maple and American beech were the most prevalent species in the pool of advanced reproduction before harvest (table 3).

Basal area in the residual stands averaged 23 ft^2/acre, which included 14 sawtimber-sized trees/acre with an average DBH of 17.1 inches (table 1). The composition of residual overstory trees averaged 50 percent yellow-poplar, 30 percent northern red oak, and 20 percent a mixture of other commercial species.

Table 1—Summary of stand characteristics before and after harvests in four central Appalachian hardwood stands.

Stand	Number of trees 6 to 10 in.[a]	Number of trees ≥12 in.	Basal area 6 to 10 in.	Basal area ≥12 in.	Volume[b] ≥ 12 in.	Average DBH ≥12 in.
	----- no./acre -----		----- ft^2/acre -----		- Mbf/acre -	---- in. ----
Riffle Creek						
Initial	120.2	68.6	41.3	87.7	1 2,932	15.3
Cut[c] 1979	118.9	57.4	40.6	70.9	10,305	15.0
Residual	1.3	11.2	0.7	16.8	2,627	16.6
Fish Trough						
Initial	48.9	72.2	16.5	112.7	20,630	16.9
Cut 1980	48.7	58.1	16.4	88.0	15,697	16.7
Residual	0.2	14.1	0.1	24.7	4,933	17.9
Shavers Fork						
Initial	110.6	63.4	34.1	81.9	14,221	15.4
Cut 1981	110.2	50.7	33.9	61.8	10,363	15.0
Residual	0.4	12.7	0.2	20.1	3,858	17.0
Red House						
Initial	65.5	120.4	21.9	156.9	27,561	15.5
Cut 1984	65.3	101.8	21.8	127.8	22,014	15.2
Residual	0.2	18.6	0.1	29.1	5,547	16.9
All						
Initial	86.3	81.2	28.5	109.8	18,836	15.8
Cut	85.8	67.0	28.2	87.1	14,595	15.5
Residual	0.5	14.2	0.3	22.7	4,241	17.1

[a] DBH.
[b] International ¼-inch rule.
[c] All trees ≥1.0 inch DBH were felled unless selected and marked as a residual tree.

Table 2—Species composition before commercial harvests in four central Appalachian hardwood stands.

Commercial species	Riffle Creek	Fish Trough	Shavers Fork	Red House	All
	-------------------------------- % basal area/acre --------------------------------				
Sugar maple	2.4	6.9	4.0	2.0	3.8
American beech	1.9	14.6	9.3	0.5	6.6
Red maple	10.0	0.6	2.2	2.4	3.8
White ash	< 0.1	1.5	4.9	2.5	2.2
White oak	20.0	0.0	1.6	0.0	5.4
N. red oak	14.7	5.2	14.8	15.7	12.6
Yellow-poplar	13.4	38.4	46.7	46.7	36.3
Black cherry	< 0.1	3.7	2.7	27.9	8.6
Others	37.5[a]	29.1[b]	13.8	2.3	20.7[c]
Totals	**100.0**	**100.0**	**100.0**	**100.0**	**100.0**

[a] 22% chestnut oak.
[b] 18% American basswood.
[c] Includes 17 species.

Table 3—Advanced reproduction ≥12 inches tall and <1.0-inch DBH present before harvests in four central Appalachian hardwood stands.

Commercial species	Riffle Creek	Fish Trough	Shavers Fork	Red House	All
	-------------------------------- no. of stems/acre --------------------------------				
Sugar maple	745	833	380	558	629
American beech	1575	450	1200	209	859
Red maple	340	0	0	47	97
White ash	43	200	520	23	197
White oak	128	0	20	0	37
N. red oak	255	33	40	140	117
Yellow-poplar	0	33	20	256	77
Black cherry	128	33	0	721	221
Others	169	18	140	279	150
Totals	**3,383**	**1,600**	**2,320**	**2,233**	**2,384**

Data were collected on woody reproduction located under and adjacent to 29 residual northern red oaks in the four study areas (table 4). Growth of the residual overstory trees in the 17- to 22-year period after deferment cutting averaged 8 inches DBH, 10 feet in total height, and 5 feet in crown radius (table 4). Based on crown radius growth the average area covered by each residual overstory tree crown increased from 660 ft^2 to 1170 ft^2, an increase of almost 80 percent during the study period.

Table 4—Mean DBH, total height, and crown radius of residual northern red oaks in two-aged central Appalachian hardwood stands.

Stand name	No. of trees observed	Year	DBH	Total height	Crown radius
			---- in. ---	---------------- ft -----------------	
Riffle Creek	6	1979	14.3	89.8	13.3
		2001	22.0	96.0	18.1
Fish Trough	7	1980	16.7	98.7	15.1
		2001	25.9	110.9	20.9
Shavers Fork	8	1981	15.7	91.1	14.8
		2001	25.4	99.6	21.6
Red House	8	1984	18.1	101.4	14.5
		2001	24.4	112.9	16.7
All	29	Variable	16.3	95.5	14.5
		2001	24.5	105.3	19.4

Effect of Overstory Trees on DBH and Height of Reproduction

DBH and total height of codominant reproduction were positively related to the position of individual trees relative to the crowns of residual overstory trees ($p < 0.01$). In general, DBH and total height increased as distance from residual overstory trees increased (figures 2 and 3). Regressions analyses also indicated that DBH and total height differed by stand ($p < 0.01$), probably reflecting subtle effects of site quality, tree age, and species composition. Mean DBH was 3.5 inches for trees located under residual overstory trees compared to 3.9 inches ($p = 0.04$) for trees located adjacent to residual overstory tree crowns (table 5). Similarly, mean total height for trees located under or adjacent to overstory trees was 41.8 feet and 44.8 feet ($p < 0.01$), respectively. The analyses revealed no interaction between stand and position, indicating that slope coefficients were not significantly different. This finding supports the general conclusion that DBH and height of codominant reproduction was strongly related to position of codominant reproduction relative to residual overstory trees, regardless of study site.

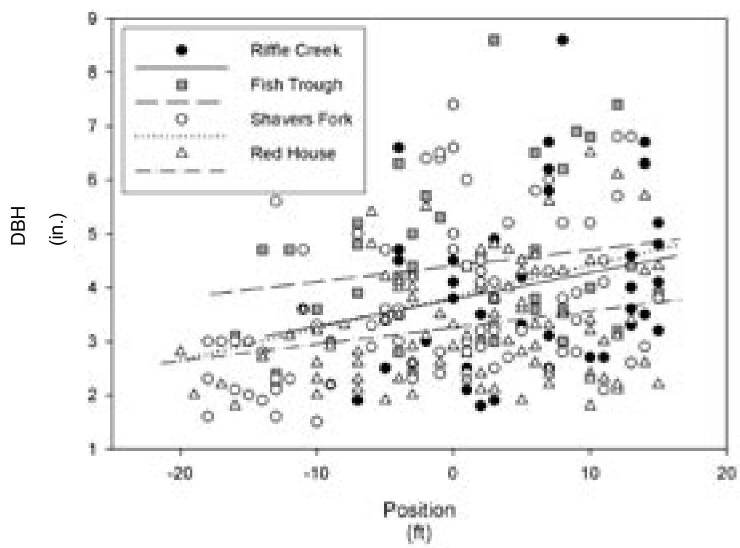

Figure 2—DBH of codominant reproduction by position of stem relative to edge of overstory tree crown.

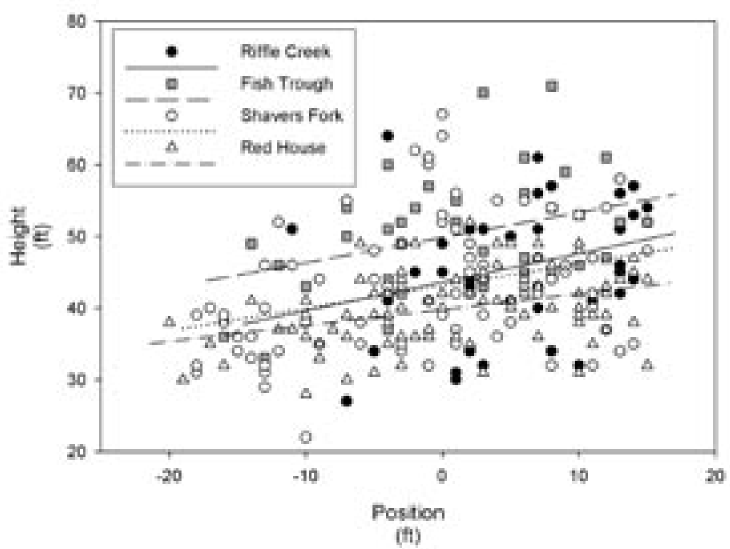

Figure 3—Total height of codominant reproduction by position of stem relative to edge of overstory tree crown.

Table 5—Mean DBH and total height of 17- to 22-year-old codominant reproduction by location relative to residual overstory trees.

	DBH			Total height		
	Location			Location		
Stand	Under	Adjacent	p-value	Under	Adjacent	p-value
	------------- in. (n)[a] -------------			----------- ft ---------		
Riffle Creek	3.92 (10)	4.10 (31)	0.76	45.0	46.1	0.74
Fish Trough	4.35 (20)	4.49 (24)	0.74	48.0	51.9	0.11
Shavers Fork	3.54 (43)	3.98 (42)	0.16	41.7	43.8	0.27
Red House	3.10 (42)	3.44 (52)	0.11	38.1	41.5	<0.01
All	3.55 (115)	3.90 (149)	0.04	41.8	44.8	<0.01

[a] Number of trees in sample.

Effect of Overstory Trees on Species Composition of Reproduction

A recent survey of the permanent sample points located in each stand indicated that yellow-poplar, black cherry, red maple, and sweet birch accounted for most of the codominant stems in the reproduction (table 6). Other valuable species such as sugar maple and northern red oak also were present in relatively small proportions.

Based on data collected in transects in 2001, the residual overstory trees significantly influenced the distribution of species occupied by codominant reproduction (table 7). Both location ($p < 0.01$) and tolerance class ($p = 0.03$) were significant class variables, while stand and cardinal direction were not. Interaction between location and tolerance class was not evident ($p = 0.69$). Basal area of codominant reproduction adjacent to residual trees averaged 49.5 ft^2/acre compared to 31.7 ft^2/acre under residual trees ($p < 0.01$). It was clear that basal area of codominant reproduction of intolerant and mid-tolerant species was greatest in the space adjacent to residual trees, where competition for sunlight is reduced (table 7). Note that basal area of mid-tolerant species, which includes the oaks, was 18.2 ft^2/acre adjacent to residual trees compared to only 10.0 ft^2/acre under residual trees ($p < 0.01$), a difference of over 80 percent.

When all crown classes of reproduction were considered in the analysis, basal area averaged 72.5 ft^2/acre and 55.5 ft^2/acre for trees adjacent to or

Table 6—Composition of 17- to 22-year-old reproduction in two-aged central Appalachian hardwood stands.

Commercial species	Riffle Creek	Fish Trough	Shavers Fork	Red House	All
	-------------------- no. of stems/acre --------------------				
Sugar maple	128[a] (36)[b]	300 (9)	75 (3)	16 (0)	130 (12)
American beech	315 (92)	70 (0)	170 (0)	16 (0)	143 (23)
Red maple	187 (100)	7 (3)	55 (10)	56 (5)	76 (30)
White ash	6 (0)	25 (2)	34 (0)	12 (0)	19 (1)
White oak	34 (6)	0 (0)	170 (0)	0 (0)	51 (2)
N. red oak	181 (79)	17 (2)	35 (8)	30 (0)	66 (22)
Yellow-poplar	87 (45)	190 (78)	318 (112)	656 (198)	313 (108)
Black cherry	57 (36)	24 (8)	9 (3)	228 (86)	80 (33)
Others	448 (225)	311 (88)	21 (20)	467 (139)	312 (118)
Totals	**1443 (619)**	**944 (190)**	**887 (156)**	**1481 (428)**	**1189 (349)**

[a] All stems ≥ 1.0 inch DBH.
[b] Dominant and codominant stems.

Table 7—Mean basal area per acre of 17- to 22-year-old reproduction by tolerance class and by location relative to residual overstory trees.

Location	Transect sample size	Tolerance class Intolerant	Mid-tolerant	Tolerant	Total
	n	-----------------------------	ft²/acre	-----------------------------	
			All trees		
Adjacent[a]	91	24.9	24.7	22.9	72.5
Under	91	19.3	17.2	19.0	55.5
			Codominant trees		
Adjacent	91	19.9	18.2	11.4	49.5
Under	91	15.1	10.0	6.6	31.7

[a] Reproduction was located under residual overstory tree crowns or adjacent to residual trees where there was no residual overstory competition.

under residual trees, respectively, but the distribution of species was not affected by location relative to the residual overstory trees (table 7).

Discussion

The analysis focused on the composition and development of codominant reproduction, while subordinate stems received little attention. At the 20-year stage of development, when crown classes are evident, codominant stems are those most likely to remain competitive for an extended time. For intolerant and mid-tolerant species, in particular, subordinate stems have little chance of exhibiting sufficient height growth to survive or maintain their position in the canopy of an aggrading stand (Trimble 1973). Intolerant species also are more likely to lose competitiveness once they become subordinate to neighboring trees of the same age or receive overhead competition from older, taller trees.

The observed influence of overstory trees on reproduction will increase as the crowns of the overstory trees expand. When transect data were recently collected, the residual overstory trees were approximately 100 years old, and the reproduction ranged from 17 to 22 years old. Overstory trees were more than 100 feet tall, while codominant reproduction was less than 50 feet tall. Height growth of the residual overstory trees was relatively slow, about 0.5 ft/yr (table 4), compared to 4.5 ft/yr (table 5) for the reproduction. At these rates of growth, the reproduction needs an additional 30 to 40 years to become competitive with overstory trees. Similarly, the crowns of residual overstory trees are likely to expand and continue to influence the development of the reproduction for many years. In this study, retaining 15 trees per acre with an average DBH of 16 inches resulted in a 22 percent canopy cover immediately after the harvest. After 20 years, average DBH had increased to 24 inches, and canopy cover had increased to 39 percent. In a few more years, nearly half of the stand area will be located under the crowns of the residual overstory trees. It is apparent from these observations that successful reproduction of valuable intolerant and mid-tolerant species will be limited to less than half the stand area in years to come.

Northern red oak has a relatively large crown area compared to other species of the same DBH (Lamson 1987). The results presented here represent a worst-case scenario in terms of how the residual overstory trees can exhibit a negative impact on reproduction, particularly shade-intolerant species. Species with smaller crowns or less aggressive crown expansion would tend to occupy less of the stand area compared to northern red oak.

Reproduction in the new age class would benefit from an intermediate cultural practice such as cleaning or crop tree release (Miller 2000). These precommercial treatments free selected codominant stems from crown competition, thus accelerating their growth and increasing their future competitiveness. In two-age stands, however, opportunities to successfully release intolerant and mid-tolerant species may be limited to areas located between the residual overstory trees.

This study did not compare reproduction in two-age stands to that observed in even-age stands, but there is evidence that the residual overstory trees in two-age stands reduced the overall growth of codominant reproduction, regardless of its location. Data collected from 17-year-old even-aged stands located a few miles from the experimental two-age stands indicated that average DBH exceeded 6 inches and average total height exceeded 50 feet for codominant reproduction on site index 70 (Miller 2000). By comparison, DBH and total height of codominant trees located adjacent to residual overstory trees in the two-age stands averaged only 4 inches and 45 feet, respectively (table 5). Codominant reproduction located under the residual overstory trees was even smaller.

There are several alternative strategies for managing two-age stands to reduce the impact of residual overstory trees on reproduction:

- Leave fewer trees per acre in the initial harvest operation.
- Leave species with relatively smaller crowns and less aggressive crown expansion.
- Leave clumps of residual trees to increase open space and reduce the effect of residual tree crown expansion.
- Start with 15 residual trees per acre, but reduce the number of residual trees by chemical or mechanical means once the new age class is established.

Finally, it is important to prepare for the regeneration of desired species many years before conducting harvests in hardwood stands (Miller and Kochenderfer 1998). An assessment of the regeneration potential should be conducted before a planned harvest to determine if preparatory silvicultural treatments are needed. The negative impacts of interfering plants and high deer populations must be controlled to allow desirable advanced reproduction to develop many years before a planned harvest. Once competitive advanced reproduction is in place, removing most or all of the overstory will provide the light and seedbed conditions for a new age class of desirable trees to develop.

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Shortleaf Pine-Bluestem Habitat Restoration in the Interior Highlands: Implications for Stand Growth and Regeneration

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***Abstract**—National Forest managers in the Interior Highlands of Arkansas are restoring 155,000 acres of unburned shortleaf pine stands to shortleaf pine-bluestem habitat. Habitat restoration consists of longer rotations, removal of midstory hardwoods, and reintroduction of fire. A study was installed in the spring of 2000 to evaluate shortleaf pine regeneration and overstory stand growth under treatment. At this point in the study, there is no difference in milacre stocking of pines related to number of growing seasons after burning. Analysis suggests that residual basal areas below 50 ft^2 per acre will be needed to develop sufficient advance growth of shortleaf pine to ensure regeneration when regeneration cutting is implemented. Over a four-year period, growth in treated and control stands is substantially less than that predicted from growth models developed in this forest type. However, there are no significant differences in growth over four years between treated stands and the control stands.*

Introduction

The Ouachita Mountains cover approximately 6.6 million acres in western Arkansas and eastern Oklahoma, of which 85 percent is forested. Nearly 40 percent of the area is in forest stands dominated by shortleaf pine, which comprises 46 percent of the live-tree volume, 50 percent of growing-stock volume, and 67 percent of sawtimber board-foot volume on timberlands in the region; over half of the shortleaf pine volume in the region is found on National Forest land (Guldin and others 1999).

These shortleaf pine forests evolved with fire. Historical accounts and General Land Office records suggest that at the time of European colonization, the forest was more open than it is today, and fires no doubt contributed to that low density (Foti and Glenn 1991). Both wildfires and fires started by Native Americans were probably common. But since the 1930s, fire suppression has been extremely effective in the region. Fire return intervals prior to European settlement were on the order of from two to 40 years. Today, fire return intervals average 500 years or longer (Foti and others 1999), although the recent and widespread increase in the use of prescribed burning in contemporary forestry practice may alter this rate downward in the future.

As managers on National Forest lands seek to restore natural patterns and processes to the shortleaf pine ecosystem, the reintroduction of frequent surface fires is critical to success. Fire exclusion and conservative approaches to thinning second-growth shortleaf pine and pine-hardwood stands have produced stands that have more pines, and more hardwoods especially in the smaller size classes, than would be expected if fires had continued to burn

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over the past seven decades. This is at odds with descriptions in the literature that refer to open woodlands of pine and hardwood, with big bluestem (*Andropogon gerardii*) and other grasses in the understory (Foti and others 1999). Among the prominent advantages for this restoration is the attendant benefits that would be provided for restoration of a healthy population of the endangered red-cockaded woodpecker (*Picoides borealis*).

The Ouachita National Forest completed a Forest Plan Amendment in 1996, encompassing 155,000 acres of national forest land in western Arkansas and eastern Oklahoma in Management Area (MA) 22. The goal of the new MA22 is to implement restoration of the shortleaf pine-bluestem ecosystem, to implement the direction of the Region 8 RCW Environmental Impact Statement, and to maintain and enhance other associated resource values and attributes. Management Area 22 has been a resounding success with resource managers and the public. For it, the Ouachita NF received the Chief's Award for Forest Stewardship in 2001. And the National Audubon Society honored the project with its designation as a nationally Important Bird Area in 2002.

The prescriptions include removal of most midstory hardwoods, thinning from below in overstory and midstory pines, and reintroducing surface fires on a one- to three-year return interval. These treatments have been effective in restoring many underrepresented species in the landscape, such as purple coneflower (*Echinacea pallida*), bobwhite quail (*Colinus virginianus*), red-cockaded woodpecker, Bachman's sparrow (*Aimophila aestivalis*), and eastern wild turkey (*Meleagris gallopavo*) (Bukenhofer and Hedrick 1997). In addition, there is roughly seven times the preferred forage for white-tailed deer (*Odocoileus virginianus*) in treated versus untreated areas (Masters and others 1996).

The details of the restoration prescription are designed to carry the existing fully stocked second-growth pine and pine-hardwood stands to restored condition. Pre-restoration stands typically contain approximately 130 ft^2 per acre of basal area, of which 100 ft^2 per acre is in the pine component and 30 ft^2 per acre in the hardwood component; of the hardwood basal area, two-thirds is in the sub-sawtimber size class and most likely originated as a result of fire control efforts implemented in the 1930s (Guldin and others 1994).

The restoration prescription involves three components. First, the overstory basal area is reduced from 100 ft^2 per acre to approximately 75-80 ft^2 per acre using essentially a thinning from below and implemented using commercial timber sales. Those sales typically remove from 2000-3000 board feet (fbm) Scribner per acre, and are a successful and integral element of the district timber sale program. The sales generate program dollars (under the Knutsen-Vandenberg Act of 1930, as amended by the National Forest Management Act of 1976) for subsequent habitat treatments. The second element of the prescription is the removal of midstory hardwoods that have encroached in the stands as a result of seven decades of fire exclusion. Material is generally felled by chainsaws or girdled and left standing to meet standing snag requirements in the MA22 plan amendment. Finally, as the felled hardwoods decompose, a program of triennial prescribed fire is implemented. Generally, two prescribed burns can be imposed prior to the closure of the five-year window for expenditure of K-V trust funds collected in the initial thinning that triggers the process. The entire sequence of restoration would not be possible in the absence of a viable district timber management program.

In the long term, two silvicultural issues are of interest to ensure long-term productivity and sustainability of the prescriptions associated with

MA22. First, a question exists about the impacts of repeated prescribed burning regimes on the growth of the existing pine overstory. This has a bearing on the length of time needed to support supplemental thinning and eventually on the yields that might be expected when stands in the area are ultimately subject to regeneration cutting.

The second question is the exact manner by which shortleaf pine regeneration can be expected when the time comes for regeneration cutting in MA22 stands. Shortleaf pine is one of the few pines with the ability to resprout at young ages if the top is killed, a trait described in the early 1900s as an adaptation to fire (Mattoon 1915). This resprouting ability might allow the development of a reproduction cutting method based on an understory storage bank of pine seedlings and saplings that can be relied upon to respond to release, if the overstory basal area is substantially reduced below fully-stocked levels. Under proper conditions, this advanced pine regeneration could be quickly released with overstory removal. That would provide for more reliable reforestation than relying either on shortleaf pine's inconsistent annual seed production or on the timing of a site preparation prescription designed to catch whatever seedfall is available in a given year.

The question, then, is to characterize the growth and yield of shortleaf pine stands under restoration, and proper conditions and silvicultural prescriptions under which shortleaf pine advance growth can be obtained and accumulated in light of the triennial regime of prescribed fire currently being used in the pine-bluestem restoration prescription. In this paper, a first look is provided at observed growth in shortleaf pine stands, and on the differences in pine regeneration size, density, and stocking after one, two, and three seasons of growth after prescribed fire in stands being managed under the shortleaf pine-bluestem restoration prescription.

Methods

Study Area

This study area is located on the Poteau Ranger District of the Ouachita National Forest. Twelve stands were randomly selected for inclusion in the study; they are all mature stands dominated by shortleaf pine and contain a minor and varying proportion of hardwoods. Site indices vary between 55 and 65 feet (base age 50 years), and stand age varies from 55 to 70.

Treatments

Nine of the stands are under active prescriptions for shortleaf pine-bluestem restoration, and three are in nearby stands comparable in age and site quality but which have not been subject to the restoration treatment. The nine treated stands had all been subject to treatment for at least five years; treatment included midstory reduction (hardwood midstory trees were chainsaw-felled), a low thinning of light intensity in the pine overstory, and had been burned at least twice using dormant-season fires. Three of the stands had last been burned during the 1999 dormant season, three during the 1998 dormant season, and three during the 1997 dormant season. Thus, treatments were identified as one (B1), two (B2), and three (B3) years after burning, respectively; the control treatment (C) remained untreated by thinning, midstory reduction, or burning. Stands in the B2 treatment were thinned recently, and thus are excluded from the growth analysis in this paper.

Plot Measurements

Within each of the 12 stands, six plots were established on a square 4-chain x 4-chain grid. At each plot, a nested series of measurements were taken. Overstory trees, defined as trees with diameter at breast height (DBH) greater than > 9.6 inches inclusive, were sampled on a 0.2-acre fixed-radius plot. Midstory trees, defined as trees between 3.6 inches and 9.5 inches in diameter, inclusive, were sampled on a 0.1-acre fixed radius plot. Twelve milacres were established on a 13.2-foot grid within the 0.1-acre fixed radius plot, with the grid point at plot center omitted.

Overstory and midstory trees were sampled by measuring DBH to the nearest 0.1 inch and recording their species identity. Milacre measurements proceeded in a different manner using two tallies: an inventory tally and a tagged-tree tally. The inventory tally consisted of a count of all seedlings and saplings greater than 6 inches tall but less than 3.5 inches in diameter, inclusive, on each milacre. The tagged-tree tally consisted of subdividing the milacre into quadrats, identifying and tagging the tallest conifer and tallest hardwood on each quadrat (if present), and recording groundline diameter, DBH, and total height.

Data Analysis

The stem density and basal area for combined overstory and midstory trees per plot were calculated by applying the appropriate expansion factor to trees from either the 0.1-acre or 0.2-acre plot. The Shortleaf Pine Stand Simulator model (SLPSS, Lynch and others 1999) was used to calculate overstory timber volumes in the spring of 2000 and the summer of 2003, providing data on stand overstory stand growth over four years. Milacre stem density per plot was obtained by tallying the inventory of regeneration stems, applying the appropriate expansion factor per milacre, and averaging for the 12 milacres per plot. Milacre stocking of shortleaf pine was calculated from the proportion of milacres in a plot having at least one shortleaf pine. Treatment mean and standard error statistics were calculated for these variables across all plots within a treatment (n=36).

Overstory growth was analyzed by calculating the difference in total merchantable volume (ft^3 per acre), total merchantable green weight (tons per acre), and sawtimber volume (fbm Scribner per acre). Growth was also compared with that predicted over four years for the control and the treated stands using the SLPSS model. Regeneration data were analyzed using analysis of variance methods found in the SAS statistics software (SAS Institute 1990). Mean comparisons among treatments were conducted using the Student-Newman-Kuels mean comparison test.

Results and Discussion

Regeneration Differences by Treatment

Pine milacre stocking was inadequate in all treatments falling below the recommended standard of 300 trees per acre and 30 percent milacre stocking (Guldin and others 2004) and did not differ significantly by treatments (table 1). Milacre stocking varied from slightly more than 10 percent of milacres stocked with shortleaf pine in the B3 treatment to less than 1 percent in the B2 treatment; the control treatment also had 10 percent milacre stocking of shortleaf pine. At this point in the study, there appears to be no

Table 1—Means (and standard error, in parentheses) for milacre stocking and regeneration density by treatment.

Treatment	Milacre stocking -percent-		Regeneration density, all species: All species -stems/acre-		Shortleaf pine -stems/acre-		Oak spp. -stems/acre-		Other hardwood spp. -stems/acre-	
B1	6.48	a	8,245	a	287	a	5,056	a	2,903	a
	(3.73)		(699)		(174)		(812)		(481)	
B2	0.93	a	7,759	a	9	a	4,352	a	3,398	a
	(0.64)		(487)		(6)		(454)		(364)	
B3	10.18	a	7,435	a	324	a	4,148	a	2,963	a
	(2.82)		(596)		(120)		(652)		(641)	
Control	9.30	a	3,106	b	268	a	1,051	b	1,787	a
	(3.08)		(360)		(118)		(153)		(363)	

difference in milacre stocking of pines related to number of growing seasons after burning beneath the residual overstory densities found in this study.

There is a 2.5-fold increase in the number of stems of regeneration of all species as a result of treatment versus no treatment; however, there was no significant difference in total regeneration density one, two, or three years after the most recent burn (table 1). The combined treatment of thinning, midstory reduction, and fire result in this enhanced cohort of regeneration, not the number of growing seasons since the most recent prescribed burn.

This treatment pattern did not exist in the shortleaf pine component. There was no statistically significant difference in shortleaf pine regeneration density by treatment, which parallels the observation about milacre stocking of pines previously observed. Pine regeneration density varied from 324 stems per acre in the B3 treatment to 9.3 stems per acre in the B2 treatment; the control contained 268 stems per acre (table 1).

Significant differences did exist in oak regeneration density between treated stands and the control stands, but not among the treated stands. Treatment resulted in between four and five times as many oaks as were found in the control stands (table 1). Oak regeneration density varied between 4,148 stems per acre in the B3 treatment to 5,156 stems per acre in the B1 treatment, compared to 1,051 stems/acre in the control.

There was a twofold difference in the stem density of other non-oak hardwood species among the four treatments. Stem density of other hardwoods varied from 1,787 stems per acre in the control stand to 3,398 stems per acre in the B2 treatment. However, this difference was not statistically significant (table 1).

The distribution of seedling-origin stems and sprout-origin stems in the regeneration cohort varied between time since last burn versus control, but again not within the burning treatments per se. The control treatment had between three and 10 times the number of seedlings as the burn treatments (table 2), which reinforces the intuitive notion that stems of seedling origin were most likely to accumulate in the absence of burning treatments over time. Conversely, and as expected, the burn treatments resulted in a three- to four-fold increase in the number of sprouts per acre compared to the control (table 2). Although the trends in the burn treatments showed an increased number of seedling-origin stems versus the number of years after treatment, these trends were not significantly different. However, the ratio of seedling-origin to sprout-origin reproduction increased markedly across treatments—roughly speaking, 1:2 in the control, 1:20 in the B3 treatment, 1:40 in the B2 treatment, and 1:100 in the B1 treatment.

Table 2—Seedling-origin versus sprout-origin stems of regeneration by treatment.

	Regeneration density			
Treatment	Seedling-origin -stems/acre-		Sprout-origin -stems/acre-	
B1	83	b	8,162	a
B2	176	b	7,583	a
B3	356	b	7,079	a
Control	1,172	a	1,990	b

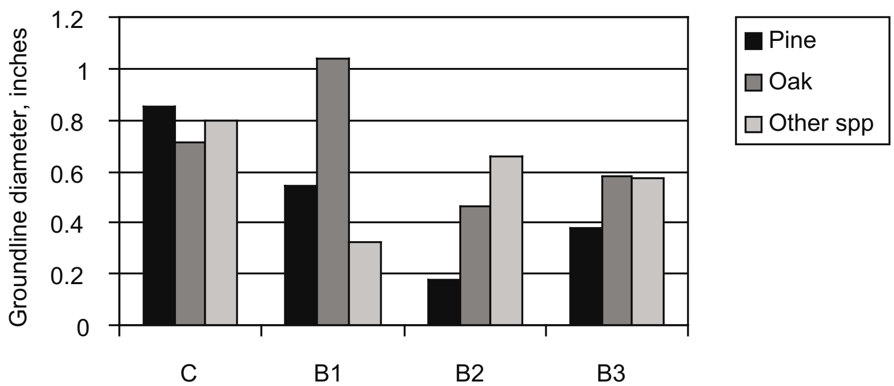

Figure 1—Differences in quadratic groundline diameter by treatment.

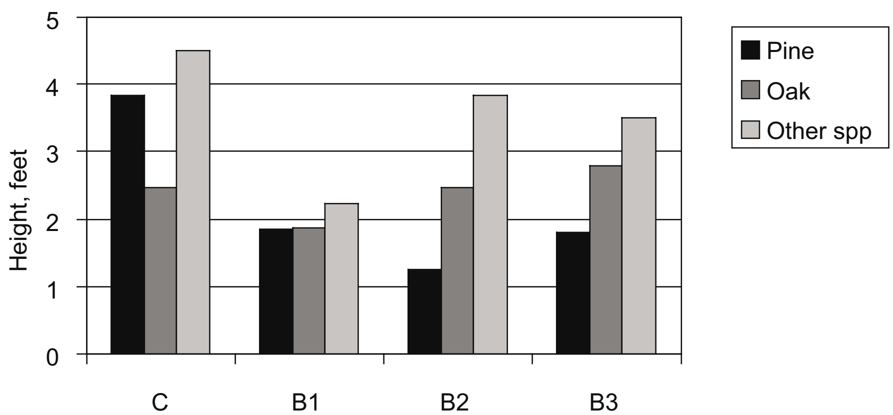

Figure 2—Differences in height by treatment.

In all but a few instances, the average size of the tallest seedling per quadrat was smaller in treated stands compared to the control stands. There was no consistent pattern in groundline diameter by treatment; in pines and oaks, the groundline diameter two years after burning was less than that one year after burning, whereas the groundline diameter was greater two years after burning in the other species category (figure 1). Similarly, height growth patterns suggest that regeneration in the control stands is generally taller than in treated stands, especially in the pine and other species component (figure 2), but trends among burn treatments are not clear. As the study matures, growth data from tagged trees measured over time will provide a better impression of regeneration development after burning.

These data are obscured by the number of observations of overstory basal area that lie beyond the range where one might expect to observe regeneration. A subset of the original data set was created that excluded all control

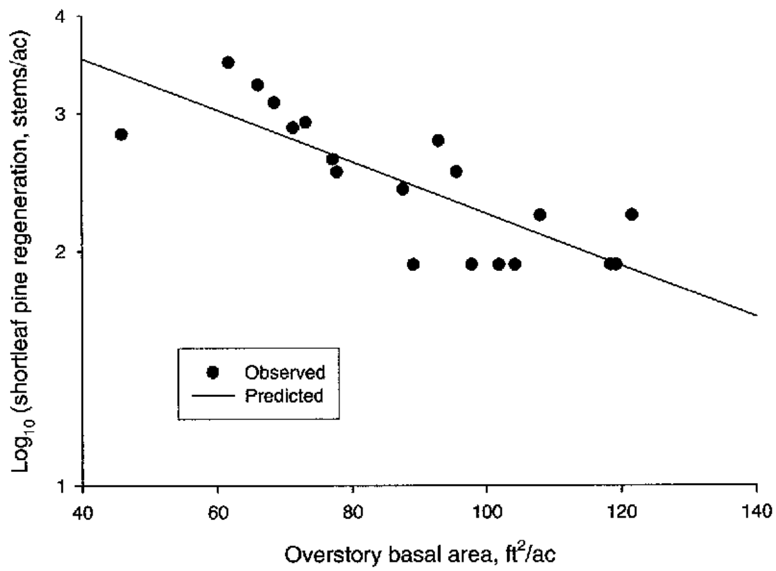

Figure 3—Relationship between overstory basal area and pine regeneration density for treated plots in the database in which milacre stocking is greater than zero.

plots and all treated plots if all 12 milacres had no pine stocking. When subject to simple linear regression, a log transformation of pine stems per acre was strongly correlated with total overstory basal area (figure 3). The predicted equation was

Log10 (SLPR) = 4.17172 – 0.01907 * (Total BA)
where
SLPR=shortleaf pine regeneration, stems/acre;
Total BA = basal area for all species, ft^2/acre; and
R^2(adj) = 0.622, Pr>F<0.0001.

Overstory Growth

After four growing seasons, there were no significant differences in volume growth between the B1 and B3 treatments and the control (table 3). Total merchantable cubic volume growth in the stands varies from -4 ft^3 per acre in the B1 treatment to 70 ft^3 per acre in the control over a four-year period. In green weight, change in tonnage varied from a slight decline in the B1 treatment to a 2.4 ton per acre increase in the control. Scribner board-foot volume changes over four years varied from an increase of 320 fbm per acre in the B1 treatment to 846 fbm per acre in the control stands.

However, growth in either control or treated stands does not appear to be at a level one would expect (figure 4). For example, the Scribner board-foot volume growth in the three treatments that is predicted using the SLPSS growth model varies from 480 to 500 fbm per acre annually. Observed

Table 3—Change in volume over four years by treatment.

Treatment	Total merch. cubic volume growth *-ft³/acre-*		Total merch. green weight growth *-tons/acre-*		Scribner volume growth *-fbm/acre-*	
B1	-4.7	a	-0.2	a	320	a
B3	16.3	a	2.4	a	536	a
Control	70.7	a	6.8	a	838	

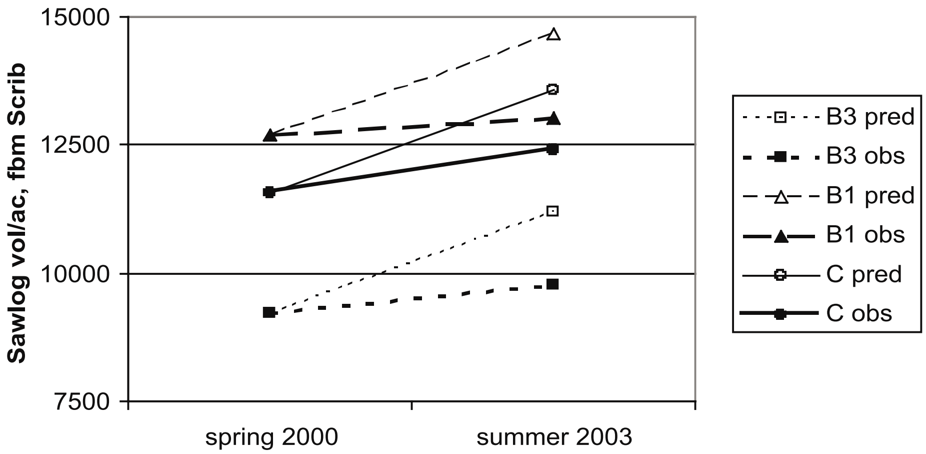

Figure 4—Board-foot volume growth, fbm Scribner per acre, in two treatments and the control. Projected growth was determined using a four-year projection with the SLPSS growth model.

growth is about 70 percent less than expected, and even the measured growth in the control stand was less than half that expected under application of the growth model.

Conclusions

Minimum stocking guidelines for shortleaf pine regeneration (Guldin and others 2004) call for 200 stems/acre and 20 percent of milacres stocked in uneven-aged stands, and 300 stems per acre and 30 percent of milacres stocked in even-aged stands. By either of these standards, obtaining a reliable cohort of advanced regeneration of shortleaf pine as a result of simply treating these stands with the traditional shortleaf pine-bluestem treatment is inadequate. Supplemental treatment of some kind will be necessary to obtain an adequate cohort of pine regeneration suitable for long-term sustainability of the stand.

One possible reason for the inadequate result under the current stands is that overstory density is greater than that required to promote the establishment of pine regeneration. When a subset of treated stands was analyzed separately based on having at least one milacre stocked with shortleaf pine per plot, significant relationships were obtained between pine milacre stem density and overstory basal area.

With lower basal areas, additional work will be needed to determine sprouting potential of different sizes of shortleaf pine regeneration, and whether the three-year burning interval must be lengthened to allow pine seedlings to grow large enough such that they will not be top-killed when the prescribed fire program is reestablished.

Additional research is required to determine the effects of treatments on stand growth. Over a four-year period, growth in either treated or control stands is substantially less than that predicted from growth models developed in this forest type. However, there are no significant differences in growth over four years between treated stands and the control stands. The growth model was developed in unburned second-growth shortleaf pine stands and may not reflect the influence of burning. In addition, a large sample of treated and untreated stands will be needed to provide a better assessment of the impacts of this restoration treatment on stand growth.

As these studies mature, the tracking of stand conditions over time will allow for better determination of whether shortleaf pine can be accumulated as advance regeneration under these stands. Ultimately, additional research will be needed to determine what combination of delay in the burning treatment and reduction in overstory basal area will result in an effective advance growth seedling bank of shortleaf pine in these restored pine-grass habitats.

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The North American Long-Term Soil Productivity Experiment: Coast-to-Coast Findings From the First Decade

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Abstract—*The National Forest Management Act of 1976 mandates that a site's productive capacity must be protected on federally managed lands. Monitoring the effects of management on a site's productive capacity is not easy, and in 1989 a national program of Long-Term Soil Productivity (LTSP) research was established to assist National Forests toward this end. The LTSP program focuses on disturbances associated with timber harvest, but findings apply to any activities altering vegetation or soil. LTSP centers on core experiments that manipulate site organic matter, soil porosity, and the complexity of the plant community. Results from a dozen decade-old LTSP installations in the Sierra Nevada and the Southern Coastal Plain do not indicate that site productivity has been impaired despite substantive soil compaction and massive removals of surface organic matter. The strongest effect of treatment on planted tree growth on sites governed by temperate and subtropical climates was the control of competing vegetation. With only one-fifth of the LTSP installations reporting, findings should not be generalized to other sites and climates.*

Introduction

The Long-Term Soil Productivity (LTSP) study began in 1989 as a "grass roots" proposal that grew to a national program of the USDA Forest Service. LTSP was founded to examine the long-term consequences of soil disturbance on fundamental forest productivity through a network of designed experiments. The concept caught the imagination of other resource managers and scientists, and partnerships and affiliations soon were forged among public and private sectors in the United States and Canada. Today, more than 100 LTSP and affiliated sites comprise the world's largest coordinated research network addressing basic and applied science issues of forest management and sustained productivity.

Background

Historical Basis

The LTSP program began in response to the National Forest Management Act of 1976 (NFMA) and related legislation (USDA Forest Service 1983). NFMA requires the U.S. Secretary of Agriculture to ensure, through research and monitoring, that forest management practices do not permanently impair the productivity of the land. This requirement seems superfluous because sustaining productivity is an obvious aim of modern forest management and has been a Forest Service goal since the agency was founded. It is remarkable only in that NFMA may be the world's first modern mandate for a forest land ethic that carries the weight of law.

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Responding to NFMA, an independent committee of scientists was appointed to form a framework for implementing the law. Their recommendations led in 1985 to a statement of responsibilities surrounding federal land management activities (Code of Federal Regulations 1985). One notable element was that the Forest Service must monitor the effects of forest management prescriptions, including "significant changes in land productivity." This monitoring requirement was developed more than a decade in advance of The Montreal Process (Canadian Forest Service 1995) and the environmental surge toward "green certification" (Anonymous 1995).

The Forest Service knew that clear and objective definitions were key to addressing its monitoring charge. "Land productivity" was a central issue. Broadly, it could be defined as a site's capacity to produce a cornucopia of timber, wildlife, watershed, fishery, and aesthetic values. All these values are legitimate expressions of land productivity, but some are less tangible, more subjective, and more variable temporally than others. Instead, and with guidance from the U.S. Office of General Council, a fundamental definition was forged. Land productivity was defined as the carrying capacity of a site for vegetative growth. This was useful, because the capacity of a site to capture carbon (C) and grow vegetation is central to its potential for producing all other values. Given the vagaries of annual fluctuations in dry matter production, consensus held that a departure from baseline would have to exceed 15 percent to be deemed significant (USDA Forest Service 1987). But what variables should be monitored?

The National Forest Approach

Trying to measure the productive potential of a site directly by assaying trends in tree or stand growth is fraught with frustrations and uncertainty because trends vary with stand age, structure, stocking, treatment history, and the lack of reference controls (Powers 2001). Consequently, soil-based indices have been proposed as more objective measures of a site's productive potential (Burger 1996, Powers and others 1990). The USDA Forest Service also saw the value in soil properties as an independent basis for monitoring potential productivity. In 1987 the Watershed and Air Management division of National Forest Systems adopted a program of soil quality monitoring that was based on the following rationale (Powers and Avers 1995):

- Management practices create soil disturbances.
- Soil disturbances affect soil and site processes.
- Soil and site processes control site productivity.

Monitoring soil and site processes directly is not feasible. Instead, the Forest Service proposed a monitoring strategy based on measurable soil variables that either reflect, or are correlated with, important site processes. Accordingly, each Forest Service Region has developed threshold monitoring standards for soil quality reflecting state-of-the-art knowledge and professional judgment (Page-Dumroese and others 2000; Powers and Avers 1995; Powers and others 1998). Threshold standards are meant to detect when significant changes have occurred in potential productivity at a statistical confidence of ± 15 percent of the true site mean. These standards await validation and are updated as findings accrue from research. Unfortunately, correlations between soil monitoring variables and potential productivity are mainly conceptual. Because they are conceptual and somewhat subjective, they can be challenged.

Research Coordination

Recognizing the difficulty inherent in developing soil quality monitoring standards based partly on professional judgment, the National Forest System (NFS) of the Forest Service asked Forest Service Research for assistance. A small but seasoned team of scientists and practitioners assembled informally in 1988 to address the problem. Extensive review of world literature revealed that two ecosystem properties most likely to impact long-term productivity were site organic matter and soil porosity. While these site and soil properties were seen clearly as of paramount importance, we concluded that existing information was sparse, site specific, often contradictory, and too anecdotal to be broadly useful. More fundamental work was needed, and we proposed a nationally coordinated field experiment to address the issue directly and unambiguously. The proposal was reviewed internally by leading Forest Service scientists and professionals, and both nationally and internationally by research scientists outside the agency. We believe that this was the most widely reviewed research study plan ever produced by the Forest Service. A final study plan was prepared (Powers and others 1989). The plan was approved as a national effort in 1989 by the Deputy Chiefs for Research and National Forest Systems in Washington, D.C., and 10-year funding was secured for implementing the study on public lands. The overview was published and circulated widely (Powers and others 1990, 1996; Powers and Avers 1995).

Partnerships

The first LTSP installation was established in 1990 on the Palustris Experimental Forest in the loblolly pine (*Pinus taeda*) forest type of the Louisiana Coastal Plain. The following year saw units established in the mixed conifer (*Abies/Pinus/Pseudotsuga*) forest of California's Sierra Nevada and in the glacial till landscape of Minnesota's aspen (*Populus deltoides/tremuloides*) forest. The experiment then expanded to other sites and Life Zones. As the LTSP program gained momentum it drew widening attention. British Columbia's Ministry of Forests adopted the LTSP concept in 1990 as a high priority program for Interior British Columbia (Hope and others 1992). Two installations were established by 1994 and several more followed (Holcomb 1996). Independently, the Canadian Forest Service began experiments in Ontario that closely paralleled the LTSP design, and the two programs merged in 1996 to expand the network. Today, the total number of installations with the core design stands at 62 (figure 1).

In the United States, forest industry voiced concern that the experiment highlighted only "negative" impacts of management and that LTSP lacked treatments aimed at enhancing site productivity. Accordingly, we invited leaders from private and public forest management groups to a 1995 working session in St. Louis, Missouri, to air concerns and to find ways of improving the study and strengthening the network. This led to an expanded affiliation that included studies on industrial lands and elsewhere. Conditions for affiliation are that (1) studies have certain elements in common with the LTSP experimental design (at least the minimal potential impact treatment), (2) treatment plots be large enough to have minimal edge effect once plots attain leaf area carrying capacity, and (3) members agree to share findings and provide mutual support (Powers and others 1996). These affiliate sites have brought the LTSP network to more than 100 installations (figure 1), making it the world's largest coordinated effort aimed at understanding how pulse disturbances affect sustained forest productivity.

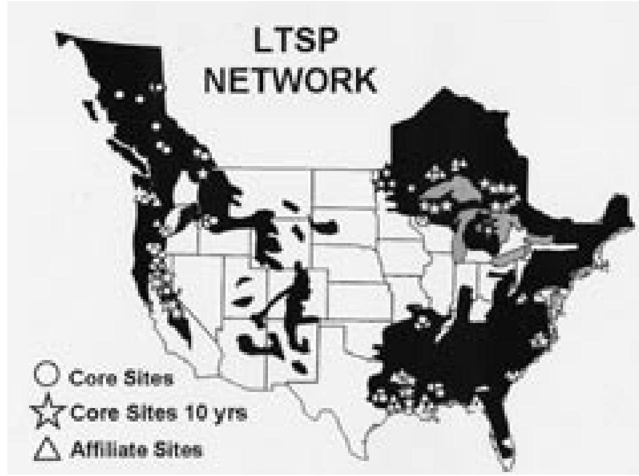

Figure 1—Location of core LTSP and affiliate installations on the approximate range of the commercial forest in the United States and two Canadian provinces. Stars indicate installations achieving at least 10 years of growth.

The Study

A Conceptual Model

The LTSP program is predicated on the principle that within the constraints of climate, a site's potential net primary productivity is strongly regulated by physical, chemical, and biotic soil processes affected readily by management. The key properties directly affected by management are soil porosity and site organic matter (OM). These two properties regulate critical site processes through their roles in microbial activity, soil aggregate stability, water and gas exchange, physical restrictions on rooting, and resource availability (figure 2).

Regardless of silvicultural strategy or harvest intensity, site organic matter and soil porosity are impacted directly by forest management operations. Therefore, they were targeted for specific manipulation in large-scale, long-term experiments meant to encompass the range of possibilities occurring under management. The experiments were designed to address these four hypotheses:

Null hypothesis	Alternative hypothesis
1. Pulse changes in site organic matter and/or soil porosity do not affect the sustained productive potential of a site (sustained capacity to capture carbon and produce phytomass).	Critical changes in site organic matter and/or soil porosity have a lasting effect on potential productivity by altering soil stability, root penetration, soil air, water and nutrient balances, and energy flow.
2. If impacts on productivity occur from changes in organic matter and porosity, they are universal.	The biological significance of a change in organic matter or porosity varies by climate and soil type.
3. If impacts do occur, they are irreversible.	Negative impacts dissipate with time, or can be mitigated by management practices.
4. Plant diversity has no impact on the productive potential of a site.	Diverse communities affect site potential by using resources more fully or through nutrient cycling changes that affect the soil.

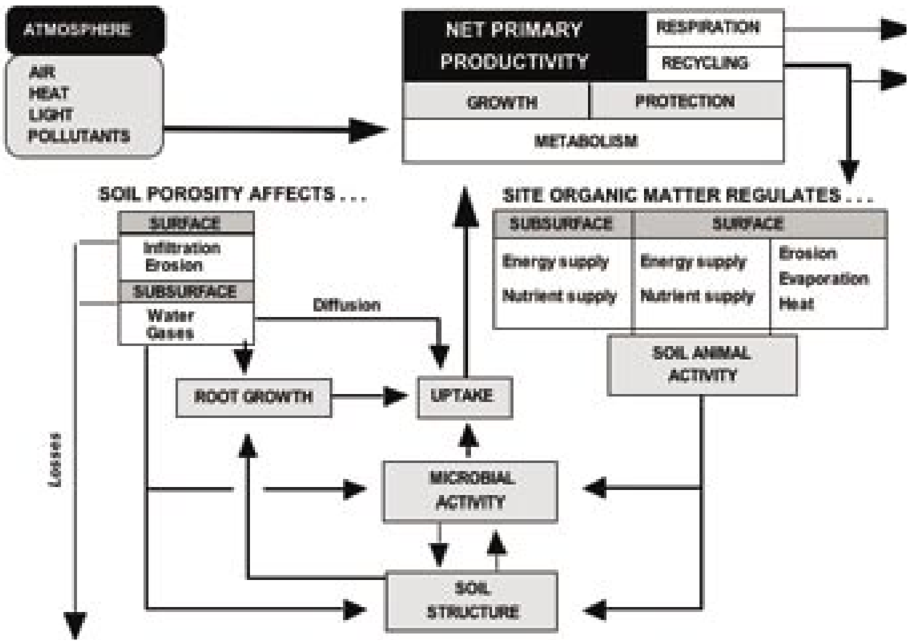

Figure 2—Conceptual model of the influence of site organic matter and soil porosity on fundamental site processes that regulate primary productivity.

Selecting Sites and Applying Treatments

The study was targeted at forest types, age classes, and soil conditions apt to fall under active forest management involving harvesting, thinning, or fuel modification. These were fully stocked, young-growth, even-aged stands—i.e., not "ancient forests" or non-forested openings. Preliminary plots of 0.2 or 0.4 ha were identified and surveyed for variability in soil and stand conditions. Those with comparable variability at a given location (similar soil type, stand density, and amounts of disturbance) were chosen for the experiment. Pretreatment samples were taken to quantify standing biomass and nutrient capital in the overstory, understory, and forest floor. Stands were then harvested under close supervision and treatments were imposed randomly. The main effect treatments were as follows:

Main effect	Symbol	Description of treatment
Modify site organic matter	OM_0	Tree boles removed. Retain crowns, felled understory, and forest floor.
	OM_1	All aboveground living vegetation removed. Forest floor retained.
	OM_2	All surface organic matter removed. Bare soil exposed.
Modify soil porosity	C_0	No soil compaction.
	C_1	Compact to an intermediate bulk density.
	C_2	Compact to a severe bulk density.

We had two reasons for choosing these levels of organic matter manipulation. First, they encompass the extremes in organic matter removal likely under any silvicultural system short of removing surface soil or extracting roots. Second, they produce a step series of nutrient removal that is disproportionate to biomass loss. Table 1 illustrates these points using six typical LTSP sites arrayed along a climatic gradient. It shows that overstory trees contain roughly 80 percent of site aboveground organic matter with about

Table 1—Absolute and proportional amounts of biomass and nitrogen removed by the three organic matter treatments on representative LTSP sites (OM_0 = bole only removed, OM_1 = whole tree removed, OM_2 = whole tree + understory and forest floor removed). Life zone codes after Holdridge (Lugo and others 1999); *BM* = boreal moist, *CTM* = cool temperate moist, *WTD* = warm temperate dry, *WTM* = warm temperate, moist, *STM* = subtopical moist.

			Biomass removed (Mg/ha) (% of above ground total)			Nitrogen removed (kg/ha) (% of above ground total)		
Location	**Life zone**	**Forest type**	**OM_0**	**OM_1**	**OM_2**	**OM_0**	**OM_1**	**OM_2**
British Columbia	*BM*	Subboreal spruce	126 *(56)*	158 *(71)*	223 *(100)*	195 *(18)*	253 *(24)*	1,068 *(100)*
Minnesota	*CTM*	Trembling aspen	175 *(61)*	214 *(75)*	286 *(100)*	194 *(30)*	316 *(48)*	653 *(100)*
Idaho	*CTM*	Mixed conifer	160 *(61)*	191 *(73)*	261 *(100)*	190 *(22)*	410 *(48)*	846 *(100)*
California	*WTD*	Mixed conifer	252 *(47)*	473 *(89)*	532 *(100)*	218 *(20)*	609 *(57)*	1,064 *(100)*
Missouri	*WTM*	Central hardwood	96 *(42)*	175 *(77)*	228 *(100)*	195 *(24)*	540 *(67)*	811 *(100)*
Louisiana	*STM*	Loblolly pine	133 *(77)*	153 *(88)*	173 *(100)*	134 *(38)*	229 *(65)*	352 *(100)*

two-thirds occurring in boles. At best, the forest floor accounts for only one-fourth of aboveground organic matter.

Nitrogen (N) shows a different trend. Although half or more of aboveground organic matter may be in tree boles, this accounts for only one-fifth to one-third of the aboveground N capital. On average (and in the absence of frequent disturbance), the forest floor of mature stands contains as much N as boles and crowns, combined. However, the actual proportion of aboveground N in the forest floor varies with climate (table 1). In moist boreal forests of British Columbia where decomposition is slowed by cool temperature and perhaps by partial anaerobia, the forest floor accumulates far more N than is contained in the vegetation. Under warm, humid conditions, the forest floor decomposes rapidly and is a relatively low reservoir of N. Regardless of Life Zone, the understory in mature forests is only a minor component of site organic matter or N (only a few percentage points of the aboveground total after canopies have closed).

Compaction was accomplished through multiple passes of heavy machinery to achieve target levels of soil bulk density varying by soil texture (Daddow and Warrington 1983). Organic removal was accomplished by full suspension of boles or crowns, or by manually raking the forest floor from the plot to expose mineral soil. Experimental treatments were not meant to mimic operational practices, but rather to bracket the extremes in disturbance likely to occur under present or future management. Generally, all factorial combinations of main effect treatments were applied, producing nine core combinations of organic matter removal and soil compaction. Treatment plots (0.4 ha) were separated from residual stands by a distance at least equivalent to the height of bordering trees. This plot size and separation avoided competitive edge effects that could mask the true impact of the treatments, a confounding factor that affects small plot studies and many historical investigations (Powers and others 1990, 1994). Only rarely were treatments replicated at a given location. High establishment costs (about $60 thousand per set of 9 treatments) and the need to generalize findings across a broad ranges of sites convinced us that the better approach was to replicate the experiment within particular soil types (soil Series of Families) but at geographically separated locations. Soil types were chosen based on their regional prevalence and on their position along a continuum of site productivity within a regional forest type. In California for example, three installations occur on each of three soil types representing low, medium and high levels of productivity (nine in all), and another three installations occur on unreplicated soil types representing levels of productivity between the extremes. Only a few installations were established in a given year and replicates in a given soil type sometimes were established in different years. In California, three installations (Central, Owl, and Vista) are replicates of a particular soil Series or Family, but Challenge and Wallace are not (table 2). The LTSP study is planned to extend several decades to at least the culmination of mean annual volume increment. Only those achieving 10 years from treatment are reported here.

Plots were regenerated with the tree species indigenous to the site and measurement trees were separated from outer plot boundaries by several rows of buffer trees. Except for aspen (*Populus*) forests and the mixed conifer sites of interior British Columbia where policy precluded herbicides, all main effect treatment plots were split. One half of each plot was kept weed-free by regular applications of herbicides, and the other half was allowed to develop naturally (thereby producing side-by-side subplots with simple and diverse forest communities). Where possible, the more severe treatments were applied and followed by mitigative measures, such as fertilization to replace nutrients and subsoiling to alleviate compaction. Each field installation was equipped with an automated climatological monitoring station, thereby linking all sites in a network characterized by precipitation, temperature, solar radiation, and relative humidity.

Post Treatment Measurements

Although many measurements could be taken, principal investigators agreed that a reduced set of eight core measurements were critical to the success of the LTSP program. Beyond treatment establishment, funds were extremely limited. Therefore, minimum measurement intervals were identified for each variable:

Measurement variable	Minimum measurement interval
Climatological data	Continuous.
Soil moisture and temperature	Monthly.
Soil bulk density	Each 5 years.
Soil strength	Seasonally each 5 years.
Soil organic matter content and chemical composition	Each 5 years.
Water infiltration and saturated hydraulic conductivity	Each 5 years.
Plant survival, growth, damage from pests, NPP	Each 5 years.
Foliar chemistry and standing nutrient capital	Each 5 years.

Methods for estimating growth and net primary productivity (NPP) were left to the discretion of each principal investigator, but generally they involved periodic destructive sampling within the treated buffer. While early findings have been reported for individual sites (Alban and others 1994; Amaranthus and others 1996; Tiarks and others 1998; Powers and Fiddler 1997; Stone and Elioff 1998), most have dealt with stand conditions short of crown closure and may not be indicative of long-term trends when sites are stocked at carrying capacity. This paper constitutes the first effort at summarizing findings from installations that have reached 10 growing seasons. It highlights installations in two geomorphic provinces with differing climates: the Sierra Nevada of California, and the Southern Coastal Plain (table 2). Analyses are principally of two types: analysis of variance and least squares regression via standard procedures.

Table 2—Site and pretreatment stand characteristics of LTSP installations achieving 10 years of growth. Life zone codes after Holdridge (Lugo and others 1999); *BM* = boreal moist, *CTM* = cool temperate moist, *WTD* = warm temperate, dry, *WTM* = warm temperate, moist; *STD* = subtropical dry, *STM* = subtopical moist. Nd = information not determined or not available.

Location	Installation name	Life zone	Forest type	Elev (m)	ppt. (cm)	Soil origin	Stand age (yr)	Preharvest biomass (kg/ha) Overstory	Understory	FF
California	Central	*WTD*	Mixed conifer	1685	114	Granodiorite	117	422,111	94	80,455
California	Challenge	*WTD*	Mixed conifer	790	173	Metabasalt	108	473,348	576	60,926
California	Owl	*WTD*	Mixed conifer	1805	114	Granodiorite	115	576,071	34	72,233
California	Vista	*WTD*	Mixed conifer	1560	76	Granodiorite	132	373,609	43	72,567
California	Wallace	*WTD*	Mixed conifer	1575	178	Volcanic ash	230	450,193	83	115,757
Idaho	Priest River	*CTM*	Mixed conifer	900	85	Volcanic ash	120	191,250	1,750	68,000
Louisiana	Glenmora	*STD*	Pine-hardwoods	61	147	Marine sediments	52	153,000	4,200	15,900
Louisiana	Malbis	*STD*	Pine-hardwoods	52	150	Marine sediments	45	91,000	5,100	Nd
Louisiana	Mayhew	*STD*	Pine-hardwoods	61	147	Marine sediments	55	236,200	1,700	15,400
Louisiana	Metcalf	*STD*	Pine-hardwoods	61	147	Marine sediments	55	203,200	1,800	20,500
North Carolina	Croatan	*WTM*	Pine-hardwoods	7	136	Marine sediments	65	167,800	3,190	52,410

Findings to Date

Findings reported too hastily can be misleading. While a decade may seem a long observational period for many studies, we have resisted making a hasty synopsis of cross-site comparisons. Even at 10 years, crown canopies have not closed on many treatment plots. However, we believe that oscillations from initial perturbations have dampened enough to give us an early glimpse of longer-term trends. We confine our analyses to simple responses of soil and vegetation to the main effect treatments on our oldest installations for which data are available, those from the Southern Coastal Plain and Sierra Nevada—two regions contrasting greatly in climate and geology. Our analyses carry the caveat that trends may change when data are available from all LTSP installations.

Organic Matter

Productivity

We tested the hypothesis that site organic matter removal affects forest productivity by comparing total standing biomass at 10 years for 12 sites, five from the Sierra Nevada and seven from the Southern Coastal Plain. Planting through logging slash sometimes reduces tree survival. Therefore, we based our analyses on total standing biomass (planted trees plus understory vegetation) on non-herbicide plots. Total vegetative production reflects site potential more fully, particularly where tree stocking has not reached site carrying capacity.

Removing all surface organic matter prior to planting had no general impact on total vegetative production at 10 years, regardless of geographic province (figure 3). The linear trend determined by regression suggests that removing surface organic matter reduces productivity more on poorer sites than on better, but the intercept is not significantly different from zero ($p = 0.33$) and the slope trend is not significantly different from 1.0 ($p = 0.62$).

Soil Chemistry

Data from the seven Coastal Plain sites indicate that organic matter removal had negligible impact on the concentration of organic C in the

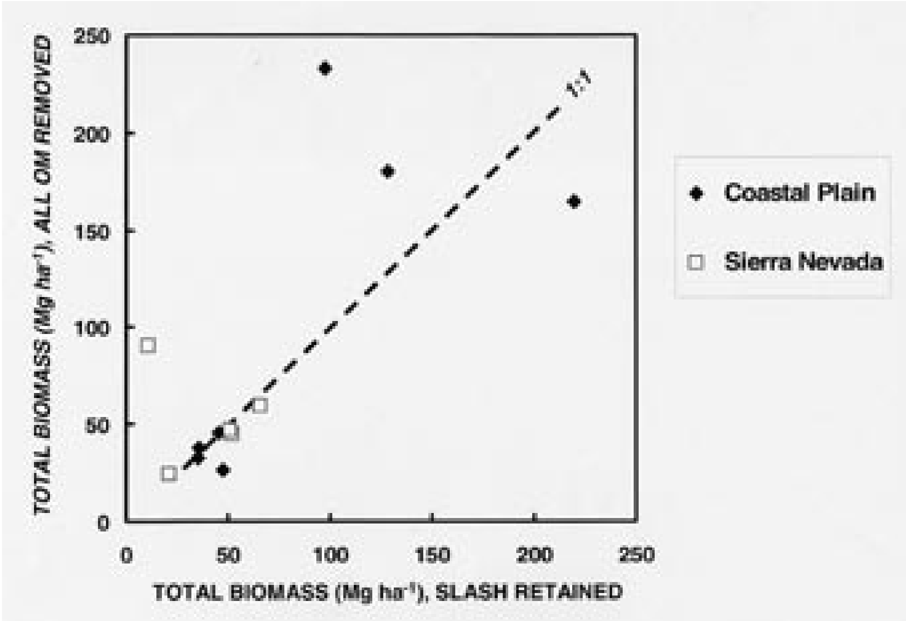

Figure 3—Standing biomass of trees and understory vegetation at 10 years as influenced by the retention or removal of organic surface residues (no soil compaction). Dashed line indicates 1:1 parity between treatments. Basis: 12 sites in California and the Southern Coastal Plain. (OM = organic matter.)

upper soil profile (figure 4). Nor did removing surface organic matter have any apparent effect on the mass of C or N in the upper soil profile at 10 years. Analysis of variance for soil C and N content on the three North Carolina installations replicated on the Croatan National Forest detected no significant effect among organic matter removal treatments (table 3). Yet, when the same soils were analyzed for organic C concentrations before treatment and at time intervals thereafter, post-treatment concentrations were greater at all depths than initial values (figure 5). This was true at all depths, even where all surface organic matter had been removed.

This presents a curious and seemingly contradictory point. On one hand, surface organic matter removal seemed to have no obvious effect on soil C storage at 10 years. On the other hand, soil carbon concentrations significantly increased following harvest. The explanation for this lies in the primary source of soil organic C. Apparently, soil inputs following disturbance depend less on decomposition of surface residues and more on the decay of fine roots that remained from the previously harvested stand. This conclusion is supported by work elsewhere. In a Tennessee study more than a decade after harvesting a mixed-hardwood forest, Johnson and Todd

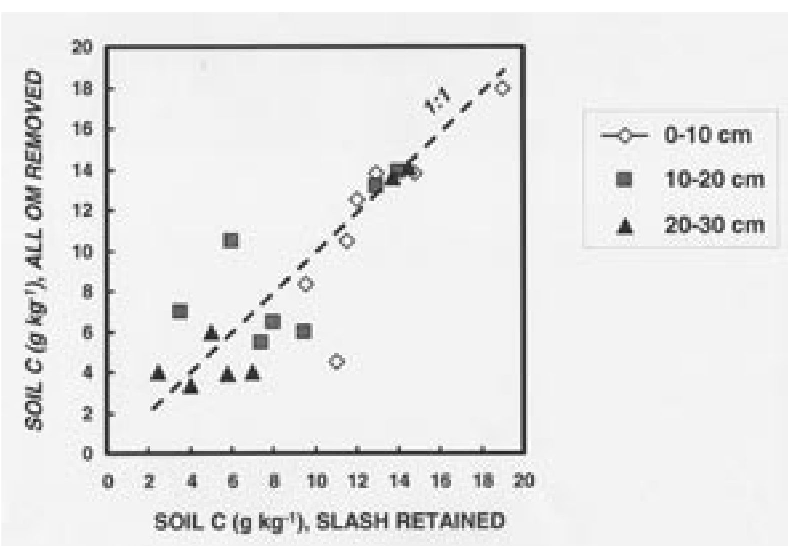

Figure 4—Concentration of organic soil carbon at 10 years for three soil depths as influenced by the retention or removal of organic surface residues. Dashed line indicates 1:1 parity between treatments. Basis: seven sites in the Southern Coastal Plain.

Table 3—Influence of organic matter removal on soil organic carbon (C) and nitrogen (N) 10 years after treatment at the North Carolina LTSP sites. Statistical significance of differences among treatments indicated by $p > F$.

Soil depth (cm)	Organic matter removal OM_0	OM_1	OM_2	$p > F$
	Organic C ($Mg\ ha^{-1}$)			
0-10	28.4	33.3	33.8	*0.21*
10-20	21.4	23.5	25.1	*0.61*
20-30	14.0	25.1	16.5	*0.69*
	Total N ($kg\ ha^{-1}$)			
0-10	807	905	882	*0.85*
10-20	542	524	540	*0.99*
20-30	352	385	368	*0.94*

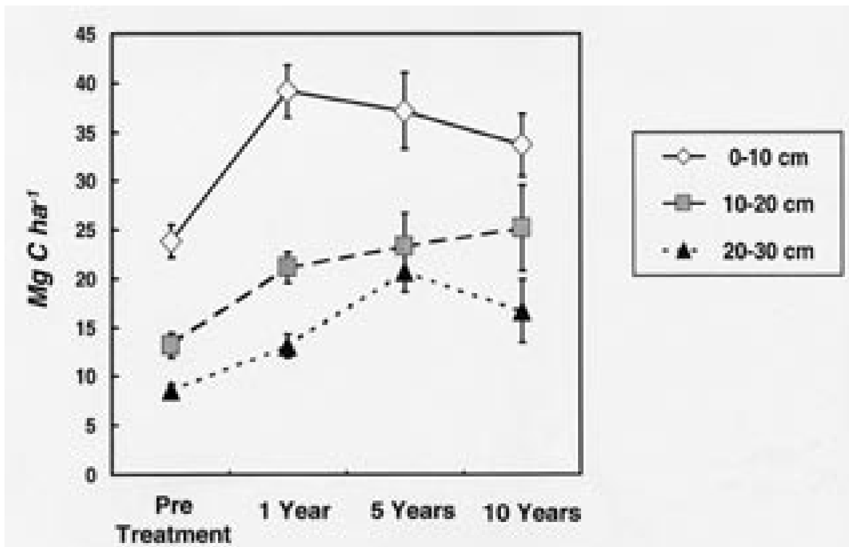

Figure 5—Quantity of fine fraction organic soil carbon stored at three soil depths before and after the OM_1 treatment on the Croatan LTSP site in North Carolina. Vertical bars indicate one standard error of the mean. Trends were similar among all OM treatments.

(1992) found no differences in soil organic matter beneath previous piles of logging slash and units free of slash. Evidently, under moderate and warmer climates, C is respired as CO_2 as surface residues decompose, and very little C is incorporated into the soil beyond. In their work on California soils similar to our California LTSP sites, McColl and others (1990) showed that dissolved organic C from mature forests contributed less than 1 Mg C ha^{-1} yr^{-1} to the mineral soil—only a fraction of the increases we found (figure 5).

On the other hand, fine roots decaying from harvested stands provide sizable C inputs in fractions small enough to pass a conventional 2 mm sieve. Van Lear and others (2000) found that soil C concentrations were more than an order of magnitude greater in the vicinity of roots remaining from a stand harvested 16 years earlier than in the general soil. The effect was evident to as much as a meter depth. Root decay apparently follows a simple Q_{10} model of rate increasing with temperature (Chen and others 2000), and should be quite rapid in soils of the warm, humid Southern Coastal Plain and in those dominated by a Mediterranean climate. We conclude that organic C from surface residues (logging slash, understory vegetation, and forest floor) most likely is respired as CO_2 during decomposition and contributes relatively little to soil C. And while organic N mineralized to ammonium during decomposition presumably is released to the soil, either it is immobilized quickly, nitrified and leached, or is too miniscule relative to organic N to be detected through conventional analysis (table 3).

Soil Compaction

Soil compaction effects on productivity through the first 10 years were assayed by comparing total standing biomass (trees plus understory vegetation) on C_0 (not compacted) and C_2 (severely compacted) treatments (figure 6). Organic matter treatment was held constant at OM_2 (complete removal) to eliminate the possibility of compaction x organic matter interactions. The regression trend suggests that in general, soil compaction leads to slightly greater productivity, but the slope of the linear trend is not significantly different from 1.0 ($p = 0.22$) and the intercept is not significantly different from zero ($p = 0.82$). We conclude that soil compaction in our most extreme treatments did not significantly or universally affect total vegetative productivity on sites in the Sierra Nevada and Southern Coastal Plain.

But findings may be biased if trees on compacted soils have lower understory competition, or if soil texture is such that both understory and overstory growth are increased by compaction as was reported by Powers and Fiddler (1997). We found that understory biomass was 55 percent greater on plots not compacted ($p = 0.08$), although this was not so on soils with a sandy texture where biomass tended to be greater on compacted soil. To reduce possible confounding, comparisons also were made of tree biomass for C_0 and C_2 treatments on plots kept free of understory vegetation. Even so, 10-year tree biomass on C_0 and C_2 plots were identical ($Y = 1.94 + 1.00X$, *adj.* $r^2 = 0.78$). Data for plots free of understory competition (open squares) are superimposed on figure 6. We found no evidence that 10-year productivity was universally impacted by soil compaction, regardless of the presence or absence of understory vegetation.

Given that soil compaction generally is believed to reduce tree growth, this result is surprising. One explanation for the lack of an overall soil compaction effect might be that our treatments did not reach compaction levels considered to be severe. To examine this, we calculated soil bulk density immediately following severe compaction as a function of bulk

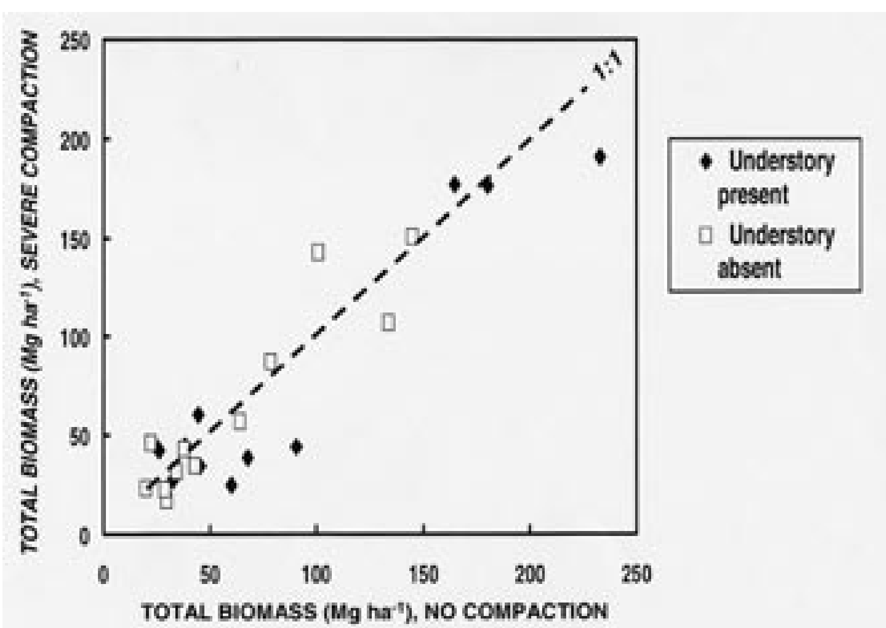

Figure 6—Effect of severe soil compaction on total standing biomass at 10 years (OM_2 treatment). Filled diamonds indicate the biomass of trees + understory vegetation where understory vegetation was present. Open squares indicate tree biomass where understory vegetation was absent. Neither trend differs significantly from a 1:1 relationship, indicating that severe soil compaction had no general effect on productivity.

density immediately before compaction for the 10-20 cm depth zone over a broad range of sites in the LTSP network. The trend was strongly linear of the form:

$$Y = 0.426 + 0.788X \qquad \text{[Eq. 1]}$$
$$r^2 = 0.95$$

Where:

X = soil bulk density in Mg m^{-3} at 10-20 cm before soil compaction

Y = soil bulk density at 10-20 cm in the first year following severe compaction.

r^2 = the proportion of variation in Y explained by the linear relationship.

This indicates that the degree to which soil bulk density was increased by compaction depends strongly on the initial bulk density. That is, soils with low initial bulk densities were compacted more than soils where bulk densities already were high. It also suggests that soil with an initial bulk density of 1.99 Mg m^{-3} can not be compacted further through the procedures we employed.

Soil compaction occurs at the expense of larger pores, resulting in the loss of aeration porosity (Siegel-Issem and others, in press). This means that soils compacted further from a very high initial bulk density may lose air-filled pore space and the soil may become waterlogged or suffer from the buildup of respiratory gases. Grable and Siemer (1968) suggest that an aeration porosity of 10 percent is a critical limit for root respiration and growth. Although we did not measure pore size distribution on most of our soils, we can solve for approximate total porosity by assuming a soil particle density of 2.65 Mg m^{-3}. Using Eq. 1 above, and solving for the bulk density at which no further compaction is possible by the means we used, we can infer that the "uncompactable porosity" remaining at a bulk density of 1.99 is 24 percent, and that this essentially defines the micropores remaining after practically all air-filled porosity has been depleted.

The highest bulk densities we achieved on any depth for any of the sites in figure 6 were in the range of 1.65 to 1.71 Mg m^{-3} (Louisiana). Based on the simple approximations above, this translates to a total porosity between 38 and 35 percent immediately following compaction, for an estimated aeration porosity of between 14 and 11 percent once microporosity is subtracted. This suggests that aeration porosity following severe compaction on the Louisiana sites remained just above the 10 percent threshold proposed by Grable and Siemer (1968). Greenhouse studies have shown that loblolly pine can grow reasonably well even under waterlogged conditions (Siegel-Issem and others, in press). This is probably because of the presence of aerenchyma cells allowing gas exchange between roots and the aboveground atmosphere.

Another possibility explaining the absence of a clear impact of soil compaction on productivity is that soils may have recovered quickly from the initial effects of compaction. We tested for recovery by comparing soil bulk densities at 10-20 cm in the first year after severe compaction treatment with those on the same plots after 10 years. Figure 7 indicates that recovery in that period has been negligible at soil depths below 10 cm.

We conclude that despite appreciable increases in soil bulk density, particularly on lower density soils, compaction has not affected productivity in a general sense over the first 10 years. In our view, the most likely explanations concern the facts that (1) soil compaction may improve soil water availability on droughty sites (Gomez and others 2002); (2) the highest soil bulk densities were associated with loblolly pine sites, a species that tolerates high bulk densities and poorly drained conditions (Siegel-Issem and others, in press);

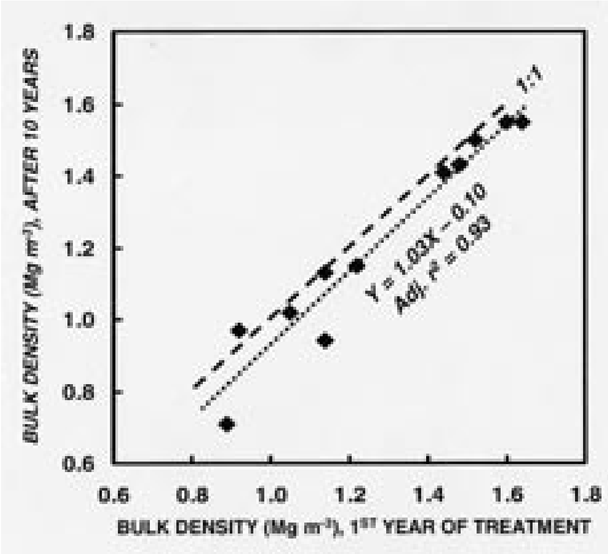

Figure 7—Soil bulk density recovery from severe compaction between the first and 10th year in the 10-20 cm depth zone on 11 LTSP installations. Understory excluded. No recovery indicated by the 1:1 line of parity. Regression line indicates that the higher the initial bulk density, the lower the rate of recovery.

and (3) soils are not compacted readily around stumps left from the previous stand (large surficial roots buffer against compaction). Friable soil bordering roots of remnant stumps maintains a favorable balance of moisture and aeration and becomes the locus for increased rooting activity and superior growth in the new stand (Van Lear and others 2000).

Evidence abounds that soil compaction reduces tree growth (Greacen and Sands 1980, Powers and others 1990) and models for estimating tree growth reduction with increasing compaction have been developed (Froehlich and McNabb 1984). But more recent findings indicate that the impacts of compaction are not universal. Instead, impacts depend largely on site conditions affecting air and water balance in the rooting zone (Gomez and others 2002; Heninger and others 2002; Miller and others 1996; Siegel-Issem and others, in press).

The Presence of Understory Vegetation

Over the first 10 years of the LTSP experiment, the single strongest factor affecting planted tree growth was the competitive effect of understory vegetation. Whether in the Sierra Nevada or the Southern Coastal Plain, tree biomass averaged about one-fifth greater where understory vegetation was excluded (figure 8). In the Sierra Nevada, where summer drought is common, planted tree productivity averaged more than three times higher in the absence of understory vegetation.

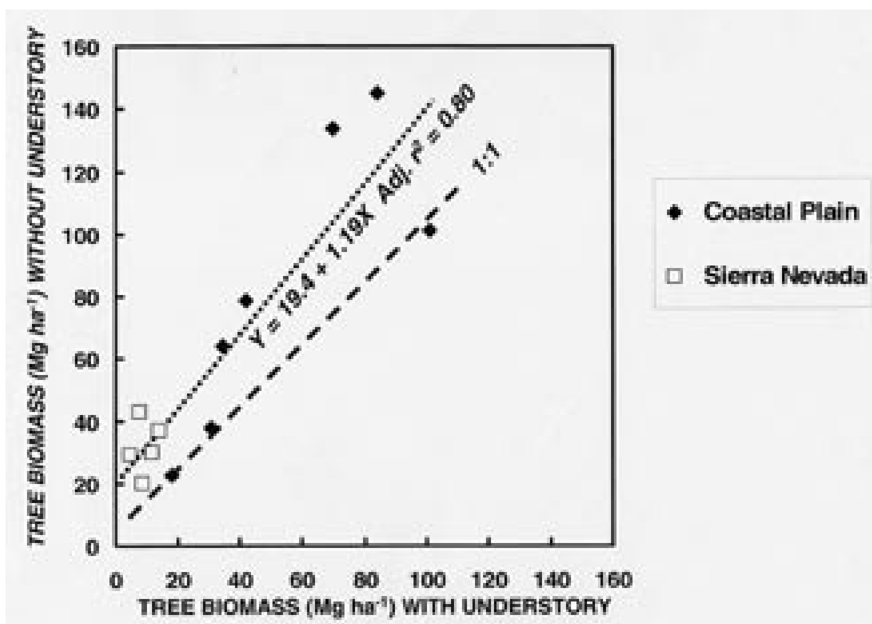

Figure 8—Effect of understory vegetation on the biomass of planted trees at 10 years for OM_2C_0 treatments on 12 LTSP installations. Regression line indicates that growth response to vegetation control is proportionally greater on lower productivity sites, but absolutely greater at higher levels of productivity. Dashed line indicates 1:1 parity between treatments.

Conclusions

The LTSP experiment is still in its infancy. Installations were established over several years, and only the oldest and most productive are approaching site carrying capacity. The findings reported here may provide the earliest glimpse into general longer-term trends. Or they may be seen as aberrations once a more complete data set emerges and vegetation more fully occupies our sites. What we can conclude for the Sierra Nevada and the Southern Coastal Plain is that there is no evidence that soil productivity has been seriously impaired in the first 10 years despite massive removals of surface organic matter and substantial soil compaction.

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Caring for a Wild and Scenic River

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Abstract*—The Middle Fork Clearwater Wild and Scenic River was established under the 1968 Wild and Scenic Rivers Act. Forest Service managers gradually became concerned with the increasing loss of the large, old ponderosa pine and Douglas-fir that characterize much of the river corridor and adjacent uplands. The perceived dilemma was how to maintain both high esthetic values and a seral forest that was resilient in the face of wildfire, insect attacks, and disease presence. The Lochsa District on the Clearwater Forest developed guidelines for management within the corridor. Prescriptions included shelterwood with reserves, group selection, and prescribed fire. These treatments maintained the highly esthetic character, improved big game winter range, reduced fire hazard, maintained soil stability on steep slopes, realized an economic return, and set up these forests for long-term resiliency.*

Introduction

The Management Setting

The Wild and Scenic Rivers Act was passed in 1968. Among others, it named the Middle Fork Clearwater as a Wild and Scenic River, designated as a recreation river. Recreation rivers are managed for their high scenic quality but are readily accessible by road and may have development along their shores. Private lands along the Middle Fork, downstream from the National Forest, are encumbered with scenic easements, authorized and funded through the Wild and Scenic Rivers Act. These easements limit development and land-disturbing activities to preserve the scenic quality of the corridor. Management actions on both private and federal lands need to preserve and enhance the outstandingly remarkable values for which the river was designated. For the Middle Fork, this includes maintaining a forested setting along the river. For many years after designation, this was interpreted to exclude timber harvest.

Years of fire exclusion and years of drought resulted in conditions that made river managers rethink that interpretation. Under natural conditions, these lower river breaks would have underburned every 25 years or so. This would have maintained seral ponderosa pine and Douglas-fir in fairly open stand conditions. Instead, 60 years of fire exclusion has allowed grand fir and additional Douglas-fir to become established and grow into dense stands. Over the past decade, north-central Idaho has experienced droughty conditions. Drought, coupled with these dense stand conditions, has put stress on the older overstory ponderosa pine, making them vulnerable to insects and diseases. Many have died. Even without active management, the character of the forest was shifting and becoming more vulnerable to drastic change

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Table 1—Fire regime condition classes.

Condition class	Fire regime	Example management options
Condition class 1	Fire regimes are within an historical range, and the risk of losing key ecosystem components is low. Vegetation attributes (species composition and structure) are intact and functioning within an historical range.	Where appropriate, these areas can be maintained within the historical fire regime by treatments such as fire use.
Condition class 2	Fire regimes have been moderately altered from their historical range. The risk of losing key ecosystem components is moderate. Fire frequencies have departed from historical frequencies by one or more return intervals (either increased or decreased). This results in moderate changes to one or more of the following: fire size, intensity and severity, and landscape patterns. Vegetation attributes have been moderately altered from their historical range.	Where appropriate, these areas may need moderate levels of restoration treatments, such as fire use and hand or mechanical treatments, to be restored to the historical fire regime.
Condition class 3	Fire regimes have been significantly altered from their historical range. The risk of losing key ecosystem components is high. Fire frequencies have departed from historical frequencies by multiple return intervals. This results in dramatic changes to one or more of the following: fire size, intensity, severity, and landscape patterns. Vegetation attributes have been significantly altered from their historical range.	Where appropriate, these areas may need high levels of restoration treatments, such as hand or mechanical treatments, before fire can be used to restore the historical fire regime.

From: Schmidt and others 2002.

as a result of intense wildfire. This is a significant departure from historic fire effects. Most of this area would be classed as Fire Regime Condition Class 3 (Schmidt and others 2002; see table 1), well outside its historic disturbance regime, and at risk of losing key ecosystem components.

The Clearwater Forest Plan (Anonymous 1987) designated the area within ¼ mile of the river as Management Area (MA) A7, to be managed as a wild and scenic river. The breaklands farther than ¼ mile from the river are to be managed for big game winter range and timber management, with a high visual quality objective (MA C4). Since the late 1990s, elk populations have declined, with at least part of the cause being lack of high quality winter forage.

Soils are shallow on these steep breaklands and are inherently unstable. Mass wasting is a natural soil movement or landslide occurrence that supplies woody debris and cobbles for anadromous fish spawning gravels. Any treatments would need to be designed to limit additional soil movement.

Private Land Guidelines

Private landowners within the Wild and Scenic River corridor, downriver from the forest, were the first to address these changing forest conditions. Their sites were drier and started showing symptoms of stress sooner. Landowners wanted to manage their forests to keep them healthy. The scenic easement holder (the Forest Service) could have said "no harvest" as long as the trees were green, as the easements only allow the landowner to cut dead trees. Rather, working with the forest landscape architect, local ranger district personnel developed harvest guidelines that would maintain the forest appearance but develop healthy stands, resilient to disturbance, over time. These guidelines were generally to remove no more than 20 percent of the canopy at a time, to keep road construction off of the steep ground, and to retain the large seral trees (Jones 1998). These guidelines were used successfully on a number of private properties over several years.

National Forest Proposal

The East Bridge project area was chosen for assessment because past harvest had created landscape patterns that did not fit the natural pattern. There were straight lines at the edges of clearcuts and "gun sight" breaks on the ridgelines. These conditions did not meet the visual goals for lands along the river corridor. It looked like an easy fix: just feather the edges and take a few more trees off the ridgeline, and things would be just fine. That isn't exactly how it worked out.

The initial proposal would have addressed the short-term scenic quality from the highway but would not have addressed the long-term maintenance of the forest or dealt with winter range concerns (Klinger 1998, Talbert 1999). It would have repaired existing problems with scenic quality but would not have developed a forest that would be healthy and resilient for many decades to come. The selected alternative for the project dealt with both the existing scenery problems and long-term forest health. The guidelines developed and tested on private lands were adopted for this project, which has now been implemented as the East Bridge Timber Sale and the East Bridge Prescribed Burn (table 2).

Table 2—Summary of East Bridge treatment units.

Treatment unit	Current vegetation	Prescription	Expected results
1, 1A	Mixed conifer, marginal stocking, root rot active	Group selection followed by underburn and spot planting	Develop a two-storied stand of early seral xeric conifers
5	Mixed conifer, very active root rot	Shelterwood with reserves followed by underburn and planting	
4, 6	Xeric mixed conifer, active root rot	Group selection followed by underburn and spot planting	
11	Xeric mixed conifer, low stocking in overstory, high stocking in small trees	Prescribed fire	Reduction in understory stocking, higher percentage of seral species in understory

Ecology and History

Fire-Resistant Species

The forest in the East Bridge area is a dry forest, dominated by ponderosa pine and Douglas-fir at lower elevations. Both ponderosa pine and Douglas-fir are fire resistant due to their thick bark. As elevation increases, grand fir and western redcedar are more common. Grand fir and western redcedar are found on moist, relatively warm sites. They are very susceptible to fire damage, especially at young ages. The low-elevation ponderosa pine and

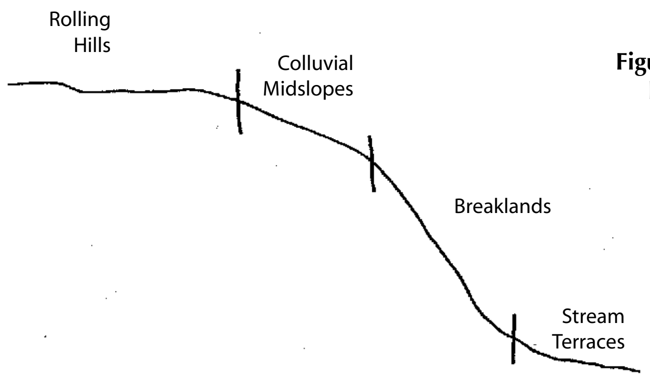

Figure 1—Schematic drawing of landtype groups.

Douglas-fir were maintained by frequent fires, returning at 25-50 year intervals. These fires removed much of the grand fir and cedar, some of the young Douglas-fir and ponderosa pine, and a few of the older trees.

Frequent Fires, Low Intensity/Severity

Prior to the early 1900s, fires burned frequently in the East Bridge area. The low, steep southerly aspect slopes dried out faster than the high-elevation rolling hills above them (figure 1). These fires left evidence in the large, old ponderosa pine on sites that will support grand fir and western redcedar, as well as numerous fire scars at the base of many of those pines and Douglas-firs. The East Bridge area is in a transition zone from low-elevation dry sites to higher elevation, more moist sites.

Landtype Associations

Landtype associations (LTAs) are landform classifications that follow the National Hierarchy of Ecological Units (Cleland and others 1997). They are aggregates of site-specific landtypes and subsets of the subsection classifications. The primary landtype association in the East Bridge area is 23A, which is composed of stream breaklands on southerly and westerly aspects, with shallow soils. These stream breakland landtypes are typically steep – with 60 percent or steeper slopes. They are some of the hottest and driest sites on the Clearwater National Forest. The parent material is micaceous gneisses and schists associated with the border zone of the Idaho Batholith. These are prone to mass wasting, and over 75 percent of the area is rated as high to very high risk of mass wasting (Mital 1998). Mass wasting is a general geological term for dislodgement and down slope movement of soil and rock material. The fire regime is a frequent fire return interval (25 to 50 years) with non-lethal or mixed severity. Stands on this LTA were typically uneven-aged, composed of small, even-aged groups. Fire suppression has successfully excluded fire from the area for about 70 years (Hazelbaker 1998).

Fire Exclusion

In the early 1900s, frequent fires swept through this area. The last large fire burned in 1934. These fires left scars on the bases of the big, old ponderosa pine but didn't kill them. Since then, human population growth and increased national forest management resulted in highly successful fire suppression. With the exclusion of fire on these sites, Douglas-fir representation increased and grand fir and cedar invaded the open understories.

Existing Stand Conditions

Current forest structures are multi-layered with shrubs, saplings, poles, and large trees. Crowns low on the boles of the trees create a ladder-fuel condition that could easily carry fire through the stand into the crowns and kill even the large, old ponderosa pine. Many of these large trees survived at least seven understory fires without major damage but have a low fire-survival potential with the current fuel conditions. Most of the seedlings, saplings, and poles are grand fir and Douglas-fir. Competition from these young trees has put the older ponderosa pine under additional stress. With this additional stress, they are beginning to succumb to insects, disease, and drought.

Prescriptions

Management Objectives

High Scenic Quality (Long Term Vs. Short Term)

River managers were concerned that maintaining the existing forest cover in the short term could set the area up for stand-replacing fires in the future. The resulting bare slopes and risk of ugly scars from mass wasting of bare soils would diminish scenic values. Managers wanted to make these stands more resilient to fire effects in order to maintain a scenic forest in the long term. Due to the heavy fuel loads and arrangement, it was unlikely this could be accomplished using only fire without harvest.

Soil Stability

Most of the project area has inherently unstable slopes, but Unit 5 caused particular concern for soil stability. There was already a small, active slope failure within the boundary. Stand composition was almost entirely Douglas-fir, with extensive root rot mortality. The concern was that the continued mortality would eventually result in reduced soil stability. Other units also had steep slopes and high risk of mass wasting.

Fire Hazard Reduction

U.S. Highway 12, a major east/west route between Idaho and Montana, runs along the river. The increased fire hazard, due to stand structure and composition, combined with a higher risk of human-caused ignitions, pointed to the need to reduce the fire hazard. The project area is adjacent to the little town of Syringa, which also raised a concern for the potential of urban interface fires.

Shrub Rejuvenation

These breaklands are low elevation sites, used by wintering big game animals. As the shrubs aged and the canopy closed, available forage was reduced. As seral species, shrubs need more open growing conditions to grow vigorously and produce abundant forage. Good winter range forage is one of the keys to maintaining the good elk herds for which central Idaho is well known.

Establishment of Seral Species

Habitat types range from mesic Douglas-fir types (*Pseudostuga menziesii/ Physocarpus malvaceus*) through the moist grand fir types, to moist western redcedar types (*Thuja plicata/Adiantum pedatum*) (Cooper 1991). Without periodic disturbance, the understories of these stands filled in with climax tree species – Douglas-fir, grand fir, and western redcedar. As the older ponderosa pine lost vigor and were subject to increased competition from understory trees, they began to succumb to insects, disease, and drought. They were disappearing from these stands. There was little opportunity to establish additional ponderosa pine, which is very fire tolerant and moderately shade intolerant, and which was much more abundant on the site historically.

Technical Details

Prescriptions were developed for shelterwood with reserves, group selection, and prescribed fire (FSM 2470).

Shelterwood With Reserves

Unit 5: This unit was not directly adjacent to the river corridor, but it is visible in the middle ground and background. Cedar and grand fir habitat types were both common, with mesic Douglas-fir/ninebark on the drier ridges. The existing forest was dominated by Douglas-fir and grand fir, with an understory of tall, old shrubs. Root rot was gradually reducing conifer stocking levels. The stand was not meeting resource management objectives to provide big game winter forage, contribute to timber production, maintain slope stability, and provide high visual quality. Harvest provided an opportunity to rejuvenate the decadent shrubs and re-establish ponderosa pine for long-term health of the breaklands landscape which would, in turn, provide for high quality scenery from the river corridor.

The prescription for this unit called for a group shelterwood with reserves harvest system, leaving about one-half of the area in untreated groups, followed by an underburn. The groups have about 135 ft^2 of basal area; so overall, the stand will be left with 60 to 70 ft^2 of basal area per acre. Ponderosa pine was favored as leave trees where it occurred. This prescription was designed to be similar to a mixed severity fire. Trees were left in swales and along active landslides to maintain short-term soil stability and provide material for large woody debris in streams when slides would occur. The openings were to be planted with ponderosa pine to assure recruitment of this early seral species. There were few ponderosa pines in the overstory, and those present were poor seed producers. These stands would be maintained as two-storied stands.

Group Selection

Units 1, 1A, 4, and 6: Units 4 and 6 are directly adjacent to the river corridor and highway. All are a little drier than Unit 5. The predominant habitat type is grand fir/ninebark (*Abies grandis/Physocarpus malvaceus*) (Cooper 1991). They have an old ponderosa pine overstory that is gradually disappearing as the trees die. Clumps of Douglas-fir and grand fir are common throughout the stands, both between the older ponderosa pine and under the pine canopies.

The group selection method was chosen for these units to produce a disturbance similar to a low-severity fire. Harvest was followed with slash burning in the openings. Removals targeted groups of grand fir and Douglas-fir in root rot pockets, leaving the old ponderosa pine where possible. The openings were one-half to one acre in size. About 25 percent of the acreage was treated. Spot planting of ponderosa pine in the small openings was prescribed to assure establishment of this desired species. Two factors reduced the likelihood of natural regeneration. First, the overstory trees are old and are not reliable cone producers. Secondly, the shrubs in ninebark habitat types often respond to disturbance with profuse growth, occupying the site and precluding seral conifer establishment (Steele and others 1992; Fire Effects Information System 2003). Planted trees would also have an advantage over naturally regenerated seedlings. They are larger and are established sooner so are more likely to stay above the ninebark. The intent was to maintain these stands as three-storied stands with even-aged groups.

Prescribed Fire

Unit 11: This unit had scattered large, old ponderosa pine and Douglas-fir trees, with an understory of smaller Douglas-fir, grand fir, and a few western redcedar and ponderosa pine. These ranged in size from seedlings to small pole-sized trees. Distribution of these younger trees was very clumpy, with some shrub-filled openings still present. The stocked areas were usually overstocked for this site. Underburning was proposed to reduce stocking levels and remove some of the small late seral and climax trees. Fuel loads were rather high, and the fire management team proposed implementing this prescription over two to four entries. The first entry would consist of burning under moderate conditions to remove the most flammable fuels and kill some of the grand fir and cedar trees. Subsequent burns would gradually reduce more of the fuel load and remove more of the grand fir, cedar, and small Douglas-fir. The intent was to develop more open, two- or three-storied stands that have a dominant component of ponderosa pine.

Implementation

Project Design With Interdisciplinary Team

The interdisciplinary team made several trips through the area to look at desired conditions. This focused the project on the key items that would make this a success: retention of the large, old ponderosa pine; retention of considerable canopy to maintain scenic quality and soil stability; and reduced stocking levels to maintain forest health.

The East Bridge Timber Sale sold in 1999. It included yarding with skidders, skyline systems, and helicopters. The majority was yarded with helicopters.

Prescribed Fire

In September 1999, Unit 11 was burned for the first time. Aerial ignition with a sphere dispenser was used. Ignition was timed to take place just before a front moved through with expected rain showers. These materialized the day following ignition and limited fire spread within the unit. About one-third of the area within the unit actually carried fire. Shrubs and small trees in those areas were top-killed as expected. A few of the large, old ponderosa pine trees were also killed because the fire was able to get inside the boles through old fire scars.

In October 2002, this unit was burned again. The same aerial ignition technique was used. This time, most of the area actually carried fire (figure 2). Shrub rejuvenation was more extensive, and more of the small tree seedlings and saplings were killed. The resulting stand is a patchy, open stand that is

Figure 2—Smoke generated over the entire burn unit.

weighted to the early seral species. Additional burning is planned in another three to five years.

Shelterwood With Reserves

This unit was harvested in 2003, using a helicopter logging system. It will develop into a two-aged ponderosa pine forest. This would be typical of forest structure and composition under periodic fires. The open stand conditions are conducive to shrub growth for winter use by big game. Shrubs that were top-killed by prescribed burning after harvest are resprouting, and redstem ceanothus (*Ceanothus sanguineus*), a preferred browse species, has germinated profusely (figure 3). Adjacent stands provide more dense vegetation for thermal cover.

Figure 3—Redstem ceanothus seedlings after harvest and underburn.

Group Selection

These were recently logged (spring of 2003) with a helicopter yarding system. There is little evidence of disturbance when viewed from the highway along the river (figure 4). There may be a short-term visual impact when the stands are underburned this fall. The emphasis in these units was maintaining visual quality. Additional entries will likely be needed to reduce fuels and improve browse conditions.

Conclusions

Each of the prescriptions met the objectives of improving forest health and resiliency while maintaining a forested appearance from the scenic river corridor. Some retained more forest cover, but all were within the range that could

Figure 4—View of group selection harvest.

be expected from natural disturbances. The group selection treatments created small patches where fuel loads were reduced but left ladder fuels in the remainder of the forest. Shelterwood treatments and prescribed fire treatments produced a more uniform fuel reduction. Group selection that included intermediate treatments (thinning) between the groups that were removed would have also reduced fuel loads more uniformly over the treated area.

The biggest challenge in implementing all of these treatments was field layout. Treatment units are located on very steep breaklands along the Middle Fork Clearwater River. Slopes over 60 percent are common. Both personnel safety and work productivity were concerns. Post-treatment monitoring is also a challenge. Fortunately, no one was injured and the layout work was completed on time, but the results could have been different.

Overall, the scenic quality has been retained, the seral forest was maintained, big game winter range was improved, fuel loads were reduced and ladder fuels that could lead to stand replacing fire were removed.

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Is Forest Structure Related To Fire Severity? Yes, No, and Maybe: Methods and Insights in Quantifying the Answer

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Abstract—*Wildfires in 2000 burned over 500,000 forested ha in the Northern Rocky Mountains. In 2001, National Fire Plan funding became available to evaluate the influence of pre-wildfire forest structure on post wildfire fire severity. Results from this study will provide information on forest structures that are resilient to wildfire. Three years of data (558 plots) have been collected from forested areas that burned in 2000, 2001, and 2002. Forests used in this study include dry ponderosa pine/Douglas-fir, cold lodgepole pine/subalpine fir, and moist western larch forests. Probability sampling of all areas within a particular fire perimeter was used to locate study sites and a sampling matrix was used to capture variation in weather, topographic setting, and pre-wildfire forest structure of which the fires represented. Fire severity (current state of soils and vegetation after the wildfire) was quantified on adjacent paired plots, with each plot representing a different forest structure. Classification trees and cluster analysis identified relations among forest structure characteristics, physical setting, and fire severity. Probability of a particular forest structure relating to fire severity was computed. This paper describes methodology used in the project, discusses challenges associated with conducting this type of study, and uses preliminary results (probabilities) from the first two years of data collection to show how forest structure relates to both crown and soil surface fire severity.*

Introduction

Fire behavior (expressed as intensity) and severity are dependent on the interaction among forest structure and composition (fuel), weather, and physical setting (Robichaud and others 2003; Rothermel 1983, 1991; Ryan 1990; Wells and Campbell 1979). In general, fuels defined as canopy bulk density (canopy weight for a given volume), live crown base height, and surface fuel conditions (amount, composition, moisture content, compactness, continuity) are key forest characteristics related to fire behavior (Albini 1976; Agee 2002; Graham and others 1999; Rothermel 1983, 1991; Scott and Reinhardt 2001). Most often the objective when altering forest fuels is not to remove fire completely from a forest, but rather to make a forest more resilient to fire and decrease a fire's unwanted and detrimental effects by altering these key forest characteristics (Agee 2002; Graham and others 1999; Scott and Reinhardt 2001).

Because fire behavior and effects are highly complex, there is still uncertainty in knowing when and where forest structure characteristics influence both wildfire behavior and/or severity, particularly during large and extreme wildfire events (Albini 1976; Carey and Schumann 2003; Cruz and others 2003; Omi and Martinson 2001; Graham 2003). In fact, there is little empirical information determining when (under what weather conditions

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and physical settings) forest structure contributes to decreasing crown-fire occurrence (Carey and Schumann 2003; Cruz and others 2003; Omi and Martinson 2002).

Moreover, it is difficult to directly quantify fire behavior (e.g., not safe for persons to closely observe fire behavior) during extreme wildfire events; however, fire severity can be evaluated for its relation to forest structure after a fire has occurred and, to a certain extent, indicate how forest structure influenced a fire's behavior. Fire behavior characteristics include rate of spread, fire line intensity, residence time, transition to crown fire, and spotting, and they are usually associated with a flaming front (Rothermel 1972, 1983, 1991; Albini 1976; Van Wagner 1977). Fire severity is dependent on what is burned and the units used for its evaluation (Simard 1991). For example, wildfire severity describes the amount of organic material consumed, its flame length, torching index, and other indicators of fire risk and fire behavior. The wildfire severity in terms of its effects on the atmosphere describes the particulates and gasses a wildfire produces and its effects on sky clarity (Finney and others 2003). In economic terms, fire severity describes the value of homes damaged, timber destroyed, or water storage losses measured in dollars (Kent and others 2003). Fire severity in relation to vegetation and soils describes the extent of char on shrubs, forest floor, rotten wood, scorch height on tree boles and crown scorch, exposed mineral soil, and the amount of soil modification (fusing of soil particles, changes in color, etc.). These descriptors and quantification of fire severity can provide interpretive possibilities as to the effect a fire would have on processes such as soil erosion, tree growth, vegetation regeneration and succession, or nutrient cycling. In addition, fire severity can relate to the fire behavior—such as all black crowns (fire severity indicator) are caused by a crown fire (fire behavior indicator), mixed black and green crowns indicate a surface fire with some torching in the crowns and green crowns with abundant organic materials remaining on the forest floor would indicate a low intensity surface fire.

In general, forest management concentrates on desired conditions to meet a particular goal or objective ranging from timber production to maintaining wildlife habitat. As indicated by the passing of the Healthy Forests Restoration Act of 2003, the development of resilient fire dependent forests is also a national emphasis. These objectives singly or in combination can be met through silviculture prescriptions that describe forest composition and structure development through time. Attributes of resilient fire-dependent forests include appropriate species, live trees, seed sources, and intact soils. Presence of these elements are important after a wildfire (Debano and others 1998; Hungerford and others 1991; Jurgensen 1997; Robichaud 2003). Because of this importance we chose to describe and quantify fire severity as the condition of the vegetation and soils after a wildfire.

The wildfires that burned in the Rocky Mountains in 2000, 2001, and 2002 provided an opportunity to study the influence that pre-wildfire forest structure has on fire severity. In addition, this replicated study will add to our knowledge of describing and quantifying fire severity. This paper introduces the study, provides some preliminary results, and provides some "food for thought" on the relation between pre-wildfire forest structure and fire severity. In this paper we present methods used in data collection, show how pre-wildfire forest structure was reconstructed from post-fire characteristics, describe ways to classify fire severity, and determine if relations between forest structure and fire severity can be identified.

Methods

Study Areas

Although this study was conducted in Montana, Idaho, Colorado, and Oregon on fires occurring in 2000, 2001, and 2003, the analysis and results of this paper only encompass data collected on sites burned during 2000 and 2001 by fires on the Bitterroot, Lolo, Kootenai, and Flathead National Forests in Montana (figure 1). In this analysis a total of 19 separate fires were sampled within the cold (lodgepole pine *Pinus contorta* and subalpine fir *Abies lasiocarpa*) moist (western hemlock *Tsuga heterophyll;* western redcedar *Thuja plicata;* and grand fir *Abies grandis*), and dry (ponderosa pine *Pinus ponderosa* and Douglas-fir *Pseudotsuga menziesii*) forests. The Bitterroot fires (eight fires) burned 144,040 ha within the cold and dry forests from July 15 through September 1, 2000. On the Lolo National Forest, three fires totaling 15,662 ha were sampled that burned from August 5 through September 6, 2000. We sampled the Moose Creek Fire in the Flathead National Forest, which burned between August 16 and October 5, 2001 and encompassed 28,723 ha of cold forest. Eight fires burning a total of 14,000 ha between July 31, 2000, and August 30, 2000, were sampled in the moist forests within the Kootenai National Forest. All fires were sampled the summer after they occurred, except for the fires on the Kootenai National Forest, which were sampled the second summer after they occurred.

Study Design

Stratified random sampling of each fire was used to ensure that the variation in forest structure, physical setting, and weather were represented within each fire. It is the interaction of these characteristics that determine fire severity (Ryan 1990, Lohr 1999). In establishing the sampling frame,

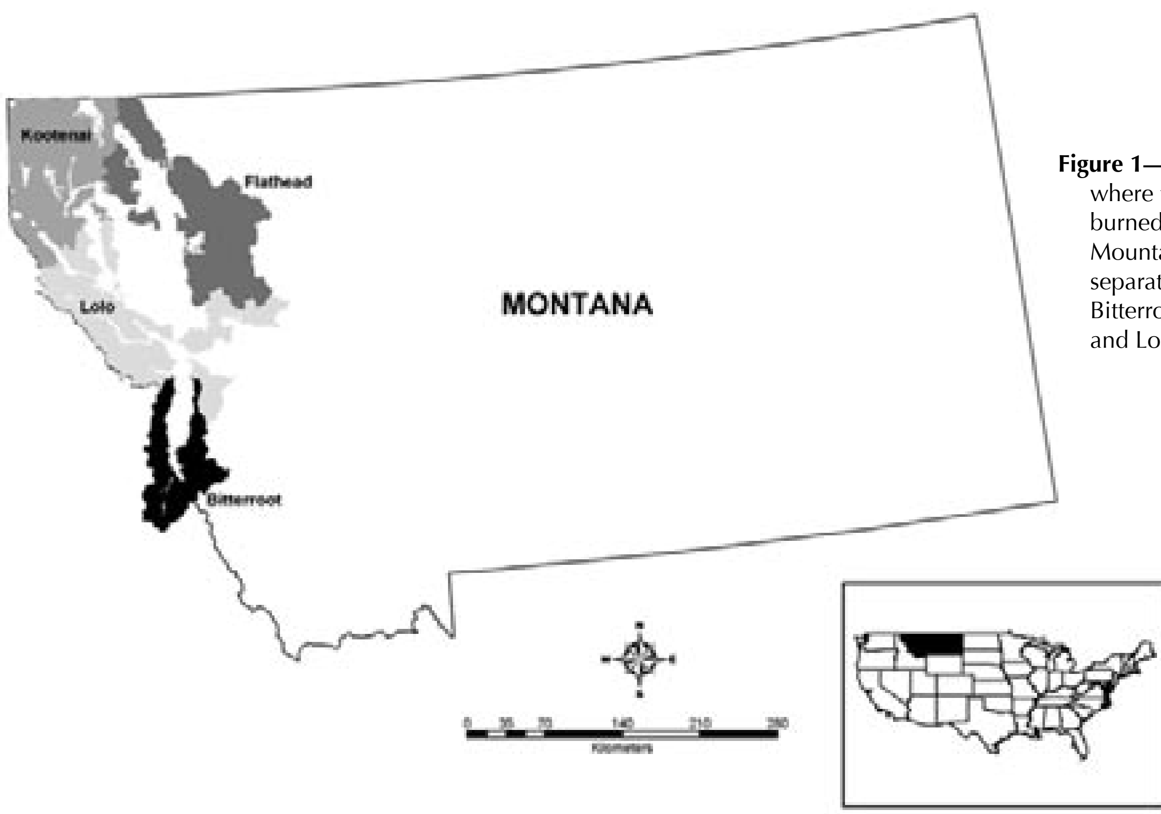

Figure 1—This map shows the area where the 2000 and 2001 fires burned in the Northern Rocky Mountains. We sampled 19 separate fires located on the Bitterroot, Kootenai, Flathead, and Lolo National Forests.

forest cover type was used to describe the broad-scale vegetation. Cover types included: ponderosa pine and/or Douglas-fir, (PP/DF), grand fir, western redcedar, and/or western hemlock (GF/C/WH), and lodgepole pine and/or subalpine fir (LPP/SAF). Within a specific cover type, burning index accounted for variation due to weather. Burning index describes the effort needed to contain a single fire within a particular fuel type within a given area. The index is based on the spread component (SC) and available energy release component (ERC) of a fire, which in turn are used to estimate flame length from which the burning index is computed (Bradshaw and others 1983, Bradshaw and Britton 2000). Wind speed, slope, fuel (including the effects of green herbaceous plants) and the moisture content of the fuels are used to determine the SC and ERC. The difference between the two components is that SC is determined on the moisture levels of the fine fuels while ERC requires moisture levels from the entire fuel complex.

Fire progression maps were used to estimate the day a particular stand burned. Using weather data for this day from the closest weather station and the most applicable fuel model for each fire, the burning index for each stand within the fire perimeter was calculated using Fire Family Plus (Bradshaw and Britton 2000). After forest cover type, the stands within the fire were stratified by high and low burning index (divided at the median burning index) for all stands burned by a particular fire. This stratification ensured that stands sampled were burned during the range of weather conditions that occurred throughout the fire.

Within each burning index class (high and low) the physical settings of the stands were placed into two strata: those with slope angles less than or equal to 35 percent and those with slope angles greater than 35 percent. In the Northern Rocky Mountains, settings with slope angles less than 35 percent usually occur on benches, within riparian areas, or along ridge tops. Settings with slope angles greater than 35 percent tend to occur on side slopes. Within a given slope class, the structure characteristics of stands were divided into those containing short, sapling- to medium-sized trees (≤13 m), and those containing tall, mature to old trees (>13 m). Within these size classes stands were divided into two density strata: those with canopy cover less than or equal to 35 percent and those with canopy cover greater than 35 percent. This stratification ensured that stands selected for sampling would have a range of horizontal structure. Therefore, the final sampling stratification contained forest cover (3 classes), burning index (2 classes), slope angle (2 classes), canopy height (2 classes), and stand density (2 classes). All stands occurring within a particular stratum and fire perimeter had an equal probability of being selected. Additional fire and physical setting characteristics not in the stratification but occurring regularly were recorded during sampling and included aspect, bole scorch height, and direction of the scorch as indicators of flame length (Van Wagner 1973) or ignition source (back fire, flank fire, or head fire).

Stand Selection

All stands within the fire perimeter contain a unique identification code. These codes were randomly assigned into the sampling matrix, which represented the designed stratification. The matrix was populated with the first 15 low-density stands that were randomly selected. Each stand was evaluated (in selection order) to determine if it (1) fit within the sampling criteria, (2) had an opportunity to burn (in some cases, stands along the fire perimeters had fire lines that prevented them from burning), (3) did not have any

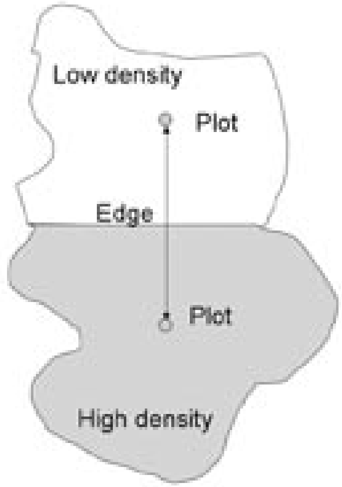

Figure 2—Illustration of paired plots between two stands. The low-density stands were paired to an adjacent stand that had a distinct change in forest structure. The adjacent stand required a change in horizontal structure (density defined by canopy cover), species composition, and/or vertical structure (number of stories). Plots were located a minimum of 50 m to plot center from stand edge.

confounding factors that may have influenced burning (e.g., fire retardant throughout, large fire lines splitting the stand), and (4) was at least 100 m by 100 m in size (large enough to establish the sample points).

In order to increase the number of stands sampled and to determine if changes in stand structure influenced fire severity in a given area, randomly selected low-density stands were paired with qualified adjacent stands (figure 2). To qualify as a paired stand it had to be adjacent to the randomly selected stand and contain a change in horizontal structure (density defined by canopy cover), species composition, and/or vertical structure (number of stories or vegetation layers). A change in stand density was defined as a differential between high and low canopy cover of at least 20 percent, (i.e., a stand with 25 percent cover was paired with a stand with no less than 45 percent canopy cover). A change in species composition was defined as a change in the cover type (e.g., lodgepole cover type to subalpine fir cover type). Vertical structure was a change in the number of stories (canopy layers) occurring in a stand, such as the selected stand containing a single story and the paired stand containing two or three stories or a selected multi-storied stand paired with a single storied stand.

Stand adjacency was determined by a rule set. The first choice for a paired stand with a different structure was downhill from the selected stand. Since fires predominantly burn uphill, this selection criteria would provide opportunities for sampling stands in which structure influenced either or both fire behavior and fire severity. If a major change in topography (such as 180° aspect change, steep side slope to riparian setting, etc.) occurred downhill from the selected stand before a suitable adjacent stand was selected, an alternate selection process commenced. Beginning on the western edge of the selected stand and continuing in a clockwise direction, forest conditions were evaluated until an adjacent stand was located. Ideally, the paired stand would be similar in aspect and slope as the selected stand, but subtle changes in slope and aspect were allowed. If no suitable stand was located adjacent to the randomly selected stand, the low-density stand was not chosen and the next stand in the matrix was evaluated.

Plot Selection

The objective of this study was to quantify the relation between pre-wildfire forest structure and fire severity among stands and not to characterize fire severity within stands. Therefore, to maximize the number of stands sampled, only one plot was placed in each selected stand. The edge of a stand was defined where the forest structure changed between the paired stands (figure 2). An aerial photo or topographic map was used to obtain an azimuth intersecting the approximate center of both stands. At a minimal slope distance of 100 m from the stand edge along this azimuth, a random number between 1 and 6 was selected (using a die). This value was multiplied by 16 and an additional distance (meters) equaling this value along the azimuth was traversed before plot installation. If the stand was too small to use this additional distance, the plot was located at least 50 meters from the stand edge. The plot was monumented with a 1 m rebar stake, the location was recorded by a GPS, and distance from the stand edge was recorded.

Data Collection

Site descriptors (aspect, slope, topographic position, and elevation) and a general stand description (species composition, number of stories, stand origin, horizontal spacing) for each plot were recorded. Our intention was to post-classify or develop a continuous variable characterizing fire severity. Therefore descriptors of soils and vegetation were collected in considerable detail. Our approaches to data collection were developed or modified from past fire severity classifications (Key and Benson 2001; Ryan and Noste 1985; Wells and others 1979) (tables 1, 2, and 3). The characterization and

Table 1—Surface components, their definitions, and char classes for fire severity. Litter fallen since fire, litter prior to fire, and humus depth were measured in cm. All measurements were conducted on a 1/740th ha circular plot. Trees were less than 12.7 cm diameter breast height (DBH).

Strata	Unburned (%)	Light char (%)	Moderate char (%)	Deep char (%)
		Surface		
Litter fallen onto surface since fire	Litter type (fir or pine, leaves) with no char classes			
Litter present prior to fire	No sign of char	Blackened but present	No moderate or deep char class	
Humus (decomposed organic matter)	No sign of char	Blackened but present	No moderate or deep char class	
Bare mineral soil	No sign of char	Blackened	Gray color	Orange color
Rock	No sign of char	Black edges	Black edges	White residue
Brown cubical rotten wood	No sign of char	Burned on surface	Charred but still present	Imprint on surface
Coarse woody debris ≤7.6 cm diameter	No sign of char	Burned on surface	Charred but still present	Not present
Coarse woody debris >7.6 cm diameter	No sign of char	Burned on surface	Charred but still present	Imprint on surface
Stumps	No sign of char	Burned on surface but intact	Completely charred	Stump hole
		Shrubs and Trees		
Shrubs – low <0.60 mm basal stem dia.	Stems intact	Stems present but charred	Base of stem present	Stump hole
Shrubs – medium 60-250 mm stem dia.	Stems intact	Stems present but charred	Base of stem present	Stump hole
Shrubs – tall >250 mm stem dia.	Stems intact	Stems present but charred	Base of stem present	Stump hole
Forbs and grasses	Growing on unburned litter	Growing on blackened surface	Growing on moderate charred soil	Growing on deep charred soil
New seedlings since fire	Growing on unburned litter	Growing on Blackened litter	Growing on charred soil	Growing on deep charred soil
Trees present prior to fire <12.7 cm DBH	No sign of char	Live trees needles present	No or brown needles	Stump hole

Table 2—Fire severity data taken on large trees (>12.7 cm diameter breast height (DBH) using a fixed (1/59th acre) and variable plot (8 m^2 ha^{-1}). Trees less than or equal to 45 cm DBH were measured on fixed plot, and trees greater than 45 cm DBH were measured on variable plot.

Strata	Un-compacted crown ratio	Green crown (%)	Brown crown (%)	Black crown (%)	Bole scorch height (ft) and direction scorch is facing (az)		Scorch at) base (%
					Low	High	
Trees >12.7 cm DBH	Total crown ratio	Green needles	Brown needles	Black stems, no needles	Scorch height and direction	Scorch height and direction	Circumference

Table 3—Forest structural characteristics derived from the FFE-FVS (Forest and Fuels Extension-Forest Vegetation Simulator) model (Reinhardt and Crookston 2003).

Density characteristics	Characteristics related to fire behavior	Biomass characteristics (Mg/ha)	Miscellaneous characteristics
Trees per ha	Height to base of crown (ft)	Foliage biomass	Average top height
Basal area (sq. m/ha)	Canopy bulk density	Live branches <7.6 cm	Number of stories
Stand density index		Live branches >7.6 cm	Species composition
Crown competition factor		Cubic volume	Dominant species
Total canopy cover (%)		Vertical distribution of crown versus stem	Quadratic mean diameter
Sum of the diameters (cm)			Dry, cold, or moist forest
			Average top height for plot

description of soils and vegetation were accomplished using five strata: (1) soil surface, (2) grass, forbs, small shrub, and seedlings, (3) medium and tall shrubs (4) saplings and large trees, and (5) woody debris (tables 1 and 2) (DeBano and others 1998).

All strata (surface and understory vegetation) except for the large trees and woody debris were measured on a 1/740th ha circular plot. For the large trees, a combination of fixed and variable radius plots was used to ensure enough trees representing all sizes were sampled. Trees greater than 45 cm diameter breast height (DBH) were sampled using a variable radius plot defined by an 8 m^2 ha^{-1} angle gauge (40 ft^2 ha^{-1}). Trees between 12.7 and 45 cm DBH were measured using a 1/59th ha fixed plot.

Soil surface characterization included total cover and the proportion of total cover dominated by new litter (deposition since the fire), old litter (present previous to the fire), humus, brown cubical rotten wood (rotten wood at or above the soil surface), woody debris less than or equal to 7.6 cm in diameter, woody debris greater than 7.6 cm in diameter, rock, and bare mineral soil. Each of these cover characterizations were divided into char classes (table 1). The second stratum described the proportion of grass and forbs growing on a specific charred surface. Cover proportion and number of basal stems were used to quantify small shrubs (<0.5 m tall or <0.60 mm basal stem diameter) (Brown 1976) (table 1). The number of new tree seedlings regenerated since the fires (1-year post fire) were counted and if the species was identifiable it was recorded (table 1). The medium (0.5 to 2 m tall or 60 mm to 250 mm basal stem diameter) and tall shrubs (>2 m tall or >250 mm basal stem diameter) were quantified using the same protocol as the low shrubs (table 1) (Brown 1976). The fourth stratum included saplings (<2.7 cm DBH) established prior to the fire and large (>12.7 cm DBH) trees (tables 1 and 2). The total number, species, and height of saplings were recorded and classified as to their fire severity (saplings with no char, charred saplings with brown needles, charred saplings with no needles, and a burned stump (table 1). Species, height, diameter, and uncompacted crown ratio were recorded for each large tree. The proportion of the total crown containing green needles, brown needles, no needles, or black stem was determined for each large tree. Scorch height on the stem was recorded and the circumference of scorch at the base of the stem was measured (table 2). The amount of woody debris on the site was determined using three 37 m linear transects (0, 120, and 240 degree azimuths) starting at plot center (Brown 1974).

Discussion

Fire behavior most often is described at the stand level with at least an elementary understanding of how forest structure, weather, and physical setting interact to create a given fire behavior (Albini 1976; Rothermel 1972, 1983, 1991; VanWagner 1977). In contrast, there is little understanding how these same characteristics interact to provide a specific fire severity where each fuelbed and combustion environment can create a different fire severity (Ryan and Noste 1983). In this study we described fire severity, forest structure, weather, and topographic characteristics across three forest types. The fires we sampled were all large (2000 to 144,000 ha) and burned dry fuels during extreme weather events. The variation in fire severity and fire behavior captured in these fires was beneficial since the inferences derived from the data will reflect a wide range of conditions. However, large amounts of variation can be detrimental because it often masks relations and makes the analysis challenging.

In general, fire models were developed to predict fire behavior and effects within "normal" burning conditions; however, fires used in this study burned outside "normal" weather conditions, limiting fire model use in the analysis (Albini 1976, Bitterroot National Forest 2000). To be effective, the analysis needs to maintain simplicity but be robust enough to answer a suite of questions useful to both managers and the scientific community: For example, how should forest structure be characterized when related to fire severity? How should fire severity be defined to provide ecological understanding as well as analytical power? Can relations between forest structure and fire severity be determined and if so, which combinations of variables best describe these relations? Is there a relation between fire severity observed on tree canopies and those associated with the soil surface and lower vegetation?

Characterizing Forest Structure

The Fire and Fuels Extension (FFE: Reinhardt and Crookston 2003) to the Forest Vegetation Simulator (FVS: Wykoff and others 1982) was used to characterize pre-wildfire forest structure. The Northern Rocky Mountain variant of FFE-FVS provided relative values of forest structure characteristics using the data collected at each sample point (e.g., tree DBH, crown ratio, total height, and species). Forest structure characteristics derived from FFE-FVS included stand density indices (basal area per ha, stand density index, trees per ha, etc.), characteristics associated with fire behavior (canopy bulk density and height to the base of the live crown), biomass estimates of foliage and branches, and other miscellaneous stand characteristics (number of stories, dominant species, etc.) (table 3).

Describing pre-wildfire forest structure based on post wildfire conditions has proven to be effective but limited. From a forest stand and tree perspective, at least in relative terms, different forest structures can be described using post wildfire data, because live tree branches and boles were seldom completely consumed in our data even during the most intense and severe fire. These post wildfire standing tree data along with FFE-FVS provided consistent data summaries within and across regions. These techniques can also be repeated within both a research and management framework and FFE-FVS provides stand structural characteristics linked to models describing fire behavior. Even with these benefits, FFE-FVS estimates of needle, branch, total biomass, canopy bulk density, number of stories, and horizontal structure are limited (Reinhardt and Crookston 2003). Subsequently, it is unknown how well they reflect true values (Cruz and others 2003). However, these relative values are extremely useful for understanding forest structure changes across sites and with the information added from this study, the capability of FFE-FVS for predicting fire severity as a function of forest structural characteristics can be improved.

Although we have good confidence in describing pre-wildfire standing tree and stand characteristics using post wildfire data, describing pre-wildfire soil surface characteristics post wildfire is problematic. Only in very limited circumstances are soil surface conditions described before a wildfire, and even recurring forest inventories such as those conducted by Interior West Forest Inventory and Analyses (e.g., USDA 1997) do not regularly describe forest floor conditions. To definitively describe or predict both fire intensity and fire severity requires pre-wildfire biomass estimates of shrub and herbaceous layers, fine and coarse surface fuels, litter, and duff. In general, fine-scale sampling is required to estimate these surface fuel characteristics, and extrapolating existing prediction equations across different regions is questionable (Brown 1976). Using habitat type, successional stage, overstory structure, or other stand or site characteristics for estimating surface fuels is limited in scope (e.g., Covington and Fox 1991, Mitchell and others 1987). A possible estimate of surface fuel conditions that existed pre-wildfire might be achieved by using scorch heights on boles of standing trees post wildfire as an indicator of flame length. In turn these data could be used to identify potential fuels and fuel loadings that could have produced these flame lengths. However, this approach for estimating pre-wildfire surface fuel conditions is highly speculative and needs thorough investigation.

Classifying Fire Severity

In our study we had the ability to describe fire severity using either continuous or categorical variables. Initially we used canonical correlation analysis using continuous variables that described soil surface fire severity such as amount of mineral soil exposed, amount of charred litter, etc. (table 1). The results from this analysis identified variables describing forest structure (e.g., basal area per ha, height, and number of stories) and the variation in fire severity on the soils and crowns and determined whether these sets of variables were related to each other. The unfortunate aspect of canonical correlation is that, although it is mathematically elegant, results are difficult to interpret (Tabachnick and Fidell 2001) because they express the data in multi-dimensional space. However, from an exploratory perspective, the analysis did reveal that the relations between soil surface fire severity and forest structure are multivariate. The variability in these data is best described in three dimensions (up to 97 percent). Because the relations between soil surface fire severity and stand structure are multivariate, there are many soil and overstory variables that describe the relations among tree and stand characteristics and soil surface fire severity. This finding quickly showed that no single overstory characteristic such as tree density controls the impact wildfires have on soil surface fire severity; rather, combinations of structural characteristics interact to determine how a wildfire impacts the soil surface.

Soil characteristics relevant to fire severity included the mineral and litter components within the unburned, light char, and moderate char classes. Deep soil char did not appear to be as related to forest structure, most likely because it only occurred in isolated areas. Similarly, shrub, grass, and herbaceous cover were not important for describing fire severity because they too were not present throughout the burned areas. Therefore, it was difficult to evaluate the importance of these variables as to their relations with forest structure. Crown severity variables included percent crown scorch within the green, brown, and black scorch classes and scorch height (table 2). Variables important for describing forest structure included those associated with tree density, total biomass, biomass distribution, and vertical crown distribution (table 3).

The fire severity variables identified by the canonical correlation were used in cluster analysis to determine if the fire severity descriptors could be grouped into distinct classes. Results from the cluster analysis were disappointing in that concise clusters of fire severity (low, medium, and high) were not identified. To address this challenge, we are pursuing several avenues, such as using ordination techniques to determine if fire severity can be analytically classified. In addition to attempting to classify fire severity analytically, we also are attempting to identify meaningful thresholds noted in the scientific literature (e.g., Hungerford and others 1991; Johansen and others 2001; Niwa and others 2001; Jurgensen and others 1997).

Relationship Between Forest Structure and Fire Severity

To evaluate whether a relation between forest structure and fire severity could be determined, we post-classified fire severity using variables identified in the canonical correlation analysis and supplemented these classifications with information on fire effects on soils and vegetation (Omi and Kalaokidis 1991; Ryan and Noste 1980; Wells and others 1979). However, the classifications we developed are preliminary and may change depending on further investigation. The purpose for using our current fire severity classifications is to investigate ways to identify relations between forest structure and fire severity.

The fire severity classification for tree crowns used four classes: (1) entire crown contained green needles (no sign of fire), (2) crown dominated by green needles but with the presence of brown needles and/or blackened crowns (charred branches with all needles consumed by the fire), (3) crown dominated by brown needles but with the presence of some green and/or black branches, and (4) crown dominated by black branches with only a trace of brown needles. We separated scorched trees from totally black trees because when brown needles fall to the forest floor they decrease soil erosion and provide organic matter to the soil (Jurgensen and others 1997, Pannkuk and Robichaud 2003). Therefore, fire severity was considered less severe on sites with brown needles present on trees compared to trees where all needles were consumed. After each tree was assigned a fire severity class, these data were summarized to an average crown fire severity for the plot. These values were placed into a severity class and used in the analysis. An average

Table 4—Cross-validation matrix showing how well the overall model correctly classified tree severity. The values on the diagonal provide the probability of correctly classifying the actual fire severity given the forest structure variables used in the model.

	Predicted class			
Actual class	**No fire**	**Green crowns**	**Brown crowns**	**Black crowns**
No fire	0.62	0.17	0.10	0.11
Green crowns	0.05	0.40	0.39	0.16
Brown crowns	0.08	0.39	0.35	0.48
Black crowns	0.11	0.28	0.26	0.34

crown severity between 1 and 1.50 was classified as green (class 1), average crown severity between 1.51 and 2.50 was classified as green to brown (class 2), an average crown severity between 2.51 and 3.50 was classified as brown (class 3), and an average crown severity >3.50 was classified as black.

The results from the canonical correlation indicated that litter and mineral soil in all char classes were related to soil surface fire severity. Moreover, surface organic matter (litter, humus, and brown cubical rotten wood) plays many roles in forest nutrition (Jurgensen and others 1997). Therefore, soil severity classes were based on the presence or absence of surface organic materials and their level of burning. The soil surface fire severity classes were defined as follows: unburned litter dominated the plot (class 1), lightly burned litter dominated the plot (class 2), unburned or lightly burned mineral soil dominated plot with litter present (class 3), moderately burned mineral soil dominated plot with litter present (class 4), and 100 percent of plot exhibited burned mineral soil with no litter present (class 5).

To identify relations between forest structure (only overstory forest structure characteristics were used) and both crown and soil surface fire severity, we used a nonparametric classification and regression tree (CART) technique (Steinberg and Colla 1997). CART does not require the normalization of data through transformations, making the results readily interpretable; it identifies interactions, maximizes homogeneity within a particular classification, and can conduct internal cross-validation (checks how a model generalizes to new data) among classes (see table 4 for cross-validation matrix). Most of our forest structure data were continuous (table 3) and our fire severity data categorical, which can be problematic for many analytical techniques that attempt to relate the two. However, CART partitions data using a binary decision process making it appropriate for both categorical and continuous data. CART produces trees with "nodes" showing where splits in the classifications occurred. Based on decision rules, CART classifies observations until either (1) every observation in the outcome is classified correctly or (2) the outcome contains equal proportions of classes or contains the minimum number of observations specified. In this particular analysis, we specified a minimum number of 30 observations left in the node. Forest structure characteristics occurring at the top of a classification tree provide an indication that they were clearly related to fire severity, compared to characteristics that appear later in the tree. CART can also identify thresholds in relations. For example, when crown base height was identified as an important characteristic for describing crown severity, it occurred at the top of the tree; CART then identified the crown base height at which the greatest number of observations were classified correctly (figure 3a). In addition, CART provides a probability of this relationship (figure 3b).

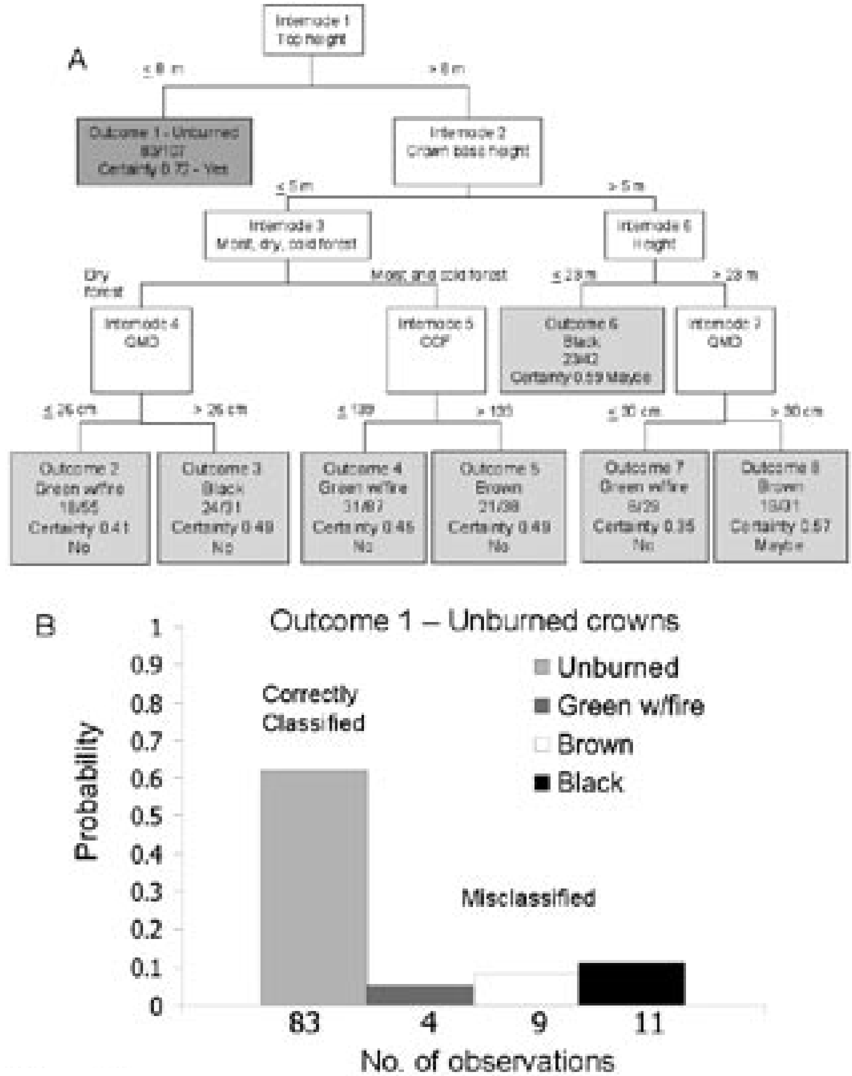

Figure 3—Figure 3a shows an eight-outcome-node classification tree used for predicting crown scorch as a function of pre-wildfire forest structure. Outcomes (shaded, 1 through 8) show number of observations correctly classified, total number of observations, probability of certainty, and whether or not the forest structure characteristic is related to crown scorch (yes, no, or maybe). The lower the probability of certainty the more likely there is no relationship. Internodes (non-shaded, 1 through 7) show the forest structure characteristics used in the split and the threshold where the split occurred (e.g., top height ≤8 m went left to outcome 1). Forest structure characteristics used to split the data at the internodes included top height (m), crown base height (m), forest type (dry, moist, and cold), quadratic mean diameter (QMD), and crown competition factor (CCF). **Figure 3b** illustrates the probabilities after cross-validation associated with predicting unburned crowns (Outcome 1, figure 3a). In this outcome, there is a 0.62 probability that trees less than ≤8 m tall were correctly classified as having unburned crowns (83 of 107 observations). Twenty-four observations were misclassified that actually contained green crowns with fire, brown crowns, or black crowns. Crown severity was placed into four classes. Unburned class was where the crown had green needles and no sign of fire. The green crown with fire class was where green needles dominated the entire crown, with the presence of scorched brown needles and/or black branches. Brown crown class was where entire crown was dominated by scorched brown needles but may have had green or no needles left. Black crown class was where the entire crown was scorched with no needles left or the crown had very few scorched needles.

There are several ways to measure forest density; for example, basal area per hectare, crown competition factor (CCF), total cubic feet per ha, trees per ha, and canopy bulk density. Both in CART and in canonical correlation analysis, canopy bulk density (key variable used in fire models) as calculated by FFE-FVS when included with other density measurements was never involved in any relations with fire severity. However, when canopy bulk density was the only density measurement used, it was included in the relationships. Similar results where noted by Omi and Martinson (2001) when they related canopy bulk density to fire severity. We inferred from this result that canopy bulk density might not reflect variation in density among sites as well as other density measurements such as CCF or basal area per ha.

Forest Structure and Crown Severity

The results from CART were encouraging because they identified forest structure characteristics that were related to fire severity, plus they provided an indication of the strength and weaknesses of these relations. When predicting crown severity as a function of pre-fire overstory forest structure, the model explained 36 percent of the variation in the data. This particular model performed fairly well at classifying sites with no evidence of fire in tree crowns versus areas that tended to contain trees with burned crowns. Sites containing non-burned tree crowns had a 0.62 probability of being correctly classified. In contrast, the model showed a 0.40 probability of classifying burned sites with green crowns present, a 0.35 probability of classifying trees with brown crowns, and a 0.34 probability of classifying trees with no needles left after the fire (table 4).

The classification tree contained eight outcomes as a function of forest structure (figure 3a). Outcomes (shaded) show the number of observations correctly classified, total number of observations, and the outcome's certainty (the probability of correctly classifying the fire severity on a new observation not included in the CART model). Internodes show the forest structure characteristics used in the split and the threshold where the split occurred (e.g., top height ≤8 m went left to outcome 1). The first split in the tree was top height at 8 m tall (figure 3a). There were 107 observations in outcome 1, which contained trees ≤8 m tall, 83 of the plots were correctly classified as containing unburned crowns resulting in a 0.72 probability of certainty. Outcome 1 indicates that, yes, there is a relationship between top height and fire severity.

The certainty of other outcomes is much less when compared to outcome 1. Moreover, a combination of forest structure characteristics is required to obtain one or more of these other outcomes (outcomes 2 through 8). For example, two outcomes (6 and 8) might (maybe) have a relation between forest structure (combination of top height, crown base height, and tree diameter) and crown severity (figure 3a). In outcome 6, trees were between 8 and 23 m (internode 1 and 6) tall and have crown base heights >5 m (internode 2), which resulted in a 0.59 probability of certainty where 23 of the 42 observations were classified correctly. Outcome 8 contains trees taller than 23 m that have crown base high heights >5 m and have a diameter >30 cm with a 0.57 probability of certainty. These outcomes either contained black or brown crowns indicating these characteristics tend to favor high fire severities in the crowns.

The other outcomes (2, 3, 4, 5, and 7) all have certainty probabilities <0.50 and can either contain black, green crowns with fire present, or brown crowns (figure 3a). Several observations were misclassified, indicating a substantial amount of variation in these outcomes. Outcome 7 was classified as containing green trees with an indication of fire; it contained crown base heights >5 m with diameters <30 cm but its probability of certainty was only 0.35. Upon further investigation, seven observations contained green trees with no sign of fire, and the residual observations contained either brown or green crowns. This ambiguous outcome may be a function of our fire severity classification and probabilities may improve with different breaks in our crown severity rating.

The strength of this model is identifying that young stands (short) with top heights less than 8 m have a low crown severity rating (green crowns) (figure 3b). After cross-validation, there is a 0.62 probability that areas with short trees (<8 m tall) were fairly resilient to fire; however, 20 observations still experienced moderate to high (brown or black) fire severities (figure 3b). There were several observations that were correctly classified and 24 of the 107 observations were misclassified.

Another important aspect of the model is to observe the entire classification tree to determine which forest structure characteristics were related to fire severity and which characteristics have either no relationship or a weak relationship to fire severity. Based on location of splits (figure 3a) (top versus bottom) and which structure characteristics were used in the splits, this model indicates that top height and crown base height play more of a role in relating to fire severity than density or size (QMD)—particularly top height, since it was the first variable used in the tree and was related to sites that contained no fire.

Forest Structure and Soil Severity

The relation of soil surface fire severity to forest structural characteristics was weak at best. The overall model explained 20 percent of the variation, and the only factor that was somewhat related to soil surface fire severity was tree height (figure 4). Observations with trees ≤15 m tall, tended to have unburned litter, but there were many observations that were incorrectly classified (139 observations) (figure 4). If sites contained trees >15 m tall,

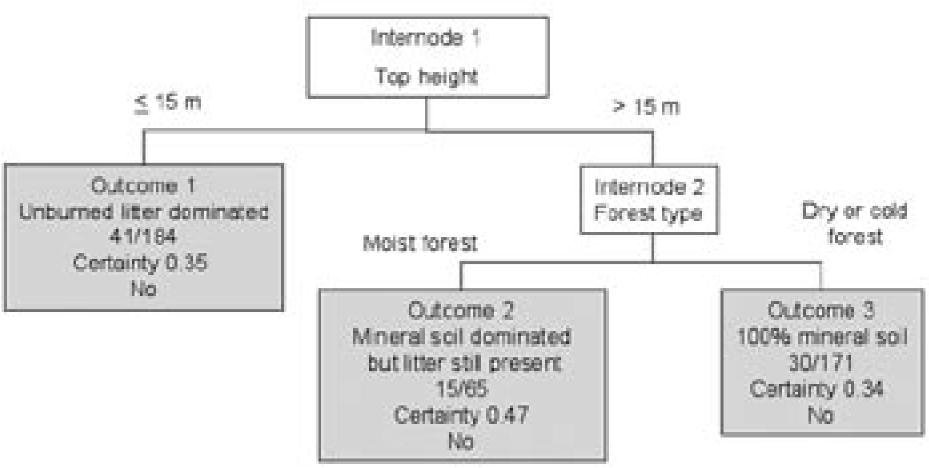

Figure 4—A three-outcome-node classification tree used for predicting soil fire severity as a function of pre-wildfire forest structure. Outcomes (shaded, 1 through 3) show number of observations correctly classified, total number of observations, probability of certainty, and whether or not the forest structure characteristic is related to crown scorch (yes, no, or maybe). The lower the probability of certainty the more likely there is no relationship. Internodes (non-shaded, 1 and 2) show the forest structure characteristics used in the split and the threshold where the split occurred (e.g., top height ≤15 m went left to outcome 1). Forest structure characteristics used to split the data at the internodes included top height (m) and forest type (dry, moist, and cold). Soil fire severity was defined as (1) unburned litter dominated the plot, (2) lightly burned litter dominated the plot, (3) unburned or lightly burned mineral soil dominated plot, litter still present, (4) moderately burned mineral soil dominated plot, litter still present, and (5) 100 percent of plot is mineral soil no litter present.

the classification tree split the difference in soil surface fire severity based on forest cover type (moist versus dry and cold forests). If sites occurred on moist forests they tended to have less severe soil surface fire severity (<100 percent mineral soil exposure with litter present) than when they occurred on either dry or cold forests (100 percent mineral soil exposure). Because of the low estimates of certainty (<0.47) and the misclassification of many observations, we inferred from this analysis that a relation between overstory forest structure and soil surface fire severity may not exist. Several factors may have contributed to these results: (1) overstory trees have little or no relation to soil surface fire severity, (2) fire severity for the soils is poorly classified, and (3) structural characteristics as currently defined are not related to soil surface fire severity.

Conclusion

Although these results are preliminary, they do provide an indication that data from this study will provide information on the relation between forest structure and wildfire severity. It will: (1) provide key structural characteristics related to fire severity, (2) identify thresholds in structural characteristics so they can be applied when treating forest stands, 3) provide useful results that can be incorporated into models, (4) give an estimate of risk or certainty of a particular fire severity within a stand containing identified structural characteristics, and (5) provide empirical probability distributions showing the relations between fire severity and forest structure, which we currently lack.

Acknowledgments

Several people played critical roles in conducting this research. Special thanks to the members of the Research Work Unit 4155; Dennis Ferguson, Project Leader for securing the funding for this study; and Bill Wykoff for providing valuable input on the sampling design. Nick Crookston and John Byrne gave excellent reviews of the study plan. John Byrne was instrumental in hiring crews and acquiring the necessary vehicles and equipment. Dan Frigard, Don Shipton, Teresa Rockwood, John Lhotka, Jonathon Sandquist, and Bob Denner truly made this work a success; without their hard work and resiliency, we would have never succeeded. We would like to especially thank Elizabeth Reinhardt and Bob Keene for taking the time to review and comment on the study plan and to Michael Battaglia and Ann Abbott for a thorough review of the manuscript. In particular, we would like to thank Kevin Ryan not only for providing an excellent review and good suggestions, but also for the strong support we have received.

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Lessons Learned in 84-Year-Old Plots at Looking-Glass Rock, North Carolina

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Abstract—*Looking Glass Rock is a large, exposed granite pluton that is a special place for recreation and wildlife in the Pisgah National Forest. Even-aged timber stands surrounding the base of the rock originated in 1916 from clearcutting of the original mixed-species virgin stands. Two species now account for 86 percent of the 211 ft^2 of stand basal area per acre: overstory yellow-poplar (81 percent) and mid-story silverbell (5 percent). Timber management is not a current option in this area. If options change, however, research results suggest several techniques for managing and regenerating this and other stands on productive sites.*

Introduction

Looking Glass Rock (LGR) is a large, exposed granitic pluton that is widely recognized as a special place in the Pisgah National Forest of western North Carolina (figure 1). Rising nearly 1000 feet above the surrounding landscape, LGR is readily visible from many locations, particularly the nearby Blue Ridge Parkway, and is proposed for the National Registry of Natural Landmarks. LGR is managed as a special interest area by the Pisgah District to provide nesting and foraging habitat for the threatened peregrine falcon, to preserve the unique botanical features characteristic of the xeric habitats associated with thin soils on the top of the pluton, and to maintain the scenic attributes desirable for recreational uses, such as hiking and rock climbing. The 1600-acre LGR area also is exceptional because it contains the oldest hardwood silvicultural research plots in the southern Appalachians.

Figure 1—The Looking-Glass Rock special interest area in the Pisgah National Forest, as viewed from the Blue Ridge Parkway. Photo by Bill Lea, www.billlea.com.

[1] USDA Forest Service, Southern Research Station, Asheville, NC.

[2] USDA Forest Service, Pisgah National Forest, Pisgah Forest, NC.

The plots were installed in the fall of 1923 to evaluate growth and mortality of natural regeneration after clearcutting.

The typical recreation user probably does not know that timber stands surrounding LGR are relatively young and largely even-aged. The stands originated after a wildfire in logging debris from cutting in 1913 of the last virgin stand on George Vanderbilt's Pisgah Forest, which later became the nucleus of the Pisgah National Forest. Of particular interest to silviculturists, however, is the mesophytic stand that developed naturally on the east-facing, high-quality site at the base of LGR. Development of this stand has been reported at ages 18 years (Abell 1935), 42 years (Wahlenberg 1950), 51 years (Della-Bianca 1971), and 60 years Della-Bianca (1983). In the 1983 report, Della-Bianca suggested that one of the pioneer species, yellow-poplar (*Liriodendron tulipifera*), would not relinquish domination of the stand in the near future. Few examples are available of long-term changes in species composition after clearcutting of stands on high-quality sites in the southern Appalachians. One purpose of our paper is to update stand development at LGR, which was last reported 20 years ago. Another purpose is to explore management options that could be applied to similar stands. Although this and other stands around LGR will not be managed for timber production, research findings from nearby Bent Creek Experimental Forest provide methods that can be used to manage stands on similar sites.

Methods

Study Area

LGR is about 30 miles south of Asheville, NC, and 5 miles northwest of Brevard. Elevation is about 3100 feet at the base of the rock and 4000 feet at the top. Climate in this area is temperate, with short, cool winters and long, warm summers. Precipitation is plentiful and evenly distributed, with over 65 inches annually. Bedrock geology is mainly gneisses and schists. The intrusive LGR pluton, however, consists of resistant Whiteside granite, which has been exposed through weathering and erosion of the softer surrounding rocks. Canopy vegetation at this intermediate elevation consists of several species of oaks (*Quercus* spp.) and pines (*Pinus* spp.) on the xeric to subxeric ridges and slopes, and yellow-poplar, sweet birch (*Betula lenta*), northern red oak (*Q. rubra*), and a number of other species on submesic and mesic lower slopes and coves. Midstory vegetation consists of sourwood (*Oxydendrun arboretum*), dogwood (*Cornus florida*), red maple (*Acer rubrum*), and silverbell (*Halesia carolina*). An evergreen shrub layer of mountain laurel (*Kalmia latifolia*) may be present on xeric and subxeric sites, and rhododendron (*Rhododendron maximum*) is common on some mesic to subhydric sites. Common forms of natural disturbance include crown and stem breakage and uprooting from extensive ice storms and small patches of windthrow from localized, intensive microbursts associated with thunderstorms.

Timber Stand Studied

The original LGR stand consisted of mixed species, mainly American chestnut (*Castanea dentata*), yellow-poplar, and some cucumbertree (*Magnolia acuminata*). Volumes averaged about 40 thousand board-feet per acre. The stand of about 120 trees per acre was cut in 1913 using a diameter limit of 14 inches DBH for chestnut and chestnut oak (*Q. prinus*), and 16 inches for yellow-poplar and other species. Surveying the stand after logging and seeing only snags and culls, Frothingham[3] said, "Lumbermen would call the area clear cut." Wildfire burned through the logging slash on May 11, 1916. By 1919, abundant seedlings of yellow-poplar had become established, as often occurred following severe fires on moist sites in the Southern Appalachians (Frothingham 1931, p. 41). However, the yellow-poplar seedlings and sprouts of other desirable species were mostly shaded by a dense stand of undesirable staghorn sumac (*Rhus typhina*) and silverbell. Silviculturists of the day were concerned that the stand of desirable species might not emerge.

Four sample plots were established in fall 1923 near the base of the northeastern side of LGR to evaluate the feasibility of releasing desirable regeneration from the competition of undesirable species. The treatment, termed "cleaning," removed sprouts of chestnut, sumac, and silverbell from two of the plots. Della-Bianca (1983), however, found little practical difference between treatments in this case study at age 60. In October 1999, after 84 years of growth, we inventoried all arborescent vegetation ≥0.6 inches DBH on each of the four 0.5-acre permanent plots. Because 1923 treatment effects were no longer evident, we pooled data from the cleaned and uncleaned plots. Site index averages 110 feet for yellow-poplar at 50 years (Della-Bianca 1971).

Results

Fifteen species are now present on the plots (table 1). In terms of stem numbers, the present LGR stand is dominated by two understory and one

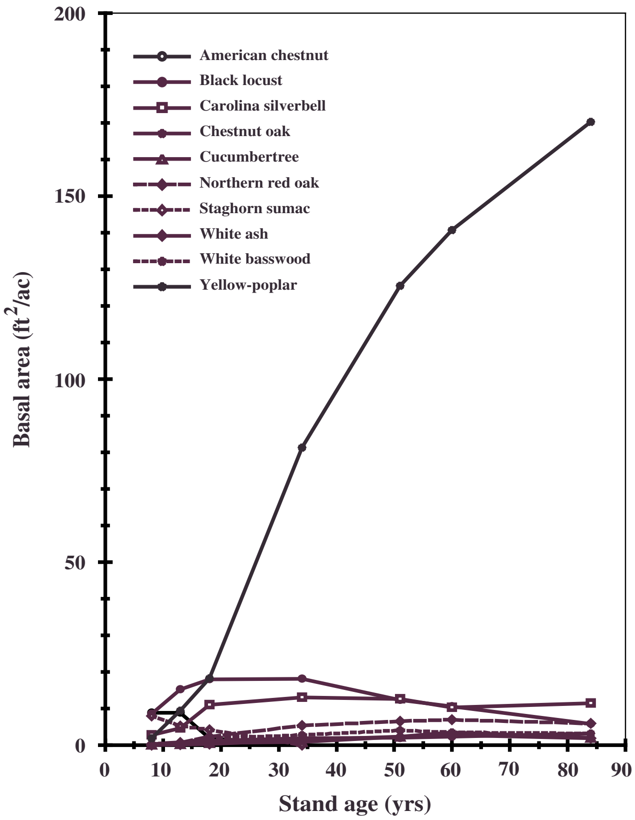
Table 1—Numbers of trees ≥0.6 inches DBH by species, at seven stand ages on permanent plots at Looking-Glass Rock, Pisgah National Forest, NC.

	Stand age (years						
Arborescent species[a]	**8**	**13**	**18**	**34**	**51**	**60**	**84**
American chestnut	324	97	108	0	0	0	0
Black locust	222	368	71	61	23	17	5
Carolina silverbell	205	227	298	171	107	235	359
Chestnut oak	21	28	29	9	2	2	2
Cucumbertree	9	18	10	8	7	8	3
Fraser magnolia	[b]	[b]	[b]	[b]	[b]	[b]	8
Northern red oak	30	41	46	23	10	8	4
Staghorn sumac	828	751	628	0	0	0	0
Striped maple	[b]	[b]	[b]	[b]	[b]	[b]	94
White ash	2	5	3	6	5	5	2
White basswood	18	32	23	10	6	6	3
Yellow-poplar	182	890	422	402	188	139	96
Other[c]	78	92	108	61	25	69	15
All live	1919	2549	1746	751	373	489	591
All standing dead	[d]	[d]	[d]	[d]	[d]	[d]	47

[a] Common and scientific names: American chestnut, *Castanea dentata*; black locust, *Robinia pseudoacacia*; Carolina silverbell, *Halesia carolina*; Chestnut oak, *Quercus prinus*; cucumbertree, *Magnolia acuminata*; Fraser magnolia, *Magnolia fraseri*; northern red oak, *Q. rubra*; staghorn sumac, *Rhus typhina*; striped maple, *Acer pensylvanicum*; white ash, *Fraxinus americana*; white basswood, *Tilia heterophylla*; yellow-poplar, *Liriodendron tulipifera*; flowering dogwood, *Cornus florida*; hickory spp. *Carya spp.*; red maple, *Acer rubrum*; sourwood, *Oxydendrun arboretum*; sweet birch, *Betula lenta*.
[b] Included with "Other" category in previous reports (Della-Bianca 1983).
[c] Includes: Flowering dogwood, hickory spp., red maple, sourwood, and sweet birch.
[d] Not measured in previous inventories.

[3] Frothingham, E.H. 1917. Report on study of cutover areas in the southern Appalachians. Unpublished report on file at Bent Creek Experimental Forest, USDA Forest Service, Southern Research Station, Bent Creek Experimental Forest, Asheville, NC.

Figure 2—Basal area by tree species at seven stand ages on research plots at Looking-Glass Rock. Not shown are basal areas of *Fraser magnolia* (striped maple) and five other minor species (listed in table 1), which totaled 7.36 ft²/ac at age 84.



overstory species. About 77 percent of the stems are silverbell and striped maple (*Acer pensylvanicum*) in the understory. Yellow-poplar is the dominant canopy species but accounts for only 16 percent of stem numbers. Silverbell and yellow-poplar have always been a major component of the stand, but staghorn sumac, American chestnut, and black locust (*Robinia pseudoacacia*) were more important in the early history of the stand. Total stem density declined until age 51, then increased as shade tolerant species grew large enough (≥0.6 inches DBH) to be tallied.

Yellow-poplar accounts for 81 percent of the total stand basal area of 211 ft² per acre (figure 2, table 2). Three other species, Carolina silverbell, black locust, and northern red oak make up 11 percent of basal area. For most species, basal area has decreased over time, particularly between ages 60 and 84, when considerable mortality occurred. Basal area of yellow-poplar, however, has increased consistently. The mean annual basal area increment for the stand was 2.51 ft² per acre. Since age 60, annual increment has averaged 1.21 ft² per acre.

Discussion

Long-term observations of stand development at LGR and elsewhere suggest that low- to moderate-elevation, high-quality sites in the southern Appalachians will regenerate quickly after clearcutting and will usually be dominated by a single desirable species: yellow-poplar. Data from the LGR plots demonstrates the consequences of clearcutting on good sites in the southern Appalachians where yellow-poplar reproduction consists of sprouts from cut stumps and seedlings from buried seeds. The fire after the harvest at LGR likely helped rather than hindered yellow-poplar establishment by

Table 2—Plotting points for figure 2 by species: basal area in ft²/acre at seven stand ages.

	Stand age (years)						
Arborescent species	**8**	**13**	**18**	**34**	**51**	**60**	**84**
American chestnut	8.84	8.88	1.83	0	0	0	0
Black locust	8.62	15.24	18.03	18.12	12.28	10.63	5.84
Carolina silverbell	2.70	4.62	11.02	13.11	12.63	10.24	11.41
Chestnut oak	0.17	0.44	1.22	1.83	2.05	2.26	3.18
Cucumbertree	0.09	0.35	0.61	0.87	2.31	3.48	1.96
Fraser magnolia	[a]	[a]	[a]	[a]	[a]	[a]	3.31
Northern red oak	0.26	0.78	2.35	5.40	6.53	6.97	5.88
Staghorn sumac	7.93	5.23	4.22	0	0	0	0
Striped maple	[a]	[a]	[a]	[a]	[a]	[a]	2.22
White ash	<0.05	0.09	0.22	1.39	2.35	2.70	1.92
White basswood	0.26	0.65	1.92	2.70	4.09	3.44	3.22
Yellow-poplar	1.83	9.49	18.29	81.27	125.44	140.77	170.30
Other [b]	0.96	1.74	3.57	5.44	1.70	2.09	1.83
All live	31.70	47.51	65.07	130.10	169.43	182.49	211.06
All standing dead	[c]	[c]	[c]	[c]	[c]	[c]	16.42

[a] Included with "Other" category in previous reports (Della-Bianca 1983).
[b] Includes: Flowering dogwood, hickory spp., red maple, sourwood, and sweet birch.
[c] Not measured in previous inventories.

top-killing competition and exposing mineral soil, which aided germination of both stored and new seeds from a few surviving seed trees. Yellow-poplar can capture the canopy of high-quality sites from a cohort of mixed species soon after harvest and hold it for many, many years. Dominance will likely be maintained until the occurrence of some significant overstory disturbance, such as crown breakage from an ice storm, which allows additional light on the forest floor to stimulate development of advance regeneration of species with intermediate shade tolerance, such as some oaks. At LGR, however, development of intermediate species on the forest floor resulting from crown disturbance is unlikely because the dense midstory of the shade-tolerant silverbell and striped maple will intercept much of the additional light.

The findings about regeneration and stand dynamics at LGR should be applicable on other high-quality sites in the southern Appalachians. Such sites, which cover moist slopes and coves, include about a third of the landscape (Frothingham 1931). On a high-quality site in the nearby Bent Creek Experimental Forest, Beck and Hooper (1986) also found that yellow-poplar dominated 20 years after the original stand of mixed species was clearcut. One interesting difference between these two stands is composition of the midstory. Silverbell does not occur naturally at Bent Creek and significant amounts of dogwood and red maple are lacking at LGR. Loftis (1989) likewise found that yellow-poplar was a major component of stands regenerating high-quality sites after clearcutting, but the proportions were more variable than those at LGR and Bent Creek clearcuts. When yellow-poplar is present in a stand at crown closure, it is likely to maintain or gain canopy space because of its rapid growth and longevity, unless natural disturbances or silvicultural treatments occur.

Although the LGR stand will not be managed for timber production, we can speculate on outcomes of hypothetical management scenarios based on results of research at Bent Creek. Even though the LGR stand is almost 90 years old, approaching a time when we might consider regeneration, continued management of the present thrifty and likely long-lived yellow-poplar stand also could be attractive. One or more intermediate cuttings could be prescribed to salvage mortality and concentrate growth on the best trees. Low thinning, for example, would increase the average diameter of the residual stand and increase radial growth for several years. Larger trees would be produced more quickly than without silvicultural activity (Beck and Della-Bianca 1981).

Due to high stand volumes and the longevity of yellow-poplar, a number of regeneration options are available. Single-tree selection has not been successful in yellow-poplar stands (Della-Bianca and Beck 1985), but group selection is feasible with group openings as small as one-quarter acre (Beck 1988). Treatment of midstory competition in the openings would encourage yellow-poplar regeneration from seed and sprouts. Representation of other species would depend on the presence of large advance reproduction at the time of cutting and stump sprouts created by cutting (Loftis 1985).

The longevity of yellow-poplar provides great flexibility in converting even-aged stands to uneven-aged stands. Two-aged stand management will work (Beck 1987). The low residual basal areas required for the creation of two-aged stands is reached with a single cut or with two cuts. The regeneration will be dominated by seedlings and sprouts of yellow-poplar as long as midstory competition is controlled. Again, the longevity of the residual yellow-poplar will provide several options as the new age class develops. At mid-rotation, one could remove the older age class in a selection thinning, leaving an even-aged stand. Alternatively one could leave the two-aged stand in place until the end of the rotation. Another option would be to begin the regeneration process again, leaving residual trees of both existing age classes to create an irregular structure with multiple age-classes.

Research has also shown that conventional shelterwood systems can regenerate these stands (Loftis 1983). Over a broad range of residual basal areas, species composition of regeneration has been roughly the same as would have been expected after application of the clearcutting method, as long as the overwood is removed in a timely manner. Yellow-poplar has typically dominated the species composition of new stands. Other species have been present when they occurred as advance reproduction in the previous stand or regenerated from stump sprouts.

Research at Bent Creek suggests that it is possible to change the species composition in stands dominated by yellow-poplar, increasing the amounts of other species such as northern red oak or white ash (*Fraxinus americana*). Loftis (1990) used herbicide treatments to reduce shading from tolerant midstory species, such as silverbell and dogwood, to allow development of desirable species with intermediate shade tolerance. This treatment can be applied in a uniform fashion across an entire stand or in a group or patchwise fashion. After large advance reproduction of these species has developed, usually in about 10 years, the overstory can be removed in any number of ways to provide timely release of the advance reproduction and representation of these species in the new stand.

In summary, LGR provides today important botanical, wildlife, and recreation resources on the Pisgah Ranger District. Equally important to silviculturists, but not so obvious, are the long-term lessons available from observations of stand development on the oldest regeneration research plots in the southern Appalachians. There and elsewhere on good-quality sites in the southern Appalachians, one desirable species, yellow-poplar, has been a significant component of stands regenerated by clearcutting. Yellow-poplar tends to dominate those stands for many years unless a natural disturbance occurs or a silvicultural treatment is imposed to achieve other management objectives.

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Poster Abstracts

Stocking Conditions Influence Strength of Small Diameter Ponderosa Pine Wood Harvested in Forest Restoration Treatments

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Treatment to reduce risk of catastrophic fire in ponderosa pine forests of the Colorado Front Range requires removal of large numbers of small diameter trees. Many of these trees have grown in highly competitive environments and it is unknown what effect these conditions have on strength properties. We measured strength properties of 9 to 11 inch diameter open-grown (n=15), suppressed (n=20), and mistletoe-infected (n=21) ponderosa pine trees. Strength properties were compared to the basal area increment (BAI), heartwood-sapwood ratio, and forest density surrounding each tree. Growth rates of open grown trees were 50 percent higher than suppressed and mistletoe trees, while age and heartwood-sapwood ratio were lower in open grown trees. Modulus of rupture (MOR) was similar for suppressed and mistletoe trees but lower for open grown trees. Modulus of elasticity (MOE) was highest in the suppressed trees and lowest in open grown trees. MOR and MOE increased with increasing stand basal area and tree ring density but decreased with increasing individual stem BAI. Different competitive environments impact growth rates, which influence the strength of small diameter wood. Small diameter trees removed from dense Front Range forests appear to be superior in strength than rapidly growing young trees of the same size.

The Forests of Washington and Oregon in the 1930s

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Many public land managers are interested in restoring stands and landscapes to a condition within the historical range of variability. To accomplish this objective, managers need information about past forest conditions. Historical vegetation information, particularly at broad scales, is not readily available.

The Pacific Northwest Forest Experiment Station surveyed Washington and Oregon forests in the early 1930s and prepared maps showing species composition and tree size. Prior to the 1930s, national forest managers relied on extensive reconnaissance mapping that tended to delineate areas based on timber volume (2,000 to 5,000 board feet per acre, etc.) rather than species composition or tree size. The extensive reconnaissance mapping had somewhat limited utility because it did not include private or other lands not in national forest ownership.

No mapping protocol is completely free of bias, but the 1930s forest survey brought a new standard of consistency and accuracy to forest mapping for the Pacific Northwest. Although the forest type maps were widely used until the late 1950s, many contemporary managers are unaware of their existence. Digital versions of the 1930s forest type maps (at an original scale of 1:253,440) have recently become available. These maps are referred to as "quarter-state" maps because four sheets were produced for each state – one sheet for each quarter.

This poster provides a brief overview of the 1930s forest survey methodology and includes examples of data from several locations, as well as the interpretation of that data. It also indicates how to obtain a CD that includes a history of the forest survey, excerpts from early publications produced by the survey unit, and the actual forest type maps in several GIS or graphics formats. These maps should be very useful to resource managers who need broad-scale information about forest composition and structure.

County-level mapping from the mid 1930s has been located for some counties. The county-level maps were published at a scale of 1:63,360. They are valuable for fine-scale analysis because the tree species and size information is mapped at a finer scale, and because they provide information about stocking, age, associated tree species, and certain disturbance factors or damaging agents such as timber harvest, wind, and fire. Note that the "quarter-state" maps described above do not include this level of detail because it was not possible to depict it on a large-scale format (1:253,440).

The county maps have been particularly useful for "ecosystem analysis at the watershed scale" (e.g., watershed analysis), a mid-scale procedure for characterizing the human, aquatic, riparian, and terrestrial components of a watershed. Ecosystem analysis is designed to set the stage for subsequent decision-making processes by providing context for fine-scale project planning. Historical maps are a critical data source for step four in a six-step ecosystem analysis process; step 4 produces a description of reference conditions for the watershed.

On the Umatilla National Forest in the Blue Mountains of northeastern Oregon and southeastern Washington, the 1930s county-level mapping is a primary data source when characterizing reference conditions. This is accomplished by using a database lookup (cross-walk) table that relates map attributes (the legend codes) with an associated cover type, stand size class, stocking category, and structural class. Reference and current conditions are then compared and interpreted before formulating management recommendations.

Information about the availability and format of the 1930s forest type mapping is available from these contact persons:

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Alternative Treatments for Fuel Reduction and Carbon Retention at the Forest/Urban Interface

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Weak vegetation control in western forests raises the risk of catastrophic wildfire from the buildup of understory fuels. The problem is particularly acute at California's forest/urban interface where rural communities often are bordered by open pine stands choked with woody shrubs. Prescribed fire is an impractical solution because of air quality and liability concerns. Burning also increases CO_2 emissions, reduces the site's carbon storage, and degrades soil quality. In our operational scale experiment, alternative fuel treatments funded through the National Fire Plan include a control, mastication + prescribed fire, hand removal, mechanical mastication, and mastication + tillage of chips into the surface soil. This affords a series of comparisons on the economic and environmental costs of reducing fuels and exporting or retaining organic carbon. This new experiment has been replicated at four sites in California. Results show that understory fuels can reach 41 tons/acre and store up to 20 tons of carbon and 700 million BTUs of potential energy/acre. Treatment costs vary ten-fold, from $400/acre (mastication) to $4,000/acre (hand removal). However, these costs pale when compared to the risks of doing nothing. All such treatments reduce fuels immediately, but managers must consider regrowth of understory fuels, and a possible solution is chemical control. A piggybacked study is tracing the environmental fate of hexazinone, glyphosate, and imazapyr in the soil in several of the treatments.

FVS-BGC: A Physiology Approach to Simulating Stand Vigor

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FVS-BGC is a hybrid stand growth model that adds the logic of the Stand-BGC physiological process model to the Forest Service's Forest Vegetation Simulator (FVS). FVS operates on a five- to 10-year time step and assumes average climate conditions and stable site conditions over each time period. In the real world, variable weather conditions affect growth rates, affect mortality rates, and create periods of low tree vigor that contribute to pest epidemics. The "BGC" portion of this model uses climate data to operate on a daily time step. This fine scale approach allows users to project variation in stand vigor conditions among years and estimate moisture stress conditions within a year. Previous studies have found a strong relationship between stand vigor and susceptibility to bark beetle attack. Traditionally, managers have used thinning prescriptions to reduce bark beetle hazards, with a goal of applying them when stand densities exceed a threshold level. Projection systems such as FVS have been used to estimate when stands will exceed that threshold. This approach assumes that stand density is closely linked to stand vigor. Using FVS-BGC, we demonstrate how stand growth and normal climate variability combine to create a wide age and density band during which a vigor-based threshold might be crossed. Using climate records for example stands, we calculate probabilities of exceeding this vigor threshold at different ages. We also demonstrate how the interactions of site characteristics influence stand growth potential and vigor conditions. The desire to maintain forests in certain conditions for nontimber purposes requires an understanding of the potential for creating physiological conditions that may not be viable under some climate conditions. FVS-BGC may be helpful in these cases.

Red River Watershed Forest Health Analysis

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A watershed assessment project of the Red River watershed in the Red River Ranger District of the Nez Perce National Forest, Idaho, provided an opportunity to test and demonstrate a set of newly developed forest health analysis and display tools. The watershed is the site of an ongoing mountain pine beetle epidemic. This disturbance has impacts on wildlife habitat, fuel conditions, and other resource values. The Westwide Pine Beetle (WWPB) Model is a recently developed extension of the Forest Vegetation Simulator (FVS). Within FVS, we used: (1) base FVS model growth and mortality routines; (2) the Event Monitor, to calculate various published bark beetle hazard rating systems; and (3) the WWPB Model to simulate the landscape-level effects of the mountain pine beetle. The impacts of past and potential silvicultural activities were included in these simulations. We used three customized ArcView GIS projects—FVS-EMAP, and two WWPB Model Mapping Tools—to spatially portray FVS output. FVS-EMAP is our newly available system that provides a simple interface to move FVS output into ArcView. We also used the Stand Visualization System (SVS) and Envision to create stand and landscape visualization images of the conditions over time. We show how the use of these tools provides a powerful linkage between stand level and landscape level analyses, helping forest managers address ecosystem management objectives.

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RMRS

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