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# Cumulative Watershed Effects of Fuel Management in the Eastern United States

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# Abstract

As a result of effective fire suppression activities over the last 75 years and a reduction in timber harvesting on the national forests, biomass has accumulated increasing the susceptibility of large and more severe wildfires. Reducing accumulated fuels is now a major management objective on the national forests. A combination of traditional silvicultural treatments such as prescribed fire and thinning and new innovations are needed to address the myriad of site conditions. Effective fuels management should improve the health of the watershed ensuring the sustainability of the goods and services that are derived from the landscape. However, since fuels management necessarily interacts with other land management considerations and often requires periodic treatments, assessing the cumulative effects can be daunting. This volume and a companion volume focusing on the Western United States (Elliott and others 2010) were designed to provide land managers with a synthesis of the science to support an assessment of the cumulative effects of fuels treatments on forested watersheds in the conterminous United States. This volume is organized to into three sections, an overview of the biophysiography of the Eastern United States, consideration of ecosystem components and how fuel treatments may affect specific processes or properties, and the third section synthesizes fuels management practices and effects in the major ecosystem types of the region. The findings provide a sound foundation for assessing the ecological effects of fuels management practices. By necessity much of the information is derived from the literature on silvicultural effects on ecosystem functions; however the authors have interpreted that work from the perspective of fuels management prescriptions. Similarly, fuels management prescriptions are evolving; accordingly, the intent is to convey the science in a way that it will be relevant to new approaches. These chapters are derived through a synthesis of well-founded research and experience, providing a much needed reference on the cumulative watershed effects of fuels management practices.

**Keywords:** Biomass, cumulative watershed effects, environmental impact, forest fuels, fuel management, prescribed fire.



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# Foreword

This volume is the result of a major interdisciplinary effort to synthesize our understanding of the cumulative watershed effects of fuel management in eastern forests. It is intended primarily for national forest field personnel who must develop credible National Environmental Policy Act documents that contain the most current available knowledge of the direct, indirect, and cumulative effects of fuels treatments on a wide variety of resources, including watershed resources. It should also prove useful to land managers in forest industry and other private forest ownerships.

Although the literature contains significant information on intense but localized effects of fire (primarily wildfire), vegetation management, and similar treatments, little information exists on less intense but extensive fuels treatment effects. To fill this knowledge gap, the Rocky Mountain Research Station (Forest Service, U.S. Department of Agriculture) hosted a national workshop in April 2005 in Utah to assess the status of knowledge and outline research needs to fill knowledge gaps. All but four of the participants were from the West and little information addressing eastern knowledge and needs was presented. A second workshop to address eastern issues was held in July 2006 in Georgia. The workshop was a collaborative effort of the Stream Systems Technology Center, Rocky Mountain Research Station (John Potyondy), Southern Research Station (Carl Trettin), Eastern Region (Russ LaFayette), Southern Region (Suzanne Krieger), and State and Private Forestry (Maureen Brooks).

This volume is a testament to what can be accomplished when National Forest Systems, Research and Development, and State and Private Forestry programs of the Forest Service pool their resources and work collaboratively to achieve a common goal. The resulting volume is the product of more than 25 authors and 62 reviewers including scientists from Forest Service Research Stations and several universities. Over a four-year period, authors drafted chapters that were peer-reviewed and edited.

This synthesis is organized somewhat differently from its western counterpart.[1] The eastern volume begins by providing background material for context and then discusses fuel management activities grouped by ecological divisions followed by the physical, chemical, and economic consequences of fuel treatments and methods for analyzing cumulative watershed effects.

The other editors and I are grateful to all authors and reviewers for their considerable efforts and patience in the development of this volume. Special recognition is owed to Carl Trettin, Southern Research Station, and Russ LaFayette, Eastern Region, who advocated for an eastern volume and committed time and talent to make it a reality. We wish to also acknowledge Lisa Audin Duarte, with the support of Dr. Bill Elliot, who contributed greatly to this project. Lisa was one of the editors of the western volume and helped us organize the eastern volume for several years. And finally, Maureen Brooks kept the process on track after Lisa's departure to a new assignment.

As with all syntheses, new knowledge and new science will supersede the contents of this volume. Readers are encouraged to seek updated and locally derived information as needed. Readers are also encouraged to obtain a copy of the western volume for additional useful background perspectives on fuel management.

My personal thanks go to all the authors, reviewers, my coeditors, and Southern Research Station publishing staff for the considerable effort necessary to develop and publish this synthesis.

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[1] Elliot, William J.; Miller, Ina Sue; Audin, Lisa. Eds. 2010. Cumulative watershed effects of fuel management in the Western United States. Gen. Tech. Rep. RMRS-GTR-231. Fort Collins, CO: U.S. Department of Agriculture Forest Service, Rocky Mountain Research Station. 299 p.

# Acknowledgments

This volume represents the collective effort of many persons and organizations. In addition to those previously mentioned in the Foreword to this book, we would like to thank the Southern Research Station's Science Delivery Group as led by Assistant Director Jennifer Plyler, and in particular the Group's Technical Publications staff, with Gary Kuhlmann (Team Leader), Maureen Merriman, Donna Burnett, and Louise Wilde, for supporting the production and publication of this volume. We also appreciate Lee Moreland's help in conducting the first round of technical editing, and Joyce VanDeWater for the cover design.

# Contents

Cumulative Watershed Effects of Fuel Management in the Eastern United States

CHAPTER 1.

# Introduction to Synthesis of Current Science

**Douglas F. Ryan, Russell LaFayette**

Preparation of this report was commissioned to a group of scientists and land managers by the U.S. Department of Agriculture Forest Service, for the purpose of synthesizing current scientific literature to answer an important question facing the managers of Federal, State, and private lands in many parts of the country: At the watershed scale, what potential cumulative environmental effects might result from implementing fuel-reduction activities on forested landscapes? The main body of this report is a compilation of their findings, including both what can and cannot be concluded from the current science.

An earlier synthesis on this topic (Elliot and others 2010) focused primarily on fire regimes, vegetation types, and management practices in the Western United States. Several participants in that initial effort recommended a parallel report focusing on eastern landscapes, species, practices, and conditions. This report is the result of that recommendation. Although western fire conditions tend to be more dramatic, eastern conditions are as important, albeit more subtle. Human uses, particularly those since European settlement, have significantly changed most of the eastern landscape, including its vegetation form and distribution (Nowacki and Abrams 2008). This report reflects these subtle but important changes.

Our synthesis is organized somewhat differently from its western counterpart. The report contains 14 chapters grouped into 3 main topic areas. Chapters 1 through 4 set the stage, providing background material to establish the context for the remaining work. To simplify the presentation, silvicultural types were grouped into several larger categories than the 26 generally accepted cover types (U.S. Department of Agriculture 1983). Chapters 5 through 9 discuss fuels management activities grouped by ecological divisions (fig. 1); the Warm Continental Division (210) and the Savanna Division (410) lack research information on fuel treatments and are therefore not included. Chapters 10 through 14 assess the effects of treatments on the ecosystems described in chapters 5 through 9, including physical, chemical, and economic consequences.

Here in chapter 1 we broadly describe fuel reduction treatments in wildlands and the concept of cumulative watershed effect analysis. For perspective we have referred to some of the primary laws and policies that influence the way that Federal land managers apply fuel reduction treatments and analyze cumulative effects.


**Figure 1.** Ecological divisions and provinces of the Eastern United States (Cleland and others 2007).

# Fuel Reductions Treatments and Policies and Laws Related to Them

Fuel reduction treatments are actions taken to reduce the threat of severe or intense wildland fire by manipulating live and dead vegetation to reduce the loading of fuel on the landscape. Fuel reduction can be accomplished in a number of ways, with the most common involving mechanical removal of fuel material (usually brush or trees), application of herbicides to reduce growth of undesirable species, and consumption of fuel using prescribed fire. These actions may be applied either alone or in combinations. Treatment of wildlands to reduce fuel was given a national mandate when the U.S. Congress passed the Healthy Forests Restoration Act of 2003. Efforts to reduce severe wildfire risk may be advanced by studies that link climate change to recent increases in the frequency and size of fires (Westerling and others 2006), raising concerns that this threat to U.S. forests may increase even further with expected changes in climate.

To effectively reduce the risk of wildland fires, fuel reduction treatments will need to be applied to large areas each year. In most circumstances, new vegetation will grow after treatments, meaning that maintaining low fuel stocking requires repeated treatments at intervals ranging from several years to a few decades. Where they are well designed and implemented, fuel reduction treatments can minimize the intensity of disturbance. However, they are carried out over many acres each year and require return treatments at regular intervals, producing project-level impacts that may be cumulatively significant at larger, watershed scales.

Cumulative effects on watersheds might be caused by combinations of individual activities directly related to removing fuels (such as felling, skidding or chipping to mechanically remove fuel, and prescribed fire to consume it). Cumulative effects might also include the impacts of supporting operations or infrastructure at some distance from the actual site of fuel reduction. Examples of supporting functions include the movement of logging, fire control, and other vehicles used in fuel management. Examples of supporting infrastructure include roads, skid roads, and access landings as well as associated drainage ditches, culverts, and stream crossings. Although the landscape responses to short duration/high intensity disturbances (such as wildfire, final harvest, and site preparation) have been reported extensively, less is known about the effects of less intense treatments implemented repeatedly on large areas over an extended period. The research needed to fill these knowledge gaps is identified below.

In recognition of the critical role of wildland fire in forest ecosystems and the risk that high fuel loads pose in forests, the Federal government has taken several actions to accelerate fuel reduction treatments. Several policy initiatives by the Forest Service and other Federal agencies (the wildland fire policy of 1995 and cohesive fire strategy of 2000, the National Fire Plan of 2000, and the 10-Year Comprehensive Strategy and Implementation Plan of 2001 and 2002) were strengthened when President Bush announced the Healthy Forest Initiative in 2002.

# What Are Cumulative Watershed Effects?

The National Environmental Policy Act (NEPA), requires that Federal agencies disclose the potential direct, indirect, and cumulative environmental effects of proposed alternative land management actions, and that they document those findings in a public report: an Environmental Impact Statement, an Environmental Assessment, or a Categorical Exclusion, depending on the nature and complexity of the action.

The basic premise of a cumulative effects analysis is to identify and consider the total effects of actions that overlap temporally or geographically and that could be missed if each action were evaluated individually. The goal of cumulative effects analysis is to provide decisionmakers and the public with comprehensive information about “the impact on the environment which results from the incremental impact of the action when added to other past, present and reasonably foreseeable future actions” (40 C.F.R. § 1508.7). Cumulative effects may arise from single or multiple actions that may be additive or interactive, direct or indirect. Cumulative effects are the net impact on watersheds of multiple management activities that may coincide geographically or temporally.

Although cumulative effects are defined by NEPA, cumulative watershed effects also come into play in the application of other environmental laws, although in more restricted circumstances. The Clean Water Act requires control of water-pollution nonpoint sources (pollution that does not have an easily identified source) as well as point (identifiable) sources, with particular emphasis on waters that States have designated as impaired (not meeting water quality standards) under section 303(d). Often, nonpoint sources are not easily attributable to well defined sources because they are geographically and/or temporally dispersed, and thus may be the result of cumulative watershed effects or may add to cumulative effects.

The Endangered Species Act protects species that are at risk of extinctions (listed under the Act as “threatened” or “endangered”) from Federal-agency actions that could reduce their numbers or their habitats. Where listed species dwell in aquatic or riparian habitats, the risk may be the result of multiple management activities occurring at a watershed scale—the definition of cumulative watershed effects. These examples are not exhaustive, and laws such as the Clean Air Act and the National Historic Preservation Act have requirements of their own that may call for a cumulative-effects analysis under some conditions. If the consideration of cumulative watershed effects for complying with multiple laws becomes an issue for fuel reduction treatments on a particular landscape, a more comprehensive watershed-scale analysis that meets the requirement of all these laws might be warranted. The synthesis presented in the report may help by providing a scientific basis for developing such a comprehensive watershed analysis.

# Evidence of Cumulative Watershed Effects

Because large-scale implementation of fuel reduction treatments has only begun recently, direct evidence of cumulative watershed effects is likely to be limited. However cumulative effects of other land management activities have been measured at watershed scales. A classic example comes from an extensive Columbia River basin study of fish habitat in 122 streams (McIntosh and others 2000): anadromous fish habitats that had been originally surveyed in the 1930s and 1940s were remeasured in the 1980s and 1990s. With the exception of streams in roadless watersheds, pools that were important stream habitat features for anadromous fish experienced significant declines over the intervening 60 years (for example, a 24-percent decrease in "large pools" and a 65-percent decrease in "deep pools"). An analysis of land management practices in these watersheds showed that no single practice or project was clearly responsible, but that a wide spectrum of land uses had occurred within the watersheds of the degraded streams, including forestry, grazing, urbanization, and road construction. It was the aggregate impact of all these practices—the cumulative effect of all the land uses—that had caused the habitat loss. By comparison, in watersheds with little or no change in land use over the period (watersheds with no roads), habitats had remained stable or had improved, reinforcing the hypothesis that cumulative effects of multiple land management activities had caused the degradation. The lesson from this study for large-scale fuel management is that widespread land management activities have the potential to cause significant, real impacts on aquatic systems in aggregate, even where the impacts of each individual, local project may be small or hard to measure.

# Considering Cumulative Watershed Effects in Fuel Management

It is critical that cumulative watershed effects be considered early as part of planning and implementing fuel reduction treatments in the current legal and policy environment. Although the Healthy Forests Restoration Act did not waive NEPA analysis, it reduced the number of alternatives that must be considered and added requirements for public collaboration. A brief discussion of NEPA requirements follows; however, the purpose of this synthesis is to assess what valid scientific information is available to analyze the cumulative watershed effects of fuels reduction treatments, not to explain the legal requirements for documenting these effects. Interdisciplinary teams can use the information presented in this report to produce environmental documents at the appropriate level.

The Council on Environmental Quality[1] gave guidance on when to include cumulative effects in NEPA analysis (Council on Environmental Quality 1997); and stated in a recent memo, "except in extraordinary circumstances, proposed actions that are categorically excluded from NEPA analysis do not involve cumulative effects analysis" [www.nrcs.usda.gov/technical/ECS/environment/pastact.pdf (Date accessed: June 27, 2011)]. For the Forest Service, extraordinary circumstances are defined as the degree of environmental impact to seven resource conditions listed in the Forest Service Handbook 1909.15, Chapter 30, Section 30.3: steep slopes or highly erosive soils; threatened, endangered, proposed, and sensitive species or their designated or proposed critical habitat; flood plains, wetlands, or municipal watersheds; Congressionally designated areas, such as wilderness, wilderness study areas, or National Recreation Areas;

[1] Connaughton, James L. 2005. Guidance on the consideration of past actions in cumulative effects analysis. 4 p. CEQ memo. Available at: http://ceq.hss.doe.gov/nepa/regs/.

inventoried roadless areas; Research Natural Areas; or Native American religious or cultural sites, archaeological sites, or historic properties or areas. Fuel reductions treatments that meet these requirements may not have to consider cumulative watershed effects.

Ultimately, it is the responsibility of the Forest Service leadership and employees to apply the broad set of laws and policies designed to create the intended effect of protecting the public from wildland fire, while at the same time complying with other sets of laws intended to protect natural resources and the environment. If members of the public disagree with specific management decisions, they have the right of challenge through appeal and in the courts. Within this changing legal and policy arena, methods for analyzing cumulative watershed effects are likely to remain important for natural resource managers. Cumulative effects are real, and sustaining multiple natural resources over the long run will require their consideration. Streamlining requirements for analysis under NEPA or other rules assumes that some practices have impacts that are either insignificant or small compared to the long-term benefits from a proposed action. Courts and public opinion will likely place the burden on land managers to demonstrate the validity of that assumption, using a standard for predicting outcomes that is usually based on current science.

## *Need for Decisions Based in Science*

The scope and scale of fuels reduction treatments being undertaken by Federal land managers provide a strong argument for developing scientifically based methods to estimate potential cumulative effects readily available to practitioners. When decisionmakers must tackle projects whose scope and scale are beyond what they have experienced, any relevant body of science may be of little practical use unless it has been interpreted and articulated in a manner directly useful for addressing the problems at hand.

## *Potential Uses of this Synthesis*

This synthesis of the current literature on cumulative watershed effects is a step towards developing useful methods for managers. It assembles in one place the current state of knowledge that was previously scattered across many sources. At the minimum it should provide managers, planners, and policymakers with a place to start describing the cumulative watershed effects of fuels treatments.

This synthesis, however, goes beyond being a central source of scientific information. Although cataloging and summarizing the literature are, in themselves, useful, this report goes further and anticipates questions that are likely to be posed by managers, planners, and policymakers. In this way, it identifies relevant questions the current science cannot answer as well as those that science can answer. The "science gaps" are at least as important as current knowledge because it is in these gaps that management and policy are vulnerable, and where caution is required because the results of actions may not be reliably predicted and may produce unforeseen outcomes. Identifying critical knowledge gaps also performs an important function for the science community. Such gaps can indicate productive areas for new research and development that have high potential both to advance scientific understanding and to serve the needs of the land management user community.

The value of this synthesis will depend strongly on how well it reinterprets existing knowledge in the face of new questions. Although it is true that a synthesis of the current literature may be a reworking of existing information, asking new questions of old data often casts them in new light. Thoughtful new questions may suggest new insights that have not previously been considered.

The questions considered here are indeed new because they involve the implications of a new management practice being applied on the landscape at expanded scales of space and time.

# Conclusions

We leave it to you, our readers, to decide how well this document meets your needs. As you use this document, you may find other pressing questions that we did not anticipate or address. We urge you to ask the science community to answer them. The value of this report will depend on its use; only through use will it serve its purpose of advancing the state of land management and policymaking and setting the stage for future research and development.

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CHAPTER 2.

# Silviculture of Forests in the Eastern United States

**Daniel C. Dey, John C. Brissette, Callie J. Schweitzer, James M. Guldin**

The forests of the Eastern United States are diverse and provide many products and amenities for people living in the area and beyond. Eastern temperate forests play an important role in determining water yield and quality. They have the potential to sequester large quantities of carbon and influence air quality, and thus climate. Our standard of living is very much linked to the health and productivity of forests. Forests cover approximately 41 percent of the Eastern United States, on average, but vary considerably at the State level, ranging from 6 percent in Iowa to 89 percent in Maine (Smith and others 2004).

This chapter provides a brief overview of the silviculture of eastern forests beginning with some fundamental definitions and concepts in silviculture that will be more fully applied in syntheses for northern conifers, northern and central hardwoods, southern hardwoods, and southern pines (table 1, fig. 1). These silvicultural overviews will allow us to address, to an extent, how silvicultural systems differ across a landscape that is highly variable in climate, soils, geology, biodiversity, and ecology.

The forest management plan considers the entire forest estate, which may range from hundreds to millions of acres. It identifies the broad goals and objectives of the landowner and guides management activities at finer spatial and temporal scales. In practice, forest operations occur at the stand scale (usually <100 acres); this is where silviculture is practiced. A recent exception is in the restoration of fire-dependent communities, where prescribed burning may be applied across landscapes of thousands of acres. Even on landscape-scale restoration projects, treatments such as thinning and midstory reduction are usually conducted in "stand-sized" areas to manage glades, fens, and other site specific communities. Also, smaller areas within the greater restoration area may need to be treated differently to create a diverse mosaic of stand composition and density as hardwood or conifer savannas, woodlands, and forests.

Good forest management requires that good silviculture be practiced.

## Silviculture

Silviculture is the science and art of cultivating forests by controlling their establishment, growth, composition, structure, health, and quality by applying planned and deliberate treatments to achieve specific objectives on a sustainable basis (Helms 1998). Silviculture is applied forest ecology: selection and implementation of treatments are

**Table 1.** Forest regions and ecological divisions with division numbers in parentheses

| Forest region | Warm Continental (210) | Hot Continental (220) | Subtropical (230) | Warm Continental Mountains (M210) | Hot Continental Mountains (M220) | Subtropical Mountains (M230) |
|---|---|---|---|---|---|---|
| Northern conifers | X | | | X | | |
| Northern hardwoods | X | | | X | | |
| Central hardwoods | | X | X | | X | X |
| Southern hardwoods | | | X | | | |
| Southern pines | | | X | | | |

Sources: Bailey (1995), Braun (1950), Hicks (1998), and Johnson and others (2002).

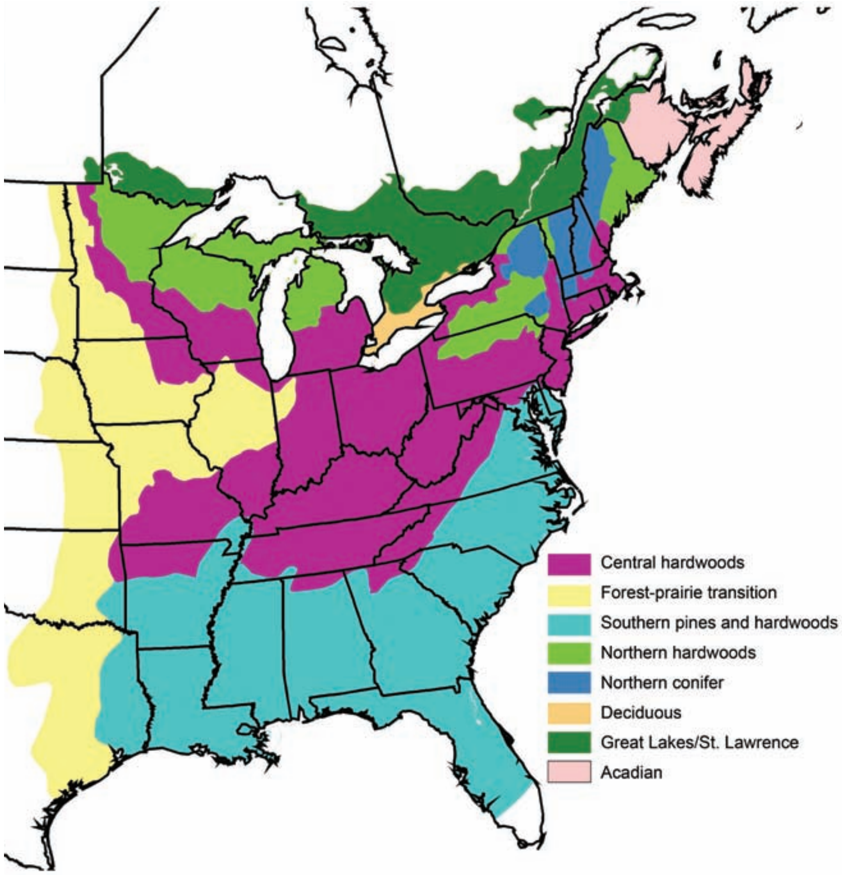


**Figure 1.** Major forest regions of the Eastern United States (Johnson and others 2002).

founded on the knowledge of ecosystem process and function, disturbance ecology, silvics, and stand dynamics. The practice of silviculture requires integration of many disciplines including ecology, genetics, entomology, pathology, soils, and other physical, biological, and social sciences.

## *Silvicultural Treatments*

Silvicultural treatments are applied to regenerate forests or promote stand development within existing forests. The clearcutting, shelterwood, and seed-tree regeneration methods create even-aged stands, in which trees are of a single age class and the range in age does not exceed 20 percent of the rotation (Helms 1998) or how long the stand is allowed to grow until it is regenerated again. Single-tree and group selection regeneration methods produce uneven-aged stands, in which there are at least three distinct age classes of trees intermingled (table 2).

**Table 2.** Silvicultural treatments for stand regeneration or intermediate tending

| Regeneration | Age structure | Method |
|---|---|---|
| | Even-aged | Clearcut |
| | | Seed tree |
| | | Shelterwood |
| | Uneven-aged | Single-tree selection |
| | | Group selection |
| Intermediate tending | Thinning | |
| | Release cutting | Weeding |
| | | Cleaning |
| | | Liberation |
| | Pruning | |
| | Sanitation cutting | |
| | Salvage harvesting | |

Tending treatments (intermediate cuttings) may be applied in conjunction with the regeneration harvest in an uneven-aged system, or at various times in an even-aged system. Tending alters stand character because it results in removal of some trees to achieve specific responses from remaining trees. The tending treatment is named according to the intended purpose or stage of stand development. For example, (1) thinning reduces stand density and increases growth (stem diameter or crown size) of residual trees; (2) release cuttings before the sapling stage free seedlings from competing vegetation (weeding), from overtopping undesirable competing trees of the same age (cleaning), or from overtopping older trees (liberation); (3) pruning removes branches to improve future tree grade and log quality; (4) sanitation cutting reduces the threat of insect and disease pests by improving tree health and vigor; and (5) salvage harvesting recovers dead or dying trees after pest outbreaks or wildfire.

## *Silvicultural System*

A silvicultural system is a comprehensive program of planned treatments, including regeneration and tending, that are applied to manage a forest stand through its life. Its name either describes the number of age classes (even- or uneven-aged) or the regeneration method, such as clearcutting, shelterwood, or selection harvesting (fig. 2). A silvicultural prescription outlines for each stand the timing and sequence of all treatments in the silvicultural system, including the specific regeneration method and the tending treatments needed to carry the stand from its existing condition to the desired future condition (the condition that meets the needs of the landowner).

Development of a silvicultural prescription for a stand begins with the assessment of the current stand and site conditions and consideration of any expected problems from insect and disease pests, nonnative invasive species, and damaging wildlife—such as white-tailed deer (*Odocoileus virginianus)* browsing. Then comes a thorough evaluation of how well alternative silvicultural systems could achieve management objectives in light of social, economic, and ecological constraints and opportunities. The prescription identifies the type and timing of activities needed to meet other objectives listed in the management plan—for example, reducing fire risk, retaining trees and coarse woody debris for wildlife habitat, sustaining native biodiversity, protecting culturally sensitive sites, mitigating soil erosion, or maintaining an ecological legacy from the previous stand—and describes which objectives will be achieved through implementation of each silvicultural treatment. The prescription also provides quantitative benchmarks at various key stages in stand development that indicate whether the outcomes of silvicultural treatments will be desirable and sustainable.

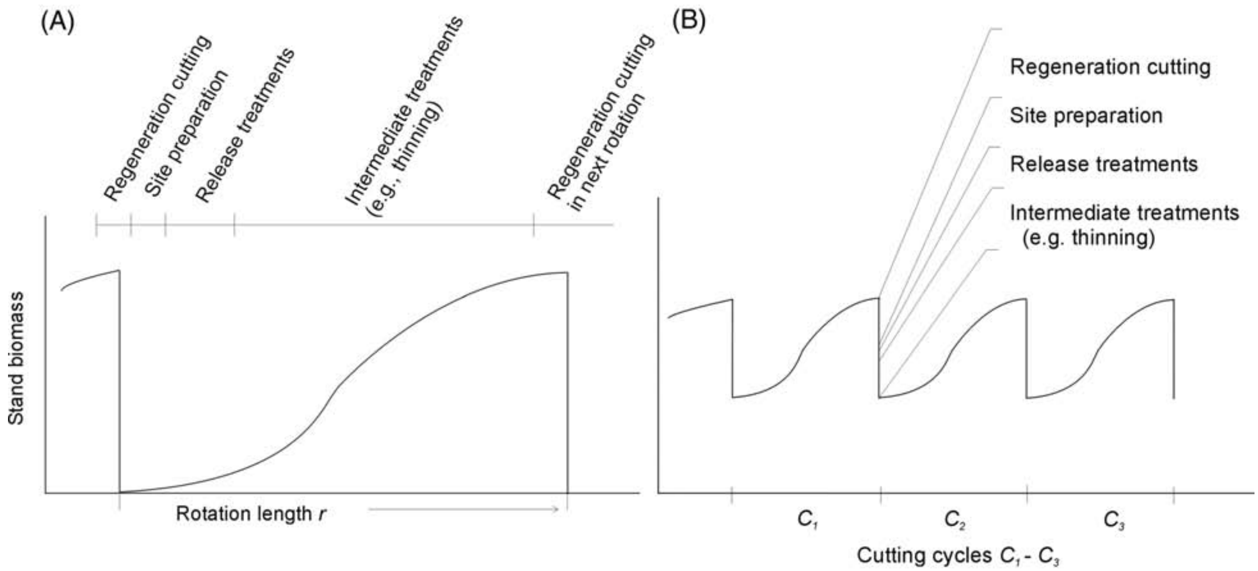


**Figure 2.** Schematics showing (A) chronosequential application of individual practices of a silvicultural system in an even-aged stand, with treatments applied across the entire stand to meet age-class silvicultural objectives during the rotation period, and (B) concurrent application of individual practices of an uneven-aged silvicultural system during a cutting cycle harvest in a balanced uneven-aged stand; with each cutting cycle harvest supporting similar treatments (Guldin 2006).

## *Regeneration Methods*

A brief review of the common regeneration methods used in the Eastern United States will provide a common understanding for the discussions of silvicultural systems for specific forests. It is important to distinguish between acceptable regeneration methods and practices—such as diameter-limit cutting, selective cutting, and high grading commonly practiced on nonindustrial private lands—that are not recommended because their main goal is to remove commercial timber with little thought for regeneration, future productivity and value, or other long-term benefits to landowners.

## *Even-Aged Regeneration Methods*

The following methods regenerate even-aged stands:

### Clearcutting

Clearcutting removes the entire stand in one operation. Some trees may be left in the clearcut to achieve goals other than regeneration, but their density is not enough to inhibit the development of reproduction. Natural reproduction is by seeding from adjacent stands and harvested trees, advance reproduction (seedlings or saplings in the understory before harvesting), stump sprouts (shoots arising from stumps of harvested trees), and root suckers (shoots arising from tree roots). Artificial regeneration methods, either by direct seeding or planting, can be applied before (but more commonly after) clearcutting.

### Seed-tree

Seed-tree is similar to clearcutting except that a small number of mature trees are left singly or in groups throughout the harvested area to supply seed for natural regeneration. The residual crown cover of seed trees is not large enough to create more modifications to the physical environment than would have been created by a clearcut.

### Shelterwood

Shelterwood removes the overstory in a series of harvests over a relatively short portion of the rotation. The goal is to retain enough seed producers to naturally regenerate the stand and enough residual overstory to shelter both newly established seedlings and existing advance reproduction from environmental extremes. Harvesting is usually from below (with trees in the smaller diameter classes and lower crown classes removed first), leaving the prescribed stocking of codominant and dominant trees of desirable species. The shelterwood is removed in a final harvest once sufficient numbers of competitive stems are established. The shelterwood system can be applied uniformly across the stand (uniform shelterwood) or in patterns such as groups (group shelterwood) or strips (strip shelterwood). In addition, a portion of the shelterwood (shelterwood with reserves) may be retained throughout the rotation for purposes other than regeneration, such as mast production, aesthetics, and structure for wildlife habitat. The shelterwood method may consist of three harvests:

- A preparatory cut removes the seed source of undesirable species and low-quality individuals and promotes the crown expansion of seed trees. This is not necessary if the existing stand has adequate seed production potential or if advance reproduction is present.
- A seed or establishment cut further reduces canopy closure in (or just before) a seed year, provides opportunities for site preparation before seed fall, and creates environmental conditions that favor germination and seedling establishment.
- A removal cut harvests the residual overstory to release well established reproduction.

## *Uneven-Aged Regeneration Methods*

The following methods regenerate uneven-aged stands:

### Single-tree selection

Single-tree selection is practiced by harvesting individual or small groups of trees indefinitely on a 5- to 25-year cutting cycle. Both regeneration and tending take place simultaneously in each harvest. Trees are considered for removal from all diameter classes in the stand with the goal of establishing reproduction and allowing recruitment of existing trees into larger size classes. Selection of an individual tree for removal is also influenced by its quality, vigor, and growing space requirements and by its potential contributions to wildlife habitat. Regeneration is largely from natural seedfall, existing advance reproduction, or stump sprouts and root suckers that develop after harvesting.

### Group selection

Group selection is applied on small patches, in which all trees are cut to create openings that are larger than single-tree gaps but smaller than clearcuts. The size of a group opening varies, depending on the regeneration requirements of the desired species, but is commonly twice the height (about 125 to 250 feet) of adjacent mature trees, or about 0.2 to 1.1 acres. The abundance and size of advance reproduction largely determines what reproduction will dominate forest openings: when advance reproduction is small, sparse, or absent, then regeneration is from seed. Group openings are often located where abundant advance reproduction occurs in patches within the stand.

### Combining prescriptions

Stand prescriptions for single-tree or group selection are guided by the desire to maintain a specified stand structure that sustainably yields a flow of products. In single-tree selection, the intensity and frequency of harvesting and the selection of trees for removal is determined by growth rate, target basal area, maximum tree diameter, and diameter distribution. In a stand or management unit, the area harvested by group selection is often regulated by area control and the length of the rotation. Practically, single-tree and group selections are applied together in a stand, with group openings being opportunistically used to increase forest diversity by favoring species that are intolerant-to-intermediately tolerant of shade.

# Northern Conifer Silviculture

Stands dominated by spruces (*Picea* spp.) and balsam fir (*Abies balsamea*) are common throughout the Laurentian Mixed Forest Province (212) of the Warm Continental Division (210) described by Bailey (1995). This province (fig. 1) stretches from northern Minnesota to northern Michigan; is found in parts of upper New York State and northern New England, especially eastern Maine; and straddles the border with Canada, where it is called the Great Lakes/St. Lawrence forest region in Ontario and Quebec, and the Acadian forest region in New Brunswick, Prince Edward Island, and Nova Scotia (Rowe 1972). Although often named after their two dominant species, the conifer stands in this province are commonly diverse with important components of pine (*Pinus* spp.), eastern hemlock (*Tsuga canadensis*), northern white cedar (*Thuja occidentalis*), and hardwoods, especially aspen (*Populus* spp.), birch (*Betula* spp.), and red maple (*Acer rubrum*). Thus, "northern conifers" is a more appropriate name. The mix of species varies across the province. White spruce (*Picea glauca*) is common throughout, but red spruce (*Picea rubens*)—the signature species of the Acadian Forest—is seldom seen west of the Adirondacks. Also, eastern hemlock is absent in most of Minnesota.

Soils are of glacial origin throughout the province. Vast areas of northern conifers are found on sites that are relatively flat with soils that are poorly to somewhat poorly drained. On very poorly drained sites, swamp species such as black spruce (*Picea mariana*) and tamarack (*Larix laricina*) dominate. As drainage improves, pines become increasingly prominent. Northern conifers are also found at high elevations in the mountainous sections (M211), but are less extensive in area than they are in the lowlands.

## *Natural Disturbances*

Natural stand-replacing disturbances are rare in northern conifer forests. Partial disturbances resulting from windthrow and isolated pockets of insects and diseases are common. With cyclic outbreaks that cause mortality and growth suppression, spruce budworm (*Choristoneura fumiferana*) has a significant impact on forest structure and composition (MacLean 1984), especially in balsam fir and (to a lesser extent) in spruce dominated forests. The extent of spruce budworm mortality is determined by the proportion of balsam fir and poor vigor trees (Baskerville 1975a, McClintock and Westveld 1946), soil drainage (Osawa 1989), and tree age (MacLean 1980, 1984). The relationship between stand structure and budworm susceptibility is less certain, and both even-aged (Baskerville 1975b) and uneven-aged (Crawford and Jennings 1989) structures have been recommended. However, when a spruce budworm outbreak is at full strength, structure may not be a factor because many of the ecological and stand relationships that normally prevail are simply overwhelmed.

## *Ecology and Silvicultural Systems*

Spruce, balsam fir, eastern hemlock, and northern white cedar are all shade-tolerant species; even eastern white pine (*Pinus strobus*) is intermediate in tolerance. They produce seed crops with regularity; and on most sites, they seldom experience water deficits of extended duration. Thus, advance natural regeneration is prolific under a broad range of overstory densities (Brissette 1996). Prevailing site conditions and silvics of the major species provide a range of silvicultural options for naturally regenerated stands, including both even-aged and uneven-aged systems. The major requirement is advance reproduction before harvesting the overstory; without it, regenerated stands are converted to hardwoods (Hart 1963).

Clearcutting is not effective for natural regeneration of northern conifers because they cannot compete with fast growing intolerant species in an open stand. Additionally, seeds of northern conifers remain viable for up to a year in the forest floor; consequently, they are not a reliable source of natural regeneration following harvesting (Frank and Safford 1970). The seed-tree method is also not effective for northern conifers because of competition with shade-intolerant species, and—perhaps more importantly—because the shallow-rooted residuals lack windfirmness (Frank and Bjorkbom 1973, Seymour 1995). The seed-tree method has been used with some success for eastern white pine (Wendel and Smith 1990), but does not provide the overhead shade that affords protection from white pine weevil (*Pissodes strobi*).

The most effective even-aged regeneration method for northern conifers is the shelterwood (Brissette and Swift 2006, Seymour 1995). Unmanaged stands being regenerated are often in the reinitiation stage of development (Oliver and Larson 1996) and will be well stocked with seedlings and saplings of desirable advance reproduction. For them, the first cut of the shelterwood may remove a third to a half of the overstory basal area (Frank and Bjorkbom 1973), followed by one or more additional removal cuts over the next decade or longer to release the new cohort. In situations where advance reproduction is insufficient for regeneration, overstory removal should be preceded by a light preparatory cut followed by a seed or establishment cut.

Perhaps the most innovative shelterwood variation in northern conifers is the Acadian *Femelschlag*, named by Seymour (2005) to describe an irregular group shelterwood with reserves. Emulating natural gap dynamic disturbances, its purpose is restoring complexity to stands that have been structurally and compositionally simplified by a century of repeated heavy partial harvests.

Thinning has not been a common practice in northern conifers (Seymour 1999). The combination of vast acreages of mature forests and a thriving pulpwood market made thinning an expense that many considered unwarranted. Predicted timber shortages, a strengthening market for small diameter sawlogs, and improvements in harvesting technology have all led to an increase in thinning over the past quarter century. And following the spruce budworm outbreak in the 1970s and 1980s, precommercial thinning became a routine means of accelerating merchantability in dense young stands.

Precommercial thinning is also an effective way to favor spruces over the typically more abundant and budworm-susceptible balsam fir (Brissette and others 1999). In the mid-1990s, new cut-to-length harvesting technology and a reduction in minimum top diameter specifications combined to make commercial thinning viable on an operational scale (McNulty 1999). The type of commercial thinning employed depends on whether a precommercial thinning has taken place. If so, crown thinning should be favored, although selection thinning may still be appropriate in some situations or in some parts of the stand. In stands that were not precommercially thinned, free thinning is recommended because it simultaneously controls spacing, captures mortality, and favors the best dominant and codominant trees.

Excessive thinning can reduce density to a point where the stands are vulnerable to windthrow. Also, tree height, crown length, diameter, and depth of rooting should all be considered when making thinning prescriptions.

Although multiaged northern conifer stands are not uncommon, the application of selection systems is not often rigorous. Some of the problems with the selection system in northern conifers have been reported by Kenefic and Seymour (2001), who showed that trees in the upper canopy generally produce more stemwood per unit leaf area than those lower in the canopy. Furthermore, trees released from suppression do not grow as well as those that have been free-to-grow; this is because older trees in uneven-aged stands grow less stemwood for the same amount of foliage compared to younger trees (Seymour and Kenefic 2002). These findings illustrate the perplexing question about applying the selection system in stands of species with quite different silvics: What is the proper structure to ensure sustainability over the long term?

In northern conifers, too much overstory suppresses the development of trees in the understory; it can also impede regeneration, although Brissette (1996) found that this is less of a concern than suppressed tree development. The amount of overstory that can be carried without suppressing smaller trees to the point of structural instability has yet to be determined for northern conifers, although species competitive advantage clearly depends on the amount and quality of overstory light (Moores 2003). In a long-term study on the Penobscot Experimental Forest in eastern central Maine (Sendak and others 2003), analyses of sapling ingrowth revealed slow growth, generating concern about the long-term sustainability of selection cutting in northern conifer stands (Kenefic and Brissette 2005).

Although it is critical to avoid carrying too many trees in the sawtimber classes, it is also important to maintain balances in other portions of the structure to (1) provide sufficient trees in each size class to replace those from larger classes as they grow or are cut, and (2) influence growth of smaller trees (Arbogast 1957, Solomon and Frank 1983). Timely regeneration of desired species is necessary, not only to sustain uneven-aged stands but also to tend immature trees and thereby accumulate high-quality growing stock (Hart 1963).

The abundance of site and species characteristics allows managers to choose from an array of silvicultural options for managing most northern conifer stands. What silvicultural system to adopt depends on stand attributes and management objectives. The key to success is having adequate advance reproduction before harvesting all of the overstory. Advance reproduction of northern conifers may already be sufficient in previously unmanaged stands that are in the reinitiation stage of development. If advance reproduction is not adequate, it can be achieved through application of shelterwood silviculture. Stand development and composition can be managed with precommercial thinning; and commercial thinning can help achieve a range of objectives. Selection silviculture can also regenerate and tend northern conifer stands. However, ensuring long-term sustainability requires careful monitoring of stand dynamics and periodic harvesting across all merchantable diameter classes to promote regeneration and sustain ingrowth of trees into larger size classes.

# Northern and Central Hardwood Silviculture

## Northern hardwood forests

According to Johnson and others (2009), northern hardwood forests extend from northern Minnesota eastward through the Northeastern United States (fig. 1). Northern hardwoods in Canada (Anderson and others 1990) occur in the Deciduous, Great Lakes/St. Lawrence, and Acadian forest regions (Rowe 1972). More than 18 forest cover types identified by Society of American Foresters (Burns and Honkala 1990) are present, representing various combinations of sugar maple (*Acer saccharum*), red maple, American beech (*Fagus grandifolia*), yellow birch (*B. alleghaniensis*), white birch (*B. papyrifera*), American basswood (*Tilia americana*), northern red oak (*Quercus rubra*), eastern hemlock, black cherry (*Prunus serotina*), aspen, and others on approximately 92 million acres of forest land in Bailey's (1995) Mixed Deciduous–Coniferous Forest Province (211) and Mixed Forest–Coniferous Forest–Tundra Province (M211b) of the Warm Continental Division (210 and M210). Northern hardwood forests are found in Fenneman's (1938) Laurentian Upland Division, Appalachian Highlands Division [Appalachian Plateaus Province (Mohawk, Catskill, and Southern New York Sections) and New England Province], and Interior Plains Division (Central Lowland Provinces).

## Central hardwood forests

Johnson and others (2009) report that the central hardwoods extend from the Ouachita and Ozark Mountains of Arkansas and Missouri; east to the Appalachian Mountains in northern Georgia and western North Carolina; northeast to southern New York, Connecticut, and Massachusetts; and west to central Minnesota (fig. 1). A significant inclusion in the west-central portion is the Prairie Peninsula of Iowa, northern Missouri, Illinois, Indiana, and central Ohio (Transeau 1935). The central hardwood range covers about 220 million acres of which 50 percent is forested today by a diversity of deciduous broadleaf species and several associated conifers [shortleaf pine (*Pinus echinata*) and eastern redcedar (*Juniperus virginiana*)]. The oaks represent the largest proportion of growing stock but many species of hickory (*Carya* spp.), sassafras (*Sassafras albidum*), flowering dogwood (*Cornus florida*), blackgum (*Nyssa sylvatica*), red maple, black cherry, yellow-poplar (*Liriodendron tulipifera*), elms (*Ulmus* spp.) and other upland hardwood species grow with the oaks. The central hardwood range includes Bailey's (1995) Eastern Broadleaved Forests (Oceanic) Province (221a) and Eastern

Broadleaved Forest (Continental) Province (221b) of the Hot Continental Division (220); and the Central Appalachian Broadleaf Forest–Coniferous Forest–Meadow Province (M221) and the Ozark Broadleaf Forest–Meadow Province (M222) of the Hot Continental Mountains Division (M220). Central hardwoods occur in Fenneman's (1938) Interior Highlands Division (Ozark Plateaus Province), Appalachian Highlands Division (Blue Ridge and Valley Province, Ridge Province, and southern sections of the Appalachian Plateaus and New England Provinces), and Interior Plains Division.

## Natural Disturbances

In western portions of northern and central hardwood forests, historic wildfire frequency and intensity were sufficient to retard tree regeneration and growth, thus creating woodlands and savannas; and in effect, extending eastward the ecotone between the Great Plains and the eastern deciduous forests. Wherever Native Americans lived, their use of fire resulted in local forest openings, barrens, savannas, and woodlands (Guyette and others 2002, Pyne 1982). However, decades of fire suppression in modern time have allowed trees to invade savannas and woodlands rapidly, transforming them into forests. Fires are still numerous, but most are kept small (<10 acres) by fire suppression. It is primarily in severe drought years that wildfires affect significant forest acreage.

Natural disturbances such as wildfire, hurricanes, tornadoes, and insect and disease outbreaks can initiate stand regeneration. The scale may be large enough to produce even-aged forests, but such severity of disturbance occurs infrequently and affects relatively small areas. More common is the mortality of individual mature trees or small groups that occurs annually in a forest stand. However, the probability of catastrophic mortality from invasive species or extreme weather events is increasing as the age structures and species diversity of forests become more homogeneous on the landscape.

Individual tree species have been seriously compromised by introduced pathogens. The chestnut blight (*Cryphonectria parasitica*) and Dutch elm disease (*Ophiostoma novo-ulmi*) have effectively eliminated the once prominent American chestnut (*Castanea dentata*) and American elm (*U. americana*) from eastern forests. Today, the gypsy moth (*Lymantria dispar*) is causing large-scale mortality and growth reductions in hardwoods throughout much of the Northeastern and Lake States. Emerald ash borer (*Agrilus planipennis*) is threatening to eliminate ash species (*Fraxinus* spp.) from all eastern forests. Large-scale homogeneity in forest composition and structure across the landscape could result in devastating ecological and economic losses following oak decline or outbreak of southern pine beetle (*Dendroctonus frontalis*).

## Ecology and Silvicultural Systems

Natural reproduction, the primary source of regeneration in northern and central hardwood forests, is from seeds produced in the current season or stored in the forest floor, advance reproduction, stump sprouts, and root suckers. Reproduction that dominates after harvesting is strongly influenced by preharvesting composition and structure of the trees in the overstory, midstory, and understory (Johnson and others 2009). The only silvicultural method not commonly used for regenerating northern and central hardwoods is the seed-tree method. The factors that affect regeneration success under alternate regeneration methods include the mix of desired species and their silvical requirements, physical environment, initial stand structure and composition, and vulnerability to deer browsing, invasive species, competing vegetation, insects, and diseases. Often critical to the success of any regeneration method is the planning and implementation of site preparation treatments—such as prescribed burning, mechanical scarification, and herbicide application—and preharvest or postharvest mechanical, chemical, or fire vegetation management treatments. Collectively, these treatments ensure establishment and dominance of the desired reproduction over its major competitors.

Even-aged systems are appropriate for most of the species found in northern and central hardwood forests. Clearcutting is effective for shade-intolerant species that move into highly disturbed environments and grow quickly in open environments (fig. 3). Aspen grows fastest in openings created by clearcutting, catastrophic wildfire, or blowdown; because its seed viability is short lived (weeks) and any seedlings that do germinate are highly susceptible to moisture stress, regeneration is primarily from root suckering, (Laidly 1990, Perala 1990). Clearcutting also favors prolific and frequent seeders that are capable of rapid growth: white birch, sweet birch (*B. lenta*), yellow-poplar, and black cherry (Burns and Honkala 1990).

Regeneration failures can result from insufficient seed supply at the time of harvesting or unsuitable seedbed conditions (deep litter and humus layers). Seed sources include mature trees in the harvested stand or adjacent forests, and dormant seed in the forest litter. Seed dispersal into openings has its limits, and the centers of very large clearcuts may experience understocking. Seed stored in the forest floor can provide a buffer to poor seed production or lack of dispersal into harvested areas. Although the seeds of most tree species either germinate or are destroyed within a year of dispersal, viability is 4 to 7 years for yellow-poplar and 3 to 5 years for white ash (*Fraxinus americana*) and black cherry (Marquis 1975).


**Figure 3.** Clearcutting is an effective way of regenerating species that do not survive or grow well in shaded conditions, provided the regeneration potential of the desired species is adequate before harvesting. Tree regeneration and stand development occurs relatively rapidly in clearcuts, and stands can reach the complex stage in less than 100 years in northern and central hardwood forests, as shown here in a set of photos taken from several stands (1-year, 2-year, 10-year, 45-year, and 85-year) in the Ozark Highlands of Missouri. (Photos by Daniel C. Dey)

Clearcutting success for oaks and many other hardwood species depends on advance reproduction in sufficient numbers and size. In these stands, it is the composition of large advance reproduction that determines what species will prevail or will share dominance with fast growing shade-intolerant species.

Natural populations of oak advance reproduction are more likely to be sufficient for successful regeneration by clearcutting on lower quality, xeric sites (Johnson and others 2009). On high-quality mesic sites, advance reproduction of oaks and other intermediately shade-tolerant species is often absent or has low regeneration potential due to its small size. Clearcutting on these sites accelerates the loss of these species and the succession either to shade-tolerant species (such as sugar maple, red maple, or American beech) or to pioneer reproduction of yellow-poplar, aspen, or other shade-intolerant species.

Light-seeded species such as birches, pines, and hemlocks germinate best on mineral soil. Therefore, the regeneration prescription for these species may include mechanical scarification or prescribed burning to provide a suitable seedbed (fig. 4). Black cherry, white ash, and yellow-poplar can establish seedlings in humus or light-to-moderate amounts of leaf litter. The nut producing species—such as black walnut (*Juglans*



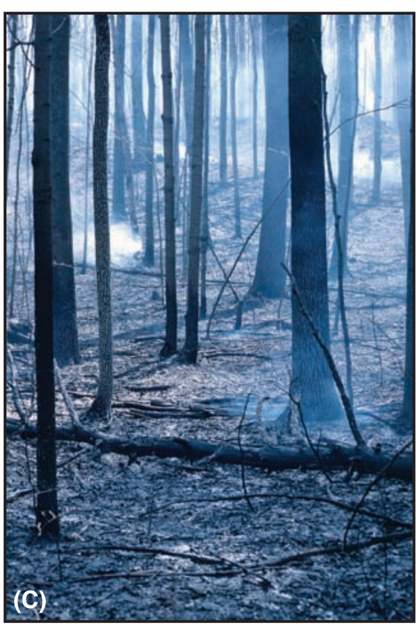



**Figure 4.** Site preparation to expose mineral soil for light seeded species, such as yellow birch, white pine, red pine, and shortleaf pine: (A) mechanical scarification of the ground by anchor chains dragged behind rubber-tired skidder, (B) mechanical scarification by dozers with brush blades, or (C) prescribed fires to consume fine litter and expose duff or soil, such as this low intensity spring burn in an Ontario northern hardwood stand (note that although low intensity fires can reduce competition from many herbaceous species and woody species up to about 5 inches in diameter, the benefits of competition control are ephemeral and treatments may need to be applied again); in contrast, (D) logging in the winter over frozen and snow covered ground, which can minimize the disturbance to soil, existing seedlings, and other desirable ground cover. (Photos by Daniel C. Dey)

*nigra*), pecan (*Carya illinoinensis*), hickory, and oak—can germinate in relatively deep litter. In fact, a covering of litter or mineral soil helps acorns and other recalcitrant seeds maintain adequate moisture content, thereby improving viability (Korstian 1927).

Control of competing vegetation may be needed to favor certain species. Beginning several years before the clearcut and extending for up to 5 years after harvesting, one or more applications of herbicides, mechanical cutting, mowing or disking, or prescribed burning may be required to allow growth of the desired reproduction (fig. 4). Small reproduction of intermediately tolerant-to-intolerant species that have relatively slow juvenile growth rates—such as the oak, hickory, and pecan—can be overtopped and suppressed by dense herbaceous vegetation or fast growing, shade-intolerant trees and shrubs.

Soil and air temperatures in clearcuts can be so high that new germinants perish, regardless of species (Dey and MacDonald 2001). Mortality of new seedlings increases further when surface soils and litter dry rapidly in spring and early summer, before tree roots grow deeper into the soil. Although growth of surviving seedlings of shade-intolerant species is greatest in the full sunlight of clearcuts, partial overstory shade moderates moisture and temperature extremes, improving the establishment of seedlings for most species.

The shelterwood method is a useful and flexible system, capable of regenerating a wide variety of hardwood species including black cherry, white ash, and oak (fig. 5). It is highly effective for species that rely on an abundance of large advance reproduction, which is typically absent or underdeveloped in species of low-to-moderate shade tolerance in mature forests. The density and arrangement (uniform, group, strip, or




**Figure 5.** Even-aged systems for regenerating a variety of hardwood and conifer species, including (A) the seed-tree method, in which often fewer than 15 trees per acre are left in the stand to provide a uniform dispersal of seed after the harvest—most often used to promote light seeded species that can regularly produce good seed crops such as eastern white pine in Ontario, and (B) shelterwood with only 30 to 50 percent of the original basal area, or (C) shelterwood more heavily stocked. (Photos by Daniel C. Dey)

irregular) of the shelterwood is calibrated to provide a favorable atmospheric and soil environment (light, moisture, and temperature) for regeneration of the desired species. Depending on the density and spatial arrangement of the shelterwood, sunlight levels in the understory may range from 20 to 60 percent (Marquis 1973), which is sufficient for species of intermediate shade tolerance to survive and grow. Under fully stocked, closed-canopied forests, light levels (1 to 3 percent of full sunlight) are only sufficient for the more shade-tolerant species to persist or grow (Dey and others 2008b).

As with clearcutting, treatments to prepare a suitable seedbed or to control competing vegetation may be needed for the shelterwood method. Many options are available for combining these activities before and after the preparatory, seed, or removal cuts. Because herbicides used to control competing woody vegetation are also lethal to the desired hardwood reproduction, the prescription should include the timing and method of application needed to avoid exposure to nontarget species.

Mechanical scarification exposes mineral soil and retards competing vegetation by uprooting and breaking stems of the woody understory. However, without followup herbicide treatment, woody shrubs and hardwood stems have a high probability of resprouting, limiting the time of effective release for desired reproduction.

Prescribed burning is effective for controlling the density of competing woody vegetation to favor the development of oaks and other fire-adapted species (fig. 6). Fires are


**Figure 6.** Prescribed burning (center), often in conjunction with tree harvesting, is done to restore and manage (A) savanna and (B) woodland ecosystems, and to (C) promote oak and pine regeneration in forest management by judicial and targeted use of fire to reduce (D) shade tolerant understory woody species and control other competing vegetation both before and following regeneration harvesting. (Photos by Daniel C. Dey)

often limited to the dormant season to reduce the risk of killing the shelterwood, especially after harvesting when fuel loading may be high. Dormant season fires can kill or cause shoot dieback in hardwood stems up to about 5 inches d.b.h. or diameter at breast height (Waldrop and others 1992). Many hardwood species can resprout after one fire, but oaks are preferentially favored by additional burning.

Finally, the density of the shelterwood can be calibrated to control, to some extent, the growth of competing shade-intolerant species such as yellow-poplar, sassafras, and aspen in the understory, while at the same time providing enough light for reproduction of oaks and other intermediately tolerant species (fig. 5). This is an effective means of favoring oak advance reproduction development on higher quality sites (Loftis 1990, Schlesinger and others 1993).

Of the uneven-aged systems (fig. 7), the single-tree selection method is effective for regenerating and sustaining forests dominated by shade-tolerant species such as sugar maple, red maple, American beech, and eastern hemlock (Hicks 1998). After a single-tree selection harvest, light levels in the understory are similar to those found in unmanaged, mature, closed-canopied forests—too low for any but the most shade-tolerant reproduction to persist. This method is not well suited for regenerating black cherry, white ash, birch, oak, and other light-demanding species in northern and central hardwood forests. However, evidence suggests that it can sustainably regenerate oak forests in the Missouri Ozark Highlands (Iffrig and others 2008, Loewenstein 2008, Loewenstein and others 2000), where a suite of competing shade-tolerant species such as American beech and maples is lacking, leaving white oak (*Q. alba*) as one of the more tolerant species. Harvesting by the single-tree method in these ecosystems is shifting forest composition toward a dominance of white oak from the current mixture of black oak (*Q. velutina*), scarlet oak (*Q. coccinea*), white oak, and shortleaf pine (Kabrick and others 2008).

The group selection method can regenerate shade-tolerant species, but is more often used for intermediate and intolerant species (fig. 7). The abundance and size of advance reproduction before harvesting largely determines what species will benefit. In most forest systems, the advance reproduction is dominated by sugar maple, red maple, and American beech, unless measures have been taken to reduce the stocking of these species in the understory and midstory before or during harvesting. If the diameter of a group opening is smaller than one-to-two times the height of the adjacent dominant trees, it will favor shade-tolerant species and will be more vulnerable to closure from lateral extension of adjacent overstory crowns before reproduction can recruit into the overstory. However, species such as maple and American beech can tolerate periods of suppression and eventually grow into the overstory, provided they are released by several periodic (every 10 to 20 years) stand harvests.

Larger openings are needed for black cherry, white ash, yellow birch, northern red oak, eastern white pine, and yellow-poplar. If the diameter of the opening equals the height of one tree (75 to 100 feet), light levels in the center can range from 20 percent on moderate north slopes to 30 percent on moderate south slopes (Fischer 1981). Increasing the diameter to two tree heights provides almost 50 percent of full sunlight on moderate north slopes and >60 percent on similarly steep south slopes, adequate for robust growth of most species of intermediate shade tolerance. Larger openings are even more beneficial to shade-intolerant species that require nearly full sunlight to achieve maximum growth.

Northern red oak and white ash should be present as large advance reproduction for successful regeneration, but black cherry, yellow birch, and yellow-poplar can regenerate from seed (Marquis 1990, Weigel and Parker 1997). Yellow birch and other light-seeded species germinate best on mineral soil, which can be exposed by mechanical scarification during harvesting.

Herbicides can be sprayed on the fresh stumps or injected into stems to prevent sprouting of unwanted shade-tolerant species, thereby reducing competition in group openings. Mechanical cutting alone will provide short-term release for intermediately tolerant–to-intolerant species, but sprouting of maple, American beech, hophornbeam

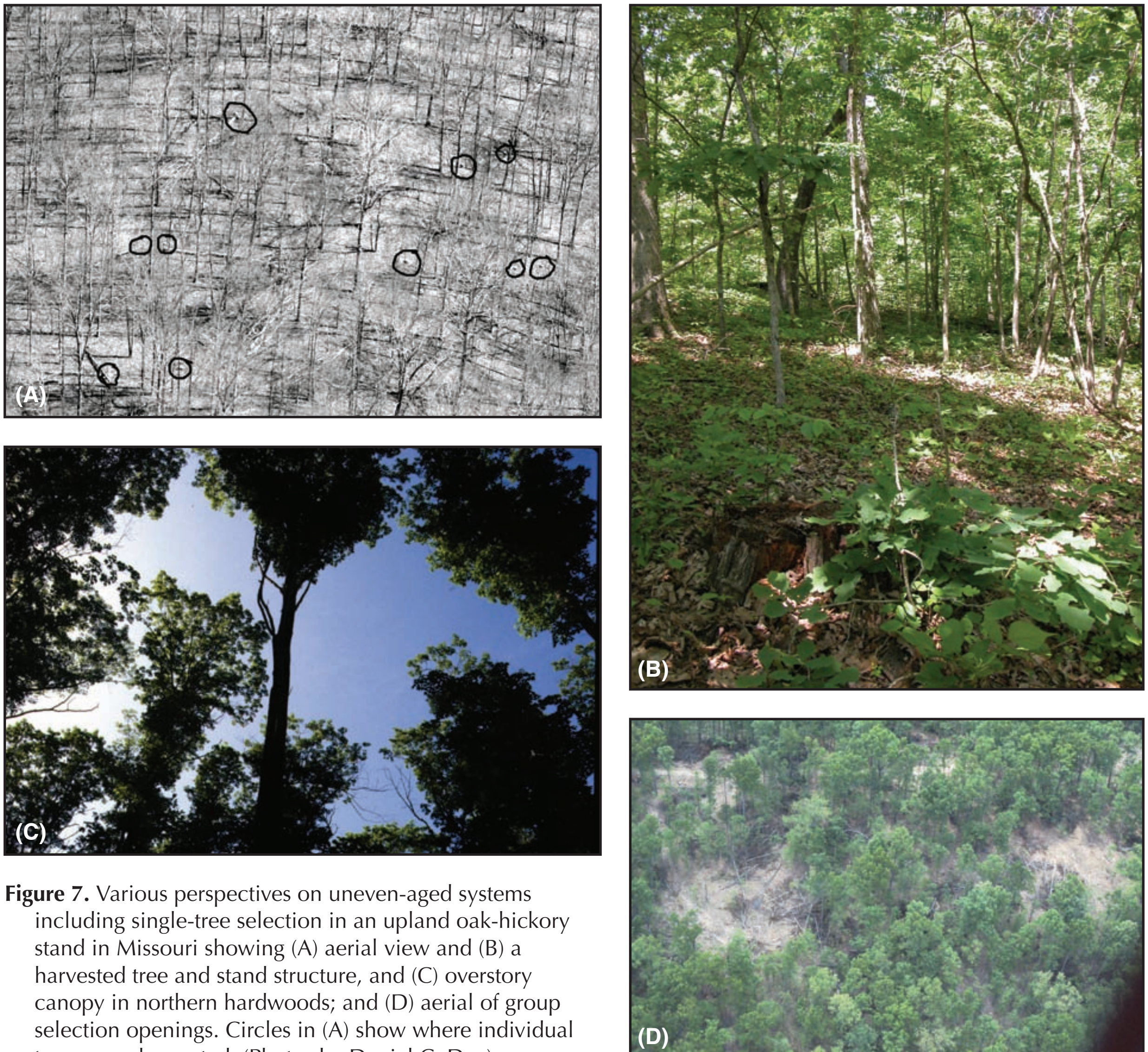


**Figure 7.** Various perspectives on uneven-aged systems including single-tree selection in an upland oak-hickory stand in Missouri showing (A) aerial view and (B) a harvested tree and stand structure, and (C) overstory canopy in northern hardwoods; and (D) aerial of group selection openings. Circles in (A) show where individual trees were harvested. (Photos by Daniel C. Dey)

(*Ostrya virginiana*), striped maple (*Acer pensylvanicum*), and flowering dogwood will be rapid, with new stems quickly suppressing desirable reproduction.

Artificial regeneration of hardwoods limited to situations where it is needed to supplement natural advance reproduction of desired species (in particular oaks) in upland forests. As summarized by Johnson and others (2009), a number of researchers have evaluated methods for underplanting oaks in shelterwoods in the Eastern United States, including Dey and Parker (1997a, 1997b), Dey and others (2009, 2008a), Johnson (1984), Johnson and others (1986), Loftis (1990), Parker and Dey (2008), Spetich and others (2002), and Weigel and Johnson (1998a, 1998b, 2000). Artificial regeneration of hardwoods is more common in afforestation of agricultural bottomlands and upland pastures; and, in combination with direct seeding, for oaks in floodplains (Dey and others 2008a).

Once regeneration is established, sustaining the stocking of desired species to maturity may require one or more tending treatments. Crop tree release is effective for improving the survival, growth, and quality of individual trees and maintaining the stocking of desired species to maturity (Dey and others 2008b, Miller and others 2008). Maintaining shade-intolerant and intermediately tolerant species in young stands requires early crop tree release, beginning about the time of crown closure.

# Southern Hardwood Silviculture

Of the 535 million acres of land in the Southern United States, 214.6 million acres are classified as forest land, the majority (95 percent) of which is commercial timberland (Smith and others 2004). This area of forest is about 60 percent of what existed at the onset of European settlement in 1630, and about 90 percent of forest acreage at the height of selective cutting in 1907 (Conner and Hartsell 2002). Over the past six decades, however, the area of commercial timberland in the South has remained more or less constant, with areas going out of timberland primarily to agriculture and urbanization balanced by a reversion of abandoned agricultural land back into forests.

Southern pines and hardwoods occur conterminously across the South (fig. 1), and are bounded on the north by central hardwood forests and on the west by the Prairie Division (250), with the remaining boundaries being the Gulf of Mexico and the Atlantic Ocean (Johnson and others 2009). Within this area are four major ecoregions in Bailey's (1995) Subtropical Division (230 and M230)—the Piedmont [Southern Mixed Forest Province (231)], the Coastal Plain [Outer Coastal Plain Mixed Province (232)], the Interior Highlands [Ouachita Mixed Forest–Meadow Province (M231)], and the lower Mississippi Alluvial Valley [Lower Mississippi Riverine Forest Province (234)]—all covering approximately 270 million acres, 60 percent of which are forested.

The Cumberland Plateau [Hot Continent Division (220)], and associated highlands of the Interior Highlands and lower Mississippi Alluvial Valley [Subtropical Division (230)] contain the majority of upland and bottomland hardwood forests. Oak forests—including white oak, scarlet oak, southern red oak (*Q. falcata*), overcup oak (*Q. lyrata*), chestnut oak (*Q. prinus*), water oak (*Q. nigra*), Nuttall oak (*Q. nuttallii*), willow oak (*Q. phellos*), northern red oak, and black oak—cover 60 percent; loblolly pine (*Pinus taeda*), shortleaf pine, longleaf pine (*Pinus palustris*), and slash pine (*Pinus elliottii*) dominate the remaining forested lands.

In uplands are mixed oak-pine stands, often consisting of mixed upland hardwood species with loblolly, shortleaf, or Virginia pine (*Pinus virginiana*). Upland hardwoods are dominated by red and white oaks, with frequent occurrences of hickory, yellow-poplar, sugar maple, red maple, American beech, black cherry, sassafras, sourwood (*Oxydendrum arboreum*), blackgum, birch, and ash.

Southern bottomland forests occur on river floodplains and are most extensive in the Coastal Plain and Mississippi Alluvial Valley. Species compositions are complex and influenced by site conditions. Bottomlands are dominated by oaks (overcup oak, Nuttall oak, willow oak and water oak), American sycamore (*Platanus occidentalis*), sweetgum (*Liquidambar styraciflua*), blackgum, elm, sugarberry (*Celtis laevigata*), eastern cottonwood (*Populus deltoides*), willow (*Salix* spp.), ash, hickory, and red maple. Conifers that may grow in floodplains include loblolly pine, spruce pine (*Pinus glabra*), baldcypress (*Taxodium distichum*), pondcypress (*Taxodium distichum* var. *nutans*), Atlantic white-cedar (*Chamaecyparis thyoides*), and eastern redcedar.

## *Natural Disturbances*

Fires occur infrequently in alluvial areas because these sites are too wet and often lacking enough litter to support combustion. More common are natural and episodic disturbances, such as severe or unseasonable flooding, drought, windstorms, and animal activities; or human modifications such as impoundments or other flood control devices, timber harvesting, and land clearing for agriculture.

In southern upland hardwood forests, natural fires (those not set by humans) have not played a major role in landscape dynamics because the climate is dominated by long, hot growing seasons and abundant rain. Conflicting opinions exist over the role and extent that human use of fire had as an ecological force in upland forests. Native Americans certainly used fire to clear along watercourses and to drive game, and occasionally these fires escaped into higher elevations. Europeans burned to improve grazing, reduce undergrowth for better visibility and accessibility, and control insects (Komarek 1974, Van Lear and Waldrop 1989).

In addition to fire, the most important shapers of today's southern hardwood forests were agriculture and extraction of coal, timber, and gas resources. Disturbances caused by storms, insect and disease outbreaks, and late-season frosts continue to alter stand structure and composition. Likely future forest influences include increasing development, invasion by nonnative species, and more aggressive coal mining.

## *Ecology and Silvicultural Systems*

In addition to landowner goals, the key factors driving management decisions in southern hardwood forests are landscape location (physiographic, edaphic, and moisture and nutrient site class) and the influences of past disturbances on current forest composition and structure. From xeric upland oak-hickory forests to species rich mesic cove forests to mixtures of oak, gum, and eastern cottonwood in hydric bottomland forests, site factors—primarily moisture and fertility—dictate the range of appropriate silvicultural prescription options. Also important is managing competition to achieve desired forest composition and structure. Regenerating desirable species is more difficult on the most productive sites, where competition is great and light availability often limits regeneration of desired species. However, ensuring that adequate light is available to reproduction can be achieved through silvicultural treatments.

Southern hardwood forests are disturbance-dependent systems. They are also diverse in species, many of which are desirable, challenging efforts to control the final composition at maturity. Therefore, we must consider the silvic requirements of each species when selecting the type and timing of silvicultural treatments. In upland and bottomland systems, oak is a focal species group—more difficult to regenerate on high-quality sites, where potential competition from other woody and herbaceous species is greater. Silvicultural prescriptions for regenerating southern hardwoods are primarily even-aged based; scant information exists on the long-term effects of uneven-aged management.

## *Bottomland Hardwood Systems*

In bottomlands, higher elevation sites on fronts and ridges of major streams have better drainage and lower soil clay content than the lower elevation flat sites. Competition is greater on higher elevation sites than lower. Manuel's (1992) decision model for managing and regenerating southern bottomland hardwoods is based stocking levels of desired species, tree-preference class, and individual tree characteristics. Belli and others (1999), Broadfoot (1976), and Putnam and others (1960) outline techniques for evaluating natural regeneration of bottomland hardwoods. These techniques take into account regeneration source (seed, seedling, or sprout) and site type based on soil series and inundation regime. If the regeneration source is adequate, the decision model recommends using the clearcut method. If the decision is to manage but not harvest and if adequate regeneration is present, care must be taken to provide the forest floor with enough light to maintain that regeneration.

Clearcutting is the most widely proven method of regenerating bottomland hardwoods because it allows for full sunlight to reach the forest floor, promoting the growth of species that are shade-intolerant and moderately shade-intolerant (Clatterbuck and Meadows 1993). Light-seeded species such as willow, eastern cottonwood, and ash also thrive under clearcutting operations that exposure mineral soil. The seed tree method can also be used for light-seeded species in bottomland hardwood stands, but it has been shown to have no benefit for regenerating oaks or other desired heavy-seeded species (Johnson and Krinard 1976).

If the regeneration is inadequate or the stand lacks an adequate mix of desirable species, managers can enhance the growth of individual stems and promote establishment of regeneration by increasing light through density-reducing harvests. The shelterwood method can promote regeneration by harvesting to reduce stand density and remove undesired species, by herbicide injection into individual stems of undesired species, or by a combination of both treatments. Oaks and other heavy-seeded species benefit from a release harvest, either at the time of a bumper acorn crop or immediately after the establishment of seedlings from a bumper crop. Final overstory removal is postponed until the desired advance reproduction is large enough to be competitive after release.

Single-tree selection, practiced in the bottomlands in the 1950s and 1960s, favored the development of shade-tolerant species and resulted in forest composition shifts towards lower-valued timber species (Hodges 1997). Group selection may also result in species composition shifts towards more shade-tolerant species; groups must be large enough to meet the regeneration needs of shade-intolerant species. Patch cutting, which combines clearcutting and group selection to create larger openings, is becoming increasingly more common in bottomlands (Meadows and Stanturf 1997). Key to success is the development of large advance reproduction before final regeneration harvesting and the control of competing vegetation by preharvest or postharvest treatments.

## Upland Hardwood Systems

Although primarily focused on oaks, Johnson and others (2009), Loftis and McGee (1993), and Spetich (2004) provide an overview of silvicultural options and recommendations that can be applied to other desirable species in upland hardwood forests. Loftis' (1989) comprehensive model for evaluating natural regeneration of southern upland hardwoods is calibrated for the Southern Appalachian Mountains, but it can easily be adjusted for other southern upland systems. For example, in the Cumberland Plateau, which differs by having abundant sugar maple and a scarcity of black cherry, users can alter the model parameters to increase sugar maple ranking on the competition scale and reduce the influence of new black cherry seedlings.

Topographic position dictates upland hardwood management. In general, clearcutting regenerates oak stands on higher elevation sites (for example, the tabletops of the Cumberland Plateau and upper ridges of the Southern Appalachian Mountains), where lower site quality, less competition from other species, and relatively high numbers of oak advance reproduction contribute to a desirable species composition in the next stand. If the preservation of the oak component for more productive stands is desired, silvicultural techniques can encourage more and larger oak advance reproduction while at the same time reducing competition.

After a regeneration harvest, germinating acorns provide new seedlings; but because oak seedlings preferentially allocate carbon to root growth, their shoot growth is slow, making them vulnerable to suppression by species (fig. 8) that exhibit rapid shoot growth (Johnson and others 2009). Consequently, regenerating oak rarely reaches a dominant or codominant position on productive sites (Loftis 1983, Sander 1972), where yellow-poplar is its major competitor. In addition to regeneration from stump sprouts and advance reproduction, yellow-poplar can also regenerate successfully from often-numerous seedlings that grow rapidly after harvesting, either from current seed production or seed stored in the forest floor (Beck 1970). In other upland hardwood systems, desirable species such as ash and black cherry can also regenerate from new seedlings and grow rapidly if given sufficient light. The diversity of shade tolerances that these species exhibit contributes to the challenge of regenerating southern upland hardwood stands.

A promising alternative regeneration method to favor oaks and other intermediately shade-tolerant species, the shelterwood requires a sequence of cuttings over a 5- to 20-year interval and multiple entries into the stand. The residual basal area in a shelterwood must be large enough to prevent light-seeded, shade-intolerant species such as yellow-poplar from growing and becoming established. The change in canopy structure and below-canopy light conditions will also favor sugar maple and other shade-tolerant species. Treating the shade-tolerant subcanopy, in addition to reducing overstory density, will promote the development of advance reproduction of the desired species.

Over time, single-tree selection in upland hardwood systems consistently results in a composition shift towards shade-tolerant species. In mature forests that initially have

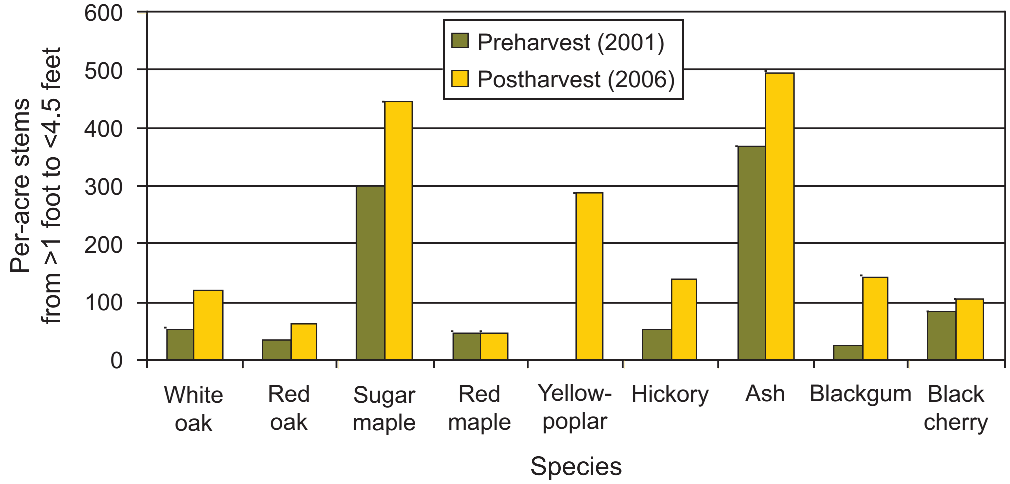


**Figure 8.** Species density in the Cumberland Plateau of Jackson County, AL, where large yellow-poplar, sugar maple, and ash reproduction increased substantially compared to slight increases in large oak reproduction, 5 years after a shelterwood harvest that removed 50 percent of the basal area.

a substantial overstory component of oaks, yellow-poplars, and other desirable but less shade-tolerant species, this method inhibits their regeneration and favors the recruitment of sugar maple, red maple, sourwood, flowering dogwood, American beech, and other shade-tolerant species (fig. 9). The desirable seedlings and sprouts that do become established will not persist in the low light of the forest canopy. This means that managers may need to consider interspecific competition when manipulating stand composition and structure in addition to ensuring sufficient light in the understory for development of large advance reproduction. Creating large openings by group selection harvesting and applying herbicides to eliminate the tolerant understory is a technique that may offer promise (Della-Bianca and Beck 1985).

Many southern upland hardwood stands originated when wildfire was more prevalent. Burning to regenerate oak most likely will require multiple fires over a decade or more. In an early study testing the use of prescribed fire on oak regeneration in the Southern Appalachian Mountains, Loftis (1990) found that one burn not only failed to increase oak-seedling growth and control the development of other competing regeneration, but it also reduced the survival rate of red oak seedlings. A regime that incorporates a high level of disturbance, such as shelterwood harvesting followed by prescribed burning, may favor oak regeneration over its competitors.

**Figure 9.** Shade tolerant species such as sugar maple are dominant in the understory of many mature hardwood forests in the Eastern United States. Single-tree and group selection favors the recruitment of these species into the overstory and the replacement of the oaks and other less shade tolerant species. Even-aged systems that fail to control the shade-tolerant competitors also accelerate successional replacement of the oak species. (Photo by Callie J. Schweitzer)

The timing and intensity of burning may also play a significant role in modifying regeneration dynamics. For example, a single spring burn several years after shelterwood harvesting produced a high-intensity fire that favored oak regeneration over yellow-poplar, compared to less-intense summer or winter burns that were insufficient for competition control (Brose and others 1999). Fire intensity affects stand density more than the height of oak and its competitors. Because a single, low-intensity fire may have little or no effect on stand composition, repeated burning may be necessary to favor oak over yellow-poplar. But regardless of fire timing or intensity, the competitive status of oak seedlings drives the response to disturbance.

# Southern Pine Silviculture

About 96 million acres of timberland in the Southern United States (fig. 1) are found in southern pine or oak-pine forests (Smith and others 2004). The southern pines consist of four major species: loblolly pine, shortleaf pine, slash pine, and longleaf pine.

Loblolly pine is found in 14 States, growing from southern New Jersey to eastern Texas. Its natural range is along the Atlantic Ocean and Gulf of Mexico, in the Piedmont Plateau, and in parts of the Cumberland Plateau and Appalachian Mountains (Baker and Langdon 1990). Loblolly is the preferred species for plantation forestry in the South, and millions of acres of native mixed pine, pine-hardwood, and hardwood-pine stands across the South have been converted to genetically improved and intensively managed loblolly pine plantations for use in timber and fiber production.

Shortleaf pine is the most widely distributed of the four southern pines. It is found in 22 States, typically in mixture with other pines (especially loblolly) or hardwoods; but in the Ouachita Mountains of Arkansas and Oklahoma, it is the only dominant naturally occurring pine (Guldin 2007, Lawson 1990).

Slash pine has the smallest native range of the four species, found from southern South Carolina through the hills of southern Georgia and virtually throughout Florida, and west along the Coastal Plain to southern Louisiana. Outside of its native range, it has been widely planted and direct-seeded in western Louisiana and eastern Texas on cutover longleaf pine sites (Lohrey and Kossuth 1990).

Longleaf pine is native along the Coastal Plain from Virginia to eastern Texas. Once occupying an estimated 92 million acres of the South, today it is much less widely distributed over roughly 3.2 million acres—the result of virgin-stand harvests, fire exclusion, and reforestation of cutover areas with loblolly and slash pine (Boyer 1990, Landers and others 1995). Efforts are underway to restore longleaf pine ecosystems, especially on Federal and State lands such as national forests and lower Coastal Plain military bases.

These four southern pines are occasionally found in association with minor pine species such as spruce pine along the Gulf of Mexico and pond pine (*Pinus serotina*) along the lower Atlantic coast; in the Appalachian Mountains are found the pines that have a more northerly distribution, such as Table Mountain pine (*Pinus pungens*), Virginia pine, pitch pine (*Pinus rigida*), and eastern white pine. Throughout the South, pines are found side-by-side with hardwoods, especially the oaks and hickories (Keys and others 1995) that would eventually dominate in the absence of disturbance.

Thirty percent of the forest land area in the South—some 66 million acres—is dominated by two southern pine forest types. About 52 million acres are in the loblolly-shortleaf forest type, and 14 million acres are in the longleaf-slash forest type; another 30 million acres are classified as the oak-pine forest type, in which pines and oaks are found in mixtures of varying percentages (Smith and others 2004):

- The loblolly-shortleaf forest type includes pure stands of loblolly pine throughout the South and pure stands of shortleaf pine in the Ouachita and Ozark Mountains—both of natural or planted origin—and mixed stands of loblolly and shortleaf pine that are typically of natural origin.

- The longleaf-slash pine forest type is generally found in pure stands of either slash or longleaf pine of natural or planted origin; only occasionally are both species present in naturally regenerated stands.
- Oak-pine stands are usually of natural origin; landowners often to manage them either for the hardwood component or—more commonly, especially on forest industry ownership—for the pine component so as to simplify species composition and increase pine growth and yield.

## Natural Disturbances

The southern pines are early successional species adapted to a range of disturbance events, most especially at larger scales. The climate of the South features a variety of large-scale disturbance events, any one of which can destroy an existing stand and create open conditions for establishing a new age cohort of pines. Wind events such as tornadoes and hurricanes can affect areas as small as a single stand or as large as an entire State. Under certain conditions, outbreaks of the native southern pine beetle can spread to cover thousands of acres if unchecked; even if controlled, they can affect hundreds of acres anywhere in the South at any time.

Fire, whether as a result of natural or human causes, is the single most important ecological determinant in southern pine stand dynamics and development. Presettlement accounts of southern pine forests commonly described mature pines with virtually no midstory and with an understory dominated by grasses, annuals, and perennials—where one could easily ride a horse and not be impeded by vegetation (Hedrick and others 2007). Native Americans used understory burning to promote hunting and community defense (Guyette and others 2006). Early settlers adopted the practice as well to attract game and provide forage for domesticated livestock. No doubt, both the Native American and European cultures appreciated the benefits that understory burning provided in controlling the ticks and chiggers that still torment those who live and work in southern forests.

## Ecology and Silvicultural Systems

Fire is especially important in southern pine regeneration dynamics. The four southern pine species have each developed interesting and unique adaptations to fire that improve the likelihood of seedling establishment and development. A shortleaf pine sapling is the only one of the four that will reliably resprout if its crown is topkilled by fire (or if mechanically severed), a fire adaptation trait that was described early on (Mattoon 1915). Small shortleaf seedlings are extremely vulnerable to even low intensity surface fires, but their ability to survive or resprout after topkill increases with increases in stem diameter (Dey and Hartman 2005), up to a maximum d.b.h. of 3 inches (Dey and Fan 2009). Thus, in sapling-sized shortleaf pine stands, a new age cohort develops through resprouting and some added seedfall if a seed source remains nearby. To prevent regeneration from accumulating and growing into the overstory, the frequency of burning must be 8 years or less (Stambaugh and others 2007).

In contrast to shortleaf, loblolly and slash pine saplings are quickly and effectively killed by fire, which may explain why these species are thought to be the more mesic of the southern pines. For example, slash pine is found naturally only in the wetter areas of the Atlantic Coastal Plain (Lohrey and Kossuth 1990), and loblolly pine has a reputation of thriving naturally on moist to wet sites (Baker and Langdon 1990). Both species are abundant and regular seed producers, producing adequate-or-better seed crops at least half the time. The loblolly-shortleaf pine type in the western Coastal Plain is arguably the most prolific pine type in North America, producing adequate-or-better seed crops 4 years in 5 and bumper crops with >1 million seeds per acre (Cain and Shelton 2001). Essentially, the adaptation strategy for Coastal Plain loblolly-shortleaf pine mixtures and slash pine is to produce enough seed on a sufficiently frequent basis to establish seedlings within any new forest opening shortly after it is created, and to grow to the sapling stage fast enough to survive the next surface fire.

One might speculate that the two strategies—resprouting and reseeding—work together in mixed loblolly-shortleaf pine stands of natural origin, and that this may explain why shortleaf is retained in the mixture. If a newly established loblolly-shortleaf pine cohort has the opportunity to grow fast enough to escape the next fire, the species mixture would favor loblolly pine, whose saplings grow faster than shortleaf pine. But a surface fire in a mixed sapling stand would kill the loblolly—requiring reseeding onsite or from a nearby seed source—whereas the shortleaf saplings would simply resprout: a dynamic that might confer an adaptive advantage to shortleaf in circumstances that would normally favor loblolly.

Longleaf pine has a different strategy entirely, featuring extended irregularity in seed crops and a distinctive seedling grass stage. While in the grass stage, the seedling builds root growth rather than shoot growth, and the terminal bud develops a pattern of bud scales and needle architecture that protects it from surface fire. Those early years in the grass stage require occasional surface fires to prevent suppression by grasses and other understory herbaceous and woody vegetation. Fires also control brown spot needle blight (*Mycosphaerella dearnessii*), which can prevent seedling emergence from the grass stage (Boyer 1979). After several years and under proper conditions, longleaf seedlings break through the grass stage and begin to grow rapidly.

All four species are generally considered intolerant of shade as mature trees, but shade tolerance is more pronounced at younger ages—especially in loblolly and shortleaf pine, both of which can tolerate somewhat more overstory shade than longleaf and much more than slash pine. All of the southern pines also have the interesting attribute of being able to respond to release from adjacent or overtopping competition at relatively advanced ages, which enables them to maintain site occupancy under partial disturbance events such as ice storms or wind events. The four species also show good ability to segregate into crown classes, which helps minimize extended periods of sapling stagnation even though poor growth can occur to a certain degree in densely stocked sapling stands.

Summaries of the silviculture of southern pines have been developed over the past four decades and are still appropriate references for landowners and the foresters who advise them. Burns (1983) describes most of the important forest cover types in North America, including the southern pines. More recently, Fox and others (2007) and Guldin (2004) have published overviews of the general principles of plantation silviculture and silviculture of naturally regenerated stands. State-of-the-art summaries of the selection method are also available, one for longleaf pine (Farrar 1996) and the other for loblolly and shortleaf pines (Baker and others 1996).

## Clearcutting and planting

Even-aged plantation silviculture is effective for all four of the southern pines, but has been most widely practiced with loblolly and slash pine. One can argue convincingly that the two most important silvicultural advancements in the 20th century were responsible for the widespread practice of plantation silviculture. First was the development of genetically improved planting stock, which was pioneered with loblolly pine and applied with varying intensities in all four species. Second was the development of chemical amendments such as fertilizers for site amelioration and herbicides for woody and herbaceous competition control. These technologies were optimally applied in association with clearcutting and intensive customized site preparation, followed by planting with careful attention to the origin and quality of planting stock. As a result, clearcutting, planting, and subsequent intermediate treatments became the standard prescription for intensive pine silviculture. The millions of acres of plantations that were created using the many variations of this practice have been the mainstay of the southern pulp and paper industry for the past four decades (fig. 10).

Because of the plasticity and success of pine plantation silviculture for rapid fiber production, southern pine forests have become the focus of the most intensive forest management activity in the South, if not the Nation. Recent data suggest that of

**Figure 10.** A loblolly pine plantation on a high-quality site in the western Ouachita Mountains; the stand is between 15 and 20 years in age, and recent treatments consisted of prescribed burning and thinning. (Photo by James M. Guldin)

the 66 million acres in the two pine-dominated forest types, 34 million acres are in stands of natural origin and 32 million acres are planted (Smith and others 2004). Most pine plantation area is in private ownership, with only 1 million acres (24 percent) on national forest lands and 750,000 acres (20 percent) on other public lands; compared to roughly 15 million acres (40 percent of the total pine-dominated area in this ownership) on nonindustrial private lands, and 15 million acres (75 percent of the pine-dominated forest area) on forest industry lands (Smith and others 2004). And the 32 million acres of forests dominated by planted pines represents 84 percent of the total plantations in the South.

Wear (2002) suggested that by 2050, a quarter of all southern forest land—50 million acres—will be planted. With 85 percent of current plantations coming from the two southern pine forest types, an additional 11 million acres, roughly, are likely to be converted to planted pines. It is unlikely that these additional planted acres will come from forest industry, which has only 5 million acres of natural stands remaining in its 20 million acres of pine-dominated forests. Instead, we are likely to see the increases come from natural pine stands on nonindustrial private land or from converting hardwood-dominated forests on forest-industry and nonindustrial private lands. And because planting with containerized planting stock is an important tool in the restoration of longleaf pine stands on the southern Coastal Plains, longleaf restoration goals may involve significant planting on public lands as well.

## Other even-aged methods

That portion of the southern pines not managed using plantations can be very effectively managed using even-aged and uneven-aged methods that rely on natural regeneration. Four areas of continuing or expanding application have been suggested (Guldin 2004):

- Plantation silviculture is costly, especially the initial capital investment into stand establishment; many landowners seek regeneration methods that have lower initial establishment costs and that retain some degree of canopy cover on their forest land.

- Some landowners seek high-quality, large-diameter pine trees to take advantage of the larger product sizes and higher unit values that sawtimber brings compared to pulpwood.
- The middle ground of silvicultural activity within streamside management zones falls between the two extremes of hands-off or high-grading; regeneration methods that retain some overstory trees may be a more robust way to manage these areas sustainably in the future.
- The shift away from clearcutting on public lands has been coupled with increased reliance on other even-aged and uneven-aged regeneration methods, which meet ecosystem needs that cannot be satisfied by clearcutting.

A key to successful natural regeneration of southern pines is in the wide range of fruitfulness among individual trees. Seed production in the pines is a highly inherited genetic trait, so foresters must pay attention to the inherent differences in capacity when selecting trees being retained as seed producers. This is easy to do with shortleaf pines, because their cones persist in the crown. For the other three southern pines, one should examine cones at the base of the tree or use binoculars to scrutinize developing cones.

The seed tree method reserves 4 to 10 dominant or codominant pines per acre, with a corresponding residual basal area of 5 to 15 square feet per acre (fig. 11). The method is most easily applied in loblolly and slash pine; both are abundant seed producers, and seedlings thrive in the open conditions found in the understory after a recent seed-tree harvest. Shortleaf pine can also be managed using this method, if attention is given to retaining effective seed producers and properly preparing the site. Zeide and Sharer (2000) outline the typical seed-tree prescriptions for mixed loblolly-shortleaf stands in

**Figure 11.** A mixed loblolly-shortleaf pine stand in the upper western Coastal Plain (Gulf of Mexico), managed using the seed tree method. (Photo by James M. Guldin)

the upper western Coastal Plain as practiced by forest industry in southern Arkansas over the last three decades of the 20th century.

Application of the seed-tree method starts with late-rotation thinning or preparatory cutting to encourage crown development in trees likely to be retained for seed production. The seed cut then removes all but a few residual trees per acre, in association with site preparation treatments to dispose of logging slash and remove competing vegetation. Frequently, the normal scarification of the site associated with logging is sufficient to expose mineral soil, which is the best seedbed for germinating and establishing the pines. A properly timed prescribed fire can help with this, especially when regenerating shortleaf pine. Several years after the new age cohort is established, the seed trees can be cut. Subsequent treatments in the first decade after the seed cut are likely to include chemical release of the pines from competing hardwoods and precommercial thinning to control pine stem density. In the second decade and beyond, a typical prescription includes commercial thinning on a 7- to 10-year cycle, an herbicide application every 10 years to control encroaching hardwoods, and reintroduction of prescribed fire on a 3- to 5-year cycle to retain open understory conditions.

The shelterwood method reserves 15 to 30 dominant or codominant pines per acre, with a corresponding residual basal area of 20 to 40 square feet per acre. The most practical use of the shelterwood method is to regenerate species that have erratic or unreliable seed production, and thus for which the seed tree method is uncertain. The extra trees retained in the shelterwood can make an important difference between marginal and adequate stocking by providing added seed production potential and helping modify the microclimate in the regeneration zone to favor pine seedling survival.

A classic example of shelterwood method in southern pines is the work done in the 1970s with longleaf pine in southern Alabama (Boyer 1979, Croker and Boyer 1975). The limitations of seed production were overcome through careful attention to the fruitfulness and the basal area of residual trees, with 30 to 40 square feet per acre of basal area deemed optimal (Maple 1977). Prescribed fires were used to control brown-spot needle blight. The shelterwood optimized the relationship between seed production and the amount of needlefall required to support regular prescribed burning. This example is essentially a silvicultural application of the stored seedling bank beneath the seed trees, which develops into the succeeding age cohort as seedlings break from the grass stage, ideally in 3 to 5 years after germination (fig. 12).

**Figure 12.** A longleaf pine stand in the lower Atlantic Coastal Plain, managed using the shelterwood method. (Photo by Dan Wilson)

As with the seed-tree method, the shelterwood method starts with late-rotation thinning or preparatory cutting that encourages crown and cone development. The seed cut then removes those trees not marked for retention, and site preparation treatments dispose of logging slash and remove competing vegetation—with logging and site preparation activities also preparing the seedbed. Prescribed fire is implemented shortly after the seed cut, especially important for longleaf pine. Several years after the new age cohort is established, the seed trees can be harvested using a removal cut. The added number of seed trees in the shelterwood (compared to the seed-tree method) can actually benefit the removal cut, because they provide harvest volumes sufficient to attract a logger. Conversely, removing the larger number of pines may result in unacceptable logging damage to the regeneration cohort, especially if stocking is marginal. Some managers may want to retain the seed trees through the subsequent rotation for reasons related to structural diversity, but this comes at a cost of reduced volume growth in the new age cohort. Subsequent treatments after the seed cut are similar to those in the seed-tree method: chemical release of the pines from competing hardwoods, precommercial thinning to control pine stem density, commercial thinning on a 7- to 10-year cycle, an herbicide application every 10 years to control encroaching hardwoods, and reintroduction of prescribed fire on a 3- to 5-year cycle to retain open understory conditions.

## Uneven-aged methods

Applying uneven-aged regeneration methods in species that are shade-intolerant seems counterintuitive, but the earliest successful examples of the selection method were in pines, for example the German Dauerwald method of forest management, patterned after nature, promoting sustainably productive, profitable, environmentally stable, biologically diverse, socially responsive forests (Troup 1952) as applied to Scots pine (*Pinus sylvestris*); and the improvement selection in Arizona (Pearson 1950) as applied to ponderosa pine (*Pinus ponderosa*). In the South, the longest record of success with uneven-aged management has been in Coastal Plain loblolly-shortleaf pine stands of southeastern Arkansas (Baker 1986, Baker and others 1996, Guldin 2004, Guldin and Baker 1998, Reynolds and others 1984), with other long-term demonstrations reported in Mississippi (Farrar and others 1989) and southwestern Arkansas (Farrar and others 1984). Uneven-aged methods have also been used for longleaf pine in Florida and Alabama (Brockway and Outcalt 1998, Farrar 1996, Mitchell and others 2006) and shortleaf pine in the Ouachita Mountains (Guldin and Loewenstein 1999, Lawson 1990). Research on uneven-aged regeneration methods in slash pine is virtually nonexistent, but results from Langdon and Bennett (1976) suggest that the group selection method may show some promise, and other methods suitable for longleaf pine should also be effective for slash pine. In short, the selection method can be used in southern pines if attention is paid to marking, regeneration, and stand structure (Guldin and Baker 1998).

The group selection method offers ecological and administrative advantages in managing the intolerant southern pines. Openings can be made without leaving seed trees, instead relying on existing advance growth, natural seedfall from adjacent trees, or supplemental planting. Retaining some residual trees at shelterwood basal areas within group openings is also an option for longleaf pine (Farrar 1996, Guldin 2006), and probably shortleaf pine as well. Once the pine seedlings are established, the relatively open conditions within group opening resemble the conditions that are most favorable for the southern pines—more so with larger group openings than with smaller ones. Administratively, followup treatments such as cleaning or precommercial thinning are targeted specifically to the openings, an easy process to work into operational contracts using maps or geographic locations.

The group selection method has one major disadvantage: although easy to initiate, group openings are difficult to maintain over repeated cutting cycles without strictly adhering to an area-based regulation system, which can eventually become an even-aged patch clearcutting system rather than an uneven-aged selection system. That is not important to the trees, but might be important for managers who have committed to specific proportions of even-aged versus uneven-aged acreages, as is often true for national forest management planning.

The single-tree selection method also offers advantages and disadvantages. The seven-decade experience with the Farm Forestry Forty demonstrations at the Crossett Experimental Forest in south Arkansas (fig. 13) had its origins in the rehabilitation of understocked stands (Baker and Shelton 1998) and was imposed using a simple marking rule—cut the worst trees and leave the best, regardless of diameter or pattern of occurrence. Stands that had initially been understocked recovered to full stocking within two decades. Details of the implementation of the selection method in these mixed loblolly-shortleaf pine stands (Baker and others 1996, Guldin 2002, Guldin and Baker 1998) serve as appropriate mensurational guidelines for any of the intolerant southern pines that are managed using volume regulation with a guiding diameter limit, or structural regulation (BDq method) using preset targets for residual basal area (B), a maximum retained diameter (D), and the rate of change in density in adjacent size classes classes (q).

The biggest disadvantage of the selection methods in intolerant southern pines is the management commitment required to maintain proper stand structure, especially with single-tree selection. The concept behind single-tree selection is to manage size classes rather than age classes, relying on the assumption that diameter approximates age in stands with three or more age classes. To maintain adequate sunlight in the understory for development of the seedling and sapling classes, the overstory and midstory diameter classes of the stand must be deliberately maintained in a slightly understocked condition (less than 75 square feet per acre, assuming that annual growth of most uneven-aged southern pine stands is 2 to 3 square feet per acre). Cutting-cycle harvests usually leave from 45 to 60 square feet per acre, which suggests that the cutting cycle must be 10 years or less to maintain acceptable understory development. If

**Figure 13.** A mixed loblolly-shortleaf pine stand in the upper west Coastal Plain (Gulf of Mexico), managed using uneven-aged methods. (Photo by James M. Guldin)

timely cutting-cycle harvests are not repeatedly maintained, the understory development needed to maintain stand structure will be lost. Midstory and overstory crown classes will revert to a homogeneous canopy profile more typical of a late-rotation even-aged stand, rather than the heterogeneous canopy profile that characterizes a well-regulated uneven-aged stand.

# Conclusions

Forest managers have many options for providing the mix of commodities and amenities desired by society thanks to the diversity of tree species in the Eastern United States, the large ecological breadth and geographic distribution that most species exhibit, and the variety of uneven- and even-aged silvicultural systems available. Silvicultural systems are designed to achieve multiple resource objectives—often simultaneously—within ecological, social, and economic constraints. Silvicultural stand prescriptions integrate resource objectives, apply ecological principles, and identify the system of treatments that are effective and efficient in achieving forest goals with a degree of certainty.

Rarely do foresters treat stands for a single reason or for a short-term goal, such as fuels management. However, available funding often drives on-the-ground management operations, and the failure of forest management plans comes when implementation of newly funded activities is not integrated with already established forest plan goals and silvicultural prescription objectives. The process of developing forest plans and silvicultural prescriptions provides an opportunity to integrate all management activities before implementation, and to coordinate and schedule treatments to achieve the desired management outcomes efficiently and effectively. This integrated planning to achieve a common mission increases the probability that treatments maximize attainment of goals and objectives, give the biggest bang for the dollar, and minimize the likelihood of outcomes that are in conflict with other resource goals. This is possible because silvicultural systems are dynamic and can be adapted as new knowledge accumulates, management goals change, and stochastic events alter forest condition and succession from the desired pathways.

Preparing for the certain future attack by nonnative invasive species, periodic outbreaks of native species, and the inevitable environmental extreme events requires the proper application of silviculture within the framework of sound forest and regional planning. Silvicultural prescriptions can be developed to treat current stand conditions, manage composition, and promote tree vigor and forest health. Healthy forests are less susceptible to attack by insects and diseases, less vulnerable when attacked, and more able to survive and recover from the biotic attacks or stress from environmental extremes. The most effective forest plans seek to diversify composition and structure of forests, woodlands, and savannas across the landscape and thereby buffer the effects of pest outbreaks and harsh climates.

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Cumulative Watershed Effects of Fuel Management in the Eastern United States

CHAPTER 3.

# Geographic Considerations for Fire Management in the Eastern United States: Geomorphology and Topography, Soils, and Climate

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Across the Eastern United States, there is on average an estimated 36 MT $ha^{-1}$ (16 tons $ac^{-1}$) of dead woody fuel (Chojnacky and others 2004). Variations in fuel type, size, and flammability make the selection of treatment options critical for effective fuels management. The region is a complex landscape characterized by highly fragmented forests, large areas of wildland-urban interface, and vast differences in geomorphology, topography, soils, and climate. For example, the Coastal Plain is generally flat, has large areas of wetlands, and is derived from sedimentary parent material. By contrast, the Piedmont and Appalachian Mountains are derived primarily from igneous and metamorphosed igneous parent materials, have complex topography, and few or no wetlands. Understanding interactions among fuel management treatments and geographic areas, and matching treatment prescriptions with physical conditions is critical.

Fire and fuel management options are constrained by complex interactions among physical, biological, and social parameters. Biological and social parameters can be altered somewhat by management activities, new technologies, and policies; whereas physical parameters are generally not easily altered. Except where major changes have been possible (such as drainage of hydric ecosystems in the Coastal Plain), variation in physical parameters constrains fuel and fire management options among and within geographic areas.

The purpose of this chapter is to describe the geomorphology, topography, climate, and soils of major landscapes in the Eastern United States. The information is derived from several publications and provides the backdrop for understanding fire and fuel management options.

Although many levels of resolution in landscape variations have been described for the Eastern United States (Bailey 1995, Cleland and others 2007, Reed and Bush 2005), they can generally be characterized by eight major ecological divisions the basic geographic units described in this chapter; figure 1 shows the ecological provinces within each ecological division. Table 1 contains the geologic time scales for reference, and table 2 is a comparison of mean annual temperature, precipitation, and elevation among ecological divisions.

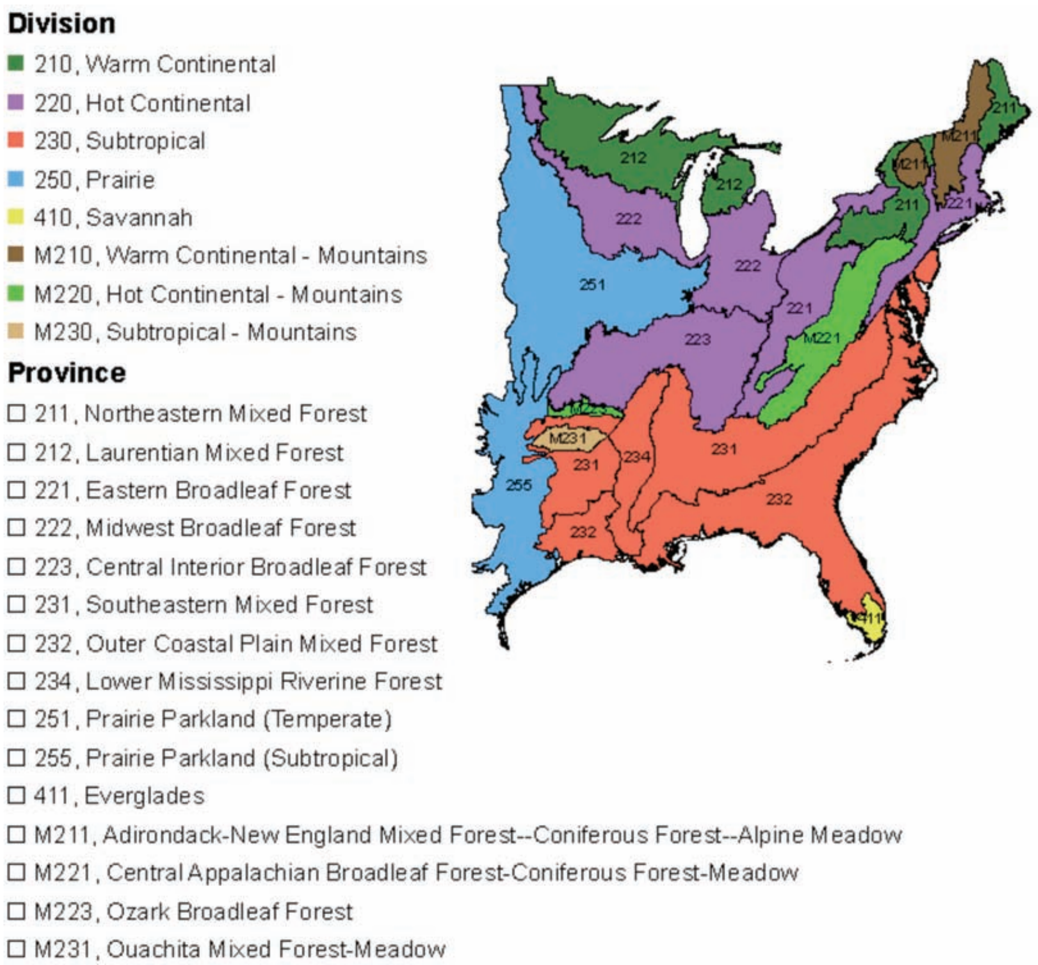


**Figure 1.** Ecological Divisions and Ecological Provinces in the Eastern United States, characterized by distinct biophysical features such as vegetation, topography, geology, soils, and climate (McNab and others 2005).

# Geomorphology and Topography

A detailed description and comparative analysis of the temporal and spatial variations in topography and geology in the Eastern United States is beyond the scope of this chapter. In this section we describe topographic and geologic variation (figs. 2 and 3) within and among numbered ecological categories (fig. 1) described by McNab and others (2005), with specific attention given to the major landscapes within each (Fenneman and Johnson 1946); as an example, the Southern Appalachian Ridges and Valleys described below are classified as M221A, Northern Ridge and Valley (McNab and others 2005) and as the Tennessee Section of the Appalachian Highlands–Valley and Ridge (Fenneman and Johnson 1946).

## *Hot Continental Mountains Division*

### Blue Ridge

This area (M221D, Blue Ridge Mountains) consists of several distinct topographic features, including the Blue Ridge Escarpment to the east, the New River Plateau to the north, interior low and intermediate mountains throughout, intermountain basins between major mountains, and the high mountains making up the bulk of acreage. Elevations range from around 275 m (900 feet) at the southern and southwestern boundaries to more than 2010 m (6,600 feet) at the crest of the Great Smoky and Black Mountain ranges.

**Table 1.** Geologic time scales

| | | Years before present |
|---|---|---|
| | | *millions* |
| Geologic era | Cenozoic | 0 to 65 |
| | Mesozoic | 65 to 230 |
| | Paleozoic | 230 to 570 |
| | Precambrian | 570 to 4,500 |
| Geologic period | Quaternary | 0 to 2 |
| | Tertiary | 2 to 65 |
| | Cretaceous | 65 to 140 |
| | Jurassic | 140 to 190 |
| | Triassic | 190 to 230 |
| | Permian | 230 to 280 |
| | Pennsylvanian | 280 to 310 |
| | Mississippian | 310 to 345 |
| | Devonian | 345 to 405 |
| | Silurian | 405 to 425 |
| | Ordovician | 425 to 500 |
| | Cambrian | 500 to 570 |
| Geologic epoch | Recent (Holocene) | 0 to 0.010 |
| | Pleistocene | 0.01 to 2 |
| | Pliocene | 2 to 10 |
| | Miocene | 10 to 25 |
| | Oligocene | 25 to 40 |
| | Eocene | 40 to 55 |
| | Paleocene | 55 to 65 |

The bedrock geology in this area consists mostly of Precambrian metamorphic rock formations with a few small bodies and windows of igneous and sedimentary rocks. The degree of metamorphism varies but generally decreases westward. The higher grade metamorphic rocks include formations of gneiss, schist, and amphibolite. Low-grade metamorphic formations in the southwest include distinct and interbedded bodies of metasandstone, slate, phyllite, metasiltstone, and metaconglomerate. The northern Blue Ridge formed during a period of post-Cretaceous uplift along the eastern coast of North America, forming a sequence of resistant minerals primarily chlorite-actinolite, schist, schistose metabasalt, siliceous metabreccia, laminated metasedimentary gneiss, and quartzite. Surficial deposits in both the Northern and Southern Blue Ridge include colluvial material on fans and aprons along the ridges and alluvial material along the major streams.

## Southern Appalachian Ridges and Valleys

Most of this area (M221A, Northern Ridge and Valley) is in the Tennessee Section of the Appalachian Highlands–Valley and Ridge. The thin stringers in the western part of the area are mostly in the Cumberland Plateau Section of the Appalachian Highlands–Appalachian Plateaus. A separate area in northern Alabama is in the Highland Rim Section of the Interior Plains–Interior Low Plateaus. The western side of the area is dominantly hilly to very steep and is rougher and much steeper than the eastern side, much of which is rolling and hilly. Elevation ranges from 200 m (660 feet) near the southern end of the area to more than 730 m (2,400 feet) in the part of the area in the

**Table 2.** Ranges in elevation, mean annual precipitation, and mean annual temperature within Ecological Divisions

| Division | | Elevation[a] | Mean annual precipitation | Mean annual temperature |
|---|---|---|---|---|
| | | *m* | *mm* | *°C* |
| Hot Continental Mountains (M220) | Minimum | 200 (SARV) | 915 (BR) | 8 (BR) |
| | Maximum | 2010 (BR) | 3000 (BR) | 17 (SARV) |
| Warm Continental Mountains (M210) | Minimum | 305 (NNU) | 815 (AD) | 1 (NNU) |
| | Maximum | 1525 (NNU) | 2665 (NNU) | 8 (AD) |
| Prairie (250) | Minimum | 200 | 485 | 4 |
| | Maximum | 300 | 1220 | 17 |
| Sub-tropical (230) | Minimum | m.s.l. (LCP) | 940 (SP) | 12 (SP) |
| | Maximum | 400 (SP) | 1830 (UCP) | 25 (SAV) |
| Sub-tropical Mountains (M230) | Minimum | 200 (BM) | 990 (AVR) | 13 (BM) |
| | Maximum | 840 (AVR) | 1675 (OM) | 17 (OM) |
| Hot Continental (220) | Minimum | 160 (IPL) | 485 (IPL) | 4 (IPL) |
| | Maximum | 505 (NP) | 1320 (NP) | 17 (IPL) |
| Warm Continental (210) | Minimum | 275 (NGL) | 660 (NGL) | 4 (NGL) |
| | Maximum | 1100 (APC) | 1755 (NNC) | 10 (APC) |
| Savanna (410) | Minimum | m.s.l. | 1015 | 23 |
| | Maximum | 5 | 1575 | 25 |

Note: Abbreviations in parentheses represent physiographic provinces within each ecological division where the minimum or maximum values occur. BR = Blue Ridge, SARV = Southern Appalachian Ridge and Valley, NNU = Northern New England Uplands, AD = Adirondack Shield, SAV = Savannas, SP = Southern Piedmont, UCP = Upper Coastal Plain, LCP = Lower Coastal Plain, IPL = Interior Plains and Lowlands, NP = Northern Piedmont, NGL = Northern Great Lakes, APC = Allegheny Plateau and Catskills, AVR = Arkansas Valley and Ridges, BM = Boston Mountains, Ouachita Mountains.
[a] m.s.l. = mean sea level
Source: U.S. Department of Agriculture, Natural Resources Conservation Service (2006).

western tip of Virginia. Some isolated linear mountain ridges rise to nearly 1500 m (4,920 feet) above sea level. This area is highly diversified. It has many parallel ridges, narrow intervening valleys, and large areas of low, irregular hills. Many ridges and valleys have a difference in elevation of 200 m (660 feet).

The bedrock in this area consists of alternating beds of limestone, dolomite, shale, and sandstone of early Paleozoic age. Ridgetops are capped with more resistant carbonate and sandstone layers, and valleys have been eroded into the less resistant shale beds. These folded and faulted layers are at the southernmost extent of the Appalachian Mountains. The narrow river valleys are filled with unconsolidated deposits of clay, silt, sand, and gravel.

## Cumberland Plateau

The northern third of this area (M221C, Northern Cumberland Mountains) is primarily in the Kanawha Section of the Appalachian Highlands–Appalachian Plateaus. The southern two-thirds is primarily in the Cumberland Plateau Section of the Appalachian Highlands–Appalachian Plateaus; a strip along the central part of the eastern edge of the area is in the Cumberland Mountain Section. Small areas along the southwestern edge are in the Highland Rim Section of the Interior Plains–Interior Low Plateaus.

This highly dissected portion occurs mainly as a series of long, steep side slopes between narrow ridgetops or crests and narrow stream flood plains. Elevation ranges from 200 m (650 feet) on the flood plain along the Ohio River to about 300 m (980 feet)

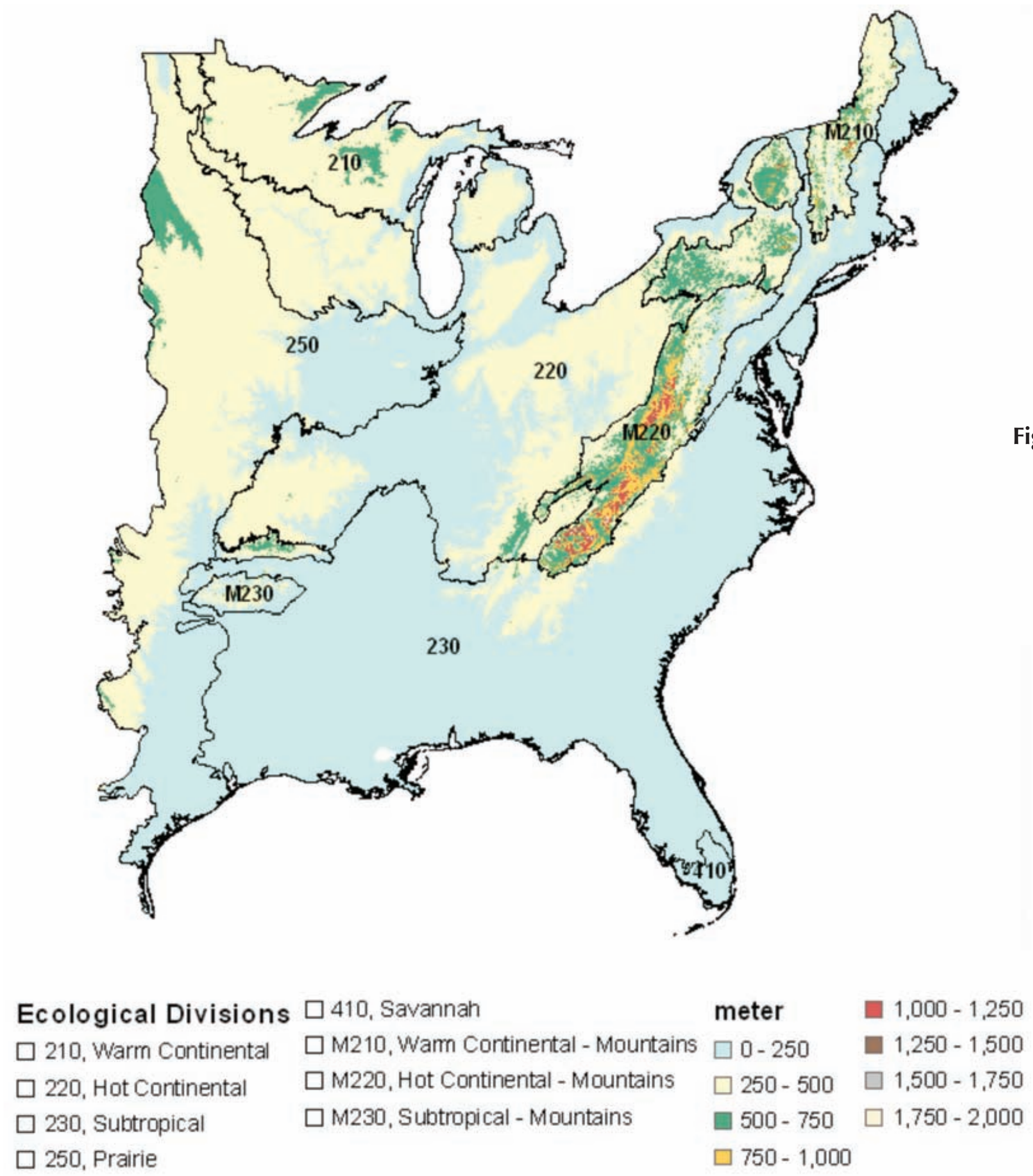


**Figure 2.** Elevations within Ecological Divisions in the Eastern United States (U.S. Geological Survey, Earth Resources Observation and Science Center 2007).

on nearby ridge tops. It gradually rises from these areas to areas near the Virginia–Kentucky border, where it is about 505 m (1,650 feet) on local flood plains and 1205 m (3,950 feet) on the higher mountains.

Cyclic beds of sandstone, siltstone, clay, shale, and coal of Pennsylvanian age form the bedrock in most of this area. Pennsylvanian limestone and dolomite bedrock is in the part of the area in Virginia and Alabama. Coal mining is the major industry. Unconsolidated deposits of silt, sand, and gravel are in the major river valleys and on terraces along these rivers. The lower parts of many hillslopes have a thin layer of colluvium.

## *Warm Continental Mountains Division*

### Adirondack Shield and Northern New England Uplands

Because of the similarities between these two areas (M211A, White Mountains; M211B, New England Piedmont; M211C, Green–Taconic–Berkshire Mountains;

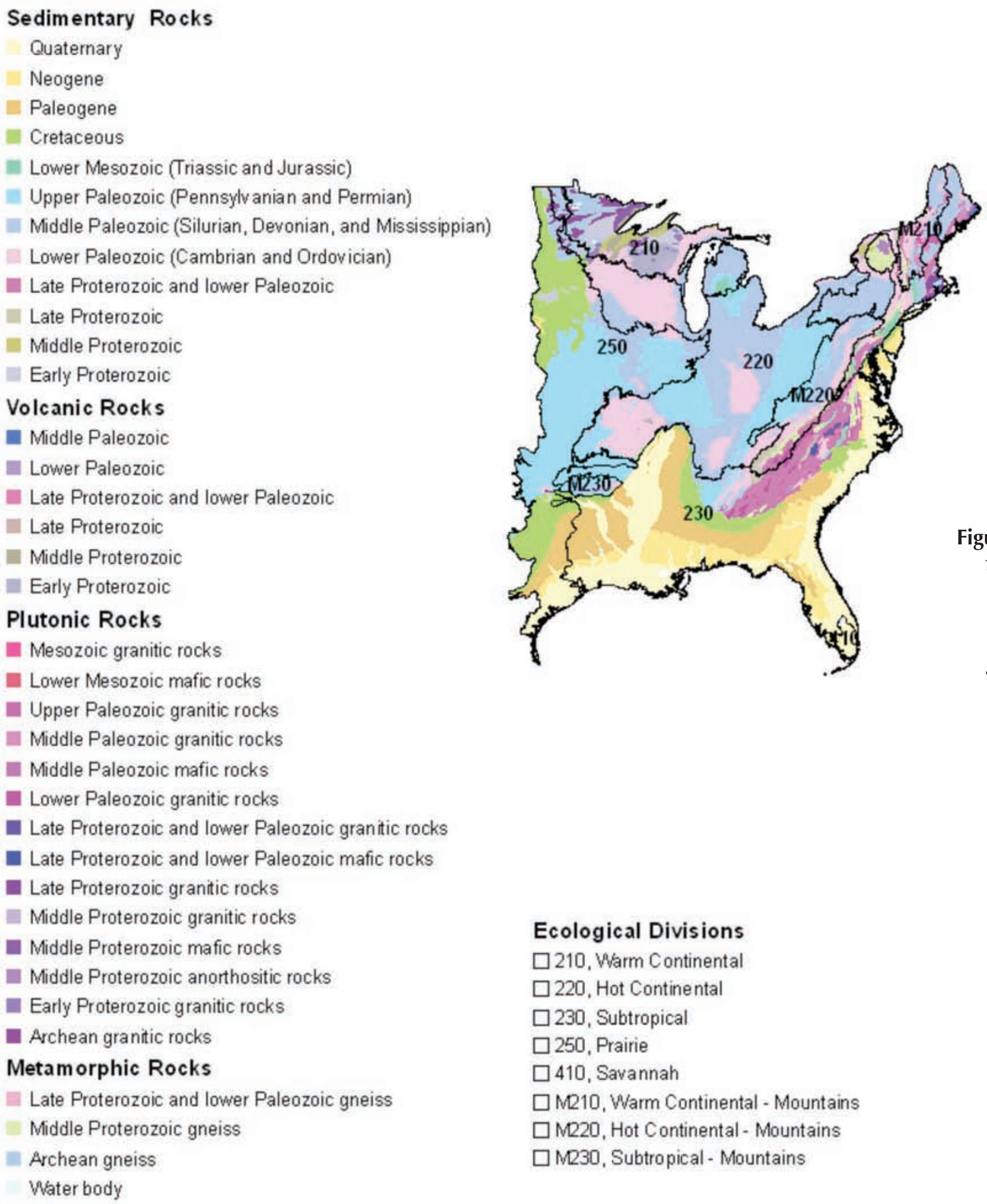


**Figure 3.** Geological formations within Ecological Divisions in the Eastern United States (Reed and Bush 2005).

M211D, Adirondack Highlands), we elected to treat this division as one unit. The westernmost part of this area is primarily in the Appalachian Highlands–Adirondack. A small area in the southern end of the western part is in the Mohawk Section of the Appalachian Highlands–Appalachian Plateaus. The easternmost part, primarily in northern Maine, is in the New England Upland Section of the Appalachian Highlands–New England, its southwestern half is in the White Mountain Section, and its middle part of this area is in the Green Mountain Section.

The mountains and foothills in the area are commonly rounded. They are underlain by bedrock and are typically covered with thin deposits of glacial till. The more rugged mountain areas are separated by high gradient streams coursing through steep areas of colluvium or talus-laden valleys. Many glacially broadened valleys are filled with glacial outwash and have numerous swamps and lakes. The mountains and foothills are moderately steep to very steep, and the valleys are nearly level to sloping. Elevation generally ranges from 305 to 1220 m (1,000 to 4,000 feet), but it is more than 1525 m (5,000 feet) on a few isolated peaks and is less than 305 m (1,000 feet) in some of the valleys, especially in northeastern Maine. Local relief ranges from moderate in some areas to high in ruggedly mountainous areas.

The entire portion of this division was glaciated by the last continental ice sheet. In addition, evidence on the more rugged mountain peaks indicates that alpine glaciation may have lingered after the retreat of Wisconsin ice. A thin mantle of till covers most of the bedrock. Sandy glacial outwash has been deposited in many stream valleys, and ice-contact, stratified drift (on kames and eskers) has been deposited on the walls of the valleys. When the European and African Continents were squeezed up against the North American Continent by plate tectonic activity, the mountains must have appeared to be similar to the present Himalaya Mountains. For the past 500 million years, as the Atlantic Ocean opened up and the European and African continental plates were pushed east, erosion has been the dominant process. Only the roots of those ancient mountains remain today. The bedrock consists primarily of igneous and metamorphic rocks. The metamorphic rocks (gneiss, schist, slate, metanorthosite, marble, and quartzite) are the oldest. The igneous rocks, primarily granite and granodiorite, were intruded into the metamorphic rocks during the Triassic and Cretaceous periods. The deformation history and the weathering of these rocks have left numerous fractures, joints, bedding plane partings, and cleavage partings that now contain freshwater.

## *Prairie Division*

Almost all the eastern portion of the area (251C, Central Dissected Till Plains; 251D, Central Till Plains and Grand Prairies) is on the glaciated Bloomington Ridged Plain in the Till Plains Section of the Interior Plains–Central Lowland, and the northern tip is in the Eastern Lake Section. The western portion is on the eastern side of the Illinois River on the glaciated Springfield Plain. The extreme western part is dominantly on the Galesburg Plain. The northern part of this western area also encompasses the Green River Lowland and the Rock River Hill Country.

The entire area was glaciated and has deposits of loess of various thicknesses. The area is on a relatively young, moderately dissected to strongly dissected, rolling plain where stream terraces are adjacent to broad flood plains along the major streams and rivers. Slopes are generally less than 15 percent but are significantly steeper in some areas along the major streams. Elevation ranges from 200 m (660 feet) in the eastern and southern parts of the area to about 300 m (985 feet) in the western and northern parts. The maximum local relief is about 50 m (160 feet) along the major streams and along the dissected drainage-ways fingering into the uplands. Relief is considerably lower in much of the area. It typically is only 1 to 3 m (3 to 10 feet) on the broad, flat uplands. The eastern portion is a relatively young, moderately dissected, rolling plain with stream terraces adjacent to the broad flood plains along the major streams and rivers. Glacial moraines are numerous and tend to form elongated ridges tending from northwest to southeast. Slopes are generally less than 5 percent but are significantly steeper on the moraines and along the major streams. Elevation ranges from 200 m (660 feet) in the southern part of the area to about 300 m (985 feet) in the northern part. The maximum local relief is about 50 m (160 feet) along the major streams. Relief is considerably lower, however, in most of the area. It typically is only 1 to 3 m (3 to 10 feet) on the broad, flat uplands.

This area is underlain by Pennsylvanian shale, siltstone, and limestone in the southern part and Ordovician and Silurian limestone in the extreme northern part. Coal beds occur in the northern part and east of the Illinois River. Glacial drift covers all of the area, except for the bluffs along the major streams where the underlying bedrock is exposed. The glacial drift is Wisconsin in age to the east and Illinoian age to the west, and consists of distinct till units as well as sorted, stratified outwash. The entire area has been covered by a moderately thin or thick layer of loess. In a few areas the loess directly overlies the bedrock.

## *Subtropical Division*

### Southern Piedmont

The Southern Piedmont (231A, Southern Appalachian Piedmont; 231I Central Appalachian Piedmont) extends from Maryland southwest to Alabama and is bounded on the southeast by the Upper Coastal Plain and to the northwest by the Blue Ridge. Almost all of this area is in the Piedmont Upland Section of the Appalachian Highlands–Piedmont. A very small part in central North Carolina is in the Atlantic Plain–Coastal Plain. A very small part around Roanoke, VA, is on the eastern edge of the Appalachian Highlands–Blue Ridge.

It can be generally described as consisting of broad ridges separated by sometimes deeply incised stream channels. It is highly weathered in geologic time and highly eroded during the recent past (200 years) by intensive agricultural activities. Past land use practices have resulted in a piedmont landscape where agricultural and silvicultural activities in the red clay B horizon is a common practice, and a common feature of this landscape. The area is a rolling to hilly upland with a well defined drainage pattern. Streams have dissected the original plateau, leaving narrow to fairly broad upland ridgetops and short slopes adjacent to the major streams. The associated stream terraces are minor. Valley floors are generally narrow and make up less than 10 percent of the land area. Elevations range from 100 to 400 m (330 to 1,310 feet).

Precambrian and Paleozoic metamorphic and igneous rocks underlie most of this area. The dominant metamorphic rock types include biotite gneiss, schist, slate, quartzite, phyllite, and amphibolite. The dominant igneous rock types are granite and metamorphosed granite. Some gabbro and other mafic igneous rocks also occur, and diabase dikes are not uncommon. The Carolina Slate terrain occurs just east of an imaginary centerline in the area. It consists of metamorphic rocks with some metavolcanics and metasediments. Scattered graben basins, which are bounded by faults where the ground between the faults has dropped down, occur from South Carolina to south of Charlottesville and Richmond in Virginia. These basins have Triassic and Jurassic siltstone, shale, sandstone, and mudstone. River valleys have recent alluvium and few terraces.

### Coastal Plain

This is an area of coastal lowlands, Coastal Plains, the Mississippi River Delta on the Gulf Coast, drowned estuaries, tidal marshes, islands, and beaches (231B, Coastal Plains–Middle; 232B, Gulf Coastal Plains and Flatwoods; 232E, Louisiana Coastal Prairie and Marshes: 232H, Middle Atlantic Coastal Plains and Flatwoods; 232H, Southern Atlantic Coastal Plains and Flatwoods). This area extends from Virginia to Louisiana and Mississippi, but it is almost entirely within three sections of the Atlantic Plain–Coastal Plain. The northern part is in the Embayed Section, the middle part is in the Sea Island Section, and the southern part is in the East Gulf Coastal Plain Section.

The area is mostly level to gently sloping and has low relief. It is strongly dissected into nearly level and gently undulating valleys and gently sloping to steep uplands. Stream valleys are generally narrow in their upper reaches but become broad and have widely meandering stream channels as they approach the coast. Elevations range from 25 to 200 m (80 to 655 feet), gradually increasing to the north. Local relief is mainly 3 to 6 m (10 to 20 feet) but is 25 to 50 m (80 to 165 feet) in some of the more deeply dissected areas.

This area is bounded on the west and north by the "fall line." This physiographic feature marks the western and northern extent of the unconsolidated Coastal Plain sediments and is an erosional scarp formed when this area was the Atlantic Ocean shoreline during the Mesozoic period. The Southern Coastal Plain is underlain by eroded igneous and metamorphic bedrock. Rivers and streams draining the Appalachians deposited a thick wedge of silt, sand, and gravel east and south of the fall line as delta deposits in the Atlantic Ocean. These Jurassic and Cretaceous river sediments were eventually exposed as the Coastal Plain uplifted and the sea level changed. When the sea level rose again, the Coastal Plain was submerged and covered by a thin layer of Cretaceous sands in the east and limestone, dolomite, and calcareous sands in the west. This area has a "benched" appearance because of the cycles of erosion and deposition that occurred as the area was exposed and submerged numerous times in its geologic history.

## *Savanna Division*

This area (411A, Everglades) is in the Floridian Section of the Atlantic Plain–Coastal Plain. It is on a level, low Coastal Plain that has large areas of swamps and marshes.

Poorly defined and broad streams, canals, and ditches drain the area to the ocean. Most of the area is flat, but in the interior, hummocks rise 1 to 2 m (3 to 6 feet) above the general level of the landscape and low beach ridges and dunes, mainly in the eastern part of the area, rise 3 to 5 m (10 to 15 feet) above the adjoining swamps and marshes. Elevation ranges from sea level to less than 25 m (80 feet).

This area is a young marine plain underlain by Tertiary-age rocks, including very fine-grained shale, mudstone, limestone, and dolomite beds. Limestone rock is the dominant subsurface material. A sandy marine deposit of Pleistocene age occurs at the surface in the northern part of the area.

## *Subtropical Mountains Division*

### Ouachita Mountains

This area (M231A, Ouachita Mountains; M231G, Arkansas Valley) is in the Ouachita Mountains Section of the Interior Highlands–Ouachita.

Most of the stream valleys are narrow and have steep gradients, but wide terraces and flood plains border the Ouachita River in western Arkansas. Elevation ranges from 100 m (330 feet) on the lowest valley floors to 800 m (2,625 feet) on the highest mountain peaks. Local relief is generally 30 to 60 m (100 to 200 feet), but can exceed 300 m (980 feet).

These steep mountains are underlain by folded and faulted sedimentary and metamorphic rock, dominantly shale and sandstone. Ordovician-age shale and sandstone are included in the Collier Shale, Crystal Mountain Sandstone, and Womble Shale. Mississippian-age shale, sandstone, novaculite, and chert are included in the Arkansas Novaculite and the Stanley Shale. Pennsylvanian-age shale, slate, quartzite, and sandstone are included in the Jackfork Sandstone, Johns Valley Shale, and upper Atoka Formations. Alluvial deposits of silt, sand, and gravel are on the wide terraces and flood plains that border the Ouachita River.

### Ozark Highlands

This area (223A, Ozark Highlands) is in the Springfield–Salem Plateaus Section of the Interior Highlands–Ozark Plateaus. The landscape ranges from highly dissected, steeply sloping wooded hills and narrow, gravelly valleys in the central and southern parts of the area to gently rolling prairie-like uplands in the northern part. Soluble carbonate rocks are responsible for a well developed karst topography in the southern part of the area. This topography includes sinkholes, caves, dry valleys, box valleys, and large springs. Elevation ranges from about 90 m (300 feet) on the southeastern edge of the Ozark escarpment to about 490 m (1,600 feet) on the western side of the area. Relief is generally 60 to 245 m (200 to 800 feet). It is highest in the southwestern part of the area. The geologic strata generally are horizontally bedded, but with a slight dip to the west and south away from the apex of the Ozark Uplift in southeastern Missouri.

This area has a variety of geologic formations. Most of the bedrock consists of sedimentary rocks, including Ordovician-age dolostone and sandstone, Lower Mississippian-age limestone and dolostone, and Pennsylvanian-age sandstone and shale. Remnants of an ancient loess deposit ranging from a few inches to several feet in thickness are on the nearly level upland divides. The loess is thickest in the northern and eastern parts of the area. Most of the exposed bedrock consists of limestone and dolostone formations that have thick layers of chert bedrock or chert fragments. The chert generally occurs in long, wavy beds less than 1 foot thick. In some areas, however, it occurs in massive layers more than 2 meters (6 feet) thick. Several old and inactive geologic faults are in the area.

### Eastern and Western Arkansas Valley and Ridges

Most of this area (M231A, Ouachita Mountains, 231G, Arkansas Valley) is in the Arkansas Valley Section of the Interior Highlands–Ouachita, and in the Osage Plains Section of the Interior Plains–Central Lowland. Elevation ranges from 90 m (300 feet) on the lowest valley floors to 840 m (2,750 feet) on the mountaintops. In the east, the topography consists of long, narrow ridges and high flat-topped mountains capped with sandstone that trend northeastward. Crests are narrow and rolling on ridges and broad and flat on mountaintops. The intervening valleys are broad and smooth. In the west, the topography of the area is characterized by long, narrow sandstone-capped ridges that trend northeastward. The ridges are dissected by valleys cut by streams at right angles to the ridges.

In the east, the ridges and valleys are underlain by slightly folded to level beds of sandstone and shale, respectively. The area principally consists of the Savanna group, McAlester group, Hartshorne sandstone group, and the upper and lower Atoka group. These are all of Pennsylvanian age. The terrace deposits along the Arkansas River include a complex sequence of unconsolidated gravel, sandy gravel, sands, silty sands, silts, clayey silts, and clays. The individual deposits commonly are lenticular and discontinuous. At least three terrace levels are recognized. The lowest is the youngest. In the west, the area principally consists of hard and soft sandstone, shale, siltstone, limestone, and some conglomerates of the Cabaniss, Krebs, and Marmaton groups. These are all of Pennsylvanian age. They may include economically viable coal deposits. The bedrock geology of the area is tilted 2 to 15 degrees from the horizontal and is gently folded in some areas. Unconsolidated clay, silt, sand, and gravel are deposited in the river valleys.

### Boston Mountains

This area (M223A, Boston Mountains) is mostly in the Boston Mountains Section of the Interior Highlands' Ozark Plateaus. The northern half of the western tip of the area is in the Ozark Plateaus' Springfield–Salem Plateaus Section. The southern half of the western tip is in the Arkansas Valley Section of the Interior Highlands' Ouachita. This area marks the southern extent of the Ozarks. It is an old plateau that has been deeply eroded. Ridgetops are narrow and rolling. Valley walls are steep. Elevation ranges from 200 m (660 feet) on the lowest valley floors to 800 m (2,625 feet) on the highest ridge crests. Local relief commonly exceeds 30 m (100 feet).

Most of this area is underlain by level to slightly tilted shale, sandstone, and siltstone strata in the Pennsylvanian-age Atoka Formation and the Cane, Boyd Shale, and Prairie Grove members of the Hale Formation. Parts of the northern edge are underlain by the Mississippian-age Pitkin Limestone,

Fayetteville Shale, and Batesville Sandstone. Alluvium consisting of an unconsolidated mixture of clay, silt, sand, and gravel is deposited in river valleys.

## *Hot Continental Division*

This ecological division is topographically quite diverse. It includes the Interior Plains and Lowlands of Indiana, Ohio, southern Illinois, and southern Michigan as well as most of Kentucky, eastern Tennessee, and portions of West Virginia and Pennsylvania. In addition, it includes the northern Piedmont of eastern Pennsylvania and northern Virginia, as well as the central New England coasts from New York to New Hampshire.

## Interior Plains and Lowlands

This area (222H, Central Till Plains–Beech–Maple) is in the Till Plains Section of the Interior Plains–Central Lowland. It is dominated by broad, nearly level ground moraines that are broken in some areas by kames, outwash plains, and stream valleys along the leading edge of the moraines. Narrow, shallow valleys commonly are along the few large streams in the area. Elevation ranges from 160 to 425 m (530 to 1,400 feet), increasing gradually from west to east. Relief is mainly a few meters, but in some areas hills rise as much as 30 m (100 feet) above the adjoining plains.

This area is underlain by Pennsylvanian shale, siltstone, and limestone in the southern part and Ordovician and Silurian limestone in the extreme northern part. Glacial drift covers all of the area, except for some areas along the major streams where the underlying bedrock is exposed. The glacial drift is Wisconsin in age and consists of distinct till units as well as sorted, stratified outwash. The entire area has been covered by a moderately thin or thick layer of loess. In a few areas the loess directly overlies the bedrock.

## Western and Central Allegheny Plateau

The physiography in the part of this area (221E, Southern Unglaciated Allegheny Plateau; 221H, Northern Cumberland Plateau) east of the Mississippi River is varied and consists of gently rolling terrain on level-bedded limestone in the Kentucky Bluegrass and Highland Rim areas. Moving eastward, the topography becomes progressively more dissected and hilly. The Appalachian Plateau, stretching from central Pennsylvania to northern Georgia, grades from a dissected plateau to a rugged band of mainly forested mountains and high hills underlain by shale, sandstone, coal, and some limestone. The Valley and Ridge features long, linear forested ridges and cropland in the valleys. The Central Allegheny Plateau is in the Kanawha Section of the Appalachian Highlands–Appalachian Plateaus. It is on a dissected plateau that is underlain mainly by horizontally bedded sedimentary rocks. The narrow, level valleys and narrow, sloping ridgetops are separated by long, steep and very steep side slopes. Elevation ranges from 200 m (650 feet) on the lowest valley floors to 400 m (1,310 feet) or more on the highest ridgetops. Local relief is about 100 meters (330 feet).

In the Western Alleghenies, cyclic beds of sandstone, siltstone, clay, shale, and coal of Pennsylvanian age form the bedrock. Similar rocks of Mississippian age occur along the southwestern edge of the area in Kentucky and southern Ohio. This area is on the eastern side of the Cincinnati Arch, so the bedrock is tilted to the east in Kentucky and Ohio. Old glacial drift deposits are in some of the major river valleys. Wisconsin-age glacial outwash deposits of unconsolidated sand and gravel are near the surface in river valleys in Pennsylvania and Ohio. Wisconsin-age glacial drift covers the surface in areas to the east and north of this area. In the Central Allegheny Plateau, the area is underlain mostly by horizontal layers of Pennsylvanian-age sandstone, siltstone, shale, coal, and some limestone. The valleys along the Ohio, Muskingum, and Kanawha Rivers have significant deposits of river alluvium (unconsolidated silt, sand, and gravel). The bedrock geology is faulted and folded shale, sandstone, and limestone.

## Northern Piedmont

Most of this area (221D, Northern Appalachian Piedmont) is in the Piedmont Upland Section of the Appalachian Highlands–Piedmont. The southwestern end and the northwestern portion of the southwestern half of this area and the southeastern portion of the northeastern half are in the Piedmont Lowlands Section. The northwestern portion of the northeastern half of the area is in the New England Upland Section of the Appalachian Highlands–New England. Most of this area is an eroded part of the Piedmont Plateau. This area is mostly gently sloping or sloping. Intrusive dikes and sills form fairly sharp ridges that interrupt the less steep terrain. Differential erosion has created low areas where rocks are soft and high areas where rocks are resistant to erosion. The steeper slopes generally are on ridges at the higher elevations or on side slopes adjacent to drainages. Elevation is dominantly 100 to 300 m (330 to 985 feet) but ranges from 25 to 300 m (80 to 985 feet) in most areas. It is as much as 505 m (1,650 feet) or more on some ridges and isolated peaks.

Most of this area is above the "fall line" on the east coast. The fall line is the boundary between Coastal Plain sediments and the crystalline bedrock of the interior uplands. The eastern third of the area is underlain mainly by Lower Paleozoic to Precambrian sediments and igneous rocks that have been metamorphosed. The typical rock types in this part of the area are granite, gabbro, gneiss, serpentinite, marble, slate, and schist. The central part of the area is a crustal trough or basin that formed during the Triassic period. This basin represents the ancestral Atlantic Ocean that formed when the European–African continental plate began its movement westward from the North American plate. Many of the rocks in this part of the area are the same rocks as those in the western British Isles, since they were deposited at a time when the North American, European, and African plates were all one landmass. The rocks deposited in the basins include Triassic sandstone, shale, and conglomerate. These ancient basins have been uplifted and are now in the uplands. Numerous Jurassic diabase and basalt dikes and sills cut the sedimentary rocks in the basins. The far western part is underlain mostly by Cambrian to Silurian limestone. The northern boundary marks the southernmost extent of the Wisconsin glaciers. Earlier periods of glaciation extend farther south in north-central New Jersey and in eastern Pennsylvania. Unconsolidated stream alluvium (primarily sand and gravel) fills the major river valleys.

## *Warm Continental Division*

### Northern Great Lakes

This area (211M, Northern Minnesota and Ontario; 212N, Northern Minnesota Drift and Lake Plains) is in the Central Lowland areas south and west of the western Great Lakes. It is a glaciated area with numerous lakes and wetlands. Slopes are nearly level to gently undulating in areas of glacial lake deposits, gently undulating to rolling on till plains and ground moraines, and steep on end moraines, on valley sidewalls, and on escarpments along the margins of lakes. In the extreme northwestern portion, these glacial lake plains have remnants of gravelly beaches, strandlines, deltas, and sandbars. The mostly level or nearly level plains are bordered by some gently sloping strandlines and rolling dune land. In this northwestern section, elevation is 410 m (1,350 feet), decreasing gradually to 275 m (900 feet) in the north. Ditches have been used in an attempt to drain the many wetlands, but low gradients commonly prevent adequate removal of surface and subsurface water.

Precambrian-age bedrock underlies most of the glacial deposits. The bedrock is a complex of folded and faulted igneous and metamorphic rocks. The bedrock terrain has been modified by glaciation and is covered in most areas by Pleistocene deposits and windblown silts. The glacial deposits form an almost continuous cover in most areas. The drift is as much as several hundred feet thick in many areas. Loess covered the area shortly after the glacial ice melted. In the extreme northwestern portion, the surface is covered mostly by silty and clayey lacustrine sediments and lake-modified glacial till. Crystalline metamorphic rocks underlie the glacial deposits.

### Glaciated Allegheny Plateau and Catskills

This area (211F, Northern Glaciated Allegheny Plateau; 211I, Catskill Mountains) is primarily in the southern New York section of the Appalachian Highlands–Appalachian Plateaus. The east-central part is in the Catskill Section. A small portion of the Allegheny Mountain Section is in the south-central part of this area, and the southwestern corner is in the Kanawha Section. The southeastern edge and a fingerlike area protruding into the southeastern corner are in the Middle Section of the Appalachian Highlands–Valley and Ridge. The top of the dissected plateau is broad and is nearly level to moderately sloping. The narrow valleys have steep walls and smooth floors. The Catskills in the east have steep slopes. Elevation is typically 200 to 305 m (650 to 1,000 feet) on valley floors; 505 to 610 m (1,650 to 2,000 feet) on the plateau surface; and 1100 m (3,600 feet) or more in parts of the Catskills.

The bedrock in this area includes alternating shale and sandstone beds of Devonian age. Some of the upper Devonian layers have been eroded away in the part of the area in New York. Glacial drift mantles the area. Significant deposits of glacial outwash, consisting of unconsolidated sand and gravel, fill most of the valley floors. Some glacial lake sediments and ice-contact and stratified drift deposits occur in most of the valleys. These deposits are the primary aquifers in this area. Younger stream deposits cover some of the glacial deposits on the valley floors.

### Northern New England Coastal Area

This area (211C, Fundy Coastal and Interior; 211D, Central Maine Coastal and Embayment; 221A, Lower New England; M211A, White Mountains; M211B, New England Piedmont) is the Appalachian Highlands–New England. The separate western part is in the Taconic Section. The rest of the area is mostly in the New England Upland Section. The part in southeastern Maine is in the Seaboard Lowland Section. This area includes the entire coastal zone of Maine and extends inland along the major river valleys. Most of the area is characterized by rolling to hilly uplands. The area has some isolated mountain peaks. In the part of the area in southeastern Maine, gently sloping to level valleys terminate in coastal lowlands. Elevation ranges from sea level to 305 m (1,000 feet) in much of the area. It is 610 m (2,000 feet) on some hills and 900 m (2,950 feet) on a few isolated peaks. Local relief is mostly low or moderate. It generally is highest in the northern part of the area and decreases as sea level is approached. An exception is the Taconic Mountains along the New York–Massachusetts border, where relief is substantial. Relief is mostly about 2 to 20 m (5 to 65 feet) in the valleys and about 25 to 100 m (80 to 330 feet) in the uplands.

Most of this area is characterized by till-mantled, rolling to hilly uplands. The northern and eastern parts of the area are underlain mostly by granite, gneiss, and schist bedrock. Limestone, dolomite, and marble beds interspersed with basalt flows occur in the southern and western parts. Stratified drift deposits of unconsolidated sand and gravel, primarily glacial outwash, fill most of the narrow river valleys. Some marine sediments occur at the lower end of the valleys that terminate in the coastal lowlands in southeastern Maine. Some glacial lake sediments occur on valley floors behind glacial moraines.

## Climate

Figures 4 and 5 show average annual precipitation and air temperature across the Eastern United States, where climate varies considerably in response to latitude, longitude, and elevation, and ranges from continental in the Interior Plains and Lowlands to marine along the coast (fig. 1). Average annual precipitation, for example, ranges from as little as 64 mm (26 inches) on the western shore of Lake Michigan to over 2500 mm (100 inches) at the highest peaks in the Southern Blue Ridge. Much of the variation in precipitation is driven by proximity and position around the Great Lakes, as well as topography. For example, although not as pronounced as in the Western United States, orographic effects can substantially influence precipitation patterns and distribution across eastern mountains, particularly in the southern Appalachians where elevational gradients are the strongest (Kittel and others 1997). For example, in the mountains of southwestern North Carolina, precipitation is approximately 30 percent greater at the high versus low elevation (a difference in elevation of approximately 700 m or 2,300 feet) (Swift and others 1988). Similarly, there is a wide range in average annual and minimum and maximum temperatures across ecological divisions. In

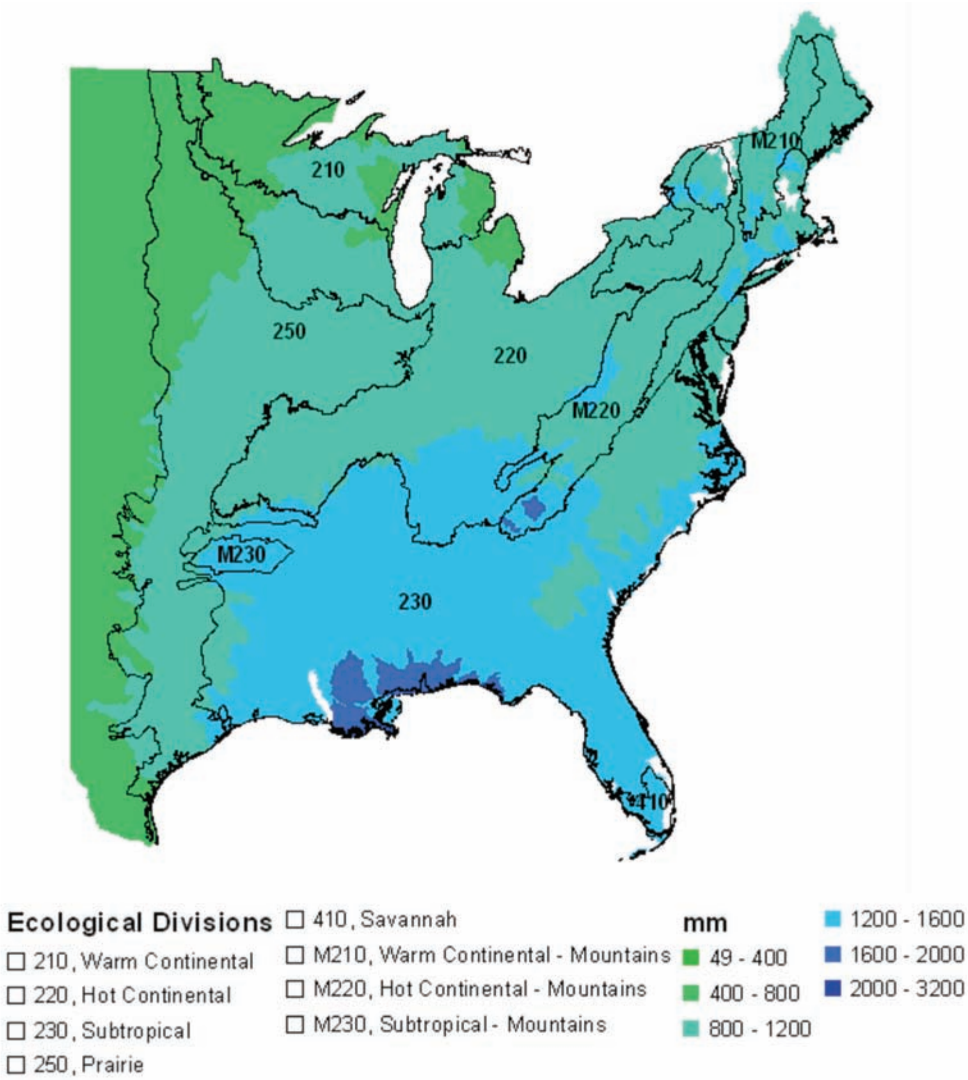


**Figure 4.** Average annual precipitation, 1963 to 1993, within Ecological Divisions in the Eastern United States (Kittel and others 1997).

the northern Great Lakes and northern Maine, average daily minimum temperature in January can reach –22 °C (–7 °F); whereas in the Savanna Division of southern Florida, average daily minimums rarely drop below 13 °C (55 °F) for the same time of year. Similarly, average daily maximum temperatures in July range from 25°C (77 °F) in northern areas of the Eastern United States and the highest elevations of the mountains to 36 °C (96 °F) in the Southern Coastal Plain.

## *Hot Continental Mountains Division*

### Blue Ridge

Average annual precipitation ranges from 915 to 1525 mm (36 to 60 inches), generally increasing with elevation and decreasing with latitude. Areas in southwestern North Carolina and northeastern Georgia rainfall amounts range from 1512 to 2300 mm (60 to 90 inches) per year and can reach totals of over 3000 mm (115 inches) on the higher peaks. Precipitation is generally lowest in October, but is well distributed throughout the year. Precipitation falls primarily as rain throughout most of the area except for the highest elevations. In the Northern Blue Ridge average annual precipitation is somewhat less than farther south and averages 915 to 1145 mm (36 to 45 inches) but can range as high as 1550 mm (61 inches) at high elevations. Unlike the Southern Blue Ridge, snow frequently covers the ground in winter and is a major contributor to total

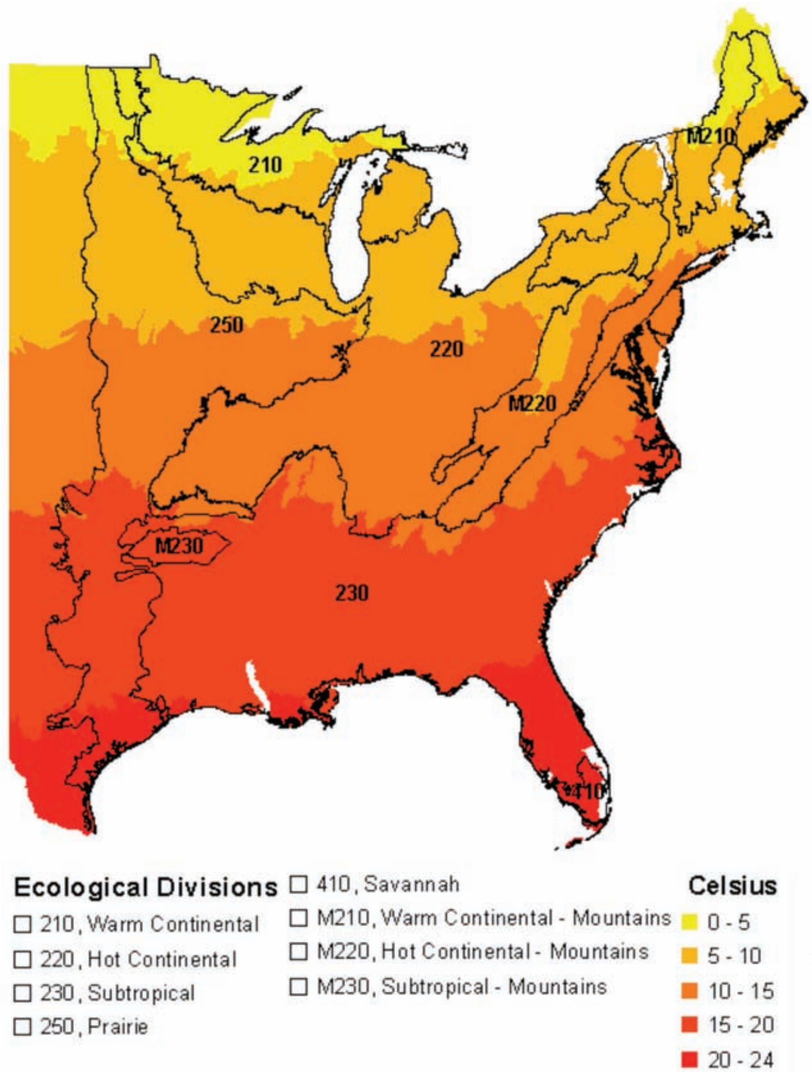


**Figure 5.** Average annual temperature, 1963 to 1993, within Ecological Divisions in the Eastern United States (Kittel and others 1997).

annual precipitation. In the Southern Blue Ridge average annual temperature ranges from 8 to 16 °C (46 to 60 °F), generally decreasing with elevation. The freeze-free period averages 185 days and ranges from 135 to 235 days. The length of this period decreases with increasing elevation, and with cold air drainage on valley floors. Strong aspect gradients exist and microclimatic differences resulting from aspect variation significantly affect the type and vigor of the plant communities in the area, driven primarily by differences in moisture and temperature regimes. South-facing and west-facing slopes, for example, are warmer and drier than north-facing and east-facing slopes and those shaded by the higher mountains. In the Northern Blue Ridge average annual temperature ranges from 9 to 14 °C (49 to 56 °F) and decreases with increasing elevation. The freeze-free period averages 195 days and ranges from 165 to 225 days, and shortens with increasing elevation.

## Southern Appalachian Ridges and Valleys

The average annual precipitation in most of this area is 1040 to 1395 mm (41 to 55 inches). It increases to the south and is as much as 1675 mm (66 inches) at the highest elevations in eastern Tennessee and the northwestern corner of Georgia. The maximum precipitation occurs in midwinter and midsummer, and the minimum occurs in autumn. Most of the rainfall occurs as high-intensity, convective thunderstorms. Snowfall may occur in winter. The average annual temperature is 11 to 17 °C (52 to 63 °F), increasing to the south. The freeze-free period averages 205 days and ranges from 165 to 245 days. It is longest in the southern part of the area and shortest at high elevations and at the northern end.

### Cumberland Plateau

Average annual precipitation ranges from 940 to 1145 mm (37 to 45 inches) in the northern third of this area and 1145 to 1525 mm (45 to 60 inches) in the southern two-thirds. It can reach 1525 mm (60 inches) at the higher elevations in the northern third of the area and can be as much as 1905 mm (75 inches) in the mountains in the southern two-thirds. Almost half of the annual precipitation falls during the growing season. Rainfall typically occurs during high-intensity, convective thunderstorms in summer. Snow may occur during winter in the northern part of the area and at the higher elevations. Average annual temperature is 10 to 15 °C (50 to 60 °F). The freeze-free period averages 200 days and ranges from 170 to 225 days. The shorter freeze-free periods are at the higher elevations and in the more northerly parts of the area.

## *Warm Continental Mountains Division*

### Adirondack Shield and Northern New England Uplands

Because of the similarities between the two areas, we elected to treat this division as one. The average annual precipitation in most of this area is 815 to 1145 mm (32 to 45 inches). It is typically 1145 to 1525 mm (45 to 60 inches) at the higher elevations in the mountains and is 1525 to 2665 mm (60 to 105 inches) on the highest peaks in the Green and White Mountains. More precipitation generally falls in summer than in winter. Most of the rainfall occurs as high-intensity, convective thunderstorms during the summer. Heavy snowfalls are common in winter. The average annual temperature is 1 to 8 °C (35 to 46 °F). The freeze-free period averages 145 days and ranges from 110 to 185 days, decreasing in length with elevation.

## *Prairie Division*

Typically, the land surface is a nearly level to gently sloping, dissected glaciated plain. The average annual precipitation is typically 815 to 990 mm (32 to 39 inches), but ranges from 485 to 1220 mm (19 to 48 inches), increasing from north to south. Most of the precipitation occurs during the growing season. In most of the area, the average annual temperature is 8 to 12 °C (47 to 53 °F), but it ranges from 4 to 17 °C (38 to 62 °F), increasing from north to south. The freeze-free period generally is 170 to 210 days, and increases in length from north to south.

## *Subtropical Division*

### Southern Piedmont

Climatic regimes fall between warm, moist-temperate and subtropical (Bailey 1989). Much of the climate is dominated by frontal activity either from off shore or continental sources. Often, the convergence of warm moist air off the coast with cooler continental air masses produces severe thunderstorms in the piedmont. Average annual precipitation ranges from 940 to 1145 mm (37 to 45 inches) at the southern end, and is as much as 1905 mm (75 inches) in a small high elevation area of northeastern Georgia. Precipitation is generally evenly distributed throughout the year but is generally lowest during the autumn months. Most of the rainfall during the growing season occurs as high-intensity, convective thunderstorms, whereas during the dormant season weather patterns tend to be dominated by less intense and more persistent frontal weather systems. Significant moisture also comes from the movement of warm and cold fronts from November to April. High amounts of rainfall are associated with tropical weather systems such as hurricanes and other significant depressions. Snowfall is typically light.

Average annual temperature is 12 to 18 °C (53 to 64 °F). The freeze-free period averages 230 days and ranges from 185 to 275 days. Both temperature and length of freeze-free period decrease from south to north and with increasing elevation.

### Upper Coastal Plain

Average annual precipitation ranges from 1040 to 1525 mm (41 to 60 inches), increasing from north to south. It is typically 1550 to 1830 mm (61 to 72 inches) in the extreme southwestern part of the area, inland along the Gulf Coast. The minimum precipitation occurs in autumn throughout the area. The maximum precipitation occurs during midsummer in the eastern part of the area and during winter and spring in the western part. Rainfall typically occurs as high-intensity, convective thunderstorms during the summer, but moderate-intensity tropical storms can produce large amounts of rainfall during winter in the eastern and southwestern parts of the area. Snowfall does not occur in the southern part of the area, but occasionally occurs in the northern part. The average annual temperature is 13 to 20 °C (55 to 68 °F), increasing from north to south. The freeze-free period averages 250 days and ranges from 200 to 305 days, increasing in length from north to south.

### Lower Coastal Plain

This area includes the Atlantic Coast Flatwoods and Tidewater. The climate is mostly temperate to hot and humid. The average annual precipitation is 1065 to 1370 mm (42 to 54 inches). It commonly exceeds 1650 mm (65 inches) along the Louisiana, Mississippi, and Alabama coastlines. The area is generally driest at the northern end and wettest at the southern end. The amount of precipitation is slightly higher during the fall and winter than during the rest of the year. Snowfall occurs in the northern third. Average annual temperature ranges from 14 to 18 °C (58 to 65 °F). The freeze-free period ranges from 220 to 305 days, increasing in length to the south.

## *Savanna Division*

This area includes the Everglades and associated areas where average annual precipitation is 1015 to 1575 mm (40 to 62 inches). About 60 percent of the precipitation occurs from June through September. The center of the area is the driest. Most of the rainfall occurs as moderate-intensity, tropical storms that produce large amounts of rain from late spring through early autumn. Late autumn and winter are relatively dry. The average annual temperature ranges from 23 to 25 °C (73 to 78 °F). The freeze-free period averages 355 days and ranges from 345 to 365 days.

## *Subtropical Mountains Division*

### Ouachita Mountains

Average annual precipitation in most of this area is 1270 to 1675 mm (50 to 66 inches). It decreases to 1040 to 1245 mm (41 to 49 inches) along the western edge of the area. The precipitation is fairly evenly distributed throughout the year. The maximum occurs in spring and early in autumn. Most of the rainfall occurs as high-intensity, convective thunderstorms. Snowfall is not common in winter. The average annual temperature is 14 to 17 °C (57 to 63 °F). The freeze-free period averages 230 days and ranges from 205 to 255 days. The shorter freeze-free periods occur at the higher elevations on the major ridges.

### Ozark Highlands

Average annual precipitation in almost all of this area is 965 to 1145 mm (38 to 45 inches). It is as high as 1245 mm (49 inches) in some small areas along the extreme southeastern and southern edges of the area. About 57 percent of the annual precipitation falls during the six warmest months of the year. Snow falls nearly every winter, but the snow cover lasts for only a few days. The annual snowfall averages about 305 mm (12 inches). Average annual temperature is about 12 to 16 °C (53 to 60 °F). The lower temperatures occur at the higher elevations in the western part. The freeze-free period averages 210 days and ranges from 175 to 245 days. It is shortest at the higher elevations along the western edge. The longer freeze-free periods occur at the lower elevations.

### Eastern and Western Arkansas Valley and Ridges

Average annual precipitation is 990 to 1170 mm (39 to 46 inches), with the western portion being the driest. Precipitation averages 1145 to 1550 mm (45 to 61 inches) in the eastern two thirds of the area. Most of the rainfall occurs as frontal storms in spring and early summer. Some high-intensity, convective thunderstorms occur in summer. Precipitation occurs as rain and snow in January and February. The average seasonal snowfall is 125 mm (5 inches). Most of the precipitation falls from April through September. The average annual temperature is 14 to 17 °C (58 to 62 °F). The freeze-free period averages 235 days and ranges from 220 to 260 days. The shorter freeze-free periods occur at the higher elevations on top of the major ridges.

### Boston Mountains

Average annual precipitation is 1065 to 1395 mm (42 to 55 inches). The maximum precipitation occurs in spring and fall, and the minimum occurs in midsummer. Most of the rainfall occurs as high-intensity, convective thunderstorms. Snowfall is uncommon in winter. The average annual temperature is 13 to 16 °C (55 to 61 °F). The freeze-free period averages 225 days and ranges from 200 to 245 days.

## *Hot Continental Division*

### Interior Plains and Lowlands

The average annual precipitation is typically 815 to 990 mm (32 to 39 inches), but it ranges from 485 to 1220 mm (19 to 48 inches), increasing from north to south. Most of the precipitation occurs during the growing season. Rainfall decreases with distance from the ocean, hence, this area is subdivided into moist oceanic and dry continental zones. In most of the area, the average annual temperature is 8 to 12 °C (47 to 53 °F), but ranges from 4 to 17 °C (38 to 62 °F), increasing from north to south. The freeze-free period generally is 170 to 210 days and increases in length from north to south.

### Northern Piedmont

Average annual precipitation is 940 to 1320 mm (37 to 52 inches). The maximum precipitation occurs as high-intensity, convective thunderstorms in spring and early in summer. Droughts of 10 to 14 days are common in summer. Snowfall occurs in winter. The average annual temperature ranges from 9 to 14 °C (48 to 57 °F). The freeze-free period averages 205 days and ranges from 170 to 240 days.

### Southern New England Coasts

Along the coast including Long Island and Cape Cod, average annual precipitation is 1040 to 1220 mm (41 to 48 inches). The precipitation is fairly evenly distributed throughout the year. Rainfall occurs as high intensity, convective thunderstorms during the summer. The seasonal snowfall is moderate to low in winter, and extended periods of no snow cover can be expected in winter because of relatively moderate temperatures. The average annual temperature is 10 to 12 °C (49 to 54 °F). The freeze-free period averages 220 days and ranges from 195 to 240 days. Farther inland, the average annual precipitation is 890 to 1145 mm (35 to 45 inches) in the Hudson Valley, which is in the northern half of the western part of this area. It is 1145 to 1370 mm (45 to 54 inches) in the southern end of the western part of the area and in most of the eastern part of the area. Precipitation generally is evenly distributed throughout the year, but decreases during the summer as you near the coast. It is slightly higher in spring and fall in inland areas. Rainfall occurs as high-intensity, convective thunderstorms during the summer. During the winter, most of the precipitation occurs as moderate-intensity storms (northeasters) that produce large amounts of rain or snow. The average annual temperature is 6 to 12 °C (44 to 54 °F), increasing from north to south. The freeze-free period averages 190 days and ranges from 145 to 240 days, increasing in length to the south.

## *Warm Continental Division*

### Northern Great Lakes

Climate varies considerably in this area. In eastern Wisconsin and around Green Bay, average annual precipitation can be as low as 735 mm (29 inches), and in portions of the Upper Peninsula of Michigan can average as low as 660 mm (26 inches). Around 20 percent of total precipitation is snowfall. Whereas on the lee side of Lake Michigan in the northern half of Michigan's southern peninsula, precipitation amounts can average 1000 mm (40 inches), and snowfall amounts can reach 3800 mm (150 inches) annually. The average annual temperature ranges from 4 to 7 °C (39 to 44 °F). The freeze-free period ranges from 120 to 175 days, increasing in length from north to south.

### Glaciated Allegheny Plateau and Catskill Mountains

Average annual precipitation in most of this area ranges from 760 to 1145 mm (30 to 45 inches). It is 1145 to 1625 mm (45 to 64 inches) in small pockets at high elevations in the eastern part. Rainfall occurs as high-intensity, convective thunderstorms during the summer, but most of the precipitation occurs as snow. Average annual temperature ranges from 4 to10 °C (40 to 50 °F). The freeze-free period averages 165 days and ranges from 130 to 200 days. The coldest temperatures and the shortest freeze-free periods are at high-elevations in the eastern part of Allegheny Plateau and Catskill Mountain portion of this area.

### Northern New England Coasts

Average annual precipitation in most of the area is 840 to 1145 mm (33 to 45 inches) and can range from 1145 to 1755 mm (45 to 69 inches) in a few scattered, higher elevation areas and along the coast. Precipitation generally is evenly distributed throughout the year. Near the coast, however, it is slightly lower during the summer months. In inland areas, it is slightly higher in spring and fall. Rainfall occurs as high-intensity, convective thunderstorms during the summer. During the winter, most of the precipitation occurs as moderate-intensity storms (northeasters) that produce large amounts of rain or snow. Heavy snowfalls commonly occur late in winter. Average annual temperature is 4 to 9 °C (39 to 48 °F). The freeze-free period averages 160 days and ranges from 120 to 195 days. Temperatures and the length of the freeze-free period increase from north to south and closer to the coast.

# Soils

The soils described in this section are classified and named in accordance with the U.S. Department of Agriculture system of classifying soils described in Soil Taxonomy (Soil Survey Staff 1999). The information and descriptions herein are derived primarily from the U.S. Department of Agriculture, Natural Resources Conservation Service (2006). Soils throughout the Eastern United States are extremely variable, ranging from glaciated tills and moraines in the north to subtropical mudflats in the south. The formation of these soils was strongly influenced by climate and mineralogy and in many areas; the surface soils are a reflection of past and present land use patterns. In the southern Piedmont, for example, past agricultural activities resulted in widespread erosion that left much of the area with surface B horizons and surface C horizons in extreme cases. The erosivity of soils when subjected to cultural activities such as farming and silviculture varies considerably, as well. For example, due primarily to mineralogy (relative proportions of clay, silt, and sand), erosivity of some soils is extreme and caution must be used when soil disturbance is planned in these areas. Steep landscapes are particularly vulnerable to erosive forces when disturbed. Hence, understanding the interactions between land management options and soil behavioral properties is critical for insuring long-term site productivity and minimal offsite impacts such as sedimentation of surface water.

## *Hot Continental Mountains Division*

### Blue Ridge

Dominant soil orders are Inceptisols and Ultisols. The soil moisture regime is udic and the soil temperature regime is mesic, but is frigid at elevations above 1280 m (4,200 feet). Soil depth ranges from shallow to very deep. The general textural class is loamy or clayey. At elevations less than 1065 m (3,500 feet), the soils on uplands generally are red, fine-loamy or fine Typic Hapludults (Evard, Junaluska, and Hayesville series). Humic Hapludults (Trimont and Snowbird series) are on northern and eastern aspects. Soils that formed in colluvium in coves are Typic Dystrudepts (Tate, Greenlee, and Northcove series), or Humic Hapludults (Saunook and Thunder series). At elevations between 1065 and 1280 m (3,500 and 4,200 feet) are generally brown, fine-loamy or coarse-loamy Dystrudepts. Humic Dystrudepts (Plott, Porters, Cheoah series) are common on northern and eastern aspects, and Typic Dystrudepts (Edneyville, Chestnut, Ditney, and Stecoah series) are common on southern and western aspects. Soils that formed in colluvium in coves are Humic Dystrudepts (Cullasaja, Spivey, Tuckasegee, and Santeetlah series) or Humic Hapludults (Saunook and Thunder series). The general soil texture class at this intermediate elevation is loamy or clayey. Soil depth ranges from shallow, mostly on the ridge tops, to very deep at the base of ridges formed by colluvium. Most soils are well drained and only in areas of alluvium near large streams do anaerobic conditions exist where drainage is poor. In areas at elevations above 1280 m (4,200 feet), the soils on uplands generally are brown, fine-loamy or coarse loamy Humic Dystrudepts with a frigid soil temperature regime (Burton, Oconaluftee, and Breakneck series). Soils that formed in colluvium also are Humic Dystrudepts (Balsam and Chiltoskie series). Soils that formed in alluvium vary with stream gradient, energy, and entrenchment into the valley floor. In the upper reaches of watersheds where flood plains are narrow, the soils are Oxyaquic and Fluvaquentic Dystrudepts (Dellwood, Reddies, and Cullowhee series). In the lower and broader river valleys, Udipsamments (Biltmore series) and coarse-loamy Dystrudepts (Rosman series) are in areas closest to rivers and streams on flood plains. Humaquepts (Ela, Nikwasi, and Toxaway series) are in low-lying, frequently flooded or ponded areas. Ultisols are most common on the more stable stream terraces. Fine-loamy Aquic and Typic Hapludults (Dillard and Statler series) are on low terraces, and fine Typic Hapludults (Braddock and Unison series) are on high terraces.

### Southern Appalachian Ridges and Valleys

The soils are mainly Udults and, to a lesser extent, Udepts. They have a udic soil moisture regime and a thermic or mesic soil temperature regime; are dominantly well drained, strongly acid, and highly leached; and have a clayenriched subsoil. They range from shallow on sandstone and shale ridges to very deep in valleys and on large limestone formations. Paleudults (Decatur, Dewey, Frederick, Fullerton, and Pailo series, commonly cherty) are in the many extensive areas underlain by southwest-to-northeast traversing limestone. Hapludults (Townley and Armuchee series) are dominant in valleys underlain by acid shale. Steep, shallow or moderately deep, shaly and stony Dystrudepts (Weikert, Wallen, Montevallo, and Calvin series) are on the sides of steep ridges. Shallow, shaly Eutrudepts (Bays and Dandridge series) are in areas of the shale formation extending along the eastern side of the area. Eutrudepts (Hamblen, Sullivan, and Pettyjon series) are on narrow bottomland.

### Cumberland Plateau

Most of the soils in the undulating to rolling areas on the Cumberland Plateau are Hapludults. Moderately deep or deep, well drained, loamy Hapludults (Lily, Lonewood, and Hartsells series) formed in sandstone residuum. Shallow, somewhat excessively drained, loamy Dystrudepts (Ramsey series) also formed in sandstone residuum. They are less extensive than the other soils in the undulating to rolling areas on the Cumberland Plateau. Most of the remaining soils in the undulating to rolling areas are deep or very deep, moderately well drained, loamy Hapludults (Clarkrange and Hendon series), which formed in a loamy mantle and sandstone residuum. The dominant soils in hilly to steep areas are Hapludults (Gilpin and Lily series) and Dystrudepts (Petros and Matewan series). They are shallow to moderately deep, well drained or somewhat excessively drained, and loamy and formed in sandstone or shale residuum. The remaining soils on steep slopes generally are deep or very deep, well drained, loamy Hapludults (Bouldin, Grimsley, Jefferson, Pineville, and Shelocta series) and Dystrudepts (Varilla, Highsplint, and Guyandotte series), which formed in gravelly or stony colluvium derived from sandstone or shale or both. Soils on flood plains are of small extent on the Cumberland Plateau and are slightly more extensive in the Cumberland Mountains. Most of these soils are well drained or moderately well drained Dystrudepts (Ealy, Pope, Philo, and Sewanee series) or Eutrudepts (Grigsby, Sensabaugh, and Chagrin series) or poorly drained Endoaquepts (Bonair and Atkins series). They are deep or very deep, are loamy, and formed in alluvium derived from sandstone and shale. Material derived from surface and deep mines is common. Udorthents (Bethesda, Cedarcreek, Fairpoint, and Kaymine series) formed in this material.

## *Warm Continental Mountains Division*

### Adirondack Shield and Northern New England Uplands

Because of the similarities between the two areas, we elected to treat this division as one. The dominant soil orders are Inceptisols and Spodosols. The soils dominantly have a frigid soil temperature regime, an aquic or udic soil moisture regime, and isotic or mixed mineralogy. At elevations above 915 m (3,000 feet) in the Adirondack Mountains, the soil temperature regime is cryic. The soils are shallow to very deep, generally somewhat excessively drained to poorly drained, and loamy. Humaquepts (Burnham series) and Epiaquepts (Monarda series) formed in dense till in depressions on till plains. Haplorthods formed in loamy till on hills, mountains, and plateaus (Berkshire, Lyman, Thorndike, and Tunbridge series) and in dense till on drumlins, hills, and ridges (Becket, Colonel, Dixfield, Howland, Marlow, Peru, and Plaisted series).

## *Prairie Division*

The soils are dominantly Alfisols, Entisols, Inceptisols, or Mollisols. Some Histosols occur on flood plains and in wetlands. The dominant suborders are Udalfs, Aqualfs, and Aquolls. The sandy soils are typically Psamments. The soils dominantly have a mesic soil temperature regime, an aquic or udic soil moisture regime, and mixed or smectitic mineralogy. In central Illinois, the dominant soil orders are Mollisols and Alfisols. Most of the soils are Udolls or Aquolls. They have a mesic soil temperature regime, an aquic or udic soil moisture regime, and dominantly mixed mineralogy; and generally are moderately deep to very deep, poorly drained to moderately well drained, and silty or clayey. Nearly level Endoaquolls (Drummer series) and gently sloping to sloping Argiudolls (Saybrook and Catlin series) formed in loess over loamy till on uplands. Hapludalfs commonly occur along the major stream valleys. They are on the gently sloping to moderately sloping uplands (Birkbeck and Mayville series) or on the steep or very steep valley bluffs (Strawn series). Nearly level Endoaquolls (Ashkum, Bryce, and Drummer series) are on broad flats and in shallow depressions. Moderately well drained Argiudolls (Graymont and Varna series) formed in loess and loamy till on gently sloping to sloping uplands. In areas of the more clayey till, somewhat poorly drained Argiudolls (Clarence, Elliott, and Swygert series) are more prevalent. Hapludalfs (Kidami and Ozaukee series) commonly occur on gently sloping to moderately sloping uplands along major stream valleys. They also occur on many of the more sloping glacial moraines. Moderately well drained Eutrudepts (Chatsworth series) generally are in the steeper areas. Haplosaprists (Houghton and Lena series) are common in wet, closed depressions. Loamy, moderately well drained and well drained Argiudolls (Proctor and Warsaw series) and Hapludalfs (Camden and Fox series) are on outwash plains or broad stream terraces underlain by sand and gravel. Somewhat poorly drained Argiudolls (Martinton series) and poorly drained Endoaquolls (Milford series) commonly are on broad glacial lake plains. Cumulic Endoaquolls (Sawmill series) and Cumulic Hapludolls (Lawson and Huntsville series) formed in alluvium on nearly level, broad flood plains and in the smaller upland drainage ways.

## *Subtropical Division*

### Southern Piedmont

The dominant soil orders are Ultisols, Inceptisols, and Alfisols. The soils have a thermic soil temperature regime, a udic soil moisture regime, and kaolinitic or mixed mineralogy. They are shallow to very deep, generally well drained, and loamy or clayey in texture. Hapludalfs (Enon and Wilkes series), Hapludults (Badin, Nason, and Tatum series), and Kanhapludults (Appling, Cecil, Georgeville, Herndon, Madison, Pacolet, and Wedowee series) formed in residuum on hills and ridges. Dystrudepts (Chewacla series) formed in alluvium on flood plains. Udults in the Rhodic subgroup (Davidson, Hiwassee, and Lloyd series) formed in old alluvium on stream terraces or in residuum derived from mafic rocks.

### Upper Coastal Plain

Dominant soil orders are Ultisols, Entisols, and Inceptisols. The soils dominantly have a thermic soil temperature regime, a udic or aquic soil moisture regime, and siliceous or kaolinitic mineralogy. They generally are very deep, somewhat excessively drained to poorly drained, and loamy. Hapludults formed in marine sediments (Luverne and Sweatman series) and mixed marine sediments and alluvium (Smithdale series) on hills and ridges. Kandiudults formed in marine sediments (Dothan, Fuquay, Norfolk, and Orangeburg series) and mixed marine and fluvial sediments (Troup series) on hills and ridges. Fragiudults (Ora and Savannah series) and aleudults (Ruston series) formed in mixed marine and fluvial sediments on uplands and stream terraces. Fluvaquents (Bibb series) and Endoaquepts (Mantachie series) formed in alluvium on flood plains. Quartzipsamments (Lakeland series) formed in sandy eolian or marine material on uplands. Paleaquults (Rains series) formed in marine and fluvial sediments on terraces.

### Lower Coastal Plain

Soils are dominantly Alfisols, Entisols, and Ultisols, but Histosols and Spodosols are not uncommon. The soils typically formed in alluvium on flood plains, in depressions, and on terraces. They dominantly have a thermic soil temperature regime, an aquic or udic soil moisture regime, and siliceous, mixed, or smectitic mineralogy. The soils of the Lower Coastal Plain are made up predominantly of Spodisols (Harris 2001). Spodisols can develop under excessively to poorly drained conditions and are commonly associated with widely fluctuating water tables within 2 m (6.5 feet) of the soil surface. Although edaphic conditions associated with Spodisols are rather specific, vegetation is less definitive because a variety of plant species assemblages are found to occur over Spodisols. However, acidifiying trees and shrubs (heaths, conifers) are commonly associated with Spodisols (Dalsgaard 1990). Soil depth to the saturated zone varies seasonally throughout the Lower Coastal Plain, and particularly in shallow soils, can be underwater during certain times of the year.

Soil texture is predominately sandy-loamy to coarse-loamy materials under humid and perhumid climates. Soil depth can vary seasonally as water tables fluctuate but is generally around 2 m deep. However, during certain times of the year local flooding is common as the water table rises. Soils are excessively to poorly drained. Seasonal flooding occurs in depressions created as karst features of the landscape underlain by deep limestone substrate beneath the saturated (aquiclude) zone. These ephemeral ponds can serve as discharge zones during periods of low water table. Immediately along the coast (The Atlantic Coast Flatwoods), the dominant soil orders are Alfisols and Entisols. The soils are characterized by restricted drainage, a thermic soil temperature regime, and an aquic soil moisture regime. The soils in the northern part of the area dominantly have mixed mineralogy, and those in the southern part dominantly have mixed clay and siliceous sand mineralogy. Very deep, loamy to clayey Endoaquults (Tomotley, Yeopim, Yemassee, and Wahee series), Umbraquults (Cape Fear and Portsmouth series), Endoaqualfs (Argent and Yonges series), and Albaqualfs (Meggett series) are extensive. Hapludults (Bertie and Tetotum series) are in the higher areas where drainage is better but is somewhat restricted. Other important soils are Alaquods (Leon and Lynn Haven series) and Psamments (Wando, Newhan, Corolla, and Fripp series). Histosols (Pungo and Belhaven series) are in large areas in North Carolina and Virginia, in the Great Dismal Swamp and in broad upland wetlands known as poquosins. Aquents (Bohicket and Capers series) are extensive throughout the brackish tidal marshes protected by the barrier islands and sea islands.

## *Savanna Division*

The dominant soil orders are Entisols and Histosols. The soils dominantly have a hyperthermic soil temperature regime, an aquic or udic soil moisture regime, and carbonatic mineralogy. They are very shallow to very deep, generally moderately well drained to very poorly drained, and loamy or sandy. Udorthents (Krome series) formed in residuum on flats. Fluvaquents (Biscayne and Perrine series) and Psammaquents (Hallandale series) formed in marine sediments on flats and in depressions and sloughs. Haplosaprists (Pahokee and Terra Ceia series) formed in organic deposits in marshes.

## *Subtropical Mountains Division*

### Ouachita Mountains

The dominant soil orders are Ultisols and Inceptisols. These soils dominantly have a thermic soil temperature regime, a udic soil moisture regime, and mixed or siliceous mineralogy. They are shallow to very deep, generally somewhat excessively drained to somewhat poorly drained, and loamy. Dystrudepts (Bismarck and Clebit series) and Hapludalfs (Clearview series) formed in residuum on hills and mountains. Hapludults formed in colluvium (Zafra series), colluvium over residuum (Bengal series), and residuum (Carnasaw, Pirum, Sherless, Sherwood, Stapp, and Townley series) on hills, mountains, and plateaus. Udifluvents (Ceda series) formed in alluvium on flood plains.

### Ozark Highlands

Most of the soils in this area are Alfisols or Ultisols. They formed in material weathered from cherty limestone. Most areas in the northern and eastern parts are partly covered with a thin mantle of loess. Physical and chemical weathering has caused the cherty limestone to disintegrate into its least soluble components, which are chert and clay. The chert remains in the form of angular fragments or wavy horizon beds interstratified with layers of clay. Downslope movement by gravitational creep and overland waterflow has altered the cherty material in the upper part of some soils. In general, the soils are shallow to very deep, moderately well drained to excessively drained, and medium textured to fine textured. The soil temperature regime is mesic bordering on thermic, the soil moisture regime is udic, and mineralogy is mixed or siliceous. Many of the soils on nearly level to moderately sloping upland divides are Paleudalfs (Gravois, Gepp, and Peridge series), Fragiudalfs (Union, Viraton, and Wilderness series), or Fragiudults (Captina, Scholten, and Tonti series). Many of the soils on moderately sloping to steep side slopes in the uplands are Hapludalfs (Gatewood, Mano, Ocie, and Wrengart series), Hapludults (Bendavis, Bender, and Lily series), Paleudalfs (Alred, Goss, and Rueter series), or Paleudults (Clarksville, Coulstone, Noark, and Poynor series). Many of the soils in glades are Mollisols (Gasconade, Knobby, and Moko series). Many of the soils on terraces and the adjacent flood plains are Hapludalfs (Razort, Secesh, and Waben series), Hapludolls (Cedargap, Dameron, and Sturkie series), Paleudalfs (Britwater and Pomme series), Eutrudepts (Gladden and Jamesfin series), or Udifluvents (Midco and Relfe) series.

### Eastern and Western Arkansas Valley and Ridges

In the eastern portion, the dominant soil orders are Ultisols. In the west, they are dominated by Udalfs or Udepts. Both areas have a thermic soil temperature regime, a udic soil moisture regime, and mixed or siliceous mineralogy. In the east, soils are stony or non-stony and are medium textured. Well drained, shallow and moderately deep Hapludults (Mountainburg and Linker series) formed on ridgetops, benches, and the upper slopes. Well drained, deep Hapludults (Enders series) and Paleudults (Nella series) formed on the middle and lower slopes and in concave areas between ledges. Fragiudults (Leadvale, Taft, and Cane series) formed in valleys. Udifluvents (Roxana series), Udipsamments (Crevasse series), Haplaquolls (Roellen series), and Hapludalfs (Gallion series) are minor soils along the Arkansas River, and Dystrochrepts (Barling series) and Hapludults (Spadra and Pickwick series) are minor soils on terraces along the smaller streams. In the west, moderately deep, gently sloping to steep Hapludalfs (Clearview series) formed on ridgetops, shoulder slopes, and side slopes. Very deep, gently sloping to sloping Paleudalfs (Stigler series) formed on the side slopes of valleys. Deep, gently sloping to steep Hapludalfs (Endsaw series) formed on side slopes and footslopes. Shallow, sloping to steep Dystrudepts (Clebit and Hector series) formed on narrow ridgetops and the upper shoulder slopes. Very deep, gently sloping to steep Paleudalfs (Larton and Porum series) and Hapludalfs (Karma series) are minor soils on terraces along streams. Nearly level to sloping Hapludolls (Verdigris series) and Udifluvents (Severn series) are minor soils along flood plains throughout the area.

### Boston Mountains

The dominant soil orders are Ultisols and Inceptisols. These soils dominantly have a thermic soil temperature regime, a udic soil moisture regime, and mixed or siliceous mineralogy. They are shallow to very deep, generally well drained, and loamy. Hapludults (Enders, Linker, Mountainburg, and Steprock series) and Dystrudepts (Hector series) formed in residuum on hills, plateaus, and mountains. Paleudults formed in alluvium or colluvium over residuum (Allen and Nella series) and alluvium or colluvium (Leesburg series) on hills and terraces.

## *Hot Continental Division*

### Interior Plains and Lowlands

Soils are chiefly Inceptisols, Ultisols, and Alfisols, rich in humus and moderately leached, with a distinct light-colored leached zone under the dark upper layer. The Ultisols have a low supply of bases and a horizon in which clay has accumulated.

The soils typically have a frigid soil temperature regime, an aquic or udic soil moisture regime, and mixed mineralogy. They generally are very deep, well drained to very poorly drained, and loamy.

### Southern New England Coasts

The soils are dominantly Entisols or Spodosols. They commonly have a fragipan. Alfisols are less extensive. They formed in limy parent material and have a fragipan. The dominant suborders are Ochrepts and Orthods at the higher elevations and Aqualfs, Aquepts, and Histosols on lowlands and in depressions. The soils on flood plains (Fluvents) are of small extent but are important for many uses. The soils dominantly have a frigid or mesic soil temperature regime, a udic soil moisture regime, and mixed mineralogy. The major soil resource concerns are water erosion, wetness, and maintenance of organic matter content and productivity of the soils. Wind erosion is a hazard in some of the northern parts where the lighter textured soils occur. Protecting wildlife habitat and preserving the quality of surface water and ground water are additional concerns in many parts of this area.

### Northern Piedmont

The dominant soil orders are Alfisols, Inceptisols, and Ultisols. The soils dominantly have a mesic soil temperature regime, a udic soil moisture regime, and mixed, micaceous, or kaolinitic mineralogy. They are moderately deep to very deep, moderately well drained to somewhat excessively drained, and loamy or loamy-skeletal. Hapludalfs (Duffield, Neshaminy, and Penn series) and Dystrudepts (Manor, Parker, and Mt. Airy series) formed in residuum on hills. Fragiudalfs (Reedington series) formed in residuum on footslopes and in drainageways. Hapludults (Chester, Elioak, Gladstone, and Glenelg series) and Kanhapludults (Hayesville series) formed in residuum on hills, upland divides, and ridges. Fragiudults (Glenville series) formed in colluvium or residuum on hills. The far northeastern extent of the northern Piedmont was affected by early periods of glaciation, and many soils formed in very deep, highly weathered till; the dominant soils are Hapludalfs (Washington and Bartley series) and Fragiudults (Annandale and Califon series).

## *Warm Continental Division*

### Northern Great Lakes

The soils are primarily Histosols, Alfisols, Spodosols, and Entisols. Some areas also have a significant acreage of Mollisols or Inceptisols. Almost all of the soils have a frigid soil temperature regime, and all have an aquic or udic soil moisture regime. Soils with a mesic soil temperature regime are in many areas in the southern part. Mineralogy is dominantly mixed, but it is isotic in some areas.

### Glaciated Allegheny Plateau and Catskill Mountains

The dominant soil order is Inceptisols. The soils dominantly have a mesic soil temperature regime, an aquic or udic soil moisture regime, and mixed mineralogy. They are shallow to very deep, well drained to very poorly drained, and loamy or loamy-skeletal. Dystrudepts (Arnot, Lordstown, and Oquaga series) formed in till on hills and dissected plateaus. Fragiudepts (Bath, Lackawanna, Mardin, Swartswood, Wellsboro, and Wurtsboro series) and Fragiaquepts (Chippewa, Morris, Norwich, and Volusia series) formed in till (dense till in some areas) on hills and till plains.

### Northern New England Coasts

The dominant soil orders are Inceptisols and Spodosols. The soils dominantly have a frigid soil temperature regime, an aquic or udic soil moisture regime, and isotic, illitic, or mixed mineralogy. They are shallow to very deep, generally excessively drained to poorly drained, and loamy or sandy. Eutrudepts (Buxton series) and Epiaquepts (Scantic series) formed in glaciomarine or glaciolacustrine deposits on coastal lowlands and in valleys. Dystrudepts formed in till on till plains and moraines (Lanesboro, Shelburne, and Colrain series) and on hills and ridges (Taconic series). Haplorthods formed in glaciofluvial deposits on outwash plains and eskers (Adams and Colton series); in till on till plains, ridges, and moraines (Bangor, Berkshire, Dixmont, Hermon, Lyman, Monadnock, and Tunbridge series); and in dense till on drumlins and uplands (Marlow and Peru series).

# Conclusions

The Eastern United States encompasses significant variation in biophysical features that constrain management practices available to reduce fuel loads. For example, in areas with generally flat topography (such as the Coastal Plain), mechanical techniques are easy to implement. By contrast, fuel management options are more limited in steeper terrain where mechanical techniques are difficult (or cost prohibitive) to implement. Variation in climate influences species composition and fuel load, and also determines site access (wetter areas may be less accessible for mechanical techniques) and unsuitable moisture levels (either too wet or too dry) for prescribed fire. Variation in soils and geology determine the sensitivity of soils to compaction or erosion after mechanical treatments, and sustainability of soil productivity after prescribed fire. In summary, variation in geomorphology, topography, soils, and climate in the Eastern United States requires understanding interactions among fuel management treatments and geographic landscapes, and matching treatment prescriptions with physical conditions.

# Acknowledgments

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Cumulative Watershed Effects of Fuel Management in the Eastern United States

CHAPTER 4.

# The Human Context: Land Ownership, Resource Uses, and Social Dynamics

**David N. Wear**

The forests and grasslands of the Eastern United States have been subject to more than two centuries of episodic change, generally characterized by forest clearing, agricultural use, abandonment, reforestation, and recovery. Today, rapid colonization of forests and other rural lands by people, the spread of many floral and faunal non-native invasive species and, in some places, structural changes in forest product companies continue to alter forests. Historical legacies and ongoing disturbances define a complex landscape in the Eastern United States where no land is without substantial human influence. Opportunities for and the practice of forest management and fuels treatments are heavily influenced by this human history and by the human context of forest settings.

This chapter describes the history of eastern forest conditions and uses, and discusses the implications these dynamics hold for future uses, management, and conditions. In particular, I examine time trends in forest area, biomass, and ownership, juxtaposed with changes in human populations and uses of these vast forest resources. The changing human-forest interface holds implications for future forest uses, including opportunities for fuel treatments and other types of forest management, the availability of timber products and ecosystem services, and the values at risk from wildfire and other disturbances.

## Conditions and Trends in Eastern Forests

One way to gauge change in forests is to examine how the area of forest cover has changed over time. Surveys of forest conditions conducted by the Forest Inventory and Analysis Program of the Forest Service, U.S. Department of Agriculture, since 1938 provide a basis for a systematic analysis of forest conditions including forest area. In addition, work by Kellogg (1909) provides estimates of forest area in the United States for 1907 and at the time of European settlement (~1630). These data are compiled for the country as a whole in a series of publications, the latest from Smith and others (2003), that provide the majority of forest data discussed in this chapter.

At the time of European settlement, forest area in the Eastern United States exceeded 650 million acres, with roughly 298 million acres in the Northeastern and North Central States and 354 million acres in the Southeastern and South Central States (fig. 1). By 1907, eastern forest area had fallen by about 43 percent to roughly 374 million acres

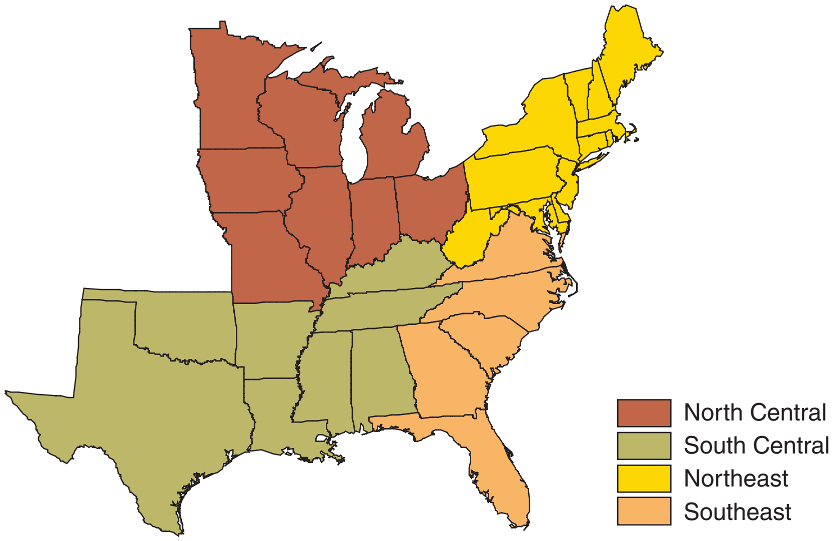


**Figure 1.** Four divisions of the Eastern United States.

overall with 139 million acres remaining in the North (a decline of about 53 percent) and 236 million acres in the South (a decline of about 33 percent). The spatial pattern of eastern deforestation was highly variable through the 19th century (fig. 2). Ohio, Indiana, and Illinois had lost at least 80 percent of their forest area by 1907, compared to <30 percent for Maine, Florida, Arkansas, and Oklahoma.

Changes in forest area from 1907 to 2002 reveal different patterns (fig. 3). In the 20th century, eastern forest losses were concentrated in Florida and west of the Mississippi River (Florida and Texas had the largest proportions of forest loss). In contrast, the more central States from Illinois to New York saw large proportional increases in forest cover, with moderate gains occurring in a few Southern States (Alabama, Georgia, Virginia, and Kentucky). Through much of the Eastern United States, extensive deforestation in

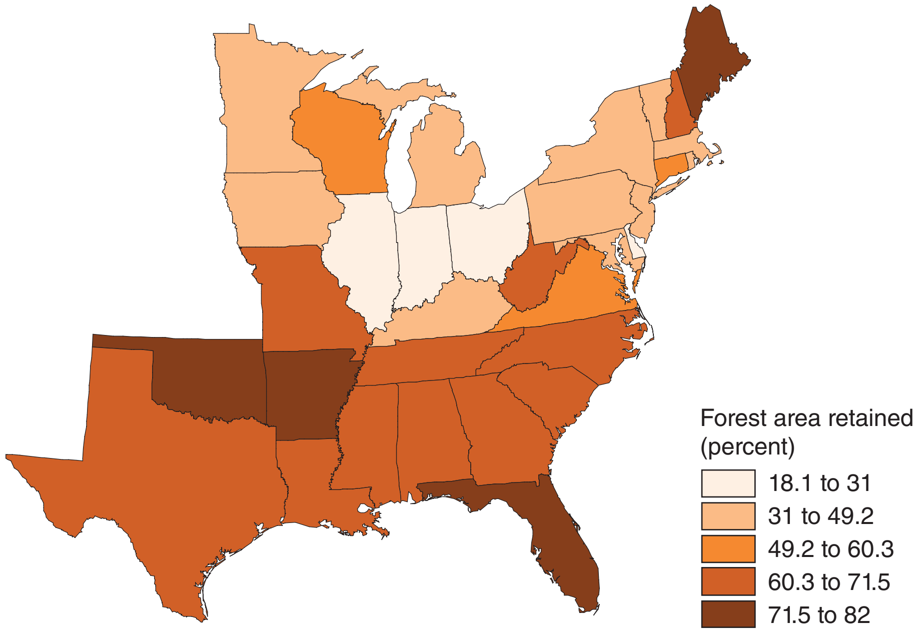


**Figure 2.** Eastern forest area in 1907 compared to 1630, by State (Smith and others 2003, Kellogg 1909).

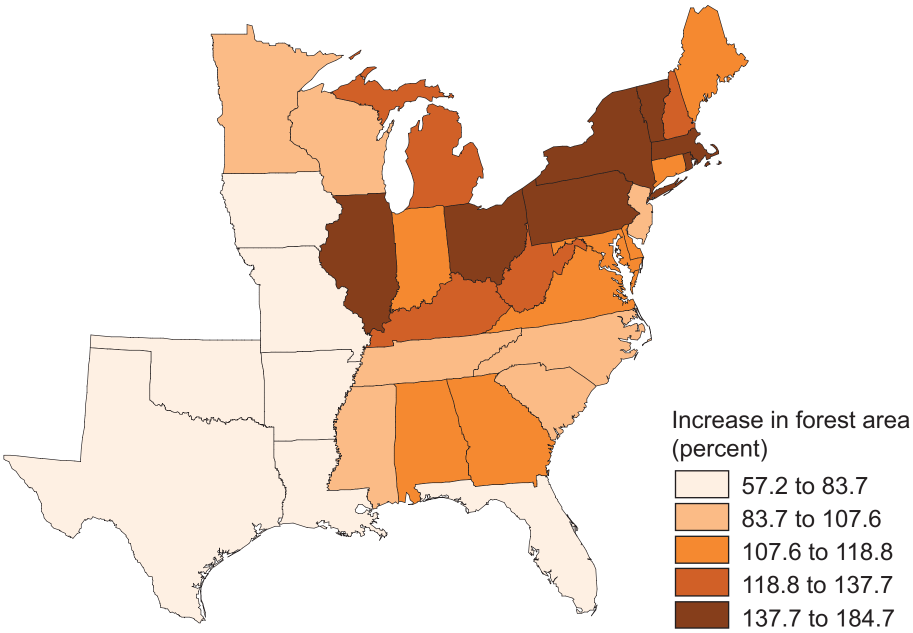


**Figure 3.** Eastern forest area in 2002 compared to 1907, by State (Smith and others 2003, Kellogg 1909).

the 19th century was followed by some forest area gains in the 20th century. In 2002, forest cover was 43 percent lower than presettlement levels in the North and 39 percent less in the South forest.

Net changes in forest area from 1630 to 2002 were highly variable among Eastern States (fig. 4). In 2002, 5 States (Maine, New Hampshire, Vermont, West Virginia, and Alabama) had more than 75 percent of their presettlement forest area; and 20 States—including 11 of the 13 Southern States—retained 50 to 75 percent. Less than 50 percent of their presettlement forest area was retained by the remaining 7 States, 3 of which are small eastern-seaboard States dominated by urban uses (Maryland, Delaware, and New Jersey). The lowest proportions of residual forests are in Texas and the territory stretching from Iowa eastward through Illinois, Indiana, and into Ohio—in these agricultural areas, residual forests are 23 to 41 percent of original forest area. From 1907 to 2002, 23 of the 33 Eastern States experienced a recovery of some forested area that had been lost before 1907. States with the greatest proportional recovery of forest area were mostly in New England (fig. 4).

The net loss of forest area understates the overall impact that European settlement and land exploitation has had on forest conditions. Even in areas where forest use was maintained over time, timber harvesting altered conditions substantially. Nearly every existing forested acre in the United States has been harvested at least once. So, in most eastern forest landscapes, biomass has been removed at least once since European settlement; in many places, several harvests have occurred. After harvesting, especially in the late 19th and early 20th centuries, a large share of cleared land had been briefly farmed before the economics of poor soils returned forest cover through land abandonment and natural regeneration. The second growth forests that remain reflect a different productivity, species composition, and structure than existed in presettlement forests.

The extent of harvest disturbances and recovery in eastern forests can be deduced from trends in tree biomass contained in these forests over time. Measures of biomass are available for only the second half of the 20th century, but they reflect the rapid recolonization and growth of cutover forests, a large portion of which was returned to forest cover after a brief agricultural exploitation between the 1920s and

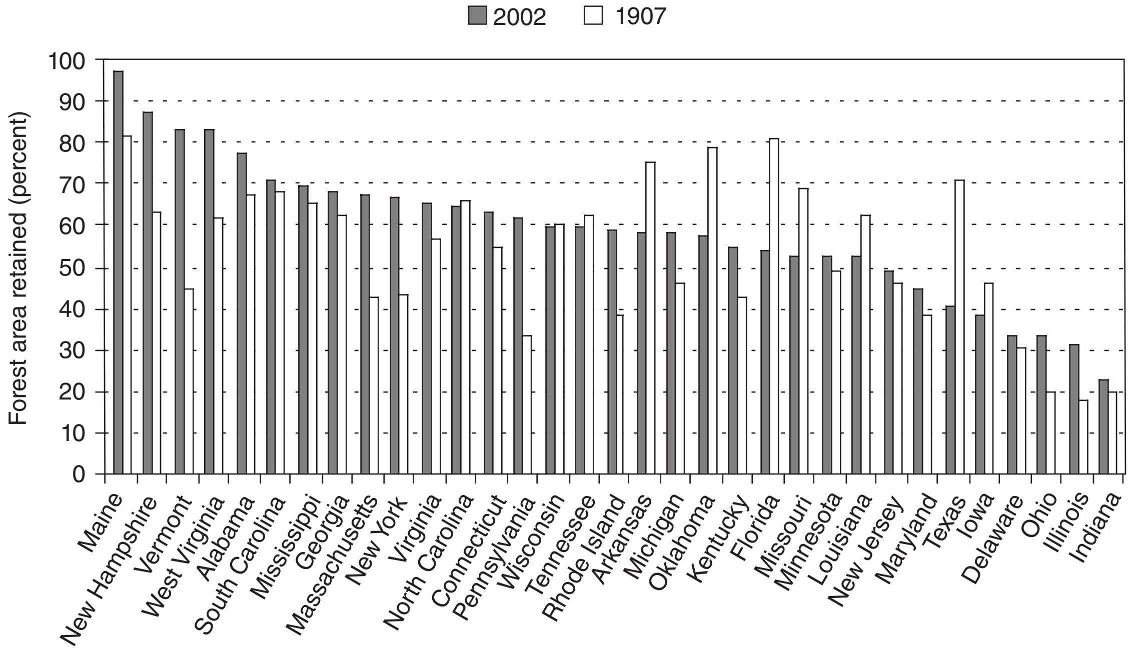


**Figure 4.** Eastern forest area in 1907 and 2002 compared to 1630, by State (Smith and others 2003, Kellogg 1909).

the Great Depression. Figures 5 and 6 show the evolution of standing biomass from the 1950s through 2002 for northern and southern forests (fig. 5) and for hardwoods and softwoods in the northern and southern forests (fig. 6). During this period, forest area was relatively stable but tree biomass (as estimated by growing stock inventories in Forest Service inventories) nearly doubled from 252 to 486 billion cubic feet. The rate of increase has slowed since the 1970s, indicating perhaps an approach to a capacity defined by soil conditions and ongoing human dynamics, including timber harvesting, and movement into and out of forest cover. However, the average biomass contained on eastern forest sites increased throughout the last half of the 20th century.

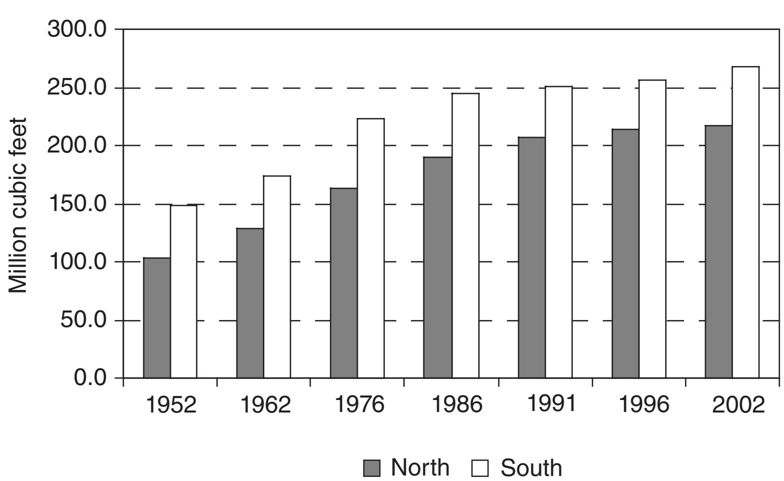


**Figure 5.** Accumulated tree biomass measured as growing stock inventory, 1952 to 2002, in the Eastern United States (Smith and others 2003).

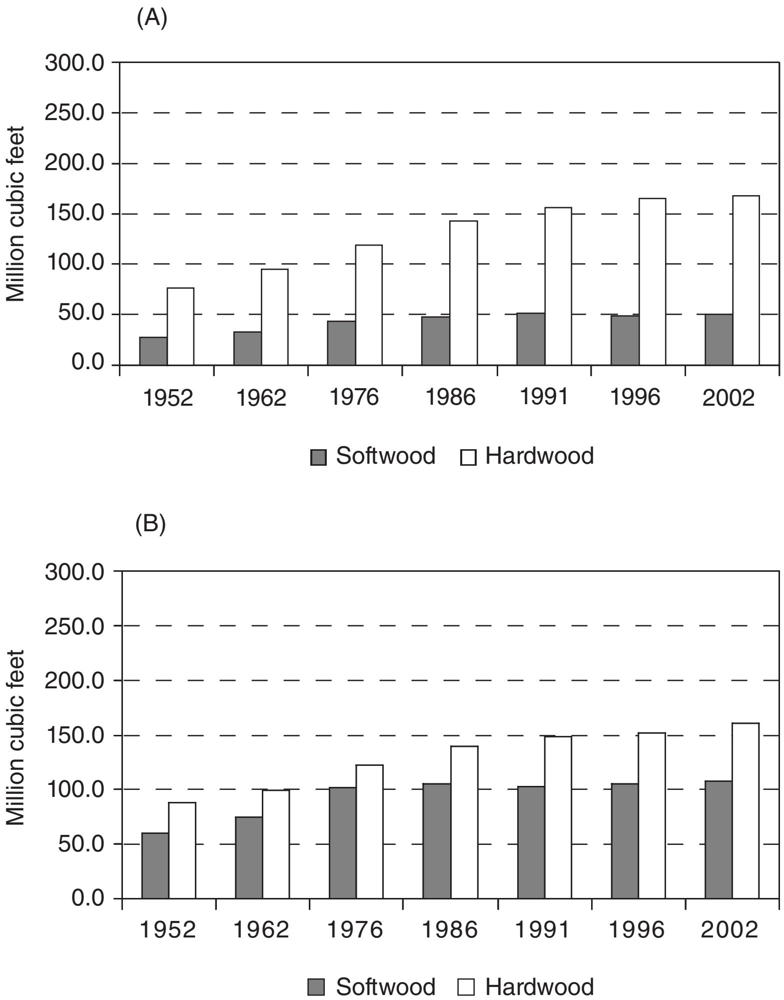


**Figure 6.** Accumulated tree biomass measured as growing stock inventory, 1952 to 2002, of softwoods and hardwoods (A) for the North and (B) for the South (Smith and others 2003).

# Forest Ownership

Unlike their western counterparts, eastern forests are dominated by private ownership (fig. 7). Roughly 85 percent of the eastern forests were privately owned in 2002. Of the 15 percent that was in public management, 40 percent was in national forests; 47 percent was owned by States, counties, or municipalities; and the remaining 13 percent was in some other type of Federal ownership (predominantly military facilities). Ownership patterns vary somewhat between the South and North (fig. 7). The North has a higher proportion of public ownership (20 percent versus 14 percent), whereas the South has a higher proportion of private ownership (86 percent versus 80 percent).

Owner objectives and management styles differ substantially between public and private owners but also vary within the private ownership group. Forest Service surveys have tracked a private owner typology over time that, at its coarse grain, splits forest industry (defined as companies that hold both forest land and wood products processing facilities) from all other private owners. Forest industry owners have differed from other types of owners in that they generally have approached forest lands with a timber-profit motive and have adopted a distinct production style of forest management (Newman and Wear 1993). The result has been a higher level of forest investment and outputs with implications for forest structure—these lands were more heavily dominated by pine plantations, retained lower levels of standing biomass overall, and were generally younger than nonindustrial private forests. Forest industry lands have also traditionally represented some of the largest contiguous blocks of forest land in the Eastern United

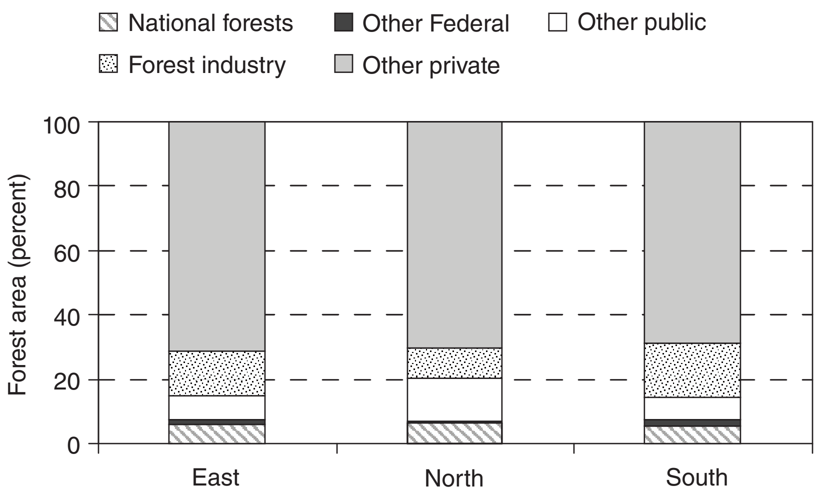


**Figure 7.** Distribution of forest area by broad ownership classes, 2002, for the Eastern United States, the North, and the South (Smith and others 2003).

States with associated values for protecting certain ecosystem services. Other, nonindustrial, private owners have a notoriously varied suite of motivations for owning forest land (Butler and Leatherberry 2004), with a perhaps less predictable management style and a more variable outcome.

During the 20th century, forest industry established and managed some of the most productive forest lands in the Eastern United States and was a fairly stable component of the forest products sector, especially in the Southeastern States (fig. 7). However, commencing in the late 1990s and accelerating since 2005, most large companies have divested their forest holdings (Clutter and others 2005). Figure 8 shows the beginning of this trend, with more recent estimates indicating a loss of about 80 percent since the late 1990s. These changes, driven by a variety of economic factors, have a new set of implications for forest structure. Many of the industry's vast holdings have been subdivided in the process of being sold, resulting in a more fractured ownership pattern. What is more, a variety of forest conditions—including those on environmentally sensitive land—had been bundled with production on industry tracts; these components are readily split apart as the land is sold in pieces, possibly removing some de facto protection. Where other uses compete for forests, the land has been sold for development.

Productive industry timberlands have largely been sold to private timber investors organized by Timber Investment Management Organizations (also known as TIMOs), which have a strong focus on a profit-maximizing forest management—not unlike the forest industry. This arrangement provides substantial capital for ongoing investment in the face of favorable markets, creating a state of investment inertia that currently keeps much land in forest production but that also has the potential for rapid land-use

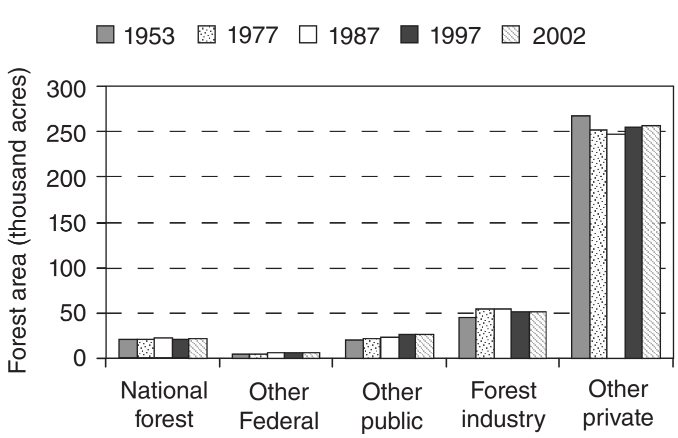


**Figure 8.** Area of timberland by broad ownership classes, 1952 to 2002 for the Eastern United States (Smith and others 2003).

switching when markets change. A general outcome of this new landownership arrangement then is a higher liquidity of land in the face of changing economic circumstances.

Another way of explaining this elevated liquidity is to contrast investor objectives and options. When companies owned land largely as a buffer against future supply shortages, they had a strong incentive to retain land over a long time frame even in the face of adverse short-term market conditions. Following divestiture, few options remained for providing this kind of timber supply insurance. Individuals are motivated to invest in timberland for returns, returns that are perceived to be countercyclical to equity markets. However, timberland is only one of many alternative investment instruments available for providing countercyclical returns, and new ownership arrangements may be less conducive for the long-term investment needed for effective forest land management.

This large-scale change in the ownership of the Nation's most productive timberlands will undoubtedly have an effect on landscape structure in some parts of the Eastern United States. TIMO holdings are often bundled for investors as closed end funds, which must be sold at the end of a fixed term. With 5- to 15-year terms, these investment vehicles imply a relatively rapid turnover of land ownership over time. What is more, each transaction offers an opportunity to split parcels and sell portions for different uses, thereby encouraging an ongoing fragmentation of forested lands with implications for the ecosystem services and management potential of remaining forest lands.

Federal forest lands also occupy a distinct portion of the landscape in the Eastern United States and provide an important suite of forest benefits. The eastern national forests were authorized by the 1911 Weeks Act and acquired through land purchases from private owners. The national forests acquired land piecemeal, mainly from 1911 until the end of the Great Depression, from cutover and unproductive lands in relatively remote areas where the value of land for any other use was very low. Referred to as the "lands nobody wanted" by Shands and Healy (1977), these forests were concentrated in mountainous areas (Ozark, Ouachita, Allegheny, and Appalachian ranges), and not in close proximity to population centers. As a result, of the way these lands were accumulated, eastern national forests are less contiguous than their western counterparts and are often interspersed with private forest holdings, where private and public good values commingle and define a challenging management context.

Taken together, these forest ownership dynamics yield several important implications. Public lands tend to be concentrated in areas that are the most remote and rugged and the least productive, and are not tightly consolidated. As amenity values increase in these areas, the value of private in-holdings and adjacent private lands also increases, and subsequent development can compromise the provision of several public values for which the public lands are especially important. Timber management and production are increasingly concentrated on productive rural lands that compete with agricultural uses of land. Forest industry set the stage for an increased concentration of production forestry on a smaller land base; with a new ownership structure, these lands are increasingly guided by shorter term market signals.

# Social Context of Forests

Humans alter the structure and extent of forests, directly through the uses to which they allocate land and indirectly by changing atmospheric and hydrologic systems and introducing nonnative (and often invasive) flora and fauna. A simple index of the pressure that people place on natural systems is the areal density of human populations. In the 2000 census, the density of counties in the Eastern United States stood at about 244 people per square mile and ranged from less than 5 in Oklahoma to more than 55,000 in metropolitan New York. From 1970 to 2000, the average density grew by about 16 percent and the total population grew from 208 million to 274 million.

Of course, this growth in population was not spread evenly across the landscape. In 2000 (fig. 9), 46 percent of counties was in what we have labeled a rural category (0 to 50 people per square mile), 32 percent in a transitional category (51 to 150), 10 percent

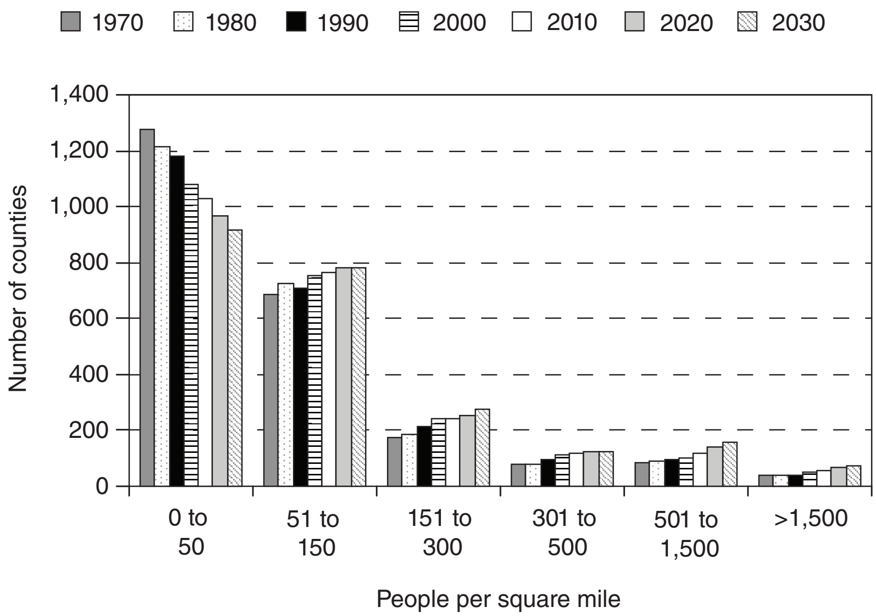


**Figure 9.** Distribution of counties by population density classes, 1970 to 2030, based on U.S. Census Bureau historical data and Woods and Poole Econometrics projections.

in a suburban category (301 to 500), and 12 percent in an urban (500 to 1500) or a high-density urban category (>1500). This distribution has changed as population has grown in the Eastern United States. The percentage of counties in the most rural category has declined substantially since 1970 (from 55 percent in 1970 to 46 percent of counties in 2000). Over the same period, the number of counties in transitional, suburban, and urban classes has increased. Figure 9 also shows that these patterns are expected to continue well into the future (to 2030) based on a set of county-level population forecasts for the United States. That is, we expect a continued movement from rural conditions toward transitional and urban conditions.

Patterns of population change differ as well. Although eastern populations have grown steadily, some areas experienced sizable depopulation from 1970 to 2000 (fig. 10). Among the areas with the largest population losses are the agricultural areas of

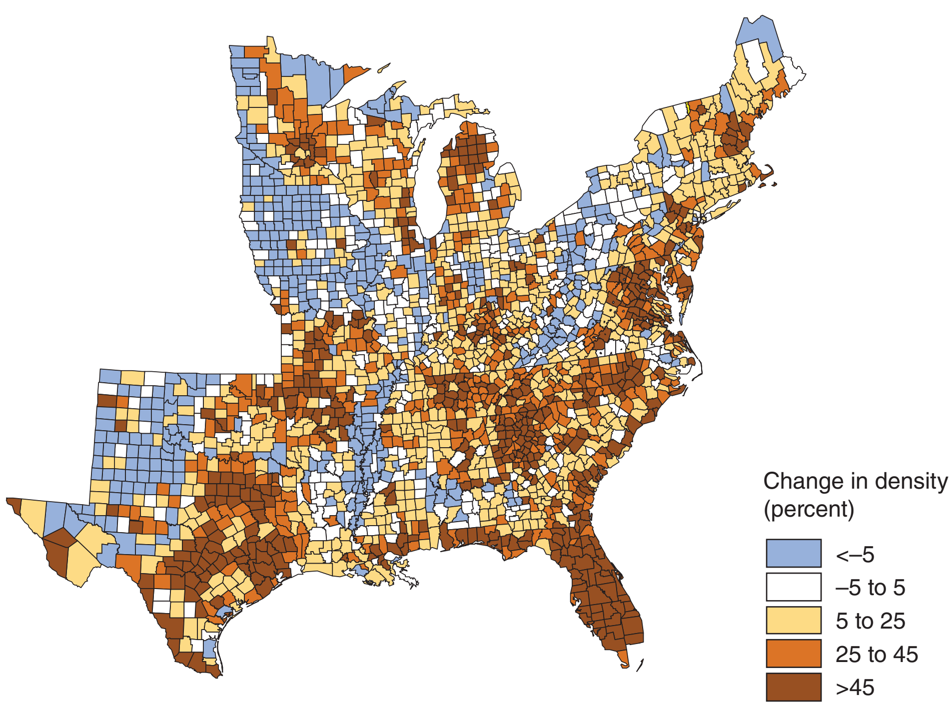


**Figure 10.** Change in population density by county in the Eastern United States, 1970 to 2000, based on U.S. Census Bureau data.

southern Minnesota, Iowa, and Illinois; the lower Mississippi Alluvial Valley from the confluence with the Ohio River to Louisiana; and the Allegheny Highlands from Kentucky and West Virginia into western New York. Smaller areas experiencing depopulation include an area north of Mobile Bay in Mississippi and Alabama; the northernmost counties in Minnesota, Michigan Upper Peninsula, New York, and Maine; and a grouping of counties in central Ohio.

Population gains were also concentrated, with three large areas experiencing the largest increases from 1970 to 2000: the metropolitan corridor stretching from Boston to Washington; the Piedmont of the Southern Appalachians from Raleigh, NC, to Atlanta; and peninsular Florida. Many moderately large cities have also experienced high rates of population growth, including Dallas, Houston, Detroit, Chicago, Minneapolis, and Nashville.

# Competing Land Uses

Land use patterns reflect the distribution of human populations (such as the density of housing and urban uses) as well as the comparative productivity of land in a variety of rural uses (such as crops). The Economic Research Service of the U.S. Department of Agriculture maintains a consistent time series of State-level land use estimates from 1945 to 2002 in their Major Land Uses series, with the latest report from Lubowski and others (2005). The data on land use changes reported below, which are taken from this series, distinguishes among four major land use groupings: total cropland (including planted and fallow), pasture (land in a grazing use including range), forest land (consistent with the Forest Inventory and Analysis definition), and urban land in densely populated areas. An all-other category includes rural transportation, defense and industrial areas, rural parks, and miscellaneous farm and other special uses.

Land use in the Eastern United States reflects a diversity of these conditions. In 2002, cropland occupied 28 percent of the land base, pasture occupied 17 percent, forests occupied 38 percent, and urban and all other uses occupied 17 percent (fig. 11).

The distribution of land uses varies greatly (fig. 11). For example, crop production is predominant in the North Central States of Iowa, Illinois, and Minnesota—reflecting soil and climatic conditions that favor crop production. In addition, crop production is a dominant land use in the Lower Mississippi Alluvial Valley and Florida. Range and pasture uses are most predominant in the South Central States, especially Texas and Oklahoma. Agricultural uses represent an areal majority of States in the western half of the study area.

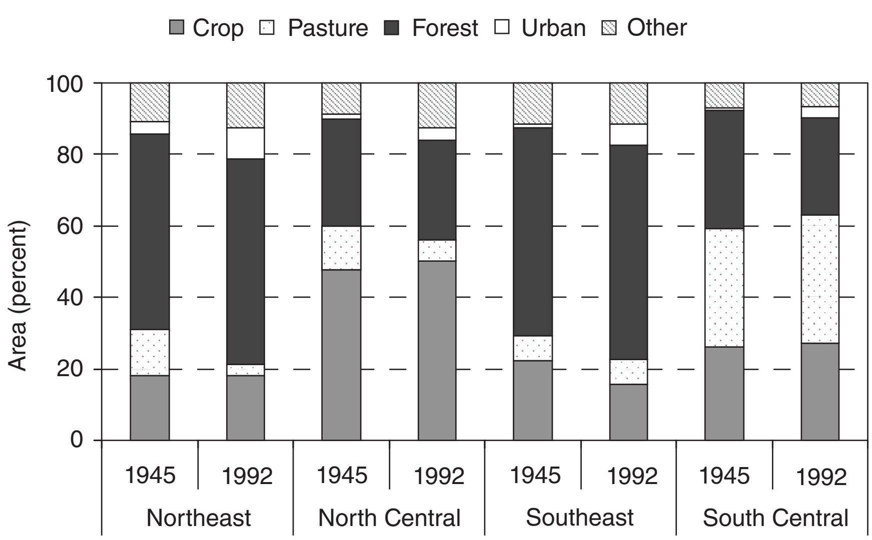


**Figure 11.** Distribution of land area by broad land use classes, 1945 and 2002, for Northeastern, North Central, Southeastern, and South Central States (Lubowski and others 2006).

Farther east, forests tend to dominate rural land uses, with comparable shares of forest land use in the Northeastern and Southeastern States. Urban and other land use (mainly transportation, parks, and rural developed area) generally make up 10 to 24 percent of the eastern landscape (fig. 11).

From 1945 to 1992, the share of land in non-rural uses expanded throughout the Eastern United States, with the greatest increase (from 10 to 17 percent) in the Northeastern States (fig. 11). The portions of States in rural uses shrank over this period and the distribution among rural uses changed as well. In the Northeastern States, pasture uses experienced the biggest losses (from 12 to 3 percent), and the area of cropland and forest remained relatively constant. In the North Central States, forest and pasture uses shrank slightly and cropland stayed constant. Conversely, cropland in the Southeastern States declined from 22 to 14 percent, and pasture and forest area remained relatively constant. The South Central States experienced a loss of forest land, and both pasture and cropland remained relatively constant.

Among the eight States that gained cropland area from 1945 to 2002, six were along the Mississippi River and the other two were Texas and Florida (fig. 12). Florida experienced the greatest gain in cropland area (29 percent). All other Eastern States lost some cropland, with the New England States experiencing the biggest losses (>50 percent). Total cropland was relatively constant across the Eastern United States, so these changes indicate a westward shift in and spatial consolidation of crop production.

The spatial distribution of pasture use also shifted from 1945 to 2002 (fig. 13). Total pasture in the Eastern United States declined slightly over the period (from 19 to 17 percent) but the distribution shifted to the south. Pasture gains were found in only five States: Florida and a four State south-central block composed of Texas, Oklahoma, Arkansas, and Louisiana. As with cropland, Florida experienced the greatest gains in pastureland use. All Northern States experienced substantial reductions in pastureland use.

The pattern of change in urban land use (fig. 14) is quite distinct from the patterns for cropland and pasture. Urban uses grew by at least 72 percent across all Eastern States and more than tripled in more than half of them. Percentage-growth rates for this period were substantially higher in the South than in the North (fig. 14) but the absolute changes in urban area were more evenly distributed between the regions (Northern States had much larger urban area at the beginning of the period). The result is expansion of metropolitan areas into formerly rural lands throughout the Eastern United States, changing the context for rural uses in many areas.

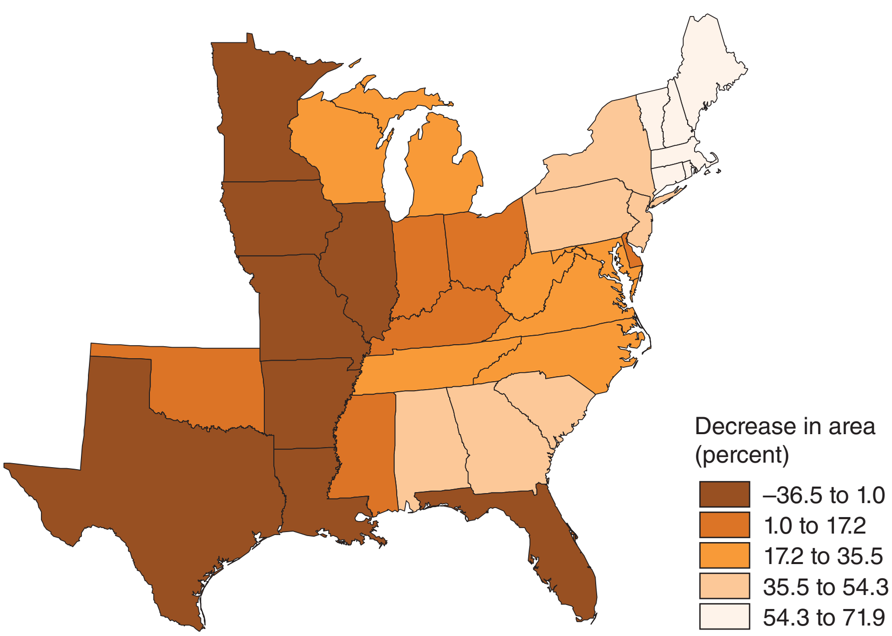


**Figure 12.** Change in cropland area, 1945 to 2002, by State in the Eastern United States (Lubowski and others 2006).

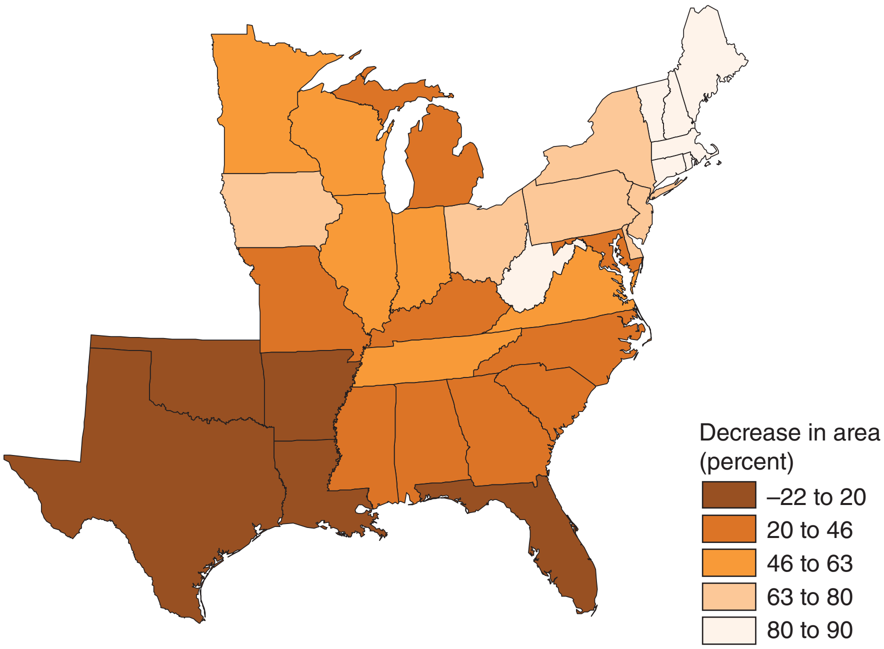


**Figure 13.** Change in pasture area, 1945 to 2002, by State in the Eastern United States (Lubowski and others 2006).

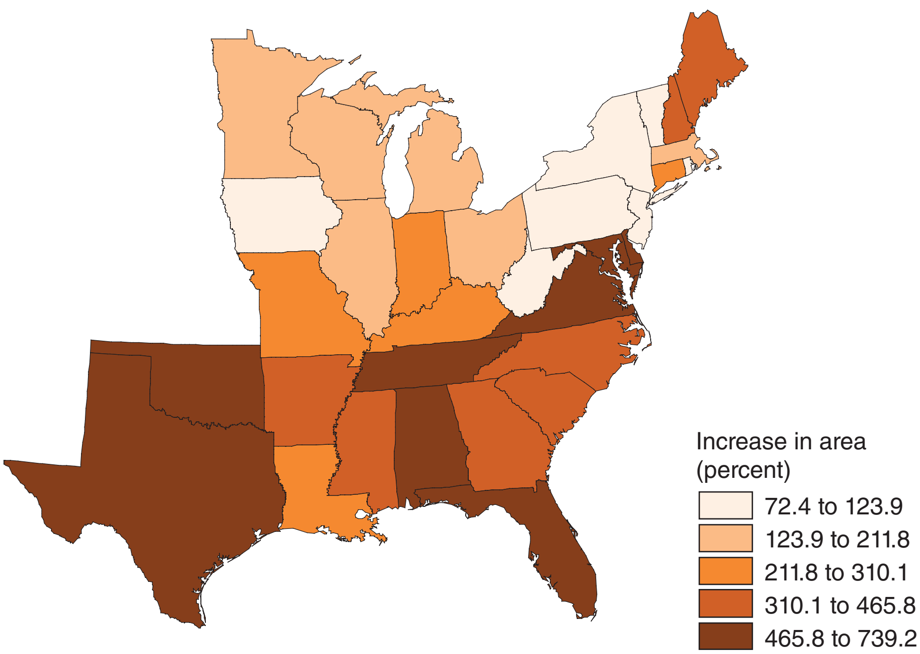


**Figure 14.** Change in urban area, 1945 to 2002, by State in the Eastern United States (Lubowski and others 2006).

# Conclusions

Eastern forests have been subjected to a series of transformative changes since European settlement. Existing forests are generally the product of multiple human-based disturbances including timber harvesting, cultivation or grazing in a previous agricultural use, abandonment, and recolonization by tree species. The restoration of forest cover was especially strong in the 20th century as much agricultural land was abandoned beginning in the early 1900s. With passage of the Weeks Act national forests were established in a few areas, generally in remote places where land was less valuable for any other kind of use.

The dynamics most relevant to current forest structure and forest management include a fairly rapid growth in human populations along with associated land development throughout much of the Eastern United States. In addition to the direct effect of losing forested area (Wear 2002), the current pattern of development places more people in the proximity to residual forests. The ability to manage these forests is compromised by this human presence through reduction in tract size, increased prevalence of restrictive regulations on forest uses, and negative spillover effects for neighboring landowners (Wear and others 1999).

The magnitude of this change can perhaps be best summarized by examining the density of human populations with respect to forest area (forest population density). Figure 15 shows the forest population density of counties in the Eastern United States ranging from less than 40 to more than 750 people per square mile in 2000. Roughly 20 percent of the forested area has less than 40 people per square mile, about 40 percent has 40 to 250, and 40 percent has >250. High forest population density can reflect a small forest area or a large human population or both, but they unambiguously reflect the relative scarcity of forest services relative to the size of the local population and a lowered propensity to manage forests. High forest population density is found in areas surrounding the large metropolitan areas as well as in areas with a high concentration of cropland.

U.S. population growth is expected to continue for the next several decades. Figure 16 shows the implications of a forecast of population growth to 2030 in the Eastern States. Future forest population density predictions are conservative, calculated by dividing forecasted populations by the current forest area within each county without accounting for the loss of forest land that would likely accompany development associated with expected population growth. Even so, figure 16 demonstrates a substantial growth in the forest population density throughout the Eastern United States. Forest population density is projected to grow fastest along the eastern seaboard especially from Washington to Maine, in the Southern Appalachian Piedmont and Florida, and surrounding the Midwestern cities of Chicago and Minneapolis. Thirty five percent of the forested area in the Eastern United States is projected to realize a growth of at least 25 people per square mile, with 15 percent experiencing >100. In these areas, the opportunities to conduct most forest management practices will likely be diminished.

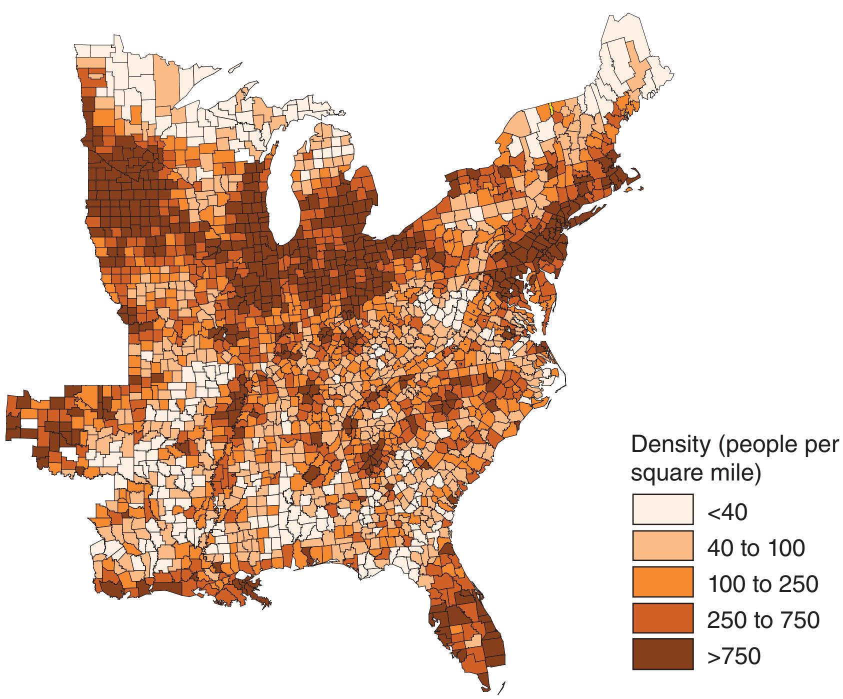


**Figure 15.** Forest population density in 2000, by county, in the Eastern United States, based on data from the U.S. Census Bureau and the U.S. Department of Agriculture, Natural Resources Conservation Service, National Resources Inventory.

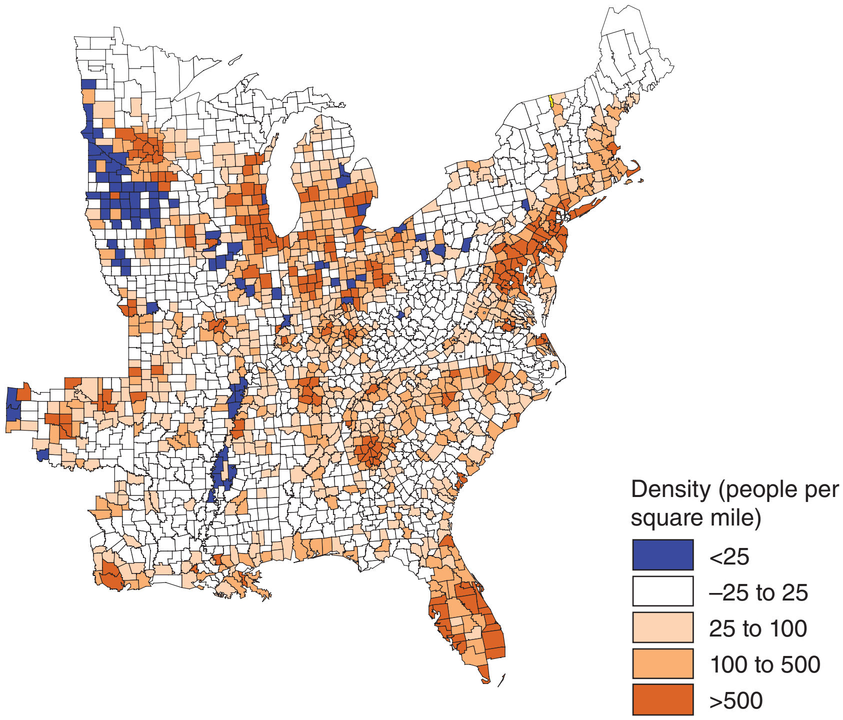


**Figure 16.** Forest population density forecast to 2030, by county, in the Eastern United States, based on Woods and Poole Econometrics projections.

In addition to population growth, changes in the forest products markets will affect the distribution of forest management in the Eastern United States. Beginning in the 1950s, the forest industry led the way in intensifying management and concentrating management on a smaller land base. This specialization of forest land uses, with some areas seeing more focus on timber production than other areas, will likely continue in spite of the sale of forest industry lands to new owners. The flow of investment capital to forests during a period when timber production and prices declined indicates a strong investor interest in forest growth and specifically in the returns to intensive management. To the extent that management becomes more concentrated on plantations and other intensively managed areas, the opportunities for management activities on the remainder of forest areas may become more limited.

These findings suggest that the practice of traditional forest management, or rural forestry, will be limited to a smaller portion of eastern landscapes. Outside the southern Coastal Plain, the Maine woods, and the northernmost counties of the Lake States, fuel treatments and other management activities normally applied in tandem with traditional forest management to support ecosystem services are not likely to occur. In rural lands throughout most of the Eastern United States, traditional management will be limited by a lack of markets for forest products and by an expanding forest population density. The greatest challenge for forest management will likely be to design practices that can be deployed in a cost efficient manner and can complement the increasingly nontimber management needs of landowners in these complex landscapes.

The potential application of fuel treatments needs to be evaluated in the context of this changing human-forest landscape:

1. An increasing human population density close to a large portion of eastern forests (rising forest population densities) is likely to result in less forest management, including fuel treatments.
2. Increased fragmentation and smaller parcels work against the economies of scale in fuel treatments, because treatments become more costly to implement on a per acre basis. As parcels become smaller, the effectiveness of treatments on management objectives also declines. Both these factors have a negative impact on the cost/benefit assessment of fuel treatments.

3. Increasing population densities and incomes in a commingled public and private ownership pose significant challenges for public forest managers. Administrative as well as management costs increase in the face of conflicting values and scale issues.
4. The trend toward forest specialization implies declining timber markets and timber management in many eastern rural areas. These areas are likely to experience increasing difficulties in applying fuel treatments or other management solely for the purposes of nontimber benefits.

All of these observations suggest challenges for the application of fuel treatments in the Eastern United States. However, expanding populations in rural lands also imply that the returns to fuel treatments, especially in the form of avoided costs of wildfire, may grow in commensurate ways, possibly leading to increased demand for the returns from fuel treatments. Realizing these returns will require innovative programs and policies to encourage management that spans parcels and coordinates the efforts of owners to deliver benefits at meaningful landscape scales.

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## Chapter 5.

# The Hot Continental Division: Oak Forests, Fire, and Ecosystem Management Frame Fuels Management Questions

**Susan L. Stout, Matthew B. Dickinson, Gregory J. Nowacki**

The Hot Continental Division is one of the larger ecoregions within the continental United States (McNab and Avers 1994), incorporating portions of 19 States and extending from the eastern seacoast to areas west of the Mississippi River (chapter 1). The Division includes the Eastern (Oceanic) and Eastern (Continental) Broadleaf Forest Provinces and two Mountain Provinces (Central Appalachian Broadleaf Forest—Coniferous Forest—Meadow Province and Ozark Broadleaf Forest—Meadow Forest), which are described in chapter 6. The entire Hot Continental Division is divided into 27 sections, 5 of which are in the mountains, and occupies 449,000 square miles (1 162 950 km$^2$), or about 12 percent of the land area of the United States, based on U.S. land area of 3,678,190 square miles [http://www.britannica.com/EBchecked/topic/616563/United-States (Date accessed: August 8, 2011)].

The Division supports many soil types, with the vast majority having mesic temperature regimes and udic moisture regimes. To the far north and west, frigid temperature regimes can be found; aquic- through xeric-moisture regimes also occur. Annual rainfall ranges from 20 to 48 inches (510 to 1320 mm). The growing season ranges from 120 to 250 days, falling mostly between 140 and 180 days. Mean annual temperatures range from 37 to 68 °F, or 3 to 20 °C (McNab and Avers 1994).

McNab and Avers (1994) list oak-dominated (*Quercus* spp.) forests among the most important natural vegetation in all nonmountainous Hot Continental Division sections, with the exception of the northern-hardwood and beech-maple (*Fagus grandifolia–Acer* spp.) dominated Erie and Ontario Lake Plain. Northern hardwoods and beech-maple forests are second in abundance to the oak-dominated forests; and coniferous evergreen species—including shortleaf pine (*Pinus echinata*), eastern hemlock (*Tsuga canadensis*), and cedar (*Chamaecyparis* spp.)—are found primarily as components of other forests. Fire is the most common disturbance agent, at least historically. Fire regimes varied widely before European settlement and in early settlement times (Guyette and others 2006), but all oak-dominated forests (from Midwest savannas to Appalachian ridges) experienced fires. Wind, ice, and insects and diseases are other disturbance agents that interacted with fire.

# History of Fire in the Hot Continental Division

A great deal of evidence, summarized in Abrams and Nowacki (2008) and other literature, suggests that oak communities in the Hot Continental Division largely owe their existence to fire regimes that favor oaks and associated species, and that the primary source of ignition for these fires was human (Abrams 2002, DeVivo 1991, Gleason 1913, Guyette and others 2002, Pyne 1982, Whitney 1994). Fire is such an important force that the boundary between this Division (fire-adapted oak systems) and the Warm Continental Division to the north (fire-sensitive northern hardwoods) was probably more attributed to regular fire occurrence than to climate (Cogbill and others 2002, Curtis 1959). The fire regimes of the Hot Continental Division included those that sustained open prairie as well as forest vegetation. Brose and others (2001) state that, "Generally, fires caused by Indians were periodic, low-intensity surface fires ignited in the spring or fall." Lightning was likely never more than a secondary source of ignition (Abrams and Nowacki 2008, Guyette and others 2006) and—in contrast to human-ignited fires—may have been most frequent in late summer and autumn when dry conditions and thunderstorms coincided (Petersen and Drewa 2006). In an analysis of presettlement and early settlement fire regimes across the central hardwoods, Guyette and others (2006) describe the spatial and temporal variability in fire regimes, attributing it to topographic resistance to fire spread, changes in human populations through time, cultural differences, drought, and continental-scale variations in climate.

Brose and others (2001), Iverson and others (2008), and Pyne and others (1996) provide evidence that early settlers used fire in ways that were similar to, and often based on, the practices they observed from local tribes, often continuing the practice of autumn or early spring burning. Although they added fires incidental to railroad logging and charcoal and iron production "fire regimes did not change enough to cause region-wide shifts in species composition" (Brose et al. 2001). As early settlers moved into topographically rough mountains, they may have increased fire frequency and the proportion of the landscape burned, compared to the final stages of the Native American period (Dey 2002, Guyette and others 2002). Frequent to annual fire favors the development of mixed oak forest types especially on high-productivity mesic sites (Brose and Van Lear 1998, Dey and Hartman 2005, Kruger and Reich 1997, Waldrop and others 1992).

The Hot Continental Division's continuum of oak forest types—from open prairies and savannas through woodlands to closed-canopy forests, all characterized by frequent, usually low-intensity surface fires and relatively high decomposition rates of woody material (Waldrop and others 2006)—resulted in continuously moderate fuel loads, ready for the next fire. More open conditions fostered fine fuels in the form of grasses and forbs that also encouraged fire. Oak foliage decays more slowly than the more mesophytic species (Hobbie and others 2006, Piatek and others 2009); as oak leaves dry on the forest floor, they curl, providing a well-aerated bed that readily supports fire spread (Loomis 1974). Stambaugh and others (2006) observed that fine fuels and litter rapidly accumulated after surface fires in hardwood-dominated forests in the Ozark Highlands. Within 4 years after burning, litter had recovered to 75 percent of the original amounts; at 12 years, litter input and decomposition reach equilibrium. A continuous cover of litter and fine fuels is essential for fire to readily move across large areas; and fuels that are able to dry in sunny, windy weather promote ignition and spread of fire in humid eastern climatic regions.

In areas with closed forests, however, fire regimes changed substantially with intensive forest harvesting and resource exploitation in the late 19th and early 20th centuries—a change fueled by the Industrial Revolution, the introduction of the railroad, and the development of railroad technology to climb steep grades and make sharp turns (Marquis 1975). The slash left by widespread industrial harvests provided unprecedented fuel loading, and the trains that transported timber provided a new ignition source. These high-intensity fires, on balance, favored chestnut oak (*Q. prinus*), although new stands could not develop until the frequency of burning subsided (during the fire suppression era that followed). Northern red oak (*Q. rubra*) was favored by the increased sunlight that followed intensive harvesting, and both red and chestnut oaks benefited from the decline of the American chestnut (*Castanea dentata*). Although red and chestnut oak experienced increases in the period between the early settlement and fire suppression eras, the once dominant white oak (*Q. alba*) declined (Abrams 2006).

Across the United States and the Hot Continental Division, the fires of the early 20th century were so intense that they produced a policy response. In Pennsylvania, nearly a million acres (400 000 ha) burned annually in the early 1900s (Banks 1960). Nationally, the fires of 1910 laid the groundwork for a policy of universal fire suppression. Lewis (2005) reports that the winter and spring of 1910 were unusually dry throughout much of the country, resulting in a fire season in which >5 million acres (2 million ha) burned, and as many as 85 firefighters died. State and Federal laws supported organizations of foresters and community volunteers to work together in fire suppression, and most fire history records show dramatic changes in fire frequency beginning around 1930 (Abrams and Nowacki 1992, Brose and others 2001, Guyette and others 2002, Iverson and others 2008). Throughout this fire suppression era, spring and autumn fires (although less frequent) continued to predominate (Haines and others 1975), a reflection that human ignition sources were still a bigger factor than lightning ignitions.

According to Nowacki and Abrams (2008), during the transition from Native American to European resource management, open systems historically maintained by frequent, human-ignited fires rapidly converted to dense, closed-canopy forests, as aged oak root stocks (“grubs”) were no longer suppressed by frequent fires (Cottam 1949, Curtis 1959, Grimm 1984). In forests of the Prairie Peninsula, continued fire suppression favored increases in shade-tolerant, fire-sensitive species, such as sugar maple (*A. saccharum*) and elm (*Ulmus* spp.). Similarly, oak savannas that were widespread in the western part of the Hot Continental Division (Leach and Givnish 1999) responded to fire suppression with increases in fire-sensitive species and overall tree density (Cottam 1949, Curtis 1959, Grimm 1984). In formerly closed-canopy systems, such as Appalachian oak forests, fire suppression generally favored the establishment of dense understories and midstories of fire-sensitive and shade-tolerant species like the maples, American beech, and flowering dogwood (*Cornus florida*). Nowacki and Abrams (2008) named this process “mesophication” and spatially quantified the phenomenon in a fire regime change map (fig. 1). The map shows large reductions in fire frequency throughout the Hot Continental Division, most dramatically in areas farthest west.

# Consequences of Mesophication

Changing forest conditions mean changing fuel conditions. With fire suppression, stand density increased—especially for fire-sensitive, shade-tolerant mesophytic species—which led to cool, moist understory conditions with forest floors dominated by fast-decaying, compact layers of moist leaves (Nowacki and Abrams 2008). Similarly, the amount of woody debris (fuel) decreases as decay-resistant oak logs are replaced by the more degradable logs of mesophytic species (MacMillan 1988, Tyrell and Crow 1994). Thus, the mesophication process is a kind of fuels management process, albeit largely unintended, that makes fire less likely without intervention through silviculture.

The absence of fire and the shift in forest management policies and practices have important ecological consequences for forest types that are adapted to fire. For example, ground flora accustomed to the variable sunlight conditions (including medium- and high-light conditions) commonly associated with the historic burning regime cannot persist in the uniform, low-light environment of today’s closed-canopy forests (Brudvig and Asbjornsen 2009). Light is the ultimate limiting factor, such that understory plant cover and diversity are inversely related to tree density and basal area (Anderson and others 2000, Taft 2009). As tree density and shading increases, plants disappear in a predictable manner, from perennial grasses through sedges to perennial forbs (Taft 2009), or convert from high-light-requiring prairie species to shade-tolerant forest


**Figure 1.** Past-to-current change in fire regime across the Eastern United States, based on spatial analysis regime maps (Nowacki and Abrams 2008), derived by applying a frequency-by-intensity fire classification to past and current vegetation maps; the departure from zero relates to the extent of fire regime change [past vegetation maps based on potential natural vegetation from Schmidt and others (2002); current vegetation maps based on the Advanced Very High Resolution Radiometer Project, http://noaasis.noaa.gov/NOAASIS/ml/avhrr.html (Date accessed: June 22, 2011), and the National Map of Land-Cover Vegetation, http://www.gap.uidaho.edu/landcoverviewer.html (Date accessed: June 22, 2011)].

species (Anderson and others 2000). In most situations, tree density has progressed to the point where remaining ground plants are few, sparsely distributed, shade-tolerant, and relatively indifferent to density increases or decreases (Taft 2009). Loss of ground cover has been associated with increases in soil erosion and runoff (Wilhelm 1991). Fortunately, the negative trends of ground flora (cover, richness, and diversity) can be reversed through active management (Apfelbaum and Haney 1991, Brudvig and Asbjornsen 2009, Taft 2009), so long as viable seed banks still exist (Anderson and others 2000). Mechanical treatments coupled with prescribed burning seem to provide the best results for restoring robust ground floras, compared to prescribed burning alone (Nielsen and others 2003).

Crow (1988), Loftis and McGee (1993), and Lorimer (1993) document widespread difficulty in producing a significant oak component when regenerating oak-dominated stands, with much of the difficulty occurring across the Hot Continental Division. The loss of oak from the forests of the Hot Continental Division is important because of its effects on wildlife, on plant communities, and on the economic benefits provided by forests. McShea and Healy (2002) emphasize the importance of acorn mast as "the most important wildlife food in the deciduous forests of North American, the ecological equivalent of manna from heaven." Van Dersal (1940) lists 101 North American bird and 104 mammal species that take sustenance from oak, including direct consumption of acorns and browsing on plant material. At least 51 of these species are in the Hot Continental Division, including white-tailed deer (*Odocoileus virginianus*), black bear (*Ursa americanus*), eastern gray squirrels (*Sciurus carolinensis*), and blue jays (*Cyanocitta cristata*). Oak timber is a highly valued resource; its loss from forests results in reduced economic benefits.

The importance of oak ecosystems, their fire regimes, their value to plant and animal species, and the widespread realization that they are being replaced by more mesic forest types have prompted most national forests in the Hot Continental Division to adopt forest plans that include some elements of oak restoration, prescribed fire, and management of fire regime class (table 1), either in the goals and objectives or in vision statements of their recently revised Land and Resource Management Plans. The Eastern Region of the U.S. Department of Agriculture Forest Service has identified oak ecosystems as a focus for restoration efforts (Nowacki and others 2009), consistent with

**Table 1.** Oak restoration and related goals and objectives of national-forest and recreation-area lands in the Hot Continental Division.

| National forest/recreation area | Oak restoration objective(s) and goals |
|---|---|
| Daniel Boone National Forest | Restore and maintain 3,000 acres of pitch pine and pitch pine-oak forest types on appropriate land-type phases. |
| | Reintroduce fire use across the landscape to increase biodiversity and improve resilience and stability of ecosystems. |
| | Move acres from FRCC 3 and 2 into FRCC 2 and 1, and reduce abundance of fire-intolerant species in fire-mediated areas. |
| | Provide adequate habitat to support populations of management indicator species. |
| Hoosier National Forest | Use prescribed fire to maintain fire-adapted ecosystems, to promote a more diverse community of plants and animals, and to manage accumulated fuels. |
| | Provide the diversity of habitats needed for viable populations of all native and desired nonnative species. |
| Land-Between-The-Lakes National Recreation Area | Use wildland fire, when practical, to protect, maintain, and enhance natural and cultural resources and, as nearly as possible, to function in its natural ecological role. |
| | Restore and maintain fire regimes and fire return intervals in fire-dependent communities (improve FRCC status) and use fire and other treatments to restore and manage for a healthy, predominantly oak-hickory forest type with respect to species composition, forest canopy structure, and associated wildlife species. |
| Mark Twain National Forest | Reestablish the role of fire in the natural communities of the Ozarks by emulating the historic fire regime. |
| | Restore FRCC 2 or 3 lands to condition FRCC 1. |
| | Facilitate restoration treatments, then emulate the range of natural variability for historical fire regimes in glades, savannas, and pine woodlands. |
| Shawnee National Forest | Use landscape-scale burning for oak-hickory forest management where coordinated, active vegetation management can be implemented, and for barrens management on shallow soils and poorer sites. |
| Wayne National Forest | Promote restoration and maintenance of the oak-hickory ecosystem by improving conditions for oak regeneration in the Historic Forest and Historic Forest with Off-Highway Vehicle Management Areas. |
| | Use all available silvicultural treatments, including precommercial and commercial thinning, prescribed fire, shelterwood harvests, and improvement cutting to promote the maintenance and restoration of the oak-hickory forest type. |
| | Use prescribed fire to conserve fire-adapted plant and animal biodiversity and to maintain and restore mixed-oak and native pine ecosystems. |
| | Use prescribed fire and mechanical treatments to modify current fuel composition and fire frequency, severity, and pattern. |
| | Use prescribed fire and mechanical treatment to maintain a current FRCC that represents a historic range of variability. |

FRCC = Fire Regime Condition Classes.

the Agency's ecosystems restoration framework (Day and others 2006). There is widespread interest in coordinating these efforts through a unified monitoring approach that would allow national forest managers to share data and learn from the experiences on other forests, and that would also incorporate scientists and other land managers into the process (Yaussy and others 2008). Several States in the Hot Continental Division have also adopted oak restoration programs [http://mdc4.mdc.mo.gov/Documents/13728.pdf and http://www.dcnr.state.pa.us/forestry/sfrmp/sfrmp_update_2007.pdf (Date accessed: June 20, 2011)], and The Nature Conservancy has also embraced oak ecosystem restoration as an important goal for some of its conservation areas [http://www.nature.org/wherewework/northamerica/states/ohio/preserves/art17415.html (Date accessed: June 20, 2011)].

## Fuels and Fuel Management in the Context of Oak Restoration

Forest floor fuel levels are remarkably stable across the Eastern United States because fuel deposition and decomposition are in balance (Graham and McCarthy 2006, Waldrop and others 2007). The fact that more productive (mesic) sites generate more fuel inputs than less productive (xeric) sites is offset by the higher decomposition rates on the more productive sites (Waldrop 1996). Consistent with this generality, differences in topographic position had little influence on fuel levels across the Southern Appalachians (Waldrop and others 2007) and aspect made little difference in fuel levels in the Missouri Ozarks (Kolaks and others 2003). Disturbance history and type, however, appreciably affect fuel accumulation. Fire reduces the litter layer (Graham and McCarthy 2006, Phillips and others 2000, Waldrop and others 2007). Waldrop and others (2007) found significant increases in 1-hour fuels for beetle-killed plots (compared to undisturbed plots) and 10-hour fuels for plots that had been attacked by beetles, harvested, and burned. After the first fire at the Ohio site of the National Fire and Fire Surrogates Study, increases in fine fuel loads were transient because of high decomposition rates (lasting <3 years); and large-diameter woody debris (>7.6 cm diameter) fuel loads increased, both in plots that were thinned and those that were thinned and burned (Graham and McCarthy 2006). Two subsequent fires appeared to create a positive feedback between fire intensity and woody fuel loads, with relatively high fire-line intensities on dry aspects resulting in high tree mortality and dense woody regeneration (often from sprouts), which in turn resulted in high woody fuel loads for the next fire [Iverson and others 2008, Dickinson (author observation)].

Fire regimes are profoundly affected by suppression-induced compositional changes—especially shifts from grass- and forb-dominated understories to closed-canopy oak forests in former savanna systems, and from closed-canopy oak-dominated forests to systems dominated by mesophytic species. In the absence of active silvicultural intervention and changes in forest management practices, many systems undergoing the mesophication process "may be approaching critical ecological thresholds and near-irreversible state shifts," according to Nowacki and Abrams (2008). They identify large contiguous blocks of public land as the optimal sites for restoration activities using prescribed fire, because burning larger landscapes would maximize benefit-to-cost ratios. Table 2 shows that some national forests in the Hot Continental Division have indeed seized upon this approach, and the achievement of a 40,000-acre (about 16 000-ha) burn target in 2010 by the Mark Twain shows that implementation is well underway in at least one.[1] In addition to oak and oak-pine restoration, several other forest types, all historically fire-adapted, are the targets of the restoration efforts that rely on prescribed fire. As with mesophication of closed-canopy oak-dominated forests, suppression-induced conversions of savannas to woodlands may become irreversible

[1] Personal communication. 2010. Michael Schanta, Resource Information Manager, Mark Twain National Forest, 401 Fairgrounds Road, Rolla, MO 65401.

**Table 2.** Target prescribed fire acreages on national-forest and recreation-area lands in the Hot Continental Division as reported in forest plans; fire acreages are given for all national forests, including the Daniel Boone, which has only a small portion of its acreage in the Hot Continental Mountain Division.

| National forest/recreation area (date of plan revision)[a] | Prescribed burning planned |
|---|---|
| Daniel Boone National Forest (2004) | Increase target from 7,500 to 22,500 acres in 2004 to 25,000 to 50,000 acres in 2014 |
| Hoosier National Forest (2006) | No targets given |
| Land-Between-The-Lakes National Recreation Area (2004) | Increase target from 2010 acreage to the desired long-term average of 10,000 to 21,000 acres |
| Mark Twain National Forest (2005) | Target of 45,000 acres |
| Shawnee National Forest (2006) | Target of 12,380 acres includes: 700 acres per year of site preparation/brush disposal at harvesting, 6,600 acres per year of landscape-scale burning for oak management, three burns totaling >10,000 acres per decade for barrens, and four burns (each >2,700 acres) per decade for management of open lands |
| Wayne National Forest (2006) | Target of 6,970 acres includes: 4,600 acres per year for oak regeneration, 20 acres per year for nonnative invasive species control, 150 acres per year for herbaceous habitat management, and 2,200 acres per year for hazardous fuel reduction |

[a] Because most of the Ozark National Forest is in a Mountain Province of the Hot Continental Division, information about the Ozark is not included in this table.

once grass and forb seed and bud banks are exhausted (Nielson and others 2003, Vogl 1964).

Tables 1 and 2 reflect a diversity of approaches in scale and in context of how and why fire is being used on national forests in the Hot Continental Division. Some forests are integrating stand-level prescriptions into their silvicultural programs with a specific focus on optimizing oak regeneration responses in mesic forests, and others are embracing the landscape-scale approach to restore oak savanna and woodland systems in more xeric areas.

## *Stand Scale Approaches to Fuels Management and Prescribed Fire*

As the growing importance of sustaining or restoring oak ecosystems has been recognized, so also has a growing body of research that combines prescribed fire with other canopy-opening treatments (Yaussy and others 2009). Van Lear and others (2000) described a silvicultural approach to regenerate oak in forests that have changed as a result of fire exclusion; the technique increases the competitive advantage of oak seedlings by introducing fire when the seedlings are the most resilient. Brose (2004), Brose and Van Lear (2000, 2003, 2004), and Brose and others (2008) elaborated on the required conditions of oak seedlings and the importance of burn-season choice and fire intensity. This work of Brose and his collaborators (2008) has been integrated into the SILVAH decision support system [http://www.nrs.fs.fed.us/tools/silvah/ (Date accessed: June 20, 2011)]. Their work has been confirmed by other research in the Hot Continental Division—such as Iverson and others (2008) and Neilson and others (2003)—showing that a combination of gap formation and prescribed fire is more effective than prescribed fire alone in promoting successful oak regeneration and reinvigorating ground flora; and that the oak regeneration period can be quite long. Ongoing research is examining combinations of mechanical thinning, herbicide application, and prescribed fire to improve oak regeneration.[2] In analogy to mechanical thinning, canopy

[2] Personal communication. 2010. Todd Hutchinson, Research Ecologist, and Joanne Rebbeck, Plant Physiologist, U.S. Forest Service, Northern Research Station, 359 Main Road, Delaware, OH 43015.

opening from the mortality of diseased, canopy white oaks in Ohio on sites that experienced multiple, low-intensity fires has led to abundant oak regeneration in contrast to unburned sites where the same canopy disturbance occurred.[3]

The techniques described in the above papers function at the stand level and require careful observation of good acorn crops and the development of oak seedlings afterwards. Understory and overstory shade is manipulated to culture oak seedlings with large root systems while competing regenerating species are growing in height. If applied in the early growing season just after leaf out—when oak seedlings still have sufficient belowground carbohydrate reserves and competitor carbohydrate reserves are at their lowest—prescribed fire favors oak, which sends out new sprouts to assume a more competitive position in the regeneration. If timber production is a principal objective, fuel management is necessary near desirable crop trees in the seed-source age class; Brose (2009) developed a photo guide to help managers conserve valuable crop trees through prescribed fires.

To control stand stocking, growth, and yield, forest managers across the Hot Continental Division use a variety of other techniques (including thinning, herbicide, and fertilization) that can affect the management of fuels. Thinning treatments and shelterwood seed-cuts add coarse woody debris to the fuel load (Kolaks and others 2004). These additions can be minor or significant depending on the size, species composition, and utilization standards of the specific locality. The persistence of high fuel loading varies by locale and depends on factors, such as climate, that control decomposition rates. Clearcuts add a large pulse of woody material to the forest floor; loadings decline rapidly over the first couple of decades until woody input from new growth produces an increase in downed biomass (Waldrop and others 2006). Like precommercial thinning, herbicide applications add fuel to the forest floor, but their other cumulative effects are small (Ristau 2010). If prescribed fire is part of the intended sequence of stand treatments, managers can achieve the silvicultural objectives of prescribed fire if they implement the other treatments in the sequence with an eye to fuels management.

When prescribed fire activities are conducted at the stand scale, their watershed cumulative effects—including effects on erosion, sedimentation, and nutrient loading—are likely to be quite small (chapter 12; Yang and others 2007, 2008). This is true whether the prescribed fires are used primarily for fuels control or natural resource management objectives, and refers to forests without an emphasis on landscape-scale burns.

Low-intensity fires to reduce canopy density in savanna and former savanna systems have been found to be less reliable than high-intensity fire combined with mechanical treatment. Haney and others (2008) found that low-intensity fires repeated over 20 years were ineffective at restoring savanna canopy species composition and structure, although evidence suggested that a longer period of low-intensity fire might be effective. In another savanna site, generally low-intensity fires repeated over 32 years incrementally reduced overstory density (Peterson and Reich 2001). Mechanical thinning is often viewed as a means to reestablish structure before fire reintroduction (Brudvig 2010). Nielsen and others (2003) found that preliminary thinning (followed by prescribed fire) was effective in reducing overstory density, but that fire alone was ineffective. High-intensity fires in former savanna sites (Haney and others 2008) and in woodlands adjacent to savannas (Anderson and Brown 1983) have resulted in substantial overstory mortality.

## *Landscape Scale Approaches to Fuels Management and Prescribed Fire*

A number of forest managers in the Hot Continental Division are adopting landscape-scale prescribed fire treatments. They include those on the Mark Twain and Daniel Boone National Forests and the Land-Between-The-Lakes National Recreation

[3] Personal communication. 2010. Todd Hutchinson, Research Ecologist, U.S. Forest Service, Northern Research Station, 359 Main Road, Delaware, OH 43015.

Area, which are the primary practitioners of landscape-scale prescribed fire. For these forests, the objectives include restoration of oak savanna and woodland conditions with open canopy structure. They not only have high acreage targets for prescribed fire ecosystem and fuels management activities, but also have relatively short return intervals for fire—often in the 3 to 5 year range. Written interviews with the fire management officers on these forests were conducted in the autumn of 2009 and excerpts of their replies are reported in the remainder of this chapter:

- Reggie Bray, Fire Management Officer, Mark Twain National Forest, Ava/Cassville/Willow Springs Ranger District, 1103 South Jefferson, Ava, MO 65608
- Michael Davis, Fire Management Officer, Hoosier National Forest, Tell City Ranger District, 248 15th Street, Tell City, IN 47586
- Jody Eberly, Fire Management Officer, Mark Twain National Forest, 401 Fairgrounds Road, Rolla, MO 65401
- Keith Kelly, Zone Fire Management Officer, Mark Twain National Forest, 401 Fairgrounds Road, Rolla, MO 65401
- James McCoy, Fire Management Officer, Land-Between-The-Lakes National Recreation Area, 100 Van Morgan Drive, Golden Pond, KY 42211
- Paul Nelson, Forest Ecologist, Mark Twain National Forest, 401 Fairgrounds Road, Rolla, MO 65401
- Charly Studyvin, Forest Silviculturist, Mark Twain National Forest, 401 Fairgrounds Road, Rolla, MO 65401
- Bennie Terrell, Fuel Specialist, Mark Twain National Forest, 401 Fairgrounds Road, Rolla, MO 65401

When asked, "Is your prescribed fire program designed to reduce the risk of wildfire or is ecosystem management the major objective of your fire program?" all respondents identified ecosystem management or restoration components; most added that fuels reduction was an ancillary but important benefit, and some weighted both goals equally. An important implication of the Nowacki and Abrams (2008) mesophication argument is that oak-dominated forests are better adapted to fire and burn more easily; thus, although reintroduction of fire to these landscapes may have short-term fuels management benefits, they also represent a choice to manage more fire-prone forests for the associated ecosystem benefits on the landscape scale.

The landscape approach to oak ecosystem management used on the Daniel Boone National Forest involves periodic ridge ignition intended to maintain oak dominance on upper slopes. After ignition, fires move down slopes and into drainages. As much as possible, natural barriers are used and, typically, the fires self-extinguish on middle and lower slopes, although some may continue to spread for several days. The overall strategy is to restore and maintain oak dominance on the more xeric portions of the landscape where oaks have a competitive advantage, instead of attempting to convert to oak on more mesic sites. An added benefit is that fire intensities and spread rates are reduced, with lower risk expected for vulnerable wildlife populations (Dickinson and others 2009). The downside is that ridge ignition takes patience and a tolerance for smoke production over multiple days and nights.

Fuels management is an important consideration when using prescribed fire to reduce stand density in woodland and savanna restoration. Controlling overstory stocking by thinning with fire is difficult, and distribution of fuels (near trees intended for removal and away from trees that managers want to retain) is a critical factor in the achievement of desired overstory mortality and eventual return to the target woodland and savanna structure. During the dormant season, low-intensity fires often fail to reduce overstory stocking sufficiently to create open woodland or savanna structure, and high-intensity fires can cause more overstory mortality than intended (Anderson and Brown 1983). At worst, fires ignited in low humidity and hot-dry weather in high fuel loading of cured slash can cause complete overstory mortality.

## *Cultural and Spatial Concerns in Fuels Management and Prescribed Fire*

Fire planning on national forests incorporates a suite of cultural and spatial concerns. Areas adjacent to campgrounds and administrative sites undergo priority fuels management treatment. Areas with recent natural disturbances such as ice or windstorms are monitored for fuel loading. For instance, fuel beds in mixed-oak forests in the Ohio River Valley that had been classified as fuel model 9 (Anderson 1982) for fire behavior were reclassified to slash models 10 and 12 after an ice storm, with model loadings[4] of 12 tons per acre (2.2 t/ha) and 36 tons per acre (6.6 t/ha). Dormant season wildfires in ice-storm fuels have resulted in areas of near complete overstory mortality on southerly aspects.[5]

Roads, in-holdings, and neighbors—both in the wildland-urban and the rural interfaces—are considered in laying out prescribed fires and in smoke management planning. A recent study just north of the Hot Continental Division identified strategies that forest managers can use to balance stewardship of fire-dependent ecosystems with the need to reduce fire risk for neighboring landowners. Using landscape-scale simulations, this study suggested that substantially reducing human-caused ignitions and redistributing fire-dependent forest types away from human ignition sources can offer "viable solutions for mitigating long-term fire risk and reducing land-use conflict in multi-owner landscapes" (Sturtevant and others 2009).

Habitats for species of concern are flagged for special attention; several of the surveyed managers identified the Indiana bat as a focus for prescribed-fire timing and placement. Potential direct effects from smoke, gases, and heat must be balanced by potential long-term habitat benefits (Dickinson and others 2009, Lacki and others 2009). Because rare, threatened, and endangered plants are known to benefit from prescribed fire, they also influence prescribed burn planning. For example, the smooth purple coneflower (*Echinacea laevigata*) that inhabits the eastern reaches of the Hot Continental Division and landscapes farther south "requires fire to maintain its preferred open habitat" (Owen and Brown 2005). Only 2 of the 186 species that Owen and Brown (2005) surveyed—the rock gnome lichen (*Gymnoderma lineare*) and the large flower skullcap (*Scutellaria montana*)—are found in the Hot Continental Division and have been classified as species "adversely affected by fire." Land managers must balance potential benefits of fire to threatened and endangered flora and fauna by potential risks from invasive species that are stimulated by disturbances, including fire.[6]

All respondents to the November 2009 manager survey identified some concern about wildfire on the landscapes they manage; especially when they are human caused, resulting from accidental escape of debris burning or arson. Keith Kelly reported that, "Wildfire is a concern, because it is our responsibility as land managers to suppress uncontrolled wildfires. We typically do not have catastrophic stand-replacing large-scale wildfires, but can have some small stand-replacing events."

Other managers also expressed anxiety about the impacts of wildfires. One reason for this anxiety is that firefighting resources, especially well trained fire personnel, are limited, and wildfires can exceed local capacity and consume resources needed to manage a prescribed fire program. The problem is made more acute by the relatively few windows of opportunity for prescribed burning during the spring and autumn seasons. Further, combating wildfires in other areas, especially the Western United States, reduces the ability of Hot Continental Division managers to achieve fuels management objectives by tapping local fire-trained personnel, particularly during the summer and

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[4] Bowden, M.W. 2003. Dean and Shawnee State Forest ice storm 2003—fuels and fire behavior assessment. 9 p. Unpublished report. On file with: Ohio Department of Natural Resources, 1855 Fountain Square Court, Building H-1, Columbus, OH 43224-1327.

[5] Personal communication. 2010. Michael Bowden, Forest Fire Supervisor, 1855 Fountain Square Court, Building H-1, Columbus, OH 43224-1327.

[6] Personal communication. 2008. Joanne Rebbeck, Plant Physiologist, U.S. Forest Service, Northern Research Station, 359 Main Road, Delaware, OH 43015.

autumn. When this happens, local burning is not possible. Combinations of wildfire and prescribed fire can reduce public support for fuels management programs that include prescribed fire; and some organizations in the Hot Continental Division have adopted opposition positions that could reduce the ability of managers to implement fuels management or ecosystem restoration programs (Buckeye Forest Council 2009).

In the limited number of studies that measure the effects of both prescribed and wild fires on water quality in Eastern North America, the most dramatic impacts have occurred where soils are shallow and fires are severe (chapter 12). This combination is rare in the Hot Continental Division. However, water monitoring may be needed if large landscapes are being burned in single, aerial ignition events and severity varies across the burn unit.

Land managers in the western reaches of the Hot Continental Division have reported hydrological benefits from reintroducing fire in areas where fire suppression and subsequent forest densification have resulted in the loss of seeps and springs. In these instances, thinning such areas by fire and mechanical treatments can restore these important aquatic habitats. Wilhelm (1991) suggests that the loss of protective ground cover associated with increased forest density has increased soil erosion and runoff, and that restoration efforts could reverse these trends.

Large woody debris serves a critical function in riparian systems, providing a substrate for invertebrates and plant life and habitat complexity that benefits fish and wildlife (Guyette and others 2008). Most streams and flood plains of the Hot Continental Division experienced continual recruitment of large woody debris from trees migrating back onto surfaces after glaciers retreated. Guyette and others (2008) found wood accumulating in Midwest streams since the late Pleistocene (about 14,000 calibrated radiocarbon years ago). The Great Cutover that swept across the Eastern United States and the conversion to agriculture during European settlement greatly reduced inputs of large woody debris into streams. This was somewhat offset by the longevity of oak logs; once submerged and integrated into fluvial deposits, they could later be resurrected and resume functionality through stream dynamics (excavation). In Missouri, the median residence time of oak boles was found to be 3,515 years (Guyette and others 2008); thus, representing one of the most persistent carbon sinks in North America. Large oak wood has more-or-less been continually replenished over time, with a substantial dropoff only within the last 150 years of logging, conversion of riparian forests to agriculture, and channel modification. The compositional shift from rot-resistant oaks to highly degradable elm and hackberry (*Celtis occidentalis*) along stream bottoms has also been a factor (Guyette and others 2008). The consequences of these recent human impacts are still unfolding, but they hint at the importance of restoring oak ecosystems along riparian zones and allowing streams to function without human intervention, migrating within their valleys.

Maintaining oak as a component in forests of the Hot Continental Division helps to regulate carbon and nutrient stores in the forest floor. Compared to other tree species, oak leaves decompose slowly, in part because of their high-lignin content (Hobbie and others 2006); this translates to higher carbon and nutrient retention (Piatek and others 2009). Oak leaves are effective in immobilizing nitrogen (Piatek and others 2009), particularly significant in light of the number of ecosystems throughout the world that are at risk of nitrogen saturation (Aber and others 1998, Fenn and others 1998, Vitousek and others 1997). Without oak, more nitrogen would be mineralized from the litter of other species, hence increasing total nitrogen availability in the system (Piatek and others 2009, Templer and others 2005). Alexander and Arthur (2010) found winter net nitrification rates of soils to be 5 to 13 times greater beneath red maple (*A. rubrum*) than oaks. Ultimately, excess nitrogen (often in the form of nitrates) is exported into and degrades rivers, streams, lakes (Peterjohn and others 1996, Piatek and others 2009, Vitousek and others 1997), and downstream estuaries. Indeed, if left unchecked, the compositional shift from oak to maple will profoundly alter forest hydrology and nutrient availability, with many unknown cascading effects (Alexander and Arthur 2010).

# Research and Operational Needs

Prescribed burning is a critical requirement for forest health across the Hot Continental Division (Nowacki and others 2009). The need arises from fire suppression during much of the 20th century, which has resulted in the mesophication of forests and the loss of savanna and woodland habitats. In response, many land and resource management agencies and conservation organizations have adopted plans to reintroduce fire to these landscapes. Emerging research further corroborates the link between fire and the ecology of oak and other fire-dependent vegetation types.

When prescribed fire is used at the stand scale, research needs revolve around its impacts on nontarget organisms—such as the timber rattlesnake (*Crotalus horridus*), the Indiana bat (*Myotis sodalis*), and a host of fire-dependent or fire-adapted plants—its interaction with invasive species, its impacts on timber growth and quality, and its continuing integration with other silvicultural practices. For example, research is needed to match the season and intensity of burn requirements for oak regeneration with the seasonal use of various bat habitats, to better understand the effects of smoke on hibernacula and roosts, and to provide a scientific basis for evaluating the tradeoffs between short-term damage to a single species and long-term, large-scale loss of oak habitat (Dickinson and others 2009, Lacki and others 2009).

The impact of projected climate change on species composition, fuels accumulation, and fire risk is daunting to imagine, but study of these interactions and impacts will be required to help managers sustain ecosystem function across the Hot Continental Division, especially those working at landscape scales. As examples, simulations with a dynamic vegetation model under climate scenarios through the 21st century suggest a general drying trend in the Eastern United States and the possible conversion of closed forests to more open ecosystems over large areas (Lenihan and others 2008). Based on current relationships between tree distribution and climate, tree distributions under future climate scenarios are projected to change significantly, with significant increases in the ranges and importance of the oak groups that favor warmer and drier conditions (Iverson and Prasad 2002). However, even under these scenarios, oaks will continue to require fire and/or fire surrogates to expand their range concomitant to shifts in climate envelopes.

Some national forest plans in the Hot Continental Division focus their goals and strategies around landscape-scale prescribed burns, repeated on 3- to 5-year cycles. Although evidence is strong that practices like these were common for hundreds—if not thousands—of years through the early 20th century, the impacts of treatments at the landscape scale have not been documented. This results in both policy and research needs.

For example, both policy guidance and new research is needed to develop methods that optimize the acreage affected by fuel-management and ecosystem-restoration treatments. Four Mark Twain National Forest (MTNF) managers articulated this need in their written interviews (Eberly, Terrell, Studyvin, and Nelson):

> The 2005 Forest Plan emphasizes restoration of fire-adapted ecosystems in MP 1.1 and 1.2 covering 19 management areas totaling 438,000 acres.[7] The objectives include treating and moving 50,000 to 150,000 acres[8] of fire-adapted ecosystems toward desired restored conditions. This ecosystem-based conservation approach is part of a statewide comprehensive wildlife strategy and follows recommendations in The Nature Conservancy's Ozark Ecoregional Conservation Assessment. The majority of the MTNF's 1.5 million acres[9] was historically fire-mediated yet we barely treat 2 percent of this total acreage annually. The present inability to substantially increase the use of fire across the MTNF will leave many areas of hazardous fuels

[7] 177 000 ha.
[8] 20 000 to 61 000 ha.
[9] 600 000 ha.

> untreated and will result in the further degradation and loss of biodiversity associated with these ecosystems.

These remarks and others suggest the need for a regional and local-scale prioritization of restoration and maintenance activities for mixed-oak forests, wherein fire and other activities are only undertaken in areas where the species of interest are most favored by climate and topography.

Michael Davis of the Hoosier National Forest indicated a pressing need for prescriptive smoke-dispersion and burning standards. This need, too, is magnified in areas of the Hot Continental Division where landscape-scale burning is or will be practiced.

Another pressing research need is continued assessment of the cumulative effects that would occur should these treatments not be undertaken, and ongoing mesophication of the landscapes were allowed to continue. Although it seems quite likely that impacts of continued unabated mesophication will interact with climate change—making adaptation more difficult—a formal assessment of this risk would be beneficial. For instance, with continued fire suppression and ongoing mesophication, favorable climate shifts (in terms of oak) may not necessarily result in projected increases of oak species.

Finally, as described in chapter 12, additional study will be required to evaluate the cumulative watershed impacts of prescribed burning at the landscape scale, especially where large-scale burns include areas of high fire intensity. A watershed-scale perspective may also be needed to determine where restoration of seep and spring habitats can be accomplished by forest thinning with fire. Also needed are new metrics to measure landscape-scale intensity, and increased attention to the effects of prescribed fire at any scale on mercury transport and accumulation in the food chain (chapter 12).

# Conclusions

Many forest ecosystems in the Hot Continental Division have historically depended on frequent, low-intensity fire—usually set by humans—to maintain their fire-adapted species and relatively open conditions. These ecosystems, especially the wide variety of oak savannas, woodlands, and forests, became critically important habitat for many wildlife (McShea and Healy 2002, Rodewald and Abrams 2002) and plant species (Anderson and others 2000, Apfelbaum and Haney 1991, Brudvig and Asbjornsen 2009, Nielson and others 2003, Owen and Brown 2005, Taft 2009). For example, at least 96 vertebrate species depend on acorns for some or all of their sustenance (McShea and Healy 2002). The prominent role that fire played in North American ecosystems is definitively reflected in the physiological requirements of plants. Of 186 Federal listed, proposed, and candidate plant species on National Forest System lands, 47 were found to require fire, 65 to tolerate fire, 70 to be unaffected by fire (Owen and Brown 2005), and only 4 (2 percent) to be adversely affected by fire.

With fire suppression, oak-dominated (and similar) forests have accumulated high densities of fire-sensitive, mesic species such as maple, beech, and blackgum (*Nyssa sylvatica*). The changes in density and composition that result from fire suppression, labeled mesophication by Nowacki and Abrams (2008), make these forests increasingly fire resistant. At the same time, the proportion of oak in these forests is declining, with important consequences for dependent or specialist wildlife and plant species. The demise of oak degrades aquatic systems—either directly by reducing the amount of long-lived, large woody debris (aquatic animal and plant habitat)—or indirectly through increased nitrogen exports to surface waters.

As a result, some conservation organizations and many public land management agencies, especially the national forests across the Hot Continental Division, have adopted ecosystem restoration goals to return fire-adapted species and forest community types to their previous condition and function on the landscape. Prescribed fire, with its ancillary fuels management benefits, is a primary tool to achieve these objectives. On some landscapes, prescribed fire is added to the stand-level toolkit for forest managers; all available evidence suggests that these practices have modest to no cumulative effects on water quality. Moreover, the application of fire at watershed scales may actually result in aquatic habitat improvements through increases in growing-season water yield. Management guidelines are emerging to strengthen these prescriptions with safeguards for endangered species—especially the Indiana bat—but more research is needed on these species, and similar research is needed on other species of concern. Management and policy guidelines are needed to balance the local and airshed impacts of smoke (Charney and others 2006) with the benefits of ecosystem restoration, particularly as air quality standards are tightened (Achtemeier 2009).

Most research needs are tied to the increasing adoption of landscape-level prescribed burning strategies. Some research has already suggested management practices to reduce the risk fire poses for neighbors in the wildland-urban or rural interfaces, but further tests under specific conditions across the Hot Continental Division would be desirable. Additional research on the impacts of these practices and their cumulative effects on watersheds is needed, especially in areas where fire intensity might be high. Such research should consider both potential benefits—such as aquatic habitat improvements through increases in growing season water yield—and potential risks. The need for fuels management treatments with ecosystem management benefits is very high in the Hot Continental Division. Increased understanding of the consequences of undertaking these treatments and the failure to do so are urgent needs.

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# Fuels Management in the Southern Appalachian Mountains, Hot Continental Division

**Matthew J. Reilly, Thomas A. Waldrop, Joseph J. O'Brien**

The Southern Appalachian Mountains, Hot Continental Mountains Division, M220 (McNab and others 2007) are a topographically and biologically complex area with over 10 million ha of forested land, where complex environmental gradients have resulted in a great diversity of forest types. Abundant moisture and a long, warm growing season support high levels of productivity across the area. Disturbances such as fire, severe windstorms, ice storms, and outbreaks of pathogens and insect infestations are common and can affect large areas. The interactions among these factors can produce a dynamic forest fuels situation, requiring frequent monitoring and updating of fuel loads. Fire exclusion since the early 20th century has allowed a buildup of fuels, both living and dead, across the Southern Appalachian Mountains. A rapidly expanding wildland-urban interface and the potential for climate change to increase the frequency and severity of wildfires will require that more resources be devoted to fuel management. In this new environment, managers will need more effective methods of fuel management to reduce the potential for hazardous wildfires and maintain landscape diversity.

## Fire History

Fire played an integral role in determining historic patterns of forest vegetation across the Southern Appalachian Mountains (Delcourt and Delcourt 1997). Historical accounts suggest that recurrent burning by Native Americans was common in Southern Appalachian forests, starting 10,000 to 12,000 years ago and extending through the arrival of Europeans (DeVivo 1991, Fowler and Konopik 2007, Van Lear and Waldrop 1989). Fowler and Konopik (2007) outline five periods with unique fire regimes, all having different impacts on vegetation as fire regimes and forest structure have been influenced by changing cultures, fluctuations in population sizes, and altered land-use priorities.

During the first period, Native Americans burned entire valleys near settlements to clear land for agriculture and selectively burned upper slopes and ridges to promote wildlife habitat and mast production (Delcourt and Delcourt 1997, DeVivo 1991). Fire frequency during this time was likely negatively correlated to distance from settlements

(Delcourt and Delcourt 1997). Some estimates suggest that the fire frequency was 7 to 12 years on ridges and upper slopes at elevations below 1000 m, but less frequent at upper elevations (Frost 1995). Others suggest that frequency was annual or biannual in some areas (Barden 1997). This presettlement landscape was likely a "shifting mosaic of open grasslands, woodlands, and closed forests with widely scattered Indian villages" (Buckner 1989).

The second period of fire use began with the arrival of Europeans in the 16th century. The new arrivals introduced pandemic diseases, and native populations plummeted. The initial result for the landscape was a reduction of fire frequency, which altered forest structure. By the 17th century, the European population increased, and much of the landscape was occupied by settlers who began adopting many Native American burning practices.

A third period characterized as the Industrial Revolution began in the latter 19th century as railroads made previously isolated parts of the mountains easily reachable and enabled large-scale transportation of commodities. Subsequent large-scale timber exploitation resulted in heavy fuel loads from slash and led to fires that were much more intense, albeit not much more frequent than in previous periods (Harmon 1982).

The high intensity and often stand replacing fires ushered in the fourth period of fire exclusion beginning in the early 20th century. Complete fire exclusion was the policy of Federal and State land management agencies throughout the century and continues to the present.

The fifth period of fire management began in the late 20th century. Currently, prescribed fires are the dominant form of fire use in the Southern Appalachian Mountains. Suppression is still practiced on most wildfires. Natural ignition by lighting is infrequent (Barden and Woods 1973, Harmon 1982)

Fire exclusion caused important changes in the structure and function of Southern Appalachian forests (Vose 2000). Stem density has increased in the shrub layer and species composition has changed with a greater dominance of shrubs such as mountain laurel (*Kalmia latifolia*). High vegetation density has inhibited regeneration of overstory species and decreased diversity of herbaceous communities in the understory (Chastain and Townsend 2008). Fuel loads have also increased for reasons not directly related to fire exclusion, such as overstory mortality resulting from native and nonnative pathogens and insects. Accelerated mortality has increased the quantity of coarse woody debris and other organic matter, which have increased carbon and nutrient pools in the forest floor. This effect varies across the Southern Appalachian Mountains and among ecosystems, presenting a difficult situation for forest management and restoration where a "one size fits all" approach may not be suitable. If managers understand how the interactions of past land use and disturbances have given rise to current stand conditions, they can take appropriate actions to mitigate fuel risks.

The recognition of the role of fire in maintaining biodiversity and its usefulness as a forest management tool resulted in the active use of prescribed fire in Southern Appalachian Mountains beginning in the 1980s. Fires today are less frequent and generally much smaller than those of the past (Barden and Woods 1973, Lafon and others 2005). Despite the usefulness of prescribed fire, its application is often limited by air quality issues and operational complexities within a rapidly growing wildland-urban interface.

Of the approximately 15.2 million forested hectares encompassed by the Southern Appalachian Mountains, 84 percent (about 13 million ha) is privately owned (Southern Appalachian Man and the Biosphere Program 1996a) and 16 percent is in public management. Approximately 2.2 million ha are under Federal management, primarily in national forests or national parks. Federal lands represent the vast majority of Southern Appalachian area where fuels are being managed. They include 10 national forests, the Great Smoky Mountains National Park, and the Shenandoah National Park. An additional 230 000 ha are managed by State agencies, a little over 40 000 ha by the U.S. Department of Energy and U.S. Department of Defense, and about 20 000 ha by the Eastern Band of the Cherokee Indians.

# Southern Appalachian Forests

## Climate, Ecosystem Processes, and Disturbance Regimes

The Southern Appalachian Mountains stretch from northeastern West Virginia through western Virginia and North Carolina, to northwestern South Carolina and northern Georgia, to northeastern Alabama (Southern Appalachian Man and the Biosphere Program 1996a). Elevations generally range from 600 m in major river valleys to over 3000 m on upper ridges and peaks. Local climate varies dramatically along latitudinal and elevational gradients. Increases in elevation are associated with decreasing temperature and increasing precipitation, relative humidity, and cloud cover. Summers are usually hot, with daytime temperatures frequently exceeding 32 °C, and below freezing temperatures are common throughout the winter. Mean temperature is 19 °C in the south, decreasing to 8.3 °C in the north. Annual precipitation is abundant, ranging from a maximum in the south of approximately 200 cm to just over 75 cm in the north. Most precipitation is in the form of springtime rain, but winter snows and summer thunderstorms are frequent. Widespread drought occurs about once every decade. See chapter 3 for additional detail on the physical setting for this area.

The Southern Appalachian Mountains are well known for biological diversity and are home to a variety of forest ecosystems that are generally distributed along strong elevational and topographic gradients (Whittaker 1956). Other factors—such as precipitation and temperature—also vary along these gradients, affecting forest composition and ecosystem processes such as decomposition (Abbott and Crossley 1982), turnover of soil carbon (Garten and Hanson 2006), and aboveground forest productivity (Bolstad and others 2001). Evidence suggests that these ecosystem processes control fuel loading (Iverson and others 2003, Kolaks and others 2004, Waldrop and others 2004), but variation in rates of input across the landscape may be balanced by corresponding rates of decomposition (Kolaks and others 2003, Waldrop 1996). Evidence from over 1,000 study plots at low to middle elevation across the farthest southern extent of the Appalachian Mountains found little difference in surface fuels across topographic positions (Waldrop and others 2007). Instead, disturbance history and type were found to play a greater role in determining fuel loads.

In addition to fire, other disturbances occur at variable frequencies and severities, with impacts ranging from single-tree mortality to large areas of mortality resulting from high wind, hurricanes, floods, pathogen outbreaks, insect infestations, drought, and ice storms. A great deal of evidence suggests that these disturbances may also vary in intensity along environmental gradients (Elliott and Swank 1994, Harmon and others 1984, McNab and others 2004, Reilly and others 2006, Stueve and others 2007). The interactions between environmental gradients and disturbance hold implications for fuels management because they alter dead and down surface fuels and patterns of regenerating live fuels in recently disturbed areas. Waldrop and others (2007) found less litter on sites that had been burned in the last 10 years and higher 1-hour fuel loads on sites recently infested by southern pine beetles (*Dendroctonus frontalis*). In areas that had been subjected to beetle attack, fire, or wind—or all three—larger woody fuels were more abundant than on undisturbed sites.

## Major Forest Ecosystems

The diverse vegetation in the Southern Appalachian Mountains has the potential to create a wide array of fuel management scenarios. We present an ecosystem-based approach using major vegetation and "macro" habitat groups (Southern Appalachian Man and the Biosphere 1996b). These forest ecosystems correspond well with those described by others (McLeod 1988, Newell and others 1999, Whittaker 1956) and provide managers with a useful classification scheme. Additionally, geographic information system data on the distribution and occurrence of these ecosystem types are readily and freely available from the Southern Appalachian Assessment Online Database (http://samab.org/data/SAA_data.html).

## Bottomland hardwood forests

Bottomland hardwood forests are found at the lowest elevations in the major river valleys and cover approximately 183 00 ha in the Southern Appalachian Mountains. These forests are dominated by several species including sweetgum (*Liquidambar styraciflua*), yellow-poplar (*Liriodendron tulipifera*), red maple (*Acer rubrum*), river birch (*Betula nigra*), American sycamore (*Platanus occidentalis*), green ash (*Fraxinus pennsylvanica*), American elm (*Ulmus americana*), silver maple (*Acer saccharinum*), and eastern cottonwood (*Populus deltoides*). Bottomland hardwood forests are very productive with rapid decomposition rates resulting from seasonal flooding and high soil moisture. Floods play a role in the disturbance regime of bottomland hardwood forests and may redistribute coarse woody debris and remove litter, especially after large events.

Invasion of nonnative species has potentially altered fuel structure in bottomland hardwood forests. Dense thickets of Chinese privet (*Ligustrum sinense*) and multifloral rose (*Rosa multiflora*) may form large patches of continuous fuels capable of spreading fire under dry conditions. Large patches of kudzu (*Pueraria lobata*) reaching into canopies along forest edges may also occur. The presence of these species may warrant the use of fuels management to reduce localized fire hazards and control further spread of invasive species.

## Oak forests

Oak forests (*Quercus* spp.) occur across a wide range of middle elevations and vary in topographic moisture. These are the most extensive ecosystems in the Southern Appalachian Mountains and cover approximately 5.4 million ha. Xeric oak forests are dominated by chestnut oak (*Q. prinus*) and scarlet oak (*Q. coccinea*) with an abundant ericaceous shrub layer. Post oak (*Q. stellata*), black oak (*Q. velutina*), southern red oak (*Q. falcata*), blackjack oak (*Q. marilandica*) and bear oak (*Q. ilicifolia*) may be found at lower elevations. Mesic oak forests are dominated by white oak (*Q. alba*) and northern red oak (*Q. rubra*). Pignut hickory (*Carya glabra*) may also be present. A thick layer of potentially flammable ericaceous shrubs composed mostly of mountain laurel with several species of blueberry (*Vaccinium* spp.) and huckleberry (*Gaylussacia* spp.) is often present throughout. Rhododendron (*Rhododendron maximum*) may be present in mesic oak forests. This shrub layer represents a major source of hazardous fuels, particularly when composed of mountain laurel, and can frequently pose a serious problem for fuel management.

Fire plays a major role in the disturbance regime of oak forests. It is hypothesized that many of these forests developed under a regime of frequent low intensity fires (Abrams 1992). Fires are thought to have encouraged oak regeneration and inhibited encroachment of more fire sensitive mesic species like red maple and blackgum (*Nyssa sylvatica*). Absence of fire in the last century has likely increased the abundance of mountain laurel and other ericaceous shrubs and created hazardous fuel conditions. Wind and logging are also part of the disturbance regime in oak forests. Both of these disturbances have the potential to increase larger woody fuels (Waldrop and others 2007).

## Southern yellow pine forests

Southern yellow pine forests (*Pinus* spp.) are present on the xeric upper slopes and ridges of low and middle elevations and make up approximately 1.5 million ha in the Southern Appalachian Mountains. The major constituents are Virginia pine (*Pinus virginiana*), pitch pine (*Pinus rigida*), and Table Mountain pine (*Pinus pungens*), with their respective importance increasing with decreasing topographic moisture and increasing elevation. A dense shrub layer consisting primarily of ericaceous species including blueberry, huckleberry, and mountain laurel is frequently present. Also frequently present in the shrub layer are hardwood species such as oaks, blackgum, and red maple. Piedmont species such as shortleaf pine (*Pinus echinata*) and loblolly pine (*Pinus taeda*) may also occur but are limited to the lowest elevations. Longleaf pine (*Pinus palustris*) has a very limited montane distribution on dry ridges up to 600 m at the farthest southwestern part of the Appalachian Mountains.

Many yellow pine stands were established early in the 20th century before the period of fire exclusion (Brose and Waldrop 2006) and are now in a decadent state (Williams and others 1990). Active programs of prescribed burning are in place to promote regeneration of fire-adapted species, such as Table Mountain pine and pitch pine, by reducing the presence of encroaching shrubs and hardwood species and allowing sunlight to reach the forest floor. Past work has assumed that regeneration of these species required intense stand replacing fires, but more recent work suggests that periodic surface fires of moderate intensity may be sufficient (Brose and Waldrop 2006, Waldrop and Brose 1999).

Southern yellow pine ecosystems represent one of the most challenging issues for fuel managers. Potentially flammable evergreen canopies and abundant vertical fuels like mountain laurel can result in high severity crown fires. In addition, disturbance such as wind, ice storms, and southern pine beetle infestations can increase the abundance of both small-diameter and large woody fuels (Waldrop and others 2007). Periodic surface fires would not only facilitate regeneration but they would also reduce dangerous fuel loads.

## Mixed pine-hardwood forests

Mixed pine-hardwood forests are found on lower and middle elevation slopes and ridges across the Southern Appalachian Mountains, covering approximately 1.3 million ha. Dominant species include the major constituents of both oak and southern yellow pine forests at varying densities. Oak species may include chestnut, scarlet, white, and northern red oak. At lower elevations, pine species may include loblolly and shortleaf. Middle to upper elevation mixed pine-hardwood forests may include Virginia, pitch, and Table Mountain pines. Fire susceptible species, such as red maple, blackgum, eastern white pine (*Pinus strobus*), and eastern hemlock (*Tsuga canadensis*) may be present in areas where fire has been excluded. A shrub layer consisting of species of blueberry, huckleberry and mountain laurel is also often present.

Disturbance regimes and productivity in mixed pine-hardwood forests are similar to those of oak and southern yellow pine forests. The mixture of species in these forests could be explained by their mid-successional status. In the absence of fire to promote pine regeneration, the most likely eventual fate of southern yellow pine forests is to succeed to oak forests. This process may be accelerated by other disturbances, particularly southern pine beetle attacks, in stands with older pines. These areas may be characterized by large amounts of both small-diameter and large woody fuels on the ground (Waldrop and others 2007). A frequent, low intensity fire regime may promote the coexistence of pine and oaks in these forests.

## Mixed mesophytic hardwood forests

Mixed mesophytic hardwoods forests are among the most diverse forest communities in the Southern Appalachian Mountains, covering approximately 1.3 million ha. Dominant trees may often include yellow-poplar, white oak, northern red oak, basswood (*Tilia* spp.), yellow buckeye (*Aesculus octandra*), white ash (*Fraxinus americana*), eastern hemlock, American beech (*Fagus grandifolia*), and sugar maple (*Acer saccharum*). These forests are typically found on moist eastern and northern facing slopes and sheltered coves above 1200 m.

Fires in these forests were historically infrequent and remain that way today. Their topography and upper elevational range likely result in higher fuel moistures relative to other ecosystem types. However, periods of prolonged drought can result in overstory mortality, which may increase both surface fuels and midstory density in resulting canopy gaps (Olano and Palmer 2004).

## White pine-hemlock-hardwood forests

White pine–hemlock-hardwood forests are typical of cool, moist ravines over a range of elevations. These forests cover approximately 606 000 ha of the Southern Appalachian Mountains. Species composition is dominated by white pine and eastern hemlock with occasional hardwoods such as yellow-poplar, blackgum, sweet birch (*B. lenta*), Fraser magnolia (*Magnolia fraseri*) and red maple. Rhododendron is common in the shrub layer. Forest structure is often composed of large diameter trees at low density with a thick layer of rhododendron in the midstory.

The historical disturbance regime of white pine-hemlock-hardwood forests was likely dominated by wind. Although generally long-lived, large white pine and eastern hemlock may be susceptible to windthrow, which promotes gap phase regeneration of the less shade-tolerant deciduous species. These forests were likely sheltered from most fires because they are located in high moisture ravines. However, when fire does occur in these forests, mortality can be high (Reilly and others 2006). The recent invasion of the hemlock woolly adelgid (*Adelges tsugae*) has resulted in large-scale mortality of eastern hemlock. High rates of tree mortality will likely cause a pulse in both small and large surface fuels as branches and snags fall.

## Northern hardwood forests

Northern hardwood forests are distributed in coves and upper slopes at elevations ranging from 1200 to 1700 m, and cover approximately 249 000 ha of the Southern Appalachian Mountains. Dominant species include sugar maple, American beech, and yellow birch (*Betula alleghaniensis*). Other species such as pin cherry (*Prunus pensylvanica*) and species found in mixed mesophytic hardwood forests may also be present. Species frequently present in the shrub layer are striped maple (*Acer pensylvanicum*) and American mountain ash (*Sorbus americana*).

Disturbance in northern hardwood forests is primarily by wind; fire was likely infrequent historically. Because of the elevational distribution of these forests, fuel moisture is likely higher relative to other ecosystems in the Southern Appalachian Mountains. The response of northern hardwood forests to droughts is likely similar to that of mixed mesophytic forests, where canopy mortality may increase surface fuels and abundant recruitment results in increased sapling densities. These effects may potentially be more dramatic with increased exposure on upper slopes.

## Spruce-fir forests

Spruce-fir forests (*Picea* spp.–*Abies* spp.) occur at the highest elevations, generally above 1500 m. These forests cover approximately 36 500 ha of the Southern Appalachian Mountains. Growing seasons are short; weather is characterized by abundant moisture, high relative humidity, and high cloud cover. Dominant species include red spruce (*Picea rubens*) and Frasier fir (*Abies fraseri*). Species common to northern hardwood forests such as yellow birch, sugar maple, and pin cherry may also be present. Woody species found in the shrub layer may include rhododendron, Catawba rosebay (*Rhododendron catawbiense*), mountain maple (*Acer spicatum*), and American mountain ash.

The disturbance regime of spruce-fir forests includes wind and ice storms. Although fires are infrequent, these forests are structurally similar to boreal forests and large high severity fires have occurred during prolonged drought. In October of 1925, one North Carolina fire in Haywood County burned approximately 10 000 ha in 3 days in what is now the Shining Rock Wilderness Area (Barden 1978). Local accounts describe a stand replacing fire near Mt. Mitchell during the early 1900s. More recently, acid precipitation and attacks of the balsam woolly adelgid (*Adelges piceae*) have resulted in large-scale mortality of canopy trees. Areas recently disturbed by ice or the balsam wooly adelgid (Smith and Nicholas 2000) may have abundant coniferous regeneration capable of spreading intense fire.

# Fuel Management in the Southern Appalachian Mountains

Current fuels management in the Southern Appalachian Mountains is performed primarily by public land managers on oak, southern yellow pine, and mixed pine-hardwood forests. The most common technique employed by land managers is prescribed fire.

Goals of fuel management in the Southern Appalachian Mountains vary; but in addition to reducing the risk of wildfire, they also include promoting biodiversity, restoring native ecosystems, and improving wildlife habitat. Decreasing wildfire risk involves reducing surface fuels, and increasing the gap between surface fuels and living crowns (Agee and Skinner 2005). Promotion of biodiversity and restoration of native ecosystems often focuses on regenerating fire-adapted species like Table Mountain and pitch pines. Fuel treatments for restoring native ecosystems also include reducing the density of mountain laurel, rhododendron, and fire-susceptible tree species like red maple (Nowacki and Abrams 2008)—species that may substantially reduce regeneration of oak and other desirable species. Fuel treatments such as prescribed fire and thinning which increase surface light levels may also be used to improve wildlife habitat by increasing the growth of new vegetation and by promoting flowering (Whitehead 2003), which increases visitation of pollinators (Campbell and others 2007) and fruit production (Blake and Hoppes 1986, Greenberg and others 2007). Most studies on fuel treatments have primarily concentrated on prescribed fire and its effects on forest structure and live fuels, with little emphasis on the forest floor and dead and downed fuels. However, results from the National Fire and Fire Surrogate Study explicitly address effects of fuel treatments on the forest floor as well as dead and downed woody fuels (Waldrop and others 2008).

## *Prescribed Fire*

Prescribed fire is by far the most frequently used fuel management technique in the Southern Appalachian Mountains. Prescribed fire has a relatively short history in the area because of fear that hardwoods and soils may be damaged and the potential difficulty in controlling fire on slopes (Van Lear and Waldrop 1989). In the early 1980s, managers first used prescribed fire for site preparation after clearcutting hardwood stands (Phillips and Abercrombie 1987). The use of prescribed fire for restoration of native communities began in the 1990s (Waldrop and Brose 1999).

The effects of prescribed fire as a fuel management technique have the potential to vary a great deal, depending largely on burning conditions and the ultimate goals of managers—both of which will inevitably vary largely across ecosystems and will alter fire intensity and severity. We summarize the effects of prescribed fire as a fuel management tool from published reports in oak, southern yellow pine, mixed pine-hardwood, mixed-mesophytic, and white pine-hemlock-hardwood forest ecosystems. Caution is advised when considering the results summarized below because they are derived from a limited number of observations and likely do not capture the full range of effects under a wide variety of site and burning conditions.

### In oak ecosystems

Prescribed fires in oak ecosystems are generally low to moderate severity surface fires (Elliott and others 1999, Vose and others 1999, Waldrop and others 2008), which can be attributed to the characteristics of surface fuels in broadleaf forests and the resilience of most oak species to fire damage. However, areas of higher intensity fire can occur where there is a thick layer of ericaceous shrubs. Phillips and others 2006 reported that fire intensity in an oak forest ranged from 9.9 to 53.6 kW/m with flame lengths ranging from 0.3 to 0.5 m, and the rate of spread ranged from 0.3 to 1.4 m/minute; at 1 to 2 m above the ground, temperatures ranged from less than 52 to 160 °C. Vose and others 1999 reported that mean soil temperature reached 59 °C from 16.8 mm downward to 52 mm.

Table 1 shows that prescribed fires in oak ecosystems generally have only minor effects on forest structure (Elliott and others 1999, Waldrop and others 2007). Although the effect on stand basal area is small, density of saplings initially decreases after treatment. However, vigorous sprouting of hardwoods and ericaceous shrubs can result in sapling density that reaches or exceeds pretreatment levels 2 to 3 years after application of prescribed fire (Waldrop and others 2007). The effects on surface fuels are mostly limited to consumption of about half the mass of small wood and litter, and the effects on the humus layer and coarse woody debris are minor (Vose and others 1999). The high productivity of most sites means that surface fuels rapidly attain pretreatment loadings.

## In southern yellow pine ecosystems

Prescribed fires in southern yellow pine ecosystems, particularly those dominated by Table Mountain and pitch pine, have the potential for high severity and are therefore likely to confront managers with some of their greatest challenges. Mountain laurel can act as a vertical fuel where it is abundant, allowing flames to reach into pine canopies. Flame temperatures have reached >800 °C, with a 59 °C heat pulse penetrating 24 mm into the forest floor (Vose and others 1999). Flame lengths can vary a considerably, ranging from as low as 1 to 3 m to as high as 12 to 46 m (Welch and others 2000). Ignition of crowns on upper slopes and ridges is also possible (Elliott and others 1999).

Prescribed fires in southern yellow pine ecosystems can have major effects on forest structure (table 2). Studies have reported reductions of 20 to 35 percent in basal area,

**Table 1.** Effects of prescribed fires on live and surface fuels in two Southern Appalachian oak ecosystems, (1) in March 2003 and 2006 on the Green River Game Lands in North Carolina and (2) in April 1995 on the Nantahala National Forest in North Carolina.

| | | Basal area | | Density | | | | | Woody debris | |
|---|---|---|---|---|---|---|---|---|---|---|
| Site (elevation) | Measurements taken | All sizes | ≥5 cm | <10 cm | ≥5 cm | 1 to 4.9 cm | Litter | Humus | <7.5 cm | ≥7.5 cm |
| | | *m²/ha* | | *stems/ha* | | | *kg/ha* | | | |
| Green River Game Land | Pretreatment | 26.5 | – | 1,500[a] | – | – | – | – | – | – |
| | 1 year after treatment | 26.3 | – | 700[a] | – | – | – | – | – | – |
| | 3 years after treatment | 26.1 | – | 1,500[a] | – | – | – | – | – | – |
| | 5 years after treatment (1 year after second treatment) | 25.9 | – | 800[a] | – | – | – | – | – | – |
| Nantahala National Forest (1500 to 1700 m) | Pretreatment | – | 28.7 | – | 1,448 | 8,518 | 3775 | 14 780 | 4234 | 8096 |
| | 1 year after treatment | – | 28.4 | – | 1,365 | 1,556 | 2825 | 13 849 | 2465 | 7308 |

– = No data available.
[a] Exact values were not reported but were estimated for this summary based on figures.
Sources: Waldrop and others (2008), Elliott and others (1999), and Vose and others (1999).

**Table 2.** Effects of prescribed fires on live and surface fuels in three Southern Appalachian yellow pine ecosystems, (1) in October 1995 and (2) in May 1996 at the George Washington and Jefferson National Forest in Virginia, (3) in May 1996 at the Pisgah National Forest in North Carolina, and (4) in April 1995 in the Nantahala National forest in North Carolina.

| | | Basal area | | Density | | | | | | Woody debris | |
|---|---|---|---|---|---|---|---|---|---|---|---|
| Site (elevation) | Measurements taken | ≥2.5 cm | >5 cm | <2.5 cm | ≥2.5 cm | 1 to 4.9 cm | ≥5 cm | Litter | Humus | ≥7.5 cm | <7.5 cm |
| | | ----------$m^2$/ha-------- | | --------------------stems/ha------------------- | | | | --------------------kg/ha----------------------- | | | |
| George Washington and Jefferson | Pretreatment | 23.4 | – | 1113 | 1525 | – | – | – | – | – | – |
| | 1 year after treatment | 16.6 | – | 2912 | 625 | – | – | – | – | – | – |
| | Pretreatment | 28.4 | – | 1788 | 1594 | – | – | – | – | – | – |
| | 4 months after treatment | 15.9 | – | 3250 | 295 | – | – | – | – | – | – |
| Pisgah | Pretreatment | 32.3 | – | 1712 | 1850 | – | – | – | – | – | – |
| | 4 months after treatment | 25.9 | – | 2295 | 888 | – | – | – | – | – | – |
| Nantahala National Forest (1500 to 1700 m) | Pretreatment | – | 26.8 | – | – | 12 178 | 1545 | 5362 | 30 609 | 8776 | 6933 |
| | 1 year after treatment | – | 18.7 | – | – | 409 | 913 | 1873 | 28 449 | 7726 | 1369 |
| | 2 years after treatment | – | – | – | – | 5 692 | – | – | – | – | – |

– = No data available.
Sources: Welch and others (2000), Elliott and others (1999), and Vose and others (1999).

and 40 to 75 percent in overstory-tree stem density (Elliott and others 1999, Vose and others 1999, Welch and others 2000). Despite a large initial reduction of density in the sapling layer, shrubs and hardwoods sprout even after these higher severity fires, potentially increasing densities in the years following fire (Welch and others 2000). Consumption of surface fuels is only 60 to 70 percent of the mass of small wood and litter, and the effects on the humus layer and coarse woody debris are minor (Vose and others 1999).

## In mixed pine-hardwood ecosystems

Studies on prescribed fire in mixed pine-hardwood ecosystems have shown the potential for large variations in fire intensity and severity from site to site (Waldrop and Brose 1999), likely driven by mountain laurel density and with the relative proportion of more flammable pine crowns and less flammable deciduous crowns. Hubbard and others (2004) reported flame lengths from 0.3 to 1.52 m; estimates from temperature-sensitive paints on ceramic tiles showed a maximum of 135 °C at 30 cm above the ground and 59 °C at 1.0 cm below the forest floor. Waldrop and Brose (1999) reported high fire intensity with crowning occurring on upper ridges.

Prescribed fires in mixed pine-hardwood forest ecosystems can also have highly variable effects on forest structure and soils (table 3). Waldrop and Brose (1999) documented the effects of this variation on stand structure, regeneration, and composition of the forest floor. Sites burning at low intensity had an average reduction in basal area of approximately 20 percent among trees >5 cm d.b.h. (diameter at breast height) compared to 96 percent for sites burning at high intensity. Decreases in the density of trees 2.5 to 4.9 cm d.b.h. ranged from 40 percent in low-intensity plots to 99 percent in high-intensity plots. Although all stems smaller than 2.5 cm d.b.h. were killed, hardwood regeneration was abundant in all sites regardless of intensity. Pine regeneration varied among fires and was highest at medium-low intensity and lowest at medium-high

**Table 3.** Effects of prescribed fires on live and surface fuels in two Southern Appalachian mixed pine-hardwood ecosystems, (1) in March 2001 on the Chattahoochie National Forest in Georgia and Cherokee National Forest in Tennessee, and (2) in April 1997 on the Chattahoochie National Forest in Georgia.

| Site (elevation) | Severity | Measurements taken | Basal area ≥5 cm | Basal area ≥3 m tall | Density >0.5 m tall | Density ≥0.5 m tall, but <5 cm | Density 2.5 to 4.9 cm | Density ≥5 cm | Litter | Humus | Woody debris <5 cm | Woody debris ≥5 cm |
|---|---|---|---|---|---|---|---|---|---|---|---|---|
| | | | ------m²/ha------ | | ------------------stems/ha------------------ | | | | -------------------kg/ha------------------- | | | |
| Chattahoochie/ Cherokee[a] (260 to 415 m) | Low | Pretreatment | 31.1 | – | 68,480 | 9,100 | – | 1,485 | 6028 | 11 435 | 6906 | 7611 |
| | | After 1 year | 28.8 | – | 138,120 | 5,900 | – | 1,362 | 1833 | 10 837 | 4425 | 6696 |
| | | After 2 years | 23.9 | – | 113,740 | 9,525 | – | 1,150 | – | – | – | – |
| Chattahoochie (885 to 1100 m) | Low | Pretreatment | – | 28.3 | – | – | 95[b] | 716[b] | – | – | – | – |
| | | After 3 months | – | 22.7 | – | – | 0[b] | 430[b] | – | – | – | – |
| | Medium Low | Pretreatment | – | 34.5 | – | – | 200[b] | 847[b] | – | – | – | – |
| | | After 3 months | – | 11.1 | – | – | 0[b] | 177[b] | – | – | – | – |
| | Medium High | Pretreatment | – | 17.4 | – | – | 105[b] | 775[b] | – | – | – | – |
| | | After 3 months | – | 1.6 | – | – | 0[b] | 45[b] | – | – | – | – |
| | High | Pretreatment | – | 27.0 | – | – | 110[b] | 776[b] | – | – | – | – |
| | | After 3 months | – | 1.0 | – | – | 0[b] | 6[b] | – | – | – | – |

– = No data available.
[a] Between the first and second year after burning, several sites were impacted by southern pine beetles, so changes are not wholly attributable to fire.
[b] Exact values were not reported but were estimated for this summary based on tables.
Sources: Elliott and Vose (2005), Hubbard and others (2004), and Waldrop and Brose (1999).

and high intensity. Regardless of intensity, consumption on the forest floor was limited to litter, with little consumption of humus or exposure of mineral soil. Other studies in mixed pine-hardwood ecosystems have found similar results of low-intensity prescribed fire on forest structure and the forest floor (Elliott and Vose 2005, Hubbard and others 2004).

## In mixed mesophytic hardwood ecosystems

Mixed mesophytic hardwood ecosystems commonly occupy sheltered sites with high moisture, and thus tend to burn at lower intensity during prescribed fires. Although mountain laurel may be present, rhododendron and mesic-hardwood saplings are generally the most abundant live fuels. Because the fire risk is lower compared to other Southern Appalachian ecosystems, fuel treatments in mixed mesophytic hardwood ecosystems may be of low priority to forest managers. As a result, studies and observations on prescribed fire in this type are limited. Available observations report that intensity is substantially lower than in other ecosystems—temperatures at 1 to 2 m above the ground were consistently below 52 °C; and on average, temperatures of 49 °C penetrated an average of only 0.5 mm into the ground (Vose and others 1999).

Low intensity prescribed fires in mixed mesophytic hardwood ecosystems have little effect on live fuels in the overstory and midstory (table 4). Elliott and others (1999) found no overstory mortality; although stems in the midstory were killed, their presence was maintained after the fire by vigorous sprouting, and the effect on surface fuels was small (Vose and others 1999). The effect on the mass of coarse woody debris, small wood, and litter was small, but the mass of the humus layer increased.

**Table 4.** Effects of prescribed fires on live and surface fuels in a mixed mesophytic hardwood forest ecosystem in April 1995 in the Nantahala National Forest in North Carolina.

| Site (elevation) | Measurements taken | Basal area | Density | | | | Woody debris | |
|---|---|---|---|---|---|---|---|---|
| | | ≥5 cm | 1 to 4.9 cm | ≥5 cm | Litter | Humus | <7.5 cm | ≥7.5 cm |
| | | ----*m²/ha*---- | -----------*stems/ha*----------- | | ----------*kg/ha*---------- | | | |
| Nantahala (1500 to 1700 m) | Pretreatment | 27.7 | 2,153 | 1,167 | 4 151 | 11 038 | 3 560 | 15 720 |
| | 1 year after treatment | 27.8 | 2,652 | 1,117 | 4 028 | 13 410 | 3 231 | 15 596 |

Sources: Elliott and others (1999) and Vose and others (1999).

### In white pine-hemlock-hardwood ecosystems

Although white pine-hemlock-hardwood forest ecosystems generally occur on moist sites, research suggests that fires of moderate intensity can occur, particularly in areas with thick layers of ericaceous shrubs. Clinton and others (1998) found that flame lengths range from 0.3 to 1.5 m for backing fires and from 1.2 to 4.5 m for head fires. The rate of spread varied from 1.8 to 3.0 m/minute for head fires to 0.3 m/minute for backing fires. Maximum flame temperatures ranged from 260 to 704 °C. On average, about half the mass of small wood (<8 cm d.b.h.) and litter was lost, and about 20 percent of the humus layer was lost.

Burning can be overly damaging to white pine because the species has thin bark a crowns low to the ground, particularly when young.

### Limitations of prescribed fire

Future use of prescribed fire may be reduced by smoke management requirements, lack of fiscal resources, operational complexities within the wildland-urban interface, and concern for litigation arising from smoke impacts or prescribed fire escapes. In the absence of prescribed fire, fuels continue to accumulate, making the application of alternative treatments necessary. These methods primarily include mechanical or chemical treatment alone or in combination of mechanical with prescribed fire.

## *Other Fuel Management Techniques*

### Mechanical treatment

Although the use of mechanical fuel reduction treatments is currently limited, they may be useful alternatives in areas where the risks associated with prescribed fires are unacceptable. Mechanical treatments may lack many of the ecological benefits of fire and are typically more expensive to apply. In the Western United States, mechanical fuel treatments usually include some degree of thinning followed by various methods of yarding and treatment of residual slash, possibly with prescribed fire (Youngblood and others 2007). Because mechanical treatment of Appalachian forest fuels has been limited, not much historical information is available on its effects.

Recent results from one site of the National Fire and Fire Surrogate explicitly addressed effectiveness of mechanical fuel treatments (Waldrop and others 2007) as well as providing a detailed look at the effects of fuel-reduction treatments on forest structure in western North Carolina. The mechanical treatment involved chainsaw felling of stems >1.8 m tall and <10.2 cm d.b.h. and of all shrubs regardless of size. In addition, two prescribed fires—one with and the other without the mechanical treatment—were conducted at a 3-year interval. After 5 years the mechanical treatment alone had no effect on basal area and structure of overstory trees. Density of hardwood saplings decreased initially but slowly returned to levels similar to pretreatment levels as a result of vigorous sprouting. Shrubs—including the dominant shrub species, mountain laurel and rhododendron—experienced a large initial decrease and recovered to less than half their pretreatment abundance by year five.

The combination of mechanical treatment with prescribed fires reduced basal area from 23.8 to 16.6 $m^2$ /ha after 5 years. Density of hardwood saplings decreased initially but was >100 percent of the pretreatment level after 3 years; a second prescribed fire reduced hardwood sapling density to just slightly higher than the pretreatment levels 2 years later. Cover of all shrubs, including mountain laurel and rhododendron, initially decreased to near zero after the mechanical and burn treatment and remained at very low levels until year five.

### Chemical treatment

Herbicides have been studied in the Southern Appalachian Mountains for competition control to favor pines and oaks (Kass and Boyette 1998, Loftis, 1985, Lorimer and others 1994, Neary and others 1984) and for habitat of some wildlife species, including small mammals (McComb and Rumsey 1982) and herpetofauna (Harpole and Haas 1999). However, no study has examined herbicide use for fuel reduction in the area. This treatment may be viable where fire or mechanical treatments are impractical—such as along the wildland-urban interface or on steep inaccessible slopes—but its impacts are unknown. Studies in the pine flatwoods of Florida (Brose and Wade 2002) and in Gulf Coast longleaf pine (Haywood 2009) show short-term increases in fuel loading, which led to increases in fire intensity and damage. Similar results could occur in the Southern Appalachian Mountains, although differences in species composition make impacts difficult to predict. Waldrop and others (2010) showed increased fire intensity for 5 years after chainsaw felling shrubs and small trees in the Southern Appalachians. Although untested, a similar pattern would likely occur if herbicides had been used instead.

## *Combining Fuel Treatments*

Despite the absence of a large body of information from different ecosystems on the effects of fuel- treatment alternatives to fire, the existing literature offers some evidence of differences in the effectiveness of treatment options. Results from the National Fire and Fire Surrogate Study in oak ecosystems strongly suggest that combining removal of shrubs and small trees with prescribed fire is the most effective way to control mountain laurel and other ericaceous shrubs, a fuel of concern in the Southern Appalachian Mountains (Youngblood and others 2007). Studies from a variety of ecosystems consistently demonstrate the ability of mountain laurel and other ericaceous shrubs to increase rapidly after treatment. Whether these fuels respond to mechanical and burn treatments in other ecosystem types is unknown. The ubiquity of mountain laurel and other ericaceous shrubs across the landscape suggests that the response of different ecosystems could be similar. However, interactions with other variables, such as moisture patterns and disturbance regimes, could produce different responses. Results from future studies in other ecosystems could shed light on whether combining mechanical and burn treatments is only effective in oak forests or could be useful across the landscape.

The feasibility of widespread application of a mechanical plus burning treatment is questionable because treating large areas is expensive and time consuming. In addition, mechanical treatments alone are not effective, so the risks associated with prescribed burning are still a factor. Although there is no one solution, the use of mechanical treatment may be most useful in areas with immediate needs for hazardous fuels treatment, such as the wildland-urban interface. Also, mechanical treatments can be very effective in preparing long unburned sites for prescribed burning. Clearly a manager must be flexible and open to cautiously experimenting with different combinations of techniques, drawing on experience and observation until more experimental data become available.

Prioritizing areas for treatment is critical to allocate resources most effectively. Fuel treatment prioritization hinges on managers making decisions that will protect vital assets without decreasing the amount of acreage that is in an acceptable condition. For example, a best management practice would be to focus initial efforts on maintaining areas that currently have low fuel loads and are simple to burn, and only afterward allocating resources to problem areas so that the total amount of untreated acreage does not increase. A burn prioritization model can streamline treatment programs and be useful for mapping current conditions and designating treatments within a spatial context (Hiers and others 2003).

The diversity and productivity of ecosystems in the Southern Appalachian Mountains coupled with a complex disturbance regime poses a challenge for fuels management. Understanding this relationship will better enable managers to understand the dynamic interactions among disturbances, which can alter fuel loads over short periods of time. Although rates of decomposition across the area are rapid, increases in dead and downed fuels following disturbance may create pulses in the abundance of hazardous fuels. In this situation, understanding the variation in the fuel distribution over time may be as important as understanding spatial variations. This is especially pertinent in the context of climate change scenarios that predict more frequent droughts and warmer temperatures that could exacerbate the effects of disturbances such as native and nonnative insects and pathogens. These effects could be especially important in long-unburned mature stands that contain older decadent individual trees and well developed shrub layers. Effective mitigation of these threats depends on effective fuels monitoring at large scales and adaptive management to meet future challenges.

## Research Needs

Prescribed burning is a relatively new tool in the Southern Appalachian Mountains. As a result, less is known about fuel reduction treatment impacts in this area than is known in other areas of the United States. Critical research needs include the studying the impacts of mechanical and chemical treatments, comparing season and frequency of prescribed burning, and identifying the cumulative effects of repeated fuel reduction treatments over many years. More information is needed to understand the impacts of these treatments on most components of the ecosystem—biotic and abiotic—and the probability of introducing new and possibly unwanted components, such as nonnative invasive plants.

Research on smoke prediction in the Southern Appalachian Mountains is just beginning and is extremely difficult because of the complex topography and weather patterns that must be considered.

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Cumulative Watershed Effects of Fuel Management in the Eastern United States

CHAPTER 7.

# Fuel Management in the Subtropical and Savanna Divisions

**Kenneth W. Outcalt**

The Subtropical Division (230) and Savanna Division (410), both based on Bailey's (1996) ecoregions, are found in the Southern United States (http://www.na.fs.fed.us/fire/cwedocs/map%20new_divisions.pdf). The Subtropical Division occupies the southern Atlantic and Gulf coastal areas. It is characterized by a humid subtropical climate with hot humid summers (chapter 3). It has no pronounced dry season but precipitation is normally higher during summer. Soils are strongly leached and rich in iron and aluminum oxides. The natural vegetation throughout much of the Subtropical Division is forest. It includes the Outer Coastal Plain Mixed Forest Province, the Southeastern Mixed Forest Province (which occupies the inner Coastal Plain area), and the Lower Mississippi Riverine Forest Province (McNab and Avers 1994).

The Savanna Division is part of the humid tropical domain. In the Eastern United States, it is found only in southern Florida represented by the Everglades Province. It has a hot wet season driven by warm maritime air masses followed by a dry period during the somewhat cooler low sun angle months (Bailey 1996). Soils are mostly organic histosols and sandy inceptisols. The natural vegetation is tall grasses and drought resistant trees and shrubs. The Savannah Division was covered with wet and dry prairie, cypress swamps (*Taxodium distichum*), pine flatwoods (*Pinus* spp.) and rocklands, hardwood and palm hammocks (*Acoelorrhaphe* spp.), and subtropical hardwoods.

A number of different forest and nonforest ecosystems historically occupied the Subtropical Division. Before European settlement, the area was mostly forested; and although many lands were cleared for agricultural and urban uses, forests currently occupy about 60 percent of the area (Conner and Hartsell 2002). Pines dominated the frequently burned forests of the lower and middle Coastal Plains. Other forest communities like cypress and hardwood hammocks were imbedded in this pine matrix. The Piedmont, which lies at a northeast to southwest direction between the Coastal Plain and the Appalachian Mountains, was a mixture of pines, pine-hardwood forests, and the oak-hickory type (*Quercus* spp.–*Carya* spp.). Mesic hardwoods occupied the river terraces and richer bottomlands.

The Subtropical and Savanna Divisions have all of the fire regimes described by Brown (2000), with much of the area burning quite frequently (Frost 2006). Many of the ecosystems had an understory fire regime with frequent low-intensity surface fires that consumed surface fuels but left the overstory unharmed. The fire-return interval in these systems ranged from 1 to 12 years. Because of this frequent fire, the forested areas tended to be open with grass and herbaceous dominated understories. Other forest communities had mixed-severity fire regimes with less frequent but more intense fires that killed a substantial portion of the overstory (Wade and others 2000). Other ecosystems had a stand replacement fire regime with periodic intense fires, which killed the overstory but created conditions that favored regeneration.

Marshes and prairies covered quite extensive areas and were maintained by frequent fires. Because the aboveground portion of the dominant life form was killed, these were mostly stand replacement fires. Now they are often burned for ecological benefits rather than fuel management, although fuel management is also important in some situations. Other systems like mixed mesophytic hardwoods, bottomland hardwoods, and subtropical hardwoods normally had a regime with little or no naturally caused fire. Because fires are rare in these systems, fuels management is not needed and they are excluded from this chapter.

The first portion of this chapter briefly describes the former and current extent and fire regime for each major community where fuel management is applied in the Subtropical and Savanna Divisions. This background information on the systems where fuel management is applied is presented by fire-regime type. The second part of the chapter is a discussion of the most often used fuel management techniques in these communities. This is not meant to be a comprehensive prescription of how to apply these techniques, but rather to put into context the potential for cumulative impacts from the different treatments. Those needing more detailed information on using these techniques should consult cited references and additional resources (such as frames.nbii.gov).

# Major Ecosystems

## *Understory Fire Regime*

### Longleaf pine

*Pinus palustris* was once the most prevalent pine in the Subtropical Division (Frost 2006), where it dominated 60 million acres (25 million ha) and was a codominant with shortleaf (*Pinus echinata*) and loblolly pines (*Pinus taeda*) on another 30 million acres (12 million ha). Its range extended south from southeastern Virginia to central Florida and west into eastern Texas (Stout and Marion 1993). Longleaf pine was native to a wide range of ecosystems including wet flatwoods and savannas along the Atlantic and Gulf Coastal Plain and higher droughty sand deposits from the fall-line sandhills to the central ridge of Florida. Longleaf pine also grew on more productive upland sites like the red hills area of southern Georgia and the loamy soils of Alabama and Louisiana (Stout and Marion 1993). Longleaf pine even extended onto the mountain slopes and ridges of Alabama and northwestern Georgia (Boyer 1990), where it was found growing at elevations up to 2000 feet (600 m).

Logging of the valuable longleaf pine forests, which began in early settlement times, reached a peak shortly after 1900 (Ware and others 1993). Clearing of forest land for urban and agricultural uses, conversion to loblolly pine (*Pinus taeda)* and slash pine (*Pinus elliottii*) plantations, and harvesting without regeneration all contributed to the continuous decline of this once dominant forest community. Only about 20 million acres (8.1 million ha) of longleaf pine forest remained by 1935 (Wahlenberg 1946), declining to 12 million acres (4.9 million ha) in 1955, 3.7 million acres (1.5 million ha) in 1985 (Kelly and Bechtold 1990), and 3 million acres (1.2 million ha) in 1995 (Outcalt and Sheffield 1996). Longleaf dominated forests recently have been increasing on public lands including the national forests, which contained 820,000 acres (332 000 ha) or 25 percent of remaining longleaf forests in 2006.

In the period before landscape fragmentation, extensive naturally caused fires occurred every 2 to 8 years across much of the South (Abrahamson and Hartnett 1990, Christensen 1981, Ware and others 1993). Mixed pine stands were found along the northern and western edges where fire-return intervals were longest. Longleaf pine dominated much of the rest of the landscape because it was more tolerant of these frequent fires than the thinner barked seedlings of loblolly and slash pine, which lacked a fire resistant grass stage (Chapman 1932). Some have argued that longleaf not only needed fire for site domination, but that it actually perpetuated frequent surface fire through the production of long flammable needles that—as litterfall—promoted the spread of frequent surface fires (Landers 1991). Another important component of the fuel matrix in longleaf communities were the grasses, whose living and dead leaves intercepted the shed needles of overstory pines, causing an accumulation of dead biomass in a very flammable configuration. Wildfires spread quickly through this fine-fuel matrix (Abrahamson and Hartnett 1990). They also spread into embedded communities such as seepage slopes, savannas, and canebrakes. Without longleaf to propagate fire on the landscape, these systems degrade and lose their diversity.

Lightning and Native Americans provided the ignition sources for the fires that shaped the vegetation in longleaf communities (Komarek 1968, Robbins and Myers 1992). Seasonal lightning activity is quite variable and weather driven. In central Florida, 75 percent of all annual strikes occur in the summer months of June, July, and August (Hodanish and others 1997); but lightning can occur anytime. Lightning frequency, however does not equate with fire ignitions or the area burned in longleaf ecosystems. The spring months of April and May are often the driest; but although strikes are less frequent in these months, because fuels are dry and precipitation with storms often limited, ignition probabilities are highest. This combined with low humidity and winds that often occur during these months should lead to larger fires from lightning ignitions. This agrees with data for area burned by lightning fires on national forests in Florida, which was greatest during May (Robbins and Myers 1992). Thus, the historical fire regime was one of frequent low-intensity fires burning across vast expanses predominantly during the early growing season but augmented by Native American ignitions during the dormant season.

## Slash pine flatwoods

Slash pine is native to the lower Coastal Plain from Georgetown County in South Carolina to Tangipahoa Parish in Louisiana and most of peninsular Florida south to Ft. Lauderdale (Lohrey and Kossuth 1990). It historically dominated the seasonally wet to flooded woodlands; on nearly level, poorly drained sandy soils with dark sandy layers (mostly Spodosols) or clay hardpans (Ultisols) and generally low pH (<4.5). Although dominated by slash pine, these flatwood sites contained some longleaf pine on the dryer fringes where they graded into the longleaf wiregrass (*Aristida beyrichiana*) flatwoods and some swamp blackgum (*Nyssa sylvatica* var. *biflora*) and pond cypress (*Taxodium distichum* var. *nutans*) in the transition zone to wetlands. The understory consisted of evergreen shrubs and trees with saw palmetto (*Serenoa repens*), gallberry (*Ilex glabra*), fetterbush lyonia (*Lyonia lucida*), and loblolly bay (*Gordonia lasianthus*) common dominants. Herbaceous species were sparse, occurring as grasses and forbs scattered among shrubs. Since longleaf pine did not occur south of Lake Okeechobee in Florida, those flatwood forests were dominated exclusively by slash pine (Little 1971). Schultz (1983) estimated the original slash pine flatwood area at about 7 million acres (2.8 million ha) with the largest concentration in Florida and southern Georgia.

These flatwood forests have been heavily impacted over the last three centuries. Much of the slash pine and mixed slash and longleaf pine were cutover from 1780 to 1860. These areas were first logged because they were accessible by water, which was needed to raft logs to the mills (Schultz 1983). Many were logged a second time, along with higher longleaf flatwoods, by crews using temporary railroad spurs and steam skidders from 1870 to 1920. Because slash pine is a prolific seed producer, it rapidly colonized cutover areas and abandoned fields including many areas formerly dominated by longleaf pine (Schultz 1983). Fire control contributed to the increase in slash pine relative to longleaf as it allowed trees to make it through the fire-sensitive seedling stage. Once it became profitable to harvest and use small southern pine for Kraft pulp forestry became more intensive, often resulting in postharvest conversion to plantations on heavily site-prepared areas and dramatically altered understories (Schultz 1976). By 1980, 52 percent of all slash pine stands were plantations (Sheffield and others 1983), and this trend has continued. Today slash pine occupies nearly 10 million acres (4 million ha), with 70 percent in privately owned plantations (Miles 2007).

The likely presettlement fire regime for slash pine flatwoods is frequent surface fires every 4 to 6 years, ranging to <8-year intervals or longer in the wettest pond sites. Most of these fires begin in the dryer longleaf wiregrass flatwoods and then carry into the adjoining slash pine flatwood areas if they are dry enough to burn. They generally burned through the understory vegetation but only consumed the dry upper portion of the litter layer. These fires were usually moderate in intensity but during extended drought periods, which occur about every 25 years, could be quite severe because the entire forest floor was dry enough to burn. When this happened, overstory mortality was often high with either total replacement or substantial thinning. Although fire could occur in any season, in presettlement times many lightning fires probably occurred from the dry late spring to early summer. Specific months varied with latitude but were generally from mid-April to June. Once the summer thunderstorm season began, these areas soon became too wet to burn. Native Americans augmented this by setting fires during dry periods of the dormant season.

## Loblolly pine

Loblolly pine has an extensive range stretching from New Jersey along the coast to eastern Texas and inland through the Piedmont to Tennessee and Arkansas. Although loblolly pine is able to grow across a wide area on many different sites, Schultz (1997) estimated that it dominated <5 million acres (2 million ha) of presettlement forests. It was often a co-dominant of uplands with longleaf and shortleaf pine (*Pinus echinata*), or a minor species in upland hardwoods. It was also found as scattered individuals or small groups on river bottoms and swamps growing among the bottomland hardwoods. It was most prevalent on moist sites that burned less frequently than adjoining longleaf-dominated habitats. Nearly pure stands of loblolly pine did exist, primarily through establishment following major disturbance from fire or wind (Skeen and others 1993).

Agricultural clearing and logging by settlers dramatically changed the southern landscape. Loblolly pine is a prolific seed producer that grows quite rapidly on a variety of sites. It was very successful at capturing many former longleaf sites following logging (Schultz 1997). This has been aided in more recent times by fire control measures that give loblolly seedlings an advantage. Loblolly pine was very successful at seeding into abandoned cotton fields, thus the common name of old-field pine, and was also widely established in forest industry plantations. The result was that loblolly replaced longleaf as the most prevalent of the southern yellow pines. By 1989 it had become the most important timber species in the United States (Schultz 1997), dominating 33 million acres (13.4 million ha). Today loblolly pine dominates 46 million acres (18.5 million ha) with 60 percent growing in plantations, many established after intensive site preparation (Miles 2007).

Loblolly pine growing in bottomland areas seldom experienced fire but the uplands of the South burned with some regularity. Low-intensity surface fires occurred every 4 to 12 years (Frost 2006) on these dryer upland locations. Although seedlings <5 years old can be killed by fire, older trees are quite resistant to low-intensity surface fires (Schultz 1997). Less frequent stand replacement wildfires likely occurred at least every 100 years somewhere in a watershed. In fact, loblolly pine was maintained in pure stands by both frequent low-intensity surface fires that kept hardwood competitors in check and periodic severe fire, which created open areas for regeneration. As with other southern pine species, growing season fires were common.

## Shortleaf pine

Shortleaf pine has the widest range of any southern pine growing in 22 States from southeastern New York to Florida and west to Texas and inland through Pennsylvania to Ohio and Missouri (Little 1971). It is found on the Coastal Plains and Piedmont in the Subtropical Division, and the Interior Highlands in the Subtropical Mountain Division (M230). Like loblolly, a common associate, it is adapted to a wide variety of soil types. Historically it dominated the drier sites west of the Mississippi in Arkansas, Louisiana and eastern Texas. Where ranges overlapped, loblolly pine dominated the moister soils and shortleaf was more prevalent on drier sites (Wade and others 2000). In addition to associations with longleaf and loblolly previously noted, shortleaf was also found in mixtures with pitch (*Pinus rigida*) and Virginia (*Pinus virginiana*) pines in the Northeastern States (Lawson 1990). It was often also found in mixed hardwood stands where it shared dominance with oaks and hickories.

Mattoon (1915) noted that agriculture and logging in the early 1900s produced a decline in shortleaf pine. With the onset of logging in the Ouachita Highlands of Arkansas, substantial declines continued through the 1950s (Smith 1986). Often cutover stands were planted with loblolly pine—even north of its native range and in forest industry operations—which has contributed to a continued loss of shortleaf dominated forests (Guldin 1986). Loblolly has also replaced shortleaf in the eastern part of the range on sites where littleleaf disease (*Phytophthora cinnamomi*) was common because it is less susceptible (Campbell and others 1953). Shortleaf is still quite widespread but is often a minor component of forest stands. It is the dominant overstory tree on 3.2 million acres (1.3 million ha) in mostly naturally regenerated stands and on just 250,000 acres (100 000 ha) of plantations.

Shortleaf is very tolerant of fire. It is a prolific seed producer that forms dense seedling stands that have rapid juvenile growth (Mattoon 1915). Trees >5 feet (1.5 m) tall will survive surface fires unless crown scorch exceeds 70 percent (Wade and others 2000). If young trees are topkilled, they will readily sprout. Older trees—those larger than 4 inches (10 cm) at d.b.h.—have thick bark that protects the bole from surface fires (Walker and Wiant 1966). Because of these characteristics, frequent low-intensity fires give shortleaf a competitive advantage over many hardwoods. The historical fire regime was frequent low-intensity surface fires every 4 to 6 years (Frost 2006). This has also been shown to be the optimal interval for prescribed burns to promote natural regeneration (Masters and others 2005).

Lightning varies considerably in both frequency and seasonal peaks across the broad range of shortleaf pine. It is most frequent on the Coastal Plain, with an early growing season to summer maximum. The northern and western portions of the shortleaf range experience much less lightning in a bimodal distribution, with both a spring peak and a late-summer to early-autumn peak (Masters and others 1995). This ignition source was certainly augmented by Native Americans, which fostered the open grass dominated shortleaf stands by increasing fire frequency (Vogl 1972).

## Oak-hickory-pine woodlands

This community is equivalent to Kuchler's (1964) oak-hickory-pine type 111. It was composed of a mixture of species in the overstory with the unifying characteristics of fire resistance. Its historical extent is not known, but it was widespread and prevalent in the Piedmont, upper hilly Coastal Plains, and Interior Highlands. The predominant group was the oaks (Sander and others 1983) including white (*Q. alba*), northern red (*Q. rubra*), and black (*Q. velutina*); and on drier steeper sites scarlet (*Q. coccinea*) and chestnut (*Q. prinus*). On more southerly sites post (*Q. stellata*), blackjack (*Q. marilandica*), bluejack (Q. *incana*), and southern red oak (*Q. falcata*) were common. Hickories included pignut (*Carya glabra*), mockernut (*Carya tomentosa*), shagbark (*Carya ovata*) and bitternut (*Carya cordiformis*). The pine component, when present, was loblolly, shortleaf, pitch, Virginia, or white (*Pinus strobus*). Often, more mesic hardwoods were present, such as yellow-poplar (*Liriodendron tulipifera*), black cherry (*Prunus serotina*), ash (*Fraxinus* spp.), and elms (*Ulmus* spp.). The pine component owed its existence to natural disturbances, such as fire and wind, or to extremely poor site conditions (Skeen and others 1993).

This community historically covered most of the Piedmont. It was greatly reduced by agricultural clearing but rebounded following soil depletion and abandonment, which fostered an increase in pines relative to hardwoods (Boyce and Knight 1980). As previously noted, loblolly and some shortleaf pine seeded into and captured a substantial number of old-field sites across the South. Significant quantities of oak and hickory were also harvested for lumber and to provide charcoal for the iron smelters that sprang up across the South. Frequent fires, which occurred until effective fire control was implemented in the 1950s, also favored pines. Once fire control was instituted and pine stands began to be harvested, pine coverage declined as hardwoods captured many sites following commercial clearcuts (Boyce and Knight 1980). More recently, significant increases in population in the Piedmont have impacted this forest community. However, the South still contains about 32 million acres (13 million ha) of oak-hickory-pine forest (Miles 2007).

The historical fire frequency in this community was 4 to 6 years with equal ignitions from lightning and Native Americans (Frost 1998). Lighting ignitions were most prevalent on more exposed and drier ridge tops and southern and western slopes. These coincided with the lightning season, which ran from March to October, but were most common during the dry spring. Native Americans also burned significant areas during the late autumn dry period. Their ignitions were more important toward the interior where the landscape is most dissected and less exposed to extensive fires from lightning. Low-intensity fires kept the forests open and favored oaks and pines (Skeen and others 1993). Early settlers continued to burn the woods to provide forage for their livestock. More recently, fires have been mostly prescribed burns to control fuel buildup, favor oaks and pines, and improve wildlife habitat.

## Pine rocklands

This community is native to southern Florida, the Bahamas, and Cuba. In southern Florida it once occupied 180,000 acres (72 900 ha) on the Miami Rock Ridge from north of Miami to Homestead and southwest through Long Pine Key in Everglades National Park (Davis 1943). It was also found on the lower Florida Keys and the southeastern portion of Big Cypress National Preserve around Pinecrest (Snyder and others 1990). It occupied the higher elevations formed by outcrops of marine limestone, thus the term rocklands. Vegetation actually grows on the bedrock, rooting within the rocky rubble in thin layers of sand, marl, and organic matter that have accumulated in depressions, crevices, and solution holes. In the lower Florida Keys, more than half of the ground can be exposed rock; in Big Cypress, most of the limestone has a thin covering of sand. The overstory was south Florida slash pine (*Pinus elliottii* var. *densa*), growing in open canopy stands over an extremely diverse understory of tropical and temperate shrubs, palms, grasses, and forbs that included many local endemics (Snyder and others 1990).

Because these pine forests were found on higher and dryer land, they were the first to be cleared for building sites beginning around 1900. Logging was limited until the railroad arrived in 1896, but then most of the pine suitable for harvest was cut over the next half century (Snyder and others 1990). Cleared pinelands were used mainly for citrus production until the 1954 introduction of the rock plow to breakup the limestone bedrock and allow large-scale row crop farming. Fragmentation and fire control have also led to succession of pine forests to hardwood hammock (Stout and Marion 1993). The combination of an expanding urban area, logging, and agricultural clearing reduced the pine rockland forest substantially. Today only 2 percent of the original pine forest remains in the Miami area. Significant areas of intact forest only exist on public lands in Everglades National Park, the Big Cypress National Preserve, and the National Key Deer Refuge on Big Pine Key.

Pine rocklands need periodic fire to control growth of hardwood species, keep the stand open, and foster pine regeneration. These forests accumulate slowly decomposing needles, which are kept from matting by the rough rock surface and understory vegetation. The rocky porous substrate allows rapid drainage, and the open pine canopy fosters rapid drying—characteristics that lead to frequent low-intensity surface fires that consume the litter and understory vegetation (Snyder and others 1990). Pine canopies are too open to support crown fires, and the thick bark of mature trees protects the cambium of lower trunks (Hare 1965). Hofstetter (1973) reported that saplings 6.5 to 20 feet (2 to 6 m) tall have a 50-percent survival rate following fire. Seedlings have a grass stage where long needles protect the central bud and can sprout from the root collar if topkilled (Ketcham and Bethune 1963). The aboveground portion of woody understory species and saw palmetto are often consumed or killed by fires but they quickly resprout. Grasses and forbs respond with rapid regrowth and flowering (Robertson 1962). Thus, fire in pinelands does not cause significant changes because the species are predominantly perennials that can rapidly recover.

Historically, fires in pine rocklands were low-intensity surface fires occurring every 2 to 15 years with most areas burning every 3 to 7 years (Hofstetter 1973). Lightning is frequent in southern Florida and was the primary ignition source, often starting fires in wet prairies that swept into adjoining pinelands. Lightning ignitions occurred from May to October during the rainy season but fires were most extensive in late May and June at the end of the dry season before water levels rose (Snyder and others 1990). Native Americans certainly augmented natural ignitions and likely burned at other times outside the normal lightning season. Since 1950, human caused wildfires have been most frequent and burned the most area in April and May.

## *Mixed Severity Fire Regime*

### Pitch pine

Within the Subtropical Division pitch pine is native to Coastal Plain areas of Maryland and Delaware through southeastern New Jersey but is also found in the Hot Continental Division (220) on Long Island and Cape Cod (Little and Garrett 1990). It was most common on infertile soils including sands and gravels deposited on glacial outwash plains or as alluvial or marine sediments. The Pine Barrens of New Jersey contained the largest concentration of pitch pine growing on glacial deposits ranging from excessively to poorly drained sands and gravels. The historical extent of pitch pine is not known, but the Pine Barrens alone contained >1.1 million acres (450 000 ha), where pitch pine was likely a major overstory species in many historical communities. Depending on site and fire history, trees ranged from 39 feet (12 m) tall in more fertile swamps to dwarfsize, <11.5 feet (3.5 m), on the driest most frequently burned sand plains. Common associates included chestnut oaks, white oaks, black oaks, northern red oaks, bear oaks (*Q. ilicifolia*), and Virginia pine with an understory of woody species like bear oak, dwarf chinkapin oak (*Q. prinoides*), blueberries (*Vaccinium* spp.), and huckleberries (*Gaylussacia* spp.).

Nearly all lands in the Northeastern United States have been impacted by 400 years of human use, which included clearing, cultivating, grazing, logging, and burning. Initially fire frequency also increased, associated with land clearing (Parshall and others 2003). Because the pitch-pine barrens were most prevalent on infertile soils, they were rarely plowed—especially in the New Jersey Pine Barrens—but were heavily harvested for firewood, fence posts, railroad ties, and building material (Howard 2003). Beginning in the mid-1800s, many cleared areas were abandoned and pitch pine became established on former pine barrens (Motzkin and others 2002). More recently, aggressive fire control has caused a decrease in open pitch-pine stands and an increase in oak and other hardwoods (Copenheaver and others 2000). Although pitch-pine dominated forests are still common in the barrens, the absence of fire has changed them (Hall and others 2002). The need for prescribed fire is widely recognized, but the practice is becoming increasingly difficult because of fragmentation from residential development (Jordan and others 2003).

Barrens were historically dominated by pitch pine because it is very fire adapted. A thick bark protects it from fire. Buds on the bole can produce new foliage. Additional adaptations are the ability to sprout, serotinous cones that release seed following fire, and cone production at very young ages—3 to 4 years—for sprouts (Little and Garrett 1990). The large flat expanses of droughty soils allowed fires to easily propagate across the landscape, thereby increasing burn frequency. Thus, the central part of the Pine Barrens where dwarf pine is common, are estimated to have a historical fire-return interval of mostly stand replacement fires every 6 to 8 years (Givnish 1981). The more isolated areas of pitch pine and scrub oak (*Q. berberidifolia*) likely burned every 15 to 25 years with fires that killed a portion of the overstory. Native Americans burned areas of the barrens near their villages in spring and autumn on a 2- to 10-year interval with lower intensity understory burns, which produced open pitch pine stands with relatively large trees (Wade and others 2000). However, Parshall and Foster (2002) concluded that natural ignitions alone were sufficient to maintain the historical barrens.

## Sand pine

Sand pine (*Pinus clausa*) scrub historically occupied three areas in Florida, inland peninsula, coastal peninsula, and coastal panhandle (Myers 1990). Ocala sand pine (*Pinus clausa* var. *clausa*) was endemic to peninsular Florida, with the largest concentration on the central ridge. It occupied a large portion of what is now the Ocala National Forest (where it was referred to as the Big Scrub) and was once prevalent on the Lake Wales Ridge (Brendemuehl 1990). Historically, smaller patches of scrub were found along the coast on old dunes stretching from St. John's County south to the northern portion of Dade County on the east coast, and from near Cedar Key south to Naples on the west coast. Sand pine scrub is a xerophytic, evergreen plant community found on excessively well drained, nutrient poor entisols (deep droughty infertile sands of marine and aeolian origin) of the quartzipsamment classification. Ocala sand pine forests have an overstory of predominantly even-aged sand pine with twisted and leaning trunks growing over an understory of evergreen shrubs. Typical understory species include myrtle oak (*Q. myrtifolia*), sand live oak (*Q. geminata*), Chapman oak (*Q. chapmanii*), turkey oak (*Q. laevis*), rusty lyonia (*Lyonia ferruginea*), rosemary (*Ceratiola ericoides*), scrub palmetto (*Sabal etonia*), and saw palmetto. Herbs and grasses are very sparse in mature scrub habitats, but lichens (*Cladonia* spp.) can form extensive patches on the forest floor. Lake Wales scrub is very similar except it often has few or no emergent sand pine.

Choctawhatchee sand pine (*Pinus clausa* var. *immuginata*) was the dominate tree in scrubs growing on sandy soils along the Gulf Coast (including offshore islands) of northwestern Florida from the Apalachicola river westward into Alabama (Brendemuehl 1990). This scrub has an overstory dominated by sand pine with an occasional longleaf pine or large sand live oak. Regeneration is a continuous process, which results in a relatively large number of trees in the intermediate and suppressed crown classes and fewer dominants (Outcalt 1997). Midstory oaks were a prominent feature of these sand pine stands with sand live oak the most common. Beneath the midstory was a tall shrub layer dominated by sand pine regeneration, oaks, and lesser numbers of ericaceous shrubs. The forest floor was composed of mostly pine litter with a few herbs growing between patches of lichens.

Because of their droughty infertile soils, scrub habitats were used only infrequently by Native Americans and early settlers (Myers 1990). Later discoveries that they were well suited to citrus production caused many in the lower portion of Florida's central ridge to be cleared. Coastal scrubs in peninsula areas were converted to urban use as Miami, Tampa, and other major cities developed. More recently, extensive areas have been disappearing to housing developments, golf courses, and other urban uses. A large concentration of sand pine remains however, occupying >250,000 acres (100 000 ha) on the Ocala National Forest (Brendemuehl 1990).

Because of its poor form, Ocala sand pine was not commercially harvested until the Kraft pulp industry became well established in the 1950s. Since that time significant areas have been harvested, but were regenerated on public lands. Choctawhatchee sand pine was restricted to the coastal areas by frequent fires; a combination of harvesting and fire suppression has allowed it to capture many areas of former longleaf pine forest (McCay 2000). Significant areas have also been planted with Choctawhatchee sand pine across the Florida panhandle. Thus, there is more Choctawhatchee sand pine now than existed in presettlement times. Longleaf once covered an estimated 85 percent of Eglin Air Force Base but now occupies about 15 percent compared to sand pine, which has increased from 10 to 40 percent or 185,000 acres (75 000 ha).

Although Ocala sand pine scrub experiences primarily stand replacement fires every 10 to 35 years, some level of fire occurs at shorter or longer intervals. Because of its sparse ground cover and compacted litter layer, sand pine scrub is virtually fire proof much of the time. However, every 10 to 100 years—usually during the spring drought—high winds and extreme conditions result in a catastrophic wildfire (Hough 1973) that kills the sand pine overstory and burns off the understory (Myers 1990). The resulting heat opens the many serotinous cones contained in the crowns of the sand pine, which releases the seed for establishment of the next stand (Cooper 1951). Because it produces cones at 3- to 5-years old, even young stands can reseed burned sites. Occasionally in stands with sparse sand pine cover, less intense fires result in only partial overstory mortality. Historically, Choctawhatchee sand pine grew on coastal areas, where fires were rare and less intense because of the less flammable understory. Most fires in the panhandle scrub were understory or mixed, killing only a portion of the overstory. Unlike Ocala, most of the cones open when mature so seeds are shed annually and will reestablish in areas opened by fire caused mortality.

## Pond pine

*Pinus serotina* is native to the Coastal Plain from the southern tip of New Jersey south through the Delmarva Peninsula across the Carolinas and Georgia to central Florida and west into the southeastern corner of Alabama (Bramlett 1990). It once occupied a significant area of poorly drained sites. The largest concentration was in North Carolina, where pond pine was the dominant overstory on 2.5 million acres (1 million ha) of raised bogs (Richardson 1981) or pocosins, characterized by organic soils with sandy humus, peat, or muck surface horizons (Richardson and Gibbons 1993). Pond pine also grew in the wettest portions of woodlands, wet flatwoods, savannas, bay forests, shrub bogs, and swamps (Wade and others 2000) where it was often embedded in communities dominated by other southern pines, cypress, swamp conifers like Atlantic white-cedar (*Chamaecyparis thyoides*), and bottomland hardwoods like swamp tupelo (*N. biflora*) and sweetbay (*Magnolia virginiana*). Pocosins often have a thick understory layer of evergreen shrubs and smilax vines. Common shrubs species are gallberry, swamp titi (*Cyrilla racemiflora*), wax myrtle (*Myrica cerifera*), and coastal sweetpepperbush (*Clethra alnifolia*). Switchcane (*Arundinaria gigantea* ssp. *tecta*), which sprouts prolifically following fire, was abundant on some frequently burned sites (Bramlett 1990).

Pond pine was cut extensively like other southern pines during the major logging of the Southern United States from the 1800s to 1920. This logging however, was not as destructive as other operations: a large portion of the original pond pine pocosin habitat has been lost to peat mining, drainage, and conversion to pine plantations or row crops. In North Carolina just 695,000 acres (281 000 ha) of the pocosin remained undisturbed by humans in 1980 (Richardson and others 1981). Conversion to plantations and agricultural crops has continued to reduce pocosin habitat. In addition, a reduction in fire has allowed shrubs to increase in dominance at the expense of grasses (Frost 2002). Even though it is recognized that pocosins need periodic fire, the expansion of urban areas is making prescribed burning ever more difficult.

Pond pine is the most fire adapted of the Coastal Plain southern pines. It has the ability to sprout if topkilled and will produce new foliage from dormant buds under the bark following intense fires (Bramlett 1990). It also produces serotinous cones that store seed that is released following fire. The historical fire-return interval is highly variable with a range from 5 to 150 years. Wet flatwood and savanna sites have the shortest fire frequency of 3 to 10 years (Florida Natural Areas Inventory 1990), woodlands burn every 10 to 20 years (Sutter and Kral 1994), pocosins every 13 to 50 years (Frost 1995), and bogs and swamps every 50 to 150 years (Florida Natural Areas Inventory 1990). Currently wildfires are common in the spring, but can occur whenever drought conditions arise (Wade and Ward 1973). Fires were probably most common in the spring, which is often a dry period when the water table and fuel moisture are lowest. Fires were quite intense because of the fuel loads and the flammability of the shrubby or grassy understory. However, most were mixed severity fires where a portion of the overstory pond pine survived because of its adaptations. Stand replacement fires occurred during extreme droughts when the underlying peat was dry enough to burn and the resulting high severity ground fire consumed accumulated organic matter, killing the overstory and shrub layer (Wade and others 2000).

## Cypress ponds and savannas

These areas are dominated by pond cypress, which is native to the Coastal Plain from Virginia to southern Florida and west to southeastern Louisiana (Wilhite and Toliver 1990). They occupy poorly to very poorly drained infertile soils that range from sands to clays and are often overlain by peat or muck. Cypress ponds or domes are isolated depressions ranging from 2.5 to 25 acres (1 to 10 ha) that are found on generally flat expanses of the coastal lowlands (Ewel 1998). They are not generally influenced by perennial flowing streams, but rather are wet because of excessive precipitation and perched water tables. The overstory is predominately pond cypress but often contains swamp blackgum and lesser amounts of sweetbay and loblolly bay with slash pine on the slightly higher rims. The understory is dominated by woody species including yaupon (*Ilex vomitoria*), swamp titi, wax myrtle, and gallberry (Wilhite and Toliver 1990). Cypress savannas containing small, slow growing pond cypress over a grassy dominated understory occur on larger broad flat areas like the Big Cypress National Preserve in southwestern Florida (Muss and others 2003), where it occupies 370,000 acres (150 000 ha). During the wet season these systems are inundated with slowly flowing water but become dry and readily flammable during droughts.

For all habitats, the water level fluctuates considerably from the wet season to the dry season. In addition, ponds and strands are found imbedded in pyric flatwood communities that burn quite frequently and have an understory dominated by ericaceous shrubs with waxy leaves. Therefore, historically they burned regularly, about every 20 years, with both understory and mixed severity fires (Ewel 1990). The most severe fires occurred in areas with accumulated peat and in conditions that were dry enough for partial or complete consumption (Cypert 1961). The cypress savanna also burned every 5 to 15 years with understory surface fires. These periodic fires kept hardwoods from encroaching, because pond cypress is more resistant to fire than hardwoods and will sprout from adventitious branches following burning. Historically most fires occurred in savannas during the spring and early summer dry periods when conditions were favorable for lightning ignited fires. Wildfires ignited by humans are more common now during the very dry dormant season. Fire severity has also increased due to widespread drainage, which can lead to replacement of pond cypress by willows (*Salix* spp.) and eventually mixed hardwoods (Wade and others 1980). Cypress ponds however, likely have less fire than historically since much of the burning in surrounding communities is done during the dormant season when they are too wet to ignite (Kirkman and others 2000). This could lead to fuel accumulations and more severe wildfires when they do burn. Pond cypress was harvested mainly for poles and posts during the extensive logging era of the 1900s (Dennis 1988). Recently, logging has become quite widespread, however, with the development of commercial production of cypress mulch for the landscape industry, and many pond cypress domes have been clearcut for mulch over the past 20 years (Black and others 1993). Pond cypress has the ability to stump sprout and thus should regenerate most harvested areas (Terwilliger and Ewel 1986). Alteration of hydrology by drainage began much earlier and has been more widespread than harvesting. This leads to dryer conditions, an invasion by pines and more frequent fires that change the area to pine flatwood vegetation.

## *Stand Replacement Fire Regime*

### Dry prairie

This community is found only in southern Florida with the largest concentrations historically along the Kissimmee River, west and south of Lake Okeechobee, and the area north of Charlotte Harbor in Sarasota and Manatee counties. Harper (1927) estimated it covered 1,285,000 to 1,927,500 acres (520 000 to 780 000 ha) but more recent data (Shriver and Vickery 1999) indicate it once occupied about 2,051,000 acres (830 000 ha). Also called palmetto prairie, it is a treeless grass dominated community that occurred on broad flat landscapes where fire was very frequent because there were no major natural fire barriers. Interspersed throughout the community were areas occupied by wet prairie, ephemeral depression ponds, marshes, flatwoods, and mesic hammocks. Soils were sandy, poorly to somewhat poorly drained, acidic, and nutrient poor. The subtropical climate of the area has a pronounced wet and dry season. During the wet season the water table often is at or above the soil surface, while during the dry season it is a meter or more below the surface (Abrahamson and Hartnett 1990). The diverse ground cover of palmetto prairie is dominated by wiregrass with scattered saw palmetto and patches of runner oak (*Quercus minima*). Other common plants include bottlebrush threeawn (*Aristida spiciformis*), broomsedge bluestem (*Andropogon virginicus*), fetterbush lyonia, coastal plain staggerbush (*Lyonia fruticosa*), shiny blueberry (*Vaccinium myrsinites*), and yelloweyed grasses (*Xyris* spp.)

Today, only about 10 percent or 385,500 acres (156 000 ha) of intact palmetto prairie remain in southern Florida (Shriver and Vickery 1999) with the largest patches found on public lands like Myakka River State Park and Kissimmee Prairie State Preserve Park. Much of the original area has been lost to conversion for agriculture to citrus, vegetables or improved pasture. Many other areas have been heavily impacted by cattle grazing and disruption of the historical fire regime. In the absence of frequent fire, this community is taken over by invading trees and emergent shrubs and converts to pine or palm flatwoods or hardwood hammock (Huffman and Blanchard 1991). Some area has also been taken for urban development.

The wiregrass, saw palmetto, and ericaceous shrubs that dominate this community are very flammable, fueling stand replacement burns, but they also resprout quickly and revegetate the site (Abrahamson and Hartnett 1990). Historically, these fires were very frequent occurring every 1 to 2 years. Harper (1927) indicated that dry prairie burned almost every year and others also report very frequent fires (Abrahamson and Hartnett 1990). Southern Florida, where palmetto prairie is found, has one of the highest incidences of lightning in the country, which served as a natural ignition source. Since there was little in the historical landscape to stop fires, an ignition could burn a very large area. Most fires occurred during the transition from dry to wet season, which is April to June, as the thunderstorms returned but the landscape was not yet remoistened (Beckage and Platt 2003).

### Freshwater marsh

In the Southeastern States, inland freshwater marshes are associated with rivers, lakes, shallow basins, and other depressions (McPherson 2008). Tidal freshwater marshes occur along the Atlantic and Gulf coasts. Marshes develop wherever topography and impermeable soils limit runoff or infiltration (Kushlan 1990). They are found on sandy alluvial soils with variable amounts of peat or marl. The historical extent is not well known but they did cover many thousands of acres across the Southern United States. The hydroperiod is variable but all marshes have sufficiently long periods of inundation to limit encroachment of many wood plants. Dominant vegetation is quite diverse with emergent aquatic species in the lower marsh while higher zones have extensive dense stands of graminoids like sand cordgrass (*Spartina bakeri*) and maidencane (*Panicum hemitomon*) with scattered patches of shrubs (Fisher 2008). Inland marshes have seasonal fluctuations in water level dependent on evapotranspiration and rainfall patterns while tidal marshes experience daily fluctuations driven by the tides.

Tidal marshes are still quite common in the South (Wade and others 2000) covering about 1,976,835 acres (800 000 ha). Inland marshes however have been heavily impacted by drainage and conversion to agricultural uses. Marshes along the St. Johns and Kissimmee river systems in Florida for example have been reduced by >70 percent (Kushlan 1990). Other areas have had an influx of nutrients from agricultural or urban areas that enriches marshlands, encouraging the growth of cattails (*Typha* spp.), allowing them to replace native vegetation on marshlands across the South. Disruption of the normal fire regime also changes vegetation by favoring woody species growth. Trees and shrubs are also encouraged by drainage of marshlands for flood control that shortens the period of flooding. The combination of reduced fire and less flooding has resulted in significant areas along major river drainages becoming dominated by wax myrtle, coastal plain willow (*Salix caroliniana*), or red maple (*Acer rubrum*). More recently with the realization of the importance of wetland ecosystems, concerted efforts have been made to remove flood control structures and restore the channel and floodplain marshes along the Kissimmee River (Toth 1993).

Fire has been an important driver in this system but different types of marshes burned with differing frequencies (Kushlan 1990). Higher zones typically dry annually and likely burned every 1 to 6 years (Frost 1995). The wetter lower zones and areas with significant peat accumulation likely burned only during periodic droughts (Kushlan 1990). Landscape location is also important with tidal marshes burning every 3 to 5 years where fire can enter from adjacent uplands but frequency declines quickly for isolated areas behind channels (Schafale and Weakley 1990). Fire frequency of smaller isolated marshes depends on fire-return interval for the surrounding community. Fires are stand replacement, but even the plants found on wetter lower zones regrow quickly, taking advantage of the nutrient flush and reduced competition (Kushlan 1990).

## Everglades sawgrass and marl prairies

These freshwater marsh communities found in southern Florida in the Savanna Division are unique because of their size, location and special character. The Everglades are also known as the "river of grass" because excess precipitation historically flowed slowly southward along a path 50 miles wide and 120 miles long (80 by 193 km) through a mostly grass dominated freshwater marsh. A subtropical climate prevails, with a wet rainy season characterized by almost daily thunderstorms from mid May to October when 80 percent of rainfall occurs followed by a dry period with little rainfall (Gunderson and Loftus 1993). Marl prairies, which are normally flooded 3 to 7 months per year, are found on shallow inorganic soils over the limestone bedrock, which is close to the surface throughout southern Florida (Olmsted and others 1980). On slightly deeper areas, inundation slows decomposition forming organic peat soils where sawgrass communities are dominant. This sawgrass marsh was the most prevalent vegetation of the Everglades, once covering 1,976,835 acres (800 000 ha) of southern Florida. It was dominated by the grasslike sedge, called sawgrass (*Cladium jamaicense*) because of the sharp edges on its leaf blades (Kushlan 1990). Composition varies from sites with nearly pure sawgrass, which grows up to 10 feet (3 m) high, to a mixture of 30 species. Other common associates are Gulf Coast spikerush (*Eleocharis cellulosa*), blue waterhyssop (*Bacopa caroliniana*), Tracy's beaksedge (*Rhynchospora tracyi*), switchgrass (*Panicum virgatum*), maidencane, and saltmarsh morning-glory (*Ipomoea sagittata*). The marl prairie is a mixture of many species usually <3 feet (1 m) tall (Jenkins 2008), including Muhly grass (*Muhlenbergia filipes*), sawgrass, black bogrush (*Schoenus nigricans*), Tracy's beaksedge, and Florida little bluestem (*Schizachyrium rhizomatum*). This community once covered about 445,000 acres (180 000 ha) of southern Florida in the Everglades (Davis 1943).

Beginning in the early 1900s, efforts were made to drain the Everglades to make the area available for agriculture and urban development. To date more than half of the area has been drained. Everglades National Park was established to protect the unique area and its biota. Together with the conservation areas north of the park, about 20 percent of the original Everglades is protected (Davis and Ogden 1994). Even the protected areas however, have been impacted by 1,250 miles (2000 km) of canals, levees, and spillways that control water flow into them. To avert flooding, water is funneled off a good portion of the landscape during the rainy season, but stored in other areas for urban use during the dry season. The result is flow through the marshes of Everglades National Park has been reduced to about 10 percent of historical amounts. Less water can lead to increased drying and loss of peat through oxidation or by combustion during wildfires. Enrichment from fertilizers flushing from vegetable and sugar cane fields to the north also impact marsh vegetation (Davis and Ogden 1994). Enrichment coupled with the more severe wildfires can lead to replacement of sawgrass with cattails (McPherson 2009). Sawgrass also tends to decline on the conservation areas used to store excess water from deeper water and more prolonged inundation (Gunderson and Loftus 1993).

Fire is needed to maintain the sawgrass and marl prairies, which historically had stand replacement burns every 2 to 15 years (Jenkins 2008). Most large fires were lightning ignited, occurring from April to June at the transition from dry to wet season (Gunderson and Snyder 1994). Fires in wetter months were smaller and patchier because of wetter conditions. Sawgrass is highly adapted to fire, as it will burn even over standing water. Its meristem is protected by a spongy leaf base that is often below water or will absorb water from below during most dry periods. It regrows rapidly after a fire, often reaching preburn levels within 2 years (Wade and others 1980). Fire is also important for controlling woody species invasions into marl prairies. There has been a shift in fire season throughout the Everglades caused by humans with a significant number of fires now occurring during the driest portion of the dry season before lightning and associated rainfall return. Ground fires that consume the underlying peat are also more common due to accidental fires during drier months but mostly because many areas have been dried by drainage.

## Canebrakes

The presettlement extent of area covered by canebrakes is not known; but based on numerous accounts of early explorers, it was thousands of acres (Platt and Brantley 1997). This bottomland community dominated by cane (*Arundinaria gigantea*) was found along every major river and stream in the Southeastern United States. William Bartram, the early American naturalist and author (Van Doren 1928), made repeated references to vast cane tracts or meadows including traveling through a cane meadow for 20 miles (32 km) in Alabama. Along major rivers cane formed pure stands 30 feet (9 m) tall up to 4 inch (10 cm) in diameter so thick that it was necessary to cut a path to traverse these areas (Platt and Brantley 1997). Cane was a ubiquitous material that Native Americans used for many daily items from firewood to containers. When traveling, if a river too deep to wade was encountered, they relied on the ever-present cane to make a raft for crossing (Hudson 1976). Large canebrake savannas with dense cane beneath a sparse hardwood canopy were found on terraces of alluvial floodplains, where flooding was frequent but inundation periods short. Although cane will grow on a wide range of soils it did best on these fertile, deep and well drained soils found along rivers and streams (Barone and others 2008). Common associate species of cane include laurel greenbrier (*Smilax laurifolia*), gallberry, swamp titi, wax myrtle, and saw palmetto (Shoonover and Williard 2003). Switch cane was also found as an understory species in a number of other evergreen and deciduous forests outside the floodplains.

The vast canebrakes of the historical landscape are gone. The scattered patches that remain cover about 2 percent of its former area (Noss and others 1995). They disappeared from overgrazing, agricultural conversion, altered fire regimes, and flood control. Cane was an important forage crop for settlers' livestock. It is the highest yielding native forage in the South and remains green and palatable throughout the year (Hilmon and Hughes 1965). However, it is very sensitive to overgrazing and rapidly declines if utilized continuously (Shepherd and others 1951). Range burning, which early cattlemen did annually, exhausted the carbohydrate reserves of the underground rhizomes, converting cane to savanna grasslands (Platt and Brantley 1997). Because it was found on fertile soils, many sites were converted to fields for crop production. Other areas, because of fire suppression, became shrub or forest dominated sites where these woody species shaded out the cane. Finally when dams and other flood control structures were constructed, they drowned many remaining canebrakes or stopped the periodic flooding they needed to keep them healthy.

Canebrakes require disturbance from periodic flooding and fire to remain viable (Brantley and Platt 2001). Some believe that the vast canebrakes found by early European explorers were the result of Native American agriculture and existed because of abandoned fields and regular burning (Platt and Brantley 1997). Others however, postulate the Native Americans picked the areas where cane was found for their fields because it indicated fertile sites (Hudson 1976). Thus, old fields covered with cane were simply converting back to their former cover. Regardless, it is known that canebrakes need regular burning to remain healthy and the Native Americans burned them every 7 to 10 years. This kept woody shrubs and trees in check and allowed the cane to flourish. Cane is quite flammable with the culms burning easily. Following fire, it quickly resprouts from underground rhizomes and can grow up to 1.2 inches per day (3 cm), rapidly reoccupying an area and out competing other species. As noted above, annual burning will eventually eliminate cane and because most reproduction is vegetative there is not a seed bank for reestablishment. A fire-return interval of 10 years is recommended to maximize productivity (Hughes 1966).

# Fuels Management

Fuel treatments can be used to accomplish a number of objectives like ecosystem restoration, wildfire hazard reduction, wildlife habitat improvement, insect or disease control, aesthetic improvement, forage production, or silvicultural enhancements. In most applications, it is used to achieve multiple objectives. In all vegetation types, the fuel treatment depends on where the site is located, its current condition, and the desired outcomes. If the major goal is wildfire hazard reduction in forested ecosystems, the objectives will be to reduce surface fuels, increase distance to the live crown, lower crown density, increase the dominance of large fire resistant trees, or a combination of all four (Agee and Skinner 2005). In grassy ecosystems, the major goal is often to control woody species, which can quickly capture an area in the absence of fire.

Originally, fuels reduction was most often limited to wildlife habitat improvement, championed by Stoddard (1931), for bobwhite quail (*Colinus virginianus*) production. Another early application was to reduce the potential for uncontrolled wildfires, with use increasing as the benefits were recognized. Early research (Davis and Cooper 1963) showed that the area burned and the intensity of wildfires was strongly related to with the amount of time since the last prescribed burn. Recent work has also documented that tree mortality from wildfires under extreme conditions is lower in sites where prescribed fire had recently been applied (Outcalt and Wade 2004a). Fuel treatments have also long been part of silvicultural prescriptions used, for example, to control hardwoods in pine stands or to prepare seedbeds for pine regeneration. As noted above, most native ecosystems are adapted to fire either quite frequently or at least periodically. If fire is excluded, then they change in undesirable ways that affect the habitat, often reducing biodiversity and contributing to the decline of endangered species. This has led to a concerted effort to restore many fire dependent ecosystem—such as longleaf pine (Brockway and others 2005), which usually requires reduction in accumulated fuels.

The South is blessed with a long growing season and plentiful precipitation, ensuring that its forest and grassland ecosystems are quite productive but also leading to a rapid accumulation of potential fuels. Fuel levels are determined by site productivity, overstory density, and years of accumulation. A typical slash pine plantation with basal area of 110 square feet per acre (25 $m^2$/ha) rapidly accumulates forest floor and will contain 6.6 tons per acre (14.8 t/ha) in just 4 years (table 1) and has another 2.7 tons per

**Table 1.** Fuel accumulation in surface layers measured over time at varying intervals for a range of vegetation types of the Subtropical Division (Bailey 1996), expressed on a dry weight basis

| | | Years of fuel accumulation | | | | | | | | | | | |
|---|---|---|---|---|---|---|---|---|---|---|---|---|---|
| Vegetation type | Component | 1 | 2 | 3 | 4 | 5 | 6 | 7 | 10 | 15 | 20 | 30 | Reference |
| | | *tons/acre (t/ha)* | | | | | | | | | | | |
| Slash pine | Litter[a] | 2.3 (5.15) | 4 (8.96) | 5.4 (12.10) | 6.6 (14.78) | 7.6 (17.02) | 9.3 (20.83) | 11.1 (24.86) | 12.7 (28.45) | 13.2 (29.57) | — | — | McNab and others (1978) |
| Loblolly pine | Litter[a] | 2.1 (4.70) | 3.5 (7.84) | 4.6 (10.30) | 5.4 (12.10) | 6 (13.44) | 7.1 (15.90) | 7.4 (16.58) | 7.4 (16.58) | — | — | — | Wade and others (2000) |
| Slash/longleaf flatwoods | Understory | 0.5 (1.12) | 2 (4.48) | 2.6 (5.82) | — | 2.8 (6.27) | — | — | 3.6 (8.06) | 6.6 (14.78) | 8.4 (18.82) | — | Wade and others (2000) |
| Longleaf flatwoods | Litter | 2.19 (4.91) | 3.71 (8.31) | 8 (17.92) | 3.99 (8.94) | 5.12 (11.47) | — | — | — | — | 18.75 (42.00) | 27.18 (60.88) | Ottmar and Vihnanek (2000) |
| Longleaf flatwoods | Understory | 1 (2.24) | 1.28 (2.87) | 3.32 (7.44) | 3.75 (8.40) | 3.41 (7.64) | — | — | — | — | 3.42 (7.66) | 5.54 (12.41) | Ottmar and Vihnanek (2000 |
| Longleaf sandhill | Litter | 1.59 (3.56) | — | 1.73 (3.88) | — | — | — | — | — | — | 8.97 (20.09) | — | Ottmar and others (2003) |
| Longleaf sandhill | Understory | 0.32 (0.72) | — | 1.02 (2.28) | — | — | — | — | — | — | 1.25 (2.80) | — | Ottmar and others (2003) |
| Loblolly pine plantation | Litter | — | — | — | 6.35 (14.22) | — | — | — | — | — | — | — | Scholl and Waldrop (1999) |
| Loblolly pine plantation | Understory | — | — | — | 0.16 (0.36) | — | — | — | — | — | — | — | Scholl and Waldrop (1999) |
| Loblolly pine plantation | Litter | — | — | — | 6.22 (13.93) | — | — | — | — | — | — | — | Scholl and Waldrop (1999) |
| Loblolly pine plantation | Understory | — | — | — | 0.54 (1.21) | — | — | — | — | — | — | — | Scholl and Waldrop (1999) |
| Longleaf wiregrass | Litter | — | 1.21 (2.71) | — | — | — | — | — | — | — | — | — | Ottmar and Vihnanek (2000) |
| Longleaf wiregrass | Understory | — | 1.59 (3.56) | — | — | — | — | — | — | — | — | — | Ottmar and Vihnanek (2000) |
| Marl prairie | Litter and understory | 0.6 (1.4) | — | 1.7 (3.75) | — | — | 1.7 (3.75) | — | — | 1.7 (3.9) | — | — | Herndon and Taylor (1986) |

— = No data available.

Note: Litter is all dead surface fuel including leaves, needles, twigs, branches, and stems; understory is all living plants <4.5 feet tall (1.5 m).

[a] Basal area = 110 square feet per acre (25 $m^2$/ha).

acre (6 t/ha) in understory fuel. Higher density stands produce more litterfall and therefore have higher forest floor fuel loads. For example, the same slash pine plantation with double the stocking would contain 10.9 tons per acre (24.4 t/ha) of forest floor fuel after 4 years without a fire. Sites with poorer soils, like longleaf pine sandhills—with their droughty, nutrient poor sands—accumulate much less fuel and therefore will be subject to less intense fires, even if not burned for extended periods. Pocosins with their shrub dominated understory can have very high fuel loads in this layer (table 2), making prescribed burning and wildfire control difficult. This is also true to a lesser extent for the palmetto-gallberry fuel type found in longleaf and slash pine flatwoods (Wade and others 2000). Grassland accumulates fine fuels very quickly in an arrangement that makes it particularly flammable. Thus, fuel treatments must consider both the rate of accumulation and the type of material.

## *Prescribed Burning*

### Longleaf pine

Prescribed burning is used in all longleaf pine communities from wet flatwoods to droughty sandhills and from the Atlantic Coastal Plain to the montane area of the ridge and valley. Frequency is tied to fuel accumulation rates with a fire-return interval of 1 to 4 years (table 3), but most stands are burned every 3 to 4 years on sandhills, flatwoods, and wet coastal sites. More productive mesic uplands, especially those lands managed for wildlife production, are typically burned on a 2- to 3-year cycle. More frequent fires are also applied to areas with excess fuel accumulations or midstory hardwoods or both (Brockway and others 2005, Outcalt 2006). The goal is to reduce wildfire hazard and restore the ecosystem to a more herbaceous-dominated understory, which can then be maintained with less frequent fire. Historically much of this burning was applied during the dormant season when weather was more predictable and air temperatures were

**Table 2.** Typical fuel loads on a dry weight basis in select vegetation types of the Subtropical Division and the Savanna Division (Bailey 1996)

| Vegetation type | Component | Age | Tons per acre | Tons per hectare | Reference |
|---|---|---|---|---|---|
| Choctawhatchee sand pine | Litter | Stand age 15 years | 9.45 | 21.17 | Ottmar and others 2003 |
| Choctawhatchee sand pine | Understory | Stand age 15 years | 2.99 | 6.70 | Ottmar and others 2003 |
| Choctawhatchee sand pine | Litter | Stand age 58 years | 24.84 | 55.64 | Ottmar and others 2003 |
| Choctawhatchee sand pine | Understory | Stand age 58 years | 5.96 | 13.35 | Ottmar and others 2003 |
| Oak and hickory | Litter | Age of rough unknown | 5.68 | 12.72 | Scholl and Waldrop 1999 |
| Oak and hickory | Understory | Age of rough unknown | 0.5 | 1.12 | Scholl and Waldrop 1999 |
| Pocosin woodland | Litter | Age of rough unknown | 4.28 | 9.59 | Ottmar and Vihnanek 2000 |
| Pocosin woodland | Understory | Age of rough unknown | 3.62 | 8.11 | Ottmar and Vihnanek 2000 |
| Pocosin high | Litter | Age of rough unknown | 3.85 | 8.62 | Ottmar and Vihnanek 2000 |
| Pocosin high | Understory | Age of rough unknown | 19.3 | 43.23 | Ottmar and Vihnanek 2000 |
| Pocosin low | Litter | Age of rough unknown | 2.79 | 6.25 | Ottmar and Vihnanek 2000 |
| Pocosin low | Understory | Age of rough unknown | 9.97 | 22.33 | Ottmar and Vihnanek 2000 |
| Oak-pine | Litter | Age of rough unknown | 6.32 | 14.16 | Ottmar and others 2003 |
| Canebrake | Understory | 4 years | 7.0 | 15.7 | Hughes 1966 |
| Sawgrass | Litter and understory | Age of rough unknown | 12.5 | 28.0 | McPherson 2008 |

Note: Litter is all dead surface fuel including leaves, needles, twigs, branches and stems; understory is all living plants <4.5 feet tall (1.5 m) except in canebrake and sawgrass where it includes all grass.

**Table 3.** Types of fuel treatments used in different vegetation types in the Subtropical Division and the Savanna Division (Bailey 1996)

| | Treatment | | | | |
|---|---|---|---|---|---|
| Vegetation type | Prescribed burn[a] | Mechanical[b] | Manual | Harvesting[b] | Herbicide[c] |
| Longleaf sandhills | Understory 2 to 4 years | Chopping and mulching | Hand clearing | Thinning and clearcutting | Understory and midstory |
| Longleaf flatwoods | Understory 1 to 4 years | Chopping and mulching | | Thinning and clearcutting | Understory |
| Longleaf uplands | Understory 1 to 4 years | Chopping and mulching | Hand clearing | Thinning and clearcutting | Understory and midstory |
| Slash pine flatwoods | Understory 1 to 4 years | Chopping and mulching | | Thinning | Understory |
| Loblolly pine | Understory 2 to 5 years | Mulching | Hand clearing | Thinning | Understory and midstory |
| Shortleaf pine | Understory 3 to 5 years | Mulching | Hand clearing | Thinning | Understory and midstory |
| Oak-hickory-pine | Understory 3 to 6 years | | Hand clearing | Thinning | |
| Pine rocklands | Understory 3 to 5 years | Mulching | Hand clearing | Thinning and clearcutting | |
| Pitch pine | Understory 5 to 15 years | Mulching | | Thinning | |
| Ocala sand pine | Stand replacing 15 to 75 years | Chopping | | Clearcutting | |
| Choctawhatchee sand pine | Understory 3 to 5 years | | | Clearcutting | |
| Pond pine | Understory 3 to 8 years | Mulching | | | |
| Cypress domes | Understory 2 to 5 years | | | | |
| Cypress savanna | Understory 10 years | | | | |
| Dry prairie | Stand replacing 1 to 4 years | Chopping | | | |
| Freshwater marsh | Stand replacing 2 to 5 years | | | | |
| Everglades prairie | Stand replacing 3 to 5 years | | | | |
| Canebrake | Understory 3 to 5 years | | Hand clearing | Thinning | |

Note: Blanks indicate that type of treatment is not being applied to that forest type.
[a] Type of burn and normal frequency of application.
[b] Typical treatments used in different vegetation types.
[c] Target layer of application.

lower. For the last 15 years however, many more areas are being burned during the growing season, which has been shown to be more effective for reducing hardwoods (Waldrop and others 1987). Because of seasonal differences in responses to burning, the best approach is to vary the time of application rather than repeatedly burning a particular site at the same time of the year. Many quail plantations avoid growing season burns to limit nest loss.

Although private lands are often burned with drip torches and strip headfires or flanking fires, the general trend on public lands has been to burn in larger blocks. Today many of the burns are ignited from helicopters that quickly create many spot fires, allowing 500 to 2,500 acres (200 to 1000 ha) to be burned as a unit in a day. This firing technique impacts more of the watershed at one time, but also more closely mimics the larger size of the fires in longleaf communities that preceded landscape fragmentation. National forest records show 820,000 acres (332 000 ha) of longleaf pine in 2006 with another 300,000 acres (121 000 ha) of longleaf on other Federal properties, mostly military installations (Miles 2007). Assuming these are being burned on average at least every 5 years, then about 223,000 acres (90 000 ha) of longleaf are burned annually on public lands. Private lands receive less frequent fire; only about 37 percent are burned every 5 years, or 148,000 acres (60 000 ha) annually (Outcalt 2000).

## Slash pine flatwoods

Because this system often has a very similar palmetto-gallberry understory fuel complex and is often mixed with longleaf flatwoods, prescribed burning can be applied with the same frequency as longleaf: 1 to 4 years with most sites burned every 3 or 4 years. More frequent burning is necessary during restoration treatments to reduce both fuel loads and palmetto and woody understory cover while increasing herbaceous growth. Because young saplings with a ground level diameter of <2 inches (5 cm) can be killed by surface fires (Johansen and Wade 1987), burning is not appropriate in young plantations or naturally regenerated stands until trees reach 3 to 5 years old. Most burning was historically applied during the dormant season, but is now being expanded to the growing season, especially to provide ecological benefits on public lands and to increase options for managers, most of whom need every burn day they can get to keep stands within the appropriate fire-return interval. Some variation in season—which includes the growing season—is desirable, but repeated burning annually or biennially during the dormant season will reduce palmetto cover and increase grasses (Outcalt and Wade 2004b).

As noted above, public land managers are increasingly using helicopter-ignited spot firing, which treats large blocks in a single burn. Thus, slash pine flatwoods are often included with adjacent longleaf pine dominated stands in large burns. Prescribed burning has largely been curtailed on many forest industry lands because of liability issues from smoke on roads. However, burning in slash pine dominated stands is still a big portion of total burning on private land because slash pine has been planted extensively and captured much of former longleaf area following logging and fire control during the first half of the 1900s. With 1.63 million acres (660 000 ha) of slash pine on public lands and two-thirds that in naturally regenerated stands, public land managers are estimated to burn 326,000 acres (132 000 ha) annually based on an average fire-return interval of 5 years.

## Loblolly pine

This species is not considered to be fire dependent and historically loblolly dominated stands were confined to wetter or sheltered sites where surface fires were not as frequent. However, Wade (1993) showed that it is tolerant of fire once saplings attain a ground level diameter of 2 inches (5 cm), which explains why historically it was a co-dominant with longleaf and/or shortleaf on many areas. Once past this stage, stands have been routinely burned to control hardwood competition, reduce fuel loads, promote forage production, and improve wildlife habitat. Grass-dominated understories can be burned annually but the usual range is 2 to 5 years with the majority burned every 3 to 5 years, which is sufficient to reduce fuel loads—especially in plantations after crown closure—that contain very little understory fuel (table 1). Shorter fire-return intervals are sometimes used to create a grassy herbaceous dominated understory with increased species richness (Glitzenstein and others 2003). Repeated growing-season burns, if applied at least every 2 years (Wade and others 2000), are more effective than dormant season burns for topkilling hardwoods and reducing hardwood rootstocks (Waldrop and others 1987). Many private owners avoid growing-season fires to protect nests of eastern wild turkey (*Meleagris gallopavo*) and bobwhite quail.

If there are no young seedlings and saplings that must be protected, large blocks of loblolly dominated forest on public lands can be burned using helicopter ignition. Considerable prescribed burning continues in loblolly pine stands on private lands for wildlife, esthetics, access improvement, and hazard reduction, but the total amount is unknown. As with slash pine, however, very little is done by forest industry. With 6.4 million acres (2.6 million ha) of loblolly forests in public ownership and 70 percent of that in naturally regenerated stands, public land managers are estimated to burn about 432,000 acres (175 000 ha) annually based on an estimated average fire-return interval of 6 years.

## Shortleaf pine

Much of the remaining shortleaf pine is found on the drier ridges in the Piedmont and Interior Highlands of the Subtropical Division. As with other southern pines, shortleaf underwent a period of fire suppression, which allowed hardwoods and other species to capture many sites. Recently, burning has increased with the goal of restoring community structure and returning shortleaf to more productive sites. Because it is quite resistant to surface fires, shortleaf can be burned every 2 to 5 years, or every 4 to 6 years for seedling establishment (Masters and others 2005) to allow saplings to grow beyond the size where fire will cause high mortality rates. Although stands can be burned at longer intervals of 12 years, the result will be denser stands with a less open structure (Masters and others 2005). Shorter fire returns of 3 years produce a grass-dominated understory but will also result in less than optimal stocking for timber production. Shortleaf pine can be burned in both dormant and growing seasons (Sparks and others 2002). Masters and others (2002) found late dormant season burns applied every 3 years greatly improved wildlife habitat. However, maintaining the health of the entire forest community requires frequent burning in all seasons (Masters 2007).

Prescribed burning in shortleaf pine stands on public lands has been mostly by hand ignition or with torches attached to all terrain vehicles. Because of safety issues and the need to increase the amount of area burned each year, helicopter burning using ping-pong ball spot ignition is becoming more common on large blocks of forest. Burn units have increased in size on the Ouachita National Forest to 620 acres (250 ha), with some as large as 7,400 acres (3000 ha). With about 951,000 acres (385 000 ha) of shortleaf pine forest in public ownership and 91 percent of that naturally regenerated, public land managers are estimated to burn about 190,000 acres (77 000 ha) annually based on an average fire-return interval of 5 years. Some burning continues on private lands, but virtually none on forest industry property.

## Oak-hickory-pine woodlands

The role of fire in perpetuating this community was only recently recognized (Lorimer 1993). Recent research has shown that prescribed burning can be used to aid establishment of regeneration, which can later be released by subsequent burns (Wade and others 2000). Burn frequency depends of landowner goals and initial conditions. Annual or biennial burns are used to reduce shading by competing hardwoods and open the stand to promote establishment of oak and hickory seedlings, or in tandem with a shelterwood cut to enhance growth for stands with established regeneration. Burning should be delayed until oak seedlings are 0.8 inches (2 cm) in root collar diameter, and then applied during the growing season to kill the regeneration layer, which removes less fire tolerant species, leaving the oaks and hickories to sprout and grow. Fire can then be applied as needed to keep competing hardwoods in check, usually every 3 to 6 years.

Burning in these habitats has been limited. Many believed that even low-intensity surface fires would damage hardwood stems of large overstory trees. In addition, most of these sites do not need hazard reduction burns because the more mesic hardwoods have captured many former habitats during 50-plus years of fire suppression. The dense shade and the accumulated litter, which is less flammable than pine needles and oak leaves, make these stands less likely to burn. Thus, uncontrolled wildfires are not a danger. The justification for burning is to restore former habitats that likely existed largely because of Native American burning. There is considerable public resistance to prescribed burning in this habitat on Federal and State properties . Some managers conduct the initial burn of mixed stands during the dormant season to remove excess fuel accumulations. Ignition is often with drip torches used to set backing or flanking fires. The total area burned is not yet large, but it is increasing with the growing recognition of the value of fire and with the support of outside environmental and conservation groups.

## Pine rocklands

What remains of these fire dependent systems is mostly on public lands, which are burned frequently, every 3 to 5 years, to control understory hardwoods and maintain ecosystem health. A slightly longer return interval may be necessary to allow young slash pine seedlings to become large enough to survive fire (Olmsted and Loop 1984), also satisfying the need for variation to more closely mimic historical fire frequency (Snyder and others 1990). Managers historically burned many areas during the dry season but began conducting burns during the wet season in 1981. Most burns in Everglades National Park are now conducted during the early part of the wet season (June and July). At Big Cypress Preserve, the largest burns are reserved for this period, but significant areas of pinelands are also burned in late winter to early spring (January and February). Research has shown that fire intensity rather than season determines whether burning is effective for reducing understory hardwoods (Snyder 1986), and other non-seasonal factors are also critical. As has been shown for other southern ecosystems, repeated burning is required to exhaust hardwood root reserves (Gunderson and others 1983). Snyder (1986) also found that understory herbaceous vegetation recovered quickly after both dry-season and wet-season fires.

In Everglades National Park, burns are ignited by helicopter or with vehicle mounted torches. Burns are managed using topographic features of the landscape to determine burn areas. Thus, burns have been increasing in size, and now consist of several-hundred acre tracts rather than individual stands surrounded by artificial barriers like roads or fire lines. Managers also use prescribed natural fire, where lightning ignited fires are allowed to burn with careful monitoring as long as they occur within predetermined prescriptions and are not likely to spread to areas outside park boundaries. Big Cypress Preserve also uses aerial ignition and has increased the size of prescribed burns up to >7,400 acres (3000 ha) and range to include multiple forest communities.

## Mixed fire regime communities

Prescribed burns have been conducted in pitch pine barrens since the 1950s to reduce fuel loads and the danger from catastrophic wildfires (Buell and Cantlon 1953). Initial burns were mostly in the winter dormant season at intervals of 1 to 5 years. Dormant-season burns are still used to reduce fuel loads, but growing-season burns are also employed. Popp (1987) showed that successive annual growing season burns are more effective for restoration because they reduce hardwood sprouts and kill many rootstocks. Once fuel loads are reduced and hardwoods restored to a low level, a longer return interval of 10 to 15 years is effective. Because many of the remaining pitch pine stands are small fragments of former forests, large burns are possible only in some of the more extensive barrens of New Jersey.

Although sand pine is the most fire sensitive of the southern pines, prescribed burning is possible for the Choctawhatchee variety. Eglin Air Force Base began low-intensity, dormant-season burns of sand pine to control understory fuel loads and improve access in the 1960s (Britt 1973). More recently, burning has been used primarily to control the spread of sand pine into adjacent longleaf pine habitat. These burns tend to be part of large compartment-size burns of 740 to 5,000 acres (300 to 2000 ha), which include both habitat types and are conducted on suitable burn days in all seasons.

Prescribed burning is also used extensively to control invasion of sand pine into longleaf sandhills on the Ocala National Forest, along with an additional 2,900 acres (1160 ha) of sand pine scrub being managed as Florida scrub-jay (*Aphelocoma coerulescens*) habitat. Stand replacement prescribed burns in this scrub-jay area have a fire-return interval of 15 years. Burns are applied to stand-size areas of 50 to 150 acres (20 to 60 ha) to maintain a mosaic of age classes. In addition, prescribed burns are being ignited in the wilderness areas and then are allowed to burn as long as they are likely to remain inside the boundaries. These burns have been quite extensive covering 7,400 to 12,000 acres (3000 to 5000 ha).

Prescribed burning can be applied in pond pine pocosins to consume understory shrubs, but not if the fire is so intense that significant mortality occurs in the overstory.

Taylor and Wendel (1964) documented the conditions needed; yet many pond pine stands have gone unburned because they are more difficult to burn than other communities. This is especially true on areas with peat accumulations and long unburned areas with high fuel loads.

Some burning is taking place on public lands. On the Croatan National Forest, the goal is to apply fire at its historical return interval to the wilderness areas, including pond pine, and at a return interval of 3 to 5 years outside wilderness areas. The Croatan burns around 5,000 acres (2000 ha) of pond pine pocosin per year, with most burning from November to February but also extending through April if the weather is favorable. Burn units are 500 to 2,000 acres (200 to 800 ha) with ignition a combination of drip torches to secure the lines followed by helicopter ignition. Other public lands, like Camp Lejeune managed by Department of Defense, are also burning pond pine habitat as part of their overall prescribed burn program. The goal for pond pine woodlands and high pocosins on these lands is a 5- to 8-year fire-return interval, with a shorter 3- to 5-year interval in areas with endangered plants that require open conditions. Most burns are small units to limit smoke production and are applied in an array of conditions. Growing season burns are favored and areas with switch cane are given high priority for burning.

Cypress wetlands are usually imbedded within surrounding longleaf or slash pine dominated communities. Historically, they were often protected from fire by plowed firelines, which compromised the hydrology and disturbed the ecotone area that is the habitat for many rare plant species. This practice has been discontinued on public lands, where cypress wetlands are now burned at the same interval as the surrounding habitats, every 2 to 5 years. Because the cypress community is wetter, most of these fires only burn the edges with depth of penetration controlled by water levels at the time of the burn. Burning takes place in all seasons, but dormant-season fires are favored in areas where former fire exclusion has led to high fuel loads. Early growing season burns at a shorter frequency are used to support the endangered plants that prefer open habitats. The larger expanses of cypress savanna in southern Florida are burned in larger units that include multiple forest communities. Late winter to early spring and the early wet season (June and July) are the most active burn windows with a fire-return interval of about 10 years.

## Stand replacement fire communities

Most of the remaining dry prairie is burned frequently, every 1 to 4 years, to control fuel levels and prevent woody species from capturing the site. Until recently most burns were conducted in the dormant season from November to March. Over the last decade, some burning has shifted to the growing season, especially on public areas that need restoration. As with the similar flatwoods, more frequent burning, annually or biennially, will speed reduction of palmetto and competing shrubs when restoring dry prairie sites. Most dry prairie is found in smaller units and is being burned with ground based ignition rather than helicopters. Public lands are on a 2- to 3-year fire-return interval. In contrast, private ranchers still manage native grasslands by burning every 1 or 2 years in winter or early spring to green up their pastures.

Freshwater marsh, like most grass dominated systems that accumulate biomass rapidly, require frequent burning to maintain ecosystem health. Typically, these communities are burned every 2 to 5 years to control invasion by woody species and to reduce fuel loads. Hydrology seems to be as—or more—important however, because flooding is more effective than fire for limiting willow invasion (Miller and others 1998). Many remaining marshes are found on public lands where managers often burn in autumn or winter, after a killing frost has increased fuel flammability, to favor vegetation preferred by waterfowl (Gordon and others 1989). Burning when soils are moist reduces the likelihood of igniting ground fires in belowground organic material, thereby avoiding problems with control and smoke (Miller and others 1998).

Sawgrass and marl prairies are fire maintained systems that are burned frequently, every 3 to 5 years, to control fuels and maintain health. Almost all remaining sawgrass and marl prairies are on public lands, where regular burning has been in effect for many years. As with most southern systems, burning was usually confined to the winter dry season (Gunderson and Snyder 1994) but has now shifted more to the transition period from the dry to wet season (May and June). In Big Cypress, aerial ignition from helicopters is common with these systems burned as part of a large block burn. Everglades National Park has also gone to large block burns that cover more of the landscape and use natural firebreaks. To avoid ground fires, prescriptions set a minimum soil moisture level of 65 percent.

For many years, fire was deliberately excluded from canebrakes because they burned quite intensely. However, when it can be burned, cane responds favorably in either winter or summer (Platt and Brantley 1997). Although burning every 7 to 10 years is sufficient to maintain this ecosystem, most remnants under active fire management are burned every 3 to 5 years. This more frequent burning keeps fuel loads down and helps reduce woody species that increased on most sites during the fire exclusion period. Because of heavy fuel loads and high flammability in canebrakes that have not been burned for a long period, the first burn is normally delayed until fuel moisture and humidity are rather high. Because this community exists as mostly isolated patches, hand ignition with drip torches is employed. Some smaller areas are burned by aerial ignition incidental to burning the vegetation that surrounds them.

## *Mechanical Methods*

Mechanical fuel treatments are accomplished with an array of different equipment—including mowers, mulchers, and choppers—developed to cut, chop, shred, or sever mostly midstory and understory fuel layers. This equipment was developed for site preparation, land clearing, or right-of-way maintenance; it is most efficient on areas without large trees, but it can be used in existing stands where the retained overstory spacing is about 10 feet (3 m) or greater. Mowers are best suited to treating smaller understory shrubs. Mulchers come in various sizes; a small unit with a front mounted cutter can quickly chew through stems about 6 inches (15 cm) in diameter and high horsepower units can take down trees up to 12 inches (30 cm). Choppers also come in a variety of sizes and configurations, from small-teethed aerator models to 20-ton double drum offset machines. They knock down and crush understory and midstory fuel layers.

These treatments are most often applied to areas with high fuel loads for hazard reduction and ecosystem restoration. Because they do not remove material from the site, they change fuel configuration but not total fuel load. Thus, the midstory and understory fuel becomes surface fuel, which may or may not reduce wildfire severity. Most often, mechanical treatments are used to prepare the area for a prescribed burn. Reducing the ladder fuels reduces the potential for crown damage to overstory trees, compacting surface fuels often allows the burn to spread under more moderate conditions of higher humidity and fuel moisture with a lowered intensity. Therefore, except in some wildland-urban interface areas where burning may not be possible because of smoke sensitive areas, mechanical treatments are a one-time application to areas subsequently maintained with prescribed burning.

Chopping has been used extensively in longleaf pine ecosystems on all site types (table 3). Small to medium single drum choppers work well on sandhill sites to knock down the scrub-oak midstory layers that developed during the fire exclusion period. Selecting a tow unit and chopper that will knock down unwanted trees but then ride mostly on top of the small hardwood stems will limit soil disturbance. Chopping has also proven very useful for controlling saw palmetto on flatwood sites and in palmetto prairies (Fitzgerald and Tanner 1992). Research at Myakka River State Park (Outcalt and Brockway 2002) shows that chopping will reduce palmetto and shrub cover, while increasing grass cover. In the absence of additional treatment, palmetto fuel biomass did recover but remained lower on chopped sites that were subjected to a second prescribed burn 3 years later. In Ocala sand pine scrub, chopping is a replacement for fire in reducing height of woody stems, thereby maintaining forage and breeding habitat for Florida scrub-jays.

Mulching has been a fuel-treat option in all pine-dominated communities. It is used to reduce midstory scrub oaks on longleaf sandhills, and loblolly pine and mesic hardwoods like sweetgum (*Liquidambar styraciflua*) on upland longleaf sites. In upland and sandhill longleaf sites in Louisiana mulching reduced midstory hardwood density by 33 percent and understory woody cover by 64 percent (Rummer and others 2002). Mulchers have also been used on long-unburned flatwoods where the saw palmetto has become a true midstory of upright palms, 10 to 13 feet (3 to 4 m) tall. In loblolly-shortleaf pine types mulching targets hardwoods and small pine seedlings and saplings in high-density stands. The shrub layer is the target of mulching operations on pocosin sites. The objective in all these forest communities is to reduce the midstory fuel layer so sites can be more easily and safely burned.

Hand clearing, which can be used for cleaning or thinning stands, is performed with hand tools like axes, saws, or machetes, or with power equipment like brush and chainsaws. It has been used extensively to fell scrub oaks in sandhill longleaf stands (Provencher and others 2001), especially as a midstory removal treatment to improve red-cockaded woodpecker (*Picoides borealis*) habitat. Hand crews have also been employed to cut invading sand pine seedlings and saplings from longleaf sandhill sites on the Ocala National Forest in central Florida. On upland longleaf sites, chainsaws are used to fell invading loblolly pine, sweetgum, and other mesic hardwoods. These felling techniques are also used in loblolly, shortleaf, and pine hardwood systems to remove unwanted stems and reduce overall density. Hand treatments can be used in pine rocklands to remove midstory tropical hardwoods or understory and midstory invasives, such as brazilian peppertree (*Schinus terebinthifolius*) and melaleuca (*Melaleuca* spp.), and to remove privet (*Ligustrum* spp.) on canebrake sites. These treatments are generally followed by prescribed burning. Although felling does change fuel configuration, the goal is more often to accelerate the restoration process compared to using burning alone.

Harvesting by clearcutting or thinning—mostly with mechanized equipment—is a normal forestry operation, but recently the benefits for restoration and fuel management have become much more important and more widely applied in pines across the Southern United States. A common example is on longleaf sites for selectively removing other pines—such as loblolly or slash—along with the midstory and overstory hardwoods that increased during fire suppression periods. On typical longleaf upland sites in Alabama, thinning reduced hardwood basal area by 55 percent (Outcalt 2005) and proved to be the quickest way to restore stand structure to the conditions normally found in frequently burned longleaf stands. Thinning is also routine for removing excess stems and mesic hardwoods in loblolly, shortleaf, and oak-pine stands of the Piedmont; and is the standard prescription for restoring red-cockaded woodpecker habitat. A very intensive form of thinning has been used to remove all sand pine, only leaving residual longleaf pines on sandhill sites in Florida (Provencher and others 2000). To support the open conditions needed by cane to flourish, hardwood overstory has been thinned as part of the restoration treatment for some bottomlands.

Thinning treatments create considerable slash-type fuel, but also reduce midstory layers and create a more open stand with a lower crown density. To remove slash and reduce wildfire hazard, prescribed burns are routinely applied afterward. Often slash is allowed to decay for a period to reduce fire intensity, but burning is also possible soon after thinning operations (Outcalt 2005). Burning has the added benefit of reducing the density of understory hardwood stems, and can stimulate growth of grasses and forbs in loblolly stands after a thinning operation (Waldrop and McIver 2006).

Clearcutting is used mostly in situations where the dominant overstory tree species is poorly adapted to the habitat and needs to be replaced with a different species; for example, removing slash, loblolly, or sand pine from former longleaf pine sites or slash pine from dry prairie. Fuel created by these harvesting operations can be chopped and burned or just burned. With slash or loblolly pine, an alternative to clearcutting is to create openings that are planted with longleaf pine. This keeps large trees on the site that will furnish litterfall to help carry fire, but it also leaves a seed source for competing seedlings, which must be controlled with frequent burns.

## Herbicides

Herbicide treatments, with a variety of modern target-specific formulations aimed at understory and midstory layers, have been applied in pine plantations for some time. Many land managers also use herbicides specifically for fuel reduction. A recent survey in Florida found that 41 percent used this fuel reduction technique (Wolcott and others 2007). Forest industry rarely burns stands following herbicide application because of volatilization of nitrogen and smoke-management issues. Even without followup burns, herbicide treatment reduces fire intensity (Brose and Wade 2002), but offers less protection than regular prescribed burning during severe droughts (Outcalt and Wade 2004a).

Often just the first step in fuel reduction and restoration treatment that includes prescribed burning, herbicide treatment is generally used in areas with dense shrub layer vegetation that is difficult to burn. The objective is to kill the aboveground stems, allow more light to reach the surface, and increase the range of conditions that are favorable for prescribed burning. This approach has the added advantage of significantly accelerating the restoration process and reducing the time that fuel loads are high. On upland longleaf pine sites, herbicide followed by burning has been shown to be more effective than burning alone for sustained reduction of understory shrubs and woody vines (Outcalt and Brockway 2007). Herbicide has also been shown to be quite effective when combined with burning for restoring longleaf wiregrass communities on sandhill sites (Brockway and Outcalt 2000).

## Comparisons and Use

Each of the various fuel treatment options has positive and negative impacts that land managers must consider (table 4). Prescribed burning is inexpensive, especially with the economies of scale that come with burning large blocks. It also provides many ecological benefits that cannot be achieved with other treatments, and it causes very little soil disturbance. The major drawbacks are smoke impacts to offsite areas and potential for damage if there is an escape. These are especially troublesome where housing developments are immediately adjacent to forest areas. In these areas, mechanical treatments with choppers, mowers or mulchers are more appropriate as a first treatment and can make subsequent burns easier to conduct and control (Glitzenstein and others 2006). However, these techniques can be more costly and allow more potential for soil disturbance.

Both manual felling and harvesting can be applied to remove specific unwanted species or stems while leaving desirable ones in place. Harvesting has a special advantage in that it produces revenue from the sale of products removed.

Herbicides are quite effective at controlling target vegetation, are fast acting, and often make followup burns easier. Public acceptance of such treatments has been uneven, especially on public lands. In addition, they create dead fuel that may temporarily increase wildfire hazard.

Prescribed burning is the most widely used fuel treatment in the South because of its history of use, its low cost, and the ecological benefits it provides. On Federal properties alone, it was applied annually to 888,000 acres (359 000 ha) in 2006 and 2007. Another 949,000 acres (384 000 ha) were burned by State and other public land managers in 2007 (www.nifc.gov/nicc/sitreprt.pdf). Mechanical fuel treatments are applied to less area but are still used extensively, with an average of 85,690 acres (34 700 ha) treated annually on Federal lands in 2006 and 2007. Treatment amounts for other public lands are not know but can be assumed to be similar—about 10 percent of total area treated with prescribed burns. Mechanical treatments are most often used in the wildland-urban interface. On Federal lands 86 percent of all mechanically treated acres are in this wildand-urban interface. Herbicide application for fuel reduction is widely used, but estimates of annual treated area for public lands are not available. Dubois and others (2003) reported herbicides were used to control hardwoods in 657,000 acres (266 000 ha) of pine plantations in 2002.

**Table 4.** Advantages, disadvantages, and costs of fuel treatment options being used in the Southern United States

| | Treatment | | | | |
|---|---|---|---|---|---|
| Attributes | Prescribed burn | Mechanical | Manual | Harvesting | Herbicide |
| Advantages | Low cost | Burning easier | Selective | Selective | Effectiveness |
| | Ecological benefits | Use in urban areas | Use in urban areas | Revenue producer | Burning easier |
| | Soil disturbance minimal | | | | |
| Disadvantages | Smoke | Can be costly | Can be costly | Fuel created | Public acceptance |
| | Potential escapes | Fuel created | Fuel created | Potential site damage | Fuel created |
| | Resource damage | Equipment breakage | | | |
| | | Potential site damage | | | |
| Cost *(per acre)* | \$23 to \$121[a] | \$120 to \$350[b] | | | \$68 to \$2,000[c] |
| | | \$35 to \$1,000[c] | | | |

[a] Cleaves and others (2000).
[b] Rummer and others (2002).
[c] Wolcott and others (2007).

# Conclusions

The Subtropical and Savanna Divisions, which include the Coastal Plain and Piedmont of the Southern United States, contained a diverse suite of forest and grassland communities before the arrival of Europeans. The common process that shaped and linked most of these communities was fire. For many, fires were frequent, low-intensity processes that maintained health, functioning, and composition. Others had less frequent but more intense mixed- or stand replacement fire regimes. Fire caused by lightning and Native Americans has long been recognized as the reason that longleaf pine was the most abundant forest community (Chapman 1932). Although other southern pines are not as resistant to fire during their seedling stage, fire was necessary on most sites to maintain pine dominance. Pine, oak, and hickory were dominants on many Piedmont sites because fire gave them a competitive advantage over more mesic hardwood species (Skeen and others 1993). Even the very wet communities—cypress domes, seepage savannas, marshes, and bottomland canebrakes—experienced frequent fire, at least on the edges, because they were imbedded in or adjacent to the pine dominated matrix. Thus, most of the South was driven and molded by fire.

European settlement significantly altered the amount, composition, and age structure of southern forests, as did subsequent years of fire exclusion. Today, active management is required if we wish to restore and maintain those key forests that remain (Van Lear and others 2005). Because the South has a long growing season and plentiful precipitation, it is quite productive and accumulates living and dead fuels rapidly. Thus, fuels management on a regular basis is a necessary part of management, required to reduce wildfire hazard and maintain ecosystem health. Because of its low cost and ecological benefits, prescribed burning is the most common fuels management technique used.

Mechanical systems that target understory and midstory layers have also become widespread, especially for treating forests in the wildland-urban interface. This technique, which includes chopping, mulching, mowing, and hand felling, is usually applied to facilitate a subsequent prescribed burn. Thinning is used in a similar fashion when the stems that need to be removed are large enough to have economic value. Herbicides are also applied to select stands, also to accelerate restoration or facilitate burning.

Most systems are being managed with short fire-return intervals of 3 to 5 years. Other fuel reduction techniques are applied only once if they are followed by burning; otherwise, they need to be reapplied at the same interval of 3 to 5 years. This frequent repetitive treatment over a significant area could cause cumulative effects on the landscape.

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# Fuels Management in the Subtropical Mountains Division

James M. Guldin

## The Ouachita Highlands

The heterogeneity of the forests west of the Mississippi River in the Southern United States is strongly influenced by physiography and topography. The west Gulf Coastal Plain of southern Arkansas, northwestern Louisiana, and eastern Texas features highly productive pine-dominated forests (*Pinus* spp.) on gentle terrain that are interspersed by major and minor alluvial bottomland hardwood forests. The Ozark Mountains are an uplifted eroded dolomitic plateau in northern Arkansas, eastern Oklahoma, and southern Missouri; they feature primarily oak-hickory (*Quercus* spp.–*Carya* spp.) forests with a minor and varying pine component that was far more widely distributed 150 years ago than it is today. Both of these areas support forests similar in species composition and fire dependency as types farther to the east.

Between these two areas lie the Ouachita Mountains of western Arkansas and eastern Oklahoma, among the most ecologically unique ecoregions of the South. Three elements contribute to that uniqueness. First, the general orientation of Ouachita ridges runs from east to west, perpendicular to most other mountains and hills in the continental United States. This points to the second unique element; forest types are closely associated with aspect, with xerophytic pine-dominated forests on the south-facing slopes, and mesophytic oak-dominated forests on the north-facing slopes. The third element is unusually important—the dominance of shortleaf pine (*Pinus echinata*) in the Ouachitas. East of the 100th meridian, shortleaf is the most widely distributed of the southern pines (Guldin 2007), and is generally found in mixture with other pines or in pure stands of limited extent. But in the Ouachitas, shortleaf reaches its ecological maximum, where it is the only naturally occurring pine and the dominant tree species in many stands.

As a result of this unusual ecological association among tree species, forest types, and physiographic conditions, the area has a separate classification as the Ouachita Mountains Mixed Forest–Meadow Province within the Subtropical Division (Division 230). It is somewhat warmer, less wintry, and wetter than the Ozark Broadleaf Forest of the Hot Continental Division (Division 220) to the north. However, it is more prominently mountainous than the Southeastern Mixed Forest Province to the south or the Mississippi Alluvial Bottomland Forests to the east, both of which also lie within the Subtropical Division. And, it is more densely forested than the Prairie Parkland Province that lies to the west—although prairie elements do exist in the Ouachita forests. Finally, the Ouachitas only cover roughly 29 000 km$^2$, making this ecoregion one of the smallest in the South.

## Geologic Origin and Soils

Through most of the Paleozoic Era, up to about 320 million years ago, the area of the current Ouachita Mountains was under ocean water, and deposition of organic and inorganic materials occurred through marine sedimentary processes. But from 320 million to 286 million years ago, during the Pennsylvanian Period, a major tectonic event called the Ouachita Orogen resulted in the collision of what is now North America with a southern landmass. Essentially, the lateral compression from south to north shifted the marine sediments in ways that resulted in considerable folding, faulting, and subduction activity from western Texas to central Alabama (Viele and Thomas 1989). Geologic evidence of metamorphic rocks suggests some volcanic activity especially near Hot Springs, AR (Loomis and others 1994), which is not unusual in the context of prevailing theories of plate tectonics that continue to shape the Earth.

Over the past 280 million years, the major geologic event in the Ouachitas has been weathering and erosion, which have reduced the sandstones and shales that were exposed during the orogeny. The linear ridges of the Ouachita hills still show their folded and faulted history, with long ridges oriented from east to west. The terrain reaches maximum elevation of about 790 m, about 460 m above the adjoining valleys. The side slopes of the ridges are often steep and rugged in the upper slopes, but gradually flatten in the lower slopes. As a result, the hillsides grade into broad U-shaped valleys, whose breadth and gentle gradient are attributed to millennia of creek meanderings, especially along the larger streams and rivers that flow between the ridges.

Soils are highly weathered Ultisols (Buckman and Brady 1969). Pedogenesis is affected by the extremely rocky terrain, the resistance of the rocks to erosion, and the high level of soil stoniness. Phillips and Marion (2004) described the soils on the hillsides and ridges of the eastern Ouachitas as primarily medium-textured, well drained stony Hapludults; on steeper slopes or higher elevation, soils are shallow, whereas on more gentle slopes and benches, soils are moderately deep to deep. Liechty and others (2005) reported that soils in the western Ouachitas are typic Hapludults with loamy surface textures, and having unusually high rock content in surface and subsurface layers.

Site productivity closely follows slope position, with poor sites on ridgetops and upper slopes grading to better sites on lower slopes and floodplains. This common pattern is the result of colluvial activity carrying soils from ridgetops to floodplains over the years, leaving shallow thin soils on upper slopes and consequently deeper soils on lower slopes. Soil depth correlates with both soil moisture and soil fertility. In addition, south-facing slopes get considerably more sunlight than northern slopes. As a result, the south-facing ridgetops are the most xeric and least productive sites, whereas the lower north-facing slopes are the most mesic and feature highly productive sites.

## Climate

Climatic conditions in North America have varied tremendously over the millennia, most recently seen in climatic variations associated with glaciation and interglacial ecosystem processes. However, over the past 4,000 years, pollen records show that the Ouachitas have supported relatively continuous vegetation under a relatively stable climate, but no doubt with annual variations in temperature and precipitation that can occasionally be ecologically important locally (Delcourt and Delcourt 1991, Smith 1984).

Current climatic conditions can be approximated by summaries of weather data over the past several decades. Comparisons using National Weather Service data—and the assumption that Little Rock represents statewide conditions—show that Ouachita Mountains are slightly cooler and slightly wetter than elsewhere in the State. Average monthly temperatures (fig. 1) vary from 3 °C in December (dropping to a negative 3 °C

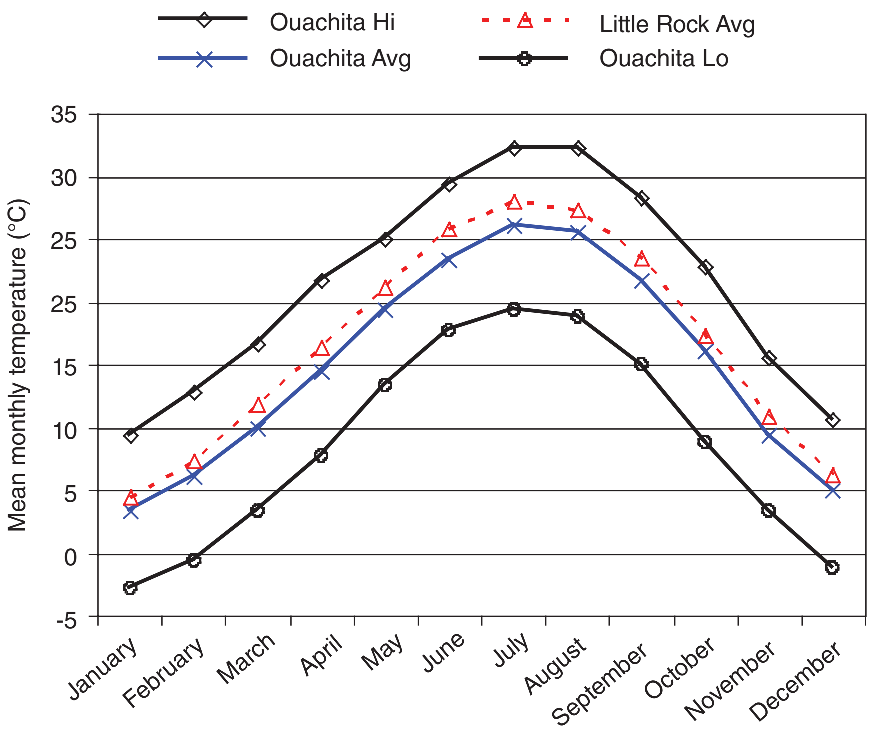


**Figure 1.** Mean monthly high, average, and low temperatures (in degrees Celsius) for the Ouachita region, compared with the mean monthly average temperature at Little Rock, AR. Source: National Weather Service, Little Rock, AR (Web access, active on 1/8/2008, http://www.srh.noaa.gov/lzk/html/climain.htm), Southern Regional Climate Center (Web access, active on 1/8/2008, http://www.srcc.lsu.edu/southernClimate/arkclim/)

average low) and January to 25 °C in July and August (rising to a 32 °C average high). Compared to the statewide average, the Ouachitas are about 1 °C cooler in winter and about 2 °C cooler in summer. Freezes are common, but continuous incidences of daytime highs remaining below 0 °C rarely last a week. Similarly, ice storms and snowstorms occur once or twice a year, but amounts of precipitation as snow are relatively low; 25 cm of snowfall in a single storm is an exceptional event, and snowpack rarely lasts >2 weeks, except on north-facing slopes. Conversely, summer daily high temperatures frequently exceed 40 °C, and the Ouachitas annually experience hot weather in July or August, with daily highs exceeding 38 °C for a week or longer.

Again, based on National Weather Service data, average precipitation in the Ouachita Mountains (150 cm) is about 15 percent higher than the statewide average of 130 cm (fig. 2). This shows the orographic effect of the Ouachitas: moisture-laden clouds that approach the mountains from the west must rise upward to clear the ridges, which condenses water vapor and increases rainfall. May is the only month when average monthly precipitation in the Ouachitas is >15 cm; whereas January, February, and August have average monthly precipitation <10 cm. In May, June, July, September, and October, the Ouachitas average 3 to 4 cm more precipitation than the State.

Scarcity of rainfall in August interacts with high temperatures to create conditions favorable for drought, which has the important ecological function of controlling which tree seedlings and other vegetation will have enough moisture from the soil to survive. Lightning is also common in the summer months, and the combination of dry vegetation and lightning strikes renders forests at risk from wildfire. Moreover, strong evidence suggests that Native Americans burned the landscape (Guyette and others 2006). This all suggests an ecological condition in which forest fires were an important agent of ecological disturbance before European settlement.

## *Physiographic Variations*

The U.S. Department of Agriculture Forest Service developed an ecoregion framework for the Eastern United States (Keys and others 1995) based on a national map of ecoregions of the United States (Bailey 1995, Bailey and others 1994). A more

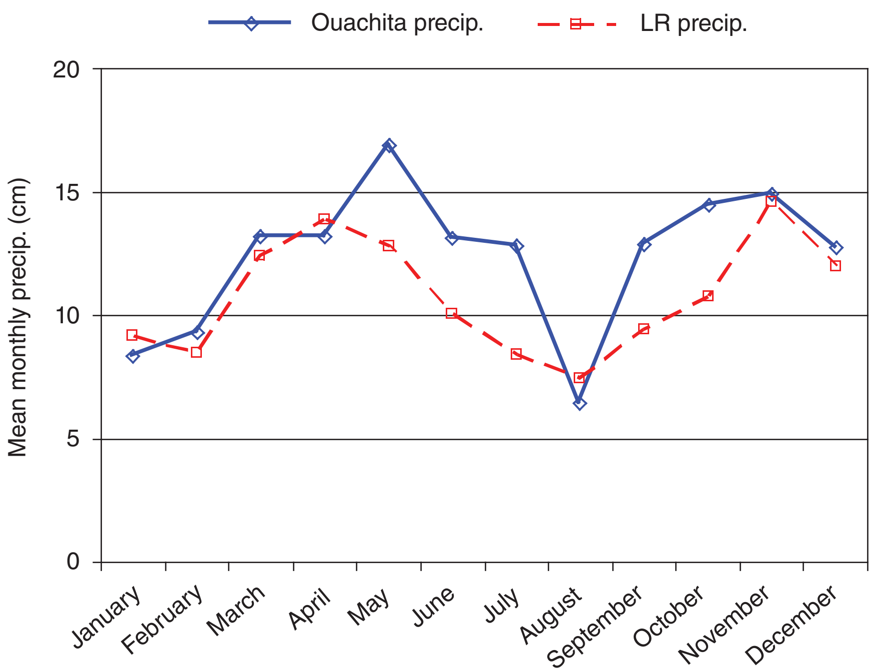


**Figure 2.** Mean monthly precipitation (cm) for the Ouachita region and for Little Rock, AR. Source: National Weather Service, Little Rock, AR (Web access, active on 1/8/2008, http://www.srh.noaa.gov/lzk/html/climain.htm), Southern Regional Climate Center (Web access, active on 1/8/2008, http://www.srcc.lsu.edu/southernClimate/arkclim/)

recentmap, which is based on slight movement of borders using local knowledge and experience, contains the best detail available with respect to coverage (Foti and Bukenhofer 1999).

The Ouachitas lie to the south of the Arkansas River Valley in west-central Arkansas and eastern Oklahoma (fig. 3). The area is classified in the Subtropical Division, Humid Temperate Domain (200), as the Ouachita Mixed Forest–Meadow Province (M231), also referred to as the Ouachita Mountains Section (M231A). Within the section are four prominent subsections—the Fourche Mountains (M231Aa) to the north, the Western Ouachita Mountains (M231Ab) to the west, the Central Ouachita Mountains

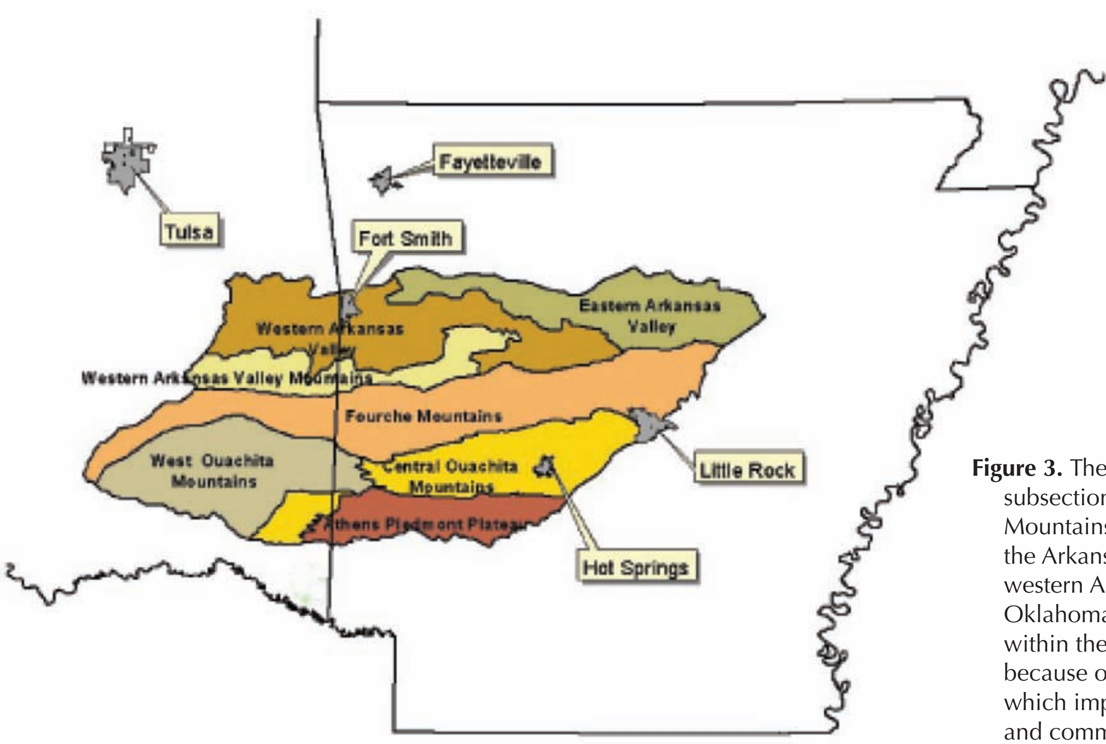


**Figure 3.** The ecological subsections in the Ouachita Mountains lie to the south of the Arkansas River Valley in western Arkansas and eastern Oklahoma. Cities are absent within the Ouachitas largely because of the rugged terrain, which impedes transportation and commerce.

(M231Ac) in the east-central area, and the Athens Piedmont Plateau (M231Ad) in the southeastern-most reaches (Foti and Bukenhofer 1999). Overall, these four sections encompass roughly 2.9 million ha.

The Fourche Mountains subsection occupies 1 180 414 ha. Ridges are moderate to high, containing some of the highest ridgetops in the area, and slope into broad valleys; elevation varies from 250 to 790 m. Ridges are underlain by Pennsylvanian and Mississippian sandstone and shale, and valleys consist of sandy residuum (Foti and Bukenhofer 1999). The area is 78 percent forested (Guldin and others 1999), the lowest percentage in the Ouachitas; stands are dominated by shortleaf pine (47 percent of forested area), oak-pine (29 percent), and oak-hickory (22 percent).

The Western Ouachita Mountains cover 679 100 ha primarily in Oklahoma, where the ridges are high and relatively steep, again with broad valleys, varying from 250 to 760 m. The area is composed of Mississippian sandstone and shale with clayey colluvium in the valleys (Foti and Bukenhofer 1999). Nearly 88 percent of this area is forested; of that, 55 percent is in pine-dominated stands, 29 percent is in oak-pine stands, and 15 percent is in oak-hickory stands (Guldin and others 1999).

The Central Ouachita Mountains encompass 663 000 ha in two separate areas—a small part in Oklahoma and the larger part in Arkansas. Elevation varies from 250 to 760 m in open wide hills, low mountains, and wide valleys; the underlying geology is Mississippian sandstone and shale with clayey colluvium in the broad valleys (Foti and Bukenhofer 1999). About 82 percent of this area is forested. It has the highest proportion of oak-hickory stands (48 percent) in the Ouachita Mountains; pine stands account for 39 percent of the forested area, and oak-pine stands only 10 percent (Guldin and others 1999).

The Athens Piedmont Plateau is the smallest subsection of the Ouachita Mountains, covering 367 500 ha in the southeast. Geologically, this area is the first uplift from the upper west Gulf Coastal Plain immediately to the south. Elevation varies from 250 to 750 m, and includes open high hills underlain by Mississippian and Pennsylvanian sandstone, with valleys built on sand and clay-loam colluvium (Foti and Bukenhofer 1999). This area has the largest percentage of forest area (91 percent) as well as a concentration of forest industry ownership. Pine-dominated stands make up 73 percent of the forested area, oak-pine stands only 6 percent, and oak-hickory stands 19 percent of the area (Guldin and others 1999). In this subsection more than the others, forest industry is converting oak-pine stands to plantations of loblolly pine (*Pinus taeda*).

### *Ownership*

Ownership of timberland in the Ouachitas is roughly divided equally among three major classes. The public owns 29 percent of the timberland, with 85 percent of that managed by the Ouachita National Forest. Forest industry holds 37 percent, with nearly two-thirds in the Western Ouachita Mountains and Athens Piedmont Plateau. Other private (nonindustrial) forest landowners hold the remaining 34 percent (Guldin and others 1999). The recent divestiture of forest industry land to other individuals and organizations represented by timberland investment management organizations (TIMOs) or real estate investment trusts (REITs), which is nearly complete elsewhere in the South, has just commenced in the Ouachitas.

## Forest Stands in the Ouachitas

The native forest types in the Ouachita Mountains vary from stands that are heavily dominated by shortleaf pine and pine-hardwood mixtures to oak-hickory stands that are dominated by hardwoods with only a minor pine component, if any. Closed-canopy forests are typical. However, open woodlands were probably more common 200 years ago, because of the changing midstory and understory forest conditions that resulted from effective fire control over the past 80 years. In addition, under forest industry ownership, large areas of native shortleaf pine-dominated stands have been converted from to plantations of loblolly pine.

## Pine Dominated Stands

### Natural pine forests

In these native stands, naturally regenerated shortleaf pine is the dominant tree. Where fire has been excluded, the result is a prominent hardwood midstory and understory (fig. 4), potentially complicating silvicultural practices intended to maintain the pine component or regenerate pines after harvesting old stands. As an alternative to fire exclusion national forest lands, specialized silvicultural systems for ecological restoration have been developed that reduce the overstory density of shortleaf pines, remove midstory hardwoods, and reintroduce a cyclical prescribed burning program (fig. 5). The goal is to maintain the production of high-quality pine sawtimber while concurrently restoring understory grasses, sensitive plant species such as the purple coneflower (*Echinacea purpurea*), and endangered animal and insect species such as the red-cockaded woodpecker (*Picoides borealis*) and the Diana fritillary butterfly (*Speyeria diana*). The value of robust local timber markets has been a critical factor in the success of this restoration (Guldin and others 2004b).

Average conditions were quantified for a "typical" Ouachita stand in a study of mature unrestored second-growth stands of shortleaf pine and pine-hardwoods on south-facing slopes on national forest land (Guldin and others 1994). This "typical" stand has a stem density of approximately 800 trees/ha, of which about half are pines and the other half are hardwoods. Basal area is roughly 30 $m^2$/ha in trees >10 cm d.b.h. (diameter at breast height); about 75 percent is pine and 25 percent is hardwood, half of which is in midstory trees 10 to 24 cm d.b.h. The average conifer is larger than the average hardwood, with the quadratic mean diameter (the diameter of the tree of average

**Figure 4.** Mature second-growth shortleaf pine-dominated stand in the Ouachita Mountains, with typical development of midstory and understory in the absence of prescribed fire. (Photo by James M. Guldin)

**Figure 5.** Mature second-growth shortleaf pine-dominated stand in the Ouachita Mountains after overstory thinning, midstory reduction, and reintroduction of cyclic prescribed burning. (Photo by James M. Guldin)

basal area) for conifers at 26 cm, compared to 16 cm for hardwoods. Pines age classes are bimodally distributed, with peaks in the 10- to 15-cm class (suppressed trees that still persist) and the 25- to 30-cm class, and with only 30 trees per ha ≥40 cm d.b.h. The dominant conifer is shortleaf pine, and the only two common overstory hardwoods in the typical stand are post oak (*Q. stellata*) and white oak (*Q. alba*). In the midstory, shortleaf pine also dominates with the most common associates being post oak, white oak, mockernut hickory (*Carya tomentosa*), and black or Texas hickory (*Carya texana*). In the understory, however, 13 tree species are prominent, including post oak, white oak, mockernut hickory, black hickory, winged elm (*Ulmus alata*), black oak (*Q. velutina*), and blackjack oak (*Q. marilandica*).

## Loblolly pine plantations

The second pine-dominated forest stand in the Ouachitas is a local exotic, the loblolly pine plantation. Forest industry and some private landowners use intensive silvicultural prescriptions to harvest shortleaf pine and oak-pine stands, and to replace that native vegetation with artificially regenerated plantations established using seedlings of genetically improved loblolly pine from Arkansas, Oklahoma, or North Carolina. These loblolly pine plantations are managed primarily for timber products over rotation lengths of 27 to 35 years. The silvicultural systems used to manage them typically include ripping to promote seedling establishment and development, broadcast herbicide application and fertilization to enhance the growth of the pines and decrease the competition, thinning to maintain adequate growth, and pruning to ensure development of clear wood in the butt log of the crop trees (fig. 6). Compared to naturally regenerating shortleaf pine, these practices are much more effective in meeting the goal of rapidly growing wood fiber, with conservative estimates of 20 to 30 percent gain in volume production (Lambeth and others 1984).

The ecological concern is that loblolly pine is native only to the southeastern-most part of the Ouachita Mountains. Throughout most of their natural southern range—and

**Figure 6.** Loblolly pine plantation after prescribed burning and first thinning near Waldron, AR. (Photo by James M. Guldin)

notably immediately to the south of the Ouachitas in the upper west Gulf Coastal Plain—loblolly and shortleaf pines can be found in mixture, with loblolly usually the dominant pine. But research and photographic evidence from 70 years ago refer to second-growth "shortleaf-loblolly" pine-hardwood type stands (Reynolds 1947), which may refer to a more robust shortleaf-pine presence in mixture with loblolly pine and hardwoods compared to domination by loblolly pine today. The difference may be due to the different regeneration dynamics of these two species. As discussed above, shortleaf is a less prolific seed producer, but resprouts if topkilled by fire; compared to loblolly, which will not recover if topkilled by fire. The tactic for loblolly seems to be in producing prolific annual seed crop, dropping adequate or better seedfall four years in five (Cain and Shelton 2001).

As one crosses the ecotone northward from the Coastal Plain into the Ouachitas, loblolly drops out of the native forest completely and abruptly, within a span of 20 to 30 miles. This unusually rapid change in species composition suggests a major ecological influence at work. But the nature of that influence is unclear and is further clouded by the generally successful development of loblolly pine plantations in the Ouachitas. These plantations grow and reach reproductive maturity rapidly, and observation shows successful loblolly pine regeneration beneath planted parents. This is not the developmental dynamic one would expect from a species that was essentially absent from the mountains during presettlement times.

Moreover, industry experience with loblolly pine plantations began in the early 1970s; in the past 40 years, only two ecological events have adversely affected these plantations to any important degree. The first—a prolonged drought in 1980—resulted in some mortality, especially of seedlings from North Carolina families that had been planted on some of the driest Ouachita hillsides (Lambeth and others 1984). The second was an ice storm in December of 2000, which only caused mortality in late-teen stands that had been recently thinned (Bragg and others 2003). So although some planted loblolly pine stands have been adversely affected by natural events, none has been sufficiently large or long lasting to fully explain loblolly pine's natural absence from the Ouachitas.

### Shortleaf pine plantations

The final and least widespread pine-dominated forest stands in the Ouachitas are shortleaf pine plantations, which are typically established on national forest lands and occasionally established on private lands after clearcutting or in response to rehabilitation of cutover or understocked stands. The use of shortleaf pine rather than loblolly on national forests relates to the general Forest Service mission of managing native ecosystems for native flora and fauna. The oldest shortleaf pine plantations on the Ouachita National Forest are ≥70 years, dating to the 1930s when Civilian Conservation Corps workers were assigned to reforestation work. Today, one would not recognize the older stands as plantations, because the easily detected rows in which seedlings were planted have generally become less obvious as stands have matured (Rosson 1995).

Establishment of shortleaf plantations generally follows the intensive treatments prescribed for loblolly plantations on private land with two differences: substitution of individual-stem applications of herbicide rather than broadcast treatment, and absence of pruning. The genetically improved seed source for shortleaf pine used in these plantations comes from Ouachita families maintained in a seed orchard in Montgomery County, Arkansas, in the eastern portion of the Central Ouachita Mountains. However, clearcutting is required when using artificial regeneration such as planting; with the decline of clearcutting on national forests from the mid-1980s to the mid-1990s, the establishment of new plantations has also declined dramatically (Guldin and Loewenstein 1999). Since 2000, clearcutting followed by planting shortleaf pine has been used on slightly >500 ha annually on the Ouachita National Forest; in all instances, the goal has been to reforest understocked stands or to convert cutover loblolly pine plantations acquired from forest industry back to shortleaf pine.

## *Oak-Hickory Stands*

Oak-hickory stands represent the opposite end of the silvicultural spectrum from pine-dominated stands in the Ouachita Mountains with respect to species composition, topography, and intensity of management. These stands are most commonly found at two topographic extremes in the Ouachitas. In the highest elevations, stands dominated by post oak, blackjack oak, some white oak and black oak, and black hickory occupy the steep south- and north-facing thin-soiled slopes and the ridgetops that are too exposed or too xeric for pines. Some of the most interesting stands are the stunted oak-hickory stands on the ridges at the Rich Mountain and Black Fork Mountain summits, where dominant oaks can be >100 years in age and yet be ≤3 to 10 m tall, in part caused by wind and ice (Johnson 1986). Stands such as these support old-growth remnants, and are important sources of dendrochronological records for analyses of disturbance and changing climate (Stahle and Hehr 1984).

Conversely, stands that feature white oak, southern red oak (*Q. falcata*), black oak, red maple (*Acer rubrum*), and sweetgum (*Liquidambar styraciflua*) can be found in mesic conditions on flat or gentle terrain along ephemeral and perennial streams on low north-facing and south-facing slopes; in many respects, these are the most productive sites anywhere within the Ouachitas. On the more mesic sites of lower slopes, white oak can become especially important, and the species could probably be commercially managed for timber under reasonable rotation lengths. The mesic Ouachita oak-hickory stands will be dominated by white oaks, especially on more mesic sites, on more xeric sites, the tendency will be toward post oak. The red oaks—such as southern red oak, black oak, and blackjack oak—are also found, though slightly less commonly than the white oaks. Other common species include winged elm, sweetgum, red maple, and flowering dogwood (*Cornus florida*).

Ouachita oak-hickory stands are rarely managed specifically for timber products, primarily because the demand for hardwood sawtimber and pulpwood is low. Most of the sawmills in the Ouachitas are dedicated to pine; the few hardwoods that loggers are willing to take for firewood or other merchandising opportunities are easily found in harvested pine stands. Moreover, recent forest plans for the Ouachita National Forest have not emphasized hardwood management for timber production, although the plans' standards and guides allow for silvicultural activities as needed in hardwood stands to improve or restore ecosystems, manage or restore key species of flora and fauna, and promote hard and soft mast production for wildlife. In landscapes that are being aggressively restored with prescribed fire, the area burned in a single fire has in some situations been >1000 ha at a time. No effort has been made to deliberately exclude hardwood stands from these large burn units, largely because periodic fires undoubtedly have a role in maintaining healthy and sustainable ecological conditions in these stands. Allowing large-scale fires to spread as they will also places an appropriate degree of natural variation within the burn unit, as the fires will burn with less intensity and even die out on the most mesic sites.

## *Oak-Pine Stands*

The delineation between pine-dominated stands and oak-hickory stands in the Ouachita Mountains is rarely discrete. A transect northward over an Ouachita ridge shows a mesic oak-hickory stand next to a creek, a pine-dominated hillside midway up the southern slope, a pine or oak-hickory stand on the ridgetop, a pine-oak or oak-pine stand on the upper northern slope, and a white oak-dominated oak-hickory stand on the lower northern slope. Almost all of these stands contain both hardwoods and pines; but hardwoods are more likely than pines to occupy the midstory and understory unless subjected to surface fires. The varying proportion of oaks and pines is as much a product of past stand development and disturbance patterns as it is a stable representation of the intermediate stand condition.

Historical accounts (Smith 1986) outline a systematic harvesting of virgin shortleaf pine stands through the Ouachitas from 1880 to 1920. This activity spread from south to north; railroads were constructed through themountain passes in the rugged terrain, and then branched out from east to west through the valleys to the ridges. Merchantable pines were cut to a 12-inch diameter limit, and horselogged through the network of creek drainages to the railroads in the valleys below. Smaller pines were not cut, and these responded to the suddenly open conditions with continued growth. Larger pines that were rotten, hollow, and otherwise not useful for lumber were also left behind, but they were capable of producing seed to reforest the site. Harvested stands may also have had some shortleaf seedlings and saplings as advance growth; if they were present, many would resprout after logging. Thus, varying amounts of pine of various size and vigor were probably left uncut after the harvesting of the virgin stands. Hardwoods, too, were left on the site, especially smaller diameter hardwoods that would not even have valued for local use as lumber, fuelwood, or railroad ties.

The next influence would be an uncontrolled surface fire. One might speculate that with frequent or intense fire, seedbeds would be created for pine seed to germinate, advance growth seedlings and saplings would resprout, hardwoods would be killed, and the subsequent stand would likely to be colonized by pines. If surface fires were infrequent or if fires were controlled, the hardwood residual trees and sprouts would be favored; the pines that successfully competed in the stand would also persist, but at lower densities. The influence of the Civilian Conservation Corps in fire suppression during the 1930s may thus have been important in the development of oak-pine stands.

Across the South, the acreage in naturally regenerated pine and oak-pine stands has decreased and pine plantations have increased (Conner and Hartsell 2002), a trend that is prominent in the Ouachita Mountains as well. Certainly some of the loblolly pine plantations being established by forest industry were planted on sites that had previously supported oak-pine stands with the goal of increasing the volume of pine.

Throughout this chapter, the focus is on deciding what to manage and with what tools. The Ouachitas have such a variety of conditions, soils, and species that foresters working within different ownership sectors can easily develop whatever spectrum of pine, oak-pine, or oak-hickory stands that they think is appropriate to meet the prevailing objectives of the landowner.

# Fire as an Element of Ouachita Ecosystems

Fire has been important in the ecosystems of the Ouachita Mountains for thousands of years. The evidence for this is found in the analysis of fire scars from old pines, from historical observations of explorers and surveyors, and from an understanding of the life cycles, requirements, and vulnerabilities and of the plants that are found in these forests. Fire scar analysis reveals changing patterns in fire occurrence over time. Presettlement fires generally occurred on the order of every 7 to 20 years (Foti and Glenn 1991). However, in the two centuries since, the fire-return interval has become much longer, with some estimates as low as one occurrence every 1,200 years (Johnson and Schnell 1985). Thus, for some unknown presettlement period, fires occurred frequently, but have since occurred much less frequently. This has important implications for the dynamics and development of forest ecosystems, and for their management.

Presettlement fire occurrence was a combination of natural and deliberate ignitions. Guyette and others (2006) compared fire occurrence to historical Native Americans populations, and showed a close correlation between population and fire scars—strong evidence that Native Americans used fire as part of their daily lives. Benefits from the use of fire were probably related to the open understory conditions that burning creates. One might speculate that those benefits would include controlling ticks and chiggers, promoting grasses and browse for wildlife, and clearing openings for agricultural use. In addition, projectiles such as arrows will fly longer and more accurately in the open rather than through brush, which would have value both in hunting and perhaps also in community defense from attacks by wildlife or aggressive neighbors.

In the 1930s, the need for fire control arose from wildfire in cutover stands that had become a threat to resource management and conservation. The first field survey of Arkansas, conducted in 1929, reported that of the 22 million total acres of land remaining in forest at that time (about two-thirds of the area of the State), 20 million had been cutover—70 percent of which had been severely damaged by wildfires, with millions of acres burned annually (Beltz and others 1992, Roberts and others 1942). The need for fire suppression and control was an important element in the expansion of the forestry profession especially in State agencies such as the Oklahoma Forestry Commission (now Oklahoma Forestry Services) established in 1925, and the Arkansas Forestry Commission, established in 1931. Firefighting was a primary reason for their establishment, but staffing in Federal and State agencies was inadequate to control wildfires effectively until the end of World War II when the GI Bill for war veterans provided an educational boost to the forestry profession.

The combination of harvesting the virgin forest, rampant wildfires, and effective fire suppression over a 70-year period (1930 to 2000) profoundly altered forest ecosystems in the Ouachita Mountains. The change was especially pronounced in reproduction dynamics and stand development. Ecologically, a vigorous midstory woody vegetation component thrived in the absence of fire. Excluding fire over these seven decades led to a change in habitat conditions from open forests and woodlands to closed canopy forests with a prominent midstory, causing a decline in species that thrived in open forest and woodland conditions such as wood bison (*Bison bison athabascae*) and North American elk (*Cervus elaphus canadensis*), both extirpated from the area in the 19th century. Also greatly reduced in extent were prairie flora such as purple coneflower, bluestem (*Andropogon* spp.), flowering plants such as birdfoot violet (*Viola pedata*), pollinators such as the Diana fritillary butterfly, and birds such as the cavity-nesting red-cockaded woodpecker, which is currently a federally listed endangered species.

The regeneration ecology of shortleaf pine and the oaks closely follow this natural dynamic. Fires benefit the establishment and development of shortleaf pine for a number of reasons. Most pines germinate best on exposed mineral soil; fires promote a patchy distribution of mineral soil for optimum seed germination and seedling establishment. Shortleaf pines up to about 8 years also have a unique trait not shared by the other southern pines; the ability to resprout if topkilled. The significance of this was appreciated early on when Mattoon (1915) described it as an adaptive advantage in response to frequent fires. The importance of this trait is that if fire burns a cutover or understocked stand, subsequent regeneration of the pines can occur either through seedfall or resprouting of the existing advance-growth seedlings and saplings.

Similarly, the oaks are adapted to advance-growth regeneration dynamics (Johnson and others 2002) in which resprouting and dieback of seedling and sapling shoots continue over time, enabling the development of a robust rootstock and eventual establishment of a sapling to grow into the midstory and overstory. In the absence of fire, the oak shoot will persist in the understory until overstory shading causes it to die back to the root collar. This shade-induced mortality is a slow process. Conversely, with frequent surface fires, the process of growth and dieback of the shoot is faster, the rootstock grows faster, and the development of the sapling into the midstory and overstory is more expeditious.

With fragmentation of the Ouachita by different ownerships and varying degrees of agricultural development, forest managers are unlikely to see widespread fires restored across the entire area, but restoration of large fires in some areas is feasible. An example of is found in the Shortleaf Pine–Bluestem Management Area on national forest land to the west. Here, managers have developed prescriptions for ecosystem restoration that use commercial timber sales to reduce overstory density and mechanical treatments to remove the midstory hardwoods that have developed under seven decades of fire exclusion, and afterward have reestablished a program of cyclic prescribed burning (Guldin and others 2004b, Hedrick and others 2007). When restoration has been fully competed, about 100 000 hectares of the Ouachita forest land will have a structure and function similar to presettlement conditions.

However, management activities on the remaining 97 percent of the Ouachita landscape will likely not support sustained cyclic prescribed burning, but nevertheless must reduce fuel levels so as to minimize risk of loss to wildfire, an increasingly important consideration for the expanding wildland-urban interface.

# Fuel Management in the Context of Silviculture

From a forest management perspective, the vegetation that has developed in the Ouachita ecosystems—as overstocked overstory trees, excessive numbers of midstory trees, and standing or downed dead trees and branches—is considered biomass that has accumulated as a result of fire exclusion. It is also flammable material that can maintain, support, increase fire intensity and otherwise exacerbate conditions associated with wildfires. Fuels treatments represent a subset of intermediate silvicultural treatments, and so are specifically designed to reduce that material in the short term (ch. 2) thereby altering the behavior of wildfires should they occur. But a more profound impact on forest management is made, not through short-term stopgap solutions to fuels, but in long-term programmatic management practices that integrate fuels treatments with the larger long-term objectives of the landowner. Fuels treatments are therefore more robust if they are examined as part of a larger and integrated program of silvicultural treatments called a silvicultural system (Smith and others 1997).

Individual silvicultural treatments can target several categories of biomass: the forest site, the forest floor, the woody vegetation in the main canopy, the woody and nonwoody vegetation in subordinate canopy positions, and the residues of vegetation. While these treatments are designed to achieve specific goals in forest stand dynamics and development, all have ancillary effects on the accumulation or reduction of biomass residues when viewed from the perspective of wildfire hazard and risk.

## *Identification of Fuels in a Silvicultural Context*

A silvicultural system is little more than a long-term plan for the stand being managed. It is implemented using a silvicultural prescription containing a planned set of treatments—each applied at a given point in time—that guide the stand to its desired future condition. But in some situations, events conspire to interrupt the long-term plan. Often, that event is triggered when enough plant material exists in the forest stand to pose a threat to the continued life of the stand if it is subjected to an uncontrolled fire.

Fuels are the living vegetation and detritus from dead vegetation that accumulates in the forest through natural or managed events. They are found as logging slash, pruned branches, and vegetation in the various strata of the forest. The term can include living trees and nonwoody vegetation, and also dead material that is still attached to live standing trees—dead snags—or dead material that has fallen to the forest floor but has not yet decomposed.

The biomass of material that can be called fuels changes during the course of a rotation. It follows that different periods of stand development differ in the amount or kind of fuels that are produced. It also follows that the main canopy of the stand is more susceptible to loss from fire at some periods than others, independent of biomass amount.

The absolute level of biomass is of less concern for fuels treatment than is the effective implementation of treatments placed in the right stands at the right time. Without timely or effective fuels treatments, fires can ignite a given level of biomass that is distributed in certain ways at highly sensitive times of year, resulting in the loss of the entire stand. In uncontrolled fire conditions, the resulting conflagration will jump from stand to stand, and losses will accumulate unacceptably across the landscape before the fire can be contained.

Questions about what constitutes fuels and when and how fuels should be treated are complicated by the fact that wood and wood fiber have monetary value. In the ideal world, fuels would be treated as an element of broader silvicultural treatments that involve identifying a desired complement of trees to retain, harvesting and selling trees that are surplus to the desired complement, and then using some of the proceeds from the commercial sale to reduce any residual fuels to an acceptable level. The situation is made less than ideal if there are no local markets for the commercial sale, if harvesting is precluded in a stand for some reason, or if natural disturbance events adversely affect the commercial value that a stand might have.

Federal forest managers have two sources of funds for fuels treatment: timber sale proceeds that can be reinvested to manage fuels in the harvested area, or funds appropriated through Congress. A program of fuels treatment that relies on sale proceeds is be more effective in the long term, because it allows larger areas to be treated more rapidly, making a faster and more durable ecological change on the landscape—essentially, a sustainable stand structure in a fuels context. That then allows the scarcer appropriated funds to be applied strategically in stands that are not in a condition, or a location, conducive for the timber sale process. Private forest landowners have far fewer opportunities to tap Federal funds for fuels treatment, except through landowner assistance funds, which are both scarce and competitively distributed. Thus, fuels treatments are unlikely on private lands unless they can be supported by proceeds from harvesting in the stand that requires treatment.

This also explains the interest in biomass as an energy source. Wood fiber that has previously been too small for commercial use might become commercially operable if markets for biomass and biofuels can be developed. That potential could lower the size threshold for commercial value, allowing smaller material (perhaps including branches and twigs) to be sold. This might have ecological implications if carried to extremes, but it would be useful if smaller standards for merchantability could allow more stands to be self-sustaining in fuel treatment costs.

## *Regeneration Treatments*

Both natural and artificial regeneration is used to regenerate shortleaf-dominated forests in the Ouachitas, whereas loblolly pine plantations are by definition established with artificial methods. Planting either loblolly or shortleaf pine after clearcutting is not a trivial matter, because of the extreme stoniness of the soils and the late summer droughts common in the area. Two approaches have enhanced plantation survival. The first is to plant a seedling with a big root collar (Brissette and Carlson 1992), which promotes root development during the growing season and enhances the chances of survival. The second is to prepare a suitable planting spot through an intensive site preparation technique called ripping or subsoiling—using a bulldozer to plow a furrow into which the seedling is subsequently planted. In combination, these practices improved plantation survival in the Ouachitas by 10 to 30 percent (Walker 1992).

Natural regeneration of shortleaf pine and hardwoods in the Ouachitas can be accomplished using either even-aged or uneven-aged methods, but some methods are more effective than others (Guldin and others 2004a). Studies show that shortleaf pine produces only three to five adequate or better seed crops per decade (Shelton and Wittwer 1996, Wittwer and others 2003); moreover, seedfall varies geographically with higher amounts in the eastern Ouachitas and lower amounts in the west. Research scientists have more work to do to better quantify regeneration dynamics and development in shortleaf pine stands, because there are some yet-to-be-answered questions about stocking and distribution of regeneration resulting from their application.

Practicing silviculturists in the area see administrative advantages in using group selection rather than single-tree selection in uneven-aged stands. Logging is less damaging because the group openings serve as logging decks, the groups can be drawn on a map to assist contractors site preparation and release treatments, and the matrix of groups can be developed to retain hardwoods for wildlife and aesthetics. However, these are attributes of convenience in application more than an indication that one method works better than the other.

The sprouting habit of shortleaf pine might be useful in silvicultural applications that are applied to increase pine regeneration by supporting both new seedlings and sprouts from established sapling rootstocks to regenerate a stand (Guldin 2007). A properly timed surface fire in a stand with some existing shortleaf pine saplings will result in top-killed seedlings that subsequently resprout, and will also create exposed seedbed conditions favorable to germination of new seedlings. Repeated fires of proper intensity thus serve the dual advantage of both controlling fuels and developing a cohort of pine saplings and sprouts to naturally regenerate the site after disturbance or reproduction cutting.

Although no studies have been dedicated to oak regeneration in the Ouachitas, many studies in upland forests (including the Ozark Highlands of Arkansas and Missouri) suggest that the principles of oak regeneration established elsewhere would most likely be successful in the Ouachitas. The commonalities are twofold: first, successful oak regeneration depends on the presence of competitive regeneration sources before substantial overstory is removed; and second, in the absence of regeneration sources, treatments to develop competitive oak regeneration sources should be applied a decade or two before harvesting (Johnson and others 2002, Loftis 2004). As with the pines, the objective is to accumulate enough sources of oak regeneration (such as seedlings, saplings, and stump sprouts), so that the probability of successful establishment is high.

The first step is to evaluate the existing oak regeneration potential in the stand using established guidelines (Sander and others 1984), and decide if supplemental regeneration sources are needed before reproduction cutting. If so, one should wait for an abundant acorn crop, underplant oak seedlings, or both. Controlling competing vegetation in the understory and midstory is important to promote the development of the oak seedlings and seedling sprouts.

The role of prescribed fire as part of a regeneration prescription for oaks is not fully understood, but one might expect fire to contribute to maintaining or increasing the vigor of seedlings and saplings through top-killing and resprouting , as well as controlling fire-intolerant competing species in the understory. Numerous studies are underway to investigate fire effects on oak regeneration and to better define how it might be used.

## *Intermediate Treatments*

Three practices form the bulk of intermediate silvicultural treatments for the Ouachita Mountains—release, thinning, and pruning. All have effects to be considered for fuels management, because the wood produced during intermediate treatments is often of marginal commercial use.

In the Ouachitas, release treatments typically remove small hardwoods or herbaceous plants that compete with young pines (<10 years). Release methods can be chemical, mechanical, or ecological. Herbicides offer the most permanent approach to competing vegetation because both shoots and roots are killed and resprouting is minimized. If topkilling the hardwoods allows the pines to prosper, mechanical treatments and prescribed burning treatments would be appropriate. However, prescribed burning in young stands requires an experienced crew and a cool fire; an effective combination is to ignite backing fires using hand tools in the coldest months of the dormant season. But winter burning may not be as effective in controlling the resprouting of hardwoods as a late spring or summer burn.

Thinning in immature and mature stands reduces stem density of trees primarily by removing trees of poor quality, form, and vigor, thereby promoting health and vigor in the trees that remain (Helms 1998). In the Ouachita Mountains, thinning is used in both pine and hardwood stands, but treatment acreage of pines is far higher—not surprising, given the emphasis that Federal land managers and forest industry foresters place on the management of pine-dominated forest stands.

Almost by definition, thinning is primary tool that foresters have to reduce the volume of fuels in forest stands. At the stand level, thinning reduces biomass in rough proportion to basal area; retaining 75 percent of basal area after thinning will result in about the same proportion of retained biomass. The pattern of thinning might affect the size class and distribution of the biomass being removed, possibly resulting in some treatments being more effective than others. A key consideration is whether the thinning can be conducted using a commercial timber sale. Payments made to the landowner from timber sales can be reinvested into treatments that further reduce fuels, especially fine fuels such as branches and tops that might otherwise have to be hauled from the stand during logging.

Precommercial thinning, or thinning in stands too small to sell commercially, is the biggest single challenge in fuels treatment for those stands. Stands that are candidates for precommercial thinning in the Ouachita Mountains are usually overstocked with small trees of marginal to no commercial value, with a high number of stems, dead trees standing or down, and dead needles draped over the lower branches of the trees. Such stands are at a high hazard of loss from fire. The two available treatment options are both costly. The first is to conduct the precommercial thinning using either appropriated dollars on public lands or out-of-pocket dollars on private nonindustrial lands. The second has high risk cost: wait until the stand grows to commercial size, hoping that it does not burn in the meantime, and then prescribe a commercial thinning.

Pruning is a relatively unusual intermediate treatment in many forests, but it is a common treatment in the Ouachitas for loblolly pine plantations that are managed by forest industry for wood production. This treatment removes living and dead branches from the stems of trees up to a certain height (3 to 6 m), so that the wood that is produced afterward is free of knots. The byproduct of the treatment is a mat of dead branches and needles around the base of the tree. Because all these branches and needles are close to the ground, natural decomposition reduces this threat after a year or so. However, should wildfire occur during that short period of time, the hazard and risk of widespread mortality is high.

## Reproduction Cutting Methods

The first indicator of forest sustainability is found at the stand level—when a reproduction cutting is made, whether a new cohort of the desired species is successfully established in conditions that will allow it to grow and develop in an acceptable manner. Even-aged and uneven-aged methods are both used for sustainable forest management in the Ouachitas.

## Even-aged pine regeneration

Clearcutting is common in the Ouachita Mountains, especially in pine-dominated stands on forest industry lands. The typical silvicultural prescription for regenerating loblolly pine is to clearcut the stand, utilizing as much biomass as can be removed; conduct supplemental site preparation treatments to dispose of logging slash and competing vegetation as needed; use ripping to prepare the site for planting; and then to plant with genetically improved stock selected for rapid growth and some degree of drought tolerance. On public lands, improved shortleaf pine planting stock is substituted. Clearcutting has been a controversial practice in the Ouachitas because of the unsightly appearance of harvested stands. But there is no question that, silviculturally, clearcutting is an effective method that quickly results in the establishment of a new fast-growing stand of desired species.

The seed-tree (fig. 7) and shelterwood methods (fig. 8) are more commonly applied on national forest lands, where management plans call for retention of some residual seed trees through the life of the new age cohort to provide structural diversity in the new stand. On private lands, the landowner may chose to remove the seed trees after the new age class is adequately established.

## Uneven-aged pine regeneration

Uneven-aged silviculture has been used in the Ouachitas since the 1950s by family lumber companies and forest industry landowners. The single-tree selection method is used occasionally to grow large high-quality shortleaf pine sawtimber (fig. 9).

**Figure 7.** Shortleaf pine and hardwood regeneration 12 years after seed cutting under the seed tree method in a shortleaf pine stand near Mount Ida, AR. (Photo by James M. Guldin)

**Figure 8.** Shortleaf pine and hardwood regeneration 12 years after seed cutting under the shelterwood method in a shortleaf pine stand near Waldron, AR. (USFS photograph by James M. Guldin)

**Figure 9.** Shortleaf pine and hardwood regeneration 12 years after the first cutting cycle harvest under the single-tree selection method in a shortleaf pine stand near Pencil Bluff, AR. Several cutting cycles will be required to develop the typical structure of an uneven-aged stand. (Photo by James M. Guldin)

Uneven-aged structure can be sustained with 10-year cutting cycle harvests that retain about 5,000 board feet of volume in 60 square feet of basal area of the best trees across all size classes. These stands have annual growth rates (in English units) of approximately 200 board feet per acre and 2 square feet of basal area, which give operable cutting cycle harvest volumes of about 2,000 board feet every 10 years. Herbicide control of competing hardwoods roughly once every 10 years is generally needed to maintain pine sapling development.

Since the early 1990s, managers on national forest lands have committed to expanding the use of uneven-aged silviculture in their pine and oak-pine stands. The single-tree and group selection (fig. 10) prescriptions that are being applied are somewhat less intensive than forest industry's single-tree selection method in that they retain some hardwood component. Most national forest sites to date are still in the early stages of the transition from mature second-growth even-aged pine and oak-pine stands to an uneven-aged structure.

As is true throughout the United States, there has been virtually no long-term experience in the Ouachita Mountains with multiple entries using the group selection method. Questions remain as to whether the group identity can be retained in the long run, and whether doing so is even important. In all likelihood, the group selection methods will gravitate more toward a single-tree selection method as multiple-age cohorts are established and stand structure becomes more balanced.

By definition, uneven-aged reproduction cutting methods in southern pine stands create discontinuous stand conditions. They provide a temporally and spatially transient distribution of logging slash and debris within the stand, resulting in a heterogeneous distribution of volatile fine fuels. This reduces the need to treat fuels, because the entire stand is unlikely to have fine fuels throughout, but it also complicates fuel treatments. Pine regeneration is being recruited in a discontinuous spatial pattern as well, and recruitment is repeated following every 10-year cutting cycle. As a result, stand-wide

**Figure 10.** Shortleaf pine and hardwood regeneration 12 years after the first cutting cycle harvest under the group selection method in a shortleaf pine stand near Mount Ida, AR. (Photo by James M. Guldin)

treatments such as prescribed burning are difficult to implement. On one hand, fuels are sufficiently heterogeneous to confound uniform fire effects and fuels treatment. On the other, the logging debris is concentrated in the openings where the desired regeneration is found, and the saplings will not survive the fire. More research is needed to better understand the degree to which uneven-aged stands can be managed with fewer age cohorts on 20-year cutting cycles—this might provide a window during the second decade when prescribed burning would not kill the youngest age cohort.

#### Hardwood regeneration

Compared to pines, few stands in the Ouachita Mountains are actively managed for hardwood species because of their lower growth and yield and the far less vigorous market for hardwood products. But for landowners interested in white oak, the dominant commercially valuable hardwood species in the area, both clearcutting and shelterwood method both can be applied successfully. As discussed previously, this requires sufficient oak regeneration potential in the stand before harvesting and appropriate followup site-preparation treatments to encourage the development of all oak regeneration sources.

The nearest successful example of successful uneven-aged silviculture in oak-hickory stands is found on the Pioneer Forest in the Missouri Ozarks (Flader 2004, Iffrig and others 2008, Loewenstein and Guldin 2004). To date, to the silvicultural approach used on the Pioneer Forest has not been translated to oak-hickory stands in the Ouachita Mountains—partly because of the low demand for hardwood products, the generally higher quality of oak sites (especially white oak-dominated lower northern slope sites) in the Ouachitas, and perhaps also the absence of efficacy testing beyond the Pioneer.

### *Unique Silvicultural Systems*

Two unique silvicultural systems merit some special consideration in the context of fuels treatments in the Ouachita Mountains. One is the work done on shortleaf pine-bluestem restoration on the Ouachita National Forest, which has a management goal of roughly 100 000 ha over time (Guldin and others 2004a, Hedrick and others 2007). The other is the intensive forest management practiced on roughly 800 000 ha by forest industry. Both practices have been successful in achieving their respective goals; and for both, the scale of application for is large enough to have considerable ecological and silvicultural effects.

## Common Fuels Treatments in the Ouachitas

Fuels treatment is in large measure a popular name for a classically established subset of silvicultural practices that contribute to a reduction in standing and down woody biomass. In that context, a number of practices that have been discussed merit specific mention.

### *Timber Harvesting Treatments*

Harvesting activity such as reproduction cutting and thinning removes large piece sizes from the stand being harvested, but adds a considerable amount of fine fuels. Although decomposition rates are rapid, as would be expected in the humid Subtropical Division, the volume of material and the hazard it presents can be a threat during the time between harvesting and decomposition. Supplemental standards in harvesting such as lopping and scattering the slash will accelerate decomposition, but this comes at a cost of extra work. The current interest in biomass utilization may result in more complete utilization of biomass during harvesting. Otherwise, supplemental site preparation treatments such as mechanical reduction of excess biomass or prescribed burning 2 or 3 years after harvest may be appropriate.

The greatest potential watershed impact from harvesting is associated with logging activities such as skid roads that can damage forest soils and log transport on permanent roads that can result in sediment delivery directly to creeks. The application of best management practices is voluntary in Arkansas and Oklahoma, but attention to the rules set forth in the voluntary guidelines for both States will help minimize adverse effects from skidding and hauling. A more indepth discussion of these effects is found in chapters 12 and 13.

## *Prescribed Fire Treatments*

Prescribed fire in the Ouachita Mountains is generally applied as either a site-preparation or intermediate treatment with a goal of cleaning and release in pine plantations and in even-aged, naturally regenerated shortleaf pine, pine-hardwood, and hardwood stands. The prescription is usually applied on Federal lands, where burns in the dormant season through the early part of the growing season typically extend from January through April. Forest industry avoids using prescribed fire in their loblolly pine plantations because of concerns about unwanted reductions in growth and yield. Private nonindustrial landowners typically do not have access to the personnel required to efficiently burn large areas. Liability issues also limit a broader application of prescribed burning on private lands.

The choice of an ignition source depends on the condition of the landscape being burned, whether there are young stands within the landscape that need special attention to withstand burning, and the proximity to private land. Drip torch ignition early in the burning season is common for burn units near or interspersed with private land, so as to better control fire intensity and the area covered by the fire. Young stands are often burned very early in the growing season, again with drip torches, so as to consume fine flashy fuels that might create too hot a fire if burned later. Aerial ignition is preferred in large well burned landscapes where sensitive stands have been preburned because of the cost and labor efficiencies that result from burning large areas.

The watershed effects of prescribed fire are usually minimal. Vegetation recovers quickly in the Ouachita Mountains, and the risk of direct erosion through overland flow is minimal. Smaller fires ignited directly with drip torches are often imposed at a stand level; in these circumstances, permanent and intermittent stream channels usually form one of the boundaries of the burn unit. The intensity of larger fires ignited by aerial ignition can be adjusted by spacing the incendiary spheres that are dropped from the helicopter, and streamside zones are likely to burn with lower intensity if not directly hit by incendiary spheres or if soil conditions are wet, as they usually are in the spring. The greatest likelihood of unwanted watershed effects is if fire lines directly cross perennial or intermittent streams—a situation that can be avoided as conditions warrant.

## *Mechanical Treatments*

Mechanical treatments associated with site preparation and intermediate prescriptions are widely applied in lieu of prescribed fire on all ownerships in the Ouachita Mountains regardless of species composition. The silvicultural objective generally depends on the ownership, the origin of regeneration (whether natural or planted), the silvicultural system being applied, and the resources of the landowner.

The goals of site preparation treatments are to reduce logging slash and competing vegetation and to prepare the seedbed. Usually, the intensity of treatments prescribed depends on whether natural regeneration or planting is to ensue, with more intensive site preparation activities usual for plantation establishment. In even-aged reproduction cutting followed by plantation establishment, all of the commercial timber is removed by harvesting and the noncommercial residual biomass is removed by mechanical felling (shearing, chopping, or chainsaw felling), which is sometimes concentrated by piling followed by either broadcast burning or burning of piles to eliminate slash. Ripping usually occurs again in late summer, with planting feasible in the following spring.

Bedding is not typically used in the Ouachita Mountains because of the extreme rockiness of the soils.

Soil displacement as a result of site preparation is a concern for cumulative watershed effects of silvicultural activity. Prescriptions that require logging debris to be raked, pushed, or dragged into rows or piles cannot be accomplished without some degree of soil movement; the less of this activity that is prescribed, the less the soil movement. Ripping deliberately moves soil so that rainfall can wash particles from the sides of the rip into the furrow, thereby creating an ideal planting medium and increasing the survival of seedlings. Cumulative watershed effects can be minimized by ripping along the contour, creating periodic discontinuities of the rip so that flow is interrupted, and stopping the rip before it reaches sensitive watershed areas such as streamside zones. However, in essence, ripping along the contour at 10-foot spacing creates a hillside of small fire lines, which impede site-preparation burning and other prescribed fires. The most effective approach for site preparation burns on ripped sites is to use drip torches and drop fire in the spaces that separate the rips. Once the rips are grassed over, though, the prescribed fire is usually able to carry across the rips without restrictions.

Intermediate mechanical treatments include activities, such as chipping or mulching, that reduce fuels. This activity is expensive, however, and thus is typically reserved for situations in which an uncontrolled fire might escape across property boundaries. Several recent disturbance events—windstorm and ice storm—in the Ouachitas over the past decade have resulted in down woody debris across landlines; these are a high priority for reduction by chipping and mulching, which produce a rather thick layer of chips and residues that usually remain in place within the stand and that do not burn easily. Cumulative watershed effects would be minimized by the simple expedient of not operating the mulchers or chippers within streamside zones.

Of course, the problem with mechanical treatments is that only the tops of trees are removed; rootstocks remain. Hardwood rootstocks without hardwood tops quickly become hardwood sprouts, and sprouting hardwoods have unwanted ecological influences on developing pine seedlings and saplings. This is not because of any inherent superiority or inferiority of hardwoods over pines ecologically, but rather because a sprout that is supported by a large preexisting root system can grow faster than a seedling supported by its own small developing root system. This imbalance threatens the seedling with suppression.

## *Herbicide Treatments*

A more permanent approach to sprout control—either through cleaning, weeding, or release treatments—is an herbicide applied in a manner that kills both the tops and the roots of the sprouts. Aerial application of herbicides is effective when the goal is to control hardwoods competing with pines; a number of chemicals and application methods are available that target hardwoods with a minimal effect on pines. For example, in late summer, hardwoods are still photosynthetically active but pines are dormant. This difference in characteristics suggests a tactic of late-summer herbicide application over large areas, using helicopters, that will affect the actively-growing hardwoods but will have little or no effect on the dormant pines. Individual-stem treatment methods are more labor intensive, but have several advantages in specificity of target application and minimization of effects on nontarget vegetation. They can be applied to cut stumps or to the foliage of the targeted tree using a backpack sprayer. Although these methods are labor intensive, they minimize the volume of herbicide applied across a stand, and they are specific to a target tree rather than a target species, meaning that they can be used in pine-hardwood or hardwood-pine stands, or in hardwood stands to release desired hardwoods from competing hardwoods. These differences in application often reflect ownership differences as well; the broadcast methods are more common on private lands, and the individual-stem applications are more common on public lands.

The cumulative effects of herbicide applications have become considerably lower over recent decades. Modern herbicides are developed to act specifically on plant metabolism—by inhibiting photosynthesis or synthesis of amino acids that are only found in plants. As a result, they have reduced adverse effects on other organisms than herbicides that were used in the past, and also have a short half life in the environment. Watershed effects are generally limited to the movement of the soil solution that contains the herbicide before being degraded in the environment, and also by the general chemistry of the inactive ingredients in carriers, surfactants, and other herbicide formulations. Following common safety precautions—such as applying setbacks from sensitive areas, avoiding direct application to streams, and employing environmentally safe loading and cleanup procedures—will also limit cumulative watershed effects.

Cumulative watershed effects are a function of the proportion of forest land in ownerships with varying of ability to engage in rigorous monitoring. In many respects, the Ouachita Mountains have an advantage because two-thirds of the forest land is managed by either forest industry or Federal agencies, more than elsewhere in the Eastern United States. These landowners and managers have a highly capable infrastructure in place to control wildfire, efficiently process unwanted forest residues (usually as part of commercial timber sale or harvesting activity), and otherwise integrate specific attention to fuels, treatment of fuels, and minimization of cumulative effects as part of their larger forest management program.

Forest lands owned by forest industry in the Ouachitas are primarily under an intensive program of even-aged forest management that emphasizes clearcutting and planting for commercial timber and fiber production. On these lands, management activities are carried out with keen attention to prompt reforestation, effective site preparation and release, timely thinning and pruning, and efficient reproduction cutting. Industry foresters have taken steps to execute this intensive program of silviculture with a minimum of adverse cumulative watershed effects, and in doing so have agreed to be bound by independently-verified standards.

Similarly, public forest management is dominated by the Ouachita National Forest. Again, these lands are managed under a comprehensive land and resource management plan that incorporates a diversity of both even-aged and uneven-aged silvicultural systems and includes comprehensive standards for ensuring that forest operations conducted under the timber, water, recreation, lands, and engineering programs are carried out in compliance with best management practices, all of which are detailed in public records.

The nonindustrial private forestry sector is more variable in this regard. Owners of these forest lands are less likely to be under a management plan, less likely to understand the hazard of fuels buildup, somewhat less likely to have sufficient resources to respond to wildfire risk (which is a responsibility of State agencies in Arkansas and Oklahoma), and less likely to be proactive in integrating fuels treatments into an overall program of silvicultural activities specified by management plans for their forested property. Finally, cumulative watershed effects on nonindustrial private lands are addressed by State-issued best management practices that are voluntary. The greater the degree to which owners can develop management plans, seek advise from professional foresters when making harvesting decisions, or get involved in public or private management assistance programs, the better will be scientific basis of the silviculture that is applied on their lands, and the fewer will be the cumulative watershed effects from improper attention to fuels and fuels treatments.

# Conclusions

Fuels are a subset of the living and dead vegetation found within every stand in the forest. They are important insofar as their size, biomass, and distribution contribute to the risk of loss to forest resources in the event of an uncontrolled wildfire. Similarly, tools such as prescribed fire, fire surrogate treatments, and fuels treatments are a subset of a broader array of general silvicultural practices that are typically applied within forest stands and landscapes as a part of general forest management activities. These tools for fuels are most effectively implemented if they fall within the context of the larger silvicultural systems being imposed within stands and landscapes, rather than as standalone treatments applied at a given point in time. According to this perspective, the cumulative watershed effects of fire, fire surrogate, and fuels treatments are best characterized as similar to those that result from all forest management activities. Unlike other areas in the South, two-thirds of the Ouachita Mountains forest landscape is under management, either by Federal agencies or forest industry. Active management under the guidance of professional foresters is the most effective way to integrate fuels treatments, and to minimize their cumulative watershed effects, as elements of a larger program of active forest management.

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CHAPTER 9.

# Ecology and Management of the Prairie Division

Roger C. Anderson

## General Features of Grasslands

### *Distribution and Climate*

Grasslands occurred on all continents, comprised almost 42 percent of the world's plant cover, and once covered approximately 46 million km$^2$ of the Earth's surface. Grasslands contain few trees or shrubs, are dominated by grasses (members of the family Poaceae), and have a mixture of nongraminoid herbaceous species called forbs. Plant families most abundant as forbs are sunflower (*Asteraceae*) and pea (*Fabaceae*) families. No single climate characterizes grasslands and they occur in areas of the Earth that receive as little as 200 mm of precipitation annually to areas that receive 1300 mm, and in areas where average annual temperatures vary from 0 to 30 °C (Oesterheld and others 1999, Risser and others 1981, Sauer 1950). Grasslands are not necessarily treeless and they are transitional to savannas, which are characterized by higher densities of drought-tolerant, fire-resistant trees. The ratio of trees to grass increases as precipitation increases (Anderson and Bowles 1999, Curtis 1971, Oesterheld and others 1999); in landscapes receiving >650 mm of precipitation, the trend is for increasing cover of woody species with "long-term fire exclusion" (Sankaran and others 2004). In areas of low precipitation, grasslands grade into desert communities. Common features found among grasslands include: periodic droughts, frequent fires, landscapes that are level to gently rolling, and an abundance of grazing animals (Anderson 1982, 1990; Risser and others 1981, Sauer 1950).

### *Drought, Fire, and Grazing Animals*

Grassland plants evolved under the influence of periodic droughts, frequent burning, and grazing animals; they are adapted to all three (Anderson 1990, Damhoureyeh and Hartnett 1997, Gleason 1922). This adaptation for grasses is manifested in their ability to die down to underground organs (rhizomes, root, and tillers); and only expose dead tops (Gleason 1922). Grasses grow tips beneath soil that are not exposed to desiccation—this allows them to escape drought. The leading edge of prairie fires can be long and historically they often extended many km in length; however, the flame depth is narrow in the range of 2 to 7 m. These fires move relatively rapidly and, because the soil is a good insulator, heat penetration into the soil can be measured in millimeters (Anderson 1982). Consequently, the growing points of prairie plants are protected from the heat of the fire. Grazers can only remove aboveground tissues; once grazing pressure is removed, new shoots can emerge from belowground (Tainton and Mentis 1984).

The adaptation of grasses to fire, drought, and grazing animals may represent a preadaptation to one or more of these factors; however, grasses and herbivores likely coevolved based on other features of grasses. The post-Miocene expansion of grasslands and savannas worldwide was associated with the adaptive radiation of large grazing mammals (Anderson 1982, 1990; Axelrod 1985, McNaughton 1993, Oesterheld and others 1999, Stebbins 1981). Adaptive responses of grasses to herbivores that reflect a coevolutionary relationship include the presence of silica in epidermal cells, perennating organs below ground level, and aboveground productivity in excess of the amount of biomass which can decompose in a single year (Anderson 1982, 1990; Stebbins 1981).

The widespread expansion of grassland is associated with the appearance of the $C_4$ photosynthetic pathway (which initially produces a 4-carbon intermediary during fixation of carbon dioxide). The $C_4$ photosynthetic pathway provides an advantage over the more common $C_3$ pathway because it provides higher quantum yields for carbon dioxide uptake under high temperatures. The $C_4$ photosynthesis is also favored over $C_3$ photosynthesis when the concentration of atmospheric carbon dioxide is below 500 ppmV or parts per million by volume (Cerling and others 1997; Ehleringer and others 1997, 2002). During the Mesozoic, carbon dioxide concentrations were thought to be >1,000 ppmV. However, in the early Miocene or late Oligocene (Kellogg 1999), perhaps 20 to 25 million years ago, decline in atmospheric carbon dioxide favored evolution of $C_4$ plants in moist tropical and subtropical climates (Ehleringer and others 1997). This photosynthetic pathway is found in <2 percent of all flowering plants but approximately half of the 10,000 species of graminoid (grasses and sedges) use this pathway. Although $C_4$ plants are a small percentage of flowering plants, they contribute 25 percent of total global productivity, largely as monocotyledons in grasslands (Eherlinger and others 2002).

$C_4$ grasslands developed quickly worldwide during the Miocene-Pliocene transition (6 to 8 million years ago) with expansion of the Antarctic Ice Sheet, which some believe increased aridity, and reduced atmospheric carbon dioxide to <500 ppmV, causing declines in forest and woodlands accompanied by an explosive evolution of grasses and forbs (Cerling and others 1997; Ehleringer and others 1997, 2002). However, Keeley and Rundel (2005) posit that the conversion of forests to $C_4$ grasslands 4 to 7 million years ago was not directly attributable to a decline in atmospheric carbon dioxide or increased aridity but rather to new climatic conditions that encouraged fire: a warm moist growing season that increased biomass productivity and a pronounced dry season that created combustible fuels. This monsoon climate likely would be accompanied by frequent lightning strikes at the end of the dry season. In this model, fire would have been a primary driver in the conversion of forest to grasslands and the maintenance of grasslands as it is today.

Expansion of open grassland and savanna habitats was associated with increased fossilized silica bodies in the epidermis of grasses, which provide protection against grazing. Concomitantly, mammals with high-crowned teeth (hypsodonty) adapted to grazing (Axelrod 1985, Stebbins 1981) increased, and more cursorial (running) and saltational (jumping) body forms began to evolve.

# Central Grassland of North America

Grasslands of North America are part of a diverse assemblage of vegetation types that occur under a wide range of climatic conditions and once covered about 15 percent of the continent. These grasslands are referred to as *prairies*, a word meaning meadow or field, which early French explorers used to describe the extensive grasslands they encountered (Curtis 1971, Risser and others 1981). Along a north-south gradient, grasslands extended from the deserts of Northern and Central Mexico to mixed-grass prairies of Alberta, Saskatchewan, and Manitoba in Canada (Risser and others 1981). Across this gradient, average annual temperature varies from 2.8 °C in the northern mixed-grass prairie around Regina, SK to 22.6 °C in Monterrey at the edge of Chihuahuan Desert. In the desert grasslands of southwestern United States, annual precipitation averages 250–450 mm, whereas along the eastern edge of grasslands precipitation varies from about 2500 mm in southeast Texas to 750 to 1000 mm in Indiana (Risser and others 1981).

## Geographic variation

The shape of the Central Grassland of North America resembled a large triangle whose base extended from the Canadian provinces of Alberta and Saskatchewan southward along the eastern foothills of the Rocky Mountains and then to southeastern Texas. The point of the triangle extended well into the Midwest in southwestern Wisconsin, Illinois, and western Indiana, with scattered outliers in Michigan, Ohio, and Kentucky (the Prairie Peninsula). This area includes the grasslands of the 12 Great Plains States, as well as the outliers east of the Mississippi River. Annual precipitation from the western Great Plains to the Prairie Peninsula increases from 260 to 1200 mm; across a north-south gradient, average annual temperature ranges from 3 to 22 °C (Sala and others 1988).

Ecologists have traditionally divided the grassland into three sectors based on annual precipitation: a western shortgrass prairie (260 to 375 mm precipitation), the eastern tallgrass prairie or "True Prairie," (625 to 1200 mm precipitation), and between the two the midgrass or mixed-grass prairie (375 to 625 mm precipitation) (fig. 1). Shortgrass prairie occupies an area dominated by grasses that are 0.3 to 0.5 m tall and include buffalo grass (*Buchloe dactyloides*), blue grama (*Bouteloua gracilis*), sideoats grama (*Bouteloua curtipendula*) and hairy grama (*Bouteloua hirsuta*). The tallgrass prairie is dominated by species that reach heights of 1.8 to 2.4 m and include big bluestem (*Andropogon gerardii*), Indian grass (*Sorghastrum nutans*), switchgrass (*Panicum virgatum*), and little bluestem (*Schizachyrium scoparium*). The mixed-grass prairie is dominated by species that are 0.8 to 1.2 m tall and include little bluestem, western wheatgrass (*Pascopyrum smithii*), and green needlegrass (*Nassella viridula*). In the

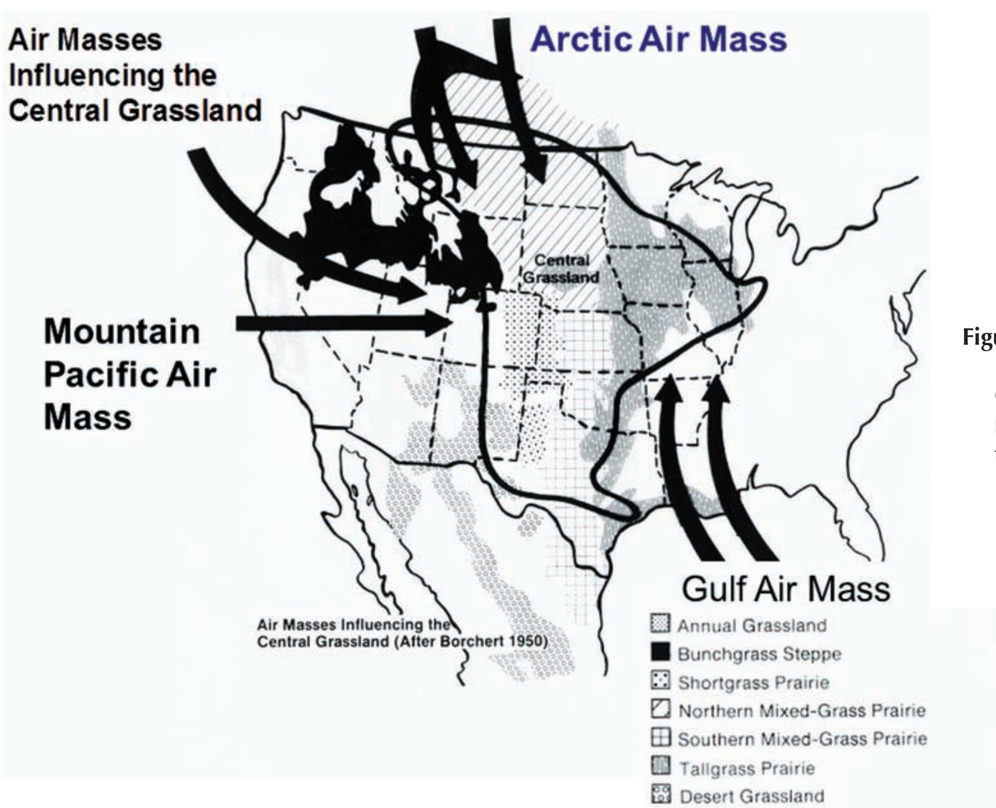


**Figure 1.** The Grassland of North America. The Central Grassland is outlined and the major area masses influencing the grassland are shown.

mixed-grass prairie, tallgrass prairie species occur in depressed areas that are moister than upland sites, and on drier sites short-grass prairie species can occur resulting in a mixture of tall-, short-, and mixed-grass prairie species. Across the Central Grassland, species composition and abundance varies continuously and with no sharp divisions among these arbitrarily designated grasslands.

## Variation within a geographical area

Within each of the major areas of the Central Grassland differences among prairie types are the result of soil, aspect, slope position and other factors. At any location in the tallgrass prairie, the diversity of prairie types depends on soils and topographic features. A primary factor causing this varied vegetation patterns is availability of soil moisture (Corbett and Anderson 2006, Curtis 1971, Nelson and Anderson 1983, Umbanhowar 1992). For example, approximately 930 ha of high quality remnant prairie in Illinois consist of diverse habitat types with varying topography and substrate. They include dry hill and bluff prairies with loess or glacial drift derived soils that often occupy western or southwestern facing slopes overlooking rivers (Evers 1955). Dry prairies also occur on deep sand deposits or on dolomitic or gravel substrates with shallow stony soils. Additionally, wet-mesic to wet prairies occur on loess-derived, till-derived, or dolomite-containing substrates (Table 1). Historically, the most common Illinois prairie types were mesic and wet prairies covering as much as 55 percent of the State (Fehrenbacher and others 1967); most have been converted to agricultural or urban uses.

## Climate

The climate of the Central Grassland is influenced by three primary air mass systems: Arctic, Gulf, and Mountain Pacific (Borchert 1950, Bryson and Hare 1974, Risser and others 1981). The Arctic air mass influence is reflected in part by the increased snow cover and decreasing temperatures from south to north (Risser and others 1981), as well as variations in vegetation (Diamond and Smeins 1988) and phenological patterns (Kebart and Anderson 1987, McMillan 1959). Gulf and Mountain Pacific air masses are most important in determining east-west variation. The Gulf air mass originates in the Gulf of Mexico. As it moves northward into the eastern part of the Central Grassland, it brings humid air and precipitation as it encounters cooler air or provides moisture for convectional storms. The Mountain Pacific air mass arrives on the west coast as a humid air mass. However, as it progresses eastward the air mass passes over several western mountain ranges (Coastal, Sierra, and Rocky). As the air mass rises, it cools adiabatically, and gives up much of its moisture and then is compressed by an increasing volume of atmosphere as it descends to lower elevations on the east side of the Rocky Mountains, causing it to become warmer and more arid as it spills out into the Great Plains. Thus, the Central Grassland occurs in the rain shadow of the western mountains.

From west to east, the influence of the Pacific air mass decreases and the influence of the Gulf air mass increases. Associated with these changes average annual precipitation increases, and periodic droughts, and periods of low humidity in summer decrease (Borchert 1950, Bryson and Hare 1974, Risser and others 1981). West-east climatic variation produces the changes in vegetation from the foothills of the Rocky Mountains to the Midwestern United States that results in the shortgrass, mixed-grass, and tallgrass prairies. Annual net primary productivity is also affected by this climatic gradient, which in years of average precipitation varies from 150 to 600 g/m$^2$.

## Grass Adaptations to Drought

There are many morphological and physiological features that allow grasses to tolerate high moisture stress including: (1) the occurrence of bulliform cells in leaves, which cause them to enroll when they lose water, thereby reducing surface area for transpiration; and (2) utilization of the $C_4$ photosynthetic pathway, which adapts plants to high temperatures, high levels of solar radiation, and periods of moisture stress. The $C_4$

**Table 1.** Leading species (with average quadrat frequency of ≥2.0 percent, values are percent (± standard error) in six community types (modified from Corbett and Anderson (2001, 2006). The second column provides the native community of maximum presence (occurrence) in Wisconsin from Curtis (1971)

| Species[a] | Community: Curtis 1971 | Dry sand | Gravel/ sand | Hill prairie | Gravel/dry dolomite | Mesic/ dry mesic | Wet/wet dolomite |
|---|---|---|---|---|---|---|---|
| | | *------percent (±standard error)------* | | | | | |
| Little bluestem | Dry prairie | 16.9 (±1.5) | 17.3 (±2.9) | 15.4 (±0.7) | 11.6 (±1.4) | 3.6 (±0.4) | — |
| Devil's-tongue | Cedar glade | 8.4 (±1.8) | — | — | — | — | — |
| Cuman ragweed | Sand barren | 6.7 (±2.2) | — | — | — | — | — |
| Prairie sandreed | Dune | 3.9 (±2.0) | — | — | — | — | — |
| Heller's rosette grass | Dry mesic prairie | 3.3 (±1.7) | 2.4 (±1.6) | — | — | — | — |
| Goat's rue | Oak barren | 3.3 (±1.7) | — | — | — | — | — |
| Hairy grama | Cedar glade | 2.3 (±1.5) | — | — | — | — | — |
| Porcupine grass | Dry mesic prairie | 2.0 (±1.0) | 4.4 (±2.4) | — | 4.5 (±1.0) | — | — |
| Flowering spurge | Oak barren | — | 5.4 (±2.0) | — | — | — | 3.6 (±0.4) |
| Pale purple coneflower | Mesic prairie | — | 3.1 (±2.0) | — | 3.4 (±0.6) | — | — |
| Carolina puccoon | Sand barren | — | 2.7 (±2.2) | — | — | — | — |
| Prairie June grass | Sand barren | — | 2.5 (±2.0) | — | — | — | — |
| Gray clustered poppy mallow | Dry mesic prairie | — | 2.3 (±1.6) | — | — | — | — |
| Sideoats grama | Dry prairie | — | | 9.1 (±0.8) | 3.8 (±0.8) | — | — |
| Indian grass | Dry mesic prairie | — | | 4.5 (±0.7) | — | 4.4 (±0.4) | — |
| Purple prairie clover | Dry prairie | — | 2.2 (±2.2) | 4.5 (±0.4) | — | — | — |
| Flowering spurge | Oak barren | — | — | 4.1 (±0.4) | 4.6 (±0.7) | — | — |
| Gray goldenrod | Dry prairie | — | — | 3.6 (±0.5) | — | — | — |
| Scurfy pea | Not given | — | — | 2.8 (±0.4) | — | — | — |
| Sky blue aster | Dry mesic prairie | — | — | 2.5 (±0.5) | — | — | — |
| Pursh leadplant | Dry prairie | — | — | 2.3 (±0.4) | 3.6 (±0.7) | — | — |
| Gray prairie dropseed | Dry prairie | — | — | — | 3.0 (±0.6) | 3.0 (±0.4) | — |
| Carolina rose | Not given | — | — | — | 2.5 (±0.6) | 2.4 (±0.3) | — |
| Snow flurry | Dry mesic prairie | — | — | — | 2.3 (±0.6) | 3.6 (±0.4) | — |
| Big bluestem | Mesic prairie | — | — | — | — | 5.0 (±0.5) | 2.5 (±0.7) |
| Virginia strawberry | Northern dry forest | — | — | — | — | 2.2 (±0.3) | 2.7 (±0.8) |
| Sedge | Not given | — | — | — | — | — | 6.3 (±1.8) |
| Giant goldenrod | Wet prairie | — | — | — | — | — | 4.3 (±0.9) |
| Virginia mountain mint | Wet mesic prairie | — | — | — | — | — | 3.7 (±0.8) |
| Bluejoint | Fen | — | — | — | — | — | 3.5 (±1.5) |
| Prairie cordgrass | Wet prairie | — | — | — | — | — | 3.2 (±0.8) |
| Upright sedge | Southern sedge meadow | — | — | — | — | — | 2.2 (±2.1) |
| Sawtooth sunflower | Wet mesic prairie | — | — | — | — | — | 2.7 (±1.1) |
| Riddell's goldenrod | Fen | — | — | — | — | — | 2.7 (±0.9) |

[a] Common names follow the USDA Plant database.
Source: Curtis (1971) modified from Corbett and Anderson (2001, 2006).

plants have high water use efficiency, photosynthetic rates, and stomatal sensitivity to water loss, and they can grow under conditions of low soil-water potential (Ares 1976, Briske and Wilson 1978). Although many dominant grasses in the Central Grassland are $C_4$ grasses—including Indian grass, big bluestem, switchgrass, little bluestem, sideoats and hairy grama grass—many species of $C_3$ grasses dominate some prairies. The $C_3$ plants maximize growth under cool moist conditions and are known as "cool season grasses." They have lower water use efficiency, photosynthetic rates, and photosynthetic temperature optima and saturation levels for solar radiation, but higher rates of photorespiration and higher carbon dioxide compensation points.

In North America, the primary separation of $C_4$ and $C_3$ grass is related to temperature (Terri and Stowe 1976). Where daytime growing season temperature are lower than 22 °C, $C_3$ plants should dominate and where growing season temperatures are above 30 °C and soil moisture is adequate, $C_4$ plants should predominate (Ehleringer and others 1997). $C_4$ plants have a higher quantum yield for carbon dioxide fixation at latitudes less than about 45° (Ehleringer and others 1978). Where the two groups of grasses grow together, the $C_3$ grasses—such as Canada wildrye, (*Elymus canadensis*), western wheatgrass, green needlegrass, porcupine grass (*Hesperostipa spartea*), and prairie Junegrass (*Koeleria macrantha*)—grow in the spring and early summer, whereas the $C_4$ grasses begin growth later and maximize growth in midsummer. Even though the $C_4$ grasses are more drought tolerant than $C_3$ grasses, western wheatgrass increased its abundance more than many $C_4$ grasses during the drought of the 1930s because it was able to use moisture that was only available in early spring—and not available to the later growing $C_4$ grasses (Monson and others 1982, Weaver 1968).

## Pleistocene history

Although grasslands may have been present on the North American continent for 20 million years (Axelrod 1985, Benedict and others 1996, Risser and others 1981, Weaver 1968), the Central Grassland is of relatively recent origin. During the Pleistocene, climate change and the continental ice sheet caused destruction of the mid-continent grassland and other vegetation types that may have replacement it. At the peak of Wisconsinan glaciation (18,000 years ago), most of what now is the Central Grassland was dominated by spruce (*Picea* spp.) and jack pine (*Pinus banksiana)* forest or covered with glacial ice. During the early Holocene, 10,000 years ago, grasslands or oak (*Quercus* spp.) savanna occurred in much of the area, but forests occurred over most of the Prairie Peninsula (Delcourt and Delcourt 1981, Nelson and others 2006). In the eastern portion of the Prairie Peninsula, aridity increased between about 10,000 and 8,500 years ago and fire sensitive trees, such as elms (*Ulmus* spp.), ashes (*Fraxinus* spp.) ironwood (*Ostrya* sp.), and sugar maple (*Acer saccharum*), decreased in abundance and prairie species expanded. Between about 8,500 and 6,200 years ago aridity declined and prairie coexisted with fire-sensitive and fire tolerant tree taxa [e.g., oaks (*Quercus* spp.) and hickories (*Carya* spp.)]. After about 6200 years ago, prairie became dominant; however, the climate was less arid than it was around 8,500 years ago. Fires set by Native Americans and lightning are thought to the primary reasons tallgrass prairies persisted as the climate became less arid.

Information from several sources, however, suggests that changes in climatic patterns and vegetation in the area of the Prairie Peninsula varied spatially during the Holocene on millennial time scales (Baker et al. 1996, Nelson and others 2006, Nelson and Hu 2008, Winkler et al. 1986). For example, based on pollen records and charcoal deposits from Lake Mendota in southern Wisconsin, the climate was hotter and drier and fire frequency greater between 6,500 and 3,500 years ago than it was earlier in the Holocene and after 3,500 years ago the climate became cooler and more moist (Winkler 1995, 1997; Winkler and others 1986). In northeastern Iowa, forest dominated from about 8,000 to 5,100 years ago and then prairie replaced forest (Baker et al. 1996). The authors suggest that a climatic shift that increased the flow of arid Pacific air and increased frequency of fire probably caused the change in vegetation.

According to Axelrod (1985), the recent origin of the Central Grassland is indicated by the occurrence of most of its species in forest and woodlands; the presence of few endemic plants (Wells 1970a), insects (Ross 1970), or birds (Mengel 1970, Risser and others 1981); the relict occurrence of a variety of tree species throughout the area; and the current invasion of woody plants. Benedict and others (1996) indicate that, among mammals, true grassland species comprise only 11.6 percent of those occurring on the central and northern plains; and that only 5.3 percent of North American bird species evolved on prairies (Knopf 1996). Similarly, many of the grass species that occur in the Central Grassland evolved in eastern forest openings, the southwestern deserts, or mountain meadows (Gleason 1922, Risser and others 1981).

## *The Prairie Peninsula*

### Location and origin

Near the Mississippi River and eastward in the Central Grassland, the climate becomes increasingly favorable for the growth of trees. The wedge-like extension of the grassland into the Midwestern United States is called the Prairie Peninsula (fig. 2), because it is a peninsula of grass extending into forests (Transeau 1935). Annually, this area receives 750 to 1200 mm of precipitation, a climate capable of supporting forest. Historically, ecologists have debated why this area had grasslands rather than forest (Curtis 1971, Transeau 1935). Several general hypotheses emerged to explain this pattern. One hypothesis focused on the importance of climate as a primary determinate of vegetation patterns. The other posited that fires set by Native Americans or soil conditions were responsible for the absence of trees.

### Climate effects

Transeau (1935) reasoned that climatic extremes are more important than averages in determining the distribution of organisms. He demonstrated that the Prairie Peninsula has periodic droughts when drier western climatic conditions shift eastward into the

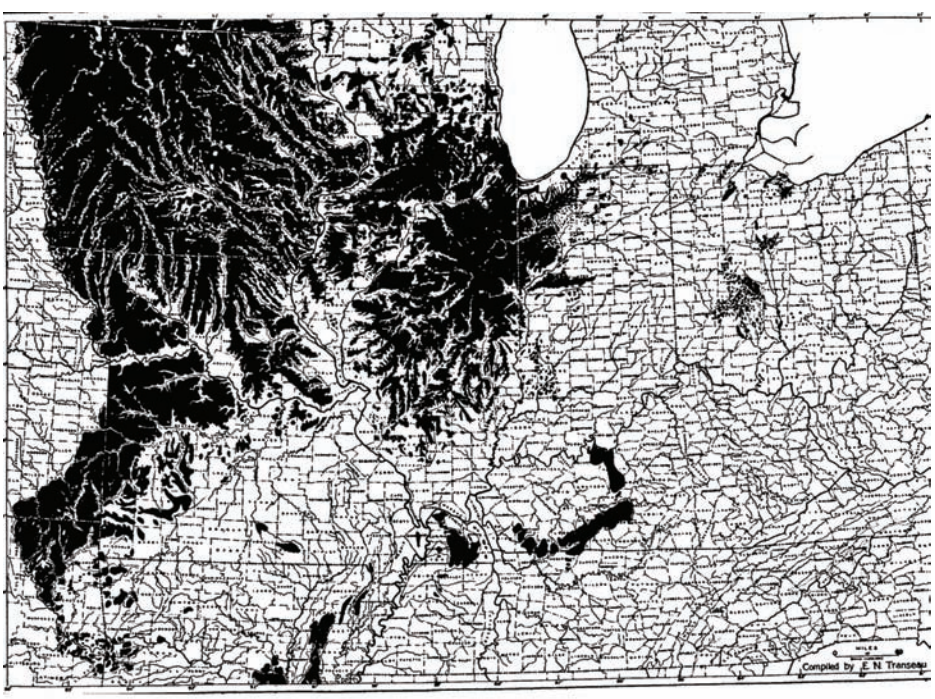


**Figure 2.** The Prairie Peninsula after Transeau (1935).

Prairie Peninsula. These periodic droughts would favor the prairie and set back the forest. Indeed, during the droughts of the 1930s trees experienced high rates of mortality in the Prairie Peninsula (Albertson and Weaver 1945). Transeau (1935) noted a loss of trees from upland sites during the droughts of the 1930s accompanied by a retreat to sheltered locations adjacent to streams. Seedlings are strongly affected by drought and by competition from grasses, both directly and indirectly—they produce flammable, finely divided fuels that encourage the spread of fire (Anderson 1990, Sankaran and others 2004).

## Drought and rooting depth

The differential effects of drought on prairie grasses and trees have been explained by the root growth forms of these two groups. In Missouri tallgrass prairie, 80 percent of the root mass occurred in the upper 25 cm of soil (Dahlman and Kucera 1965), with similar results reported by others (Bartos and Jameson 1974, Old 1969, Risser and others 1981, and Zink and Weaver 1946). Although prairie plants have most of their roots in the upper layers of the soil, many also have deep root penetration; to illustrate, for 14 grasses and 15 forbs, the rooting depth range is 0.5 to 7.0 m; average ± SE is 2.36±0.24 m, according to original data from Weaver (1954), recalculated by Risser and others (1981). Scholes and Archer (1997) suggest that in habitats with grasses, maximum rooting depth of trees generally exceeds that of grasses. They also note that both trees and grasses have the maximum amount of their root mass in the upper soil layers. Nevertheless, grasses may be less dependent on deep soil moisture than trees (Schimper 1903, Walter 1971).

Britton and Messenger (1969) suggest that when droughts disrupt recharge of deep soil moisture, trees are more affected than grasses. Grasses use their diffuse root system to take advantage of light showers that recharge upper soil surface layers. In the Prairie Peninsula recharge of deep soil moisture usually occurs during the dormant season, because high rates of evapotranspiration during the growing season reduce the likelihood of deep soil moisture recharge. In the Midwest, areas that did not experience deep soil moisture recharge during the 1933–1934 winter corresponded to the location of the Prairie Peninsula (Britton and Messenger 1969). This finding supports the hypothesis that drought is an important factor in determining the occurrence of the Prairie Peninsula.

## Fire as a factor

As previously noted, beginning about 6,000 to 3,000 years ago the Prairie Peninsula climate became cooler and moister and more favorable for trees. Following this change of climate to one that can support prairie, savanna, or forest, stabilization of the vegetation in the Prairie Peninsula is thought to be the result of fires set by Native Americans and occasional lightning strikes (Anderson 1990, 1991b, 1998; Curtis 1971).

Prairie fires can reach temperatures of 83 to 680 °C on the surface of the soil (Rice and Parenti 1978, Wright 1973). Gibson and others (1990) reported that on Konza Prairie in Kansas, fire temperatures ranged from 166 to 343 °C as a function of habitat, whether the fire occurred on an upland or lowland site, and time since last fire—all of which affected fuel loadings. As previously noted, prairie grasses are protected from fire because their growing points are located beneath the surface of soil and the penetration of heat below the soil surface is minimal (Anderson 1982, Reichert and Reeder 1972). Fire is detrimental to trees because their aboveground growing points, shoot apical meristems and vascular cambium, are exposed and vulnerable to fire. Woody species can be killed by fire or their shoots destroyed, and even if they resprout, growth is suppressed for several years, reducing their competitiveness against grasses. Anderson and Brown (1986) reported that after a single central Illinois fire in a forest adjacent to sand prairie, 34.1 percent of blackjack oak (*Q. marilandica*) and black hickory (*Carya texana*) trees >9.0 cm d.b.h. (diameter at breast height) suffered mortality in the first year following the fire, 4.9 percent in the second year, and 8.5 percent in the third years.

Frequent fire and periodic droughts may have interacted to effectively control woody plant invasion into grasslands, especially on sites supporting fire-sensitive-mesic species—such as sugar maple (*Acer saccharum*), ashes (*Fraxinus* spp.), elms (*Ulmus* spp.), and American basswood (*Tilia americana*)—that are more susceptible to fire and droughts than the oaks. Even if the trees resprout, browsing by elk (*Cervus canadensis*) and deer (*Odocoileus* spp.) may have kept woody species from dominating grasslands (Anderson 1982). Sankaran and others (2004) suggest that grazers reduce fuels and therefore favor trees, but conversely browsers favor grass.

## *The Vegetation Mosaic*

### Topography and fire spread

In the eastern portion of the Central Grassland, the occurrence of the three community types (prairie, savanna, and forest) in the vegetation mosaic was the result of climate and of fire frequency, which was strongly influenced by topographic features and distribution of waterways (Anderson 1983; Gleason 1913, 1922; Grimm 1984, Wells 1970b). In North American grasslands, sharp transitions to distinctly different vegetation types are associated with topographic changes (Wells 1970a, 1970b). The main effect of topography appears to be its control of fire frequency. Landscapes that are nearly level to slightly rolling can support the nearly annual fires that grasslands need for maintenance (Anderson 1982, 1990; Curtis 1971, Risser and others 1981). Fire might not be able to eliminate fire resistant woody species from grasslands, but it can keep them in a reduced state and dependent for survival on recurring annual growth from roots (Bragg and Hulbert 1976, Curtis 1971, Heisler and others 2003). In dissected landscapes, fire moves rapidly up slopes, as it is carried upward by rising convection currents. Conversely, as fire moves downslope, its movement is impeded by the upward flow of the convection currents; and steep slopes and ravines can function as firebreaks and provide sheltered locations were forests can survive (Anderson 1998). Using a map of the historical distribution of "timber" (forest/savanna) and prairie in Illinois (Anderson 1970) and distribution of average slope range in the State, Anderson (1991a) examined the relationship between topography and historical distribution of vegetation; about 60 percent of the State was tallgrass prairie (Anderson 1991a, 1991b; Robertson and others 1997). Most of the prairie (82.3 percent) occurred on landscapes with average slope ranging from of 2 to 4 percent, whereas only 23 percent of the forest and savanna was associated with landscapes in this slope range, mostly in flood plains. Seventy-seven percent of forest vegetation occurred on landscapes with average slope of >4 percent (4 to 7 percent slope = 35.2 percent; and >7 percent = 41.8 percent). Most of the forested areas were associated with glacial moraines, highly dissected portions of the older Illinoian glacial till plain, nonglaciated landscapes, and waterways.

### Waterways and vegetation distribution

The distribution of waterways has a pronounced effect on the occurrence of prairie vegetation. Fires are generally swept from west to east so that areas to the west of waterways support prairie, but areas to the east support forest (Gleason 1913). Clear skies and dry weather conditions favorable for fires are associated with high-pressure systems, which have a clockwise flow of air and move from west to east. As the high-pressure system moves into an area, the leading edge of the front has wind in a westerly direction. Fire at this time would be carried to the west side of a waterway, but vegetation on the east would be sheltered. As the system passes, the winds originate from the back side and shift to an easterly direction. Fires started under these conditions would be carried to the east of waterways. However, as the system passes, low pressure replaces it and brings in high humidity and increased probability of precipitation, thereby reducing the likelihood of fire.

# Fire affects on Grasslands

## Factors Influencing Fires

Golley and Golley (1972) showed that grasslands can produce 20 percent more biomass than decomposes in a single growing season, and if the excess biomass is not removed by fire or grazing, the productivity of the grasslands declines. However, the response of grasslands to burning varies depending upon factors that include the amount of precipitation, grazing (which reduces fuel loading), fire frequency, timing of the fire, climatic conditions (especially rainfall and temperature) immediately before and after the burn, species composition, and fuel loading.

Oesterheld and others (1999) summarized the positive and negative effects of fire on productivity across a wide precipitation gradient (439 to 1129 mm annually) that included sites from North America, Africa, and the Mediterranean. Productivity increased by as much as 300 percent with fire and was reduced by as much as 80 percent in unburned control sites. The productivity increases were associated with sites receiving in the range of >700 mm of annual precipitation; conversely negative effects of fire on productivity occurred where precipitation was <600 mm. In the eastern portion of the tallgrass prairie, burning enhances productivity (Hadley and Kieckhefer 1963, Hulbert 1969, Kucera and Ehrenfield 1962, Old 1969, Peet and others 1975, Rice and Parenti 1978, Vogl 1974). Exceptions to this generalization have been reported for xeric sites (Dix and Butler 1954, Zedler and Loucks 1969), although Dhillion and Anderson (1994) and Anderson and others (1989) reported an increase in productivity following burning on a deep sand site in central Illinois. However, in the arid western portions of the North American Grassland, an increase in productivity does not always follow burning (Anderson 1976, 1982; Augustine and others 2010, Heirman and Wright 1973, Hopkins and others 1948, Launchbaugh 1972, Oesterheld and others 1999; Wright 1969, 1972).

## Time of the Burn and Productivity

In the Flint Hills of Kansas, at the western edge of the tallgrass prairie, the time of the burn influences grass productivity on native grass pastures. Burned sites had lower productivity than unburned sites following winter or early-spring burns, but late-spring burns and nonburned areas had equal productivity. Decline of productivity on early burns compared to late-spring burns is attributed to litter being removed from the site for a longer period of time followed by early growth, which depletes soil moisture (Knapp 1985). Absence of litter increases runoff and evaporation of moisture from the soil surface. The resulting decline in soil moisture is the primary cause for a decline in productivity (Anderson 1982, Bragg and Hulbert 1976, Knapp 1985, McMurphy and Anderson 1965, Owensby and Anderson 1967, Owensby and Smith 1972, Svejcar 1990). Nevertheless, cattle (*Bos taurus*) grazing on forage on burned sites make faster weight gains than those grazing on unburned sites, because the forage on burned sites is more palatable and higher in protein (Anderson 1976, Dyer and others 1982, Knapp and others 1999, McNaughton and others 1982).

## Ignition by Lightning and Humans

Lightning as an ignition source was important in the western portion of the Central Grassland and can cause prairie fires during the growing season if the vegetation is dry (Anderson 1982, Bragg 1995; Howe 1994a, 1994b). In the western portion of the tallgrass prairie, lightning fires in Nebraska averaged 138 per year from 1971 to 1975 and the historical fire season was from spring through autumn. Grasslands in the Great Plains originated during the Holocene and Native Americans have been on the continent for the past 30,000 years (Bragg 1995). They used fire as a vegetation management tool for a variety of reasons—including encouraging the growth of the prairie and preventing the encroachment of woody species—and as a tool for hunting, controlling insect, and easing travel (Anderson 1990, 1997; Curtis 1971; Pyne 1983, 1997; Stewart 1956). Consequently, lightning and Native Americans both had important roles in igniting grassland fires, especially in the frequency of summer fires (Devoto 1963). Summer fires are smaller those set during the dormant season, and they are often extinguished by rains associated with the storm that generated the lightning strikes (Bragg 1995).

In the wetter eastern portion of the Central Grassland, where rainfall also usually accompanies lightning storms, most fires were apparently set by Native Americans (Curtis 1971, Pyne 2001). In the eastern tallgrass prairie, fires occurred most frequently during the dormant season. Historical evidence suggests that many fires occurred in autumn, during the period known as "Indian summer" (McClain and Elzinga 1994), a warm, dry period following killing frosts (late October and early November).

## *Litter Removal*

### Fire and litter removal

The increased productivity on burned eastern tallgrass prairie is related to litter removal (Ehrenreich 1959, Hadley and Kieckhefer 1963, Hulbert 1969, Knapp 1984, Kucera and Ehrenreich 1962, Peet and others 1975, Weaver and Roland 1952). Old (1969) reported that litter removal increased productivity whether it was removed by fire or by mechanical means. One of the mechanisms whereby litter removal enhances productivity is through the alteration of microclimatic conditions on the burned site to conditions more favorable for the growth of the dominant $C_4$ grasses.

Litter is a good insulating surface and it has high reflectance of solar radiation and low net radiation, which is the difference between incident (irradiance striking a surface) and reflected solar radiation. Consequently, the soil warms up slowly in the spring (Peet and others 1975). On the burned surface, however, the insulating and highly reflective litter layer and standing dead biomass is removed by burning and replaced by a darkened highly absorptive surface. At the Curtis Prairie in University of Wisconsin-Madison Arboretum, daytime soil temperatures at 3-mm depths were warmer on the burned site than the unburned site. At night, the burned prairie has a good radiating surface (a good absorbing surface is also a good radiating surface) and cools rapidly, compared to the unburned site whose insulating litter cover retains heat. Consequently, nighttime temperatures were cooler in the upper layers of soil on the burned site. However, at 25 cm depth, the unburned site was constantly cooler than the burned site. The differences in microclimate between burned and unburned sites decreased as a grass canopy developed on the unburned site (Anderson 1972b, Brown 1967, Peet and others 1975).

The warmer soil temperatures during the day in early spring resulted in plants beginning growth earlier on burned prairie than the unburned prairie. Emergence of vegetation on the unburned site can be 7 to 14 days or as much as 30 days later (Knapp 1984). Peet and others (1975) reported that a burned site established a larger standing crop of vegetation (43.6 g/m$^2$) than the unburned site (1.24 g/m$^2$) by May 31 at the Curtis Prairie. They reported no difference in maximum photosynthetic rates of big bluestem on burned and unburned prairies. The higher productivity on burned prairies was attributed to the larger standing crop of green biomass earlier in the growing season (Peet and others 1975).

As leaves develop underneath standing dead biomass on the Konza Prairie, they are shaded and acquire the shade-leaf characteristics of low light saturation and photosynthesis. Standing dead litter reduces solar radiation and slows wind speed (89 percent lower than above the canopy), which reduces convectional cooling. Leaf temperatures of 30 to 35 °C (Black 1973) can rise above the optimum for $C_4$ photosynthesis (Knapp 1984). On burned grasslands, however, leaves develop in full sunlight as they emerge and have characteristics of sun leaves with high light saturation and photosynthesis. Additionally, on burned prairie, leaf temperatures are near the optimum for photosynthesis; the absence of standing dead biomass results in greater convectional cooling and higher wind speeds (57 percent lower than above the canopy). For example on June 10, leaf temperatures for big bluestem were 41.5 °C (7.9 °C above air temperature) on unburned sites and 39.4 °C (4.0 °C above air temperature), on burned sites (Knapp 1984). However, big bluestem plants on burned sites had greater water stress early in the growing season than plants on the unburned prairie (Knapp 1984). Microclimatic differences related to warmer spring temperatures on burned sites (Peet and others 1975), greater availability of solar radiation, and temperatures more favorable for optimum photosynthesis on burned sites are important factors in determining high productivity (Knapp 1984). Differences in results in photosynthetic rates of big bluestem on burned and unburned sites between the Kansas and Wisconsin may be due to differences in the height of standing dead tissue, which were 42 cm in Kansas and 10 cm in Wisconsin (Peet and others 1975, Knapp 1984). Leaves on unburned sites in Wisconsin should grow above of standing dead tissue more quickly than in Kansas and develop sun leaf characteristics resulting in similar photosynthetic rates between plants on burned and unburned sites. In contrast, in Kansas leaves would be shaded longer by standing dead tissues than in Wisconsin, retaining shade leaf characteristics and having lower rates of photosynthesis than plants on burned sites.

### Inorganic nutrients

The presence of a litter layer reduces the availability of inorganic nutrients, especially nitrogen which is thought to be the most limiting nutrient in grasslands. Annual burning of litter on prairies will reduce available nitrogen by about 1.0 to 4.0 g/m$^2$/year, about twice as much as is input by rainfall (Knapp and Seastedt 1986). Although compensating mechanisms are available to replenish nitrogen lost by burning, grasslands that are burned annually are subject to a long-term net loss of nitrogen (Ojima and others 1990).

## Grasslands and Grazers

In North America, expansion of the grassland biome occurred in the Miocene–Pliocene transition 5 to 7 million years ago and was associated with a concomitant increase in animals adapted to grazing, as was true for other areas (Axelrod 1985). Through the Pleistocene (1 to 3 million years ago), a diverse grazing megafauna (weight > 44 Kg) (Doughty, Wolf and Field 2010) on the continent included 32 genera and dozens of species of mammals such as camels (*Camelus* spp.), horses (*Equus caballus*), rhinoceroses (*Rhinoceros* spp.), antelopes (*Antilocapra* spp.), bison (*Bison bison*), and elephants (*Elephas maximus*). Near the end of the Pleistocene beginning about 25,000 years ago, the number of grazing species sharply declined. This sharp decline has been attributed to the appearance of efficient human hunters or climatic change or both (Ehleringer and others 2002, Flores 1996). The peak of the American-evolved megafaunal crash occurred about 14,800 to 13,700 years ago, and by 10,000 years ago only about a half dozen browsing and grazing forms remained. When Europeans arrived, the bison, elk, and other animals that characterized the grasslands were the remnants that survived the massive extinction at the end of the Pleistocene (Doughty, Wolf, and Field 2010, Flores 1996, Gill and others 2009).

Because of the long-term association of grazing animals and grasslands, it is not surprising that several lines of evidence suggest that grazers strongly influence the productivity and diversity of grasslands. Golley and Golley (1972) suggest that the productivity of biomass in excess of that which can be decomposed is a response to grazing. Grazing, like burning, accelerates the rates of mineralization of inorganic nutrients (Frank and others 1998). For example, grazers like bison are effective in changing some recalcitrant species of nitrogen to urea that is easily converted to ammonia, a plant-useable form of nitrogen. The increased availability of inorganic nutrients can enhance grassland productivity (Knapp and others 1999). Grazing removes the physiologically older, less productive leaf tissue and these changes increase light and moisture for younger more photosynthetically active tissue, which enhances aboveground productivity (Frank and others 1998). Some authors (McNaughton 1979, 1993; Owen 1981) have proposed a symbiotic relationship between grasses and grazers. Aboveground productivity of grasslands has been said to increase with moderate grazing (Knapp and others 1999, McNaughton 1979), although others have questioned the beneficial effects of grazing (Belsky 1986, Painter and Belsky 1993). Additionally, evidence suggests that increased shoot productivity occurs at the expense of belowground productivity and nitrogen and carbon are transferred from below ground to facilitate compensatory aboveground growth following grazing (Collins and others 1998, Knapp and others 1999), meaning that excessive grazing will eventually cause a decline in productivity (Anderson 1982).

## *Bison as a Keystone Prairie Species*

### Grazing patterns and preferences

The role of bison in tallgrass prairie has not been understood until the last two decades, when reserve areas became available that were large enough to support reasonably sized populations and allow them to graze in a way that simulated historical conditions. Knapp and others (1999) delineated a keystone role for bison in maintaining diversity of tallgrass prairie. On Konza Prairie, bison fed primarily on grasses (90 percent) and consumed only small quantities of forbs and essentially no woody vegetation (Fahnestock and Knapp 1993, Hartnet and others 1996, Knapp and others 1999, Steuter 1997, Vinton and others 1993). Although grasses constituted the largest portion of the bison diet, the proportion of $C_3$ and $C_4$ grasses consumed varied seasonally.

Generally, mammalian herbivores prefer $C_3$ grasses, which have higher digestibility and protein content but lower carbon:nitrogen ratios than $C_4$ grasses (Ehleringer and others 2002). Nevertheless, while some studies reported that $C_4$ grasses have more fibers and higher silica concentration in their leaves than $C_3$ grasses (Kaiser 1998, Kepart and Buxton 1993), other studies found no difference in the two traits between $C_3$ and $C_4$ grasses (Heidorn and Joern 1984, Scheirs and others 2001). In South Dakota, $C_4$ grasses constituted 33 to 44 percent of the bison diet from early June through August and then declined to 15 percent by the end of September. Bison use of sedges and $C_3$ grasses increased from 52 to 58 percent in summer (mid-June to mid-August) to >80 percent after the beginning of September (Plumb and Dodd 1993). Similar patterns in seasonal consumption shifts were found on the Konza Prairie (Vinton and others 1993).

Bison grazed in two patterns, creating distinctive grazing patches that were 20 to 50 $m^2$ in area and more extensive patches that were >400 $m^2$. During the growing season, bison revisited previously grazed sites in preference to ungrazed locations. The grass that grew after grazing was higher in nitrogen, more palatable, and not intermixed with dead tissue. Grazed areas initially experienced short-lived productivity increases after grazing, but productivity eventually declined as movement of carbon reserves from belowground compensated for loss of aboveground tissues. By repeatedly grazing the same areas, bison encouraged the growth of non-palatable species that are the forbs. This grazing pattern eventually encouraged a shift to other areas as forage quality declined. On average, 6 to 7 percent of the grazing patches were abandoned annually (Knapp and others 1999).

### Enhancing grassland plant diversity

Bison grazing can offset negative effects of frequent burning on plant species diversity (Gibson and Collins 1990, Knapp and others 1999). Burning favors $C_4$, warm season grasses and late flowering forbs. Frequent fires, especially annual burns, can encourage these grasses at the expense of $C_3$ plants, which include many species of forbs (Gibson and Collins 1990, Knapp and others 1999, Kucera and Koelling 1964). And forbs contribute most of the species richness to the prairie (Hartnett and Fay 1998, Howe 1994a). Bison graze on the $C_4$ grasses and reduce their abundance, which favors unpalatable $C_3$ forbs, which enhances the plant diversity of the prairie.

### Effects on animal diversity

Bison enhance spatial heterogeneity in the prairie through grazing patterns that result in patches of lightly grazed to heavily grazed areas with sparse grass cover and little litter (Fuhlendorf and Engle 2001, Knapp and others 1999). This spatial heterogeneity is important for grassland bird diversity. In the eastern tallgrass prairie, some birds, such as the killdeer (*Charadrius vociferus*) and upland sandpiper (*Bartramia longicauda*), require sparse vegetation across large areas. Other species, such as eastern meadowlark (*Sturnella magna*) and bobolink (*Dolichonyx oryzivorus*), utilize medium-height vegetation with moderate amounts of litter, whereas species, such as Henslow's sparrow (*Ammodramus henslowii*) and marsh wren (*Cistothorus palustris*), occur where the vegetation is tall and litter is abundant (Herkert and others 1993). Endemic birds of western Great Plains also have characteristic distributions related to historical grassland types and grazing patterns (Knopf 1996).

Fire and bison grazing affect the diversity and density of grasshoppers (suborder Caelifera) Joern (2005) found that upland or lowland topographic position and fire frequency had no significant affect on grasshopper species richness or diversity (Shannon Index) on the Konza Prairie. However, bison grazing increased species richness, diversity, and evenness of grasshoppers. Grasshopper species richness was positively related to plant species richness and heterogeneity in plant height. Joern (2005) concluded that fire influences grazing patterns, which effects structure and plant species richness in grasslands (Fuhlendorf and Engle 2001, Hartnett and others 1996, Knapp and others 1999, Pfeiffer and Hartnett 1995, Pfeiffer and Stueter 1994, Vinton and others 1993). Consequentially, fire and large mammalian grazing are crucial features for maintenance of grasshopper diversity.

Grassland small mammals (microtine rodents) also require a diversity of vegetation and litter density. Even though their responses to burning are mixed—positive for deer mice (*Peromyscus maniculatus*) and negative for western harvest mice (*Reithrodontomys megalotis*) and prairie voles (*Microtus ochrogaster*)—those responding negatively recover in 2 to 3 years after the burn (Kaufman and others 1990, Schramm 1970, Schramm and Willcutts 1983). Fires of varied intensity and completeness should favor diversity of animals in grasslands. The mosaic of vegetation resulting from grazing creates uneven patterns of fire intensity as a result of fuel loadings that vary from heavy fuel loading in sparsely grazed areas to low fuel loading in areas subjected to intensive grazing pressure.

## *White-tailed Deer in Remnant Tallgrass Prairie*

Historically, the bison was the most important large mammalian herbivore in much of the Central Grassland tallgrass prairie; although its abundance may have been substantially lower in the eastern portion (Leach and others 1999). Today, the white-tailed deer (*O. virginianus*) is the large native mammal with the most impact on remnant and restored tallgrass prairies. Although bison graze almost entirely on grass, forbs (which are little used by bison) are favored by white-tailed deer. Anderson and others (2001) reported that deer browsed very little on grasses or sedges during the late spring and summer, but browsed from 3.5 to 18.9 percent of the standing crop of forb stems depending upon time of sampling. Because forbs contribute most of the diversity to the prairie (Howe 1994a) excessive white-tailed deer browsing can reduce the prairie diversity. Anderson and others (2005) demonstrated that diversity of prairie forbs was maximized at an intermediate level of deer browsing—supporting the intermediate disturbance hypothesis, which posits that diversity is maximized at intermediate levels of disturbance (Connell 1978). However, the community quality of forbs, based on the degree to which species were associated with relatively undisturbed remnant prairies, declined as the duration of intense deer browsing (disturbance) increased. Forb quality was highest after 8 years of protection from browsing, suggesting a potential tradeoff between maximizing diversity and maintaining quality of forb communities (Anderson and others 2006).

# Managing Tallgrass Prairie

## *Overview of Grassland Management*

Successful prairie management requires knowledge about the ecology of individual prairie species, and how functional groups (such as $C_4$ grasses, $C_3$ grasses, early flowering forbs, and litter dwelling invertebrates) will respond to various management prescriptions. Fire is the most widely applied tool for managing tallgrass prairie and—because of rainfall being associated with lightning storms and habitat fragmentation, both of which limit the spread of fire—prescribed burning is necessary to maintain prairies and preserve plant species diversity (Leach and Givnish 1996). Nevertheless, the response of the whole community must be considered when fire or other management practices are applied. Compromises will have to be made in deciding on which practices to use when species or groups of species respond differently to management. This chapter does not provide a comprehensive management guide to prairie management as occurs in Packard and Mutel (1997). Rather it discusses some issues that are often of concern in the management of prairies.

## *Timing of the Burn*

Prescribed fires most frequently occur in spring because the opportunity for burning is generally longer than in the autumn. Spring burns also have the advantage of retaining winter cover for wildlife. Nevertheless, autumn is apparently the time when most of the Native American set fires occurred; the effect of autumn vs. spring burns may be a yet unknown factor in community response to fire. Autumn burns occur when birds and mammals are not actively breeding. Large mammals readily move away from the fire and are rarely directly affected by burning. Direct small mammals losses in prescribed burns are usually small, even in head fires, but they do occur. Nevertheless, it is relatively common to burn cottontail rabbit (*Sylvilagus floridanus*) nests during spring burns, especially if the burn is delayed so that nonnative cool season plants like Kentucky bluegrass (*Poa pratensis*) and sweet clover (*Melilotus* spp.) are actively growing and can be set back by the burn (Curtis and Partch 1948; Kline 1986). Snakes can be active in the spring and suffer mortality in spring burns, although snakes seek shelter in holes and animal burrows during fires.

Summer burns have been proposed as a way of enhancing plant species diversity on prairies and also controlling invading woody species. Adams and others (1982) compared woody vegetation response to a summer burn (July) and a late-winter dormant season burn (March) in south central Oklahoma. The authors tested the hypothesis that summer burns would be more detrimental to woody species than dormant season burns, because the plants would have invested resources in building new leaves but would not have returned resources belowground to replace those used in the current year's growth. Unexpectedly, the late-winter burn was more detrimental than the summer burn to woody species, which they attributed to an unusually severe drought following the winter burn. Additionally, woody and herbaceous species can regrow in the same growing season in which a summer burn occurs; this may mitigate the affects of summer burns (Anderson 1972a).

The application of summer burns to enhance the diversity of prairie plants has been examined in several studies (Copeland and others 2002; Howe 1994a, 1994b). Burning in the summer when the dominant $C_4$ plants are actively growing should reduce their competitiveness against $C_3$ plants and reduce $C_4$ dominance. Copeland and others

(2002) reported a twofold increase in the species richness and average frequency of subdominant species in plots subjected to a late-summer fire, but these two measurements remained unchanged in plots subjected to early spring fires. Similarly, Abrams and Hulbert (1987) found that spring burning had no effect on plant species richness.

Although summer burns may have applicability, we should move cautiously on the use of summer burns, because information is lacking on animal responses and is insufficient on plant responses (Anderson 1997). In addition, under some conditions—for example, when green vegetation is dry enough to burn but still has high moisture content and is actively growing, with little dead biomass—summer burns can generate abundant smoke, which is substantially more irritating to respiratory systems than dormant season burns.

## *Fire Frequency*

### Vegetation response

For mesic tallgrass prairies, fires every 2 to 3 years will normally be appropriate. On dry prairies fire interval should be longer, perhaps in the range of 3 to 5 years. However, careful monitoring of the vegetation may indicate more or less frequent burning. Factors to be considered include the rate, at which litter accumulates, control of woody species, abundance of invasive weeds, and the relative abundance of forbs and $C_4$ grasses. Some biennial weeds, such as sweet clover (*Melilotus alba* and *M. officinalis*), may benefit from a burning schedule that occurs at intervals equal to or greater than two years and will require a varied burn schedule and/or mowing before seed set of sweet clover to achieve effective control (Kline 1986, Coles 2007).

### Small mammals

Fire alters vegetation composition and structure, benefiting some small mammal species and causing others to have less favorable habitat conditions (Kaufman and others 1990). However, as previously noted species decreasing in abundance following fire recovered within 2 to 3 years. Burning sections of the prairie on a 2- or 3-year rotational basis should meet the habitats needs of most mammals.

### Preserving invertebrates

The response of invertebrates to burning is varied and depends on a number of factors, including where the invertebrate is located at the time of the fire (Macfadyen 1952, Reichert and Reeder 1972, Seastedt 1984, Warren and others 1987), the microclimatic and vegetation structural changes after fire, and the ability of the invertebrate to adapt to the changed environment (Anderson 1964, Anderson and others 1989; Evans 1984, 1988). For example, during a burn that had surface temperatures of 200 °C, species of spiders that were active on the soil surface at the time of a burn were eliminated, whereas others survived in subsurface burrows, under rocks, or protected in the bases of caespitose (clumped) grasses (Riechert and Reeder 1972). Similarly, mixed responses of species to fire were reported for mites (Seastedt 1984), Collembola (springtails) (Lussenhop 1976, Van Amburg and others 1981), and grasshoppers (Anderson and others 1989, Evans 1988).

Deciding on appropriate grassland management methods to accommodate the needs of arthropods can be complicated. For example, some species of insects, such as butterflies and leafhoppers decrease in abundance after fire (Panzer 1988; Swengel 1996, 1998; Swengel and Swengel 2001). Grasshoppers that feed on forbs increased in frequency as fire frequency decreased; however, some grasshopper species increased or showed rapid recovery after fire (Anderson and others 1989, Evans 1988). For specialist butterfly species—those restricted to prairie, savanna, or barrens—occasional single wildfires were more favorable than rotational burning, and mechanical cutting more favorable than grazing. However, widely distributed butterflies were favored by more frequent management; mechanical cutting was not more favorable than grazing, occasional wildfires, or rotational burning (Swengel 1998). Thus, it is not possible to have a single management prescription that will be optimal for all insects; an increasing number of entomologists are expressing concern that prairie insects are being harmed by current prescribed burning practices, and if continued, the outcome could be the loss of a substantial number of species (Pyle 1997, Schlicht and Orwig 1992). Swengel and Swengel (2007) recommend that permanent non-fire refugia be established to conserve Lepidoptera and these areas be managed with methods, such as brush cutting and mowing, if necessary. However, Panzer and Schwartz (2000) concluded that the current rotational plan (burn about every 2 to 3 years) in Illinois has been compatible with conservation of insect biodiversity.

Historically, some portions of extensive grasslands likely remained unburned each year and provided “refugia” for fire sensitive insects. However, under current conditions, which often involve burning fragmented remnant prairies or restorations, all or nearly all of the area is burned. Possible solutions to this management conundrum include burning only a portion of each site on a rotational basis, leaving 50 to 70 percent as unburned refugia so that fire sensitive species can reinvade the burned site after it regrows (Andrew and Leach 2006; Panzer 1988, Panzer 2003; Panzer and Schwartz 2000). Additionally, recommendations for burning practices to favor fire sensitive insects include leaving areas missed by the fire unburned, avoiding “hot fires” by burning early in the morning, and using spring burns to preserve clumps of grasses that are used as wintering sites (Panzer 1988).

### Control of woody species

For a variety of reasons, fire does not always keep woody species under control or prevent their invasion into grasslands. Herbicide application is often necessary to achieve a reduction in woody vegetation (Solecki 1997).

## *Grazing and Fire Management*

### Patch-burn grazing

Grazing by bison has been shown to increase the plant diversity and spatial heterogeneity of grasslands; however, until recently, there were few studies (Fuhlendorf and Engel 2001, Steuter 1997) that examined the potential of grazing and burning combined as a grassland management tool. Nonetheless, “patch-burn grazing,” which combines the two, may increase prairie diversity, especially in the western portion of the tallgrass prairie in Kansas and Oklahoma, where tallgrass prairie is used for cattle grazing. A common range management practice is annual spring burns followed by early cattle grazing and removal in mid- to late-summer, with double the cattle stock rates of year round grazing operations (Coppedge and others 2008, Powell 2006). This application of grazing and burning, called intensive early stocking (IES), results in low diversity grassland system with little habitat heterogeneity. Patch-burn grazing is an alternative to this management approach (Fuhlendorf and Engle 2001, Helzer and Steuter 2005, Steuter 1997, Towne and others 2005). Grazing animals (cattle or bison) are permitted to graze freely across the prescription area that has recently burned and unburned patches. Typically one-third to one-fourth of the area is burned annually on a rotational basis. The burning increases forage production and quality, and consequently, grazing pressure on the burned areas in the first year after the fire (Fuhlendorf and Engle 2001, Knapp 1999, Towne and others 2005). The combination of recently burned and heavily grazed sites and sites that have been unburned for 2 to 3 years, depending on rotation time, provides sites with varied structural features and plant species abundances. Recently burned sites have less cover of tallgrass, litter, and shorter vegetation height, but more bare ground than unburned sites (Fuhlendorf and others 2006). Birds are especially sensitive to these structural changes and species diversity of grassland bird species increases with patch-burn grazing as compared to annually burned grasslands (Coppedge and others 2008, Fuhlendorf and others 2006, Powell 2006).

### Endangered species management

Although grazing can have positive and negative affects, it returns a historical function to grasslands that has the potential to increase diversity. For example, at the Midewin National Tallgrass Prairie in northeastern Illinois, cattle grazing on cool season domesticated grasses has provided habitat for the State's largest nesting population of upland sandpipers, "prairie plover," an endangered species in Illinois (Meyer 2002). We need to know if native grazers or their surrogates can produce similar habitats, short grass with bare ground, on restored native tallgrass prairie. Heavy grazing before and during the breeding season—with periods of cessation so the native grasses can recover—and rotating the portions of the grassland that are grazed annually might provide breeding habitat for the upland sandpiper and retain plant species diversity.

### Prairie restoration

The intense grazing on the patch-burn areas creates openings and reduces competition from the $C_4$ grasses that are preferred and encourages the growth of ruderals and cool-season plants including less palatable forbs, which increase in abundance in the year after the burn (Fuhlendorf and Engle 2001, Towne and others 2005). On restored prairies, the heavily grazed patch-burn areas could be sown with forb seeds to enhance species richness. This procedure may be especially useful to increase forb diversity on restorations that have a heavy dominance of $C_4$ grasses and low diversity of forbs. Stocking rates may have to be modified for each specific site to prevent overgrazing.

### Management decisions

Grazing is not an option on all prairies because of size limitations and other factors. Leach and others (1999) have proposed that the eastern portions of the tallgrass prairie supports plant species that are sensitive to grazing; historically, bison were not abundant on these prairies so a strong interaction between grazers and prairies did not develop. This concern remains an unresolved issue (Henderson 1999, Howe 1999). Moreover, the effect of patch-burn grazing on the diversity of prairie forbs is not as well documented as the effect grassland on birds, and further studies are need in this area. Moreover, in the highly fragmented grasslands of the eastern portions of the tallgrass prairie in Iowa and Missouri, no treatment effects were observed for bird species richness, grassland obligate bird species richness, or diversity for patch-burn grazing, burn only and grazed-and-burned treatments. Differentiation of community structure was most strongly correlated with visual obstruction and wooded edge density, to which birds are responsive, and are well developed in fragmented habitats (Pillsbury 2011).

The choice of whether to use bison or cattle for grassland management depends on a number of factors. Their grazing patterns differ somewhat, with cattle consuming more forbs and browse. Economic returns are greater from cattle than bison and space needs and facility and management costs are greater with bison, but bison provide better management for natural areas (Plumb and Dodd 1993).

## Conclusions

Managing vegetation requires an understanding of the major ecosystem functions that originally maintained the system and how these functions can be reestablished or manipulated to ensure stability and health. The practice of adaptive management can be applied and the health of the ecosystem monitored to determine if management goals are achieved. Management prescriptions should be continued, modified, or abandoned (with alternate procedures adopted) to achieve desired goals. Records should be kept of the management practices applied and their effectiveness. This information would be valuable to others managing prairies. Additionally, more research on prairie management practices is needed—if the management is designed appropriately it can serve as an experiment to test options.

The Central Grasslands are of recent origin, and have depended on human intervention, through fire management, at least for the past 6,000 years or longer. Persistence of this ecosystem will require continued fire management or appropriate surrogate practices. The most effective management will have goals that are holistic in their scope and will attempt to preserve the diversity and stability of all trophic levels. Large mammalian grazers and browsers have keystone roles in grasslands and, to the extent that it is possible, their effects on this ecosystem should be included in management practices and goals.

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Cumulative Watershed Effects of Fuel Management in the Eastern United States

Chapter 10.

# Cumulative Effects of Fuel Management on the Soils of Eastern U.S. Watersheds

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Fuel management treatments in the Eastern United States encompass diverse activities that have a range of potential impacts on the soils within watersheds of managed forests and grasslands. In industrial or production forests, the predominant fuel management strategies are intensive site preparation (bedding, roller chopping, and burning slash), use of herbicides, and pre-commercial or early rotation thinning; these activities probably impact the most land area in the East. On public lands that are managed for natural resources, the fuel treatment strategies often are more varied and can include herbicide applications and thinning, prescribed fire, grazing, or targeted chainsaw-felling of specific understory species. Thus, effects of fuel management on forest soils can be very subtle or protracted such as a plant-soil-microbe feedback resulting from removal of a single plant species; or they can be acute and profound such as the direct soil-profile disrupting disturbances associated with site preparation and logging. Because the functions of forest soil arise through complex interactions among physical, chemical, and biological components, this chapter will address the effects of individual fuel treatment practices on each of these components (Burger 1994).

A wide range of different ecosystem types occupies the eastern landscapes of North America, and this diversity is reflected in the underlying soils. Eastern soils differ from one another across broad ranges of climatic conditions, parent material, topography (elevation and aspect), age, disturbance history, and the biota that they support—all factors that influence the long-term development of soil and ultimately determine what type of soil will be found in a given location (Jenny 1941). Soils in the Eastern United States fall into nearly every order, and are classified into hundreds of series (see chapter 3). Here we attempt to review the effects of fuel management practices (specifically prescribed fire and mechanical fuel treatments) on soils of eastern North America by collecting and synthesizing available soil-related data from as many different ecosystem types and soil types as possible. The reviewed material is therefore necessarily very broad in scope.

## Prescribed Fire Effects on Eastern Soils

Prescribed fire is probably the most widely used treatment for fuel reduction in the ecologic divisions of the Eastern United States (Cleland and others 2007). These fires may be applied to logging slash as a component of site preparation for new plantings, or they may be applied as surface fires to reduce understory vegetation or promote certain desirable plant and animal species. Furthermore, fire serves a crucial functional role in many (if not most) wildland ecosystems of the Eastern United States. This relationship is particularly well known in the Subtropical Division (230) pine-dominated (*Pinus* spp.) ecosystems of the Atlantic and southeastern Coastal Plains, and equally so in Prairie Division (250) tallgrass prairie ecosystems of the Midwest. Prescribed fires are also increasingly used for fuels management in the Warm Continental (210) and Hot Continental (220) Division pine forests of the Lake States, but less is known about their effects on ecosystem properties. Finally, although the role of fire in eastern hardwood forests, primarily in the Hot Continental and Hot Continental Mountains (M220) Divisions is less well known than for pine forests, much work has been performed in recent years to shed light on this important question.

## Physical Effects of Prescribed Fire

The predominant physical effects of fire on forest soils (table 1) include heat transfer, development of hydrophobic conditions, higher soil temperature, increased risk of erosion, and degradation of soil aggregate structure. Heat transfer and hydrophobicity in soils are closely linked because heat causes volatilization of waxes and oils in organic material; these diffuse into soils and then condense around soil particles, causing them to be water repellent. The degree to which this process occurs depends on fire temperature, residence time, and the characteristics of the organic matter (DeBano 2000). The development of hydrophobicity in eastern soils does not appear to be a substantially negative consequence of prescribed fires—we were unable to find any documented cases of this phenomenon in the East.

The degradation of soil aggregate structure as a potential physical effect of prescribed fires has been hypothesized for oak (*Quercus* spp.) savanna ecosystems of Missouri, but this phenomenon has yet to be directly measured (Rhoades and others 2004). These authors suggested that destruction of aggregate structure might partially explain the slow recovery of plant communities observed in soils where large downed logs had "burned out" in a prescribed fire. Such aggregate destruction may be related to the observed changes in soil texture, as well as changes in water infiltration and water-holding capacity of the soils impacted by the intense "burn outs." In any event, the net watershed effect of such impacts will be dependent upon the amount of large down wood in burned areas and how these materials are consumed.

Increased soil erosion has been observed in wildfire-impacted areas, but evidence for large soil losses from erosion in burned areas is limited. For example, in relatively steep slopes (35 to 45 percent) in the Southern Appalachian Mountains of the Hot Continental Mountains Division, Swift and others (1993) observed localized movements of soil in an area that had been burned in a prescribed fire, but they also reported no net soil loss from the treatment area. These authors attributed the sediment retention observed in their study to entrapment of sediments in the remaining intact forest floor, which was ≤66 percent consumed in the fires. Perhaps more important than soil erosion from the prescribed burn is erosion associated with fire control activities to prevent escape, and in particular the use of plowed fire lines (Van Lear and others 1985).

## Chemical Effects of Prescribed Fire

### Carbon

The pools of carbon that are likely to be affected by prescribed fire include plant roots, total soil organic carbon, microbial biomass carbon, and "black" carbon (charcoal and soot). All of these pools are more or less tightly related to one another, and fire-induced changes in one pool are likely to be associated with changes in others. The magnitude of fire effects on soil carbon pools largely depends upon the intensity and frequency of fires, soil type, and forest type (table 2).

**Table 1.** Physical effects of prescribed fire or treatments including prescribed fire (such as thinning or herbicide application) on soils in Eastern North America

| Fire management objective | Ecosystem | Location (State/ Province) | Ecological division[a] | Organic horizon[b] | Moisture | Temperature | Texture | Bulk density | Reference |
|---|---|---|---|---|---|---|---|---|---|
| Habitat improvement | Outwash sandplain | MA | 220 | n/a | n/a | n/a | n/a | = | Neill and others 2007b |
| | Scrub oak | FL | 230 | + charcoal | n/a | n/a | n/a | n/a | Alexis and others 2007 |
| Restoration | Oak | IL | 220 | – mass | n/a | n/a | n/a | n/a | Brand 2002 |
| | Oak savanna | MO | 220 | n/a | n/a | n/a | + clay particles | = | Rhoades and others 2004 |
| | Oak-hickory | OH | 220 | n/a | = | = | n/a | n/a | McCarthy and Brown 2006 |
| | Oak-hickory | OH | 220 | – mass | n/a | n/a | n/a | n/a | Dress and Boerner 2004 |
| | Oak-hickory | TN | M220 | n/a | = | n/a | n/a | n/a | Jackson and others 2006 |
| | Oak-hickory-grass | KY | 220 | | – | + | | | Rhoades and others 2002 |
| | Oak-pine | MA | 220 | – summer burns | n/a | n/a | n/a | + in organic horizon | Neill and others 2007a |
| | Oak/pine | TN/GA | M220 | – mass Oi only | + short term | = at 10 cm | n/a | n/a | Hubbard and others 2004 |
| Site preparation | Shortleaf pine-grass | AR | M230 | Site dependent – or = | n/a | n/a | n/a | n/a | Liechty and others 2002 |
| | Black spruce | NF | 210 | – in Oi and Oa | n/a | n/a | n/a | n/a | Scheuner and others 2004 |
| | White pine | NC | M220 | – mass Oi | n/a | n/a | n/a | n/a | Vose and Swank 1993 |
| | White pine | NC | M220 | n/a | + in fell/burn | + in fell/burn | n/a | n/a | Swift and others 1993 |

n/a indicates that no data is available.
+ indicates a statistically significant increase in the measured variable.
= indicates no statistically significant change in the variable.
– indicates a statistically significant decrease in the variable.
[a] 210 is Warm Continental, 220 is Hot Continental, M220 is Hot Continental Mountains, 230 is Subtropical, M230 is Subtropical Mountains (Cleland and others 2007).
[b] Symbols are consistent with U.S. Department of Agriculture Natural Resources Conservation Service soil taxonomy (Oi is litter layer, Oa is fermentation layer, Oe is humus layer).

**Table 2.** Effects of prescribed fire or treatments including prescribed fire (such as thinning or herbicide application) on carbon (C), black carbon (BC), total nitrogen (N), carbon to nitrogen ratio (C:N), ammonium ($NH_4$), nitrate ($NO_3$), nitrogen mineralization ($N_{min}$), and dissolved organic nitrogen (DON) in the soils of Eastern North America

| Fire management objective | Ecosystem | Location (State/ Province) | Ecological division[a] | C | BC | N | C:N | $NH_4$ | $NO_3$ | $N_{min}$ | DON | Reference |
|---|---|---|---|---|---|---|---|---|---|---|---|---|
| Fuel reduction | Many | Many | 220, M220, 230 | = | n/a | n/a | n/a | n/a | n/a | n/a | n/a | Boerner and others 2008 |
| | Oak savanna | MN | 210 | n/a | n/a | – | n/a | + | = | + | + | Dijkstra and others 2006 |
| | Oak-hickory | OH | 220 | = | n/a | n/a | n/a | = | = | n/a | = | Giai and Boerner 2007 |
| | Pine barrens | NJ | 230 | n/a | n/a | – | n/a | + | + | n/a | n/a | Gray and Dighton 2006 |
| | Prairie | AR | | + | n/a | + | + | n/a | n/a | + | n/a | Brye 2006 |
| Habitat improvement | Scrub oak | FL | 230 | –/= | + | –/= | n/a | n/a | n/a | n/a | n/a | Alexis and others 2007 |
| | Longleaf pine | GA | 230 | – | n/a | – | n/a | + | = | n/a | n/a | Boring and others 2004 |
| | Outwash sandplain | MA | 220 | = | n/a | = | = | = | = | = | n/a | Neill and others 2007a |
| Restoration | Grassland | MD | | = | n/a | n/a | n/a | n/a | n/a | n/a | n/a | Sherman and others 2005 |
| | Oak-hickory | OH | 220 | + | n/a | n/a | n/a | n/a | n/a | =/– | n/a | Boerner and others 2005 |
| | Oak savanna | MO | 220 | = | n/a | = | + | = | + | n/a | n/a | Rhoades and others 2004 |
| | Oak-hickory | OH | 220 | n/a | n/a | n/a | n/a | n/a | n/a | n/a | n/a | Huang and Boerner 2007 |
| | Oak-pine | MA | 220 | = | n/a | = | = | n/a | n/a | n/a | n/a | Neill and others 2007a |
| | Oak-pine | TN/GA | M220 | –/= | n/a | –/= | n/a | = | = | n/a | n/a | Hubbard and others 2004 |
| | Pine-bluestem | AR | M230 | + | n/a | + | + | n/a | n/a | n/a | n/a | Leichty and others 2005 |
| | Mesquite savanna | TX | 310 | = | + | n/a | n/a | n/a | n/a | n/a | n/a | Dai and others 2005 |
| | Oak/Hickory - grass barrens | KY | 220 | n/a | n/a | n/a | n/a | + | = | + | n/a | Rhoades and others 2002 |
| Site preparation | White pine | NC | M220 | n/a | n/a | = | n/a | + | =/+ | +/= | n/a | Knoepp and Swank 1997 |
| | White pine | NC | M220 | = | n/a | = | n/a | + | =/+ | +/= | n/a | Knoepp and others 2004 |
| | White Pine | NC | M220 | –/= | n/a | –/= | n/a | n/a | n/a | n/a | n/a | Vose and Swank 1993 |
| | Jack pine | ON | 210 | | n/a | n/a | n/a | n/a | n/a | n/a | n/a | Staddon and others 1998 |
| | Black spruce | NF | 210 | = | n/a | = | = | n/a | n/a | n/a | n/a | Scheuner and others 2004 |

n/a indicates that no data is available.
+ indicates a statistically significant increase in the measured variable.
= indicates no statistically significant change in the variable.
– indicates a statistically significant decrease in the variable.
Note: When two symbols are presented for a given variable, the first represents a shallower soil depth (typically organic horizon soil), and the second represents a deeper soil.
[a] 210 is Warm Continental, 220 is Hot Continental, M220 is Hot Continental Mountains, 230 is Subtropical, M230 is Subtropical Mountains, 310 is Tropical/Subtropical Steppe (Cleland and others 2007).

## Plant root carbon

A large proportion of management-induced changes in soil organic-matter carbon can be traced to cumulative effects on carbon dynamics associated with plant roots. Among other management practices, prescribed fire can strongly influence the plant community found in forested stands, and this depends largely on the frequency and intensity of fire. In general terms, the shorter the fire-return interval, the more prevalent perennial grasses become in understory vegetation. This pattern is typical of mesic grassland systems, for example in Prairie Division tallgrass prairies of eastern Kansas, where fire frequency and the cover of warm-season perennial grasses are clearly related (Knapp and others 1998), and where more total root biomass can be found in frequently burned soils than unburned soils (Kitchen and others 2009). Because grass root tissues typically have very wide carbon:nitrogen ratios, the decomposition of this material is slower than analogous root tissue from forbs or woody species; the net effect is of larger accumulations of total soil organic carbon in systems that have higher warm-season grass cover (Knapp and others 1998).

Increases in perennial grass cover with frequent fire are also well known from forested systems such as the Subtropical Division longleaf pines (*P. palustris*) on the southern Coastal Plain (Brockway and Lewis 1997, Glitzenstein and others 2003) and loblolly and shortleaf pines (*P. taeda* and *P. echinata*) on the Southern Piedmont (Phillips and Waldrop 2008); and the Subtropical Mountains (M230) Division shortleaf-bluestem (*Andropogon* spp.) systems in the Arkansas Ouachita Mountains (Liechty et al. 2005). In other systems where fire-return interval is longer, or where fire has been excluded for a long period and prescribed fires have only recently been reintroduced, there has been little documented change in understory plant community with fire. This has been true for the Ohio hardwood forests in the Hot Continental Division (Hutchinson and others 2005), and in jack pine (*P. banksiana)* systems in Ontario (similar to those found in the Warm Continental Division's Great Lakes States), where prescribed (site preparation) fires reduced grass cover in the first year following fire, but effects were negligible after the second year (Tellier and others 1995).

## Soil organic carbon

One of the long-term consequences of increased inputs of grass-derived detritus is the accumulation of soil organic carbon. This is particularly true for grassland soils, which have long been noted for their high organic matter content, but it is a pattern that holds for any system with extended periods of increased grass cover. Organic matter accumulation in soils with a large component of grass in the understory is the result of the much higher carbon-to-nitrogen ratio in grass material. The carbon-to-nitrogen ratio in organic matter is of critical importance because material with a high ratio takes longer to decompose, and gives rise to more recalcitrant forms of organic matter in the later stages of decomposition (with potential to ultimately change the amount of carbon stored in a particular soil profile). Thus, the net effect of frequent prescribed fire is increased inputs of organic matter that often have longer turnover time (relative to organic matter in unburned systems); thus, an indirect effect of prescribed fire is an increase in the net storage of carbon in mineral soil horizons. Other forms of soil organic carbon that are influenced by the occurrence of prescribed fire include microbial biomass carbon and charcoal and soot (black carbon or BC), which are discussed below.

## Black carbon

Not all ecosystem carbon subjected to prescribed burning is volatilized to carbon dioxide. Depending on the fire severity, a fraction will remain in the ecosystem in the form of highly recalcitrant carbon (black carbon). The importance of black carbon in the total carbon cycle of fire-impacted ecosystems is increasingly being recognized (DeLuca and Aplet 2008). However, several aspects of the input and cycling of black carbon, for example in response to different fire frequencies, have not been thoroughly examined. Charcoal, elemental carbon, and soot derived from biomass burning are generally considered as a recalcitrant pool with a very long turnover time from centuries to millennia (Deluca and Aplet 2008). The chemical interactions between black carbon and other organic matter constituents (microbial pools, humus, soil organic matter, and fresh litter), however, are complex and not well studied, with a few notable exceptions such as Wardle and others (2008) and Czimczik and Masiello (2007). Available published data on black carbon formation and its interactions are primarily derived from ecosystems with long fire-return intervals (DeLuca and Aplet 2008; Wardle and others 2008), and these systems likely will have black carbon dynamics very different from the pine savanna systems of the Southeastern United States. We have observed formation and storage of black carbon in the mineral soil horizons of a longleaf pine flatwoods site with an annual fire regime[1], and we expect this to significantly affect the net storage and turnover of carbon in these systems.

## Nitrogen

Nitrogen is frequently the limiting nutrient in forested ecosystems, and this element occurs in many different forms that can be influenced by fire. Nitrogen is an integral part of all biomass in ecosystems, and nitrogen concentrations in organic detritus (or necromass) are highly influential on the rate of detritus decomposition (Coleman and others 2004). Finally, the inorganic forms of nitrogen (nitrate or $NO_3^-$; and ammonium or $NH_4^+$), and the rate at which these forms are released from detritus or supplied by nitrogen-fixing plants and microbes usually has a profound influence on the overall fertility of a given soil volume.

Prescribed fire can have dramatic effects on nitrogen cycling, particularly when fires are frequent. One of the principal effects is the volatilization and loss of nitrogen from the organic horizons of soil. This is directly related to the intensity of the fire and the relative proportion of the organic horizon that is consumed. Also important is the temperature at which combustion occurs and the depth to which high temperatures penetrate the organic horizon. For example, in a laboratory study, Gray and Dighton (2006) found that the temperature at which different litter materials were burned had strong influence on the amount of nitrogen volatilized. Temperatures <400 °C resulted in 90 to 100 percent loss of nitrogen whereas temperatures from 100 to 200 °C retained ≥75 percent of the original nitrogen content. The long-term consequences of nitrogen loss can be significant, whether through chronic loss from frequent repeated fires or through a large loss from a single high-severity fire. For example, a site that had experienced a more severe site preparation fire (with relatively large proportions of organic horizons consumed), had lower tree seedling growth several years after the fires than did sites with less severe fires—an effect that was attributed to the loss of nitrogen capital from the system via volatilization (Elliott and others 2002).

In other aspects of nitrogen supply and cycling, however, prescribed fire has been demonstrated in many systems to have a positive effect. For example, nitrogen mineralization (microbial processing of organic nitrogen into plant available mineral forms) is either not affected by prescribed fire or is increased following prescribed fire (table 2). The net overall effect of prescribed fire on nitrogen dynamics in soil is most likely a function of fire frequency and intensity. Very frequent or very intense fires are likely to have negative effects on total nitrogen, but fires of intermediate frequency or lower intensity may increase nitrogen availability.

## Phosphorus

Phosphorus is often the second most limiting nutrient in forested ecosystems, and its availability is also influenced by prescribed fire. As a major component of ash, it should not be surprising that phosphorus would be affected by fire occurrence (table 3), but the chemistry of phosphorus in soils is highly complex and usually is strongly influenced

[1] Callaham, M.A., R.J. DiCosty, and J.J. O'Brien [N.d.]. Unpublished data. On file with the Center for Disturbance Science, U.S. Department of Agriculture Forest Service, 320 Green Street, Athens, GA 30602.

by the pH (acidity or basicity) of soil. Because phosphorus is chemically bound to aluminum (Al) and iron (Fe) oxides at low pH, and similarly, is bound to calcium at higher pH (Schlesinger 1993), the availability of phosphorus in ash is somewhat dependent upon the pH of the underlying soil. Further complicating the chemistry of phosphorus in relation to fire is the fact that the ash produced by the fire has other constituents that can change the pH of soil, at least in the short term. Thus, depending on the pH of soil before and after fire, the availability of phosphorus will be variably affected. In general, for pine dominated soils (and indeed for most forest soils in Eastern North America), the pH is typically in the range where phosphorus becomes chemically bound with iron and aluminum (5.7 and below), and the tendency for ash addition would be to temporarily increase the soil pH to a more favorable condition relative to phosphorus availability. However, such effects are usually short term (on the order of months to a few years) as the capacity of soil to buffer changes in pH is very large. Finally, it is notable that at very high temperatures (>770 °C approximately), phosphorus can be volatilized and lost from ecosystems (Neary and others 1999), and as such, fire intensity can be of great importance to overall phosphorus availability following prescribed fire.

### Other cations

In addition to the two macronutrients already discussed (nitrogen and phosphorus), several other essential nutrients may be affected by the incidence of prescribed fire in forested landscapes (table 3). The most widely studied of these are cations such as calcium, magnesium, potassium, and sodium. All these cations serve critical functions in various aspects of plant cell metabolism, and thus their availability for uptake can influence site productivity and even plant community composition to some extent. Because cations are typically not subject to volatilization, their availability generally goes up after a fire, when ash is deposited into the soil. Again, because biological demand for these cations is relatively high—plants, microbes, and animals all compete for them—the duration of fire-mediated spikes in availability is typically short and on the order of weeks to months.

## *Biological Effects of Prescribed Fire*

### Plant roots and fire

A large amount of information is available on responses of plants to fire in eastern forests (table 4). Effects range widely, from completely positive to completely negative, depending largely on the community of plants present in a forested landscape (fire tolerant species, fire sensitive species, or a mixture) and on the intensity of the fire (low intensity prescribed fire, high intensity wildfire, or something in between). Fire almost always results in the death of some plants in a given system, and the extent to which plants are killed has a strong relationship to the effects of fire on roots. The killing of fire sensitive plants aboveground results in an input of dead roots belowground—this input of new material has the potential to influence the decomposers (microbes) as well as the entire soil food web at least in the short term.

Another effect of prescribed fire on plant roots is a change in root distribution throughout the soil profile. In grasslands such as tallgrass prairie, annual fire causes roots to be distributed more deeply throughout the soil profile (Kitchen and others in press). In forested ecosystems, data on root distribution responses to fire is scarce, but evidence from longleaf pine systems suggests that frequent prescribed fire has similar effects on fine root distribution in mineral soil. In longleaf pine systems where fire is excluded for the long term, fine roots proliferate in the organic horizons of the soil; but in frequently burned sites, the organic horizons are much reduced or eliminated completely, and thus fine root biomass is increased in mineral soil horizons (O'Brien and others 2010). The degree to which prescribed fire affects root distribution in other eastern ecosystems has not been extensively studied.

**Table 3.** Effects of prescribed fire or treatments including prescribed fire (such as thinning or herbicide application) on acidity or basicity (pH), phosphorus (P), potassium (K), calcium (Ca), magnesium (Mg), and sodium (Na) in the soils of Eastern North America

| Fire management objective | Ecosystem | Location (State/ Province) | Ecological division[a] | pH | P | K | Ca | Mg | Na | Reference |
|---|---|---|---|---|---|---|---|---|---|---|
| Fuel reduction | Longleaf pine | LA | 230 | n/a | –/= | = | = | n/a | n/a | Haywood 2007 |
| | Pine barrens | NJ | 230 | n/a | + | + | + | + | n/a | Gray and Dighton 2006 |
| | Prairie | AR | 230 | = | – | = | = | = | - | Brye 2006 |
| Habitat improvement | Longleaf pine | GA | 230 | n/a | = | n/a | n/a | n/a | n/a | Boring and others 2004 |
| | Outwash sandplain | MA | 220 | +/= | n/a | – | = | =/– | - | Neill and others 2007 |
| Restoration | Grassland | MD | 230 | + | = | = | = | = | = | Sherman and others 2005 |
| | Oak savanna | MO | 220 | + | + | + | + | = | n/a | Rhoades and others 2004 |
| | Oak-hickory | OH | 220 | n/a | – | n/a | n/a | n/a | n/a | Huang and Boerner 2007 |
| | Oak-pine | MA | 220 | + | n/a | = | = | = | n/a | Neill and others 2007a |
| | Pine-bluestem | AR | M230 | + | n/a | = | + | = | n/a | Leichty and others 2005 |
| | Oak-hickory - grass barrens | KY | 220 | = | =/+ | = | = | = | n/a | Rhoades and others 2002 |
| Site preparation | White pine | NC | M220 | + | n/a | + | + | n/a | n/a | Knoepp and others 2004 |
| | Black spruce | NF | 210 | + | +/+ | –/= | +/= | + | n/a | Scheuner and others 2004 |

n/a indicates that no data is available.
+ indicates a statistically significant increase in the measured variable.
= indicates no statistically significant change in the variable.
– indicates a statistically significant decrease in the variable.
Note: When two symbols are presented for a given variable, the first represents a shallower soil depth (typically organic horizon soil), and the second represents a deeper soil.
[a] 210 is Warm Continental, 220 is Hot Continental, M220 is Hot Continental Mountains, 230 is Subtropical, M230 is Subtropical Mountains (Cleland and others 2007).

**Table 4.** Biological effects of prescribed fire or treatments including prescribed fire (such as thinning or herbicide application) on soils in eastern North America

| Fire management objective | Ecosystem | Location (State/ Province | Ecological division[a] | Enzyme activity[b] | Soil respiration | Plant responses | Organism responses | Reference |
|---|---|---|---|---|---|---|---|---|
| Fuel reduction | Oak-hickory | OH | 220 | AcPh =; PhOx + in thinned, = in second site | n/a | n/a | Bacterial activity + in burn and thin/burn | Giai and Boerner 2007 |
| | Longleaf pine | LA | 230 | n/a | n/a | Pine leaf had lower nitrogen at one site and higher phosphorus at both sites | n/a | Haywood 2007 |
| | Longleaf pine | LA | 230 | n/a | n/a | Root variables were generally not affected by fire; root length was shorter with summer burn | n/a | Sword-Sayer and Haywood 2006 |
| | Red pine | MI | 210 | n/a | n/a | Fine root production was not affected by fire | n/a | Zeleznik and Dickmann 2004 |
| | Loblolly-longleaf pine | GA | 230 | n/a | n/a | Root biomass was not affected by fire | Microbial N fixation + in clay soil but – in sandy soil | Lajeunesse and others 2006 |
| | Loblolly pine plantation | SC | 230 | AcPh + after 4 years in thin/ burn sites; PhOx + after 4 years in thin/ burn sites; Chit – after 4 years in burn only sites | n/a | n/a | n/a | Boerner and others 2006 |
| Habitat improvement | Outwash sandplain | MA | 220 | n/a | = | n/a | n/a | Neill and others 2007a |
| Restoration | Oak-hickory-grass | KY | 220 | n/a | – | n/a | n/a | Rhoades and others 2002 |
| | Oak-hickory | OH | 220 | AcPh +; PhOx +; Chit +; a-Gluc =; L-Glut = | n/a | n/a | n/a | Boerner and others 2005 |

*continued*

**Table 4.** Biological effects of prescribed fire or treatments including prescribed fire (such as thinning or herbicide application) on soils in eastern North America (continued)

| Fire management objective | Ecosystem | Location (State/ Province | Ecological division[a] | Enzyme activity[b] | Soil respiration | Plant responses | Organism responses | Reference |
|---|---|---|---|---|---|---|---|---|
| Restoration (continued) | Oak-hickory | OH | 220 | n/a | n/a | Nitrogen release from decaying roots was slightly higher in burned sites initially, but remained unchanged after one year; fire had no effect on live root nitrogen | n/a | Dress and Boerner 2003 |
| | Oak-hickory | OH | 220 | n/a | n/a | n/a | Oribatid – with annual fire, but = with less frequent fires | Dress and Boerner 2004 |
| | Oak-hickory | OH | 220 | n/a | n/a | Root nitrogen was reduced in one site with fire but made no difference in the other; root phosphorus was not affected by fire | n/a | Huang and Boerner 2007 |
| | Oak-hickory | OH | 220 | n/a | = | n/a | n/a | McCarthy and Brown 2006 |
| | Oak | IL | 220 | n/a | n/a | n/a | Total epigeic springtails density = with fire, species richness – with annual fire | Brand 2002 |
| | Oak-pine | TN/GA | 220, 230 | n/a | – short term, = thereafter | n/a | n/a | Hubbard and others 2004 |
| | Oak-pine | KY | 220 | n/a | n/a | n/a | Litter dwelling arthropods were – with fire with no recovery after 2 years; ground dwelling arthropods = with fire; grasshoppers + in burned plots | Coleman and Rieske 2006 |

*continued*

**Table 4.** Biological effects of prescribed fire or treatments including prescribed fire (such as thinning or herbicide application) on soils in eastern North America (continued)

| Fire management objective | Ecosystem | Location (State/ Province | Ecological division[a] | Enzyme activity[b] | Soil respiration | Plant responses | Organism responses | Reference |
|---|---|---|---|---|---|---|---|---|
| Site preparation | Jack pine | ON | 210 | n/a | n/a | n/a | Microbial community diversity – 5 years after burn in a clearcut compared to unburned clearcut at the whole plot level; microbial diversity = at finer scales. | Staddon and others 1998a |
| | Jack pine | ON | 210 | AcPh in burned plots not different from clearcut and unburned, but lower than untreated stands; ArSu lower in burned plots relative to others | n/a | n/a | n/a | Staddon and others 1998b |
| | Red pine | ON | 210 | n/a | n/a | Experimental removal of organic horizon soils resulted in better emergence of pine seedlings, but addition of ash diminished this response | n/a | Herr and Duchesne 1996 |

n/a indicates that no data is available.
+ indicates a statistically significant increase in the measured variable.
= indicates no statistically significant change in the variable.
– indicates a statistically significant decrease in the variable.
[a] 210 is Warm Continental, 220 is Hot Continental, M220 is Hot Continental Mountains, 230 is Subtropical, M230 is Subtropical Mountains (Cleland and others 2007).
[b] AcPh is acid phosphatase, PhOx is phenol oxidase, Chit is chitinase, a-Gluc is a-glucosidae, L-Glut is L-glutaminase, and ArSu is aryl sulfatase.

## Soil microbes and fire

The effects of fire on soil microbes in eastern forests seems to depend to a large extent on the intensity of the fire. Joergensen and Hodges (1970) and Renbuss and others (1973) found that the responses of soil microbes to fires range from no detectable effect (low intensity prescribed fires) to total sterilization of the surface layers of soil (very hot wildfires). This early work focused primarily on the abundance of microorganisms and not their activity levels. However, others have observed that although there may be a decrease in abundance of microbes following fire, the remaining microbes can have higher activity levels than that of the preburn community (Poth and others 1995). These authors, working in tropical savanna systems in Brazil, found that the increased rates of microbial processes, such as denitrification and production of methane and carbon dioxide, persisted for a year following fire. The nature and duration of microbial responses to fires in eastern forests are not well known. In one study examining soil carbon dioxide efflux (the combined production of carbon dioxide from plant root respiration and microbial and soil animal respiration) in loblolly pine stands of the South Carolina Piedmont, Callaham and others (2004) observed that soil respiration (one indicator of microbial activity) decreased in plots that had been burned or had been thinned and burned—a response attributed to warmer soils in these two treatments, along with increased inputs of belowground detritus in the form of dead plant roots.

Most of the more recent work on soil microbes and their responses to fire has made use of new techniques designed to facilitate examination of the diversity or functional capacity of the microbial community. The most frequently used approaches are the enzyme-based assay of microbial activity, which uses the actual concentrations of ecologically important enzymes in soils to make inferences about the makeup and function of the microbial community at the time of sampling; and the community carbon utilization profile, which uses an array of different carbon sources to evaluate the potential metabolic capacity of the microbial community from the sampled soils.

- The carbon utilization profiles give an estimate of microbial-community function diversity; if the microbes from a site can use more of the different carbon sources in the assay, then that community is considered functionally more diverse.
- Changes in the concentrations of enzymes in soil can be attributed to changes in the relative importance of various functional groups of microbes. Of the many such enzymes present in soil, only a few are particularly well characterized and have standardized methods of measurement (Tabatabai 1982): acid phosphatase (indicative of total microbial biomass, and phosphorus mineralizing organisms), phenol oxidase (indicative of white rot fungal biomass), chitinase (indicative of bacterial decomposition of more recalcitrant organic matter), aryl-sulfatase (indicative of microbes processing sulfur containing organic matter), a-glucosidase (indicative of fungal metabolism of cellulose and hemicellulose), and L-glutaminase (indicative of microbes involved in metabolism that results in nitrogen mineralization). Results from enzyme assays in studies comparing burned to unburned soils seem to indicate shifts in the microbial community towards a community that is geared toward metabolizing more recalcitrant materials, but these results are somewhat site dependent and responses differ in terms of duration after fire (table 4).

## Microinvertebrates and fire

In one of the few studies dealing with microinvertebrate responses to fire in eastern forests, Metz and Farrier (1971) reported a general reduction of microarthropods (mainly springtails and mites) with increasing prescribed fire frequency in loblolly pine stands on the Coastal Plain of South Carolina. In this study, the authors compared the abundance of microarthropods in plots that had been burned every year, burned every 3 to 4 years, or left unburned for many years. They found that abundances of mites and springtails were reduced a small amount (~25 percent) by periodic prescribed fires, but that the reduction was dramatic (75 to 80 percent) with annual fires. Similar studies in midwestern Hot Continental Division forests showed similar results in that reduction of litter mass with prescribed fire generally reduced microarthropod numbers (Brand 2002, Dress and Boerner 2004). The consequences of these reductions for the decomposition of new leaf litter have not been thoroughly addressed.

The response of microarthropods to fire has also been studied in many other systems including eastern Prairie Division grasslands such as the tallgrass prairie systems in eastern Kansas and Oklahoma; in these studies, microarthropods are decreased in abundance with frequent fire (Seastedt 1984). This negative effect of fire is mostly attributed to decreased habitat for mites and springtails, because many of these organisms live in decomposing leaf litter, much of which is lost in fires.

### Macroinvertebrates and fire

The few scientists who have studied the responses of soil invertebrates to fire in forested ecosystems of the Eastern United States found that response is often driven by changes in habitat structure or by changes in the amount or the quality of food resources (Coleman and Rieske 2006). Thus, whenever fire affects vegetation, temperature, moisture, or the nutrient status of a soil, the potential exist for impact on the soil invertebrate community. These impacts are not always predictable, as demonstrated by a study of ground and litter dwelling arthropods conducted by Hanula and Wade (2003). They found that the frequency of prescribed fires (plots burned annually, every 2 years, every 4 years, or unburned for 40 years) in longleaf pine flatwoods of northern Florida had dramatic effects on numerous organisms. Interestingly, most of the arthropod groups collected during the 5-year study had negative responses to fire, but some groups were favored by fire. For example, among 28 different spider groups that were collected, four responded positively to the frequent fires employed in the study.

Another study of litter dwelling and soil dwelling macroinvertebrates showed that the density of macroinvertebrates was significantly reduced a year after a prescribed fire in the upland forests of the Cumberland Plateau in Kentucky (Kalisz and Powell 2000). Reduction in the number of beetle larvae accounted for a large proportion of the difference following fire, and the authors proposed that repeated fire in a single location could potentially have long-term negative effects populations and on the functions these beetles perform within the system.

Several studies on the responses of soil macroinvertebrates to fire have been conducted in Prairie Division grassland soils of eastern Kansas. Studies have repeatedly shown that earthworms are strongly affected by fire in tallgrass prairie soils, and the usual pattern observed is for fire to increase their abundance (James 1982). Interestingly, in areas close to human habitations (with nonnative earthworms present), prescribed fire had the effect of limiting the colonization into soils under frequently burned vegetation (Callaham and others 2003). Results of this study suggest that native earthworms in grassland soils are adapted to the warmer soil conditions often found under frequently burned vegetation; also that because fire improves the performance of grasses, native earthworms may have strong preferences for soils with abundant grass roots. This effect of fire on nonnative earthworms may have potential application as a control strategy in eastern forests where invasions of European or Asian earthworms are currently underway—this idea is in need of further research.

## Mechanical Fuel Treatment Effects on Eastern Soils

Mechanical fuels treatments have the potential to alter soil properties and processes dramatically; but under many conditions they may have little to no impact on soils. These treatments affect soils by using heavy equipment, which may change physical and hydrological processes, and by cutting and removing vegetation and site organic matter (fuels), which changes soil fertility and soil chemical and biological processes (Powers and others 1990). Mechanical treatments can vary from single-entry understory mowing or mulching treatments with small tractors to multiple-entry whole-tree thinning and harvesting followed by harvest residue raking and piling (table 5). In addition,

**Table 5.** Mechanical fuels treatment practices and their relative potential for soil impacts in the Eastern United States

| Practice | How used | Mechanism | Modifiers |
|---|---|---|---|
| Mulching, mowing, chopping, crushing | Precommercial thinning, reduction of ladder fuels, site preparation | Equipment traffic | Number of passes, soil type, and conditions |
| Commercial bole harvest | Ladder fuel reduction, stand development (thinning), salvage/sanitation cuts, regeneration cuts | Equipment traffic, low-nutrient product removal | Degree of harvest; tree age, species, soil type, and conditions |
| Intensive harvest | Same as above, plus: understory fuel reduction and biofuel production | Equipment traffic +[a], high-nutrient product removal | Degree of harvest, tree age, species, season of harvest, soil type, and conditions |
| Harvest residue removal | Prepare site for regeneration, esp. planting | Equipment traffic +[a], high-nutrient product removal, soil displacement | Degree and method of removal, soil type, and conditions |

[a] + refers to the generally greater number of passes with intensive harvest and site preparation as well as a reduced amount of debris upon which equipment can be driven, which increases the potential for physical property change.

mechanical treatments are applied under stand and soil conditions that are both resistant and resilient to impact, or they can be applied in conditions that provide little resistance to soil disturbance or nutrient removal and few mechanisms for recovery. No mechanical treatment is without the potential for impacting soil function, but conditions do exist under which any mechanical treatment can be used effectively without degrading essential soil functions such as supply of adequate rooting medium, water and nutrient supply to plants, and water infiltration (without excessive runoff or erosion).

In intensive production forestry, soil quality is restored or even improved after soil disturbance if other practices, such as soil tillage and fertilization, are used (Fox 2000). These practices are feasible because they ameliorate damages and usually increase production. In extensive forest management systems practiced by families and other nonindustrial owners, especially those for whom timber yield is not the primary goal, the focus is to minimize negative disturbances impacting soil productivity and rely on natural recovery processes and inherent site productivity (Grigal 2000). Therefore, a complete understanding of how mechanical treatments affect soil properties and processes is necessary to avoid degrading soil quality to the extent that natural processes cannot restore it.

Much of the basic knowledge we have regarding mechanical treatments and soil impacts was developed quite some time ago, and most of the important foundational principles that describe how mechanical treatments impact soil were developed in agricultural and forestry systems. Unfortunately, the potential set of conditions to which the principles apply is virtually infinite, and it is only through continued, site-specific research that we will be able to better understand how to minimize negative impacts. Therefore, we will only briefly review the basic concepts and widely accepted principles of soil disturbance effects and concentrate on describing the most current evidence available from studies on eastern forests.

## Effects on Physical Properties and Processes

Mechanical treatments have the potential to cause changes to soil physical properties and processes (Greacen and Sands 1980, Lull 1959, Miwa and others 2004), and these changes have been linked to reductions in germination (Pomeroy 1949), establishment and early survival of seedlings (Bates and others 1993, Brais 2001, Foil and Ralston

1967, Hatchell and others 1970, Lockaby and Vidrine 1984, Scheerer and others 1994), sprouting or suckering success (Smidt and Blinn 2002, Stone 2002, Stone and Elioff 2000, Zenner and others 2007), seedling root growth (Jordan and others 2003, Mitchell and others 1982, Siegel-Issem and others 2005, Simmons and Ezell 1982, Simmons and Pope 1985, Tworkoski and others 1983), seedling shoot growth (Farrish and others 1995, Hatchell 1981, Lockaby and Vidrine 1984), and growth of remaining trees (Moehring and Rawls 1970). However, soil disturbance and damage during mechanical operations is not a given (King and Haines 1979), and many studies have shown that soil physical disturbances do not necessarily lead to reduced tree survival or growth (Carter and others 2002, Reisinger and others 1993, Sanchez and others 2006, Scott and others 2007, Tiarks 1990, Xu and others 2000). Although the overwhelming majority of research on soil physical disturbance in eastern forests has been conducted in the pine forests of the Southern States or in the aspen forests of the North Central States, the general relationships hold for most forest types. Unfortunately, general relationships are often not useful in determining the impact across different site types or for particular soil functions within a given site type.

Several classification systems have been created to define soil disturbances. Most of these systems describe various degrees of harvesting, forest floor removal, and mineral soil disturbance; and all have evolved from those defined by Dyrness (1965) for Pacific northwestern forests. Miller and Sirois (1986) and Aust and others (1998) developed classification systems in the South, and Steber and others (2007) recently used a nationally based system to evaluate disturbance in the Great Lakes States. These disturbance classification systems are used widely for two reasons: first, they provide an easy and rapid assessment of forest sites; and second, unlike chemical or biological changes, soil physical disturbances have a clear and usually negative visual impact. Although visually based classification systems are useful for rapidly assessing and monitoring impacted areas, they are not generally effective at discerning quantitative changes in soil properties or processes (Aust and others 1998, Steber and others 2007). However, these systems are quite useful in determining the spatial extent of disturbance, which is an important component to determining actual site disturbance.

Soil physical disturbances have generally been classified as compaction, rutting, and puddling or churning. Compaction occurs whenever the load applied to a soil is greater than its strength, resulting in an increase in bulk density and a reduction in porosity. Mechanical traffic causes compaction when the soil contains enough water to reduce friction between soil particles—and thus reduce soil strength—but not enough to cause soil flow. Puddling occurs when the soil is wet enough to flow, traffic causes rutting, and repeated tire slippage smears pores and destroys soil structure (Miwa and others 2004).

Bulk density is the most common method of quantitatively describing disturbance. Other properties and processes commonly affected by soil physical disturbance include soil strength (for example, resistance to penetration by roots), porosity and the distribution of pore sizes or quantity of air- or water-filled pores, hydraulic conductivity, and infiltration rate. Comparing bulk density among different soils is prone to imprecise interpretation because the bulk density at which root growth is limited depends on soil texture (Daddow and Warrington 1983). In general, the more coarse textured (sandy) a soil is, the higher its bulk density; and the more fine textured (clayey) a soil is, the lower its bulk density. Organic soils or topsoils with high organic matter content generally have the lowest bulk density. Within a given soil, comparing one bulk-density value to another is can also be misleading. A large absolute increase in bulk density from a relatively low value to a moderate value will have little effect on the properties that actually influence root growth—soil strength, aeration porosity, and water availability. Conversely, a small absolute increase in bulk density from an already elevated value to an even higher value will likely constitute soil damage. For example, an absolute increase in a loam bulk density from 1.2 to 1.4 mg/m$^2$ (0.2 mg/m$^2$ or 17 percent) is larger, both in absolute and relative terms, than an increase from 1.4 to 1.5 (0.1 or 7 percent). Under current U.S. Department of Agriculture Forest Service standards, a 17-percent increase in bulk density constitutes a significant impairment while the

7-percent increase does not, even though the increase from 1.4 to 1.5 would likely create much more growth-limiting conditions. Thus, change in bulk density is only useful given the initial or undisturbed value. For this reason, other parameters are better indicators of soil function.

The interactions among soil strength, aeration porosity, and water availability have been illustrated by Letey (1985) and have been updated by Da Silva and others (1994) with the creation of a single parameter, the least limiting water range. This parameter has been used successfully to explain loblolly pine response to soil physical disturbance (Kelting and others 2000), and although laborious and data intensive, could be used to monitor effects of soil physical disturbance on plant growth. Compaction increases soil strength, which becomes limiting to root growth at around 204 t/m$^2$ of pressure (Taylor and others 1966), although this value is species specific. Rutting and churning tend to decrease macroporosity and hydraulic conductivity substantially, and soils with <10 percent aeration porosity are not supportive of root growth. Similarly, reductions in hydraulic conductivity can alter the surface hydrology of sites, causing shifts in a host of physical and chemical processes. Because soil type determines which of these particular properties may have greater influence on tree response, Aust and others (1998) suggested that soil strength is the best indicator of damage on dry to moist soils, the decrease in aeration porosity <10 percent is the best indicator of site damage on seasonally saturated soils, and the reduction of hydraulic capacity is the best indicator on frequently saturated soils.

Tree response to soil disturbance is not always a good indicator of soil function, because responses are subject to other factors, such as competing vegetation (Brais 2001). For example, compaction reduced understory competition on the Mississippi long-term soil productivity study sites, which have moderately well drained silt loam soils (Aquic Paleudalfs). One of the treatments was soil compaction at three levels: none, moderate, and severe. The moderate and severe compaction levels were induced by pulling a weighted wobble-wheel road compactor across the plot six times to achieve uniform compaction. The treatments were effective with soil bulk density of 1.3 in the uncompacted plots and 1.4 in the compacted plots (Scott and others 2004). Planted pine biomass after five growing seasons was 5.9 mg/ha for no compacting, 7.2 mg/ha for moderate compacting, and 7.1 mg/ha for severe compacting (Stagg and Scott 2006). Competing understory biomass was 5.6, 2.0, and 1.8 mg/ha on the same plots, and these differences were statistically significant. Total biomass, however, was not significantly different among the compaction treatments. Furthermore, although most understory species were affected similarly, some species, such as flowering dogwood (*Cornus florida*) and some oaks were virtually eliminated from the compacted plots, presumably to the result of greater sensitivity to either increased soil strength or decreased aeration. These findings all underscore the fact that although dominant tree survival and growth is the easiest and most common bioassay of soil disturbance, all plants have individual responses to soil properties and processes (Burns and Honkala 1990); whereas one plant may not respond negatively to a given change in soil properties or processes, others may be negatively impacted.

In rare circumstances, soil disturbance can create soil conditions that are actually more conducive to tree growth. If a site is characterized by coarse-textured or very loosely packed soils, water-holding capacity is often the soil property that influences tree growth. On these soil types, compaction can increase micropores by reducing the size of macropores; and even though overall aeration may decrease, water-holding capacity can be increased. This has been shown most definitively by Gomez and others (2002) in ponderosa pine (*P. ponderosa*) forests in California, but the phenomenon has been described in eastern forests as well (Brais 2001). Clearly, this phenomenon is very site specific, and careful planning and site evaluations should precede any management prescriptions that involve soil compaction.

Compaction and other physical soil disturbances may impact soil functions other than tree growth. Surface compaction reduces infiltration, which increases runoff and the potential for erosion. However, mechanical treatments rarely cause erosion and sediment transport except on areas where the forest floor is removed, such as on main skid trails and roads. Although mechanical on treatments Eastern U.S. sites increased disturbance and water yield, measurable increases in sediment and nutrients are slight, especially where best management practices are employed to limit the amount of bare soil created (Aust and Blinn 2004). Similarly, rutting can obstruct surface drainage, and rutting and churning can impede drainage by reducing hydraulic capacity. Better drained soils more impaired by these treatments than inherently poorly drained soils (Aust and others 1995).

## *Effects on Chemical Properties and Processes*

Organic matter disruption or removal affects a number of soil properties and cycling processes. The most direct impact of forest fuel removal is direct removal of carbon and nutrients from the forested site. The factors that govern the cumulative removal of carbon and nutrients from a site include the frequency of removals, the intensity of harvest or removal at each entry, the species and age of the plants being removed, and even the season of year. In general, multiple entries over a rotation or an equivalent length of time—such as with frequent selection-cutting cycles or multiple thinnings—remove more nutrients and organic matter than single-entry harvests (even to include clearcuts) over the same length of time; and thus, harvest intensity is clearly a determinant of nutrient removal (Freedman 1981). Leaves, branches, and bark represent about 70 percent of the aboveground nutrients held in mature trees, and these materials represent an even greater percentage in smaller trees (Mann and others 1998). Younger plants generally have much higher nutrient concentrations than older plants. Finally, the season of the year controls the quantity of nutrients held in the foliage. For example, newly flushed leaves in the spring have greater overall nutrient content compared to senescent leaves in the autumn, which lose nutrient content as trees translocate nutrients to belowground storage pools. Additionally, even in conifers the total amount of foliage in tree crowns varies by season (peaking summer and lowest in winter). Although these factors are known to control plant growth and other soil functions, some uncertainty remains as to the conditions under which removal of these materials may degrade soil function.

Concerns over harvesting and nutrient removal in eastern forests began in the early 1970s as a result of the work by Bormann and Likens (1968), who showed increased nutrient loss following clearcut harvesting; and Keeves (1966), who documented losses in productivity in the second rotation of pines on nutrient-deficient Australian soils. Interest increased dramatically in the late 1970s during the energy crisis when whole-tree harvesting (clearcut harvesting of entire trees) was first being considered to provide biomass for energy. The result was a number of experiments across the Eastern United States that were designed to determine the potential nutrient loss from harvesting and other mechanical treatments.

The general nature of these nutrient loss experiments was regional because of differences in the management systems that were in place at the time. In the North Central States (Warm Continental Division) concerns generally focused on the effects of whole-tree harvesting on soil fertility and subsequent growth, whereas studies in the South were mostly focused on harvesting and effects of subsequent site preparation practices on soil nutrient availability and pine growth. In the Warm Continental Mountains (M210) Division of northeastern landscapes and in the less intensively managed southern forests in the Hot Continental Mountains Division, studies have focused on direct effects of whole-tree harvesting removals as well as the potential for increased leaching losses following the harvest. Finally, many of the northeastern studies also examined the interactive processes related to harvest-caused losses and the losses and gains associated with acid precipitation. To further address these issues in a systematic way, a long term soil productivity program was installed in the 1990s in a variety of locations across southern and north central landscapes to examine both harvest intensity and forest floor removal.

Harvesting, especially whole-tree harvesting, removes large quantities of nutrients from a site (Freedman 1981, Kimmins and others 1985, Powers and others 2005). Recent reviews of long-term soil carbon and nitrogen responses to harvesting have shown little evidence that harvesting, even whole-tree harvesting, reduces soil carbon and nitrogen (Johnson and Curtis 2001, Johnson and others 2002, Knoepp and Swank 1997). These reviews were mostly centered in eastern forests; Knoepp and Swank (1997) reviewed harvesting studies in five watersheds in the Southern Appalachians, Johnson and Curtis (2001) did a worldwide metanalysis of 26 studies (of which 11 were from the Eastern United States), and Johnson and others (2002) resampled five long-term studies in a variety of southeastern ecosystems. Evidence from long-term soil productivity studies (Powers and others 2005) indicate only slight decreases in soil carbon through 5 years since harvesting in Louisiana, no decreases in North Carolina (Laiho and others 2003), and no general decreases at 5 or 10 years post harvest across 21 installations (including the North Carolina, Louisiana, and Mississippi locations).

While much of the initial concern over harvesting-induced deficiencies dealt with carbon and nitrogen, later studies became concerned with other nutrients, such as calcium, magnesium, potassium, and phosphorus depletion, especially in northeastern forests where acid precipitation promotes additional calcium and magnesium losses. Federer and others (1989) reviewed the literature on losses of these nutrients in response to harvesting across the Eastern United States and found that total soil magnesium, potassium, and phosphorus may decrease only by 2 to 10 percent in 120 years, depending on site and harvest intensity; and total calcium losses from leaching and harvest removals could amount to 20 to 60 percent. Huntington (2000) further reviewed the evidence from several southeastern studies and found that harvesting and leaching losses are likely to be in excess of weathering-induced additions to supply and cautioned that this could have a widespread (>50 percent of forested area) impact on productivity. Yanai and others (2005) showed that apatite, a calcium-bearing mineral found in soils with granitic parent materials, is capable of maintaining soil calcium on many sites previously thought to be sensitive to depletion, but noted that soils with sedimentary parent materials may not have adequate supply rates of calcium to maintain current levels of productivity.

Harvesting-induced phosphorus removals have also been linked to reduced availability of phosphorus and growth declines. Yanai (1998) showed that whole-tree harvesting doubled the phosphorus removed compared to a similar bole-only harvested site and that harvesting reduced soil phosphorus net mineralization by 40 to 70 percent compared to an unharvested control. Scott and others (2004) compared whole-tree harvesting and whole-tree harvesting followed by forest floor removal to bole-only harvesting on Louisiana and Texas long-term soil productivity locations, and found that the former reduced extractable phosphorus compared to the latter by 23 percent; on North Carolina or Mississippi long-term sites, whole-tree harvesting and whole-tree harvesting followed by forest floor removal had no effect on extractable phosphorus. Scott and Dean (2006) and Scott and others (2007) linked loblolly pine productivity declines caused by whole-tree harvesting (compared to bole-only harvesting) to the preharvest quantity of extractable phosphorus in Louisiana, Mississippi, and Texas.

In addition to nutrients removed in harvested material, traffic and site preparation actions, such as windrowing and root raking, can cause forest floor removal. Forest floor displacement has been conclusively linked to nutrient loss and productivity declines (Conde and others 1986, Fox and others 1989, Gaskin and others 1989, Morris and others 1983, Pye and Vitousek 1985, Riekirk and others 1981, Stone and others 1999, Tew and others 1986), and is the primary cause of erosion and sediment losses from skid trails and landings in managed forests (Aust and Blinn 2004).

## *Effects on Biological Properties and Processes*

Mechanical treatments affect soil biological functions both through physical disturbances to soil properties and processes and through impacts to organic matter and chemistry, but responses are quite variable. Because of this variability and complexity, few generalized statements can be made about the relationship between mechanical treatments and biological processes and properties.

Biological activity—commonly measured through carbon dioxide evolution, nitrogen mineralization, or enzyme assays—is usually more indirectly (than directly) affected by mechanical treatments. Biological activity depends on both substrate and aboveground environment, both of which are altered by mechanical treatments, as discussed above. Reducing forest cover warms soils, which to a point will increase biological activity. Reduced evapotranspiration increases soil water content, which generally increases activity. If sites become waterlogged or if aeration is reduced by mechanical treatments, activity decreases. These basic processes have been described in many ecosystems and forests and a detailed review is beyond the scope of this chapter.

In general, mechanical treatments have produced biological responses in places where the affected organisms specifically use the forest floor as habitat or are particularly sensitive to soil climatic conditions, such as reduced aeration and soil temperature. On the Missouri shortleaf pine-oak, long-term soil productivity sites, earthworm activity was reduced by compaction but unaffected by forest floor removal. Forest floor removal had little impact on earthworm abundance or biomass, but compaction reduced the density of *Diplocardia ornate*, which is about 5 mm in diameter, while the density increased for Oligochaetes (*D. smithii*), which is about 2 mm in diameter (Jordan and others 1999).

Microbial communities varied little in functional diversity with compaction or forest floor removal in Subtropical Division loblolly pine-dominated sites of Louisiana and North Carolina (Busse and others 2006). Li and others (2004) found that microbial biomass and diversity varied more on two similar soil series (two adjacent series) within a single research site than in response to compaction and forest floor removal.

Biological activity is clearly affected by soil disturbances caused by mechanical treatments, but responses are not consistent across treatments or soil types. Compaction reduced microbial biomass nitrogen in a Subtropical Division pine site (Li and others 2003), but changes in soil climate did not affect nitrogen mineralization. Neither compaction nor intensive harvesting affected soil carbon dioxide efflux on temperate hardwood sites in Missouri (Ponder 2005) 4 years after treatment, nor did intensive harvesting have an effect on a Subtropical Division pine site at 10 years after harvest (Butnor and others 2006). Although nitrogen mineralization was lower two and five years after compaction in North Carolina pine stands (Subtropical Division), harvest intensity had no effect on nitrogen mineralization; and the within-site differences in soil water content on the two soil types in the stands caused the greater differences in nitrogen mineralization than any treatment (Li and others 2003), similar to findings for microbial biomass and diversity discussed earlier.

# Conclusions

One overarching conclusion that must be drawn from this review of soil responses to fuel management strategies in the Eastern United States is that the responses (chemical, physical, or biological) can be extremely context dependent. In other words, depending upon the conditions under which prescribed fires or mechanical fuel treatments are conducted, the impacts on soils can be quite variable. Generally speaking, the more intense the physical disturbance (heating or consumption of forest floor for prescribed fire, or compaction or erosion in mechanical operations), the more profound and long-lived the damage to soils. Managers who take soils into special consideration when planning fuel management activities will minimize these intense perturbations. The research summarized here provides a reasonable reference point for these considerations, but we have also identified several limitations to our knowledge, and we suggest that more research on the effects of fuel management on soils would be useful.

Most of the studies cited in this review were conducted at the small plot or stand scale, and therefore do not provide much insight into watershed-level effects, or cumulative effects to the watershed. Detailed, spatially explicit modeling exercises will be needed to derive estimates of how fuel treatments likely to affect whole watersheds. Any models developed to assess whole watershed-level effects of fuel treatments on soils will likely be parameterized with the plot level data from the studies summarized in this review. Because such a modeling effort has yet to be undertaken, this represents one major avenue for future research.

Degradation of soil aggregate structure as a potential physical effect of prescribed fires has been hypothesized for oak savanna ecosystems of Missouri, but this phenomenon has yet to be directly measured (Rhoades and others 2004). These authors suggest that destruction of aggregate structure might partially explain observations of slow plant community recovery in soils where logs had "burned out" in a prescribed fire. Such aggregate destruction may be related to the changes in soil texture—more work on the dynamics of soil aggregate formation and stability will be needed to fully evaluate the effects of prescribed fires on soils in eastern North America.

The response of roots to fire in eastern forests is an area needing much future research. Root work is tedious and time-consuming, but the potential effects of fires on root dynamics and the attendant effects on landscape-scale carbon sequestration make this a critical issue for researchers and forest managers to understand.

Central questions as outlined in Czimczik and Masiello (2007) surrounding the behavior and processing of "black" carbon in frequently burned soils constitute another area where a good deal of research remains to be conducted. Major areas of uncertainty include questions about how this material varies in chemical composition when formed under different combustion conditions, how it moves into the soil profile (bioturbation or water infiltration), how it influences water quality, whether it enters the dissolved fraction of suspended organic carbon, whether microbial communities evolve to process it, and whether the its particle size affects any or all of the above processes. Overall, this and other aspects of how prescribed fire influences the carbon balance of forested ecosystems in the Eastern United States would benefit from a much more detailed accounting than is currently available.

Although soil biota, both macroarthropods and microarthropods, have been demonstrated to have substantial effects on soil processes in eastern agricultural (and some forested) ecosystems, their responses to fuel management practices are not well known. More work examining the responses of the soil invertebrate community to prescribed fire and mechanical fuel treatments would improve understanding of how these activities influence the functioning of soils.

Nearly all of the soil responses to fuel treatments discussed in this chapter have some temporal dimension that is extremely difficult to evaluate in short-term studies. Further complications arise from the fact that different soil functional responses to fire (for example, nutrient mineralization rate versus loss or accrual of soil organic matter) will take different amounts of time to manifest themselves. In other words, some responses of soil ecosystems may be clear in a year or two following fire, but others may take decades to reach equilibrium. Scientists from the Forest Service and partner research organizations maintain long-term studies including soils-based studies, such as those on experimental forests and co-located long-term ecological research sites (established with National Science Foundation funding) as well as the long-term soil productivity plots described above. Such long-term experimentation will be critical to guiding the management of natural resources (including soil) in the future. The resulting data will be of great value when models are developed to fully address these issues at the landscape scale (Richter and others 2007).

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Cumulative Watershed Effects of Fuel Management in the Eastern United States

CHAPTER 11.

# Water Yield and Hydrology

**Pamela J. Edwards, Charles A. Troendle**

Investigations of hydrologic responses resulting from reducing vegetation density are fairly common throughout the Eastern United States. Although most studies have focused on the potential for increasing water yields or documenting effects from intensive practices that far exceed what would be done for fuel-reduction objectives, data from some less-intensive manipulations—such as thinnings, understory removals, and controlled burns for seedbed establishment—that are more easily related to fuel-reduction activities are available. In this chapter, findings from the entire range of available manipulation intensities are presented so that results can be applied to various levels of fuel reductions. Even though site preparation is a silvicultural technique and is not traditionally considered in the context of fuels reduction, activities such as shearing, roller chopping, and windrowing are included in this review because they affect the architecture, mineralization rates, and surface area of materials left onsite, and thus, have relevance to combustibility and fuels management.

The ways and extent to which hydrologic responses from vegetation manipulation occur depend on whether they are expressed as surface flows, such as streamflow, or changes in water-table elevations. Surface flows typically are associated with uplands (Sun and others 2004) because the steeper terrain results in rapid runoff, which encourages the concentration of water and channel formation (Jackson and others 2004). Hydrologic expression via water-table changes typically is associated with flat or depressional terrain (Sun and others 2004) because the lack of slope slows water movement and limits channel network formation and the presence of surface flows (Grace and others 2003, Jackson and others 2004). Surface flow in southeastern wet flatlands occurs primarily within drainage ditches created to make lands more amenable to forest plantation or agricultural growth (Amatya and others 1996, Lebo and Herrmann 1998). Water contributing to these ditches comes principally from saturated or nearly saturated lateral subsurface flow (Amatya and others 1996, 1997; Sun and others 2004); to reflect that this source water results from situations that differ from typical streamflow, drainage in these ditches is sometimes referred to as outflow (Amatya and others 1997, 2002; Grace and others 2006; Lebo and Herrmann 1998).

The various hydrologic responses are described by similar equations. Streamflow in a given time period is described and predicted by the water balance equation:

$$\text{Streamflow} = \text{Precipitation} - \text{Evapotranspiration} + \Delta\ \text{Soil Moisture Storage} + \Delta\ \text{Ground Water Storage} \quad (1)$$

Often ground water changes are assumed to be approximately zero, which simplifies the equation for calculations on a water year basis. Changes in soil moisture storage can be substantial in the short term or seasonally, but over a water year, this term generally is assumed to approach zero. Thus, annually, the equation further simplifies so that total streamflow is determined by how much incoming precipitation is lost to evapotranspiration (ET), which is defined as the cumulative losses of evaporated canopy interception, soil evaporation, and vegetative transpiration. Obviously, climate is a dominant term in controlling ET. But in forest land, ET can be substantially affected by differences in species composition, vegetative density, and microclimate resulting from forest management activities; consequently, streamflow also can be substantially affected.

Equation 1 can be used to predict total stream discharge in the short term, but doing so would require inclusion of changes in soil moisture because these changes are important in controlling streamflow yields. By contrast, the shape of the storm hydrograph cannot be predicted from only the water balance equation—in fact, hydrograph behavior is extremely difficult to predict accurately because precipitation events are unique and random, and physical factors of the watershed affecting the timing of water delivered to stream channels are not constant with time.

Streamflow or outflow in channels or drainage ditches supplied primarily by saturated lateral water movement is similarly described, with one additional component to account for lateral seepage across watershed boundaries (Amatya and others 1996):

$$\begin{aligned}\text{Streamflow} = {} & \text{Precipitation} - \text{Evapotranspiration} + \Delta\ \text{Lateral Seepage} \\ & + \Delta\ \text{Soil Moisture Storage} - \text{Deep Seepage}\end{aligned} \tag{2}$$

The deep seepage term for wet flatlands often also is considered to be approximately zero because the soils involved often are poorly drained (Amatya and others 1996; Grace and others 2003, 2006; Riekerk 1989).

The change in the height of a wetland water table for a given time period is described by the equation (Sun and others 2001):

$$\Delta\ \text{Water table height} = (\Delta\ \text{Inflow} - \Delta\ \text{Outflow} - \Delta\ \text{Evapotranspiration})\ /\ \text{Soil Specific Yield} \tag{3}$$

Although inflow and outflow rates can have substantial effects on water-table height, ET becomes the dominant factor in controlling water-table height if water exchange is slow. Soil specific yield, also known as drainable soil porosity, is the ratio of the volume of water that drains from a saturated soil as a result of lowering the water table relative to the volume of that soil. Its value ranges between zero and one, but it is not actually a constant and depends on position of the water table, rate of water-table change, and soil characteristics (Hillel 1982). Fuel-reduction activities primarily would affect the variables in the numerator of equation 3 (Sun and others 2001).

To ensure that changes in water-table responses are measured and interpreted accurately, measurement wells must be at least as deep as the lowest water-table levels expected during monitoring. If the well is not deep enough, a water table may rise or fall, but documenting the change will be impossible. In these instances, “no measured effect” should not be interpreted as “no effect.”

# Hydrologic Groupings of Provinces

Hydrologic studies of vegetation manipulation have been performed in all the major landforms of the Eastern United States; commonly, the study areas have been experimental forests operated by either the U.S. Department of Agriculture Forest Service or universities, although in the South studies have been applied fairly broadly, particularly on forest-industry lands in the Coastal Plain.

For consistency throughout this volume on Eastern landscapes, the approach is to classify and describe responses by ecological divisions and provinces (chapter 3) to the extent possible. Ecological divisions are defined by “regional climatic types, vegetational affinities, and soil order”; and provinces are defined by “dominant potential natural vegetation, and highlands or mountains with complex vertical climate-vegetation-soil zonation" (U.S. Department of the Interior 2003). As a result, these boundaries do not fall strictly along those that largely define hydrologic behavior, such as the more commonly employed physiographic boundaries defined by geology and topography (U.S. Department of the Interior 2003). Thus, in an effort to keep to the ecological division/province approach as much as possible, we have defined groups of provinces with fairly similar hydrologic characteristics (table 1) and present subsequent discussions based on those groupings. However, because most groupings include provinces from different divisions, these groupings have been assigned more traditional physiographic names (North Central States, Northeastern States, Ozark Mountains and Ouachita Plateau, Central and Southern Appalachian Mountains, Piedmont, and Coastal Plain) because these can be more concisely described and easily understood. The results and interpretations from the reviewed literature should be generally applicable throughout the area encompassed by the respective provinces within the grouping. The principal exception to grouping by distinct ecological province is our separate consideration of the Piedmont and Coastal Plain. Hydrologically, these two areas behave very differently from one another, but the boundaries of the ecological provinces involved (chapter3) do not coincide with the boundaries of the Coastal Plain and Piedmont physiographic areas (fig. 1). Consequently, the Southeastern Mixed Forest Province within the Subtropical Division (230) is included in both the Piedmont and Coastal Plain (table 1).

**Table 1.** Groupings of ecological divisions and provinces that are expected to have similar hydrological responses to fuel-reduction treatments (to simplify discussions in the text, these groupings are assigned physiographic titles)

| Physiographic area | Division and provinces |
|---|---|
| North Central States | 210 Warm Continental<br>212 Laurentian Mixed Forest<br><br>220 Hot Continental<br>222 Midwest Broadleaf Forest |
| Northeastern States | 210 Warm Continental<br>211 Northeastern Mixed Forest<br><br>M210 Warm Continental—Mountain<br>M211 Adirondack—New England Mixed Forest—Coniferous Forest—Alpine Meadow<br><br>220 Hot Continental<br>221 Eastern Broadleaf Forest (northern portion only) |
| Ozark Mountains and Ouachita Plateau | M220 Hot Continental—Mountain<br>M223 Ozark Broadleaf Forest—Meadow<br><br>M230 Subtropical—Mountain<br>M231 Ouachita Mixed Forest—Meadow |
| Central and Southern Appalachian Mountains | M220 Hot Continental—Mountain<br>M221 Central Appalachian Broadleaf Forest—Coniferous Forest—Meadow<br><br>220 Hot Continental<br>221 Eastern Broadleaf Forest (southern portion only) |
| Piedmont | 230 Subtropical<br>231 Southeastern Mixed Forest |
| Coastal Plain | 230 Subtropical<br>231 Southeastern Mixed Forest<br>232 Outer Coastal Plain Mixed Forest<br>234 Lower Mississippi Riverine Forest<br><br>250 Prairie<br>255 Prairie Parkland (Subtropical) |

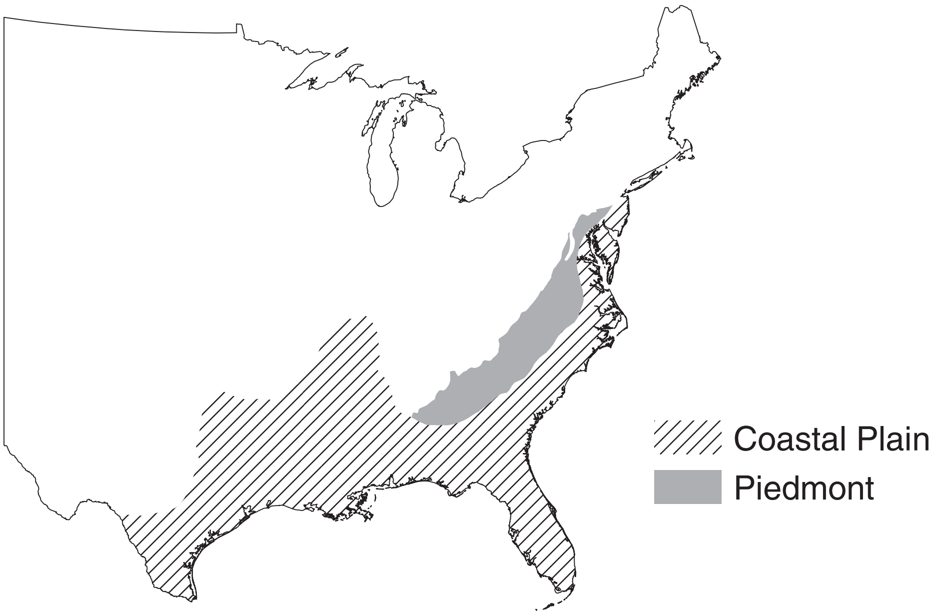


**Figure 1.** Piedmont and Coastal Plain boundaries.

No hydrologic response data applicable to the Prairie Parkland (Temperate) Province (251) were found in the literature, so this province is not included in our chapter. Likewise, the Everglades Province (411) is not included, but fuel-reduction activities are not being practiced in this area.

# Hydrologic Responses

Physiographic areas are discussed in order from north to south in subsequent sections, so that responses from similar climates are grouped closely. When data exist, water-yield and water-table results are presented seasonally (growing and dormant) as well as annually. Storm event responses also are described where data were available. The term stormflow—also called quickflow—is used in this chapter to describe the volume of flow composed of the sum of precipitation falling directly into the channel, surface (overland) flow reaching the channel, and precipitation delivered to the channel by subsurface flow during and immediately following precipitation or snowmelt or both events combined (Hewlett and Hibbert 1961). Peakflow is defined as the instantaneous maximum magnitude or rate of discharge during a precipitation or snowmelt event.

## *North Central States*

The North Central States are characterized by two different types of sites: those that have unsaturated mineral soils (often uplands), and those that are lowland bogs with organic (peat) soils. Often the elevational or topographic differences between the two are not great, but they are large enough to result in different soil characteristics that substantially affect hydrologic responses. Hydrologic changes from vegetation manipulation in organic soils usually are measured as water-table fluctuations, and those in mineral soils usually are measured as streamflow changes.

Streams can exist in lowland bogs, but hydrologic expression in them is generally less demonstrative than in water tables because peatland soils tend to transmit water laterally very slowly (Boelter and Verry 1977). For example, strip cutting followed later by clearcutting black spruce (*Picea mariana*) in a lowland bog on the Marcell Experimental Forest in northern Minnesota did not change streamflow yields (table 2), partially because of the low hydraulic conductivity (Verry 1981). However, the type of peat soil present in the area of harvesting substantially influences hydrologic responses in streams, if present, or water tables. Water moves laterally fairly freely and rapidly in poorly decomposed peats, making these soils hydrologically quite responsive. Conversely, well decomposed organic soils, which are most common in the North Central States, have many small pores that hold water tightly because of extremely low hydraulic conductivities, so losses from these organic soils are primarily as ET (Boelter and Verry 1977).

Water-table responses also are affected by the type of ground water involved. If the harvested stand is over a ground-water fed water table, removing or reducing the ET will have little effect on water-table fluctuations or levels because aquifer supplies greatly exceed precipitation inputs. By contrast, harvesting over perched water tables can result in measurable changes in water-table levels. If precipitation frequency is adequate, water tables in harvested areas will rise because interception losses are reduced. If precipitation is infrequent, the water table will drop after harvesting because there is increased ET caused by winds and increasing transpiration by sedges, which can access deeper moisture than many other plants (Boelter and Verry 1977). This is the type of water-table response that Verry (1981, 1986) reported in the 4 years after clearcutting a bog. Water tables rose 100 mm during wet periods (table 2), because interception was reduced by approximately 170 mm (30 percent), thereby adding that much more precipitation to the peat soils. Conversely, during dry periods, water tables lowered by as much as 190 mm after clearcutting—the result of high ET attributable to more wind and solar radiation, higher surface temperatures, and rapid herbaceous vegetation growth (Verry 1981). Water tables also fluctuated to a greater degree after clearcutting during years of higher than average or lower than average precipitation compared to preharvest conditions.

Harvesting on mineral soils can increase soil moisture (Blackmarr and White 1964, Verry 1972), or water-table levels (Urie 1971), or both; but because mineral soils transmit water to streams quickly, measurements of hydrologic change often are focused on streamflow. Harvests in areas with mineral soils often cover a higher percentage of total watershed area than those on organic soils, further contributing to the degree of hydrologic changein mineral soils. Upland clearcutting of aspen (*Populus tremuloides*) over two-thirds of watershed 4 on the Marcell Experimental Forest in Minnesota resulted in significant increases to annual runoff for 9 years following harvesting (Hornbeck and others 1993, Verry 1987), with approximately half of the 9-year change occurring during harvesting and the 3 years after harvesting (table 2). Changes during those 3 years were 40 to 70 percent above those when trees were present on the watershed. Most of the annual stream augmentation occurred during the growing season (Verry 1972, 1987). No change in annual yield was reported after clearcutting an oak-hickory (*Quercus* spp.–*Carya* spp.) stand in a 0.67-ha watershed at Rose Lake Wildlife Experiment Station in southern Michigan, and ET was estimated to have returned to pretreatment levels within 5 years after the clearcut (Blackmarr and White 1964).

Stormflow effects in the North Central States tend to be a function of whether snowmelt or rainfall is involved and how much of the stand is harvested. Harvesting only about half of watershed 4 of the Marcell Experimental Forest reduced peak runoff during spring snowmelt by 35 percent because the melt in the forest and open areas became desynchronized (Verry 1972, Verry and others 1983). But increasing the area harvested to approximately two-thirds of the watershed increased spring snowmelt peaks from 11 to 143 percent for 7 years (Verry and others 1983), although effects may have lasted for as many as 15 years (Verry 1986). The increases presumably occurred from less desynchronization of snowmelt—resulting from increased heat transfer to the snowpack (from solar radiation) and reradiation of longwave radiation to the snowpack by the regrowing sprouts (Verry 1986, Verry and others 1983). Consequently, snowmelt peak discharges began 3 to 5 days earlier (Verry 1972, Verry and others 1983); however, none of the changes to snowmelt peaks resulted in significant increases to total snowmelt volumes (Verry and others 1983).

By contrast, stormflow volumes from rain events increased by 100 percent or more the first 2 years after harvesting two-thirds of the watershed, but they were not significantly

**Table 2.** Water yield and water table responses to harvesting treatments in the North Central States

| Location | Area, aspect, soils | Treatment description | Time period | Annual changes to water yields | Changes to water table: Annual | Changes to water table: Wet periods | Changes to water table: Dry periods | Reference |
|---|---|---|---|---|---|---|---|---|
| | | | | ----mm---- | ----mm---- | ----mm---- | ----mm---- | |
| Marcell Experimental Forest, Minnesota Watershed 1 | 33.2 ha, no aspect given, mineral soil uplands, peat soils, central bog | Strip cut 43 percent of spruce in bog, strips run northwest to southeast | Years 1 to 5 | NS | NS | | | Verry 1981 |
| | | Year 6 clearcut remaining spruce in bog (total harvested = 8 percent of watershed) | Years 6 to 10 | NS | | 100[a] | –190[a] | |
| Marcell Experimental Forest, Minnesota Watershed 4 | 34.8 ha, flat topography, sandy loams overlaying clay loams in uplands, and peat soils in lowlands | Clearcut aspen 3 m and taller in uplands on 25.3 ha, no harvesting in lowlands 9.5 ha | During harvest | 90[a] (42 percent) | | | | Hornbeck and others 1993; Verry 1972, 1987 |
| | | | Year 1 | 85[a] (39 percent) | | | | |
| | | | Year 2 | 117[a] (58 percent) | | | | |
| | | | Year 3 | 88[a] (70 percent) | | | | |
| | | | Year 4 | 38[a] (20 percent) | | | | |
| | | | Year 5 | 77[a] (34 percent) | | | | |
| | | | Year 6 | 34[a] (45 percent) | | | | |
| | | | Year 7 | 51[a] (34 percent) | | | | |
| | | | Year 8 | 52[a] (21 percent) | | | | |
| | | | Year 9 | 40[a] (15 percent) | | | | |
| | | | Year 10 | 19 (18 percent) | | | | |
| | | | Year 11 | 6 (3 percent) | | | | |
| | | | Year 12 | –22 (–8 percent) | | | | |
| | | | Year 13 | –9 (–9 percent) | | | | |
| | | | Year 14 | 33[a] (16 percent) | | | | |
| | | | Year 15 | 12 (4 percent) | | | | |
| | | | Year 16 | 56[a] (26 percent) | | | | |
| | | | Year 17 | –1 (0 percent) | | | | |
| | | | Year 18 | –25[a] (–8 percent) | | | | |
| | | | Year 19 | 1 (0 percent) | | | | |
| | | | Year 20 | –12 (–10 percent) | | | | |
| | | | Year 21 | –4 (–4 percent) | | | | |
| Sand Plain, Northwestern lower Michigan | 40-acre blocks, no aspect given, sands | Two replicate 16-ha blocks, strip cut 50 percent of pines, strips southeast to northwest orientation | Years 1 to 3 | | 66[b] (20 to 30 percent under strips) | | | Urie 1971 |

NS = nonsignificant change indicated, but no value was given.
[a] Indicates a statistically significant change at the alpha level used by the original authors. Unless otherwise indicated, values without an [a] are nonsignificant.
[b] Significance/nonsignificance information was not provided for this result.

affected by the third year. Rainfall-induced peakflow rates were significantly higher for 8 years—the increases during the first 3 to 5 years were about double preharvest levels. Increases in rainfall-associated peakflows were the result of reductions in soil moisture deficits after harvesting (Verry and others 1983). Peakflow rates from the annual series for 2-year events increased about 1.5 times preharvest levels, compared to 2.5 times for 10-year events (Verry 1986, Verry and others 1983). Similarly, flow duration curves showed that average daily flows increased across all ranges of flow rates with the exception of the very highest flows associated with maximum snowmelt peaks (Verry 1972).

## Northeastern States

Most of the studies in the Northeastern States involve clearcutting or whole-tree harvesting (table 3). First-year water-yield increases from these intensive harvests generally were in the range of 150 to 350 mm or 20 to 40 percent (table 3), although occasionally higher percentage increases were reported from whole-tree harvests (Pierce and others 1993). Small single harvests or small sequential harvests—such as progressive strip cuts—yielded substantially lower annual discharges (Hornbeck and others 1987, Mrazik and others 1980). Regardless of the amount of augmentation, increases were short lived, lasting ≤6 years (table 3); and after 10 to 15 years, water yields commonly fell to levels lower than pretreatment (table 3). This may be because regenerating species had higher transpiration rates than the original stand (Hornbeck and others 1987).

With one exception (Mrazik and others 1980), seasonal data show that annual augmentation in the Northeastern States was almost entirely the result of increased discharge during the growing season, and that water-yield changes during the dormant season were very small (table 3). Water-yield increases of ≥300 percent have been reported from a clearcut during the first one to two growing seasons (Hornbeck and others 1970). Thus, as growing season augmentation diminished, so did annual water yields. Mrazik and others (1980) found that percentage increases in streamflow were higher during the growing season, but the actual volumes of streamflow augmentation during the growing and dormant seasons were similar (table 3). They attributed the lack of seasonal differences to the milder climate in central Massachusetts (such as the study sites at Caldwell Creek and Dickey Brook) compared to other New England study sites, such as Hubbard Brook in New Hampshire.

Increases in streamflow were expressed primarily during low flows. Shifts in flow frequency curves for Caldwell Creek and for watersheds 2, 4, and 5 at Hubbard Brook indicated increases in the numbers of days of occurrence across all flows; but the greatest displacement of the curves was at the lowest flows (Hornbeck and others 1997, Mrazik and others 1980), primarily during the growing season (Hornbeck and others 1997). This same pattern was observed for basal area reductions ranging from about 35 percent (Mrazik and others 1980) to 100 percent of the watershed, although curve displacement was greatest when herbiciding followed clearcutting (Hornbeck and others 1997). For example, average daily growing-season flows equaling or exceeding 1 mm occurred an average of 26 days before clearcutting watershed 2 at Hubbard Brook. After clearcutting and herbiciding, growing-season flow equaled or exceeded 1 mm for 116 days. Removing overstory vegetation by clearcuts, block cuts, and strip cuts resulted in changing the timing of spring snowmelt, but it did not change the overall volume of spring discharge (Hornbeck and Pierce 1970; Hornbeck and others 1970, 1987, 1997; Pierce and others 1970, 1993). More extensive and continuous overstory removal resulted in slightly earlier snowmelt peaks than did light cuts that had substantial residual shade from edge vegetation (Hornbeck and others 1987). On all of the harvests at Hubbard Brook, peakflow from spring snowmelt occurred an average of 4 to 8 days earlier than from a fully forested watershed, although in one year clearcutting caused a shift forward of 17 days on one watershed (Hornbeck and Pierce 1970, Pierce and others 1970). Resulting streamflow and peakflow were higher than normal during these earlier periods of snowmelt and lower than predicted later in the snowmelt season. Snowmelt also ended 2 to 4 days earlier in a clearcut watershed than in an uncut watershed with the same aspect (Hornbeck and Pierce 1970).

**Table 3.** Water yield responses to harvesting treatments in the Northeastern States

| Location | Area, aspect, soils | Treatment description | Time period | Changes to water yields: Annual | Growing | Dormant | Reference |
|---|---|---|---|---|---|---|---|
| | | | | mm | | | |
| Hubbard Brook Experimental Forest, New Hampshire Watershed 2 | 15.8 ha, south, sands and sandy loams | Winter clearcut primarily northern hardwoods, trees left in place, herbicide stumps and ground cover | Year 1 | 347[a] (40 percent) | 315[a] (344 percent) | 25 | Hornbeck and others 1970, 1993, 1997; Pierce and others 1970 |
| | | | Year 2 | 278[a] (29 percent) | 231[a] (310 percent) | 36 | |
| | | | Year 3 | 240[a] (26 percent) | | | |
| | | | Year 4 | 200[a] (22 percent) | | | |
| | | | Year 5 | 146[a] (17 percent) | | | |
| | | | Year 6 | 44[a] (6 percent) | | | |
| | | | Year 7 | 12 (1 percent) | | | |
| | | | Year 8 | 52 (4 percent) | | | |
| | | | Year 9 | 67[a] (8 percent) | | | |
| | | | Year 10 | 3 (0 percent) | | | |
| | | | Year 11 | 4 (5 percent) | | | |
| | | | Year 12 | 64[a] (6 percent) | | | |
| | | | Year 13 | –13 (–1 percent) | | | |
| | | | Year 14 | –13 (–2 percent) | | | |
| | | | Year 15 | –34 (–4 percent) | | | |
| | | | Year 16 | –41[a] (–3 percent) | | | |
| | | | Year 17 | –70[a] (–8 percent) | | | |
| | | | Year 18 | –62[a] (–6 percent) | | | |
| | | | Year 19 | –64[a] (–9 percent) | | | |
| | | | Year 20 | –44[a] (–5 percent) | | | |
| | | | Year 21 | –80[a] (–9 percent) | | | |
| | | | Year 22 | –82[a] (–10 percent) | | | |
| | | | Year 23 | –56[a] (–8 percent) | | | |
| | | | Year 24 | –34 (–3 percent) | | | |
| | | | Year 25 | –48[a] (–4 percent) | | | |
| Hubbard Brook Experimental Forest, New Hampshire Watershed 5 | 22 ha, aspect not given, sandy loam | Whole tree clearcut of northern hardwoods applied to 20 ha | Year 1 | 152[a] (23 percent) | | –49[a b] | Hornbeck and others 1997 |
| | | | Year 2 | 47 (5 percent) | | –46[a b] | |
| | | | Year 3 | –15 (–2 percent) | | | |
| | | | Year 4 | –11 (–1 percent) | | | |
| | | | Year 5 | 4 (1 percent) | | | |
| | | | Year 6 | 46 (5 percent) | | | |
| | | | Year 7 | 51[a] (5 percent) | | | |
| | | | Year 8 | 66[a] (8 percent) | | | |
| | | | Year 9 | 47 (6 percent) | | | |
| | | | Year 10 | 20 (2 percent) | | | |
| | | | Year 11 | 21 (4 percent) | | | |
| | | | Year 12 | 46 (4 percent) | | | |

*continued*

**Table 3.** Water yield responses to harvesting treatments in the Northeastern States (continued)

| Location | Area, aspect, soils | Treatment description | Time period | Changes to water yields: Annual | Growing | Dormant | Reference |
|---|---|---|---|---|---|---|---|
| | | | | --------mm-------- | | | |
| Great Northern Paper Co., Maine[c] | 47 ha, aspect not given, sandy loam | Spruce-fir whole-tree harvest | Year 1 | 310 (63 percent) | | | Pierce and others 1993 |
| | | | Year 2 | 290 | | | |
| | | | Year 3 | 210 | | | |
| Success, New Hampshire[c] | 7 ha, aspect not given, sandy loam | Northern hardwoods whole-tree harvest | Year 1 | 220 (45 percent) | | | Pierce and others 1993 |
| | | | Year 2 | 160 | | | |
| | | | Year 3 | 170 | | | |
| Chester, Connecticut[c] | 6 ha, aspect not given, sandy loam | Central hardwoods whole-tree harvest | Year 1 | 270 (22 percent) | | | Pierce and others 1993 |
| | | | Year 2 | 100 | | | |
| | | | Year 3 | 70 | | | |
| Hubbard Brook Experimental Forest, New Hampshire Watershed 4 | 36 ha, southeast, sandy loams | Northern hardwoods progressive strip cutting in east-west orientation | Year 1 | 22 (3 percent) | 28[a] | –4 | Hornbeck and others 1987, 1997 |
| | | | Year 2 | 46[a] (4 percent) | 36[a] | –5 | |
| | | | Year 3 | 114[a] (8 percent) | 91[a] | 14 | |
| | | | Year 4 | 67[a] (8 percent) | 38[a] | 33 | |
| | | | Year 5 | 55[a] (4 percent) | 81[a] | 33 | |
| | | | Year 6 | 81[a] (9 percent) | 38[a] | –22 | |
| | | | Year 7 | 69[a] (7 percent) | 27[a] | 34 | |
| | | | Year 8 | –14 (–2 percent) | 2 | 42[a] | |
| | | | Year 9 | –30[a] (–4 percent) | 6 | –15 | |
| | | | Year 10 | –27[a] (–3 percent) | 0 | –35 | |
| | | | Year 11 | –18 (–2 percent) | | | |
| | | | Year 12 | –45[a] (–5 percent) | | | |
| | | | Year 13 | –33[a] (–3 percent) | | | |
| | | | Year 14 | –21[a] (–3 percent) | | | |
| | | | Year 15 | –44[a] (–5 percent) | | | |
| | | | Year 16 | –67[a] (–8 percent) | | | |
| | | | Year 17 | –29 (–4 percent) | | | |
| | | | Year 18 | –59[a] (–8 percent) | | | |
| | | | Year 19 | –42[a] (–4 percent) | | | |
| | | | Year 20 | –63[a] (–5 percent) | | | |
| | | | Year 21 | –42[a] (–5 percent) | | | |
| | | | Year 22 | –60[a] (–6 percent) | | | |
| | | | Year 23 | –30[a] (–3 percent) | | | |
| | | | Year 24 | –36[a] (–6 percent) | | | |
| | | | Year 25 | –23 (–2 percent) | | | |

continued

**Table 3.** Water yield responses to harvesting treatments in the Northeastern States (continued)

| Location | Area, aspect, soils | Treatment description | Time period | Changes to water yields: Annual | Growing | Dormant | References |
|---|---|---|---|---|---|---|---|
| | | | | ---mm--- | | | |
| Hubbard Brook Experimental Forest, New Hampshire Watershed 101 | 12 ha, southeast, sandy loam | Northern hardwoods block cuts | Year 1 | 278 (36 percent) | 237 | 8 | Hornbeck and others 1987 |
| | | | Year 2 | 155 | 140 | 41 | |
| | | | Year 3 | 92 | 85 | 15 | |
| | | | Year 4 | 39 | 45 | 7 | |
| | | | Year 5 | 41 | 36 | –6 | |
| | | | Year 6 | 20 | 22 | 5 | |
| | | | Year 7 | 15 | 10 | 20 | |
| | | | Year 8 | 8 | 6 | 15 | |
| | | | Year 9 | 1 | 5 | 5 | |
| | | | Year 10 | 4 | 5 | 1 | |
| Caldwell Creek watershed, central Massachusetts | 163 ha, south, sands, and sandy loams | 49 percent riparian overstory mixed oaks and northern hardwoods and understory vegetation chemically deadened; 21 percent of upland pine plantations chemically deadened; uplands harvesting in patch clearcuts; 34.4 percent of basal area in watershed deadened or harvested | Year 1 | 103[a] (21.6 percent) | 56[a] (38.4 percent) | 53[a] (16.1 percent) | Mrazik and others 1980 |
| | | | Year 2 | 155[a] (22.5 percent) | 89[a] (43.9 percent) | 76[a] (15.9 percent) | |
| | | | Year 3 | 91[a] (19.7 percent) | 47[a] (30.9 percent) | 50[a] (16.6 percent) | |
| | | | Year 4 | 79[a] (13.5 percent) | 31[a] (21.0 percent) | 53 (12.2 percent) | |
| | | | Year 5 | 133 (14.1 percent) | 66[a] (20.0 percent) | 89 (14.9 percent) | |
| | | | Year 6 | 72 (10.9 percent) | 49[a] (20.9 percent) | 36 (8.5 percent) | |
| | | | Year 7 | 36 (5.7 percent) | 8[a] (4.2 percent) | 38 (8.8 percent) | |

NS = nonsignificant change indicated, but no value was given;.
[a] Indicates a statistically significant change at the alpha level used by the original authors. Unless otherwise indicated, values without an [a] are nonsignificant.
[b] Based only snowmelt runoff only.
[c] Significance/nonsignificance information was not provided for this site.

Reducing vegetation can result in other hydrograph responses, but the limited results reported in the literature from the Northeastern States indicate that changes to peakflow and stormflow were small even after clearcutting an entire watershed (table 4). No changes in average peak discharges were observed for 3 years at Dickey Brook after 32 percent of the basal area was removed using a combination of clearcutting and thinning (Bent 1994). High flows (> 0.2 $m^3$/second/$km^2$) increased an average of 13 percent during the first 3 years after clearcutting and herbiciding watershed 2 at Hubbard Brook (Hornbeck and others 1970). The average annual increase in stormflow during the first 3 years was 21 percent (Hornbeck and others 1970), but the largest relative increases in storm peaks and the largest increases in stormflow volumes have been reported primarily during the largest events (Hornbeck 1973, Mrazik and others 1980) during the growing season (table 4). Average annual stormflow on Hubbard Brook watershed 2 increased 99 mm during the first 3 years, with two-thirds of that occurring during the growing season (table 4). Dormant season stormflow increases were restricted primarily to spring melt events, because snowmelt was concentrated in only one short period or a few short periods. Individual stormflow totals during these spring melts can be much higher than during other times of the year. For example, the maximum increase in spring stormflow from Hubbard Brook watershed 2 was 50 mm, compared to a maximum summer stormflow increase of 30 mm (Hornbeck 1973). By contrast, changes to average peak discharge at Caldwell Creek were distributed relatively evenly between growing and dormant seasons during the first 4 years after deadening or harvesting approximately 35 percent of the watershed (Mrazik and others 1980).

## *Ozark Mountains and Ouachita Plateau*

Unlike the other areas described in this chapter, available discharge data from the Ozark Mountains and Ouachita Plateau focus on stormflows and peakflows rather than annual yields, because runoff data have been collected primarily from ephemeral channels.

In the Ozark Mountains, clearcutting a third of a 6.6-ha oak watershed did not change stormflow even though half of the harvested area was cut using a logger's choice method and soil disturbance was substantially more than what would have occurred with best management practices (Settergren and others 1980). The lack of change was attributed to the limited area that was harvested and the fact that disturbance was confined to the ephemeral headwaters. Had the harvesting been in lower portions of the watershed closer to the nonephemeral portions of the channel, stormflow increases via reductions in soil infiltration and subsequent overland flow may have occurred as a consequence of the extensive soil disturbance. For example, mechanical removal of litter significantly reduced infiltration rates of four Missouri soil series by 11 to 25 percent, with an average reduction of 18 percent (Arend 1941). Annual burning of the hardwood litter layer for 5 to 6 years across a variety of soils exposed mineral soil and reduced soil infiltration 20 to 62 percent, with an average reduction of 38 percent (Arend 1941).

Even though Settergren and others (1980) observed no changes to stormflow after clearcutting a third of a watershed, some local soil moisture augmentation may have occurred because of reductions in transpiration. Substantial differences in soil moisture deficits were observed between clearcut and forested plots on the Koen Experimental Forest (Rogerson 1976). Average maximum soil water deficit in the clearcut plots was 78 mm, only 29 percent of the average maximum deficit of 267 mm in the forested plots. Soil moisture deficits in the clearcut plots were present only during summer and autumn; recharge occurred earlier than in the forest because summer deficits grew only at an average daily rate of 0.6 mm in the clearcut plots compared to 2.1 mm in the forest.

The soil disturbance associated with site preparation following clearcutting of shortleaf pine (*Pinus echinata*) substantially affected hydrology in three small watersheds in the Ouachita Mountains of Oklahoma (Miller 1984). Site preparation following clearcutting included roller chopping, burning, and contour ripping the subsoil. The resulting soil disturbance increased roughness and detention storage in the furrows, and cut off soil

**Table 4.** Changes in stormflow volumes and peakflow magnitudes resulting from harvesting treatments in the Northeastern States

| | | | | Hydrologic changes | | | | | | |
|---|---|---|---|---|---|---|---|---|---|---|
| | | | | Mean peak discharge | | | Stormflow | | | |
| Location | Area, aspect, soils | Treatment description | Time period | Annual | Growing | Dormant | Annual | Growing | Dormant | Reference |
| Hubbard Brook Experimental Forest, New Hampshire Watershed 2 | 15.8 ha, south, sands and sandy loams | Winter clearcut, trees left in place, herbicide stumps and ground cover | Years 1 to 3[a] | 13 percent | 118 percent[b] | 0 percent | | | | Hornbeck 1973, Hornbeck and others 1970 |
| | | | Years 1 to 3[a] | | | | 21 percent | 197 percent[b] | 13 percent | |
| | | | Years 1 to 3 for all storms | | | | 99 mm[b] | 64 mm[b] | 28 mm[b] | |
| Dickey Brook watershed, New Salem, Massachusetts | 308 ha, west, fine sandy loam and sandy loam | 9 percent watershed whole-tree harvest clearcut and 5 percent thinned; 32 percent basal area removed | Years 1 to 3[c] | NS | | | | | | Bent 1994 |
| Caldwell Creek watershed, central Massachusetts | 163 ha, south, sands and sandy loams | 49 percent riparian overstory mixed oaks and northern hardwoods and understory vegetation chemically deadened; 21 percent of upland pine plantations chemically deadened; uplands harvesting in patch clearcuts; 34.4 percent of basal area in watershed deadened or harvested | Years 1 to 4 | | 0.12 $m^3$/second/$km^2$[b] (50 percent) | | | | | Mrazik and others 1980 |
| Hubbard Brook Experimental Forest, New Hampshire Watershed 5 | 22 ha, no aspect given, sandy loam | Whole-tree clearcut of northern hardwoods applied to 20 ha | Year 1 | | 18 percent | 15 percent | | | | Hornbeck and others 1997[d] |
| | | | Year 2 | | 63 percent | –2 percent | | | | |
| | | | Year 3 | | 31 percent | –2 percent | | | | |
| | | | Year 4 | | 19 percent | –30 percent | | | | |
| | | | Year 5 | | 15 percent | 10 percent | | | | |
| | | | Year 6 | | | –12 percent | | | | |
| | | | Year 7 | | 29 percent | –2 percent | | | | |
| | | | Year 8 | | | –13 percent | | | | |
| | | | Year 9 | | 28 percent | –30 percent | | | | |
| | | | Year 10 | | | –40 percent | | | | |
| | | | Year 11 | | | –14 percent | | | | |
| | | | Year 12 | | | –13 percent | | | | |

NS = nonsignificant change indicated, but no value was given.
[a] For storms with peaks >1.5 $m^3$/second/$km^2$ or storms with stormflows >1.5 $m^3$/second/$km^2$.
[b] Indicates a statistically significant change at the alpha level used by the original authors. Unless otherwise indicated, values without a [b] are nonsignificant.
[c] For storms with 2-day precipitation totals >25.4 mm.
[d] Changes to mean peak discharge in this study were calculated from only statistically significant peaks that were >10 mm per day.

macropores connected to ephemeral channels. Precipitation then became routed into the subsoil rather than laterally to streamflow. As a result, average stormflow in the clearcut watersheds fell to levels below the uncut controls (table 5) even though transpiration and probably interception losses were reduced greatly by harvesting and site preparation. Only during the second year after treatment, which was unusually dry, were the reductions in ET in the clearcut watersheds enough to significantly increase stormflows. Annual average peakflow rates also were not affected by harvesting and site preparation.

In the Ouachita Mountains of Arkansas, stormflow responses resulting from clearcutting followed by roller chopping and burning were compared to those from selection harvesting with no site preparation and with uncut controls (Miller and others 1988). The large average annual stormflow increases that were observed (table 5) indicated

**Table 5.** Water yield responses (as stormflow volumes) to harvesting treatments in the Ozark Mountains and Ouachita Plateau

| Location | Acreage, aspect, soils | Treatment description | Time period | Stormflow changes | Reference |
|---|---|---|---|---|---|
| Ouachita Mountains, Oklahoma | 1.6 to 4.2 ha, southwest, loam overlaying silt clay | Three replicate watersheds, clearcut, roller chop, burn, contour soil ripping (subsoiling), hand plant | Year 1 | –94 | Beasley and others 2000, Miller 1984 |
| | | | Year 2 | 49 [a] | |
| | | | Year 3 | –11 | |
| | | | Year 4 | –17 | |
| Ouachita Mountains, Arkansas | 4.08, 5.11, and 5.91 ha, north, southeast, and northwest, loam overlaying clay | Three replicate watersheds, clearcut and roller chop, burn, hand plant | Year 1 | 101 | Beasley and others 2000, Miller and others 1988 |
| | | | Year 2 | 92 | |
| | | | Year 3 | 193 | |
| | 4.15, 4.35, and 5.74 ha, north, south, and west, loam overlaying clay | Three replicate watersheds, selection harvest | Year 1 | 101 | Beasley and others 2000, Miller and others 1988 |
| | | | Year 2 | 74 | |
| | | | Year 3 | 149 | |
| Ouachita Mountains, Arkansas | 0.52 ha, northeast, stony silt loams | Overstory pine thinned, 57 percent basal area removed, mixed hardwood understory herbicided annually for 3 years | Year 1 | 109 [a] [b] (79 percent) | Rogerson 1985 |
| | | | Year 2 | 57 | |
| | | | Year 3 | 82 | |
| | | | Year 4 | 66 | |
| | | | Year 5 | 0 | |
| | | | Year 6 | 49 | |
| | | | Year 7 | 41 | |
| | 0.59 ha, northeast, stony silt loams | Overstory pine clearcut, mixed hardwood understory herbicided annually for 3 years | Year 1 | 259 [a] [b] (193 percent) | Rogerson 1985 |
| | | | Year 2 | 141 | |
| | | | Year 3 | 113 | |
| | | | Year 4 | 135 | |
| | | | Year 5 | 143 | |
| | | | Year 6 | 160 | |
| | | | Year 7 | 102 | |
| Athens Plateau, Arkansas | 2 to 5 ha, aspect not given, fine sand or fine loam | Three replicate watersheds, clearcut, shear, windrow, plant | Year 1 | 166 [a] | Beasley and others 1986, 2000 |
| | | | Year 2 | 388 | |
| | | | Year 3 | 237 [a] | |
| | 2 to 5 ha, aspect not given, fine sand or fine loam | Three replicate watersheds, clearcut, chemical site preparation, plant | Year 1 | –3 | Beasley and others 1986, 2000 |
| | | | Year 2 | 176 | |
| | | | Year 3 | –4 | |

[a] indicates a statistically significant change at the alpha level used by the original authors. Unless otherwise indicated, values without an [a] are nonsignificant.
[b] Significance/nonsignificance for this study was specified only for year 1.

that stormflow increased roughly proportionally to the amount of harvesting and site disturbance; however, the increases were not significant because of substantial variability in responses across the replicated sites. One of the three replicated sites in each harvesting treatment consistently yielded much higher annual stormflow volumes than the controls, presumably because they had more lateral moisture movement through the soil (Miller and others 1988). Hydrologic increases calculated from those sites provide a measure of high-end responses that could be expected: the stormflow increase for the clearcut/site-prepared watershed would have been 98 mm more than the average reported for year 1 and 100 mm more than year 2 (table 5). By contrast, selection-harvest values from the first 2 years essentially would have been unchanged (–5 and 0 mm different). These larger stormflow values suggest that all clearcut/site-prepared watersheds (including those dominated by vertical soil moisture) had substantial short-term reductions in transpiration and interception compared to the selection harvests. The changes in soil moisture from all clearcut and selection-harvest watersheds were large enough to increase the number of stormflow events that occurred the first 2 years after harvesting and also lengthen the time that stormflows, albeit in small volumes, were present. After harvesting, periods of stormflow extended farther into the summer and began earlier in the autumn. Peakflow increases also were related to the intensity of treatment, but peakflows did not differ significantly among the treatments and the controls.

In other nearby watersheds in the Ouachita Mountains, clearcutting and thinning pine followed by 3 years of herbiciding to control hardwood regrowth also decreased soil water deficits, which in turn increased discharge from ephemeral channels (table 5). The first growing season after treatment, soil water deficits were reduced by as much as 51 to 76 mm on the thinned watershed and 76 to 102 mm on the clearcut watershed (Rogerson 1985). Elevated soil moisture levels continued for at least another six growing seasons. Dormant-season soil moisture was not affected by either harvesting treatment. Resulting first-year water-yield increases for both types of harvests were substantial, but those from the clearcut were about 2.5 times larger, both in volume and percentage (table 5). Water yields over the 7-year study increased an average of 23 percent from thinning and 67 percent from clearcutting; more than half of those volume increases occurred during the growing seasons (thinned 52 percent, clearcut 61 percent) (Rogerson 1985).

Clearcutting with mechanical site preparation in the Athens Plateau of Arkansas increased stormflow significantly the first and third years after treatment (Beasley and others 1986), but clearcutting followed by chemical site preparation did not increase streamflow during any of the years (table 5) because vegetation deadening was incomplete, stump sprouting was common, and overall disturbance to watershed soils was less. Thus, more soil moisture augmentation was needed before stormflow could be generated. Stormflow increases in year 2 were very large because that year was unusually wet with several large rainfalls; however, stormflow was not statistically different than pretreatment because of substantial variability in responses among replicate watersheds attributable to variable soil depths. The Athens Plateau is an area of transition between the Ouachita Mountains and the west Gulf Coastal Plain, and watershed replicates in the west Gulf Coastal Plain had deeper soils than the replicates located in the thinner rocky soils of the Ouachita Mountains. The result was almost no stormflow discharge in the Coastal Plain area except during the unusually wet second year; replicates in the Ouachita Mountain soils yielded much more stormflow.

## *Central and Southern Appalachian Mountains*

The Appalachian Mountains cover a fairly extensive north-to-south range. In the northern portions, snow is an important component of the hydrologic cycle, although snowpacks are typically not continuous throughout most winters (U.S. Department of Agriculture 1987). In the southern portion, snow comprises only a small percentage of the hydrologic budget (Hewlett and Hibbert 1961). The vast majority of the available data for this area is from the Fernow Experimental Forest in north central West

Virginia and Coweeta Hydrologic Laboratory in western North Carolina. Limited amounts of data also are available from central Pennsylvania (including the Leading Ridge Watershed Research Unit) and the Cumberland Plateau. These studies provide data from a wide variety of experiments, including thinnings, other partial harvests, and understory removal/reduction experiments.

In general, more intensive levels of harvesting in the Appalachian Mountains result in greater augmentation of annual flows (table 6), and first-year water-yield increases are proportional to the basal area removed (Hewlett and Hibbert 1961, Kochenderfer and others 1990). Thinnings in which only small percentages of the basal area were removed typically resulted in small, nonsignificant changes in annual discharges, whereas with few exceptions clearcutting most or all of a watershed increased annual water yields by at least 100 mm (and often more) during the first year or two after harvesting. Site preparation following clearcutting at Clover watershed in West Virginia (Kochenderfer and Helvey 1989) and Coweeta watershed 6 (Hibbert 1969) did not increase annual water yields (table 6) more than from clearcutting alone (Douglass and Swank 1972, Hewlett and Helvey 1970, Hoover 1944, Johnson and Kovner 1954, Kochenderfer and others 1990, Kovner 1956, Lull and Reinhart 1967, Meginnis 1959). However, vegetative reductions do not have to be restricted to the overstory to increase annual discharge. Removal of a thick understory of mountain laurel (*Kalmia latifolia*) and rhododendron (*Rhododendron maximum*) that accounted for 22 percent of the basal area of Coweeta watershed 19 resulted in significant, albeit short-term increases in annual yields (table 6).

In the Appalachian Mountains, yields typically decline quickly because of rapid regrowth and restoration of ET encouraged by high precipitation levels and relatively long growing seasons. For example, after clearcutting at Fernow, Leading Ridge, and Coweeta, discharges returned to pretreatment levels in 5 to 10 years (Hornbeck and Kochenderfer 2001, Hornbeck and others 1993, Swank and others 2001). After recovery, streamflow can fall below that of the uncut stand because of changes in species composition and/or leaf area index of the regrowing stand (Swank and others 2001).

More severe deforestation treatments using herbicides to kill residual vegetation and prohibit regrowth (Fernow watersheds 6 and 7) resulted in higher annual increases than from clearcutting alone (table 6). This was likely because nearly all of the transpiration on the watersheds ceased from the deadening, whereas in traditional clearcuts substantial live vegetation remains in residual saplings and understory plants. Because the denudation lasted several years and regeneration occurred primarily by seed sources rather than root or stump sprouts (Hornbeck and others 1993), the effects lasted about 15 years, which is substantially longer than harvest-only studies at Fernow (table 6). Annual cutting for almost 15 years to eliminate regrowth on Coweeta watershed 17 also elevated streamflow during the entire period (Johnson and Kovner 1954). The annual discharge levels were similar to initial levels from clearcutting other north-facing watersheds at Coweeta (table 6).

High road density or a lack of best management practices or both factors had little effect on annual water yields. Fernow watershed 1 had both a high density (7.3 percent of watershed area) of skidroads and no best management practices applied during or after harvesting (Reinhart and others 1963), and annual stream discharge was similar to other clearcut watersheds on the Fernow (table 6). Likewise the absence of best management practices in Kentucky (table 6) resulted in only slightly higher annual water yields (~30 mm) than a nearby watershed that had the same cutting treatment and application of best management practices (Arthur and others 1998). Coweeta watershed 28 had a high road density with 66 percent of basal area removed but lower yields than watershed 37, which has a similar aspect and only 50 percent basal area removed (Hewlett and Helvey 1970).

A major difference between watershed responses on Fernow and Coweeta is the influence that aspect has on annual water yields after clearcutting and other intensive treatments. Aspect at Fernow did not affect annual discharge; at Coweeta annual increases from watersheds with a northerly aspect were almost always higher than those with a southerly aspect (Hewlett and Hibbert 1961). Discharges from most south-facing

**Table 6.** Water yield responses to harvesting treatments in the Appalachian Mountains

| Location | Area, aspect, soils | Treatment description | Time after treatment | Changes to water yields: Annual | Growing | Dormant | Reference |
|---|---|---|---|---|---|---|---|
| | | | | | ----------mm---------- | | |
| Robinson Forest, Kentucky Watershed B | Area not reported, east, loams and fine loams | Clearcut hardwoods and pines >35.5 cm d.b.h., cut and left stems <5 cm, BMPs employed | During clearcutting | NS | | | Arthur and others 1998 |
| | | | First 17 months | 178 mm[a] (123 percent) | | | |
| | | | Year 8 | 15 percent[b] | | | |
| | | | Years 1 to 8 | 37 percent[b] | | | |
| Robinson Forest, Kentucky Watershed C | Area not reported, east, loams and fine loams | Clearcut hardwoods and pines >35.5 cm d.b.h., cut and left stems <5 cm, no BMPs employed | During clearcutting | NS | | | Arthur and others 1998 |
| | | | First 17 months | 206 mm[a] (138 percent) | | | |
| | | | Year 8 | 12 percent[b] | | | |
| | | | Years 1 to 8 | 48 percent[b] | | | |
| Leading Ridge, Pennsylvania Watershed 2 | 42.9 ha, southeast, silt loams, stony loams, and cobbly loams | Riparian clearcut ~one-third of watershed (8.6 ha), herbicides to control stump sprouts during next 3 summers | Year 1 | 70 mm[a] | 53[a] | –27 | Hornbeck and others 1993, Lynch and others 1972 |
| | | | Year 2 | 32 mm[a] (11 percent) | 23[a] | –5 | |
| | | | Year 3 | 63 mm[a] (16 percent) | 25[a] | 40[a] | |
| | | | Year 4 | 73 mm[a] (14 percent) | 42[a] | 26 | |
| | | | Year 5 | 50 mm[a] (10 percent) | | | |
| | | | Year 6 | 94 mm[a] (10 percent) | | | |
| | | | Year 7 | 79 mm[a] (17 percent) | | | |
| | | | Year 8 | 132 mm[a] (25 percent) | | | |
| | | | Year 9 | 61 mm[a] (9 percent) | | | |
| | | | Year 10 | 193 mm[a] (32 percent) | | | |
| | | | Year 11 | 239 mm[a] (35 percent) | | | |
| | | | Year 12 | 138 mm[a] (21 percent) | | | |
| | | | Year 13 | 63 mm[a] (9 percent) | | | |
| | | | Year 14 | 73 mm[a] (28 percent) | | | |
| | | | Year 15 | –25 mm (–4 percent) | | | |
| | | | Year 16 | –37 mm (–7 percent) | | | |
| | | | Year 17 | 49 mm[a] (9 percent) | | | |
| | | | Year 18 | 22 mm (4 percent) | | | |
| | | | Year 19 | 56 mm[a] (11 percent) | | | |
| | | | Year 20 | 64 mm[a] (11 percent) | | | |
| | | | Year 21 | –28 mm (–7 percent) | | | |
| | | | Year 22 | 26 mm (6 percent) | | | |
| | | | Year 23 | 36 mm[a] (6 percent) | | | |
| | | | Year 24 | 27 mm (3 percent) | | | |

*continued*

**Table 6.** Water yield responses to harvesting treatments in the Appalachian Mountains (continued)

| Location | Area, aspect, soils | Treatment description | Time after treatment | Changes to water yields: Annual | Growing | Dormant | Reference |
|---|---|---|---|---|---|---|---|
| | | | | | ----------mm---------- | | |
| Leading Ridge, Pennsylvania Watershed 3 | 104.0 ha, southeast, silt loams, stony loams, and cobbly loams | Commercial clearcut hardwoods on 44.5 ha with BMPs | Year 1 | 137 mm[a] | 146[a] | –31 | Lynch and Corbett 1990 |
| | | | Year 2 | 39 mm | 27[a] | 10 | |
| | | | Year 3 | –61 mm[a] | 25[a] | –89[a] | |
| | | | Year 4 | 51 mm[a] | 1 | 60[a] | |
| | | | Year 5 | –37 mm | –33 | –11 | |
| | | | Year 6 | –20 mm | 0.5 | –20 | |
| | | | Year 7 | 17 mm | 8 | 1 | |
| | | | Year 8 | 8 mm | 2 | 6 | |
| | | | Year 9 | 22 mm | 3 | 7 | |
| Fernow Experimental Forest, West Virginia Watershed 1[d] | 29.9 ha, northeast, silt loams | Commercial clearcut hardwoods >13 cm d.b.h., no BMPs employed; 74 percent basal area removed | Year 1 [c] | 56 mm[a] | 30[a] | 26 | Kochenderfer and others 1990, Lull and Reinhart 1967 |
| | | | Year 2 | 130 mm[a] | 107[a] | 16 | |
| | | | Year 3 | 86 mm[a] | 76[a] | 19 | |
| | | | Year 4 | 89 mm[a] | 44[a] | 44[a] | |
| | | | Year 5 | 61 mm | 2 | 9 | |
| | | | Year 6 | 46 mm[a] | 30[a] | 29 | |
| | | | Year 7 | 36 mm | –25[a] | 19 | |
| | | | Year 8 | 28 mm | –10[a] | 26 | |
| | | | Year 9 | 20 mm | 8 | 42 | |
| | | | Year 10 | 15 mm | 8[a] | 8 | |
| | | | Year 11 | 13 mm | 2 | 14 | |
| | | | Year 12 | | 16[a] | 7 | |
| | | | Year 13 | | 14[a] | 4 | |
| | | | Year 14 | | –4 | –0.2 | |
| | | | Year 15 | | 5 | 19 | |
| | | | Year 16 | | –27[a] | 14 | |
| | | | Year 17 | | –11[a] | 28 | |
| | | | Year 18 | | –5 | 27 | |
| | | | Year 19 | | 4 | 15 | |
| | | | Year 20 | | 3 | 10 | |
| | | | Year 25 | | –10[a] | 2 | |
| | | | Year 30 | | 26[a] | 41[a] | |

*continued*

**Table 6.** Water yield responses to harvesting treatments in the Appalachian Mountains (continued)

| Location | Area, aspect, soils | Treatment description | Time after treatment | Changes to water yields: Annual | Growing | Dormant | Reference |
|---|---|---|---|---|---|---|---|
| | | | | | mm | mm | |
| Fernow Experimental Forest, West Virginia Watershed 2[d] | 15.4 ha, south, silt loams | 43-cm diameter limit cut 1958, 32 percent basal area removed; 43-cm diameter limit cut 1972, 12 percent basal area removed | Year 1[c] | 25 mm | 18[a] | 5 | Kochenderfer and others 1990, Reinhart and others 1963, Reinhart and Trimble 1962, Trimble and others 1963 |
| | | | Year 2 | 64 mm[a] | 46[a] | 20 | |
| | | | Year 3 | 36 mm[a] | 18[a] | 22 | |
| | | | Year 4 | | –5 | 15 | |
| | | | Year 5 | | 9 | 33 | |
| | | | Year 6 | | –22[a] | 18 | |
| | | | Year 7 | | 2 | 34[a] | |
| | | | Year 8 | | 5 | 39[a] | |
| | | | Year 9 | | 6 | 30[a] | |
| | | | Year 10 | | 6 | 19 | |
| | | | Year 11 | | 11[a] | 5 | |
| | | | Year 12 | | –1 | 26 | |
| | | | Year 13 | | 10 | 30 | |
| | | | Year 14 | | –4 | 27 | |
| | | | Year 15 | | 6 | 34 | |
| | | | Year 16 | | 26[a] | 37 | |
| | | | Year 17 | | 14 | 53[a] | |
| | | | Year 18 | | 1 | 31 | |
| | | | Year 19 | | 8 | 15 | |
| | | | Year 20 | | 9 | 31[a] | |
| | | | Year 21 | | –6 | 60[a] | |

*continued*

**Table 6.** Water yield responses to harvesting treatments in the Appalachian Mountains (continued)

| Location | Area, aspect, soils | Treatment description | Time after treatment | Changes to water yields: Annual | Growing | Dormant | Reference |
|---|---|---|---|---|---|---|---|
| | | | | | ----mm---- | | |
| Fernow | 34.4 ha, south, silt | Intensive selection harvests | Year 1[c] | –3 mm | –14 | 8 | Kochenderfer and |
| Experimental | loams | 1958 13 percent basal area | Year 2 | 8 mm | 8 | –0.3 | others 1990, |
| Forest, West | | removed | Year 3 | | 10 | 25[a] | Reinhart and |
| Virginia | | 1963 8 percent basal area | Year 4 | | –3 | 8 | Trimble 1962, |
| Watershed 3[d] | | removed | Year 5 | | 3 | 9 | Reinhart and |
| | | | Year 6 | | –17[a] | 17[a] | others 1963, |
| | | 0.2-ha patch cuts 1968 | Year 7 | | 0.2 | 15 | Trimble and |
| | | 6 percent basal area | Year 8 | | –2 | 21[a] | others 1963 |
| | | removed | Year 9 | | 8 | 32[a] | |
| | | | Year 10 | | 9 | 17 | |
| | | Clearcut 1968 to | Year 11 | | 7 | 22[a] | |
| | | 2.5-cm d.b.h. | Year 12 | | 35[a] | 21[a] | |
| | | 91 percent basal area | Year 13 | | 171[a] | 56[a] | |
| | | removed | Year 14 | | 64[a] | 27[a] | |
| | | | Year 15 | | 36[a] | 36[a] | |
| | | Riparian buffer cut 1972 | Year 16 | | 45[a] | 40[a] | |
| | | 9 percent basal area | Year 17 | | 29[a] | 53[a] | |
| | | removed | Year 18 | | 9 | 40[a] | |
| | | | Year 19 | | 2 | 17 | |
| | | | Year 20 | | 21[a] | 38[a] | |
| | | | Year 25 | | 11 | 32[a] | |
| | | | Year 30 | | –17[a] | 36[a] | |
| Fernow | 36.4 ha, northeast, | Extensive selection harvest | Year 1[c] | 25 mm [a] | | –3 | Kochenderfer and |
| Experimental | silt loams | 20 percent basal area | Year 2 | 18 mm | 36[a] | –15 | others 1990, Lull |
| Forest, West | | removed 1958 | Year 3 | | –8 | 8 | and Reinhart |
| Virginia | | | Year 4 | | 0 | –18 | 1967, Reinhart |
| Watershed 5[d] | | | | | | | and others 1963, |
| | | | | | | | Trimble and |
| | | | | | | | others 1963 |

*continued*

**Table 6.** Water yield responses to harvesting treatments in the Appalachian Mountains (continued)

| Location | Area, aspect, soils | Treatment description | Time after treatment | Changes to water yields: Annual | Growing | Dormant | Reference |
|---|---|---|---|---|---|---|---|
| | | | | | ---mm--- | | |
| Fernow Experimental Forest, West Virginia Watershed 6[d] | 22.3 ha, southeast, silt loams | 1964 clearcut lower half, removed 51 percent basal area; herbicide through 1969; 1967-68 clearcut upper half, removed 49 percent basal area; herbicide through 1969 | Year 1[c] | | 30 | 20 | Hornbeck and others 1993, Kochenderfer and others 1990, Patric and Reinhart 1971 |
| | | | Year 2 | | 79[a] | 86[a] | |
| | | | Year 3 | | 107[a] | 36[a] | |
| | | | Year 4 | | 99[a] | 15 (second one-half cut) | |
| | | | Year 5 | | 201[a] | 58[a] | |
| | | | Year 6 | | 228[a] | 31[a] | |
| | | | Year 7 | | 130[a] | 54[a] | |
| | | | Year 8 | | 116[a] | 55[a] | |
| | | | Year 9 | | 64[a] | 31[a] | |
| | | | Year 10 | | 49[a] | 44[a] | |
| | | | Year 11 | | 78[a] | 57[a] | |
| | | | Year 12 | | 61[a] | 73[a] | |
| | | | Year 13 | | 85[a] | 50[a] | |
| | | | Year 14 | | 94[a] | 72[a] | |
| | | | Year 15 | | 90[a] | 104[a] | |
| | | | Year 16 | | 90[a] | 45[a] | |
| | | | Year 17 | | 49[a] | 81[a] | |
| | | | Year 18 | | 132[a] | 57[a] | |
| | | | Year 19 | | 142[a] | 45[a] | |
| | | | Year 20 | | 60[a] | 71[a] | |
| | | | Year 21 | | 85[a] | 47[a] | |
| | | | Year 22 | | 86[a] | 58[a] | |
| | | | Year 23 | | 57[a] | 56[a] | |
| | | | Year 24 | | 20 | 31 | |

*continued*

**Table 6.** Water yield responses to harvesting treatments in the Appalachian Mountains (continued)

| Location | Area, aspect, soils | Treatment description | Time after treatment | Changes to water yields: Annual | Growing | Dormant | Reference |
|---|---|---|---|---|---|---|---|
| | | | | | ---mm--- | | |
| Fernow Experimental Forest, West Virginia Watershed 7[e] | 24.3 ha, east, silt loams | 1963 clearcut upper half, 49 percent basal area removed; herbicide through 1969 | Year 1[c] | 165 mm[a] (23 percent) | 91[a] | 64[a] | Hornbeck and others 1993, Kochenderfer and others 1990, Patric and Reinhart 1971 |
| | | | Year 2 | 142 mm[a] (36 percent) | 74[a] | 71[a] | |
| | | | Year 3 | 157 mm[a] (23 percent) | 124[a] | 25[a] (second one-half cut) | |
| | | 1966-67 clearcut lower half 51 percent basal area removed; herbicide through 1969 | Year 4 | 251 mm[a] (38 percent) | 218[a] | | |
| | | | Year 5 | 258 mm[a] (40 percent) | 191[a] | 33[a] | |
| | | | Year 6 | 246 mm[a] (33 percent) | 217[a] | 71[a] | |
| | | | Year 7 | 224 mm[a] (33 percent) | 149[a] | 21 | |
| | | | Year 8 | 175 mm[a] (20 percent) | 118[a] | 66[a] | |
| | | | Year 9 | 164 mm[a] (16 percent) | 93[a] | 48[a] | |
| | | | Year 10 | 157 mm[a] (17 percent) | 74[a] | 69[a] | |
| | | | Year 11 | 187 mm[a] (20 percent) | 81[a] | 82[a] | |
| | | | Year 12 | 104 mm[a] (17 percent) | 36 | 104[a] | |
| | | | Year 13 | 65 mm (11 percent) | 20 | 80[a] | |
| | | | Year 14 | 89 mm[a] (12 percent) | 28 | 53[a] | |
| | | | Year 15 | 112 mm[a] (11 percent) | 39 | 62[a] | |
| | | | Year 16 | 99 mm[a] (12 percent) | 40 | 70[a] | |
| | | | Year 17 | 62 mm[a] (8 percent) | 21 | 68[a] | |
| | | | Year 18 | 61 mm (6 percent) | 12 | 53[a] | |
| | | | Year 19 | 109 mm[a] (14 percent) | 53[a] | 60[a] | |
| | | | Year 20 | 103 mm[a] (12 percent) | 30 | 63[a] | |
| | | | Year 21 | 71 mm[a] (8 percent) | 25 | 68[a] | |
| | | | Year 22 | 52 mm (5 percent) | 24 | 55[a] | |
| | | | Year 23 | 88 mm[a] (13 percent) | 20 | 20 | |
| | | | Year 24 | 48 mm (9 percent) | –3 | 71 | |
| | | | Year 25 | 55 mm[a] (7 percent) | | 54 | |
| | | | Year 26 | 37 mm (4 percent) | | | |
| | | | Year 27 | –2 mm (0 percent) | | | |
| Clover Run, West Virginia Watershed 9 | 11.6 ha, south, silt loams | Clearcut trees >15 cm d.b.h. removed; brush windrowed onto contoured roads and along perimeter of watershed, root raking with stumps left intact; trees >2.5 cm d.b.h. on 1.2 ha of steep land herbicided; buffer strip uncut | Year 1 | | 99[a] | NS | Kochenderfer and Helvey 1989 |
| | | | Year 2 | | 71[a] | NS | |
| | | | Year 3 | | 38[a] | NS | |

*continued*

**Table 6.** Water yield responses to harvesting treatments in the Appalachian Mountains (continued)

| Location | Area, aspect, soils | Treatment description | Time after treatment | Changes to water yields: Annual | Growing | Dormant | Reference |
|---|---|---|---|---|---|---|---|
| | | | | | ----------mm---------- | | |
| Coweeta Hydrologic Laboratory, North Carolina Watershed 1[b] | 16.2 ha, south, deep and permeable, (texture not reported) | Cove hardwoods deadened 25 percent basal area 1954; clearcut 100 percent basal area, no products removed, slash scattered, partial control burn 1956; planted to pine 1957 | Year 1[f]<br>Year 2<br>Year 3<br>Year 4[f]<br>Year 5<br>Year 6<br>Year 7<br>Year 8<br>Year 9 | 30 mm<br><25 mm<br><25 mm<br>145 mm<br>18 mm<br>58 mm<br>41 mm<br>71 mm<br>69 mm | | | Swank and Miner 1968 |
| Coweeta Hydrologic Laboratory, North Carolina Watershed 3[b] | 9 ha, south, texture not reported | Clearcut | Year 1 | 127 mm | | | Hewlett and Hibbert 1961 |
| Coweeta Hydrologic Laboratory, North Carolina Watershed 6[b] | 8.9 ha, northwest, deep and permeable (texture not reported) | Riparian area 5 m from stream cut 1941; clearcut hardwoods 1958; remaining vegetation cut piled and burned; seedbed prepared by burning, grubbing and harrowing; planted to grass; foliar herbiciding subsequent years to control hardwood regrowth; fertilized and limed | First 17 months<br>Year 1[g]<br>Year 2<br>Year 3<br>Year 4<br>Year 5 | 74 mm<br>–17 mm<br>47 mm<br>64 mm<br>147 mm<br>149 mm | | | Hibbert 1969 |
| Coweeta Hydrologic Laboratory, North Carolina Watershed 7[b] | 59.0 ha, south, texture not reported | Clearcut and yarded using mobile cable system; <10 percent of watershed area with soil disturbance | Year 1<br>Year 2<br>Year 3<br>Year 4<br>Year 5<br>Year 6 | 260 mm (28 percent)<br>200 mm<br>170 mm<br>120 mm<br>40 mm<br>40 mm | | | Swank and others 1982, 1988 |
| Coweeta Hydrologic Laboratory, North Carolina Watershed 10[b] | 86 ha, south, texture not reported | Commercial clearcut | Year 1 | 25 mm | | | Hewlett and Hibbert 1961 |

*continued*

**Table 6.** Water yield responses to harvesting treatments in the Appalachian Mountains (continued)

| Location | Area, aspect, soils | Treatment description | Time after treatment | Changes to water yields: Annual | Growing | Dormant | Reference |
|---|---|---|---|---|---|---|---|
| | | | | | ----------mm---------- | | |
| Coweeta Hydrologic Laboratory, North Carolina Watershed 13[b] | 16.2 ha, northeast, deep and permeable (texture not reported) | Clearcut trees and shrubs, 100 percent basal area cut, no products removed; allowed to regrow | Year 1 | 367 mm (57 percent) | | | Johnson and Kovner 1954, Kovner 1956, Meginnis 1959 |
| | | | Year 2 | 277 mm (40 percent) | | | |
| | | | Year 3 | 278 mm (31 percent) | | | |
| | | | Year 4 | 248 mm (28 percent) | | | |
| | | | Year 5 | 200 mm (27 percent) | | | |
| | | | Year 6 | 252 mm (28 percent) | | | |
| | | | Year 7 | 201 mm (24 percent) | | | |
| | | | Year 8 | 185 mm (21 percent) | | | |
| | | | Year 9 | 132 mm (15 percent) | | | |
| | | | Year 10 | 131 mm (15 percent) | | | |
| | | | Year 11 | 145 mm (17 percent) | | | |
| | | | Year 12 | 147 mm (15 percent) | | | |
| | | | Year 13 | 127 mm | | | |
| Coweeta Hydrologic Laboratory, North Carolina Watershed 17[b] | 13.4 ha, northwest, deep and permeable (texture not reported) | Clearcut all stems >1.2 cm d.b.h., 100 percent basal area cut; wood products left onsite; little soil disturbance; recut regrowth each growing season for most years during next 15 years | First 6 months | 207 mm | | | Hoover 1944, Johnson and Kovner 1954 |
| | | | Year 1 | 425 mm (65 percent) | | | |
| | | | Year 2 | 271 mm | | | |
| | | | Year 3 | 229 mm | | | |
| | | | Year 4 | 152 mm | | | |
| | | | Year 5 | 152 mm | | | |
| | | | Year 6 | 330 mm | | | |
| | | | Year 7 | 279 mm | | | |
| | | | Year 8 | 279 mm | | | |
| | | | Year 9 | 279 mm | | | |
| | | | Year 10 | 254 mm | | | |
| | | | Year 11 | 279 mm | | | |
| | | | Year 12 | 279 mm | | | |
| Coweeta Hydrologic Laboratory, North Carolina Watershed 19[b] | 28.3 ha, northwest, loams, sandy loams, and sandy clay loams | All laurel and rhododendron on 28.3 ha cut close to ground, 22 percent basal area cut; slash left onsite, little soil disturbance | Year 1 | 71 mm | 49 | 36 | Johnson and Kovner 1956, Meginnis 1959 |
| | | | Year 2 | 64 mm | 39 | 15 | |
| | | | Year 3 | 55 mm | 30 | 45 | |
| | | | Year 4 | 47 mm | 20 | 22 | |
| | | | Year 5 | 39 mm | 11 | 26 | |
| | | | Year 6 | 31 mm | 1 | 9 | |
| | | | Year 7 | | –8 | | |
| | | | Years 1 to 6 | 4 percent | | | |

*continued*

**Table 6.** Water yield responses to harvesting treatments in the Appalachian Mountains (continued)

| Location | Area, aspect, soils | Treatment description | Time after treatment | Changes to water yields: Annual | Growing | Dormant | Reference |
|---|---|---|---|---|---|---|---|
| | | | | | ----mm---- | | |
| Coweeta Hydrologic Laboratory, North Carolina Watershed 22[b] | 34.4 ha, north, texture not reported | All vegetation in alternate 10.1-m wide strip cuts herbicided, 50 percent basal area cut; no products removed | Year 1 | 198 mm | | | Hewlett and Hibbert 1961 |
| Coweeta Hydrologic Laboratory, North Carolina Watershed 28[b] | 144.0 ha, northeast, deep and permeable (texture not reported) | All trees and shrubs clearcut on 77 ha, cove vegetation thinned on 39 ha 66 percent basal area removed, 28 ha no harvest; products removed; high road density | Year 1<br>Year 2<br>Year 3<br>Year 4<br>Year 5<br>Year 6<br>Year 7 | 165 mm<br>102 mm<br>79 mm<br>23 mm<br>28 mm<br>81 mm<br>104 mm | | | Douglass and Swank 1972 |
| Coweeta Hydrologic Laboratory, North Carolina Watershed 37[b] | 43.7 ha, northeast, texture not reported | Clearcut watershed, 100 percent basal area cut, no products removed, no roads constructed | Year 1<br>Year 2<br>Year 3 | 264 mm (18 percent)<br>91 mm (6 percent)<br>94 mm (6 percent) | | | Hewlett and Helvey 1970 |
| Coweeta Hydrologic Laboratory, North Carolina Watershed 40 | 20.2 ha, southeast, texture not reported | Commercial logging, selection cut, 22 percent basal area removed | Year 1 | 0 mm | | | Hewlett and Hibbert 1961 |
| Coweeta Hydrologic Laboratory, North Carolina Watershed 41[b] | 28.7 ha, southeast, texture not reported | Commercial logging, selection cut, 35 percent basal area removed | Year 1 | 51 mm | | | Hewlett and Hibbert 1961 |

d.b.h. = diameter at breast height; BPMs = best management practices; NS = nonsignificant change indicated, but no value was given.
[a] Indicates a statistically significant change at the alpha level used by the original authors. Unless otherwise indicated, values without an [a] are nonsignificant.
[b] Significance/nonsignificance for this result or watershed was not provided.
[c] Includes the harvesting or first harvesting period.
[d] Significance/nonsignficance information for published data for this watershed was obtained from U.S. Department of Agriculture Forest Service records.
[e] Significance/nonsignificance information for published growing and dormant season data for this watershed after year 5 was obtained from U.S. Forest Service records.
[f] Year 1 includes deadening, year 4 includes clearcutting.
[g] After grass establishment.

clearcut catchments at Coweeta were similar to those from clearcut watersheds at Fernow (table 6) and were typically 100 to 200 mm less than north-facing Coweeta watersheds the first year or two after clearcutting. Complete reduction of the forest on north-facing slopes yielded an average of 400 mm of discharge the first year following harvest. Removing half of the basal area correspondingly reduced yields by about half (200 mm) on Coweeta watershed 22 (Hewlett and Hibbert 1961). South-facing Coweeta watershed 7 was the exception to the aspect differences. For unknown reasons, it had first- and second-year increases that were similar to those of north-facing watersheds at Coweeta (Swank and others 1982, 1988). Aspect responses could not be evaluated from the Pennsylvania or Kentucky data because harvesting was not performed on multiple aspects in either location.

The causes for the differences in runoff between north- and south-facing aspects at Coweeta have not been definitively identified. Hewlett and Hibbert (1961) initially suggested that they might be due at least partially to soil depth, watershed configuration, and aquifer characteristics. However, a more likely reason is that substantially different solar energy inputs affect north- and south-facing slopes (Douglass 1983). First-year streamflow yield increases from the Appalachian Mountains are explained primarily by basal area removed—positively related—and incoming energy—negatively related (Douglass 1983, Douglass and Swank 1975). South-facing hillsides receive more radiation year round than north-facing ones, so that changes in ET, and subsequently discharge, after harvesting on the south-facing slopes may not be as dramatic as on north-facing slopes. Sites in the Central Appalachian Mountains may not experience aspect differences because the watersheds are not as steep and do not have as large elevational ranges (Hibbert 1966), so all aspects may receive more similar energy inputs.

Aspect differences at Coweeta also influence the way that water-yield increases are expressed seasonally. Clearcuts on north-facing watersheds tend to have their largest quantitative augmentation of flow during the late dormant season—such as January to April (Hewlett and Hibbert 1961)—because of the lag that results from the time needed for these deep soils to recharge (Kovner 1956, Meginnis 1959). South-facing clearcut watersheds at Coweeta tend to express the majority of their water-yield increases during the late growing season (Hewlett and Hibbert 1961) because reductions in ET caused by harvesting elevate soil moisture, which subsequently becomes streamflow (Swank and others 2001). Lower intensity treatments at Coweeta tend to display associated water-yield increases during the growing season (Hewlett and Hibbert 1961), but seasonal expression of flow is less consistent and predictable. Riparian clearing in Coweeta watershed 6 created only small water-yield increases restricted to the growing season (Dunford and Fletcher 1947); understory removal on watershed 19 produced small increases distributed throughout the year (Johnson and Kovner 1956).

At both Leading Ridge and Fernow (table 6), water-yield increases during the first 1 to 3 years after clearcutting predominantly occur during the growing season (Kochenderfer and others 1990, Lynch and others 1972, Reinhart and Trimble 1962, Reinhart and others 1963). Significant dormant season increases also can occur during those first years, but the magnitude of increase is usually substantially less than during the growing season. Typically growing season yields return to preclearcut levels after only 5 to 7 years at both Leading Ridge and Fernow (table 6), but dormant season increases at Fernow tend to last much longer (Kochenderfer and others 1990). For example, growing season yields for Fernow watershed 3 returned to preharvest conditions in about 5 years, but dormant season yields remained elevated for all but 2 years during 18 years of postharvest monitoring (Kochenderfer and others 1990). Delaying regrowth with herbicides following clearcutting extended the duration of both growing and dormant season responses on Fernow watersheds 6 and 7. On watershed 6 significant increases for both growing and dormant seasons lasted about 20 years, reaching similar levels during both seasons 8 years after the first-half clearcutting. On Fernow watershed 7, most of the growing season increase disappeared after 10 years, but the dormant season increase lasted at least another 15 years (Kochenderfer and others 1990).

Low-intensity thinnings on Fernow watersheds 2 and 5 had small but significant effects on augmenting growing season flows, but these lasted only a year or two (table

6). Even the second thinning on watershed 2 that removed only 12 percent of the basal area increased growing and dormant season streamflow significantly for 2 years. However, dormant season flow behavior became somewhat erratic in subsequent years, so it is unknown if the changes from such a light thinning actually were attributable to the treatment (Kochenderfer and others 1990).

Regardless of location and seasonality of streamflow increases, most measurable increases in water yields occur during periods of low flow in the Appalachian Mountains (Douglass and Swank 1975). Flow frequency curves show shifts in the position of the curves in the low-to-moderate ranges of average daily flow after treatment compared to preharvest conditions at Fernow, Coweeta, and Leading Ridge (Johnson and Kovner 1954, Johnson and Meginnis 1960, Lynch and others 1972, Patric and Reinhart 1971, Reinhart and Trimble 1962), indicating these lower end flows occurred more frequently after harvesting (table 7). Changes in high-end flows were much smaller or nonexistent. For clearcut Coweeta watersheds 13 and 17, there were no significant shifts in the flow frequency curves for flows >50 L/second/km$^2$ (Johnson and Meginnis 1960). Curve positions for flows ≥55 L/second/km$^2$ were not shifted during either half or total watershed clearcutting and herbiciding on Fernow watersheds 6 or 7 (Patric and Reinhart 1971). Following riparian clearcutting and control of sprouting at Leading Ridge watershed 2, flow frequencies during growing and dormant seasons were not changed for flows >8.7 L/second/km$^2$ (Lynch and others 1972). The actual changes in the volumes associated with the low flows are each relatively small (table 6), but because these flows occur so frequently, their accumulated totals over a year or a season are quite sizable and much larger than the small increases to moderate or higher flows. In general, the higher intensity of vegetation removed, the larger the shift in the frequency curve (Reinhart and Trimble 1962) for a given site.

Excluding clearcut watershed 3, table 7 shows that all of the other harvested Fernow watersheds had much higher percentage increases of low flows than Coweeta—this is

**Table 7.** Flow frequency curve results for the Appalachian Mountains (refer to table 6 for watershed and treatment descriptions)

| Location | Time period | Flow level equaled or exceeded[a] | Average increase | Volume increase | Reference |
|---|---|---|---|---|---|
| | | *percent* | *percent* | *L/second/km$^2$* | |
| Fernow Watershed 1 | First 2 growing seasons | 84 | 1700 | 0.9 | Reinhart and others 1963[b] |
| | | 50 | 500 | 4.9 | |
| | | 16 | 132 | 22.3 | |
| Fernow Watershed 2 | First 2 growing seasons | 84 | 200 | 0.7 | Reinhart and others 1963[b] |
| | | 50 | 221 | 3.4 | |
| | | 16 | 84 | 15.7 | |
| Fernow Watershed 3 | First 2 growing seasons | 84 | 20 | 0.1 | Reinhart and others 1963[b] |
| | | 50 | 33 | 0.3 | |
| | | 16 | 0 | 0 | |
| Fernow Watershed 5 | First 2 growing seasons | 84 | 100 | 0.4 | Reinhart and others 1963[b] |
| | | 50 | 38 | 1.4 | |
| | | 16 | 20 | 6.5 | |
| Coweeta Watershed 13 | First 7 years | 84 | 62 | 7 | Johnson and Kovner 1954, Johnson and Meginnis 1960 |
| | | 50 | 41 | 8 | |
| | | 16 | 17 | | |
| Coweeta Watershed 17 | First 7 years | 84 | 124 | | Johnson and Kovner 1954 |
| | | 50 | 50 | | |
| | | 16 | 35 | | |

[a] A flow level of 50 percent represents median flow; 84 percent represents the median flow plus one standard deviation; 16 percent represents the median flow minus one standard deviation.
[b] Volume data were determined from contemporary reconstruction of flow frequency curves presented in Reinhart and others (1963).

true even for thinned Fernow watersheds 2 and 5, which were cut much less heavily and treated less intensively than clearcut Coweeta watersheds 13 and 17. However, the data are not fully comparable, as the Coweeta values represent average responses over 7 years and the Fernow data are average responses over only the first two growing seasons. Even with this longer "averaging time," the median absolute increase from Coweeta watershed 13 was about double that from the Fernow watersheds (table 7). This is expected because of the lower absolute increases observed across the Fernow compared to north-facing Coweeta watersheds.

Supplements to low flows can measurably decrease the number of low-flow or no-flow days in Appalachian headwater channels (table 8). Streams on clearcut and herbicided Fernow watersheds 6 and 7 always dried up for at least a month each year before deforestation; but when each was only half deforested, streamflow never dropped <0.55 L/second/km$^2$. When each was fully deforested, flows were always ≥3.3 L/second/km$^2$ (Patric and Reinhart 1971). Clearcutting on Fernow watersheds 1 and 3 and thinning on Fernow watersheds 2 and 5 reduced the number of days that streamflow was <0.55 L/second/km$^2$ (Troendle 1970). Discharge was doubled on clearcut Coweeta watershed 7 during low-flow months (Swank and others 2001). Cutting only the riparian zone on Coweeta watershed 6 added 10 to 13 m$^3$ of extra water daily to the stream during rainless days the first growing season after treatment and 4 to 8 m$^3$ during the second growing season (Johnson and Kovner 1954). More intensive harvests tend to result in a larger reduction in the number of low-flow days (Trimble and others 1963) and greater loss of the diurnal fluctuations in streamflow that are typically observed during low flows (Dunford and Fletcher 1947). A reduction in low-flow days contributes to prolonging ground-water depletion rates during baseflow hydrographs, at least for watersheds subject to intensive harvests (table 9). For example, clearcutting Coweeta watershed 17 resulted in lengthening the time needed for flow to decrease from 20 to 4.7 L/second/km$^2$ by 25 days (Johnson and Meginnis 1960).

Fewer years of stormflow data and analyses are available for the Appalachian Mountains compared to annual and seasonal analyses. However, the available results

**Table 8.** Decreases in the number of days during which designated low-flow levels occurred following harvesting in the Appalachian Mountains (refer to table 6 for watershed and treatment descriptions)

| Location | Flow level | Time period | Decrease in number of days during which low flow occurred | Reference |
|---|---|---|---|---|
| | *L/second/km²* | | | |
| Leading Ridge Watershed 2 | <7.34 | Year 1 | 40[a] | Lynch and others 1972 |
| | | Year 2 | 5 | |
| | | Year 3 | 61[a] | |
| | | Year 4 | 46[a] | |
| Fernow Watershed 1 | <3.67 | Year 1 | 72[a] | Reinhart and others 1963, Trimble and others 1963 |
| | | Year 2 | 38[a] | |
| | | Year 3 | 63[a] | |
| | | Year 4 | 39[a] | |
| Fernow Watershed 2 | <3.67 | Year 1 | 22[a] | Reinhart and others 1963, Trimble and others 1963 |
| | | Year 2 | 47[a] | |
| | | Year 3 | 27[a] | |
| Fernow Watershed 3 | <3.67 | Year 1 | 21[a] | Reinhart and others 1963, Trimble and others 1963 |
| | | Year 2 | 14[a] | |
| Fernow Watershed 5 | <3.67 | Year 1 | 5 | Reinhart and others 1963, Reinhart and Trimble 1962, Trimble and others 1963 |
| | | Year 2 | 13[a] | |
| | | Year 3 | 5 | |

[a] Indicates a statistically significant change at the alpha level used by the original authors. Unless otherwise indicated, values without an [a] are nonsignificant.

are consistent, showing that changes to most hydrograph components are small and nonsignificant (tables 10 and 11). Even though clearcut Coweeta watersheds 13 and 17 had the largest annual augmentation of streamflow of any watersheds shown in table 6, neither experienced significant annual changes to peakflow rates or stormflow volumes (table 10); thus, streamflow increases were almost entirely from baseflow (Kovner 1956). Coweeta watershed 37 had only a 7-percent increase in average annual peakflows and an 11-percent increase in average annual stormflow for the 30 largest storms during the first 4 years after clearcutting (Hewlett and Helvey 1970), so even more extreme events were affected little. The largest increase in stormflow volume for a single storm on watershed 37 was 25 percent (Hewlett and Helvey 1970). Generally changes to stormflows at Coweeta only occur for larger storms, as the moisture storage associated with the deep soils prohibits changes to stormflow volumes that are <25 mm (Hewlett and Helvey 1970). Clearcut Coweeta watershed 7 had the most consistent responses to larger precipitation events across all variables, although the increases to stormflow components were fairly small (tables 10 and 11). The small magnitudes of the change were attributable to the lack of disturbance to the soil surface during harvesting and low road density resulting from preplanning (Swank and others 1982, 2001). The principal changes to Coweeta watershed 7 hydrographs were to the recession limbs (Swank and others 2001).

The largest percentage changes to peakflow and stormflow at Coweeta were on watershed 28 (table 10), which was conventionally clearcut to remove 66 percent of the basal area but had a high density of roads, to which these changes were attributed (Swank and others 1988). Clearcut Fernow watershed 1 also had a high density of poorly located roads (Reinhart and others 1963), but only responses for the runoff events that were in the top 23 percent of events were examined. For these highflow events, growing season peakflows increased 21 percent, and stormflows increased 24 percent; change was minimal annually and almost nonexistent during the dormant season (table 10). Although changes to Fernow watershed 1 storm hydrographs were not large, Reinhart (1964) observed sharp, short- duration peaks at the start of some larger storm hydrographs. These first peaks were attributed to contributions of overland flow directly to the stream from the poor road layout and drainage from the road, which was exacerbated by road interception of subsurface flows. Trimble and others (1963) noted that the location and number of roads in a watershed can affect stormflow responses, as roads can direct concentrated flow directly to stream channels. The higher the road density and closer the roads are to streams, the more that hydrograph components—including peakflow—can be expected to change. However, even with the presence of roads, total streamflow increases in Fernow watershed 1 primarily were caused by decreased soil moisture deficiencies from harvesting, and road-induced changes were small (Reinhart 1964).

The greatest absolute and percentage changes to stormflow occurred on Leading Ridge watershed 2 (riparian clearcut) and on Fernow watersheds 3 (clearcut) and 6 (clearcut+herbicide). On each of these catchments, average peak discharge during the growing season increased by >300 percent (table 10). Although the 300-percent

**Table 9.** Changes in depletion times during low flows following basal area reductions in the western North Carolina highlands of the Appalachian Mountains (refer to table 6 for watershed and treatment descriptions)

| Location | Flow depletion | Before clearcut | After clearcut | Resulting streamflow increase |
|---|---|---|---|---|
| | | Requirement for depletion to occur | | |
| | *L/second/km²* | *number of days* | | *mm* |
| Coweeta watershed 13 | 20 to 6 | 65 | 82 | 10 |
| Coweeta watershed 17 | 20 to 4.7 | 38 | 63 | 14 |
| Coweeta watershed 19 | 14 to 9 | 12 | 27 | 3 |

Source: Johnson and Meginnis (1960).

**Table 10.** Changes in stormflow volumes and peakflow magnitudes following harvesting for the Appalachian Mountains (refer to table 6 for watershed and treatment descriptions)

| | | Hydrologic changes | | | | | | |
|---|---|---|---|---|---|---|---|---|
| | | Mean peak discharge change | | | Mean stormflow change | | | |
| Location | Time period | Annual | Growing | Dormant | Annual | Growing | Dormant | Reference |
| Leading Ridge Watershed 2 | Years 1 to 3.5[b] | 115 L/second/km$^2$[a] (118 percent) | 280 L/second/km$^2$[a] (351 percent) | | 0.5 mm[a] (32 percent) | 1.7 mm[a] (171 percent) | | Lynch and others 1972 |
| Fernow Watershed 1[c] | Year 1[d]<br>Years 1 to 4[d] | 4 percent | 21 percent | <br>–4 percent | 7 percent | 24 percent | <br>2.5 percent | Lull and Reinhart 1966, Reinhart 1964, Reinhart and Trimble 1962 |
| Fernow Watershed 3 | Year 1[e] | | 300 percent[a] | NS | NS | | | Patric 1980 |
| Fernow Watershed 6 | Year 1[f] | | 400 percent[a] | NS | | | | Patric and Reinhart 1971 |
| Fernow Watershed 7 | Year 1[f] | | NS | NS | | | | Patric and Reinhart 1971 |
| Coweeta Watershed 7 | Years 1 to 3[g]<br>Years 1 to 4[g] | 19 L/second/km$^2$[a] (15 percent)<br>17 L/second/km$^2$[a] (15 percent) | | | 0.3 mm[a] (10 percent)<br>0.3 mm[a] (10 percent) | | | Swank and others 1982, 1988, 2001 |
| Coweeta Watershed 13 | Year 1 | NS | | | NS | | | Meginnis 1959 |
| Coweeta Watershed 17 | Years 1 to 2 | NS | | | NS | | | Hoover 1944, Meginnis 1959 |
| Coweeta Watershed 19 | Year 1 | | | | NS | | | Johnson and Kovner 1956, Meginnis 1959 |
| Coweeta Watershed 28 | Years 1 to 9<br>Years 1 to 2 | <br>30 percent[a] | | | 17 percent[a] | | | Swank and others 1988, 2001 |
| Coweeta Watershed 37 | Years 1 to 4[h] | 65.6 L/second/km$^2$[a] (7 percent) | | | 5.8 mm[a] (11 percent) | | | Hewlett and Helvey 1970, Swank and others 2001 |

NS = nonsignificant change.
[a] Indicates a statistically significant change at the alpha level used by the original authors. Unless otherwise indicated, values without an [a] are nonsignificant.
[b] For events with stormflow >0.025 cm. Annual extends from April through November each year. Growing season includes May through October each year.
[c] Significance/nonsignificance information was not provided for this watershed.
[d] For storms >109 L/second/km$^2$.
[e] After clearcutting.
[f] After full deforestation.
[g] For precipitation ≥2 cm.
[h] For 30 largest events.

**Table 11.** Changes in hydrograph parameters following harvesting in the Appalachian Mountains (refer to table 10 for information about changes to peakflow magnitudes and total stormflow volumes; refer to table 6 for watershed and treatment descriptions)

| Location | Time period | Time to peak | Recession time | Stormflow duration | Stormflow before peak | Stormflow after peak | Initial flow | Reference |
|---|---|---|---|---|---|---|---|---|
| | | *percent* | | -----------------------*percent*---------------------- | | | | |
| Coweeta Watershed 7[a] | Years 1 to 4 | 0 | 10 percent[b] | 5[b] | 6[b] | 11 | 14 percent[b] | Swank and others 2001 |
| Coweeta Watershed 37 | Years 1 to 4 | NS | NS | NS | | | | Hewlett and Helvey 1970, Swank and others 1988 |
| Leading Ridge Watershed 2 | Years 1 to 3.5[c] | –3 during both growing and dormant seasons | 4.2 hr (33 percent)[b] growing season; no change dormant season | | | | 2.54 mm[d] (12 percent) annually; 5.08 mm[d] (123 percent) growing season; 1 percent[d] dormant season | Lynch and others 1972 |

NS = nonsignificant change indicated, but no value was given.
[a] For precipitation ≥2 cm.
[b] Indicates a statistically significant change at the alpha level used by the original authors. Unless otherwise indicated, values without a [b] are nonsignificant.
[c] April through November events with stormflow >0.0.25 cm; annual period extends from April through November; growing season includes May through October; dormant season includes only April and November.
[d] Significance/nonsignificance information was not provided for this result.

increase on Fernow watershed 3 was significant, it represented only moderate peakflow increases (Patric 1980). Growing season stormflow on Leading Ridge watershed 2 also increased by 171 percent but the increases were associated with storms that produced relatively low initial flow rates (Lynch and others 1972).

Activities applicable to fuels reduction in the Appalachian Mountains have primarily involved mechanical actions, and few investigations have examined the hydrologic effects of controlled fires. This may be partially attributable to the fact that the Appalachian Mountains are fairly moist (Swift and others 1993), supporting ground conditions that make the severity of fires relatively light even when the burn is high intensity (Van Lear and Kapeluck 1989). After commercial clearcutting—followed by application of high-intensity burning of standing residuals on one plot and felled residuals on another—in South Carolina, soil infiltration rate was not different from the unburned clearcut plots (179 cm/hour): 183 cm/hour for the standing-residual plot and 157 cm/hour for the felled-residual plot (Van Lear and Danielovich 1988). Even though the burn was intense, a substantial amount of organic matter remained on the surface, and soil macropores in the deep soils were not changed by the burning; thus, soil infiltration rates remained high (Van Lear and Danielovich 1988) and soils hydrophobicity did not develop (Van Lear and Kapeluck 1989).

Swift and others (1993) also found that soils did not become hydrophobic after a low-intensity fire that followed felling (both overstory and understory) and burning on a poor-quality site in western North Carolina. Humus as well as some charred litter was present over much of the area after burning, so relatively little soil became exposed.

Consequently, although infiltration rates were not measured, they apparently were not changed much as overland flow showed no evidence of increasing. The lack of change to soil infiltration allowed soil moisture levels to increase in late summer immediately after harvesting in the top 60 cm of soil and even somewhat farther in autumn after burning. Soil moisture increases were present consistently during the second growing season in the top 30 cm of soil, but they were only about half what they had been the previous summer and autumn. Most of the soil moisture increases were attributed to reductions in transpiration from the combination of cutting and burning, and augmentations were greatest in the headwaters of ephemeral channels, making it likely that stormflow increased (Swift and others 1993).

## *Piedmont*

The history of the Piedmont includes widespread agricultural activities that have resulted in extensive and often severe erosion. The current expressions of this past erosion are shallow soils, incised stream channels, and gullies that also serve as channels for runoff (Hewlett 1979, Van Lear and others 1985). Shallow soils and denser and incised channel networks reduce the potential for soil-moisture storage by increasing the potential for soil moisture to reach channels, and allow channels to intersect water tables at deeper levels (Hewlett 1979). These characteristics mean that streamflow in Piedmont watersheds can be highly responsive to even moderate changes in the other variables of the water balance equation (equation 1).

Hewlett (1979) found that clearcutting 32 ha of loblolly pine (*Pinus taeda*) in the Georgia Piedmont followed by double roller chopping increased water yields by 254 mm the first year after harvest and site preparation, and 126 mm the second year (table 12). Similarly, after harvesting and site preparation using a KG blade and discing in North Carolina, average runoff increased 345 mm the first year and >200 mm in both the second and third years (table 12). Replanting with grass following those same treatments apparently influenced infiltration and ET substantially, because runoff was 6 to 7.5 times less during those same 3 years on planted plots (Douglass and Goodwin 1980). Employing shearing without discing resulted in runoff values that were between the other two treatments, but generally closer to the lower end values for planted grass.

For considerably less intensive, short-term treatments, annual water yields do not change. In the upper Piedmont of South Carolina, water yields did not change when one low-intensity controlled fire was applied annually for 3 years in each of three pine stands before harvesting (Van Lear and others 1985). However, harvesting coupled with a high-severity burn in Georgia is believed to have increased runoff, even though it was not measured directly (Van Lear and Kapeluck 1989). After treatment the 0.35-ha watershed developed a network of gullies, which acted as channels. These apparently were intercepting and conveying significant amounts of soil water or local ground water or both, because the gully sides were eroding—in part, because of flowing water. The gullies were expected to continue to grow in length and width for several years. Although infiltration also was not measured, the authors discounted the probability that a hydrophobic layer had formed based on other fire/soil research results.

By contrast, very temporary hydrophobic conditions—only a few minutes in duration—were observed during simulated rain applications to plots that were cut and burned (Shahlaee and others 1991). The hydrophobicity was present only when unburned organic material at the soil surface was dry. However, elevated runoff attributable to hydrophobicity was observed only on the steepest plots (30 percent slope) and for only the higher of the two rainfall application rates (~102 mm/hour). Plots with 10- and 20-percent slopes also displayed hydrophobicity, but the runoff from the same rain intensity during the initial period of water repellency was much less. Average depth of runoff across all slopes for a 30-minute period averaged 1.11 mm for high-intensity applications and only 0.78 mm for low-intensity applications (71 mm/hour), and the maximum runoff for any plot was 5.97 mm over 30 minutes. So even with initial hydrophobic conditions, actual runoff volumes were low because the forest floor was not fully consumed by burning.

**Table 12.** Changes to annual stream discharge following site preparation and harvesting in the Piedmont

| Location | Area, aspect, soils | Treatment description | Time period | Discharge change | Reference |
|---|---|---|---|---|---|
| | | | | *mm* | |
| North Carolina [a] | 0.25 to 0.75 ha, aspect not given, sandy clay loams, sandy clays, clay loams | Site preparation using KG blade shearing, windrowing, burning, and disking; 4 replicated watersheds | Year 1<br>Year 2<br>Year 3 | 345<br>242<br>223 | Douglass and Goodwin 1980 |
| | 0.43 to 0.62 ha, aspect not given, sandy clays, sandy clay loams, clay loams, sandy loams | Site preparation using KG blade shearing, windrowing, burning; 4 replicated watersheds | Year 1<br>Year 2<br>Year 3 | 70<br>142<br>71 | |
| | 0.33 to 0.53 ha, aspect not given, clay, sandy clay loams | Site preparation using KG blade shearing, windrowing, burning, discing, planting to grass; 4 replicated watersheds | Year 1<br>Year 2<br>Year 3 | 46<br>40<br>35 | |
| Georgia [a] | 32.4 ha, southwest, loam overlaying sandy loam | Harvest, roller chop twice | Year 1<br>Year 2 | 254<br>126 | Hewlett 1979, Hewlett and others 1984 |

[a] Significance/nonsignificance information was not provided for these sites.

The degree of disturbance similarly influences the extent to which storm hydrographs are affected by treatments. Clearcutting alone increased peakflow by 55 to 60 percent and increased stormflow significantly in South Carolina, but blading the slash increased average peak discharge by 150 percent and doubled average stormflow (table 13). Stormflow volumes before and after the peak increased, but time to peak and event length did not change. Clearcutting with road construction, roller chopping, and machine planting increased stormflow by only 27 percent, but peakflows <1.1 $m^3$/second/$km^2$ increased by 100 percent (Hewlett 1979). Peakflow changes were attributed largely to channel extension during storms by reactivation of old gullies and rills (Hewlett and Doss 1984). Peakflows in wet antecedent conditions were most susceptible to change, increasing as much as 35 to 50 percent during large events (Hewlett 1979), whereas stormflows in moderate-to-dry antecedent conditions were

**Table 13.** Changes in stormflow volumes and peakflow magnitudes to harvesting treatments in the South Carolina Piedmont (Douglass and others 1983)

| | | | Hydrologic change | | |
|---|---|---|---|---|---|
| Area, aspect, soils | Treatment description | Time period | Mean peak discharge | Mean stormflow | Other parameters |
| | | *months* | -------------------*percent*------------------ | | |
| 0.65 and 1.25 ha, aspect not given, sandy loam overlaying clay | Two watersheds with 3 consecutive years of control burns, then clearcut pine, slash left in place | First 21 | 55 to 60 [a] | Increased significantly | Nonsignificant change in time to peakflow and event length |
| 1.1 ha, aspect not given, sandy loam overlaying clay | One watershed with 3 consecutive years of control burns, then clearcut pine, slash bladed off with bulldozer | First 21 | 150 [a] | 100 [a] | |

[a] Indicates a statistically significant change at the alpha level used by the original authors.

most commonly changed (Hewlett and Doss 1984). The percentage of precipitation that became stormflow during the first year after clearcutting, roller chopping, and machine planting was 31 percent, compared to 22 percent during pretreatment (Hewlett and Doss 1984). On four 1-ha watersheds in the upper Piedmont of northern Georgia, clearcutting overstory vegetation after ground herbiciding approximately doubled average stormflow for 2.5 years (Neary and others 1986). Controlled fires applied in the absence of other disturbances or treatments have had little effect on hydrograph responses in South Carolina. Three consecutive years of controlled burning did not change average peak discharges or stormflow (Douglass and others 1983).

## Coastal Plain

The Coastal Plain covers a large area, with a fairly broad range of precipitation and temperature regimes. The Southeastern States tend to have more rain in the growing season than in the dormant season, and the Midsouth States (Arkansas, Texas, Mississippi, and Louisiana) generally are wetter in the dormant season and drier in the growing season (Langdon and Trousdell 1978). Hydrology is expressed in terms of water-table levels or surface flows or both. Water-table measurements are most common in the flatter terrain of the lowlands (Grace and others 2003), although surface flows also can be present—particularly in artificial structures, such as drainage ditches, dikes, and canals with single outlets. Water is present and can be measured using weirs and other devices in these ditches when water tables rise and intersect the bottoms of these structures (Riekerk 1983a). In these situations, outflows are very strongly dependent on precipitation events; so total annual water yields may be close approximations of total annual stormflow. Streamflow, in its traditional sense, primarily occurs in some areas of the Upper Coastal Plain in which more topographic relief exists.

Increases in water-table levels are the most common hydrologic responses reported following harvesting of the forests growing on soils with shallow water tables (Aust and Lea 1992, Bliss and Comerford 2002, Lockaby and others 1997a, Sun and others 2001, Trousdell and Hoover 1955, Williams and Lipscomb 1981, Van Lear and Douglass 1982, Xu and others 2000). Although typically short lived (Lockaby and others 1997c; Xu and others 2000, 2002), average annual water-table increases of at least 100 mm can be expected during the first 2 to 3 years after harvesting or longer (table 14). They are short lived (table 14) because revegetation is very rapid in these warm, long growing seasons (Beasely and others 1986) and reductions in ET control water-table fluctuations (Amatya and others 2006b, Aust and Lea 1992, Riekerk 1989, Xu and others 1999). ET is the dominant output term in the hydrologic budget throughout most Coastal Plain forests, making up 60 to 80 percent of the annual hydrologic budget (Amatya and others 2002, 1996, 1997; Chescheir and others 2003; Skaggs and others 1991; Sun and others 1998). As a result, water-table augmentation from harvesting most often is expressed during the growing season when changes to ET would be most marked (Grace and others 2006; Lockaby and others 1997c; Xu and others 1999, 2000).

Even though ET reductions are responsible for creating postharvest water-table increases, antecedent water-table levels and precipitation characteristics are the most important factors in determining the amount of change that ultimately occurs (Langdon and Trousdell 1978, Williams and Lipscomb 1981). Water-table increases are highest and most easily detectable during dry years or periods when they have space to rise in the soil column (Amatya and others 2006b, Langdon and Trousdell 1978, Riekerk 1989). In four studies in the Lower Coastal Plain of South Carolina in which longleaf (*Pinus palustris*) or loblolly pine and mixed hardwoods were harvested, Williams and Lipscomb (1981) reported that the longleaf and loblolly dominated plots had similar average first-year water-table rises (table 14) after the lightest cuts (18 percent of basal area) and the heaviest cuts (67 percent of basal area). Not much water-table rise was detected because the heavy cut was made when the water table was near the ground surface. Thus, only small rises could occur before the water reached the soil surface and was no longer ground water (Riekerk 1983a, Williams and Lipscomb 1981). The presence of more wet days in winter and early spring of one year resulted in only half the water-table increase (61 mm),

**Table 14.** Average changes to water table elevations following harvesting or harvesting plus site preparation in the Coastal Plain

| Location | Area, aspect, soils | Treatment description | Time period | Average changes to water tables | | Reference |
|---|---|---|---|---|---|---|
| | | | | Annual | Growing | |
| | | | | ------------*mm*------------ | | |
| Lower Coastal Plain South Carolina[a] | Area not given, flat topography, sandy loams over sandy clays | Dry weather harvest, no bedding, plant | First year after cut | 140 | | Xu and others 2000 |
| | | | 1.75 years after cut | 430 | | |
| | | | 2.75 years after cut | 280 | | |
| | | | 3.75 years after cut | 140 | | |
| | | Wet weather harvest, no bedding, plant | First year after cut | 210 | | |
| | | | 1.75 years after cut | 450 | | |
| | | | 2.75 years after cut | 360 | | |
| | | | 3.75 years after cut | 210 | | |
| | | Dry weather harvest, conventional bedding, plant | After bedding | 280 | | |
| | | | Year 1 | 250 | | |
| | | | Year 2 | 130 | | |
| | | Wet weather harvest, conventional bedding, plant | After bedding | 270 | | |
| | | | Year 1 | 280 | | |
| | | | Year 2 | 160 | | |
| | | Wet weather harvest, mole plowing+conventional bedding, plant | After bedding | 270 | | |
| | | | Year 1 | 300 | | |
| | | | Year 2 | 180 | | |
| Lower Coastal Plain South Carolina[a] | Area not given, flat topography, fine sands | Seed tree cut in pine stand 67 percent basal area removed | Year 1 | 100±37[b] | 119±43[b] | Williams and Lipscomb 1981 |
| | | Selection cut in pine stand 18 percent basal area removed | Year 1 | 146±70[b] | 155±116[b] | |
| | | Selection cut in pine stand 56.9 percent basal area removed | Year 1 | 226±46[b] | 171±91[b] | |
| | | Commercial clearcut of pine and mixed hardwoods 41 percent basal area removed | Year 1 | 323±61[b] | 219±88[b] | |
| Coastal Plain North Carolina[a] | 25 ha, flat topography, fine sandy loam | Clearcut pine, site preparation, bedding | Year 1 | 74 | | Amatya and others 2006b |
| | | | Year 2 | 107 | | |
| | | | Year 3 | 146 | | |
| | | | Year 4 | 30 | | |
| | | | Year 5 | 22 | | |
| | | | Year 6 | –8 | | |
| | | | Year 7 | 40 | | |
| | | | Year 8 | –14 | | |
| | | | Year 9 | –50 | | |
| | | | Year 10 | –42 | | |

[a] Significance/nonsignificance information was not provided for these sites.
[b] Plus one standard deviation.

compared to drier conditions of the same time period a year earlier (133 cm) in a harvested and site-prepared watershed in North Carolina (Amatya and others 2006b).

Small-to-moderate water-table rises can result from soil damage, such as compaction and/or rutting by skidder operation in wet conditions, although the changes are usually short lived (Aust and others 1993, 1995; Blanton and others 1998; Grace and others 2007; Perison and others 1997; Xu and others 1999). The average water-table increase (table 14) during the year after harvesting in wet conditions was 210 mm, compared to only 140 mm for dry weather harvesting; but most of the increase was confined to the growing season (Xu and others 2000). The mechanism for water-table increases is typically an increase in bulk density, particularly through losses of larger soil pores, which reduces saturated hydraulic conductivities and drainable porosities and disrupts lateral or vertical subsurface drainage (Grace and others 2007, Skaggs and others 2006, Sun and others 2004). Thinning alone reduced saturated hydraulic conductivities from 100 to 32 cm/hour in an organic soil in North Carolina (Grace and others 2007). Because water drainage or movement is retarded, water-table levels remain elevated (Grace and others 2007, Skaggs and others 2006, Sun and others 2004), at least within the local area of soil damage (Aust and others 1993, 1995). Aust and others (1995) and Xu and others (2000) suggested that better drained soils may be more vulnerable to soil damage than poorly drained soils; so that changes to water-table levels may be much larger on damaged, better drained soils than on damaged, poorly drained soils. However, better drained soils typically have longer periods of drier conditions and shallower damage, making any needed mitigation easier to accomplish (Aust and others 1995).

Some forest management practices have resulted in lowering water-table levels. Both conventional bedding and mole-plow bedding site preparation in poorly drained soils in South Carolina reduced water-table depths by nearly equal amounts (~150 to 180 mm) for about 2 years following site preparation, compared to nonbedded harvested sites (Xu and others 2000). There also was little difference in effects to water-table levels or duration of effects whether the initial harvesting occurred during wet or dry conditions. Lockaby and others (1994, 1997b) observed similar water-table reductions from clearcutting bottomland hardwoods in the Upper Coastal Plain of Alabama using two types of systems—helicopter and feller buncher-skidder. Water-table elevations were significantly lower (for example, ~0.2 m) beneath harvest blocks than outside the harvest boundaries in July; but data were not separated by harvest type, so it is impossible to determine if soil disturbance from the feller buncher-skidder operation influenced the water-table response. In this study, water-table lowering was attributed to increased evaporation caused by increased wind speeds or higher temperatures (or both) in cut areas, even though only modest soil temperature increases of 2 °C to 4 °C have been reported in clearcuts elsewhere in the Coastal Plain (Aust and Lea 1991, Messina and others 1997).

Outflow and streamflow increases after harvesting in the Coastal Plain are related to the amount of forest vegetation harvested (Beasley and others 2000), again because these increases are largely controlled by reductions in ET (Amatya and others 2006b, Riekerk 1989, Sun and others 2000). Neary and others (1982) found that first-year water-yield increases in the Coastal Plain typically were <0.4 mm for every 1 percent of basal area removed. However, at least some of the sites included in that analysis also involved site preparation, which may affect measured changes. It often is difficult to separate harvesting and site preparation effects, especially in the Coastal Plain, because very few harvest-only studies have been conducted. Summer and others (2006) noted that streamflow increased significantly after clearcutting and thinning of the streamside zone in two watersheds in southwestern Georgia, but the amount of increases were not specified. Studies in which harvesting and site preparation are separated sufficiently in time provide evidence that water-yield increases originate primarily from harvesting. For example, Swindel and others (1981) did not observe a secondary increase in outflows after intensive site preparation following mechanized logging. But because harvest-only studies are lacking for the Coastal Plain, it is probably more correct to state that changes in discharges are related to the level of devegetation and site disturbance (Riekerk 1983b).

Clearcutting followed by intensive mechanical site preparation that included shearing on three watersheds in southeastern Arkansas increased first-year water yields by

122 mm (table 15)—a thirteenfold increase (Beasley and Granillo 1988, Grace 2005). Outflow increases did not extend beyond that first year (Beasley and Granillo 1988). Much less intensive selection harvests and deadening of the residual hardwoods on three other watersheds increased average annual water yields fivefold, but the absolute increase was only about 41 mm, which was not significant (Beasley and Granillo 1988, Grace and others 2003). Beasley and others (2000) reported similar first-year increases (120 mm) from harvesting and shearing in eastern Texas; harvesting with roller chopping at the same Texas site resulted in outflows, 57 mm, that were only slightly larger than outflows from harvesting and deadening in Arkansas (41 mm) (table 15). In an analysis of harvesting followed by two levels of site preparation—minimum disturbance (clearcutting pine, roller chopping, bedding, and planting) and maximum disturbance (clearcutting pine, stump removal, burning, windrowing, harrowing, bedding, and planting)—in Florida, the maximum intensity treatment resulted in significant

**Table 15.** Changes to annual outflow or stream discharge following harvesting and site preparation in the Coastal Plain

| Location | Area, aspect, soils | Treatment description | Time period | Outflow or discharge change | Reference |
|---|---|---|---|---|---|
| | | | | *mm* | |
| Lower Coastal Plain, Florida | 67 ha, flat topography, sands overlaying clay | 33-ha clearcut pine, chop, bed, plant (low-disturbance level) | Year 1 | 30 | Riekerk 1989 |
| | | | Year 2 | 40 | |
| | | | Year 3 | 0 | |
| | | | Year 4 | –100[a] | |
| | | | Year 5 | –180[a] | |
| | | | Year 6 | 30 | |
| | | | Year 7 | –130[a] | |
| | 49 ha, flat topography, sands overlaying clay | 36-ha clearcut pine, stump removal, burn, windrow, harrow, bed, plant (high-disturbance level) | Year 1 | 150[a] (150 percent) | |
| | | | Year 2 | –60 | |
| | | | Year 3 | 30 | |
| | | | Year 4 | –130[a] | |
| | | | Year 5 | 80 | |
| | | | Year 6 | 100[a] | |
| | | | Year 7 | 130[a] | |
| West Gulf Coastal Plain, Arkansas | 2.3 to 4.0 ha, flat topography, silt loams, and clays | Three replicates, clearcut mixed hardwoods and pine, shearing, windrowing, burning, hand plant (high-disturbance level) | Year 1 | 122[a] | Beasley and Granillo 1988 |
| | | | Year 2 | 137 | |
| | | | Year 3 | 153 | |
| | | | Year 4 | 120 | |
| | | Three replicates, selective harvest of pine, harvest of all commercial hardwoods, herbiciding all remaining hardwoods, plant (low-disturbance level) | Year 1 | 41 | |
| | | | Year 2 | 28 | |
| | | | Year 3 | –30 | |
| | | | Year 4 | –36 | |
| Coastal Plain, North Carolina[b] | 25 ha, flat topography, fine sandy loam | Clearcut pine, site preparation, bedding | Year 1 | 91 (99 percent) | Amatya and others 2006b |
| | | | Year 2 | 260 (38 percent) | |
| | | | Year 3 | 207 (54 percent) | |
| | | | Year 4 | 98 (13 percent | |
| | | | Year 5 | 56 (10 percent) | |
| | | | Year 6 | –31 (–4 percent) | |
| | | | Year 7 | 8 (18 percent) | |
| | | | Year 8 | 21 (5 percent) | |
| | | | Year 9 | 116 (9 percent) | |
| | | | Year 10 | –2 (–0.5 percent) | |

[a] Indicates a statistically significant change at the alpha level used by the original authors. Unless otherwise indicated, values without an [a] are nonsignificant.
[b] Significance/nonsignificance information was not provided for this site.

first-year outflow increases of 150 mm (150 percent) compared to a nonsignificant 30-mm increase (23 percent) from the minimum intensity treatment (table 15). The maximum treatment left almost no residual vegetation, but the minimum left some intact vegetation and allowed sprouting (Riekerk 1983b). Overall, increases to first-year outflows for the maximum-disturbance watershed were exhibited soon after treatment was completed, and they were well distributed over all seasons and weather conditions. Subsequent changes to outflows from the maximum-disturbance watershed diminished in the second year (Riekerk 1983a, 1983b). By contrast, most of the first-year increase in outflows from the minimum-disturbance watershed was primarily from precipitation during one wet month; other increases that contributed to the first-year augmentation were small, intermittent, and strongly dependent on precipitation and season (Swindel and others 1981, 1982). Lebo and Herrmann (1998) reported that increases in outflows in several drained watersheds in North Carolina lasted only about a year after site preparation—shearing, burning, and bedding—applied within a year of clearcutting the pine overstory. The outflow increases were seasonal—mostly during the summer. The largest summer increases ranged from 70 to 110 mm (56 to 95 percent) but still represented only about 33 percent of precipitation totals for the same time period. Amatya and others (2006b) reported longer lived outflow increases from harvesting followed by site preparation and bedding activities in coastal North Carolina. Increased outflow in a drained watershed was measurable for 4 to 5 years (table 15) until planted regeneration sufficiently reestablished ET rates, which reduced soil moisture storage.

Prescribed burning, regardless of whether it is done before or after harvesting, is the one site-preparation technique that generally has little or no effect on surface flows. One reason may be that controlled burns may not completely combust the organic layer, so soil infiltration rates are retained (Mohering and others 1966, Shahlaee and others 1991). Burning 20 percent of a watershed in the Santee Experimental Forest in South Carolina did not increase streamflow (Amatya and others 2006a). Burning an additional 60 percent of the watershed over the next 3 years also did not increase streamflow. A later prescribed fire covering 84 percent of the watershed was followed by an increase in outflow of 64 percent in the first year and 70 percent in the second year after burning, suggesting a delayed increase in flow from the burn. However, this burn followed salvage harvesting after Hurricane Hugo and understory mowing, so some of the effect may have been caused by the combination of reduced ET from burning understory vegetation and those previous disturbances rather than just the fire (Amatya and others 2006a). Even long-term applications of burning have had limited effects on watershed hydrology. Neither the time required for surface runoff to begin nor the soil infiltration capacity was changed by 20 years of biennial burning on sandy loam plots, or by biennial burning for 10 years, or annual burning for 10 years in silt loams plots supporting longleaf pine (Dobrowolski and others 1992).

Augmentation of streamflow and outflow volumes that result from harvesting and site preparation can increase the number of days in which flow is present in nonperennial systems. In a 23-ha hardwood-dominated clearcut in North Carolina, flow began 2 weeks earlier than in an adjacent control, and the duration of surface flow was extended (Grace and others 2003). Over the 16-month period after clearcutting, the number of days during which streamflow occurred (190 days) was nearly double that of the control (99 days). Little analysis of flow frequencies has been done in the South because surface flows tend to be ephemeral or intermittent at best, and typically storm driven. However, examination of flow frequencies from a study of harvesting plus maximum site preparation (Riekerk 1983b) showed that the resulting water-yield increases, which were only 2.54 mm of daily flow, came primarily from intermediate-sized storms that occurred about 2 percent of the time.

Like overall water yields, storm hydrograph components also are affected differentially by various combinations of harvesting and site-preparation operations. In flatter portions of the Coastal Plain, operations that involved clearcutting, shearing, and windrowing had larger increases in stormflow and peakflow compared to other clearcutting and site-preparation techniques (table 16). In eastern Texas, the first-year increase was 49 L/second for peakflow and 146 mm for stormflow (Blackburn and others 1986), compared to

**Table 16.** Changes in stormflow volumes and peakflow magnitudes to harvesting treatments in the Coastal Plain

| Location | Area, aspect, soils | Treatment description | Time period | Hydrologic change: Mean peak discharge | Hydrologic change: Mean stormflow | Reference |
|---|---|---|---|---|---|---|
| Coastal Plain eastern Texas | 2.57 to 2.72 ha, aspects not given, fine sandy loam overlaying clay | Three replicates, clearcut pine and mixed hardwoods, roller chopping and burning, plant (low-disturbance level) | Year 1 | 14 L/second[a] | 57 mm[a] | Blackburn and others 1986 |
| | | | Year 2 | 1 L/second | 24 mm[a] | |
| | | | Year 3 | 2 L/second | 23 mm | |
| | | | Year 4 | 1 L/second | 21 mm[a] | |
| | | Three replicates, clearcut pine and mixed hardwoods, shearing, windrowing, and burning, plant (high-disturbance level) | Year 1 | 38 L/second[a] | 120 mm[a] | |
| | | | Year 2 | 11 L/second[a] | 38 mm[a] | |
| | | | Year 3 | 10 L/second[a] | 41 mm[a] | |
| | | | Year 4 | 12 L/second[a] | 47 mm[a] | |
| Coastal Plain Virginia | 7.9 ha, southeast, sandy loams | Clearcut pine and hardwoods with no BMPs, herbicide, moderate intensity burn, hand plant | 38 month postharvest | 3 percent | –12 percent[a] | Wynn and others 2000 |
| | | | After site prep. | –6 percent[a] | –31 percent[a] | |
| | 8.5 ha, southeast, sandy loams | Clearcut pine and hardwoods with BMPs, herbicide, moderate intensity burn, hand plant | 38 month postharvest | 15 percent[a] | –21 percent[a] | |
| | | | After site prep. | 4 percent | –6 percent[a] | |
| Gulf Coastal Plain Arkansas | 2.3 to 4.0 ha, flat topography, silt loams and clays | Three replicates, clearcut mixed hardwoods and pine, shearing, windrowing, burning, hand planting loblolly pine (high-disturbance level) | Year 1 | 14.1 L/second[a] | 125 mm[a] | Beasley and Granillo 1983, 1988 |
| | | | Year 2 | 8.5 L/second | | |
| | | | Year 3 | 8.5 L/second | | |
| | | | Year 4 | 4.1 L/second | | |
| | | Three replicates, selective harvest of pine, harvest of all commercial hardwoods, herbiciding all remaining hardwoods (low-disturbance level) | Year 1 | 0 L/second | 41 mm | |
| | | | Year 2 | 2.8 L/second | | |
| | | | Year 3 | 0 L/second | | |
| | | | Year 4 | 1.4 L/second | | |

*continued*

**Table 16.** Changes in stormflow volumes and peakflow magnitudes to harvesting treatments in the Coastal Plain (continued)

| | | | | Hydrologic change | | |
|---|---|---|---|---|---|---|
| Location | Area, aspect, soils | Treatment description | Time period | Mean peak discharge | Mean stormflow | Reference |
| Northern Mississippi [b] | 0.8 ha, northwest, sandy loams, silt loams | Harvest, brush chopping | Year 1<br>Year 2 | | 479 mm<br>316 mm | Beasley 1979 |
| | 1 ha, west, sandy loams, silty clay, silty clay loams | Harvest, shearing and windrowing | Year 1<br>Year 2 | | 422 mm<br>252 mm | |
| | 0.7 ha, west-southwest, sandy loam, silty clay, silty clay loam | Harvest, bedding on the contour | Year 1<br>Year 2 | | 478 mm<br>208 mm | |
| Upper Coastal Plain Mississippi | 0.86 ha, aspect not given, silt loam, sandy loam | Upland hardwoods, slow burned, hand plant pines, deaden overstory >2.54 cm d.b.h. with herbicides | Year 1<br>Year 2<br>Year 3 | 94 L/second[a] (53 percent)<br>38 L/second[a] (31 percent)<br>14 L/second (13 percent) | 31 mm[a] (22 percent)<br>26 mm[a] (22 percent)<br>47 mm[a] (54 percent) | Ursic 1970, 1982 |
| | 0.86 ha, aspect not given, silt loam | Upland hardwoods, slow burned, hand plant pines, deaden overstory >2.54 cm d.b.h. with herbicides | Year 1<br>Year 2<br>Year 3 | 83 L/second[a] (33 percent)<br>24 L/second[a] (15 percent)<br>46 L/second (34 percent) | 69 mm[a] (22 percent)<br>34 mm (16 percent)<br>75 mm[a] (46 percent) | |

BMPs = best management practices; d.b.h. = diameter at breast height.

[a] Indicates a statistically significant change at the alpha level used by the original authors. Unless otherwise indicated, values without an [a] are nonsignificant.

[b] Significance/nonsignificance information was not provided for these sites.

14 L/second for peakflow and 125 mm for stormflow in southeastern Arkansas (Beasley and Granillo 1983, 1988). Peakflows and stormflow volumes remained somewhat elevated for several more years, but these later year increases typically were much less than the first-year increases (table 16). Although the shearing component of site preparation apparently was important to increasing annual yields, windrowing was the most important variable related to the changes in peakflow when windrows were oriented toward the stream (Swindel and others 1983). Presumably the windrows directed surface runoff to the drainages (Riekerk 1989).

By contrast, in steep Coastal Plain terrain (30 to 50 percent slope) various site-preparation techniques produced minimal differences in stormflow (Beasley 1979). Regardless of whether the site preparation involved brush chopping, shearing and windrowing, or bedding on the contour, first-year average increases in stormflow were well over 400 mm, and even second-year values in steep terrain remained above first-year increases in flat terrain (table 16). Topographic influences controlled storm runoff and overrode any effects of site preparation (Beasley 1979).

Herbiciding to kill the overstory in the Upper Coastal Plain in Mississippi (Ursic 1970, 1982) had larger effects on peakflow the first 3 years than did harvests with intensive site preparation (Beasley and Granillo 1983, 1988, Blackburn and others 1986; table 16). However, the watersheds on which the herbicides were applied were very small (each ~0.86 ha), and small watersheds have less moisture storage capacity, particularly if the soils are shallow; this can result in large responses to a given disturbance (Douglass and others 1983, Van Lear and others 1985). Where clearcutting was followed by herbiciding on 7.9- and 8.5-ha watersheds, the peakflow and stormflow responses were small or were (most often) less than predicted (Wynn and others 2000), particularly from the combination of herbiciding and burning (table 16).

Regardless of location, increases in stormflow translate to increases in the percentage of precipitation that becomes stormflow. In flat Coastal Plain areas, 11 to 12 percent of precipitation became stormflow after clearcutting plus shearing, compared to 5 to 6 percent after clearcutting with roller chopping and selective harvesting with herbiciding (Beasley and Granillo 1983, Blackburn and others 1986). In steeper Coastal Plain terrain, 33 to 37 percent of precipitation became stormflow during the first year after harvesting with various mechanical site preparation techniques, and second-year values ranged from 19 to 28 percent (Beasley 1979). In the absence of treatment in all of these Coastal Plain watersheds, only 1 to 3 percent of precipitation became stormflow (Beasley 1979, Beasley and Granillo 1983, Blackburn and others 1986).

# Comparisons among Physiographic Areas

Forest hydrology research related to stream responses from harvesting has been ongoing for at least a half century in parts of the Eastern United States (Ice and Stednick 2004). By contrast, investigations into how wetland water tables are affected by harvesting or similar activities are relatively new, so much less information has been compiled from wetland-dominated watersheds (Shepard and others 1993). One of the oldest wetland studies focusing on harvesting effects on water-table levels in the Eastern United States is from the Marcell Experimental Forest in northern Minnesota (Verry 1981). Studies of wet, flatlands in the South are much more recent—rarely present before the 1980s and increasing markedly beginning in the 1990s (Sun and others 2001).

Even with these vastly different amounts of available information, one characteristic common to both wetland and stream systems is that augmentation of water-table levels and water yields occurs primarily because ET losses from the watershed have decreased. Because forest ET is greatest during the growing season, hydrologic changes caused by reducing vegetation generally are expressed during the growing season. However, changes to streamflow or water-table levels may not be measurable during the growing season if soil moisture deficits are large due to dry antecedent conditions. Most precipitation inputs will go toward fulfilling soil moisture storage needs before water is released to aquifers or streamwater. Conversely, if soil moisture is very high in the growing season and precipitation remains above normal, water yields or water-table levels on harvested and unharvested sites may not differ much, and treatment effects may be undetectable.

Hydrologic changes from treating only a small percentage of the vegetation on a watershed are more difficult to detect than from larger reductions. Overstory vegetation treatments also typically result in larger hydrologic changes than understory removals, probably because the higher leaf area indexes of overstory trees promote faster transpiration and more interception of water. But when understory vegetation comprises a substantial percentage of the basal area removed (Johnson and Kovner 1956, Meginnis 1959), hydrologic changes are observable, although they tend to be much smaller and shorter lived than those occurring with heavy or complete reductions in overstory vegetation. Overall, literature on eastern landscapes most commonly is focused on more intensive harvest and soil disturbance practices, which have the most potential for creating the most extreme hydrologic changes. The vastly larger number of hydrologic studies involving clearcutting or clearcutting+site preparation makes these the most useful for comparing responses across landscapes.

Although data from the North Central States are limited (table 2), they suggest that water-table elevations increase much less in response to harvesting than in the Coastal Plain (table 14); although in the Coastal Plain, additional disturbances associated with site preparation often accompany harvesting. The difference is attributable to the higher ET rates in the Coastal Plain. Net radiation is low in northern latitudes because cold soils act as sinks for heat. Because ET is dependent on net radiation, transpiration rates are lower in the North (Verry 1997), which contributes to smaller water-table changes from harvesting. ET from peatlands in the Marcell Experimental Forest averaged 63 percent (50.5 cm) of precipitation (Verry and Timmons 1982), compared to as much as 60 to 80 percent in the Coastal Plain, and overall rainfall levels tended to be higher in the Coastal Plain (Amatya and others 2002, 1996, 1997; Chescheir and others 2003; Skaggs and others 1991; Sun and others 1998).

In the South, when water-table responses were measured, harvesting almost always led to an increase in water-table elevations. Lockaby and others (1994, 1997b) and Xu and others (2000) were the exceptions to this finding. They reported decreases in water-table elevations in the Coastal Plain, which they attributed to increasing wind exposure and ground temperatures after harvesting. In wetlands of the North Central States, Verry (1981) reported similar decreases in water tables during dry years following harvesting, which was attributed to higher evaporation from increased wind exposure and solar radiation and elevated transpiration by understory vegetation. In wet years, water-table levels could increase because higher precipitation inputs offset any changes in these other losses.

Aspect played a major role in affecting runoff from harvesting only at the Coweeta studies in the Southern Appalachian Mountains. In general, discharges from northern aspects following clearcutting exceeded those found elsewhere in the Eastern United States. However, despite the colder climate and lower ET rates (450 mm) in the Northeastern States (Likens and Bormann 1995) compared to 704 mm in the Southern Appalachian Mountains (Kovner 1957), runoff from whole-tree harvesting (table 3) rivaled some of the more moderate increases associated with northern aspects at Coweeta (table 6). Annual discharges after clearcutting from south-facing watersheds in the southern mountains were similar to those from clearcutting watersheds in the central mountains and clearcutting uplands in the North Central States (tables 6 and 2). Northeastern responses were similar to these levels (tables 2 and 6) only when partial cutting was employed (table 3).

Annual discharges from clearcutting and site preparation in the Ozark Mountains and Ouachita Plateau(which used stormflow totals because the monitored streams are ephemeral) are comparable to those from the Central Appalachian Mountains, south-facing slopes in the Southern Appalachian Mountains, and the North Central States (tables 2, 5, and 6). However, because streamflow comes as stormflow in this area, the increases are expressed during much shorter periods than in the Appalachian Mountains and elsewhere, where the dominant expression of harvest effects is during growing season baseflow. That harvesting effects are expressed over vastly different time periods and during different flow regimes is evident in the magnitude of stormflow responses (table 5) compared to those for the Appalachian Mountains (table 10). Note that the watersheds treated in Ozark and Ouachita studies tend to be much smaller than those elsewhere in mountainous areas; thus, although depth (mm) is comparable across sites, the total annual runoff volumes (L) from the Ozark Mountains and Ouachita Plateau are much smaller.

In the few available Piedmont studies that involved site preparation following clearcutting, annual discharge varied tremendously (table 12). Runoff ranged from values similar to high-end values in the Southern Appalachian Mountains to low-end values reported elsewhere in the Central Appalachian Mountains (table 6). By comparison, the Piedmont is generally more susceptible to streamflow changes from disturbances than the Coastal Plain, even if the disturbance is more extensive in the Coastal Plain. For example, harvesting without site preparation in the Piedmont resulted in first-year flow increases (table 12) that exceeded those with even the most intensive site preparation in the Coastal Plain (table 14). And even though roller chopping is considered less disturbing to a site than shearing (Blackburn and others 1986), first-year increases in water yield in the Piedmont (table 12) were substantially more than those associated with clearcutting and shearing in the Coastal Plain (table 14). The more deeply incised/gullied channels and thinner eroded soils of the Piedmont account for these differences (Hewlett 1979) and probably explain much of the hydrologic variability observed after harvesting across various sites. The contrasting responses between the Coastal Plain and Piedmont provide good examples of how secondary factors—such as physical channel characteristics and land management practices—interact with the primary drivers of hydrologic responses (precipitation, antecedent soil moisture conditions, ET, land cover, and topography) in the Eastern United States to influence outflow and streamflow responses (Amatya and others 2006a, Douglass and others 1983, Grace 2005, Miwa and others 2003, Riekerk 1983b, Young and Klawitter 1968).

# Application to Fuel-Reduction Practices

The vast majority of literature reviewed in this chapter involves activities in which fuel sources were reduced for purposes other than reduction of hazardous fuels for wildfire suppression. However, the results still are applicable to fuel reductions because hydrologic responses are a result of on-the-ground activities, not the purpose of the activities. As noted previously, the majority of available studies have involved harvesting intensities that far exceed what would be done during typical fuel management in forests, with the exception of large-scale salvage harvests. If harvesting follows soon after the event that led to salvage logging, the total change in annual, seasonal, and/or storm hydrology will be similar to what would be expected from clearcutting. If salvage logging is done in stands where much of the overstory is already dead, most of the hydrologic changes will be associated with the decline, not the removal of that dead, standing fuel (Douglass and Van Lear 1983, Van Lear and others 1985). Overstory removal will reduce only the interception component of ET, which has been reported to range between 10 and 26 percent of annual precipitation in eastern landscapes, depending on species and stand age (Helvey 1967, Lull and Reinhart 1966, Swank and others 1972). But because these data are for trees with leaves, the crown condition of the overstory prior to removal (such as salvage logging) will determine the importance of interception. Interception will not go to zero after harvesting, however, because slash on the ground, residual vegetation, and litter all intercept precipitation (Helvey 1967, Lull and Reinhart 1966).

Eastwide, a minimum of 20 to 30 percent of a watershed's basal area must be removed before removals produce measurable changes in annual water yields (Hornbeck and Kochenderfer 2001, Hornbeck and others 1997). Fuel reductions for the sole purpose of fire suppression (other than salvage logging) normally would affect a small percentage of basal area in a watershed and be widely dispersed, thereby retaining a substantial proportion of antecedent interception and transpiration from adjacent vegetation (Lull and Reinhart 1966). Therefore, little change in hydrologic response would be expected in most situations, and changes that did occur should be short lived, particularly in forests of the South, as changes there usually last only a year or two. This is fortuitous, because the Coastal Plain and lower Piedmont are the areas where fuel-reduction activities most likely may be a regular part of land management activities because wildfire regimes are more frequent there than in other eastern landscapes (Van Lear and Harlow 2002).

Overall, where hydrologic responses of prescribed fires have been studied in the Eastern United States, they have resulted in little effect to hydrology. Low fire intensity may be partially responsible for the lack of hydrologic response (Cushwa and others 1970, Mohering and others 1966, Shahlaee and others 1991), but fire also can stimulate herbaceous growth and seed production (Lewis and Harshbarger 1976), which can quickly restore litter to the soil surface and promote root growth. However, high severity controlled burns can affect hydrology. Changes most commonly result from reductions in soil infiltration and soil moisture storage when the litter and duff layer are completely combusted and soil becomes exposed (Wells and others 1979). Reductions in infiltration rates in the Eastern United States appear to be caused primarily by pore clogging from fine soil particles once soil is exposed (Arend 1941, Wells and others 1979) rather than by physicochemical changes to soil that result in water repellency (DeBano 1966); this is because hydrophobicity is rarely reported and very short lived in the Eastern United States.

Particular care should be taken when burning in the Piedmont, as this area is perhaps the most susceptible to major hydrologic changes from soil disturbance. Relatively dry soils from warm temperatures, coupled with thin organic layers overlaying thin soils, can make this area more susceptible to gullying and erosion than the steeper areas that are typically thought to be highly erodible (Van Lear and Kapeluck 1989). Gullies can change hydrologic responses and increase runoff in the long term. In both the Piedmont and Coastal Plain, special care also should be undertaken when applying practices that increase fuel loads on the soil surface before burning or that increase soil temperatures during burning. Practices—such as felling and burning, or shearing and burning—increase the fuel load in contact with the soil surface. Likewise, windrowing or piling concentrates fuels, so that burning them produces much higher soil temperatures than burning dispersed materials (Cromer and Vines 1966, Robert 1965, Well and others 1979). These activities increase the probability that soil will be negatively affected and hydrology changed.

It is clear from the studies reviewed in this chapter that antecedent soil conditions and the degree of soil disturbance or damage can play an important role in controlling hydrologic responses. Therefore, fuel management plans should consider those factors when estimating potential hydrologic changes. Because fuel reduction activities typically can be planned and applied during more appropriate conditions compared to wildfire suppression, it should be possible to keep most soil disturbance at or below acceptable levels.

Soil disturbance by new fire line construction may be one of the biggest long-term impacts of fuel-reduction activities. Hand-constructed firebreaks will have little if any effect because litter can quickly be restored to the surface from wind action or annual leaf fall or both. Soil infiltration rates also should not be substantially affected by hand-constructed fire lines. By contrast, mechanically constructed fire lines such as bulldozed lines are more like roads, or at least skidroads, and may have some of the same potential effects—such as intercepting subsurface flows, increasing bulk densities and reducing soil hydraulic conductivity, concentrating overland flow, and diverting overland flow to streams. Although fire lines lack the repeated trafficking that roads have and tracked equipment that often is used to construct fire lines exerts lower pressure compared to wheeled equipment, the largest proportion of soil compaction occurs after just a few equipment passes (Jansson and Johansson 1998, McNabb and others 2001, Wang and others 2005). As a result, significant compaction and other changes to soil physical properties can occur during fire line construction. Fire lines often are subjected to all-terrain and other vehicle use during prescribed fires, which can result in further compaction. Therefore, the same care needed for planning and constructing and closing roads should be used for fire line construction. Appropriate best management practices also should be applied, particularly those that focus on proper location, water control, and soil protection and coverage.

From the perspective of cumulative watershed effects, the influence of fuel-reduction activities on hydrology probably will be small if the landscape is reforested and not converted to another use. The primary hydrologic cumulative effect from harvesting that has been raised as a possible concern is downstream flooding, which results from simultaneous accumulation of large volumes of water from upstream sources (Hewlett 1982). But even in watersheds where vegetation removal has been substantial, stormflow volumes from each subwatershed are desynchronized, thereby reducing the risk of downstream flooding (Hewlett and Doss 1984). Furthermore, most of the hydrologic change from harvesting anywhere in the Eastern United States occurs during growing seasons or low flows (or both), when flooding is least likely to occur. Consequently, the overwhelming consensus within the scientific literature is that contemporary forest management practices do not increase the risk of downstream flooding (Hewlett 1982, Hewlett and Doss 1984, Hornbeck and others 1997, Rogerson 1976, Verry 1972, Woodruff and Hewlett 1970).

# Research Needs

In this chapter, studies involving harvesting or other types of vegetation reductions have been used as a proxy for understanding how hydrology might change from fuels reduction practices in the Eastern United States. This approach was needed because information specifically pertaining to fuels reduction is largely missing from the literature. Most of the available investigations have involved much larger reductions of ET than would occur for controlling fuels, so we predominantly have information about upper end or “worst case” effects. However, from the standpoint of being able to accurately describe and disclose expected effects in environmental documents required by the National Environmental Policy Act and other legislation, studies are needed that specifically focus on fuel-reduction activities and their effects on soil and water resources. The public would be better served if the data used in these environmental documents directly applied to the proposed activities, so that direct and cumulative effects could be more accurately evaluated.

Furthermore, our knowledge about the effects of controlled burns is extremely limited, despite the fact that burning is becoming an increasingly used management tool. Controlled burns in forests usually are applied to reduce dead, downed fuels and possibly to reduce the density of understory brush, while limiting the damage to standing trees (Biswell 1975). The intensities and severities of burning to control only understory fuels may be quite different from those associated with fell-and-burn activities or postharvesting site preparation; if so, the effects would likely be different. However, until a body of scientific evidence shows that the effects from understory burning are small, it is not appropriate simply to make that assumption based on current limited data; the effects or lack thereof should be determined in replicated studies. It is now particularly important to perform these types of studies for several reasons: there is new interest in employing controlled burns during the growing season (Outcalt and others 2006) when soil moisture is lower and potential effects on soil condition and hydrology may be greater than the traditional application of fires during the dormant season; repeated burning is being used or considered for a variety of uses (Bowles and others 2007); and burning is being considered for application where it has long been excluded, which can result in severe initial burns (Knapp and others 2007). These new applications may have effects that are measurably different from what one might expect based only on currently available, sparse datasets.

# Conclusions

The initial foundation of what we know about forest management effects on water balance and overall hydrologic expression in the Eastern United States comes primarily from studies at Coweeta Hydrologic Laboratory and Fernow Experimental Forest. These sites have the most comprehensive sets of long-term hydrologic studies related to vegetation management, including a variety of low-intensity vegetation removals that have not been performed elsewhere but are applicable to fuel-reduction activities. However, substantial data also have been collected from other sites and provide additional, valuable information to further complete the current base of knowledge.

Although biological, physical, and climate conditions are quite varied throughout the Eastern United States, the similarity of results among study sites is striking. In general, water-yield increases from reducing vegetation do occur when the level is >20 to 25 (approximately) percent of the watershed basal area. Larger percentages of basal area removal result in proportional increases in annual water yield, but they primarily augment low and moderate flows. Water-yield changes from reducing vegetation typically are short lived, although retarding vegetative regrowth mechanically or chemically prolongs the time during which yields are elevated. Storm hydrograph components also can change, but these are primarily associated with small and moderate-sized runoff events. Aspect is important in controlling total annual yields only in mountainous areas that are steep and have great relief. Aspect becomes unimportant in mountains that are less steep or forests with lower topographic relief. The timing of the spring snowmelt hydrograph can be changed by varying the size of harvested sites and the character of the opening and associated regeneration. Wetland soil characteristics in both the most northern or southern landscapes play a large role in controlling how hydrologic responses will be expressed in flatlands: whether hydraulic conductivities are rapid or slow largely determines the degree of influence of ET on water-table levels. On steep hillsides, the extent of water delivery to channels is at least partially dependent on the characteristics of the channel network, such as density, length, and degree of incision. These and other commonalities among vastly different physiographic areas illustrate the broad transferability and application of findings, particularly once one adjusts for differences in precipitation, climate, topography/relief, soils, and species composition.

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Cumulative Watershed Effects of Fuel Management in the Eastern United States

CHAPTER 12.

# Effects of Fire and Fuels Management on Water Quality in Eastern North America

**R.K. Kolka**

## Introduction

Fuels management, especially prescribed fire, can have direct impacts on aquatic resources through deposition of ash to surface waters. On the terrestrial side, fuels management leads to changes in vegetative structure and potentially soil properties that affect ecosystem cycling of water and inorganic and organic constituents. Because surface water systems (streams, lakes and wetlands) are tightly linked to terrestrial systems, these changes in the terrestrial system can also impact surface waters.

Notable reviews of fire effects on water have been conducted at the North American scale (Tiedemann and others 1979, Neary and others 2004), however, these reviews have been mainly focused on the western U.S. and Canada where research has historically been the most prolific (see Stednick 2006 for a Western Synthesis). Still, a number of studies have assessed the influence of fuels management or wildfire on various water quality parameters across Eastern North America (table 1). Because fuels management is an important component to pine management in the Southeast, more research has been conducted in the Southeast than the Midwest, Northeast, and Eastern Canada.

Prescribed fire and mechanical approaches to fuels management (such as precommercial thinning) are used quite extensively in certain parts of the Eastern United States. Although some research has been conducted on the effects of fire on water quality (both prescribed fire and wildfire), little has been conducted on the effects of mechanical treatments. Other fuels management approaches such as herbicide and other chemical applications and biological treatments such as grazing are also practiced in the East, but again little relevant research has been conducted to assess impacts to surface waters.

Although wildfires tend to burn more extensive areas, burn hotter, and consume more fuel than prescribed fires, the effects on surface waters can be analogous to prescribed fire. Many prescribed fires, especially in the South, are intended for site preparation rather than fuels reduction. In this chapter, we review responses of surface water quality to all prescribed fire—independent of intent—and wildfire.

## Fire Effects on Hydrology

Either because of increased flows resulting from lower interception and transpiration or because of soil hydrophobicity, the potential exists for higher surface and subsurface runoff following fire. Increased surface runoff and higher instream flows increases the

**Table 1.** Effects of fire (prescribed fire or wildfire) on hydrology and water chemistry in Eastern North America: cations, anions, total suspended sediment (TSS), nitrate ($NO_3$), sulfate ($SO_4^{2-}$), nitrogen (N), total nitrogen (TN), phosphorus (P), total phosphorus (TP), potassium (K), mercury (Hg), calcium (Ca), chlorine (Cl) dissolved inorganic carbon (DIC), dissolved organic carbon (DOC), acid neutralizing capacity (ANC), and acidity or basicity (pH)

| Location | Fuels altering event | Aquatic system | Parameters | Findings | Reference |
|---|---|---|---|---|---|
| Tennessee, Georgia | Prescribed fire | Streams | Cations, anions, TSS, pH | No differences found following fire | Elliot and Vose 2005 |
| Western North Carolina | Prescribed fire | Streams | $NO_3$ | Streams with autumn burns showed increases in $NO_3$, with increases persisting for <1 year; streams with spring burns showed no increase | Vose and others 2005 |
| Northwestern Ontario | Wildfire | Streams | Bedload | Bedload increased 20 fold following fire, with recovery in 5 to 6 years | Beaty 1994 |
| Western North Carolina | Prescribed fire | Streams | $NO_3$ | Elevated in one burned stream for 6 months | Knoepp and Swank 1993 |
| Northwestern Ontario | Wildfire | Streams | N and P species | Fluxes of most N species and fractions increased and remained elevated up to 9 years following fire; effects on P flux were short term | Bayley and others 1992a |
| Northwestern Ontario | Wildfire | Streams | Cations, anions, DIC ANC, pH | Increases in concentrations and fluxes of anions and cations with an overall increase in stream acidity and decrease in pH 2 years following fire | Bayley and others 1992b |
| Western South Carolina | Prescribed fire | Streams | Sediments, nutrients, cations | No differences found following fire | Van Lear and others 1985 |
| Western South Carolina | Prescribed fire | Streams | Nutrients, cations | No differences found following fire | Douglass and Van Lear 1983 |

*continued*

**Table 1.** Effects of fire (prescribed fire or wildfire) on hydrology and water chemistry in Eastern North America: cations, anions, total suspended sediment (TSS), nitrate ($NO_3$), sulfate ($SO_4^{2-}$), nitrogen (N), total nitrogen (TN), phosphorus (P), total phosphorus (TP), potassium (K), mercury (Hg), calcium (Ca), chlorine (Cl) dissolved inorganic carbon (DIC), dissolved organic carbon (DOC), acid neutralizing capacity (ANC), and acidity or basicity (pH) (continued)

| Location | Fuels altering event | Aquatic system | Parameters | Findings | Reference |
|---|---|---|---|---|---|
| Eastern South Carolina | Prescribed fire | Streams | Cations, anions | No differences found following fire | Richter and others 1982 |
| Western South Carolina | Wildfire | Streams | Nutrients, cations, TSS | Only difference was elevated $NO_3$ in the first year | Neary and Currier 1982 |
| Northwestern Ontario | Wildfire | Streams | N, P, K | Increases in concentrations and fluxes of N, P, and K at least 3 years following fire | Schindler and others 1980 |
| Minnesota | Wildfire | Surface runoff, streams/lakes | Cations, N, P, pH condition | Differences in fluxes for 2 years following fire, no differences in concentrations | McColl and Grigal 1977 |
| Minnesota | Wildfire | Surface runoff, streams/lakes | P | Increases in P fluxes in surface runoff 1 year following fire, no other differences | McColl and Grigal 1975 |
| Western South Carolina | Prescribed fire | Surface runoff | Sediments | Increase in sediment 40-fold for low-severity and high-severity burns for 1 year | Robichaud and Waldrop 1994 |
| Louisiana | Prescribed fire | Surface runoff | Sediments | Small short-term effects on interrill erosion following biennial fires | Dobrowolski and others 1992 |
| Western South Carolina, Piedmont Georgia | Prescribed fire | Surface runoff | Sediments | Low sediment production from low-severity burns; high production from high-severity burns | Van Lear and Kapeluck 1989 |
| Western South Carolina | Prescribed fire | Surface runoff | Sediments | No differences found between burned and unburned clearcut plots | Van Lear and Danielovich 1988 |

*continued*

**Table 1.** Effects of fire (prescribed fire or wildfire) on hydrology and water chemistry in Eastern North America: cations, anions, total suspended sediment (TSS), nitrate ($NO_3$), sulfate ($SO_4^{2-}$), nitrogen (N), total nitrogen (TN), phosphorus (P), total phosphorus (TP), potassium (K), mercury (Hg), calcium (Ca), chlorine (Cl) dissolved inorganic carbon (DIC), dissolved organic carbon (DOC), acid neutralizing capacity (ANC), and acidity or basicity (pH) (continued)

| Location | Fuels altering event | Aquatic system | Parameters | Findings | Reference |
|---|---|---|---|---|---|
| Wisconsin | Prescribed fire | Surface runoff | Sediments | No differences found following fire | Knighton 1977 |
| South Carolina | Prescribed fire | Surface runoff | Nutrients, cations | Few differences, “no firm conclusions” following fire | Lewis 1974 |
| Quebec | Wildfire | Lakes | Nutrients, cations anions, DOC, ANC | Increases in concentrations of $NO_3$, TP, Ca, K, $SO_4^{2-}$, and Cl 1 year following fire; most still above reference after 3 years | Carignan and others 2000 |
| Quebec | Wildfire | Lakes | Hg (in fish), TP, TN, Ca, $SO_4^{2-}$, DOC pH, ANC, Chlorophyll-a | TP, TN concentrations higher in burned lakes, Hg in fish, Ca, and DOC no different | Garcia and Carignan 2000 |
| Quebec | Wildfire | Lakes | Nutrients, cations, anions, DOC | Increases in K, TN, TP, Mg, $NO_3$, and $SO_4^{2-}$ export rates following fire; rates highest 1 year following fire but still above reference after 3 years | Lamontagne and others 2000 |
| Minnesota | Wildfire | Lakes | Cations, anions, ANC, pH, condition, Chlorophyll-a | Small increases in Ca and K 1 year following fire | Tarapchak and Wright 1986 |
| Minnesota | Wildfire | Lakes | Cations, P | Increases in P and K but “minimal impacts” following fire | Wright 1976 |
| Southwestern Georgia | Prescribed fire | Wetlands | Nutrients, DOC, DIC, ANC, pH | Increases in pH, alkalinity, DIC, DOC, and $NH_4$ 1 month following fire | Battle and Golladay 2003 |

potential for higher sediment production following fire. Flows are expected to increase, depending on the fire's severity and the extent in the watershed (Baker 1988, Gresswell 1999), but little about effects on water yield or sediment production, especially in the East.

### *Water Yield*

Van Lear and others (1985) reported no increases in streamflow following low-intensity prescribed fire in South Carolina, but increases in runoff were reported for other prescribed fire studies in South Carolina (Robichaud and Waldrop 1994), Louisiana (Ursic 1970), Georgia (Battle and Golliday 2003); Battle and Golliday (2003) reported higher water levels in wetlands; and several wildfire studies reported higher water levels of lakes in Minnesota (McColl and Grigal 1975, Wright 1976). The Minnesota study estimated a 60 percent increase in water yield following wildfire (Wright 1976), and a study in Ontario indicated similar increases of 60 to 80 percent (Schindler and others 1980) with levels that remained above normal for up to 5 years following fire.

Studies on hydrophobic soils are not common in the Eastern United States, although they have been assessed in Wisconsin (Richardson and Hole 1978), in the Upper Peninsula of Michigan (Reeder and Jurgensen 1979), and on the Georgia Piedmont (Shahlaee and others 1991). In the Michigan study, the authors concluded that water repellency following fire was not an important long-term management issue (Reeder and Jurgensen 1979) although studies in Georgia indicated slight hydrophobicity following prescribed fire.

In general, low-intensity prescribed fires appear to produce to little or no additional increases in flows. However, as prescribed fires intensify and consume more forest floor and vegetation layers, possibly including the canopy, effects would be comparable to wildfires or forest harvesting (Baker 1988).

### *Sediment Production*

As noted above, little work has been done in the East on the effects of fire on sediment production or total suspended sediment. From the few studies that do exist, prescribed fire—or wildfire as reported by Neary and Currier (1982)—in the East does not appear to alter infiltration or percolation rates or lead to significant increases in surface runoff; and, hence, will not lead to higher sediment transport or more suspended sediments in surface waters (Elliot and Vose 2005, Knighton 1977, Swift and others 1993, Van Lear and Danielovich 1988, Van Lear and others 1985). Studies in Louisiana that have prescribed burned on a biennial basis for 20 years indicate short-term increases in sediment produced through interrill erosion on irrigated runoff plots (Dobrowolski and others 1992). The caveat is that all of these studies are results from prescribed burns, which tend to be less destructive to upper soil layers, forest floor, and vegetation than wildfires. Studies of a wildfire in Ontario indicate that bedload sediment production increased 20-fold with those increases persisting for 5 to 6 years (Beaty 1994). A high severity prescribed fire (similar in impact to a wildfire) in South Carolina led to 40-fold increase in sediment production compared a low severity prescribed fire (Robichaud and Waldrop 1994). Similarly, a high severity prescribed fire on the Georgia Piedmont led to high losses of sediment the first year following fire (Van Lear and Kapeluck 1989). Other studies in the West indicate that fire, especially severe fires, can have dramatic impacts on sediment production (Gresswell 1999).

## Fire Effects on Water Chemistry

A number of studies in the Eastern United States have assessed the effect of fire on nitrogen, phosphorus, and cation concentrations in surface waters. Fewer have assessed the effect of fire on nutrient fluxes.

## Nitrogen

Total nitrogen, organic nitrogen, nitrate and ammonium have been measured on a number of studies to assess the effects of fire on nitrogen cycling and fluxes to surface waters. In stream systems, studies in western South Carolina found no change in either nitrate or ammonium concentration or flux following prescribed burning (Douglass and Van Lear 1983). In other South Carolina prescribed fire studies, Lewis (1974) also found no difference in surface runoff nitrate between burned and control areas and Richter and others (1982) found no change in volume-weighted concentrations of total nitrogen, nitrate, and ammonium. Similarly, Elliot and Vose (2005) found no differences in stream nitrate and ammonium concentrations in southeastern Tennessee and northern Georgia. However, in another western South Carolina study, Neary and Currier (1982) found elevated nitrate (300 percent), but similar ammonium concentrations in streams the first year following wildfire. Vose and others (2005) found that following prescribed burning conducted in the autumn, two streams had increases in nitrate concentrations with increases persisting for <1 year, compared to no increases for two streams with spring burns. Similarly, Knoepp and Swank (1993) found that stream nitrate increased about 300 percent for some six months following prescribed burning in western North Carolina. After a wildfire in Minnesota, McColl and Grigal (1977) found no differences in surface-runoff total nitrogen or nitrate, but they did see increases in fluxes (about 150 to 200 percent) of both in the first 2 years. In northwestern Ontario, Bayley and others (1992a) found increases in nitrate (about 300 to 800 percent), ammonium (about 150 to 200 percent), suspended nitrogen (about 150 to 200 percent), total dissolved nitrogen (about 150 to 200 percent) and total nitrogen concentrations (about 150 to 200 percent) after two wildfires in the same watershed (6 years apart); after the second fire levels remained elevated for 9 years. Fluxes followed similar patterns (Bayley and others 1992a). In southwestern Quebec, Lamontagne and others (2000) estimated that watershed export rates to lakes of total nitrogen and nitrate were elevated the first year following wildfire and were still elevated 3 years later.

Nitrogen concentrations in northern Minnesota lakes gave no indication of elevated fluxes following prescribed fire (Tarapchak and Wright 1986, Wright 1976). In southwestern Quebec, Carignan and others (2000) found total organic nitrogen and ammonium concentrations doubled, and nitrate concentrations were up to 6000 percent higher in lakes present in watersheds with wildfire compared to lakes in watersheds that were unburned. The increases persisted for up to 3 years. Studies in depressional wetlands in southwestern Georgia indicate increases in ammonium but not for nitrate the first 2 years following prescribed fire (Battle and Golladay 2003).

The solubility of nitrogen species and volatilization of nitrogen from consumed plants and soils during fire could explain why nitrogen species generally do not respond or respond only shortly after fire. Although considerable nitrogen is lost to volatilization during fire (McRae and others 2001), the ash left behind is also concentrated in nitrogen—which quickly succumbs to nitrification processes and becomes available to leaching through forest soils (Knighton 1977). Overall, the preponderance of data suggests little influence of fire on nitrogen; and where differences exist, they usually do not persist more than 1 to 3 years, unless on shallow soils like those found on the Boreal Shield (Bayley and others 1992a).

## Phosphorus

Phosphorus is generally the limiting nutrient in surface waters, and excess phosphorus can lead to eutrophication of lakes, wetlands, and streams (Smith 2003). Following a disturbance such as fire, the largest fraction of phosphorus entering surface waters is typically associated with upland sediment sources (Prepas and others 2003). Total phosphorus is typically measured on unfiltered samples and comprises dissolved phosphorus and phosphorus suspended in sediment. Soluble reactive phosphorus, generally considered to be the same measure as ortho-phosphorus, is the inorganic phosphorus that passes through a filter, usually 0.45 µm. Soluble reactive phosphorus and ortho-phosphorus are considered the active form of phosphorus available for uptake.

Total phosphorus, ortho-phosphorus and soluble reactive phosphorus have been measured in streams, lakes, and wetlands following fire in the Eastern United States Because phosphorus is generally bound to particulates, similar results exist for the transport of total phosphorus and phosphorus suspended in sediment. Numerous studies have found no stream response of phosphorus to prescribed fire—or wildfire as reported by Neary and Currier (1982)—including those in southeastern Tennessee and northern Georgia (Elliot and Vose 2005), western South Carolina (Douglass and Van Lear 1983, Van Lear and others 1985), and eastern South Carolina (Richter and others 1982). Lewis (1974) also found no increases in phosphorus in surface runoff following prescribed fire in South Carolina. McColl and Grigal (1975) found no increases in stream phosphorus following wildfire in Minnesota, but they did see a 300-percent increase in phosphorus in surface runoff the first year following fire. Total, suspended, and dissolved phosphorus concentrations and fluxes in streams increased 140 to 320 percent the first 2 years following wildfire in northwestern Ontario (Schindler and others 1980), but these increases did not persist even after a second wildfire in the same area (Bayley and others 1992a).

Although phosphorus concentration did not differ on burned watersheds in northern Minnesota lakes when compared to a lake in an unburned watershed (McColl and Grigal 1975, Tarapchak and Wright 1986), estimated fluxes to burned lakes increased by 93 percent the first year following fire (Wright 1976). In Quebec, lakes in burned watersheds had 200 to 300-percent higher total phosphorus concentrations and 150 to 200-percent higher flux rates than lakes that were in unburned watersheds, with increases persisting for at least 3 years (Carignan and others 2000, Lamontagne and others 2000). Studies in depressional wetlands in southwestern Georgia indicate no differences in soluble reactive phosphorus concentration the first 2 years following prescribed fire (Battle and Golladay 2003).

Similar to nitrogen, phosphorus does not appear to be a major water quality concern following fire (prescribed or wildfire) in the East, unless located on shallow soils such as those found on the Boreal Shield. Even where shallow soils exist, the bulk of the data suggests that impacts are relatively short term.

## Cations

Because cations (calcium, magnesium, sodium, and potassium) are concentrated in ash, the potential exists for these nutrients to be transported via surface runoff or easily leached through soils following fire. Studies in the South indicate no differences in surface runoff or stream cation concentration following fire (Douglass and Van Lear 1983, Elliot and Vose 2005, Lewis 1974, Neary and Currier 1982, Richter and others 1982, Van Lear and others 1985). Wildfires in northern Minnesota, Ontario, and Quebec indicate short-term increases in cation concentrations and fluxes.

In northern Minnesota, lake concentrations of calcium and potassium increased following wildfire (Tarapchak and Wright 1986). For the same fire, Wright (1976) showed ≤265 percent increase for potassium in runoff; for the first 2 years, McColl and Grigial (1977) showed increased calcium, magnesium, and potassium in surface runoff but increases in streams were limited to potassium. Similarly, potassium fluxes in streams following wildfire in northwestern Ontario were 140 to 290 percent higher than those prior to fire (Schindler and others 1980), with calcium (190 percent), magnesium (190 percent) and sodium (170 percent) increasing as well (Bayley and others 1992b). In Quebec, potassium concentrations increased ≤600 percent in lakes on burned watersheds, compared to 200 to 400 percent for calcium and magnesium (Carignan and others 2000); levels stayed elevated for 3 years following wildfire. In the same set of watersheds, exports rates estimated for potassium (300 to 700 percent), calcium (200 to 300 percent) and magnesium (200 to 300 percent) were higher in burned watersheds than unburned watersheds the first 3 years following wildfire, steadily decreasing with time (Lamontagne and others 2000).

Similar to the effects on nitrogen and phosphorus, prescribed fires do not appear to have a dramatic influence on the concentration and transport of cations in the South. However, for wildfires in the North, some cation concentrations and fluxes (especially potassium) increase in streams and lakes following fire and those increases can persist for 3 years or more.

## Carbon

Interest in effects on ecosystem carbon has increased over the past 15 to 20 years because of the implications for climate change. Fires have been shown to be large sources of carbon dioxide (Amiro and others 2001); for example vegetation is about 50 percent carbon, leaf litter about 50 percent, surface mineral soils about 1 to 8 percent, and organic soils about 20 to 95 percent. Little work has been done to assess the effects of fire on the concentration or transport of water-soluble carbon, otherwise known as dissolved organic carbon. Dissolved organic carbon is operationally defined as the carbon that passes through a filter, usually 0.45 or 0.7 µm, and is considered mobile in water. Research in Quebec showed no effect of wildfire on lake dissolved organic carbon concentrations (Carignan and others 2000) or export rates to those lakes (Lamontagne and others 2000) following fire. Similarly, Battle and Golladay (2003) found no difference in dissolved organic carbon the first month following prescribed fire in Georgia wetlands in 2000, but did find significantly higher dissolved organic carbon following prescribed fires conducted in 2001. They suggest that field conditions are very important in determining fire's effect on the generation of dissolved organic carbon (Battle and Golladay 2003). No other studies from Eastern North America were found that assessed the effect of fire on dissolved organic carbon transport. The paucity of data makes generalizations difficult, but based on these few studies, fire does not appear to dramatically affect dissolved organic carbon concentration or transport.

## Mercury

Mercury is of great concern in the environment because it biomagnifies up the food chain in aquatic ecosystems (U.S. Environmental Protection Agency, Office of Research and Development 2002). Although we are beginning to understand the cycling of total mercury and methylmercury (bioaccumulative form) in forested watersheds (Hintelmann and others 2002, Kolka and others 2001), little work has been done understanding the role of fire in mercury cycling. Nearly 100 percent of mercury stored in plant-derived fuels is emitted into the atmosphere, 85 percent of which is elemental mercury and 15 percent particulate mercury (Friedli and others 2003). Newly released elemental mercury enters the global cycle whereas particulate mercury has the potential to be redeposited locally during the fire event. Soils are also sources of mercury during fires. Studies indicate that upper soil layers experience significant decreases in mercury following fire (Amirbahman and others 2004, Dicosty and others 2006). Zooplankton and northern pike (*Esox lucius*) in lakes on burned Quebec watersheds showed no significant difference in mercury concentrations compared to lakes in undisturbed watersheds, although average fish concentrations were about 160 percent higher in burned lakes (Garcia and Carnignan 1999, Garcia and Carnignan 2000). Although somewhat outside the geographic scope of this chapter, a Canadian study of a wildfire in Alberta found elevated methylmercury in lake and stream water following fire (Kelly and others 2006). Although this study suggests that the dynamics that increase nutrients and affect on the food chain are complex, Kelly and others (2006) did find higher mercury (500 percent) in rainbow trout (*Oncorhynchus mykiss*) in burned watersheds than in unburned watersheds. In an Alberta study, few differences were found in aquatic biota when comparing lakes in burned watersheds to ones in unburned watersheds, with even short-term (three month) decreases in mercury content of aquatic biota following fire (Allen and others 2005). Based on what little data we have, fire does not appear to affect mercury cycling and bioaccumulation in the aquatic food chain but further investigation is needed.

### *Other Water Constituents*

Some of the studies discussed above have measured other various ions such as sulfate, chloride, dissolved inorganic carbon, acidity or basicity (pH), alkalinity, conductivity, and chlorophyll-a. Richter and others (1982) found no differences in sulfate, chloride or alkalinity concentrations following prescribed fire in South Carolina. Similarly, no differences were found in acidity or sulfate concentrations in northern Georgia and southeastern Tennessee following prescribed fire (Elliot and Vose 2005). After a month, water in depressional wetlands in burned watersheds had higher pH (indicating less acidity) and alkalinity (ability to neutralize acids) that those in unburned Georgia watersheds (Battle and Golladay 2003). Studies in northern Minnesota indicate little to no differences in lake pH, alkalinity, and conductance following wildfire but did see an apparent decrease in chlorophyll-a (Tarapchak and Wright 1986). Studies in Ontario indicate decreases in stream pH and concomitant increases in concentrations and fluxes of sulfate and chloride, leading to lower alkalinity for 2 years following wildfire (Bayley and others 1992b). Research on lakes in Quebec indicated no difference in lake alkalinity but considerably higher sulfate, chloride, and chlorophyll-a concentrations persisting 3 years after wildfire (Carignan and others 2000). Not surprisingly, export rates from drainage areas for these lakes were also high for sulfate and chloride (Lamontagne and others 2000).

## Effects of Mechanical, Chemical, and Biological Treatments

Although mechanical, chemical, and biological fuels treatment are non uncommon in Eastern North America, we found no studies that have specifically addressed the effects of these treatments on water quality. However, numerous studies and a number of reviews have examined mechanical, chemical, and biological approaches for vegetation management.

Certainly mechanical fuels treatment is similar to other types of vegetation management or site preparation practices. A number of papers that evaluate water-quality responses to vegetation management or site preparation are available for those planning mechanical approaches to fuels treatment (Binkley and Brown 1993, Dissmeyer 2000, Grace 2005, Shepard 1994, Thornton and others 2000).

Chemical treatments, predominantly herbicides for the purposes of this chapter, are typically used to control competing vegetation. Chemical approaches to fuels management would likely have similarly impacts on water quality as those used for vegetation management. Several papers that review water-quality responses to chemical application are available for those planning chemical approaches to fuels management (Dissmeyer 2000, Larson and others 1997, Micheal and Neary 1993, Neary and others 1993).

Few studies have assessed biological approaches to forest vegetation management, especially in Eastern North America. The most common biological controls for plants are predation by insects or fungi or grazing by domesticated ungulates such as cows (*Bos taurus*) or goats (*Capra* app.). Although considerable research has been conducted on the biological control of invasive plant species, Markin and Gardner (1993) indicate that only a small portion focused in forest systems for the purpose of vegetation management, and none were found that assessed biological control in the context of water quality. Numerous studies have assessed or summarized grazing impacts on water quality (Patric and Helvey 1986) but again, none in the context of fuels or vegetation management in forest systems.

## Conclusions

In general, prescribed fire and other fuels management approaches appear to have little impact on water quality in Eastern North America. When soils are deep and fire severity is low, few water quality changes have been observed, and those that have been reported are generally short lived (less than a year). The most dramatic impacts have occurred where soils are shallow and fires are severe; in these situations, some water quality parameters remained elevated for 3 or more years.

Certainly, more research on the effects of fire and other approaches to fuels management (mechanical, chemical, and biological) on surface water quality in Eastern North America is needed. Although considerable work has been accomplished on various forest types in the South, little has been done in the rest of Eastern North America, even in places where prescribed fire is being used as a tool for fuels management—such as red (*P. resinosa*) and jack pine (*P. banksiana*) management in the Lakes States. Also, considering the growing importance of carbon, carbon cycling, and the importance of carbon in aquatic food chains, more could be done to assess the influence of fire on dissolved organic carbon. Finally, mercury is the number one contaminant in surface waters (with more Environmental Protection Agency advisories than any other substance), and we know little about how fire affects mercury transport and accumulation in the food chain.

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# Economic Analysis of Fuel Treatments

**D. Evan Mercer, Jeffrey P. Prestemon**

## Introduction

The economics of wildfire is complicated because wildfire behavior depends on the spatial and temporal scale at which management decisions made, and because of uncertainties surrounding the results of management actions. Like the wildfire processes they seek to manage, interventions through fire prevention programs, suppression, and fuels management are scale dependent and temporally and spatially dynamic. The objective of this chapter is to describe the status of research into the economics of fuels management. We review studies describing the economic question of fuel treatment choices in wildfire management. We discuss the importance of framing the questions and issues surrounding wildfire management to include influences of space and time on wildfire processes. Finally, we offer a case study that provides one example of evaluating the economics of fuel treatments.

### *Approaches to Economics of Wildfire Management*

#### Defining the issues

Initial studies into the economics of wildfire management expressed the problem simply, but research in more recent years has begun to examine the full complexity of the issues involved. For example, Headly (1916) and Sparhawk (1925) described the situation of a fire boss or agency seeking to minimize the sum of wildfire losses (damages) and the costs of fire suppression for a single fire. Forty years later, Davis and Cooper (1963) and Davis (1965) recognized that fire managers could alter the distribution and quantity of fuels on a managed landscape—not just suppress fires—thereby describing a multiple-input problem with much greater spatial complexity. The task of the manager had evolved to include finding the level of fuels management that would alter the likelihood that fires reach a particular size. The economic problem, then, was to minimize the sum of fuels management costs, which included the costs of suppressing fires that do occur and the losses created by the wildfires on the landscape in a fire season. Note that in this chapter, wildfire damages or wildfire-related degradation of ecosystem goods and services are defined as "losses," the expenses incurred to manage wildfire on the landscape (prevention, fuels management, suppression, and rehabilitation after a fire) are defined as "costs," and the economic goal is described as minimizing the sum of costs plus losses (also known as net value change).

Subsequent to these pioneering analyses, researchers have begun to describe the issue even more broadly. Donovan and Rideout (2003), Mercer and others (2007), Prestemon and others (2002), and Rideout and Omi (1995) identified the problem of wildfire management as one in which a manager can take any number of actions to alter the fire regime in support of an overall wildfire program or societal benefit objectives. These actions include preventing fire ignitions, managing fuels, building firebreaks, positioning firefighting resources before fire seasons begin, suppressing fires, evacuating local residents, and reducing the impact of wildfire through timber salvage and site rehabilitation. The issue can be specified either as maximizing values protected minus the costs of protection, or as minimizing the sum of losses and costs of actions taken to affect those losses. Further, many researchers have properly described wildfire management as a long-term dynamic optimization problem in which management actions and wildfire in previous periods affect wildfire in the current period. For example, fuel treatments—including fuel breaks, prescribed fire, and mechanical removals—may have long-term or multiple-season effects on many measures of fire activity and therefore affect the levels of expected damages over many years (Mercer and others 2007).

### Determining tradeoffs

Given any of the available objectives of wildland management, the extent of wildfire managers interventions into wildfire processes depends on the costs of those interventions and on the degree, duration, and spatial extent of the intervention's effectiveness. For example, if the per-acre cost of prescribed fire on a given landscape is greater than the value of the expected wildfire damages, then prescribed fire would not, from the perspective of wildfire alone, be an economical option for management. A key objective for fuels management is maximization of effectiveness, reduction of cost, or a combination of the two. Maximizing effectiveness can mean applying fuels management in the places likely to do the most "good," and reducing cost means applying fuels management in places where it is inexpensive compared to the values at risk and in ways that require fewer or cheaper inputs.

The question of fuel treatment efficiency and the role of treatments in wildland management become more complicated when considering other costs and benefits derived from the treatment itself. For example, fuel treatments can provide many benefits that extend beyond wildfire management—prescribed fire may reduce vegetative competition and therefore enhance the growth of desired tree species (Crow and Shilling 1980), enhance production of nutritious forage for livestock, and provide habitat for fire-dependent species (González-Cabán and McKetta 1986). Mechanical fuel treatments offer benefits similar to prescribed fire, and they also produce wood that can be sold (Abt and Prestemon 2006, Rummer and others 2005). Fuel treatments applied in one location, or to one property, can offer benefits in other locations or properties by breaking up fuel contiguity, slowing the spread of wildfires, or enhancing the efficiency of fire suppression activities. Consideration of these benefits can lower the overall per-acre (or other unit) net costs of implementation, viewed from the broader perspective of the landscape or society as a whole.

At the same time, treatments may carry costs that go beyond their explicit implementation costs. For example, treatments can degrade environmental attributes and other environmental values important to society and may produce risks for neighboring landowners. Prescribed fire can affect water quality by altering vegetative cover (González-Cabán and others 2004), and it produces smoke that reduces air quality (Martin and others 1977). Mechanical treatments can increase siltation and compact the soil, reducing productivity and damaging residual trees and other plants. Chemical methods for reducing fuels also may have negative environmental impacts that need to be considered when evaluating treatment costs.

## Results of Past Fuel Treatment Studies

Economic studies of fuel treatments can be divided into three broad classes: (1) those focused on the factors that affect the costs of fuel treatments, (2) those concerned with how fuel treatments can lead to changes in wildfire processes, and (3) those evaluating how fuel treatments can be applied to achieve societal or landowner goals. Some studies straddle all classes, recognizing that choices about the best locations, timing, and extent of fuel treatments depend on their costs and on whether, where, and when fuel treatments are effective. Other studies quantify how the location and characteristics of fuels management affect their costs and hence their net contributions to the achievement of desired goals.

## *Factors Affecting Fuel Treatment Costs*

Although current knowledge on the costs of fuel treatments cannot be summarized in a few paragraphs, for the sake of brevity, this chapter focuses on some of the more influential studies. For more detailed reviews see Hesseln (2000) and Kline (2004).

Perhaps the first refereed journal article on prescribed fire costs in the Southern United States was by Vasievich (1980), who emphasized the importance of vegetation characteristics and scale of activity in influencing costs. Vasievich found that the thicker the vegetation, the higher the cost, because of denser undergrowth and ladder fuels that require higher labor, capital, and materials requirements. Recognizing the importance of fixed and variable costs in fuels management, the study showed that prescribed burns greater than 2,000 acres cost nine times less per acre than 50-acre burns. Because all treatment actions must contend with fixed costs such as burn management and larger fires require less perimeter management relative to area contained within the perimeter, greater efficiencies in the use of labor, capital, and other inputs can be achieved. This effect is termed in economics as an "economy of scale." Jackson and others (1982) examined the costs of fuel treatments in the Western United States focusing on how prescribed fire could enhance wildlife habitat rather than change wildfire risk. Their analysis of fuels treatments on national forests in Montana and northern Idaho was one of the first modern studies to document economies of scale for prescribed fire in the West—larger treatments are less costly per acre treated, a central focus of their analysis.

González-Cabán and McKetta (1986) focused on prescribed fire with and without mechanical treatment on two national forests in Montana and Oregon. Linear regression quantified the average effects of the fuel treatment method used and site factors on the per-acre cost. To identify fuel treatment effects beyond reducing fire probabilities and damages, they conducted surveys that asked fire managers to allocate the costs of treatments to benefits; managers allocated 45 of the treatment cost to reduction of wildfire damage risks on the Lolo National Forest, compared to 36 percent on the Willamette National Forest. Fuel treatment costs were found to be influenced slightly by the size of the treatment (indicating economies of scale) but more significantly by the type of treatment, the type of stand, the initial fuel conditions in the stand, the primary objective for the treatment (fuel reduction versus silviculture), and the seasonal variables that contribute to weather conditions at the time of the burn. González-Cabán and McKetta concluded that an important factor influencing costs is the objective of the treatment. Treatments that are focused on reducing wildfire risk appear to be significantly more costly than those focused on silvicultural objectives. Presumably, this is because risk-reduction treatments are designed to decrease fuel contiguities and reduce crowning and torching potentials, which require significantly more handling and fire treatment of downed woody debris.

A subsequent study by González-Cabán (1997) focused on the role that managerial and institutional factors play in the cost of prescribed burning. Based on a survey of U.S. Department of Agriculture Forest Service managers in the West, González-Cabán found that efforts to reduce the negative impacts of prescribed fire—such as risk of escape and smoke emissions—led to higher per-acre costs, indicating an important tradeoff between the two. The size of the burn was a significant explainer of costs, with larger burns lowering per-unit costs. Slope and other site factors also mattered, but management objectives did not. Thus, efforts to minimize the externalities from risk reduction appear to increase costs, but these costs can be reduced by treating large areas simultaneously.

Cleaves and others (2000) conducted a nationwide survey of national forests that sought to quantify the variability of prescribed fire costs. The study, based on a survey of managers, illustrates the combined importance of interforest variations in the availability of prescribed fire services, management objectives, site factors, prescribed-fire escape risks, and other constraints. Nationwide, from 1985 to 1994, the costs of prescribed fire were found to vary by tenfold or more, with some of the most expensive prescribed fire occurring in the western mountains (more than $300 per acre) and the cheapest in the South ($20 per acre). Mercer and others (2007) used the same dataset to further identify the factors that influence costs of prescribed fire on national forests. Their analysis showed that part of the differential is explained by differences in labor costs, with the average cost per acre in each national forest mirroring statewide labor costs. However, significant cost differences also existed from one region to another and according to the amount of forest available for treatment, indicating the importance of other influences.

Rideout and Omi (1995) concentrated their analysis on understanding how the scale of operations affects the cost of fuels treatments on national parks. They found strong economy-of-scale effects on costs, with lower per-acre costs associated with larger burn areas. They also found that costs are higher when managers take additional steps to reduce prescribed-fire escape risks or protect key resource values, when treatments are accompanied with mechanical, chemical, or biological pretreatments, and when significant effects of ecosystem structure or other factors related to location are present. In short, efforts to protect valuable resources, property, and people from the dangers of prescribed fire tend to drive up costs. This implies that ecosystem restoration and fire risk reduction activities may be more frequent in areas where there are fewer values at risk—a practical reality that complicates actions and economic analyses of where best to place treatments.

The importance of management constraints is also highlighted in a study of the role of private property protection by Berry and Hesseln (2004). This analysis found that per-acre mechanical and prescribed fire treatment costs in the Pacific Northwest were higher in the wildland-urban interface than in other areas. Size of treatment was negatively related to cost, confirming once again the effect of economies of scale. As stands became denser, they required more fuels management, increasing the cost of treatment, as would be expected. Validating previous research, treatments on sites with high fuels levels that were closer to values at risk tended to carry a higher cost. Because treatment costs and risk reduction benefits appear to be positively related, careful economic and statistical analyses are required to identify where best to place treatments on landscapes.

Research into the economics of fuel treatments has also been advanced by recent studies into the costs of mechanical fuel treatments and the factors affecting them. Rummer and others (2005) quantified the costs of fire- and ecosystem-enhancing mechanical fuel treatments for all forest lands in the western United States, and found that fuel treatment costs varied greatly according to the location of treatment and stand type. Costs per acre were high, ranging from a few hundred to thousands of dollars, suggesting that restoring ecosystems to fire-adapted conditions may be very costly. Working with data from Rummer and others (2005), Abt and Prestemon (2006) evaluated the timber market consequences of selling the products from fuel treatments in western timber markets. Focused on Federal lands, Abt and Prestemon showed that significant revenues were possible, thereby reducing their overall costs, but unintended consequences in the market may also occur. Concentrating treatments on higher risk sites could mean less outlay and possibly greater overall benefit than spending money on all at-risk sites without regard to degree.

Prestemon and others (2008) evaluated the costs of fuel treatments using the same framework as Abt and Prestemon (2006) but expanding the scope to all Federal lands in the South and the effects of treatments on fire risk and having to return to stands to apply additional treatments when risky conditions return. Prestemon and others (2008) also controlled for the effects of slope and stand conditions when conducting treatments based on a modified stand-density index or a thin-from-below type treatment in fire prone stands. They concluded that fuel treatments of these types could be less costly if marketable materials are removed and sold, but that the net cost of conducting treatments was still well over $600 per acre.

Much of the historical analyses of the factors influencing treatment costs have focused on U.S. public land management. Only rarely have studies focused on private lands. The important influence of legal and institutional constraints in driving up fuel treatment costs on private lands is clarified in a study by Yoder and others (2003). Liability issues affect the amount of effort a land manager puts into reducing the risk of fire escape from a prescribed burn. Strict liability laws that penalize managers for escapes tend to increase treatment costs compared to laws that only penalize managerial negligence. Liability laws vary across the United States, implying that the use of prescribed fire will be less frequent in places with strict liability compared to those with more relaxed rules. Enforcement of liability laws may therefore conflict with societal and managerial goals of ecosystem restoration and wildfire risk reduction, but the practical effect of the laws has not been fully evaluated. In locations where escapes are important, strict laws would not necessarily be as much in conflict with achievement of societal objectives as they would be in places where escape risk is low. Yoder and others (2003) showed that liability laws inflict costs on society in two ways: through increased treatment costs or through increased losses and suppression costs from wildfires that result after landowners begin reducing fuel treatments. However, the total of these costs must be weighed against the potential losses and costs associated with escaped prescribed fires.

## *Effects on Fire Processes*

Since the groundbreaking work by Davis (1965) and Davis and Cooper (1963), many wildland managers have recognized that the economical use of fuel treatments depends on how effective they are at changing wildfire activity across broad landscapes. More recently, Prestemon and others (2002)—bolstered by subsequent analyses by Butry (2006, 2009), Mercer and others (2007), and Mercer and Prestemon (2005)—described the importance of understanding the spatial and temporal dynamics of fuel treatments in managing fire activity. The effects of fuel treatments can span large areas and long time spans—accounting for them must not be limited to short-term responses to actions taken in one confined location.

Davis (1965) described wildfire activity as a conditional probability distribution across a range of wildfire sizes and frequencies, with the level of wildfire activity conditional on actions taken to affect fire activity. Davis (1965) and Davis and Cooper (1963) offered evidence of shifts in the expected amount of area burned in a management unit during a fire season in California and Florida. More broadly, Butry (2006, 2009), Mercer and others (2007), Prestemon and Butry (2005), and Prestemon and others (2002) provided evidence that prescribed fire and other treatments have long-term impacts, and that their effects are felt across space and time. Generally, prescribed fire was found to reduce wildfire area burned, with or weighting for intensity, in the long term with an elasticity ranging from about -0.05 to -0.30. In other words, each percentage increase in prescribed fire is expected to yield a long-term decrease in wildfire activity by 0.05 to 0.30 percent.

Many of the above studies used statistical techniques—actual data on wildfire and fuel treatment amounts—to quantify fuel treatment effects on wildfire. In the absence of historical data or when new types of treatments are being considered, statistical analyses are difficult. In these cases, simulation approaches are often used. For example, Finney and Cohen (2003) focused on how fuel treatments may affect wildfire area burned and the number of structures damaged. Emphasizing the scale of analysis for evaluating fuel treatment effects and the desired outcome measures, their simulations focused especially on how the placement of treatments may affect overall wildfire risk on the landscape and how fuel treatments can reduce fire intensity. Random location of fuel treatments produced fewer beneficial fire-control outcomes than a more systematic pattern of treatment.

Mercer and others (2007) confirmed the dual effects of fuel treatments on wildfire intensity (the rate at which a fire produces thermal energy) and area burned. They combined intensity and area burned data from Florida to identify the effectiveness of prescribed fire on the landscape to show that prescribed fire reduces wildfire in both the season following treatment and up to two subsequent seasons afterwards. The long-term impact of prescribed fire, however, was about 60 percent less than the short-term impact because of the dynamic impacts on wildfire. Fuel treatments were found to be less effective than wildfires in reducing the size and intensity of future wildfires, suggesting that short-term reductions are partially offset by subsequent wildfire activity. Mercer and others (2007) and Prestemon and others (2002) found that roundwood removals have various impacts on wildfire activity, serving to increase or decrease wildfires and intensities. This result, they speculate, is due to temporary increases in fine fuels in the aftermath of thinning.

Fernandes and Botelho (2003) reviewed the effectiveness of prescribed fire in achieving societal objectives—largely from studies of Mediterranean forest types—reporting that prescribed fire is quite effective at reducing fine fuels and therefore fire intensities and possibly the amount of area burned. They found empirical data suggesting that prescribed fire is most effective at reducing fire intensities and areas burned only when weather conditions during fires are not extreme. They also showed that strategic application of fuel treatments may be the most effective approach to reducing fire activity (Keeley 2002).

Piñol and others (2005) developed a simulation model that documents the importance of fire weather in modifying the effectiveness of fuel treatments. An interesting finding from their analysis is that the amount of fire area burned is about constant, regardless of whether the fire burns as a fuel treatment (prescribed fire) or as a wildfire. Nevertheless, prescribed fire must be done in places with high fuels levels if its purpose is to be a sufficient surrogate to wildfire; this can be operationally challenging.

The importance of targeting fuel treatment locations is confirmed by Hof and others (2000). In a simulation model of wildfire and fuel treatments, they found that effectiveness of treatments at protecting valued resources or property depends on the spatial distribution of the treatments, with layout and segmentation of the landscape important determinants. Implied here, given other research on prescribed fire and mechanical treatments (González-Cabán 1997, González-Cabán and McKetta 1986), is a tradeoff between costs and protection offered. Given that specific and effective treatment spatial orientations may be more costly than typical layout designs, the finding on the importance of layout, with support by Finney and Cohen (2003), also implies tradeoffs between treatment design and implementation costs.

## *Achieving Landowner and Societal Objectives*

To our knowledge, the first published (in a refereed journal) assessment of the role of fuels management in achieving improved desired outcomeswas by Saveland (1987). But even this analysis was based on simulation and was highly theoretical. Based on untested assumptions about the costs of prescribed fire and the damages from wildfire, Saveland estimated the level of prescribed fire efficacy that would result in positive net societal benefits (reducing total fire program costs and losses). The analysis, however, advanced consideration of the long-term effects of a fuel treatment program, a major contribution.

Although researchers such as Bellinger and others (1983) and Rideout and Omi (1995) recognized the importance of the potential role of making “presuppression” interventions in a wildfire program, prescribed fire was not considered part of the toolkit. Omi and others (2000) revisited the overall question of cost effectiveness, using a simulation approach to assess how fuel treatments might affect area burned. The analysis was purely theoretical, although it was based on simulations from national parks.

It was not until Prestemon and others (2002) broached the issue of the long-term, broadscale scope of the fuels management problem that actual historical data were used to evaluate the economics of fuel treatments. In their statistical analysis of wildfire in Florida, Prestemon and others (2002) showed that prescribed fire has long-term and short-term impacts and that its effects can be identified at broad spatial scales. Butry (2006) also showed how the economics of fuels management depend on the recognition of the tradeoffs among fuel treatments, suppression expenditures, and wildfire damages. Mercer and others (2007) extended these analyses to show how prescribed fire can lead to aggregate net benefits for reducing damages to timber, housing, and the broader economy.

Other research addressed fuel treatments at a stand rather than a landscape level, still recognizing the inherent spatial and long-term complexities. Using a Faustmann model of optimal timber rotation, Amacher and others (2006) described how landowners may choose a level of fuel treatment given the random nature of wildfire, the changed intensities of wildfires after treatments, the changed value of salvage timber resulting from treatments that followed wildfires, and the long-term nature of the effects. This study was the first to mathematically describe the spatial and temporal complexities of fuels management within the context of a multi-ownership landscape.

# Prescribed Fire and Sediment Production: Costs and Benefits

Soil erosion from wildfires is an important contributor to accelerated sedimentation which can result in lost storage capacity and increased treatment costs for municipal watersheds (González-Cabán and others 2004). A study by Wohlgemuth and others (1999) studied areas affected by recent wildfires and found that sediment production was about 90 to 95 percent lower in areas previously burned compared to nontreated areas. Gonzáles-Cabán and others (2004) present a methodology for estimating the costs and benefits of using prescribed fire to reduce sedimentation following wildfires; and then apply it in a case study of the watersheds in the Los Angeles foothills (encompassing the Angeles National Forest and adjacent private lands). They found that a wildfire interval of 22 years produced $2.5 million in sediment management and watershed rehabilitation costs for local, State, and Federal agencies. A multiple regression analysis of the impact of fire interval on sediment yield indicated that using prescribe fire to reduce the fire interval to 5 years would decrease sediment yield by 2.6 million cubic yards in the 33.3 square-mile watershed adjacent to the Angeles National Forest. A 1 percent decrease in the fire interval was found to reduce annual sediment yield by 0.58 percent. Implementing a prescribed burning program on a 5-year interval would save the County of Los Angles Department of Public Works $24 million per year in debris basin cleanout costs (Gonzáles-Cabán and others 2004).

## *Case Study: Defining Socially Optimal Fuel Reduction Programs*

Defining the "best" level of fuel treatment to apply to a forested landscape remains one of the most important and difficult issues for wildfire management because: (1) treatment effectiveness is difficult to measure and varies over time, (2) treatment costs are variable and are influenced by the scale of operations, (3) wildfire damages are complex and vary regionally, and (4) future fire occurrences are inherently uncertain. Over the past 5 years, Forest Service scientists have completed a series of studies to address this large question (Butry 2006, Butry and others 2001, Butry and Prestemon 2005, Mercer and Prestemon 2005, Mercer and others 2007, Prestemon and Butry 2005, Prestemon and others 2002, Prestemon and others 2008, Pye and others 2003). Using data on forest resources, meteorology, fire occurrence, and economic impacts within a probabilistic modeling framework, they built a state-of-the-science assessment of prescribed burning efficacy in Florida. Unlike previous studies, this work goes well beyond natural resource impacts to address how prescribed fire programs affect total societal benefits at a broad scale (for the purposes of these studies, societal benefits and costs are defined as the diversion of resources to vegetation management and away from other economically productive activities in the economy; in other words, the opportunity cost of foregone uses of these resources in the economy).

Mercer and others (2007) applied the approach to evaluate the optimal prescribed burning regime for a broad range of potential fire scenarios in Volusia County, Florida. Their results indicate that the current prescribed burning regime generated expected net gains in societal benefits, and in addition these gains would exceed the increased cost of a considerably expanded prescribed burning program. Although landowners currently burn about 4 or 5 percent of their forests each year, the optimal treatment to achieve societal benefits is approximately 13 percent.

These results provide information for forest managers in Florida but also suggest broader policy and program implications: (1) the available supply of fuel treatment providers plays a key role on the ability to accomplish goals; (2) understanding and predicting the potential fire severity and burned areas under different management regimes are crucial to identifying optimal policies; and, (3) the use of private sector prescribed burning services by public land agencies may drive up fuels treatments costs for private forest landowners—an unintended consequence of public programs can be a reduction in beneficial activities on private land.

## Model Construction

Next is a brief description of the methods used for the analysis conducted by Mercer and others (2007). In general, determining the optimal level of prescribed burning to achieve societal benefits requires solving a stochastic dynamic optimization problem. Therefore, to find the optimal levels of prescribed fire (or other vegetation management) inputs for wildfire risk reduction, we maximize the sum of expected current and future net present value of societal benefits:

$$\max_{x_t} A = E\left\{VW_t - v(x)' x_t + \sum_{i=t+1}^{T} e^{-ri}\left(VW_i - v(x)' x_i\right)\right\},$$
$$\text{and } W_t = W\left(Z_t, W_{t-j}, x_{t-k}\right) + \varepsilon_t, x_t \geq 0(\forall t) \tag{1}$$

where

$A$ is the maximization criterion (a measure of societal benefits)

$V$ is the net value change per acre of wildfire (a negative value per unit, measuring damages per unit of wildfire realized)

$W_t$ is current area (acres) burned by wildfire (if expressed as a quantity measure of resources "saved" by applying resource inputs, $V$ would be a positive number, reflecting positive values) for the spatial unit of observation in year $t$

**v** is a vector of the prices per acre of suppression, presuppression, and vegetation management inputs

$\mathbf{x} = (\mathbf{x_t}, \mathbf{x_{t+1}}, \ldots, \mathbf{x_T})$ is a vector of the amount of suppression, presuppression, and vegetation management inputs for year $t$ through $T$ (the planning horizon)

$\mathbf{Z}_t$ are exogenous inputs to wildfire production including stochastic climate variables

$\mathbf{W}_{t-j}$ is a vector of $j$ lags of wildfire area

$r$ is the discount rate

Solving this optimization problem produces a $T\times1$ vector of optimal input quantities and a $T\times1$ vector of wildfire quantities over time. The uncertainty associated with random events (errors in prediction of weather, for example) means that the area burned by wildfire is known only with error, complicating the solution process. In the presence of such error, simulation techniques may be used to identify, for example, the amounts of prescribed burning most likely to maximize the societal benefits criterion. Hadar and Russell (1969) describe how to evaluate these types of uncertain prospects.

Identifying the long-term expected impact of prescribed fire requires accounting for variable weather and the uncertainties associated with the "true" form of equation (1). Although equation (1) was estimated using historical data on fire output and wildfire production inputs, observed wildfire output always differs from that predicted by empirical models because of the random nature of the phenomenon and the imprecision of statistics. To identify the "best" level of prescribed fire to apply in a fire-prone landscapetwo versions of equation (1) were estimated—one expressing wildfire output in area burned without including fire intensity and one expressing wildfire output in area burned weighted by the intensity of the wildfire. Research has shown that wildfire intensity is closely related to the resulting damages to forests. So, measuring how prescribed fire affects the intensity of wildfire output should provide a more accurate prediction of the impacts of prescribed fire on wildfire damages.

## *Methods*

Next, the results from the empirical estimates of equation (1) were used to forecast the expected damages from wildfire under different prescribed fire scenarios for Volusia County, which was representative of the fire-prone landscape of Florida. Forecasts of each year's wildfire activity were made for 100 years into the future using the following procedure:

1. Select a fixed level of prescribed fire to apply every year,
2. Randomly select the values of two climate variables found to influence wildfire in Florida (an ocean temperature measure of the El Niño-Southern Oscillation and an index of sea level air pressure quantifying the North Atlantic Oscillation),
3. Randomly select a forecast error for wildfire area burned—with and without weighting for intensity—from the historical distribution of weather factors and from prediction errors,
4. Calculate the total annual expected wildfire damages and suppression costs and the annual cost of applying the fixed amount of prescribed fire to the county,
5. Vary the amount of prescribed fire chosen in step one and then repeat steps 2 through 4.

This process was continued, starting from 5,000 acres prescribed burned per year, up to about 100,000 acres per year (out of 313,000 acres of forest in the county). After all of the simulations were completed, the total, long-term discounted cost plus losses associated with wildfire and prescribed fire were compared across all levels of prescribed fire to identify the acreage of annual prescribed fire where the sum of costs and losses was smallest.

Data were obtained from many State and Federal agencies. Florida Division of Forestry fire data on State and private lands from 1981 to 2001 provided daily records of the location and the features of each wildfire, sufficient information to construct a damage measure that incorporated fire intensity into each county's metrics of acres burned per year. Data on wildfires on Federal lands were obtained from the Forest Service, U.S. Fish and Wildlife Service, and the National Park Service. The prescribed fire data from 1994 to 2001 were obtained from permits granted by the State for prescribed fire. The U.S. Department of Commerce National Oceanic and Atmospheric Administration provided data on the Niño-3 sea-surface temperature anomaly from 1994 to 2001 (U.S. Department of Commerce National Oceanic and Atmospheric Administration 2003a)—necessary because fires burn more in Florida when the Niño-3 anomaly is negative; and also provided the values of the North Atlantic Oscillation from 1994 to 2001, another ocean temperature measure linked to wildfire in Florida (U.S. Department of Commerce National Oceanic and Atmospheric Administration (2003b). The Forest Service provided count-level survey data on the forest extent. Data on annual housing counts in each county—the instrument for measuring the impact of available wildfire suppression resources—were provided by the Florida Bureau of Economics and Business Research (2002). Weighting for wildfire intensity was calculated by categorizing all wildfires into intensity classes using actual observations of the average flame length for each fire, based on research by Byram (1959). Annual intensity-weighted risk was derived by summing for each county the product of the annual number of acres burned in each intensity class times the average intensity for that class divided by the county's total forest area.

Models were estimated for fire at the county level. County fixed-effects time series models estimated of area burned with and without weighting for wildfire intensity. The dependent variables for the two models were: (1) intensity-weighted acres per acre of forest area in the county in the year and (2) the area of wildfire area burned in the county per acre of forest area in the county.

Losses associated with wildfire were calculated based on the 1998 wildfires (Butry and others 2001). Two versions of losses were generated: one version assembled losses in terms of societal benefits—consumer plus producer surplus. Another version assembled losses in terms of market values—prices multiplied by quantities. Losses accounted for timber losses from wildfire, housing losses, and suppression expenditures.

### *Results*

The original statistical models, relating fire area burned with and without weighting for intensity, show that prescribed burning at the county level has a large, statistically significant effect in the county. The elasticity of the area burned with weighting for intensity with respect to prescribed fire permitted area was −0.9 in the short term and −0.31 in the long term, −0.72 in the short term and −0.28 without weighting for intensity.

We also estimated a model for the supply of prescribed fire service providers, which showed a long-term supply elasticity of about 0.54. This indicates that the cost of prescribed fire per acre would increase twice as fast as the increase in the areal increase in prescribed fire. This extra cost associated with higher levels of prescribed fire was included in the cost plus loss simulations.

The simulation results in Figure 1 show that the optimal levels of prescribed fire depend on whether wildfire is measured purely by area burned or if intensity is also included. Based on the losses and costs associated with wildfire and prescribed fire applied to achieve these levels of losses and costs, the expected value of losses plus costs is minimized when prescribed fire is set at about 19,000 acres per year for simulations that take wildfire intensity into account, compared 14,000 acres per year for simulations that rely solely on area burned. From 1994 to 2001, actual wildfires in the area averaged about 13,000 acres per year, which is 30 percent less than the prediction that incorporated intensity data, but very close to the prediction that was based on the amount of area consumed without considering intensity data.

## Conclusions

The most striking finding from our survey of the economics of fuel treatments is mainly how little work has been done to evaluate net economic benefits. Most economic studies of wildfire management have focused on assessing wildfire economic impacts, understanding the economic questions surrounding wildfire suppression and prepositioning of suppression resources (an area that we did not thoroughly review), and evaluating the costs of fuel treatments and the variables affecting those costs. Many of the economic analyses of fire suppression have been theoretical, which limits their usefulness to managers; almost no experimental applications have been reported in the refereed literature.

Still less work has been published that quantifies how treatments may affect wildfire processes. Without models that can quantify the impacts of treatments on

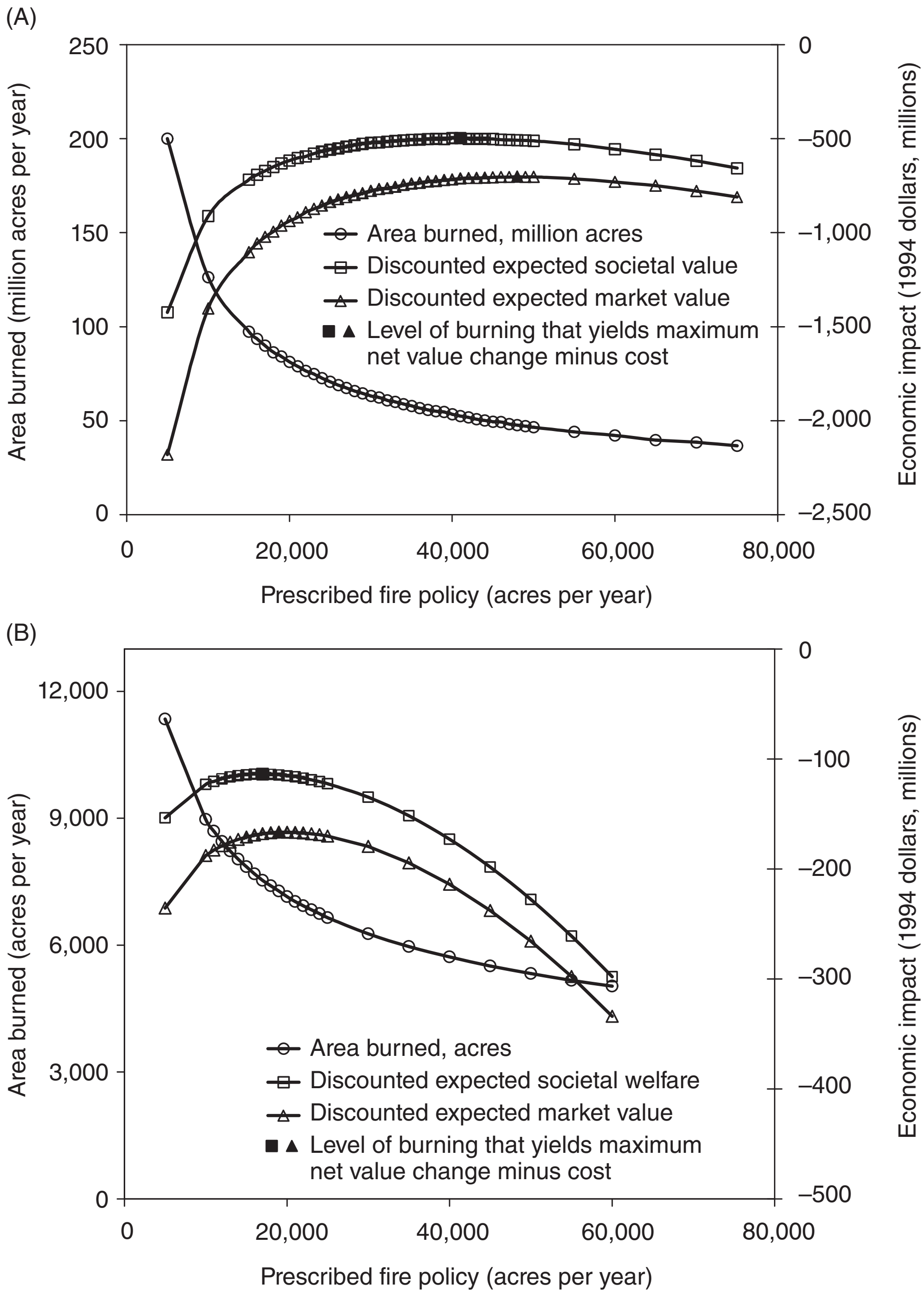


**Figure 1.** The economic value (avoided losses minus costs) of applying prescribed fire to reduce wildfire impacts, derived from (A) simulations that take wildfire intensity into account and (B) simulations that rely solely on area burned by wildfire (Mercer and others 2007).

wildfire activity, answering questions about the economics of treatments is not possible. Simulation models of wildfire, such as FARSITE, that can account for the effects of fuel treatments are available (Finney 1998, Finney and Andrews 1999), but they are in early development stages. The only known data-driven assessments in the refereed literature that we could find are analyses for Florida (Mercer and Prestemon 2005, Prestemon and others 2002). Only Davis and Cooper (1963), Davis (1965), and Mercer and others (2007) have done actual empirical analyses that place fuel treatments into the question of wildfire management economics.

Even the received literature has many information gaps that need to be addressed. For example, all of the literature ignores many of the externalities and nonmarket effects of fuel treatments. No study has integrated into an economic analysis of fuel treatments the possible damages associated with treatments, including the risks from escaped prescribed fires, the negative site impacts from mechanical thinnings, or the smoke from prescribed fires. Similarly, no study has integrated the benefits of fuel treatments on processes other than wildfire. These benefits include ecosystem restoration and improvement of timber growing conditions.

Additionally, we found only one study that examined impacts on watersheds (González-Cabán and others 2004), and it was not published in the refereed literature. These gaps in knowledge about the connections among wildfire, fuel treatments, and watersheds do not necessarily indicate lack of interest. We believe that the most recent advances in the science have occurred due to the availability of new historical data on wildfire and fuel treatments in the same landscapes, enhanced computational power (which allows fine-scale simulations), and the emergence and application of new statistical methods. We anticipate that future research in the area of fuel treatment economics will advance rapidly. In terms of water, the most significant research need that we see is for experimental and field experiments that measure water and watershed responses to wildfires and fuels management.

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Cumulative Watershed Effects of Fuel Management in the Eastern United States

CHAPTER 14.

# Methods Used for Analyzing the Cumulative Watershed Effects of Fuel Management on Sediment in the Eastern United States

**Daniel A. Marion, J. Alan Clingenpeel**

Previous chapters have described how various resource systems within a watershed can experience cumulative effects from fuel management activities like prescribed burning. As noted before, a cumulative effect is "the impact on the environment which results from the incremental impact of the action when added to other past, present, and reasonably foreseeable future actions, regardless of what agency (Federal or non-Federal) or person undertakes such other actions. Cumulative effects can result from individually minor but collectively significant actions taking place over a period of time" (40 C.F.R. § 1508.7).

A cumulative watershed effect is any cumulative effect that involves water movement through a landscape, either because water-related resources are affected or because a change in watershed processes generates the effect (Reid 2010). Because sediment production and movement affect water-related resources and are tied to water movement, a cumulative sediment effect is clearly a cumulative watershed effect.

Reid (2010) has identified the expectations for cumulative effect analysis that Federal courts have expressed in recent decisions that involved two Federal agencies—National Forest System of the Forest Service (U.S. Department of Agriculture) and the Bureau of Land Management (U.S. Department of the Interior)—after reviewing 62 district and appellant court decisions issued from 2000 to 2005. Some of these expectations are unaffected by which method is chosen to analyze sediment cumulative effects. However, others would be affected by what method is chosen or the content of the documentation for a given method (table 1).

For eastern national forests (Forest Service Southern and Eastern Regions), a technical guide (Tetra Tech 2002) provides the sanctioned strategy for determining if a cumulative watershed effect assessment is needed. The guide provides a step-by-step decision process (after MacDonald 2000) for conducting a cumulative watershed effect analysis and for determining the "level of effort" to be applied for a given situation. Five effort levels are identified depending on the degree of controversy involved, the linkages between activities and resources of concern, and the risk to those resources; with level 1 being the lowest effort and level 5 the highest. The guide does not specify that any particular analysis method be used at a given level; rather the analyst is expected to select the method most appropriate given the resource concerns and level of effort required.

**Table 1.** Legal expectations for cumulative effect analyses from 65 Federal court decisions, 2000 to 2005 involving the Forest Service (U.S. Department of Agriculture) or the Bureau of Land Management (U.S. Department of the Interior) (Reid 2010); and ways that the analysis method can affect the user's ability to address these expectations

| Expectation number | Expectation | Does the analysis method … |
|---|---|---|
| 1 | The area potentially affected by cumulative effects must be identified. | … allow the user to define an analysis area that includes all relevant individual effects, represents the processes and linkages by which a cumulative effect could result, and includes the location of resources or entities that could be affected? |
| 2 | The impact of the proposed project must be identified. | … consider both direct and indirect impacts of the new project activities on the resource of concern? Are results for the project clearly distinguished from other results and evaluated over a relevant time period? |
| 3 | The expected impacts of other individual actions in the past, present, and foreseeable future must be identified. | … list past, present, and foreseeable future actions (and their related impacts) in sufficient detail that they can be compared to those predicted for the proposed action? |
| 4 | The expected cumulative effect from the individual actions must be identified. | … determine the aggregate impact resulting from the combined individual impacts of past, present, and future actions? |
| 5 | Current and future impacts should be interpreted relative to naturally occurring conditions. | … define a baseline case upon natural conditions that would be expected to exist if no changes had occurred in the past? Can results from this case be compared to those for the past, present, and future conditions, and the proposed project? |
| 6 | Model validity for the present applications must be demonstrated. | … have documented tests of cumulative predictions using conditions similar to those now being analyzed? |
| 7 | Model shortcomings must be disclosed. | … have documentation that identifies the scientific reasoning used and any methodological assumptions, data gaps, or other problems that could affect prediction accuracy? |
| 8 | Reasoning behind significance interpretations must be stated. | … provide an interpretation of results significance to the resource of concern? If so, is the justification for this interpretation available in the documentation? |
| 9 | Effectiveness of mitigation must be evaluated. | … demonstrate how impacts are reduced if mitigation is necessary to lessen impacts to acceptable levels? |

Understanding the difference between a "method" for evaluating a cumulative watershed effect and the cumulative watershed effect analysis itself is important. The method is a tool used to predict how a specific watershed feature or process (such as stream temperature or sediment yield) will respond to a proposed activity; whereas the analysis uses these predictions to assess how a resource of concern (such as water quality or a freshwater mussel population) will be affected. The methods are a necessary first step to the analysis, but they do not constitute the cumulative watershed effect analysis itself, and the reader should keep this difference clearly in mind. Our review will focus largely on the methods in use, but we will briefly describe how outputs from these methods have been used in past analyses. The eastwide technical guide (Tetra Tech 2002) is recommended for those wishing a more comprehensive discussion of how the results from the analysis methods should be incorporated into a cumulative watershed effect analysis.

The purpose of this chapter is to describe the methods currently being used to conduct cumulative watershed effect analyses of fuel management projects within the forest lands of the Eastern United States and to evaluate how well they provide the information needed to meet legal expectations. To determine what methods are being used, we contacted soil scientists, hydrologists, and other specialists from all national forests within the Eastern United States who might be involved with such analyses.

We also contacted a limited number of resource specialists within environmental agencies of Eastern States who were recommended to us. While our survey indicates that cumulative watershed effect analysis of fuel-management projects is presently limited to Federal forest lands, the methods we discuss below could be used for any forest lands in the Eastern United States. Currently, the only cumulative watershed effect issue related to fuel management that is being analyzed is sediment; therefore only methods addressing sediment are covered. Moreover, although many techniques are available for managing fuel loads, prescribed fire is the one most commonly used; and is the technique, along with its concomitant fireline construction and use, that occasions cumulative watershed effect analyses most frequently in the Eastern United States. We limit our review to those methods that have been employed since 2000.

The remainder of this chapter is organized as follows. First, we review the methods currently being used to assess possible cumulative sediment effects from fuel management practices. Second, we discuss how well the methods provide the information required to meet the legal expectations identified by Reid (2010). Lastly, we identify several issues that should be considered in developing future models for assessing cumulative sediment effects.

# Analysis Methods Used in Eastern United States

Based on our survey of resource specialists, we found that the sediment analyses conducted in eastern national forests have several features in common. The responses from these specialists indicate that sediment is the only cumulative watershed issue related to fuel management that is being addressed in environmental analysis documents. The fuel-management practices of greatest concern are fireline construction and prescribed burning. The reasons why sediment is a primary concern can be found in chapter 12. Within eastern national forests, sediment analyses have most often been done during the forest planning process, and only rarely as part of project assessments. Sediment cumulative watershed effect analysis has not yet been applied to wildfires.

Past cumulative watershed effect applications for fire in eastern national forests seem to fall into just two effort levels—based on the eastwide technical guide (Tetra Tech 2002)—and employ similar methods within each level. Level 2 applications occur most often and result when sediment concerns are low and existing protection or mitigation methods are considered sufficient to address any concerns. Level 4 applications occur when concerns are moderate to high and existing controls may not be sufficient. Level 2 applications use a "narrative analysis" that describes the extent and potential severity of potential sediment effects, reviews the relevant literature on fire effects on sediment production, and states conclusions as to likelihood of a sediment effect and the effectiveness of proposed mitigation measures. For level 4 situations, "hazard rating models" are used to assess sediment effects. Hazard rating models use measured or categorized input variables that are mathematically manipulated (based on some conceptual or empirical model) to compute the combined effect of these variables on a response variable (in this case, sediment). Hazard rating models differ from deterministic models—such as the Water Erosion Prediction Project (WEPP) model (Flanagan and Nearing 1995)—in several ways. Perhaps the most important difference is that the rating model output is explicitly recognized as not representing a real amount; rather it is interpreted as an index value that can be used to compare different action scenarios and rate the potential risk of occurrence (high, moderate, or low). Although a number of models, both hazard rating and deterministic, have been developed over the years to assess the cumulative effects associated with fuel management activities (Elliot and others 2010), the only two models currently being used specifically for fuel management effects in the Eastern United States are the Erosion and Sediment Yield (EASY) model and the Aquatic Cumulative Effects (ACE) model. Both of these models produce outputs that are labeled as "sediment," however the documentation for both models states that these values are not to be considered physical quantities, but rather are relative values to be used in comparing alternatives and judging relative risks.

## Narrative Analysis

The use of narratives to assess cumulative watershed effects from fuel management practices is common in the forest plans of eastern national forests as well as in project level analyses. These narratives vary widely in detail and content, with sediment being the predominant concern. Conclusions are often based on professional opinion and the implementation of mitigation practices. Because of the wide range in detail and content of this method type, we did not attempt to evaluate narratives or assess how well they addressed the expectations listed in table 1.

## Hazard Rating Models

As noted above, the two hazard rating models currently being used in eastern national forests to assess sediment cumulative watershed effects related to fuel management are the EASY[1] and ACE[2] models. Both models predict the amount of erosion and sediment yield that will occur based on conditions within an analysis area. Erosion, also called soil loss, is the detachment and displacement of soil material from the ground surface. Sediment yield is the amount of eroded material that moves across the land surface, reaches a stream channel, and is transported as stream sediment to a given outlet point downstream. Erosion is typically expressed as a volume per unit area per unit time (tons per acre per year) whereas sediment yield is generally expressed as a total volume per unit time (tons per year).

Both the EASY and ACE models are applied using the same general procedure:

1. Delineate the analysis area, which is the total land area addressed by the analysis, including the area that will be directly affected by the activity prompting the analysis effort, plus all upstream and downstream areas that may contribute to or be affected by the possible effect being considered. Because both the EASY and ACE models predict sediment yield for entire drainage basins, the analysis area boundary is typically delineated using the one or more basins that encompass all of these land areas. Separate analyses can be done when multiple basins are used.
2. Identify all condition types within the area for the past, present, or future (proposed) scenario being analyzed. The condition type is a classification of the land-use activity or site conditions occurring or proposed over a contiguous land area. Examples include undisturbed forest land, forest area with a specific silvicultural practice applied (such as a clearcutting or shelterwood), road area, cropland, orchard, pastureland, urban land, or abandoned land. The classifications used vary somewhat between the models, but both models require an inventory of the existing condition types within the analysis area. Increasingly, this is accomplished using relevant Forest Service geographic information system (GIS) datasets for Forest Service lands and spatial data sets like National Land Cover Data (U.S. Department of the Interior U.S. Geological Survey 1992) and Topologically Integrated Geographic Encoding and Referencing (U.S. Census Bureau 2006) datasets for other lands.
3. Compute the total erosion from all condition types.
4. Compute the total sediment yield at the outlet of the area.
5. Repeat steps 2 through 4 for all project alternatives.
6. Interpret the sediment hazard associated with all project alternatives.

Details on how the analysis area is delineated, condition types are identified, erosion and sediment yield are computed, and results are interpreted for both the EASY and

[1] Hansen, William F.; Henderson, Jerry; Law, Dennis. 1994. Erosion and sedimentation yield background information using the Sumter National Forest Plan process records. 25 p. plus unpaged materials Unpublished paper. U.S. Department of Agriculture Forest Service, Francis Marion and Sumter National Forests, Columbia, SC.

[2] Clingenpeel, J. Alan; Crump, Michael A. 2005. A manual for the aquatic cumulative effects model. 42 p. Unpublished paper. U.S. Department of Agriculture Forest Service, Ouachita National Forest, Hot Springs, AR.

ACE models are given in the following sections. Differences between the two models are also noted.

## *The EASY Model*

The EASY model has been used on the Francis Marion and Sumter National Forests since the late 1980s to evaluate potential sediment impacts from existing or proposed conditions. A Microsoft Excel® spreadsheet program[3] is used to compute the model outputs from input data. To apply the EASY model to any area other than the Francis Marion and Sumter National Forests, the spatial data for all relevant condition types would have to be compiled for the new area.

### Analysis area delineation

Analysis areas are determined by the user; the EASY model places no restrictions on how large or small the area may be. Past applications have used areas up to 50,000 acres. Analysis areas generally correspond to watershed boundaries, but not always. On coastal areas, the terrain is very flat and watershed boundaries are difficult to discern with confidence, thus analysis areas there have not always been constrained to match watersheds. Past decisions have been based on what was deemed appropriate for the potentially affected terrain and the project being analyzed.

### Condition type determinations

For Forest Service lands, land areas for each existing or proposed condition type are obtained from Forest Service GIS datasets or relevant planning documents. Past applications have estimated the length of new or existing firelines from sample data when GIS data were not available. For other forest lands, existing conditions are determined by manual measurements from GAP imagery (South Carolina Cooperative Fish and Wildlife Research Unit 1993) or aerial photographs. The EASY model distinguishes several different condition types related to fire. Burned areas are classified as a site-preparation burn, dormant-season burn, or growing-season burn—with the latter two used for either fuel reduction or wildlife improvement. Firelines are classified as either hand or bulldozer constructed. Wildfire burns are not included, but could be classified using the existing types that best match the wildfire severity and suppression activities.

### Erosion and sediment yield computations

For each condition type, the soil loss is computed using

$$SL_i = area_i \times A_i \times t_{r_i} \tag{1}$$

where

$SL_i$ = soil loss (tons) from the $i$th condition type for the recovery period

$area_i$ = total area (acres) of the $i$th condition type in the analysis area

$A_i$ = erosion rate (tons per acre per year) for the $i$th condition type with the given soil region

$t_{r_i}$ = recovery period (years) for the $i$th condition type.

The EASY erosion rates are calculated for each condition type using equation (2):

$$A_i = R_{ave} SL_{ave} K_{ave} \left( \frac{C_{low} + C_{ave}}{2} \right) \tag{2}$$

[3] Hansen, W.F. [Undated]. [Untitled]. Spreadsheet. Available from the Francis Marion and Sumter National Forests, 4931 Broad River Road Columbia, SC 29212.

where for the $i$th condition type

$R_{ave}$ = average rainfall factor

$SL_{ave}$ = average slope length factor

$K_{ave}$ = average erosivity factor

$C_{low}$ = low cover type factor

$C_{ave}$ = average cover type factor[4]

Equation (2) is a variation of the Universal Soil Loss Equation (USLE) model (Wischmeier and Smith 1978) and primarily uses factor values developed by Dissmeyer and Stump (1978) for large soil regions throughout the South. Dissmeyer and Stump (1978) determined low, high, and average factor values for a variety of condition types (including burned forest land) in each soil region. The low, high, and average values are interpreted by Dissmeyer and Stump (1978) as those that would result from "minimum," "heavy," and "average" impacts, respectively, to a given land area. The values given in Dissmeyer and Stump (1978) are mean annual values for the entire recovery period for each condition type. The recovery period was the time (in years) it took for the values to return to pre-disturbance levels. Recovery rates vary from 1 to 2 years for most vegetation removal practices, to the entire analysis period for roads. Dissmeyer and Stump (1978) provide computational procedures and a map showing the soil regions and tables listing the low, average, and high factor values and their related erosion rates.

Although values for most of these factors are taken from Dissmeyer and Stump (1978), some were estimated based on available research and consultation with relevant specialists (see footnote 1). Users can readily change the erosion rates provided by the EASY model if they have more specific data for their analysis area.

In applying the EASY model, it is assumed that all cover type values fall somewhere between the low ($C_{low}$) and average ($C_{ave}$) values given by Dissmeyer and Stump (1978); therefore the model uses the simple average of these two values in computing a representative erosion rate for each condition type [equation (2)]. This assumption is based on the reasoning that current practices are not as disruptive to the groundcover as those measured by Dissmeyer and Stump (1978); thus the typical response should fall within the lower part of the range (see footnote 4).

Soil losses from forests not managed by the Forest Service are included in the analysis, but are assumed to be constant over the analysis period and the same for all planning alternatives.

The total sediment yield is the product of the total predicted erosion and the sediment delivery ratio (DR) for the analysis area [equation (3)].

$$Yield = DR \times \sum_i SL_i \tag{3}$$

Sediment delivery is the integrated result of the various processes between onsite erosion and downstream sediment yield, whereas the sediment delivery ratio is the ratio of total yield at the basin outlet to total erosion within the basin (Walling 1983). Sediment delivery ratio values have been determined two different ways in the past, depending on the spatial scale of the model application. For coarse spatial scales, a single delivery ratio value has been determined for each of the three landforms that make up the Francis Marion and Sumter National Forests: Appalachian Mountains, 0.38; Piedmont, 0.34; and Coastal Plain, 0.1.

The sediment delivery ratio values for the Appalachian Mountains and Piedmont were determined by Goddard (see footnote 1), while the value for the Coastal Plain is assumed to be 10 percent. This assumption is based on the estimated delivery ratio for third- and fourth-order basins in the Appalachian Mountains and Piedmont that is reduced by 30 percent for the lower drainage density in the lower Coastal Plain (U.S. Department of the Interior Forest Service 2006). For finer spatial scales (such

[4] Personal communication 2007. William Hansen, Hydrologist, Francis Marion and Sumter National Forests, 4931 Broad River Road Columbia, SC 29212.

as individual projects or timber sales), individual delivery ratio values are determined from Roehl's (1962) model using basin area.

### Results interpretation

The EASY analysis produces estimates of total soil loss and sediment yield for each condition type and total sediment yield for each analysis area and planning alternative (fig. 1). The spreadsheet can be modified by the user in any way desired to show how sediment yields vary between alternatives, condition type, land ownership, time period, or other categories of interest. The model does not include explicit direction on how to interpret the sediment yields; it is expected that the results will be presented and interpreted in whatever form is most appropriate for the problem at hand. Past applications on the Francis Marion and Sumter National Forests have presented EASY model results in a number of ways. One common approach has been to compute sediment yields for similar analysis units and to then judge the potential impacts between alternatives by the relative differences in their predicted sediment totals. A second method has been to compare the magnitude of sediment concentrations between alternatives. Sediment concentration is computed for the analysis area over the entire recovery period using an assumed mean water yield (based on local data) and the predicted sediment yield value for each alternative. A third method has been to determine the sediment yield value for the analysis area that is judged to be the worst case, and assume that impacts in other areas will be less than the worst-case value.

One concern with the EASY model is the way sediment delivery ratio values are often applied. Sediment yields are generally computed for each condition type within an analysis area (fig. 1). While this produces mathematically accurate results, it is nonetheless conceptually incorrect. Sediment delivery ratio values provided by Roehl (1962) and others are based both on the *total* erosion produced within the entire catchment [$\sum_i SL_i$ in equation (3)] and the *total* drainage area for the catchment. Applying delivery ratio values to the erosion produced from an area that covers only a portion of a drainage basin implies accuracy that is unsupported by Roehl (1962). This problem in no way invalidates the past analyses using the EASY model since

$$DR \times \sum_i^n SL_i = DR[SL_1] + \cdots + DR[SL_n] \tag{4}$$

However, we recommend that sediment yields only be listed for entire watersheds in future applications.

## *The ACE Model*

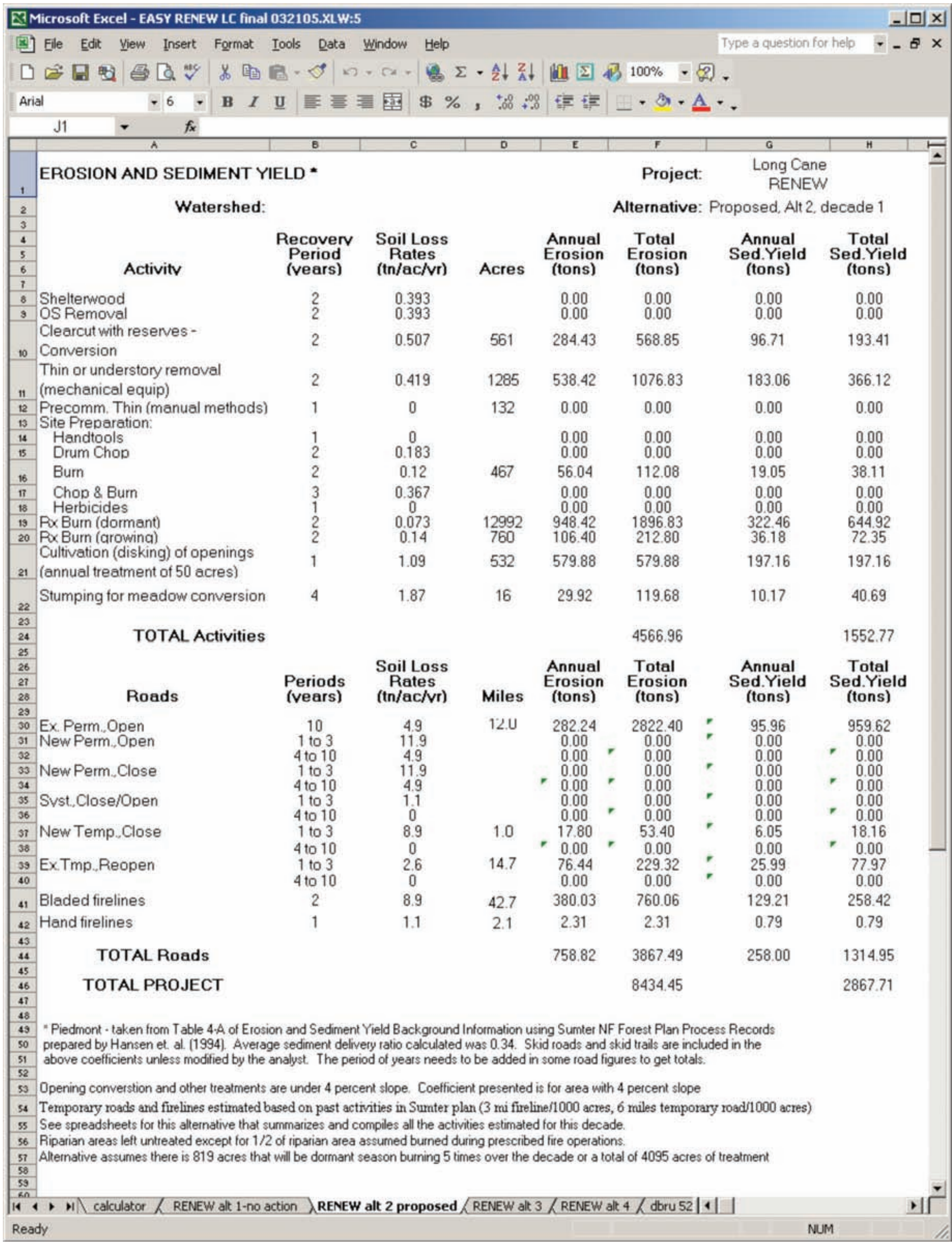
The ACE model for the Ouachita and Ozark-St. Francis National Forests[5] is the most current version of a cumulative watershed effect model that has evolved since 1990. Previous versions[6 7 8] differ in certain components of the model, but the overall methodology has remained fairly constant. The model runs through a Microsoft Excel® workbook file. Spatial data for current conditions (the compilation date varies by area) have been compiled for all fifth-level hydrologic units on the National

[5] Clingenpeel, J.A. [Undated]. [Untitled]. CD-ROM of model and spatial data for the Ouachita and Ozark-St. Francis National Forests. U.S. Department of Agriculture Forest Service, Ouachita National Forest, Hot Springs, AR.

[6] Clingenpeel, J.A. 2003. Sediment yields and cumulative effects for water quality and associated beneficial uses (process paper for forest plan revisions). 39 p. Unpublished paper. U.S. Department of Agriculture Forest Service, Ouachita National Forest, Hot Springs, AR.

[7] Clingenpeel, J.A.; Mersmann, T. 1999. Cumulative effects analysis for water quality and associated beneficial uses-national forests in Mississippi. Unpublished paper. 20 p. U.S. Department of Agriculture Forest Service, Ouachita National Forest, Hot Springs, AR.

[8] U.S. Department of Agriculture Forest Service. 1990. Cumulative impacts analysis-water quality and associated beneficial uses, Ouachita National Forest, Arkansas–Oklahoma. 13 p. Unpublished paper. U.S. Department of Agriculture Forest Service, Ouachita National Forest, Hot Springs, AR.



Microsoft Excel - EASY RENEW LC final 032105.XLW:5

EROSION AND SEDIMENT YIELD * — Project: Long Cane RENEW

Watershed: — Alternative: Proposed, Alt 2, decade 1

| Activity | Recovery Period (years) | Soil Loss Rates (tn/ac/yr) | Acres | Annual Erosion (tons) | Total Erosion (tons) | Annual Sed.Yield (tons) | Total Sed.Yield (tons) |
|---|---|---|---|---|---|---|---|
| Shelterwood | 2 | 0.393 | | 0.00 | 0.00 | 0.00 | 0.00 |
| OS Removal | 2 | 0.393 | | 0.00 | 0.00 | 0.00 | 0.00 |
| Clearcut with reserves - Conversion | 2 | 0.507 | 561 | 284.43 | 568.85 | 96.71 | 193.41 |
| Thin or understory removal (mechanical equip) | 2 | 0.419 | 1285 | 538.42 | 1076.83 | 183.06 | 366.12 |
| Precomm. Thin (manual methods) | 1 | 0 | 132 | 0.00 | 0.00 | 0.00 | 0.00 |
| Site Preparation: | | | | | | | |
| Handtools | 1 | 0 | | 0.00 | 0.00 | 0.00 | 0.00 |
| Drum Chop | 2 | 0.183 | | 0.00 | 0.00 | 0.00 | 0.00 |
| Burn | 2 | 0.12 | 467 | 56.04 | 112.08 | 19.05 | 38.11 |
| Chop & Burn | 3 | 0.367 | | 0.00 | 0.00 | 0.00 | 0.00 |
| Herbicides | 1 | 0 | | 0.00 | 0.00 | 0.00 | 0.00 |
| Rx Burn (dormant) | 2 | 0.073 | 12992 | 948.42 | 1896.83 | 322.46 | 644.92 |
| Rx Burn (growing) | 2 | 0.14 | 760 | 106.40 | 212.80 | 36.18 | 72.35 |
| Cultivation (disking) of openings (annual treatment of 50 acres) | 1 | 1.09 | 532 | 579.88 | 579.88 | 197.16 | 197.16 |
| Stumping for meadow conversion | 4 | 1.87 | 16 | 29.92 | 119.68 | 10.17 | 40.69 |
| TOTAL Activities | | | | | 4566.96 | | 1552.77 |

| Roads | Periods (years) | Soil Loss Rates (tn/ac/yr) | Miles | Annual Erosion (tons) | Total Erosion (tons) | Annual Sed.Yield (tons) | Total Sed.Yield (tons) |
|---|---|---|---|---|---|---|---|
| Ex. Perm.,Open | 10 | 4.9 | 12.0 | 282.24 | 2822.40 | 95.96 | 959.62 |
| New Perm.,Open | 1 to 3 | 11.9 | | 0.00 | 0.00 | 0.00 | 0.00 |
| | 4 to 10 | 4.9 | | 0.00 | 0.00 | 0.00 | 0.00 |
| New Perm.,Close | 1 to 3 | 11.9 | | 0.00 | 0.00 | 0.00 | 0.00 |
| | 4 to 10 | 4.9 | | 0.00 | 0.00 | 0.00 | 0.00 |
| Syst.,Close/Open | 1 to 3 | 1.1 | | 0.00 | 0.00 | 0.00 | 0.00 |
| | 4 to 10 | 0 | | 0.00 | 0.00 | 0.00 | 0.00 |
| New Temp.,Close | 1 to 3 | 8.9 | 1.0 | 17.80 | 53.40 | 6.05 | 18.16 |
| | 4 to 10 | 0 | | 0.00 | 0.00 | 0.00 | 0.00 |
| Ex.Tmp.,Reopen | 1 to 3 | 2.6 | 14.7 | 76.44 | 229.32 | 25.99 | 77.97 |
| | 4 to 10 | 0 | | 0.00 | 0.00 | 0.00 | 0.00 |
| Bladed firelines | 2 | 8.9 | 42.7 | 380.03 | 760.06 | 129.21 | 258.42 |
| Hand firelines | 1 | 1.1 | 2.1 | 2.31 | 2.31 | 0.79 | 0.79 |
| TOTAL Roads | | | | 758.82 | 3867.49 | 258.00 | 1314.95 |
| TOTAL PROJECT | | | | | 8434.45 | | 2867.71 |

* Piedmont - taken from Table 4-A of Erosion and Sediment Yield Background Information using Sumter NF Forest Plan Process Records prepared by Hansen et. al. (1994). Average sediment delivery ratio calculated was 0.34. Skid roads and skid trails are included in the above coefficients unless modified by the analyst. The period of years needs to be added in some road figures to get totals.

Opening converstion and other treatments are under 4 percent slope. Coefficient presented is for area with 4 percent slope

Temporary roads and firelines estimated based on past activities in Sumter plan (3 mi fireline/1000 acres, 6 miles temporary road/1000 acres)

See spreadsheets for this alternative that summarizes and compiles all the activities estimated for this decade.

Riparian areas left untreated except for 1/2 of riparian area assumed burned during prescribed fire operations.

Alternative assumes there is 819 acres that will be dormant season burning 5 times over the decade or a total of 4095 acres of treatment

calculator / RENEW alt 1-no action / RENEW alt 2 proposed / RENEW alt 3 / RENEW alt 4 / dbru 52

**Figure 1.** Example of EASY model output showing the cumulative watershed effect analysis of sediment for one proposed fuels management alternative. (Source: Hansen, William F.; Henderson, Jerry; Law, Dennis. 1994. Erosion and sedimentation yield background information using the Sumter National Forest Plan process records. 25 p. plus unpaged materials Unpublished paper. U.S. Department of Agriculture Forest Service, Francis Marion and Sumter National Forests, Columbia, SC.)

Forests in Alabama, Chattahoochee-Oconee National Forest, Cherokee National Forest, Daniel Boone National Forest, Sumter National Forest, National Forests in Mississippi, Ouachita and Ozark-St. Francis National Forests, and Jefferson National Forest; and for all sixth-level hydrologic units on the Ouachita and Ozark-St. Francis National Forests.

## Analysis area delineation

The ACE model is designed to be applied at the fifth-level hydrologic unit (approximately 39 to 386 square miles) for forest planning efforts and sixth-level hydrologic unit (approximately 4 to 39 square miles) for project level analysis (see footnotes 1 and 9). Unlike the EASY model, where users can bound the analysis area however they choose, the ACE model computes sediment yields for these two scales only. At the project level the user simply selects the sixth-level hydrologic unit or units that contain the project areas and the model analyzes the entire area within each selected unit.

## Condition type determinations

The data sources used to compile condition types for the ACE model are described in Clingenpeel and Crump (see footnote 1). Forest Service GIS data were used to determine condition type, road, land ownership, and ecoregion type for Forest Service lands within each fifth- or sixth-level hydrologic unit. Topologically Integrated Geographic Encoding and Referencing data from 1995 (U.S. Census Bureau 2006) were used to determine road types and lengths on forests not managed by the Forest Service. Condition types outside Forest Service lands were classified using 1992 data from the National Land Cover Data (U.S. Department of the Interior U.S. Geological Survey 1992). The slope class that each condition type fell within was determined by deriving slope class polygons using ArcView®'s GIS Spatial Analysis 2.0a extension (ESRI 2001) and 100-foot digital elevation models. Ecoregion, condition type, slope class, and ownership layers were then overlaid and rasterized on a 100-foot grid using ESRI's ArcView® 3.2 so that each grid cell was assigned a single value based on the combined layers present in the cell. Total areas for each combination type were then computed using the grid cells for each fifth- or sixth-level hydrologic unit. A similar overlay analysis was done to determine total length of road types by ecoregion and ownership by hydrologic unit.

The ACE model uses four condition types related to fire: fuel reduction and site preparation burns (for areas), fireline constructed, and fireline reconstructed. No distinction is made for type of fire (prescribed versus wildfire), preburn vegetation cover, or vegetation growing period.

## Erosion and sediment yield computations

The ACE model uses an overall computational process that is similar to the EASY model [equation (1)]; however there are several important differences in how erosion rates are determined and applied. Whereas the EASY model uses the factor values, the ACE model uses the actual erosion rates for each condition type provided by Dissmeyer and Stump (1978). More precisely, the ACE model uses the "average" rate determined by Dissmeyer and Stump (1978) for slopes less ≤35 percent, and the "high" rate for slopes >35 percent. Although this probably overestimates erosion associated with Forest Service activities, the higher erosion rates compensate for steeper slopes and management practices on other lands "that may not have the same standards as Forest Service lands"—where erosion rates are presumed to be higher (see footnote 9). The basis for erosion rates from burned areas is a second difference: Where measured rates are lacking (such as the Ouachita Mountains), the ACE model assumes burned areas erode at twice the rate of comparable undisturbed forest areas.[9] The length of the recovery period and how erosion rates vary during this period is still a third difference. For

[9] Personal communication. Various dates. J. Alan Clingenpeel, Hydrologist, U.S. Department of Agriculture Forest Service, Ouachita National Forest, P.O. Box 1270, Hot Springs, AR 71902.

forested areas, the ACE model assumes that all burned areas recover fully after 1 year and all harvested areas recover after 3 years. The decrease in erosion rates during the second and third years after harvest is based upon past research and field observations within the Ouachita National Forest (see footnote 1).

Soil loss from sample agricultural (such as pasture land or cultivated cropland) and urban condition types ($SL_{nf}$) was determined using the WEPP model (Flanagan and Nearing 1995). Representative soil characteristics from the WEPP database were applied to morphologic data (ecoregion, area, and slope) for each area of agricultural and urban condition type with a given hydrologic unit to compute the soil loss from each area (see footnote 1).

Total sediment yield from non-road areas ($SY_{nr}$) within a hydrologic unit is computed by summing the soil loss values computed for all forest and nonforest condition types, and multiplying this value by the sediment delivery ratio given by Roehl (1962) for the basin area [equation (5)].

$$SY_{nr} = DR \times (SL_f + SL_{nf}) \tag{5}$$

Still another important difference with the ACE model is how sediment from road areas was determined. The WEPP model (Elliot 2004) was used to compute representative sediment yield values (tons per mile) for roads, firelines, and all-terrain vehicle trails within each ecoregion based on sample data from each (see footnote 1). Separate yields were computed for each combination of usage type (road, all-terrain vehicle trail, or fireline), surface type, and maintenance level that occurs. Note that these are sediment yields, not soil loss values. The WEPP model includes a channel routing algorithm for estimating how much eroded sediment is delivered to and moves through the channel to the mouth. Total sediment yield from roads ($SY_r$) for an analysis area is determined by multiplying the appropriate unit yield value ($SL_{r_i}$) times the length of road by surface type and maintenance level ($l_i$), and summing these for all road types/maintenance levels within the hydrologic unit [equation (6)]

$$SY_r = \sum_i l_i \times SL_{r_i} \tag{6}$$

where

$i$ = the given road type and maintenance level.

A more detailed explanation of how road sediment yields were modeled is given in Clingenpeel (see footnote 9).

Total sediment yield from the analysis area is the sum of road and non-road sediment yields [equation (7)].

$$SY = SY_{nr} + SY_r \tag{7}$$

The ACE model is designed to require a minimum of user input. Areas for all condition types and existing roads are already determined for each sixth-level hydrologic unit. Erosion rates for all condition types have also been computed and compiled for each hydrologic unit. The user is only required to input any road types not previously captured, the various areas and condition types associated with the project alternatives being analyzed, and an assumed rotation age for private forest lands. Once these data are entered, the ACE model computes sediment yields for past, present, and proposed future conditions within each hydrologic unit. To compute the past condition sediment yield, the model assumes undisturbed forest cover over the entire basin. Present sediment yield is based on conditions existing as of when the spatial data were compiled (1992 for the Ouachita and Ozark-St. Francis National Forests), plus any updates for roads. The condition types for nonforest lands are assumed to remain constant between the compilation date and the analysis date; whereas forest land condition types have their erosion rates adjusted based on recovery or new harvest disturbances during this intervening period. The future sediment yield is based on conditions proposed in each alternative plus assumptions about harvesting on private forest lands. The ACE model also provides a routine that takes into account erosion mitigation efforts on roads that reduce soil losses (for example, through road obliteration or closure).

Unlike the EASY model, the ACE model only computes the total sediment yield of the past, present, and future scenarios for the first year of implementation of the proposed project. For condition types with erosion rates that decrease over time (or "recover"), the model uses the rates appropriate for the implementation year. For the future scenario, all proposed activities are assumed to occur during the first year of implementation. For example, a proposed project with new road construction, harvesting, and burning is modeled in ACE as if all of these activities occur in the first year. While recognized as inaccurate for most situations, this assumption eliminates the need to know the year in which each activity will occur and provides something of a "worst case scenario" by forcing all effects into a single year (see footnote 1).

### Results interpretation

The ACE model presents its results in a standard format which the user cannot modify. An example of the summary page, which displays the model results, is given in figure 2. The summary displays the total areas for each condition type under Forest Service or other management, road lengths and densities, and the total sediment yield from road and non-road areas under the past, present, and future scenarios. Lastly, the model displays ratings of relative risk to aquatic biota from sediment for each alternative.

The risk ratings show the significance of sediment effects relative to the beneficial use—providing aquatic habitat—common to all streams draining Forest Service lands. These risk ratings (also called watershed condition ranks) are based on the percent increase in sediment yield over what is predicted for the past (undisturbed forest) condition (fig. 2). Percent increases are listed for the current combination of condition types, and the combinations associated with each proposed alternative. In addition, the risk ratings for the current condition and all alternatives are also listed. These ratings were determined for each Arkansas ecoregion from bivariate analysis of various fish community metrics and predicted sediment increases using the ACE model. Based on the rationale of Terrell and others (1996), this analysis identifies when sediment increase—despite the influence of other habitat factors—becomes a limiting factor for fish population numbers. This analysis was used to establish criteria for rating the condition of fish communities given current sediment levels, and the hazard posed by potential sediment increases from proposed projects. A similar analysis was used to develop risk ratings for four ecoregions outside Arkansas: the Coastal Plain, the Central Appalachian/Western Allegheny, the Piedmont, the Blue Ridge, and the Ridge and Valley (see footnote 9).

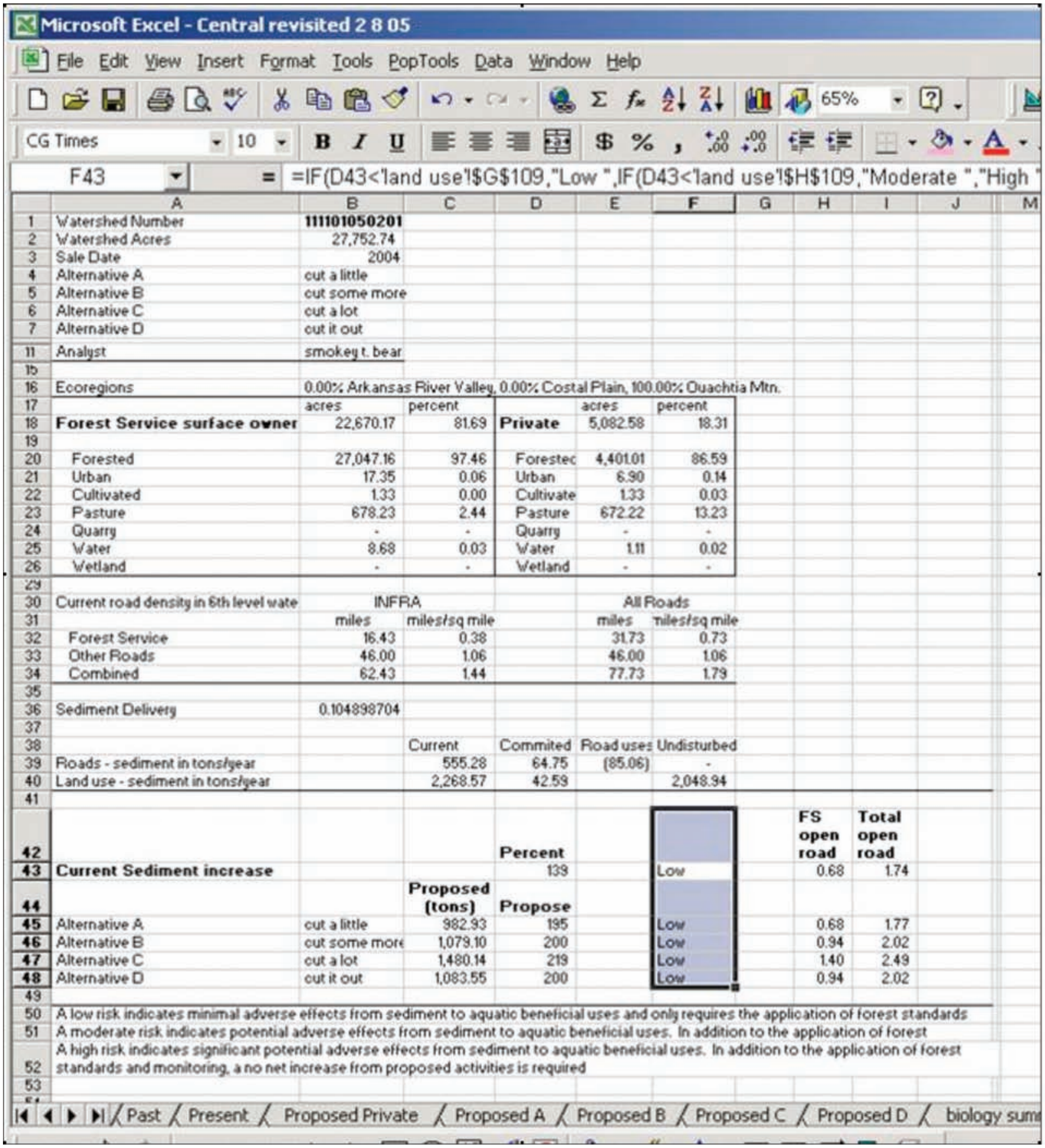
# Discussion of Current Methods

Both the EASY and ACE models are based on specific analytical procedures, compute a variety of outputs, and provide supporting documentation. Following is a comparison of how well these procedures, outputs, and documentation provide the information needed to meet the legal expectations for cumulative watershed effect analysis as identified by Reid (2010) and as summarized in table 1.

## *Model Features*

### Analysis area identification

The EASY and ACE models are very different in how the analysis area is identified. The EASY model imposes no constraints on the user; the analysis area can be whatever size the user deems appropriate. One advantage of this approach is that the user can model at a series of catchment sizes to better determine the scale at which the cumulative effect becomes insignificant. A disadvantage is that the user would also have to compile the spatial data at each analysis scale, as the model itself does not provide these




| | A | B | C | D | E | F | G | H | I |
|---|---|---|---|---|---|---|---|---|---|
| 1 | Watershed Number | 111101050201 | | | | | | | |
| 2 | Watershed Acres | 27,752.74 | | | | | | | |
| 3 | Sale Date | 2004 | | | | | | | |
| 4 | Alternative A | cut a little | | | | | | | |
| 5 | Alternative B | cut some more | | | | | | | |
| 6 | Alternative C | cut a lot | | | | | | | |
| 7 | Alternative D | cut it out | | | | | | | |
| 11 | Analyst | smokey t. bear | | | | | | | |
| 15 | | | | | | | | | |
| 16 | Ecoregions | 0.00% Arkansas River Valley, 0.00% Costal Plain, 100.00% Ouachtia Mtn. | | | | | | | |
| 17 | | acres | percent | | acres | percent | | | |
| 18 | Forest Service surface owner | 22,670.17 | 81.69 | Private | 5,082.58 | 18.31 | | | |
| 19 | | | | | | | | | |
| 20 | Forested | 27,047.16 | 97.46 | Forestec | 4,401.01 | 86.59 | | | |
| 21 | Urban | 17.35 | 0.06 | Urban | 6.90 | 0.14 | | | |
| 22 | Cultivated | 1.33 | 0.00 | Cultivate | 1.33 | 0.03 | | | |
| 23 | Pasture | 678.23 | 2.44 | Pasture | 672.22 | 13.23 | | | |
| 24 | Quarry | - | - | Quarry | - | - | | | |
| 25 | Water | 8.68 | 0.03 | Water | 1.11 | 0.02 | | | |
| 26 | Wetland | - | - | Wetland | - | - | | | |
| 29 | | | | | | | | | |
| 30 | Current road density in 6th level wate | INFRA | | | All Roads | | | | |
| 31 | | miles | miles/sq mile | | miles | niles/sq mile | | | |
| 32 | Forest Service | 16.43 | 0.38 | | 31.73 | 0.73 | | | |
| 33 | Other Roads | 46.00 | 1.06 | | 46.00 | 1.06 | | | |
| 34 | Combined | 62.43 | 1.44 | | 77.73 | 1.79 | | | |
| 35 | | | | | | | | | |
| 36 | Sediment Delivery | 0.104898704 | | | | | | | |
| 37 | | | | | | | | | |
| 38 | | | Current | Commited | Road uses | Undisturbed | | | |
| 39 | Roads - sediment in tons/year | | 555.28 | 64.75 | (85.06) | - | | | |
| 40 | Land use - sediment in tons/year | | 2,268.57 | 42.59 | | 2,048.94 | | | |
| 41 | | | | | | | | | |
| 42 | | | | Percent | | | | FS open road | Total open road |
| 43 | Current Sediment increase | | | 139 | | Low | | 0.68 | 1.74 |
| 44 | | | Proposed (tons) | Propose | | | | | |
| 45 | Alternative A | cut a little | 982.93 | 195 | | Low | | 0.68 | 1.77 |
| 46 | Alternative B | cut some more | 1,079.10 | 200 | | Low | | 0.94 | 2.02 |
| 47 | Alternative C | cut a lot | 1,480.14 | 219 | | Low | | 1.40 | 2.49 |
| 48 | Alternative D | cut it out | 1,083.55 | 200 | | Low | | 0.94 | 2.02 |
| 49 | | | | | | | | | |
| 50 | A low risk indicates minimal adverse effects from sediment to aquatic beneficial uses and only requires the application of forest standards | | | | | | | | |
| 51 | A moderate risk indicates potential adverse effects from sediment to aquatic beneficial uses. In addition to the application of forest | | | | | | | | |
| 52 | A high risk indicates significant potential adverse effects from sediment to aquatic beneficial uses. In addition to the application of forest standards and monitoring, a no net increase from proposed activities is required | | | | | | | | |
| 53 | | | | | | | | | |

**Figure 2.** Example of ACE model output showing the summary sheet for the sediment cumulative watershed effect analysis for an entire fuels management project. (Source: Clingenpeel, J. Alan; Crump, Michael A. 2005. A manual for the aquatic cumulative effects model. 42 p. Unpublished paper. U.S. Department of Agriculture Forest Service, Ouachita National Forest, Hot Springs, AR.)

data. Although this approach does provide maximum flexibility, it also provides no guidance as to what spatial scale may be too small or too large for accurate results. The ACE model limits the analysis area to either fifth- or sixth-level hydrologic units; scales that were thought to be appropriate for forest planning (see footnote 9) and project planning (see footnote 1), respectively. The user has no ability to vary from these two options; however, most of the spatial data have already been compiled and the user only has to compile data for updates to the current condition types and the proposed actions. Thus, the EASY and ACE models provide different advantages and disadvantages with respect to analysis flexibility and data compilation requirements.

### Identification of impacts from proposed action

Both models identify the sediment impact from proposed actions, and present these results in similar ways. The ACE model uses a separate sheet within the Excel® workbook for each alternative to list the areas associated with each proposed action (such as seed-tree harvest, fuel reduction burn, or fireline construction) and predicted soil losses associated with each action. The EASY model produces the same results, but the format depends on how the user chooses to present the data. The main difference between the two models is that the ACE model only computes sediment for the first year and assumes all actions are implemented during that year; whereas the EASY model computes sediment over multiple years after project implementation. The length of time modeled with EASY typically depends on the longest recovery period associated with the condition types involved.

### Identification of past, present, and future impacts

Both models have the ability to predict sediment impacts from past, present, and future actions, however, only the ACE model currently does so explicitly. The ACE model computes sediment yields for past, present, and proposed future scenarios, and displays the results on separate sheets within the workbook. The EASY model typically uses the present scenario as a no-action alternative, and it displays this alternative along with all proposed future scenarios in separate tables or sheets. The EASY model does not compute sediment values for an assumed past situation that would represent natural sediment production. However, it would be a relatively simple matter to revise the model to do this based, for example, on the assumption of uniform natural forest cover. Both models also provide ways for predicting sediment from areas burned during past wildfires, site preparation, or fuel reduction burns; although they differ in the erosion rates used and recovery period lengths.

### Identifying effects

The two models identify sediment cumulative watershed effects in very similar ways. They both list the relevant condition types either occurring within or proposed for the analysis area, compute the total erosion associated with each type, and the total sediment yield for present and proposed future scenarios. Both models include the ability to model expected changes in sediment yield from assumed actions on private lands in assessing future scenarios.

### Interpretation of impacts relative to naturally occurring conditions

The ACE model provides an explicit comparison of cumulative sediment effects under natural conditions, whereas the EASY model does not. The ACE model estimates natural (past) sediment yield by assuming a uniform forest cover, computes the percent increase in sediment yield for the present and each proposed future scenario, and lists the relative risk ratings based on these predicted increases. The EASY model does not provide an estimate of natural sediment yields. Past applications of the EASY model have compared future to present—not past—sediment yields, and based risk interpretations on this comparison. As noted earlier, it would not be difficult to revise the EASY model to produce predictions for an assumed natural scenario and thus satisfy this expectation.

## *Demonstration of Model Validity*

No direct validation of the EASY model output has been done. This could be accomplished by comparing measured sediment yields before and after some activity like prescribed burning in a basin, and determining how well the change compares with that predicted by the model. Another approach would be a treatment/control design wherein two basins are used that are similar in every way except that one experiences the activity (such as burning), and their sediment yield differences are compared to model predictions.

A less direct approach to model validation would be to validate the model components (soil loss and sediment yield). To our knowledge, no direct validation of the erosion rates given in Dissmeyer and Stump (1978) has been done, nor have the sediment delivery ratios of Roehl (1962) been validated. A later modification of the USLE by Dissmeyer and Foster (1984) was validated against observed sediment yield data from four plots (0.22 to 0.32 acre) and 35 small basins (0.5 to 2.5 acres) located in the Coastal Plain and Piedmont regions of the South (Dissmeyer and Foster 1984). This later model was similar to that used by Dissmeyer and Stump (1978)—the difference being in how the combined C and P factors in the two models were determined—and produced sediment yields very close to those observed ($R^2 = 0.90$). However, the EASY model is typically applied over much larger areas than those used in validating the Dissmeyer and Foster (1984) model.

Like the EASY model, the ACE model output has not been directly validated, nor has the soil loss component based on Dissmeyer and Stump (1978), or the sediment delivery component based on Roehl's (1962) sediment delivery ratios. Soil loss predictions based on WEPP have been validated for several types of forest (Elliot 2001, Elliot and Foltz 2001), nonforest (Laflen and others 2004, Soto and Díaz-Fierros 1998), and road (Grace 2005, 2007; Elliot and Tysdal 1999) conditions, but not for areas within the Ouachita and Ozark-St. Francis National Forests. Although the risk ratings have not been independently tested, they are based on actual fish collections from 178 different locations within the Arkansas ecoregions that encompass the Ouachita and Ozark-St. Francis National Forests (see footnote 1). The size of this sample and the fact that these data were collected within the same ecoregions as those undergoing assessment lends support to the assumption that the functional relationship proposed between the relative abundance of fish assemblages and predicted sediment yields is real.

## Disclosure of model shortcomings

The available documentation for both models lacks a thorough discussion of the shortcomings of the data sources, computation processes, and assumptions used to evaluate sediment CWEs. The current documentation of both models focuses on providing sufficient information to users so that they can understand how the model works and how to use it. Users are left to determine for themselves how well the reasoning behind the model stands up to current scientific knowledge, how complete are the data sources, and what counterarguments could be made to challenge the validity of each model.

## Reasoning behind significance interpretations

The two models take very different approaches to how significance is interpreted and justified. The ACE model provides an explicit procedure for interpreting model results by relating predicted sediment yields to relative abundance of fish assemblages. The method by which the relationship between predicted sediment yield and relative abundance, and how the risk levels are determined is explained in the model documentation (see footnotes 1 and 9). In contrast, the EASY model does not provide explicit interpretation of model outputs; rather it is expected that the user will decide how best to interpret the results based on each project's circumstances. Past applications of the EASY model have interpreted model results by comparing the percentage increases in sediment yield and by evaluating relative differences in sediment concentration, but these comparisons are not part of the standard model output. Although users bear the responsibility of justifying how they interpret the EASY model results, they have the flexibility to tailor the interpretation process to best meet the needs of each analysis.

## *Mitigation Effectiveness*

This expectation could be addressed outside of whatever method is used to assess sediment cumulative watershed effects; however, a model could provide a very straightforward way of demonstrating mitigation effectiveness. Of course, the same expectation for demonstrated validity would apply to modeling mitigation as applies to all other aspects of a cumulative effects model (table 1, expectations 6 and 9). The ACE model currently incorporates a limited capacity to calculate the effects of mitigation on erosion from proposed alternatives. The lengths of new or existing roads and off-highway vehicle or equestrian trails that are proposed to be closed, obliterated, or have only controlled use can be entered as part of an alternative; and the model will reduce soil losses computed for these areas based on the lower erosion rates. The EASY model also provides for reducing soil losses from temporary roads by specifying them to be closed after the activities requiring them are completed. Past EASY applications have also included additional condition types to assess the effects of closing trails, reconstructing roads to higher standards, and improving road surfacing (see footnote 4); although these refinements are not included in the model documentation.

# Future Modeling Issues

As our knowledge grows and our technology improves, there will be an ongoing need to periodically revisit and improve whatever models are used to assess sediment cumulative watershed effects. This need seems self evident and requires no further comment. What we think is worthy of comment are several issues related to sediment modeling because they significantly influence how future models will work and who will use them. These are often the issues that are set aside or overlooked in the urgency to develop and implement tools that are needed immediately. We may have overlooked other issues concerning assessment of cumulative watershed effects, but what we think is more important is that these issues be considered up front in future modeling efforts.

## *Appropriate Spatial and Temporal Scales*

In setting out guidance for how to accomplish a cumulative watershed effect analysis, the Council on Environmental Quality (1997) notes that choosing the appropriate geographic scale is critical. The choice of an appropriate temporal scale is, no doubt, of equal importance. The chosen scales should set the boundaries for the space and time within which a cumulative watershed effect will occur. A number of factors require consideration when choosing the spatiotemporal scales. These include the spatial magnitude and location of past, present, and future disturbances; how long it takes for ecosystems to recover from disturbances; how individual impacts might accumulate, feedback upon, or negate each other; the location and extent of the resource prompting the analysis; and the processes translating impacts through time and space (Tetra Tech 2002). Given the number and complexity of factors, the choice of the appropriate spatiotemporal scale for analysis will likely be unique for each situation (Bunte and MacDonald 1999). Furthermore, the situation, not the model, should determine the choice. Unfortunately, this is often not the case. Practical considerations have led to a single scale or two being selected and used for all situations. A major consideration behind the single-scale decision is the difficulty in compiling the spatial data for all condition types. Despite the widespread availability of GIS software, we still seem to lack the ability to readily generate the needed spatial data at any chosen spatial scale and output these data into existing sediment models. Another consideration that affects the time scale selected is the increased model complexity required to deal with activities scheduled over multiple years. No doubt these are significant challenges; however, they must be addressed if our models are to allow the selection of appropriate temporal and spatial scales based on the situation.

## Effects of Natural Disturbances

The inclusion of natural disturbances deserves more attention in future models. Models like EASY and ACE that use mean erosion rates are very limited in their ability to account for sediment produced by infrequent natural events, especially those, like hurricanes or wildfires, that affect extensive areas. Moreover, mean erosion rates, while believed to represent erosion produced over long time periods—50 years in the case of Dissmeyer and Stump (1978)—are generally derived from plot studies conducted over a limited number of years. The impact of severe storms is very likely not well represented within these values; thus estimates of sediment yield from undisturbed and disturbed areas may well be too low. Possibly these underestimates are proportional and thereby do not change interpretation of model results—the point is that we do not know. Future models will hopefully address more accurately the range of erosion amounts and how these are affected by natural disturbance processes.

## Impact of Past Human Activities

One issue that is particularly relevant to modeling sediment cumulative watershed effects in the Eastern United States is the impact of past human activities on current sediment dynamics. In many forested areas, the combination of highly erosive soils with abusive land-use practices beginning with European settlement and continuing through the early 1900s produced extensive areas of severely eroded terrain and massive sediment storage within drainage systems (Trimble 1974). Although improved practices and extensive reforestation have reduced soil loss significantly, a legacy of oversteepened slopes, compacted soils, and stored sediment remains in many areas, and can dramatically affect sediment production. Future models need to provide the capability to deal with such historical influences where necessary, and model users must be careful to consider and account for these influences when appropriate.

## Balancing Accuracy and Practicality

Future models, like EASY and ACE, will be developed through compromises between model accuracy and application practicality. Such compromises do not invalidate the use of such models. None of the legal expectations noted by Reid (2010) mandate use of a "perfect" or even "state-of-the-science" model; rather the courts expect that the analysis address the concern at relevant spatiotemporal scales, that the analysis and interpretations be reasonably thorough and scientifically defensible, that methodological validity be demonstrated, and that methodological shortcomings be disclosed. Therefore, our objective should be to produce the best model we can given the resources we have available. Resources would go farther if future models could be designed so that when new understanding emerges about erosion processes or sediment routing, the relevant model components could be revised without disrupting the unaffected components. Such models would require less resource investment over time and reduce the need to start from scratch to just those times when changes in the science or technology make such a decision desirable.

## Deterministic versus Lumped-Parameter Models

While lumped-parameter models like ACE and EASY may serve as satisfactory interim solutions for those areas where they can be validated, the future in sediment models probably lies in deterministic models. Deterministic models, like WEPP, provide both theoretical credence to the processes being modeled and a modeling structure that facilitates both computer programming and incremental refinements. In contrast, lumped-parameter models like USLE and its descendants are easier to program and require fewer data inputs, but they lack a direct theoretical basis and therefore must be validated through numerous empirical trials. The choice between deterministic and lumped-parameter models seems to us less a question of model accuracy than investment efficiency. If we have captured the relevant process mechanics correctly in a deterministic model, then scientific theory holds that these processes should function in the same manner at different locations. Testing is necessary to build confidence, but such testing should produce a model that is more broadly applicable because the process mechanics have been improved by making them more robust to input variations. In contrast, when testing shows lumped-parameter models to be inaccurate, all that can generally be done is to apply calibration factors that fine tune the model to the specific situation being tested, but produce no improvement for model applications elsewhere.

As they prove themselves, deterministic models should require less testing over time because we can better evaluate how changing environmental factors might affect model outputs as we better understand which factors most affect model behavior. This is more difficult with lumped-parameter models in which several environmental variables are often combined into a single factor (such as cover type) and it remains uncertain how variation in one or more variables might affect the combined influence of the lumped factor. Adopting deterministic models will likely come with a cost, however: it will require that model users have sufficient scientific knowledge to use them effectively.

### *User Competence*

Our assessment of these two models, plus our discussions with the model developers, leads us to question the ability of nonspecialists to adequately use these or other models for any but the most simple and straightforward applications. Both the EASY and ACE models were developed to facilitate evaluating sediment cumulative watershed effects for a variety of forest management projects that occur at varying spatio-temporal scales. Furthermore, these models were developed in part with the hope that personnel with adequate training—but not necessarily a background in sediment sciences—could use these models to perform sediment CWE sediment cumulative effects analyses. Most of the legal expectations identified in table 1 do not necessarily require improved skill on the part of the model user; they could be met through revisions to modeling procedures or improvements to model documentation. For example, expectations 2 to 5, and 9 in table 1 could be addressed by revising the current calculations and formats with both EASY and ACE. Items 7 and 8 could be addressed through more thorough model documentation. Item 6 (demonstrating model validity) would require some rigorous program of testing and analysis to assess model accuracy and to justify the accuracy standards that are chosen as being acceptable; however, such work would only have to been done where models have not been validated, standards are elevated, or new knowledge emerges to challenge past assumptions.

Conceivably, these improvements could all be made without requiring more knowledge or skill on the part of the user. However, the choice and justification of an appropriate analysis area (expectation 1 in table 1) can only be made by someone who understands the science of sediment processes, how these are affected by past and present land-use practices, how these processes are affected by the spatial and temporal scales at which they are evaluated, and how sediment is linked to other resources. The analyst must also appreciate how well we can actually model sediment and what that accuracy means in terms of our ability to judge the severity of cumulative sediment effects, because he or she will be the primary person interpreting the model results and defending the conclusions drawn from these results. It is unrealistic to expect that anyone without this knowledge and skill could apply and interpret these models effectively.

## Conclusions

Sediment appears to be the only concern related to cumulative watershed effects from fuel management being addressed in environmental analysis documents prepared by eastern national forests. Two types of analysis methods have been used: narrative analysis and hazard rating models. Narrative analysis is used when the level of concern is low and a discussion of the given environmental situation, the relevant scientific literature, and why the analyst thinks sediment effects are unlikely is deemed sufficient to meet legal expectations. Hazard rating models are employed when the level of concern is high. Two models are currently in use; the EASY model and the ACE model. Both models produce predictions of sediment yield at the outlet point of a watershed based on the condition types present or hypothesized. The models use the same general procedural steps, compute erosion for forest areas using similar data sources, and provide outputs for comparing alternatives. They primarily differ in how the analysis area is determined, how they compute nonforest and road erosion, and how results are interpreted. Both models provide much of the information needed to meet the legal expectations identified by Reid (2010), but both suffer from lack of validation.

Future models for evaluating sediment cumulative watershed effects will have to overcome current operational constraints that limit the ability to tailor analysis area delineation and spatiotemporal scale selection to the particular circumstances of each project. To improve their practicality, future models should be designed with updating and revision in mind. Although models like EASY and ACE may suffice in the near term for those regions where they can be validated, deterministic models would seem to offer a more efficient way to develop more broadly applicable tools to assess sediment cumulative watershed effects. Lastly, all analysis methods are merely tools that can only be applied effectively by practitioners who have the necessary scientific training to understand the strengths and limitations of the methods they employ.

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