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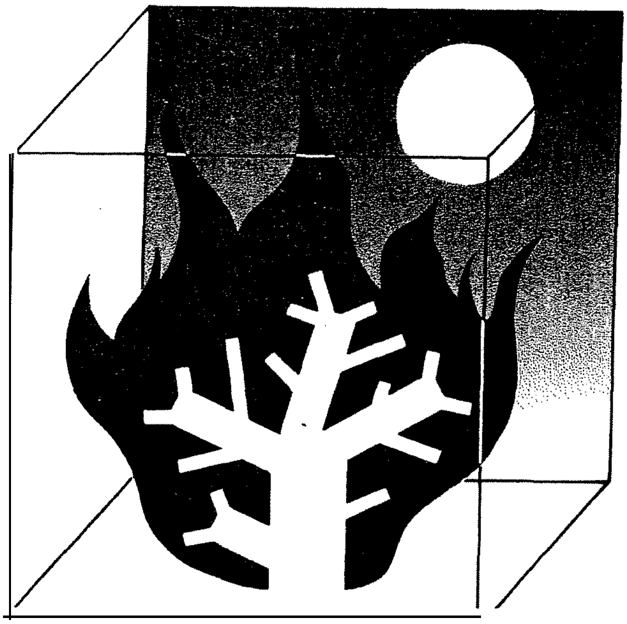
Fire and the Environment: Ecological and Cultural Perspectives



Proceedings of an International Symposium

Knoxville, Tennessee
March 20-24, 1990

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Forty-one papers based on oral presentations are included under four categories: Fire Ecology; Fire Management; Cultural; and Fire History. In addition, three papers are presented from a special session on the 1988 tires in the Greater Yellowstone Area and fourteen papers are presented from a poster session.

Fire and the Environment: Ecological and Cultural Perspectives

Proceedings of an International Symposium

Editors

Stephen C. Nodvin and Thomas A. Waldrop

Knoxyille, Tennessee
March 20-24, 1990

Sponsored by:

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PREFACE

Fire and the Environment: Ecological and Cultural Perspectives was an international symposium held on March 20-24, 1990, in Knoxville, TN. The meeting was attended by over 150 researchers, land managers, and wildlife managers. Forty-one papers based on oral presentations are included under four categories: Fire Ecology; Fire Management; Cultural; and Fire History. In addition, three papers are presented from a special session on the 1988 tires in the Greater Yellowstone Area and fourteen papers are presented from a poster session.

Papers and posters were selected by the program committee based on title summaries submitted prior to the meeting. The major objective of the editorial committee was to compile a proceedings covering a broad range of topics with papers representing new results, ongoing research, overviews of past research, and new ideas or hypotheses. Preference was given to papers covering cultural aspects of fire; such as public perception of tire, fire policy, wildland/urban interface, historical and prehistoric roles, fire and climate, use of fire toward management objectives, and effects of fue exclusion; and ecological effects of fire on climate, air quality, water quality, nutrient cycling, wildlife, fisheries, vegetation, and soils. After the meeting, papers were submitted to the editorial board for review. Each paper was given a blind review by two peers and one grammatical editor. Reviewer comments were incorporated by authors and submitted to the editorial board for approval. Some papers required additional revision but all papers were accepted. These proceedings have been prepared electronically from copy supplied by the authors. Authors are responsible for the content and accuracy of their papers as well as any stated opinions or conclusions.

The steering committee gratefully appreciates the efforts of authors and reviewers who contributed to a successful and informative program. Our appreciation is given to Brian Ostby and John Mullins, who arranged the poster session, and to Janet Paces, Ellen Williams, and Julie Smith, who served as assistants to the Program Chairman and proceedings editors. A special note of thanks is given to the moderators who provided additional insight to each topic and kept each Session on schedule. Moderators included William Boyer, Southern Forest Experiment Station; Bob James, USDA Forest Service, Region 8; Gary Schneider, The University of Tennessee; Eugene McGee, The University of Tennessee; Joe Abrell, USDI Park Service; Frank Woods, The University of Tennessee; Joe Clayton III, Tennessee Division of Forestry; Dale Wade, Southeastern Forest Experiment Station; David Van Lear, Clemson University; Larry Landers, Tall Timbers Research Station; Thomas Waldrop, Southeastern Forest Experiment Station; and Stephen Nodvin, USDI Park Service.

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VARIABLE FIRE REGIMES ON COMPLEX LANDSCAPES: ECOLOGICAL CONSEQUENCES, POLICY IMPLICATIONS, AND MANAGEMENT STRATEGIES

Norman L. Christensen, Jr.'

INTRODUCTION

Half a century ago, fire policy in most public and private agencies charged with the management of wilderness was neatly summarized in the so-called 10 A.M. Rule: If a fire starts, it should be extinguished by 10 the next morning. Our attitudes toward fire and other natural disturbances in wilderness landscapes have changed during recent decades. We now recognize that disturbances caused by fire, wind, insects, and pathogens play key roles in a variety of ecosystems processes. The folly of excluding or trying to exclude agents of disturbance from landscapes is now obvious to most wilderness managers.

Despite this knowledge, articulation of operational policies and management strategies for wilderness preserves has proven to be a daunting task. What arc the proper fire regimes for our diverse wilderness ecosystems? How and why have the frequency and behavior of tire changed through time? How have human activities such as a century of tire exclusion, landscape fragmentation, and alteration of ignition patterns affected fire regimes? How can we reintroduce fire into landscapes so altered? How can wilderness fire managers accommodate nonwilderness values such as recreation, timber and watershed resources, and air quality? Finally, how do we know when we are managing fire correctly?

In this paper, I shall argue that questions such as these can only be answered in the context of a clear understanding of wilderness processes and overall wilderness management objectives. I shall assert that wilderness management should be based on the answers to three questions, and since we have generally answered two of these questions incorrectly presents us with our most difficult management challenges.

NATURAL DISTURBANCE AND MANAGEMENT POLICY

Wilderness management can be reduced to answering correctly three questions: 1. What should be preserved? 2. How should preserves bc configured? 3. How should management be executed (Christensen 1988)?

What should we preserve?

The actual foci of preservation in particular wilderness ecosystems are often identified in nebulous, nonoperational terms or are not stated at all. The question must be answered in both philosophical and practical terms. We must first agree on the categories of items that will be the objects of preservation (i.e., genotypes, species, ecosystems, landscapes, etc.). Having made this decision, we must then determine which items within a category are worthy of preservation; that is, we must produce "shopping lists"--lists of rare and endangered species, or inventories of various ecosystem types, for example. The formulation of such lists is often the occasion for battles over the dedication of land to wilderness or nonwildemess management. In many regions, our ignorance of ecosystem variability prevents us from making such lists.

How should preserves be configured?

Patterns of natural disturbance have rarely, if ever, been a major consideration in the spatial configuration of wilderness preserves. However, if a wilderness preserve is to include the full range of patterns generated by natural disturbance, the frequency, area1 extent, and behavior of such disturbances must be considered. Ideally, a preserve should be sufficiently large to include not only the variety of post-disturbance age classes typical of a pristine wilderness landscape, but also the range of variation in disturbance severity. When disturbances occur on small spatial scales and at high frequency, small preserves will suffice. However, in environments in which large-scale tires are the norm (e.g., western coniferous forests and shrublands), few existing preserves are large enough. In addition, those who determine preserve boundaries should understand the effects of landscape features on the behavior of disturbances.

How should management be executed?

The details of procedures for managing natural disturbances depend on a variety of considerations. For unpredictable and uncontrollable disturbances such as the Mt. St. Helens eruption, management will focus on post disturbance intervention and will likely involve compromises between the wish to allow the affected area to recover as it would have in our absence and the potential consequences of the lack of intervention for areas outside the preserve (e.g., flooding, siltation, etc.). In the case of fire, where management intervention may alter the course of the disturbance, additional issues must be addressed. The manager must

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decide the extent of disturbance that is acceptable within the constraints of the design of the preserve. For example, assuming suppression is a real option, should a naturally ignited fire be allowed to become so large that containment is impossible or allowed to consume a major portion of the preserve? We know very little about the ecological consequences of artificially limiting the size of such disturbances. Are there wilderness processes that depend on the occurrence of fires of large spatial extent?

FIRE REGIMES AND ECOSYSTEM RESPONSE

Many of our views about the role of natural disturbance in general, and of wilderness fire in particular, reflect the evolution of ecological theories pertaining to succession, the recovery of ecosystems from disturbance. Clements' (1916) theory of succession portrayed wilderness as the touchstone of stability and order. Natural disturbances such as tire not only alter the structure of ecosystems, but render the landscape less habitable. Pioneer species that colonize such disturbed areas are usually organisms with high dispersal ability and considerable tolerance of harsh conditions; such organisms are rarely effective competitors under favorable growing conditions. Clements posited that these vagrant species become established and alter the environment of a disturbed site by stabilizing and enriching soil and by creating a microclimate that favors establishment of other species. Similarly, these new invaders alter their environment so as to favor yet another wave of immigrants. This succession of species continues until a community of organisms is established that maintains its environment in such a way as to perpetuate itself. This so-called climax ecosystem was considered to be the most stable assemblage of organisms that could exist on a given site within the constraints of regional climate.

Thus, ecosystems develop much like individual organisms from simple beginnings toward increasing complexity and stability. Ecologists of Clements' era recognized that individual organisms would necessarily die, but asserted that the regeneration of climax community species was associated with small-scale disturbances such as the deaths of individual trees.

Over spatial scales of interest to wilderness managers, the composition and structure of climax ecosystems was thought to remain constant over long periods of time. In Clements' (1935) words, "Under primitive conditions, the great climaxes of the globe must have remained essentially intact, since fires from natural causes were undoubtedly infrequent and localized." The quantity of data supporting this theory was quite small However, we generally require little data to support theories that portray the world as we wish it to be. It was indeed appealing to view wilderness as inevitably converging on stability. The vision of a climax ecosystem as a "super organism" composed of species performing specific functions as if they were roles much like the organs and tissues of an individual organism was proof of the "balance of nature."

This model of ecosystem change provided clear guidelines for management of wilderness. The object of management (*what* should be climax ecosystems. The supposed structure of attributed to climax communities was often based on romantic accounts by early explorers and naturalists. The view that species comprising natural communities were regenerated by small disturbances implied that questions about preserve configuration could be answered by reference to economic and political factors. If the dominant species can reproduce in openings created by the demise of individual trees, then relatively small areas should be sufficient to perpetuate the entire community. Consequently, the boundaries of our major wilderness parks have very little in common with natural ecological barriers or divides. Finally, it was clear that the key to preserving wilderness areas (*management execution.?)* thus defined was the prevention of catastrophic disturbances. Only in this way could we nurture wilderness to its climax state.

Research over the past three decades has taught us that Clements' theory was at best too simple and at worst flat wrong. Long-term studies demonstrate that the process of ecosystem change is not nearly so predictable or simple as Clements imagined (See Christensen 1991 for a review).

Perhaps the most startling discovery was that many ecosystems become increasingly ***unstable*** during succession, particularly in areas where fire is an important factor. Early in the development of most forests there is an abundance of green, moist, and relatively non-flammable plant tissue. However, as communities develop, dead woody debris accumulates increasing the ability of such communities to carry a fire. Exclusion of tire from such systems may result in additional fuel accumulation, increased flammability, and higher fire intensity when fires eventually occur. We have learned that fire is an essential and inevitable agent of biomass decomposition in many forest ecosystems.

If succession following disturbance is not a linear, predictable process that terminates with the establishment of a stable climax community, perhaps we can view it as a deterministic and stable cycle, driven by internal feedbacks such as fuel accumulation and inevitable fires. In this view, ecosystems are dynamically stable, like a pendulum with a period determined by return times between disturbances.

Alas, wilderness landscapes are not so easily modeled. Disturbances are indeed regulated in part by internal feedbacks, but actual periods or return times are quite variable and heavily dependent on extrinsic factors such as variations in regional climate. Furthermore, variations in disturbance intensity result in highly variable trajectories of postdisturbance change. Thus, the pendulum may swing in a highly chaotic random fashion. Wilderness landscapes are best viewed as an ever-changing "patch mosaics," and the frequency distribution of patch types may vary as a consequence of short- and long-term changes in climate and other chance factors (See Pickett and White 1985).

The pattern of successional change from less to more habitable conditions proposed by Clements is often reversed in the case of postfire ecosystem succession. As forests develop following fire, mineral nutrients such as nitrogen and phosphorus become less available in the soil as they are taken up and stored in plant tissues. Thus, low fertility, organic debris in the forest floor, and shade cast by mature trees may severely limit opportunities for seedling establishment and growth of the young plants in late succession ecosystems. The ash raining onto burned soils is an effective fertilizer, and postfire microclimate often favors successful plant reproduction.

This pattern of environmental change in relation to tire has resulted in evolution of plant adaptations that concentrate reproductive effort in the period immediately following the tire. Such adaptations include underground buds that are protected from heat, production of seeds that germinate only when heated, cones or fruits that open only when heated, and production of flowers only following fire. The evolution of such adaptations has not simply made some ecosystems tolerant of fire; it has made them dependent on it. Burned areas quickly lose their bleak aspect and, by the end of a single growing season, are carpeted with new growth.

Describing the evolution of knowledge in another area, Samuel Clements quipped that "the researches of many commentators have shed considerable darkness on this subject, and should they continue we shall soon know nothing about it." What once appeared to be a tidy linear process leading inevitably to stability on relatively small spatial scales is now seen as a dynamic, chaotic, and complex pattern of change in which the word "stable" may have little if any meaning.

POLICY, MANAGEMENT, AND RESEA RCH IMPLICATIONS

Policy Considerations

Given the variability in tire history, tire behavior, fire effects, and fire responses, it is clear that fire policies must vary from preserve to preserve. Nevertheless, I suggest that successful policies will have three common characteristics: (1) clearly stated operational goals, (2) identification of potential constraints, and (3) recognition of the variability and complexity of the successional process.

It is not enough to acknowledge that tire is an important natural process in an ecosystem and then simply reintroduce tire to the ecosystem. We must formulate specific operational goals for fire management programs. We do not set aside wilderness preserves in order to bum them. Rather we should withhold, apply, regulate, and respond to tire in order to accomplish specific management goals. The specification of these goals is made more difficult by the complexity of change on many landscapes.

The difficulty of setting operational goals is illustrated by the problems that the National Fire Management Policy Review Team (Philpot 1988) encountered when it attempted to define the specific goals of the Fire Manager Programs in federal wilderness areas. The overall goal was relatively simple--i.e., "to restore fire to a more natural role." But this formula begs the question "What is natural?" The Review Team defined natural as "those dynamic processes in components which would likely exist today and go on functioning, if technological humankind had not altered them." Putting aside the implication that Native Americans lacked technology, this statement seems to suggest that if natural processes are simply allowed to operate, ecosystems will converge to some preferred state. Although the details are far from clear, we are beginning to appreciate that landscape change is more chaotic than convergent.

Specification of objectives requires a clear understanding of the specific elements for which a preserve was dedicated. These may include historical features, species preservation, or preservation of entire wilderness areas. With regard to specific objects or species populations, policy objectives will likely be clear. However, a great deal of confusion exists regarding what constitutes wilderness. Wilderness is usually defined in contrast to human-altered landscapes, where wilderness represents the lack of human intervention. Given this definition, the phrase "wilderness management" should be considered an oxymoron. However, the pervasiveness of human influence ranging from the dissection and fragmentation of landscapes to global climate change may create conditions in which the most potent form of human intervention may be restraint.

We cannot simply set aside a piece of real estate and expect that, in the absence of human intervention, "those dynamic processes and components" will go on functioning as if "technological humankind had not altered them." We have created a world that we are obliged to manage. Given this situation, we are obliged to formulate policy based on operational definitions of wilderness. In particular we need to be explicit about such goals as preservation of ecosystem processes, biodiversity, and heterogeneity.

It is essential that policymakers understand the potential constraints on management in wilderness preserves. Within the realm of the "natural," a wide variety of landscape configurations is possible. However, within the constraints of preserve design, not all these configurations are equally desirable. Million-acre fires may be natural phenomena on the Yellowstone Plateau, but the desirability of such fires in the context of the altered landscape can be determined only by evaluating the costs and benefits of events on this scale.

In many cases, policymakers arc faced with competing or conflicting preserve objectives. For example, the Organic Act of 1916 that established the National Park Service extols managers to "conserve the scenery in natural and historic objects and the wildlife therein and to provide for the enjoyment of the same in such a manner and by such means as will leave them unimpaired for the enjoyment of future generations." It does not take a lawyer to detect the multitude of ambiguities and possible interpretations in this statement. For that matter, the 1963 Leopold Committee's assertion that the proper goal of national park conservation is the preservation of a "vignette of primitive America" is open to various interpretations. Some would view it as a mandate for the so-called "living museum" approach to park management, where the goal is to preserve "snapshots" of the past. Alternatively, the term vignette can be defined as a "moving picture" so as to include the process orientation of current Park Service policy.

The constraints on tire management posed by liability to other public and private resources are considerable. This is particularly true in wilderness preserves where goals include recreation or watershed management, and in situations where arbitrary borders separate wilderness from land dedicated to nonwildemess functions. The constraints on conservation of wilderness in an increasingly urbanized context are exemplified by issues such as air quality and smoke management. For example, burning in Sequoia National Park contributes to air quality problems in California's Central Valley. It may bc natural that wilderness fires inject particulates into the atmosphere, but the emission of these particulates may be deemed unacceptable by air quality authorities who must consider that the atmosphere is already polluted with a host of anthropogcnic emissions.

Fire policies must recognize the constraints set by preserve design. We have chosen to preserve relatively little of once vast expanses of wilderness, and the borders of most preserves bear precious little relation to the natural processes necessary for their preservation. The acceptability of fire events of particular intensities or spatial extent cannot be based solely on the naturalness of such events. Given the constraints of preserve design, many natural events may be deemed unacceptable or at least undesirable. This is particularly true where we can preserve only small fragments of formerly large landscapes. In these situations it is important to understand the ecological costs of not allowing large scale or high intensity events to occur.

Perhaps the most significant constraint on policy development is ignorance. Stewards of wilderness cannot claim, nor does the public have a right to expect, perfect knowledge. The only fair expectation is good faith. Policy makers and the public must understand the limits of our understanding.

When I was in my late teens, my grandmother took me to task for some transgression, the specifics of which now escape me. My excuse was that I was just trying to do what I thought was right. Grandma's reply? "Norman, you should not do what you think is right, you should do what is right!" I thought for a moment and said "Grandma, that's the dumbest thing i have ever heard."

The public's expectations are much like my grandma's, Managers must be ever cognizant of what I shall call Grandma's Law: "All we can ever do is what we think is right.' However, what managers consider right has changed markedly as our understanding has developed over the past several decades. Thus, I propose a corollary to Grandma's Law: "Just because you think you are right does not guarantee that you are.

Fire policies must allow for the variability and complexity of the process and its context. We are learning that variability is an essential component of fire regimes and that policies should not necessarily seek to replicate mean values of intensity, return time, etc. Furthermore, policy options and goals will vary considerably across the spectrum of fire regimes. Because we can prescribe low intensity tires like those that occur in grasslands with high scientific precision, we expect similar precision in the application of tire in heavy forest fuels. However, our management options in these latter situations may be more akin to those for large scale disturbances such as hurricanes and volcanic eruptions.

Finally, policies must be developed against a backdrop of constant change. In his classic paper on succession, Henry Chandler Cowles (1901) characterized succession as "a variable converging on a variable." Given the pervasiveness of human-caused environmental change, the notion of "natural" may be moot at best.

Management Considerations

Management involves the development of interventions to achieve specific policy objectives. Recognizing our considerable uncertainty and ignorance about the processes wc must manage, management should be thought of as a direct application of the scientific method. Its success depends not only on a clear understanding of available options (hypotheses), but also on a monitoring system that provides direct feedback to managers regarding management consequences (experiments and tests).

Fire management options include complete suppression, planned-ignition prescriptions, natural emission prescriptions, "let bum" strategies, and a range of fire surrogates. Where complete fire suppression is necessary, guidelines for managing the effects of suppression are critical. Such effects include those of employing tire retardants, plow lines, and heavy equipment. Where fires cannot be allowed to burn, managers may need to consider surrogates for burning such as mechanical field manipulation and artificial cutting.

Planned emission prescriptions must differentiate between the means and the ends. Historically, most prescribed burning protocols have been developed in the context of siivicultural management in which the end goals are field reduction and discouragement of competitors (i.e., reduced diversity). Management goals in wilderness areas will likely be quite different and require different burning protocols. In developing bum plans it is important to distinguish between fires set to restore fuel conditions to some "natural" state and fires set to simulate a "natural" process.

Prescribed natural ignition programs allow fires ignited by natural causes to bum so long as they are within prescribed guidelines. In a sense, such fire management programs substitute knowledge for intervention. They assume that threshold levels of fire behavior can be established beyond which tires can and should be suppressed. These is serious question whether such fire programs are realistic and natural. For example, such plans may call for the suppression of ecologically important but intense fire events. Furthermore, given the extent of landscape fragmentation and alteration, it is unlikely that fire regimes developed in this manner will simulate the full range of natural processes.

Only in the largest wilderness areas will an unmodified "let bum" fire management plan be a viable alternative. Nevertheless, wildfires will occur and fire management plans must provide clear guidelines for specific postdisturbance interventions. These guidelines must include appropriate measures for erosion mitigation, reforestation, and wildlife management interventions. Those who formulate guidelines for postfire interventions should consider the benefits of the intervention and their environmental and monetary costs, and should consider the likelihood of their success.

Any reasonable management system must have a built-in program for evaluating management's success in accomplishing policy goals. Such a monitoring program should be viewed as a set of research hypotheses especially designed to test whether management is providing the desired effects on specific dependent variables such as fire diversity, decomposition and nutrient cycling, and landscape heterogeneity. Monitoring programs not only provide information that can be used to adjust to management protocols, but also serve to inform basic scientific research programs.

Research Needs

it is clear that there is much that we do not know regarding the role of fire in wilderness ecosystems. I feel three areas deserve special attention. (1) There is much to learn about the causes and consequences of variability in fire regimes. For example, the Yellowstone fires taught us that models of fire behavior are not easily transferred among ecosystems. Even within a landscape, interactions between climate and fuels may result in multiple patterns of fire behavior. The consequences of variation in fire behavior are also little known. Fire often results in a pulse of resource availability which, although cphcmcral, may greatly influence patterns of species establishment. The variability in such pulses may have much to do with the biodiversity of landscapes throughout the fire cycle. (2) Although we know that variation in the spatial and temporal scale of fire events can greatly affect patterns of species response, the specifics of such patterns and their mechanisms are poorly understood. Implicit in much of wilderness fire management is the notion that many small events can be substituted for a single large event, but in most cases this assertion has not been tested. (3) Given the constraints on fire management programs, it is important to understand the ecological consequences of departures from "normal" fire regimes. Are them ecological costs associated with the exclusion of high intensity fires from certain parts of the landscape?

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FIRE ECOLOGY

EFFECTS OF FIRE EXCLUSION ON TALLGRASS PRAIRIE AND GALLERY FOREST COMMUNITIES IN EASTERN KANSAS

Marc D. Abrams and David J. Gibson[1]

Abstract-The purpose of this review is to synthesize a long-term body of research dealing with fire exclusion effects on tallgrass prairie and gallery forest communities on Konza Prairie in eastern Kansas. Upland and lowland prairie communities burned in spring at intervals ranging from 1-11 years were consistently dominated (70-90 percent cover) by Andropogon gerardii. With this increasing interval between fires other dominant warn-season grasses, A. scoparius and Sorghastrum nutans, had decreased cover, whereas forbs and woody species had increased cover. Aboveground biomass was higher on an annually burned versus unburned lowland prairie, due to stimulated graminoid production. Sites unburned for 10 or more years were converting to woodlands dominated by Juniperus, Ulmus, Gleditsia and Celtis. Older gallery forests occurred in stream channels and ravines and were comprised of overstory Quercus and Celtis and understory Theise Cercist and fUlmus llery forests on Konza Prairie dramatically increased from the time of European settlement (18.50) to present; this has been attributed to decreased tire frequency and intensity in the region. With continued fire exclusion this century further succession in these forests has caused oak replacement by more shade tolerant species.

INTRODUCTION

Eastern Kansas receives approximately 33 inches (83.5 centimeters) of precipitation annually, which is enough to sustain forest vegetation on all but the most xeric sites. Historically, however, the region has been dominated by Andropogon Sorghastrum Wanicum tallgrass prairie. n forest vegetation does occur it is usually restricted to thin bands along ravines and stream channels, called gallery forests. It is well recognized that frequent fire in the region limits woody vegetation expansion and helps to maintain tallgrass prairie (Abrams 1988c). Moreover, the composition, structure and productivity of tallgrass communities can vary dramatically with relatively small changes in life frequency (Abrams and Hulbert 1987, Gibson 1988). Following European settlement in the mid-1800s, the number, extent and intensity of fire most likely decreased in eastern Kansas, resulting in changes in the ecological characteristics of prairie and gallery forests in the region (Bragg and Hulbert 1976; Abrams 1986). Similar changes in woody and prairie vegetation occurred in eastern Oklahoma after decades of fire suppression (Collins and Adams 1983).

Since the late 1970s the effects of fire and fire exclusion in tallgrass prairie and gallery forest communities have been studied on Konza Prairie in northeast Kansas. Studies conducted in one or both community types include fire effects on plant species composition, structure and productivity. The purpose of this review is to synthesize these studies with special reference to the effects of fire exclusion the grassland and forest communities on Konza Prairie.

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KONZA PRAIRIE

Konza Prairie Research Natural Area is 3,487 hectares of tallgrass prairie habitat in Riley and Geary counties in the Flint Hills of northeast Kansas. The Flint Hills are along the western border of the tallgrass prairie province and because of the steep and rocky topography include the only extensive area of unplowed tallgrass prairie in North America. Gallery forests in the region are dominated by Quercus spp. (oak) and Celtis occidentalis (hackberry) and range from about 10 to 300 meters wide in protected portions in the prairie interior.

Specific methods for prairie and gallery forest data collection and analysis can be found in papers by the authors cited hereinafter. Prairie composition and productivity studies were concentrated on 10-100 hectare watershed units, burned in mid- to late April at various intervals since 1972. Each watershed contains relatively broad, level upland (Florence cherty silt loam or cherty silty clay loam soils) and lower slope (Tully silty clay loam soils) sites. Florence soils are relatively thin, well-drained and have numerous chert fragments in the top soil, whereas Tully soils are deep, gently sloping and were formed from thick colluvial and alluvial deposits, with few rocks (Jantz and others 1975). Because the upland Florence soils store less water than Tully soils, plants present become water-stressed sooner during dry periods than plants on deep Tully soils (Abrams and others 1986).

The climate of the study area is continental, characterized by hot summers, cold winters, moderately strong surface winds and relatively low humidities (Brown and Bark 1971). The average length of the frost-free season is 180 days. Mean annual temperature was 12.8" C (range = -2.7 to 26.6" C) and mean annual precipitation was 83.5 centimeters based on a 30 year record (1951-1980). Precipitation ranges from 2.1 to 13.4 centimeters per month with May and June being the wettest and December- February being the driest months.

Annual precipitation varies greatly and droughts occur frequently. In northeast Kansas, drought occurred during 38 percent of the months between 1931 and 1968 and of these 11 percent were rated severe or extreme (Brown and Bark 1971).

COMPOSITION OF PRAIRIE

Presettlement Conditions

Although accurate presettlement conditions are not known, it is generally accepted that the tallgrass prairie developed and spread in an environment that included tire at frequent intervals in the range of once every year to once every ten years (Kucera 1981; Axelrod 1985; Hulbert 1973). In the Kansas Flint Hills, frequently burned tallgrass prairie is dominated by big bluestem (Andropogon gerardii) and other tall warm-season grasses (e.g., Andropogon scoparius, Sorghastrum nutans, and Panicum virgatum). Species along with a number of forbs, e.g. Solidago missouriensis and Vemonia baldwinii, occupy the majority of space in the community and are referred to as matrix species (Collins and Gibson 1990), whereas a large number of rarer interstitial species occupy the spaces, e.g. Ambrosia psilostachya, Artemisia ludoviciana.

Effect of Fire Exclusion on Species Composition

On Konza Prairie, studies of permanent plots carried out since 1981 (Abrams and Hulbert 1987, Gibson and Hulbert 1987; Gibson 1988) indicate that in the absence of grazing species richness increases with time since fire until approximately 8 years, after which richness declines again (Gibson and Hulbert 1987; Collins and Gibson 1990). This effect was only observed in the context of the long-term study and was not necessarily discernible on a yearly basis (table 1). Fire exclusion from the tallgrass prairie allows the build up of a soil seed bank (Abrams 1988a), which along with the more suitable microsite conditions and heterogeneous community structure (Collins and Gibson 1990) results in a more species-rich community.

Andropogon perardii is the dominant species (cover = 70-90 percent) on Konza Prairie irrespective of fire treatment or topography (table 1). Nevertheless, the cover of A. scooarius, S. nutans and other warm-season species decrease significantly with time since burning (table 1) (Abrams and Hulbert 1987; Gibson and Hulbert 1987). In contrast, Panicum virgatum showed no response to fire but had higher cover on the deeper, moister soil of lowland sites. Cover of

Table 1. Community and plant species cover data (x- percent + standard error) for three unburned and two annually burned upland (Florence cherty silt Loam) and lowland (Tully silt clay Loam) soils on Konza Prairie in northeastern Kansas in 1984. Values for each community parameter or species followed by the same Letter are not significantly different: * • values (0.05 percent. (After Abrams and Hulbert 1987)

Community data	Burned treatment	Lowland	Upland
Total species cover	burned	176.5 + 3.2a	179.0 + 4.9a
	unburned	164.0 + 6.4a	171.4 + 7.1a
Species richness	burned	16.3 + 1.0a	20.1 + 0.8b
	unburned	18.3 + 0.9a	21.2 + 0.7b
species/treatment	burned	46.0 + 2.8a	54.0 + 0.7a
	unburned	57.0 + 2.8b	60.3 + 2.4b
Species data			
-Grasses-			
Andropogon gerardii	burned	83.5 + 5.1a	80.6 + 2.2a
	unburned	84.5 + 2.2a	79.6 + 2.5a
A. scoparius	burned	43.3 + 4.4a	24.0 + 4.9b
	unburned	5.4 + 1.4c	8.7 + 1.3c
Sorghastrum nutans	burned	21.5 + 3.0a	33.4 + 4.6b
	unburned	4.4 + 1.0c	3.8 + 0.9c
Panicum virgatum	burned	12.5 + 4.5a	5.6 + 2.1bc
	unburned	5.6 + 2.4ab	1.9 + 0.9c
Poa pratensis	burned	0.0 a	*a
	unburned	21.9 + 4.0b	24.9 + 4.8b
-Forbs and woody plants-			
Aster ericoides	burned	1.3 + 06a	0.5 + 0.3a
	unburned	a.7 + 2.0b	7.8 + 1.7b
Salvia azurea	burned	*a	14.1 + 5.0b
	unburned	*a	1.1 + 0.3c
silostacha	burned	2.8 + 1.1a	2.5 + 1.1a
	unburned	10.2 + 2.3b	8.2 + 1.56
Artemisia ludoviciana	burned	0.1 + *a	0.1 + *a
	unburned	12.6 + 2.9b	a.0 + 1.7b
Amorpha canescens	burned	4.4 + 1.9a	1.1 + 0.4b
	unburned	0.3 + O.lb	1.6 + 0.56

the dominant cool-season grass Poa pratensis, was not affected by topography but was greatly reduced by annual burning (table 1). The sensitivity to burning of this and other cool season species is due to the loss of terminal growth from spring burning (Abrams and Hulbert 1987). Warm-season species are still dormant during spring burning and do not show such sensitivity.

Cover of most forb and woody species increased with fire exclusion. Salvia azurea (= S. pitcherii) and Amorpha canescens are exceptions to this, with cover being significantly higher on annually burned upland and lowland soils, respectively (table 1). Artemisia ludoviciana, Ambrosia psilostachva and Aster ericoides are the dominant forbs on fire excluded sites. Overall, woody species and forb species cover increase with time since burning (Abrams and Hulbert 1987; Gibson and Hulbert 1987).

Multivariate analyses of species cover (Gibson and Hulbert 1987; Gibson 1988) have indicated that the species show an individualistic response to fire frequency and topographic position (fig. 1). This indicates that although it is clear that fire exclusion from the tallgrass prairie leads to an increase in the cover of many species, especially forbs (table 1), the rate of increase varies between species in a manner typical of secondary successions (e.g., Pickett 1982). In contrast to such typical models however, different species do not successively attain and then lose predominance. Rather, A. gerardii remains the dominant species throughout. s likely a reflection of the fact that given continued fire exclusion, grass dominated prairie is not the end-point of the successional pathway. indeed, studies in Oklahoma indicate an eventual dominance of tallgrass prairie by woody vegetation after 32 years without fue (Collins and Adams 1983).

Sites that are burned every four years show cyclic fluctuations in community composition, although soil effects and landscape heterogeneity show a stronger relationship to the plant community (Gibson 1988). Ungrazed prairie maintained under such a frequent burning regime on Konza Prairie is considered to be perhaps as comparable to presettlement conditions as is possible under present day constraints. Exclusion of fire for three year periods under this regime is, however, sufficient time to allow for an invasion of woody species (Briggs and Gibson, unpublished data).

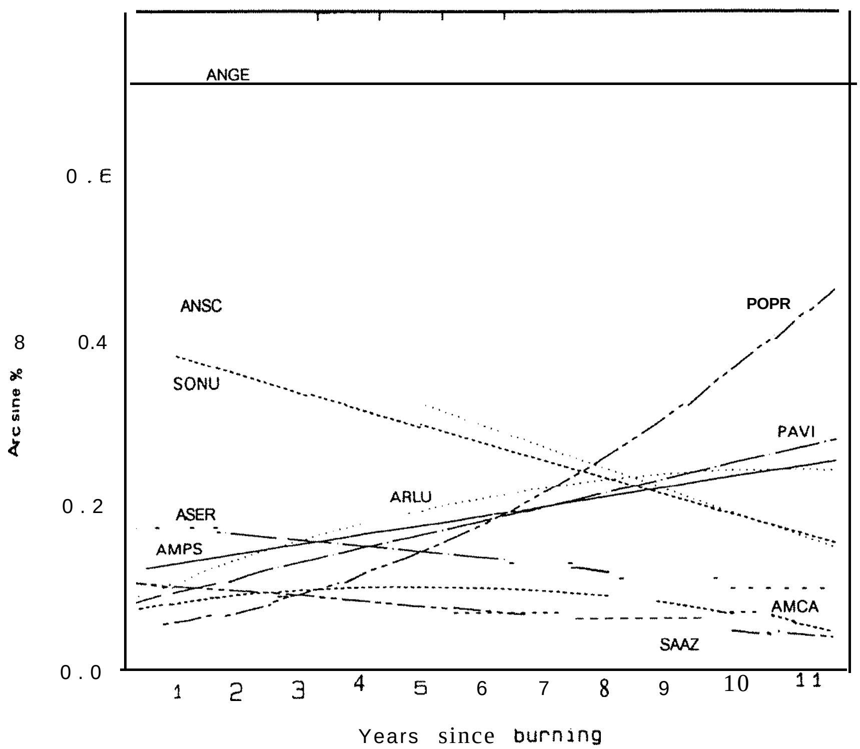

Fig. 1. Fitted 1st and 2nd degree polynomial regression lines of species distribution along a tire interval gradient identified by ordination analysis (Detrended Correspondence Analysis) (After Gibson and Hulbert 1987). ANGE = Andropogon gerardii; ANSC = A. scoparius; SONU = Sorghastrum nutans; PAVI = Panicum virgatum; POPR = Poa pratensis; SAAZ = Salvia azurea; AMPS = Ambrosia osilostachva; ARLU = Artemisia ludoviciana; AMCA = Amorpha canescens; ASER = Aster ericoides..

Prairie Productivity

Substantial differences in the seasonal (1984) production of aboveground biomass by graminoids and forbs were evident between an annually spring- burned and unburned watershed on lower slopes (fig. 2). Peak standing crop of aboveground production was significantly greater in the burned (430 + 26 grams per square meter) than unburned (368 + 31 grams per square meter) watershed. This difference between burned and unburned lowland prairie is consistent with the results of long-term studies of productivity on Konza Prairie (Abrams and others 1986; Briggs and others 1989). Peak live graminoid biomass was also greater in the burned (285 + 20 grams per square meter) than unburned (205 + 22 grams per square meter) site, whereas forb and woody plant biomass was typically two-three times greater in the unburned (maximum 94 + 15 grams per square meter) than the burned watershed (maximum 45 + 13 grams per square meter). Woody plants, the smallest component of the total, contributed little to total production. Both aboveground production and the live graminoid component showed a mid-season peak in late July-early August. In contrast, the biomass of forbs and woody plants showed little seasonal variation.

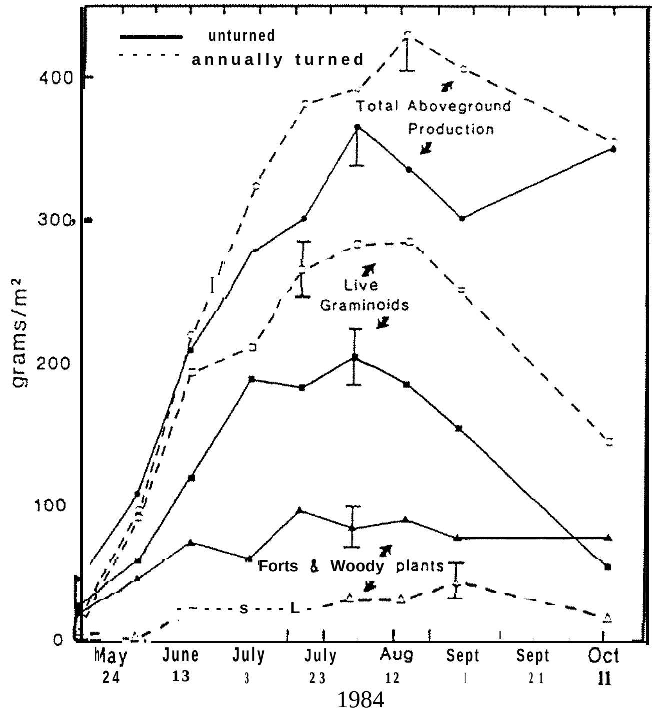

Fig. 2. The seasonal pattern of several components of aboveground biomass on a burned and unburned lowland soil during 1984 on Konza Prairie . Total aboveground production includes live graminoids, forbs, and woody plants, and current year's dead biomass. Vertical bar represents + standard error of the mean. (After Abrams and others 1986).

Invasion of Tree Species into Open Prairie

Woody species will rapidly invade open prairie in the absence of fire and grazing, given a sufficient time and local seed source (Gleason 1913; Weaver 1960; Grimm 1983; Bragg and Hulbert 1976). On Konza Prairie, the invasion of trees has been documented since 1971 by direct stem counts on over 500 hectares of open prairie. The principal tree species are triacanthos, Populus deltoides, Salix spp., Ulmus americana, and Celtis occidentalis (table 2). In frequently burned prairie, densities of 2-3 trees per hectare have been recorded, while in areas where tire has been excluded for 10 or more years densities range from 12 to '77 trees per hectare. Over a five year period, recruitment of all species into long-term unburned areas is 6-7 individuals per hectare (Briggs and

Table 2. Stem density, species number, diversity and dominant species (> 15 percent of total stem density) of tree species in the open prairie on annually burned and unburned (> 10 years) watersheds on Konra Prairie in 1986. Watersheds N4D and NIB contain large areas of gallery forest along the stream channels.

Site and Burn Treatment	Tree Density (number per hectare)	Number of Species	Diversity (H')	Dominant Species (Density per hectare)
Annually Burned				
1A	2.5	6	1.23	Gleditsia triacanthos (1.5)
ID	2.9	4	0.94	Poputus deltoides (0.5)
Unburned (IO years)				
UB	12.0	a	1.68	Salix sp. (4.1) Ulmus americana (2.8) G. triacanthos (0.7)
UC	29.6	9	1.28	G. triacanthos (12.6) P. deltoidas (12.6)
N4D	54.7	18	1.50	U. americana (28.5) G. triacanthos (8.8)
N1B	76.9	13	1.20	U. americana (48.0) Celtis occidentalis (12.1)

Gibson, unpublished data). In areas of open prairie adjacent to stream channel gallery forests, Ulmus americana and Celtis occidentalis are the dominant invasive species. This is a reflection of their importance in the gallery forests (Abrams 1986). Other gallery forest dominants such as Quercus muehlenbergii, Q. macrocama and Cercis candensis are only occasionally found in open prairie. The ability of these forest species to invade open prairie is related to their physiological ability to withstand the relatively more xeric open prairie habitat (Abrams 1988b). In areas further removed from the gallery forests, fire exclusion leads to an increase in the density of species that normally persist in frequently burned prairie albeit at low densities along the stream margins, i.e. Cleditsia triacanthos. Populus deltoides, and Salix spp. (table 2). These are short-lived, early successional species common in river floodplains and stream courses (Bellah and Hulbert 1974).

The spatial pattern of species invading open prairie from which fire has been excluded is a function of species dispersal vectors. Species such as Junioerus virginiana, which are bird dispersed, show a random pattern of distribution. In contrast, wind dispersed species such as Ulmus americana show an aggregated pattern. Juveniles of all species are clustered around adults, but at a greater distance for the bird dispersed species. At the landscape scale, invading tree species are (except J. virginiana) associated with the stream channels.

Upstream of mature gallery forest, attenuated gallery forest, as seen on watersheds N4D and N1B (table 2) represents the first stages of gallery forest development in open prairie in the absence of tire.

GALLERY FOREST

Stand Classification and Ordination

Eighteen stands were method divided into four ecological groups along the polar ordination axis according to importance values of the three dominant species (fig. 3). Group 1 (stands 1,2,6,18) included Celtis occidentalis - Quercus macrocama dominated stands, with Q. muehlenbergii and Ulmus as subdominants. Group 2 stands were dominated by Q. macrocama (stands 3,9,10) or Q. macrocama and Q. muehlenbergii (stands 4,7) with lesser amounts of Celtis and Quercus muehlenbergii and Q. macroeama were the dominants and Ulmus and Cercis canadensis the subdominants in group 3 (stands 5,8,11,12,14,16). Ouercus muehlenbergii dominated stands in group 4 (stands 13,15,17), with Q. macrocarna, Cercis, and Ulmus as subdominants. Stand positions along the polar ordination axis were highly correlated with increasing slope and decreasing silt, which may be interpreted as a moisture gradient from mesic (group 1) to xeric (group 4) (Abrams 1986).

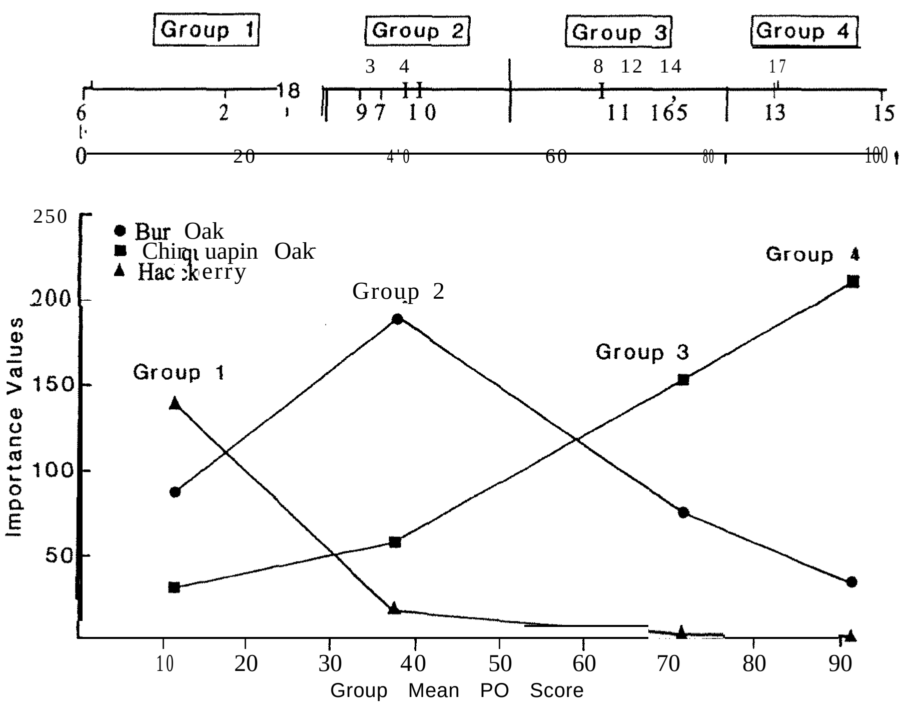

Fig. 3. Polar ordination analysis and the mean importance values for the dominant species in 18 gallery forest stands on Konza Prairie. The four stand types are identified.

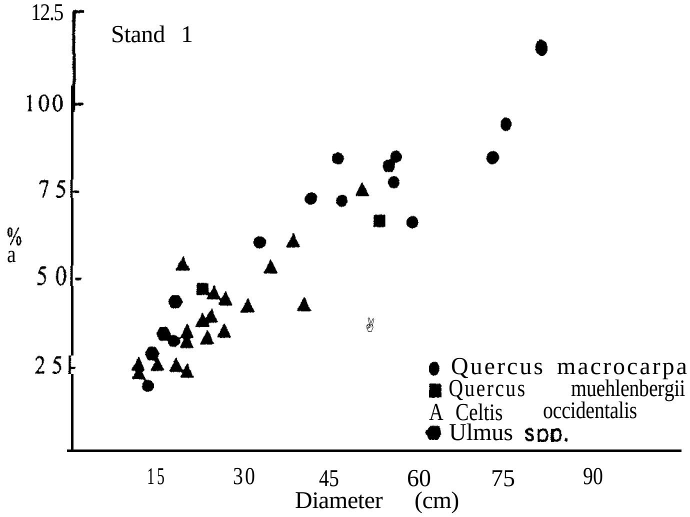

Fig. 4.
Age-diameter data
for gallery forest stand 1
on Konza Prairie.
(After Abrams 1986).

Age-diameter Data

Species age-diameter data from two representative gallery forest stands are shown in figures 4 and 5. Stand 1 (fig 4) is a Celtis • Q. macrocarua oak stand in which Q. macrocama stems were the largest and oldest present; most Q. macrocarna were over 40 centimeters diameter and 70 years old and formed an even-aged canopy. The size and age of Q. macrocarna was distinct from that of Celtis, which generally ranged from 10-40 centimeters diameter and 23-53 years old. In stand 8, a Q. muehlenbergii • Q. macrocarpa stand, oak species dominated the larger and older diameter and age classes, whereas Cercis, Ulmus and, to a lesser extent, Celtis dominated the smaller and younger classes (fig 5). A predominant age gap of 25-35 years separated these species groups.

HISTORICAL DEVELOPMENT OF GALLERY FORESTS ON KONZA PRAIRIE

Using data from the 1858 Original Land Office Survey of Konza Prairie and aerial photographs taken in 1939 and 1978, it was possible to determine changes in the extent of gallery forests during that 120-year period (fig. 6). In 1858 only two areas of continuous forest comprising about 5 hectares were noted. Occasionally, a few trees or scrubby timber and shrubs were recorded in other areas of Konza Prairie, especially along the stream channels and ravine bottoms, but in general this area was described as rolling prairie devoid of woody vegetation. The aerial photographs of Konza Prairie taken in 1939 and 1978 were in marked contrast to that described in 1858. During those periods a large expansion of the gallery forests to approximately 111 and 206 hectares occurred, respectively, with widespread invasion of shrublands and forests onto the prairie and development of shrublands into forests.

The distribution and overall ecology of the gallery forests on Konza Prairie has been greatly affected by anthropogenic factors. The limited extent of the gallery forests in 1858 was probably due to higher tire intensity and frequency prior to European settlement, which started about 1840 (cf. Penfound 1962). Following settlement, the number, extent and intensity of fire most likely decreased in the Flint Hills due to road construction, expansion of towns, agriculture, continuous cattle grazing, suppression of wildfire and recommendations against burning during the mid-1900s (Bragg and Hulbert 1976; Abrams 1985). Therefore, after settlement forests

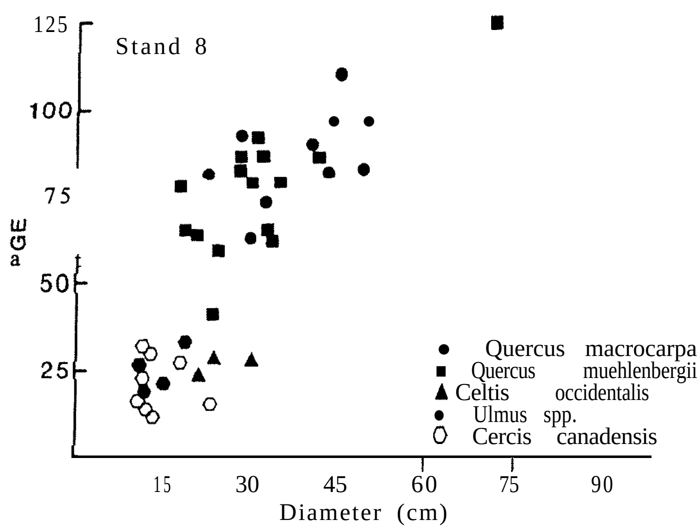

Fig. 5.
Age-diameter data
for gallery forest stand 8
on Konza Prairie.
(After Abrams 1986).

increased rapidly, which suggest that tire, rather than low precipitation, limited growth of woody vegetation in northeast Kansas (c.f Abrams 1988c).

It appears that a succession from shade intolerant Quercus species to the more tolerant Celtis and Cercis is taking place, despite these forests burning at intervals of about 1 I-20 years since the mid-l 800s (Abrams 1985; Abrams 1986). Quercus macrocarpa and/or Q. muehlcnbergii, which represented the largest and oldest species in each stand, showed very little recruitment into the tree size class for over half a century. In contrast, numerous Celtis, Ulmus and Cercis trees younger than 50 years old were present in these stands. On the mesic sites Celtis may be the future sole dominant. Already in group 1 stands (1,2,6,18) Celtis is the dominant despite it being younger and smaller than Q. macrocama. On the steeper, drier sites on Konza Prairie, where Celtis is rare, Cercis may replace Q. muehlcnbergii. The small stature of Cercis does not rule it out as a potential replacement species here because the size of Q. muehlcnbergii is limited on these harsh sites. Even though Ulmus is a dominant reproducer in nearly ail stands, its potential as an overstory dominant is probably limited by the Dutch Elm Disease. This blight was discovered in Kansas in 1957 and has depleted many area forests of mature elms (Thompson and others 1978). The less advanced successional status of the xeric versus mesic forests on Konza Prairie suggests that the rate of succession in xeric forests is constrained by harsher environmental conditions and/or higher fire frequency.

CONCLUSIONS

Subtle and gross changes in tire frequency dramatically alter landscape patterns on Konza Prairie in the absence of grazing Annual burning treatment resulted in the greatest dominance for warn-season tallgrass species. Less frequently burned areas develop progressively less cover of several dominant grasses. However, cover of Andropogon gerardii, the dominant prairie grass, remained relatively unchanged in upland and lowland prairie burned at 1- to 1 l-year intervals. With increasing intervals between fire, total cover of fort) and woody species increased. Lowland sites, especially along stream channels, unburned for 10 or more years show definite signs of conversion to forest dominated by Juniperus, Gleditsia, Ulmus and Celtis. established woodlands along stream channels and ravines had overstories dominated by Quercus and Celtis with Celtis Cercis and Ulmus in the understory. increased distribution of these gallery forests since 1850 and successional changes resulting in oak replacement by more shade tolerant species are attributed to reduced fire frequency. Thus, our work on Konza Prairie provides further evidence that fire is a primary factor controlling community composition, productivity, structure and successional processes in tallgrass ecosystems, and that a frequent fire interval and possibly grazing and periodic drought are necessary maintain tallgrass prairie in a "'pristine" condition.

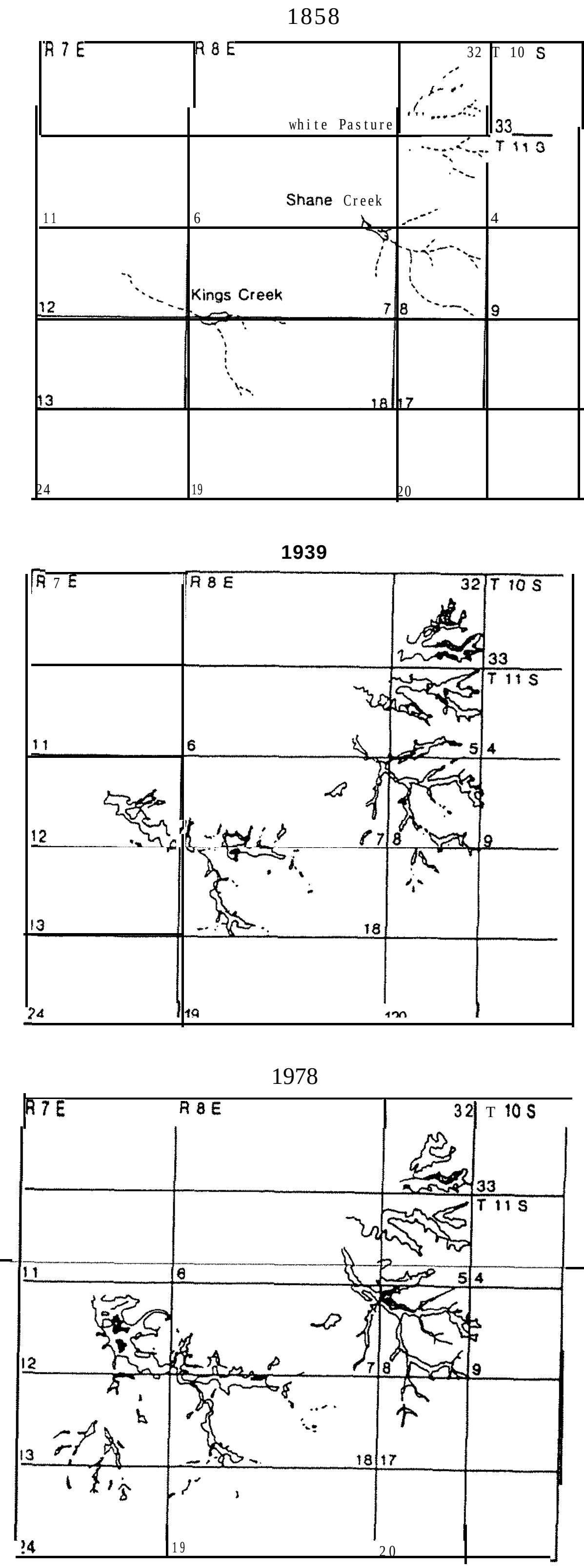

Fig. 6. Areal extent of gallery forests and shrublands (shaded) in 1858, 1939 and 1978 on Konza Prairie.

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FIRE MANAGEMENT FOR MAXIMUM BIODIVERSITY OF CALIFORNIA CHAPARRAL

Jon E. Keeley'

Abshact-Two reproductive modes present in chaparral shrubs are affected very differently by fire. Some species, called "tire-recruiters," are dependent upon fire for seedling establishment. These species have contributed to the notion that the chaparral community is dependent upon tire for rejuvenation. In the absence of fire, chaparral is often described in pejorative terms which imply that long unburned conditions represent an unhealthy state. However, many shrub species, called "fire-persisters," do not establish seedlings after fire, rather they require long fire-free periods in order to establish seedlings. These species are vigorous resprouters, not only after fire, but throughout their lifespan. Older stands of chaparral are continually rejuvenated by recruitment of new resprouts and seedlings of these fire-resister species. It is suggested that the long-term stability and diversity of chaparral requires a mosaic of tire frequencies.

INTRODUCTION

California chaparral is considered a "fire-type" vegetation based on the fact that all species are resilient to the modem fire regime of tires every few decades (Keeley and Keeley 1988). Resilience of the vegetation is reflected in the relatively minor changes in community composition resulting from fire. Species present before fire return rapidly afterwards, either regenerating aboveground parts from basal resprouts or by seedling establishment.

In addition to being considered a fire-type vegetation, chaparral is also often described as a fire-dependent vegetation. This is based on both population and community level phenomenon. Certain species, Adenostoma fasciculatum (Rosaceae), Arctostaphvlos spp. (Ericaccae) and Ceanothus spp. (Rhamnaceae) for instance, require tire for seedling establishment. Seeds are dispersed in a dormant state and accumulate in the soil until germination is triggered by fire, either from heat or a chemical leaching from charred wood (Keeley 1987). These species have specialized their reproductive biology to the extent that they are dependent upon fire for completion of their life cycle and may be referred to as "fire-recruiters". At the community level, fire-dependence is implied by the frequent suggestion that stands require fire for rejuvenation. Chaparral unburned for 60 years or more is often referred to as decadent, senescent, senile and trashy (Hanes 1977).

This fire-dependent paradigm of chaparral has guided fire management strategy in southern California, although it is perhaps generous to call the modem fire regime "a strategy," since most acreage in southern California bums by catastrophic wildfires. Nonetheless, federal, state and county agencies have prescribe bum programs for chaparral sites under their fire jurisdiction. Some areas that escape wildfires are burned under prescription at return intervals of approximately 15-25 years. Such a prescription follows logically from the commonly accepted dogma about the tire-dependence of chaparral. This, however, is not the whole story.

FIRE RESILIENCE VS. FIRE DEPENDENCE

While it is true that the chaparral community is highly resilient to fire, all species within the community are not fire-dependent. In fact, a large component of chaparral, while persisting in the face of recut-rent fire, may actually decline after repeated tires. Included here are species such as Ouercus dumosa (Fagaceae), Heteromeles arbutifolia Rosaceae), Prunus ilicifolia (Rosaceae), Cercocarpus betuloides (Rosaceae) and Rhamnus spp. (Rhamnaceae). These shrubs are resilient to tire by virtue of the fact that they are vigorous resprouters, yet they do not establish seedlings after fire. These species are "fire-persisters" but not "fire-recruiters." A management plan oriented towards long-term stability and maintenance of biodiversity needs to consider the conditions necessary for reproduction of these taxa.

The conditions under which these species recruit seedlings have not been well worked out. It is clear that these species do not establish seedlings after fire, and there are aspects of their seed germination physiology which account for this (Keeley 1987). On the other hand, studies of mature chaparral have consistently pointed out the lack of seedling reproduction under the closed canopy of this dense shrub vegetation (Sampson 1944; Horton and Kraebel 1955; Hanes 1971; Christensen and Muller 1975).

One clue to this mystery is an observation made in an early paper by Patric and Wanes (1964). These authors studied a stand of chaparral unburned for more than 60 years and noted seedlings of Quercus dumosa, Prunus ilicifolia, and Rhamnus crocea. Spurred in part by these early findings I have been investigating the fate of chaparral in the long absence of fire.

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My focus has been on the demographic structure of stands free of tire for 100 years or more in some cases. This study has revealed large seedling populations in older stands of chaparral; from 1,000 to 40,000 seedlings per hectare for taxa such as Quercus, Rhamnus, Prunus, Cercocamus and Heteromeles (Keclcy unpublished data). It is apparent that long fire-free conditions are required for seedling establishment by these tire-persister shrub species.

In summary, chaparral is dominated by shrubs that are resilient to tire. Some are fire-dependent taxa that recruit seedlings only in the first season after fire, and these are called fire-recruiters. Other shrubs, however, are not fire-dependent. They persist after fire but these fire-persisters require long fire-free conditions for seedling establishment (figure 1).

What is the best strategy for management of these systems. It appears that fire intervals on the order of every 20 years would potentially benefit fire-recruiters. Fire-persisters, while not obviously damaged .by this fue return interval, over long periods of time will be threatened by the lack of opportunities for seedling establishment. I suggest the coexistence of these modes reflecs the natural stochastic fire regime. Under natural conditions, the eventuality of tire on any given site would have been nearly certain, however, the return interval over time would have been variable. Short return intervals would have provided opportunities for population expansion of fire-recruiters and long return intervals would have provided opportunities for population expansion of tire-persisters.

RESILIENCE TO LONG FIRE-FREE INTERVALS

Community stability is dependent on both fire-recruiters and tire-persisters being resilient to both short and long fire return intervals. The current fire regime of relatively short intervals of 20 years between fires does not pose an immediate threat to either group. I suggest that all chaparral shrubs are also resilient to long tire-free periods, although few chaparral sites remain unburned for more than a few decades.

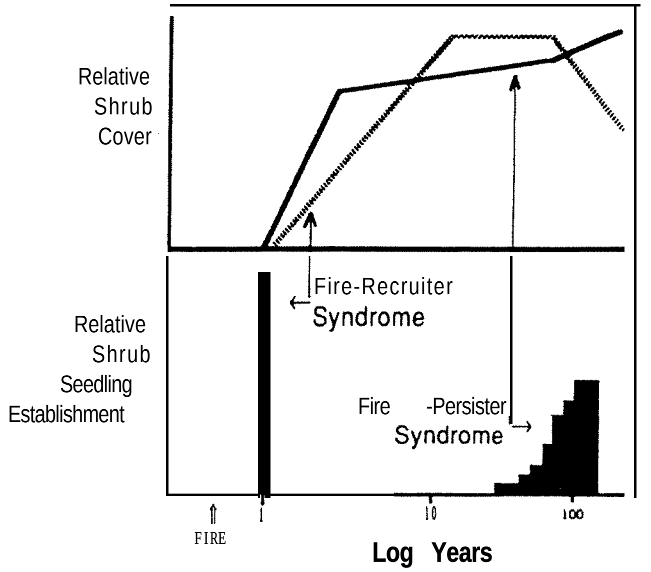


Figure 1.- Schematic illustration of the timing of seedling recruitment for chaparral shrubs described as fire-recruiters and as fire-persisters and longterm changes in relative shrub cover for tire-recruiters (dashed line) and tire persisters (solid line).

This notion would seem to be contrary to much of the dogma about the decadence, senescence and senility of chaparral stands older than 60 years. These terms, however, are seldom defined; a former student once suggested that a senile chaparral shrub was one which forgot to close its stomates, and this definition is about as good as any proposed in the literature. Most certainly these terms derive from observations that, due to natural thinning of shrubs (e.g., Schlesinger and Gill 1978), dead wood accumulates. However, something that is seldom appreciated is that dead stems are continually replaced by basal sprouting in all sprouting shrubs (figure 2). Consequently, the age structure of sprouting shrub populations are not even-aged and exhibit continuous recruitment and turnover of stems (figure 3). In other words, resprouting, in addition to functioning to rejuvenate shrubs after fire, functions to rejuvenate the canopy throughout the life of the stand.

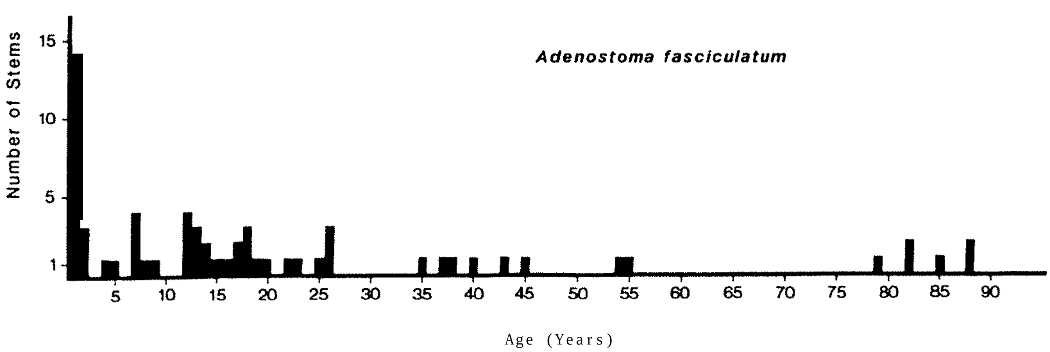


Figure 2.- Number of stems of different ages on a single resprouting shrub of Adenostoma fasciculatum in a stand of chaparral last burned 89 years ago (Keeley unpublished data).

How then did old stands of chaparral come to bc described as senescent and unproductive? This idea is apparently derived from early studies which investigated browse production by different aged stands of chaparral (Biswell and others 1952; Hiehle 1964; Gibbens and Schultz 1963). These studies concluded that chaparral became very unproductive within several decades following tire. However, these studies were only concerned with production available of wildlife. Consequently they did not present valid measures of productivity, because production above 1.5 meters, which is unavailable for deer, was not included. Since most new growth in older stands occurs above 2 meters, it is not surprising that one would conclude that frequent fires were a necessity for maintaining productive chaparral communities. Since the concept of stand senescent seemed logically consistent with the tire-dependent nature of many chaparral species, this myth of low productivity in older stands of chaparral was not questioned by many chaparral ecologists and foresters. Modem studies, however, reveal that live biomass increases with age in chaparral (figure 4), and the terms decadence, senescence, and senility, while possibly true of some species, should not be used to describe chaparral communities.

In conclusion, chaparral is resilient to short and long tire-free intervals, and different tire-return intervals, favor different components of the vegetation. Longterm stability and biodiversity of chaparral communities may require a mosaic of fire regimes.

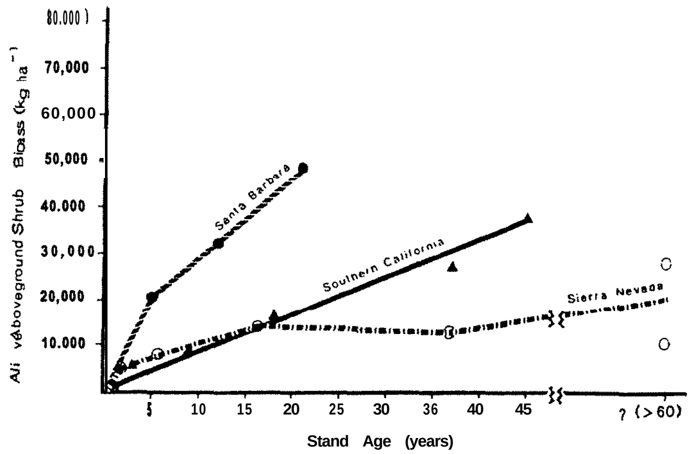
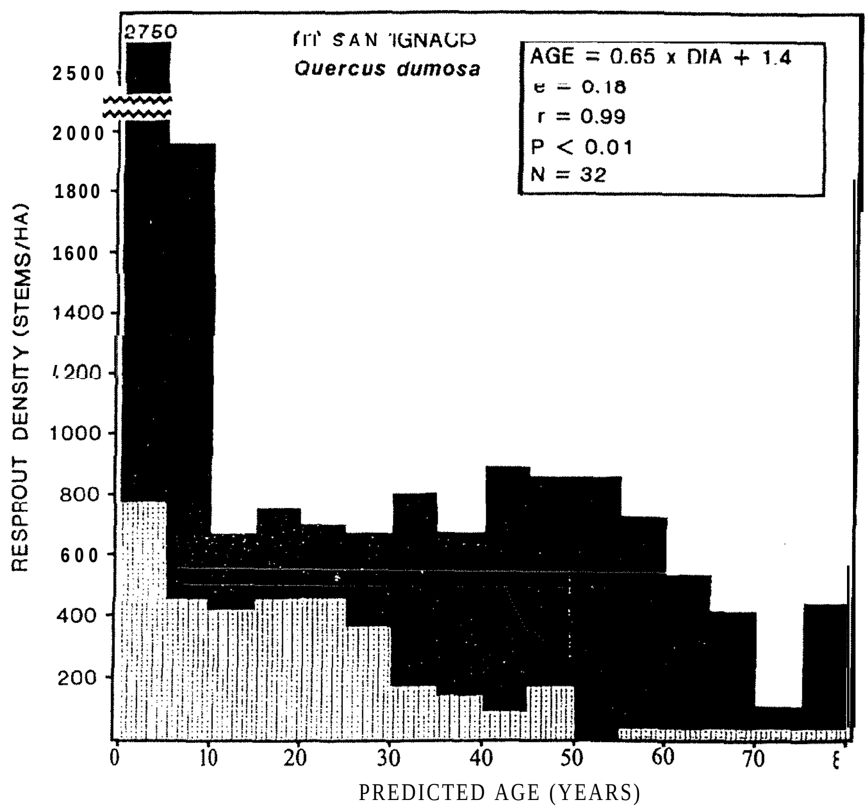


Figure 4.- Standing living biomass in chaparral stands as a function of age since last fire (from Keeley and Keeley 1988, with permission of Cambridge University Press, data from studies by Specht 1969, Conrad and DeBano 1974, Schlesinger & Gill 1980, Rundel and Parsons 1979, Stohlgren and others 1984, as cited in Keeley and Kcelcy 1988.)

Figure 3.- predicted population age structure of Quercus dumosa stems sprouted from root crowns of mature shrubs in a stand of chaparral last burned 76 years ago (solid bars are living stems, vertical lines arc dead stems). Stem diameters wcrc measured in 36 4x4 m plots randomly placed in the stand. Age was predicted from the indicated regression line based on 32 stems aged by ring counts. In addition to the correlation coeflicicnt, the estimate of relative error was calculated as the standard error divided by the mean value of y.



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FIRE AND OAK REGENERATION IN THE SOUTHERN APPALACHIANS

David H. Van Lear[1]

Abstract-Oak forests throughout the southern Appalachians have been historically maintained in a regime of frequent tire. Frequent fire over an indefinite time period favors oak establishment by reducing understory and midstory competition from tire-intolerant species and by creating preferred conditions for acorn caching by squirrels and bluejays. Fire also reduces populations of insects which prey on acorns and young oak seedlings. Once established in the understory, oaks have a tenacious ability to resprout when tops have been killed repeatedly by tire. The ability to continually resprout when numbers of other sprouting hardwoods have been reduced by tire allows oak to accumulate in the advance regeneration pool and dominate the next stand when suitable conditions prevail. Intense fires in logging debris also favor establishment and development of high quality oak-dominated stands. Tentative guidelines for the silvicultural use of fire to regenerate oak are discussed.

INTRODUCTION

The abundance of oak in the southern Appalachians and throughout eastern North America is related to past land use and extensive disturbance (Crowe 1988). Most of the river basins throughout the southern mountains were cut over and subsequently burned repeatedly around the turn of the century (Secretary of Agriculture's Report to Congress 1902). This type of disturbance regime evidently favored oak regeneration because oaks presently dominate mature mixed hardwood stands on mesic to xeric sites throughout the region.

Today oaks arc often replaced by other species when mature stands are harvested, especially on better quality sites (Sander and others 1983). Failure to consistently regenerate oaks following harvest is widely recognized as a major problem in hardwood silviculture. Even though researchers generally agree that fire played a role in the establishment of oak-dominated stands at the turn of the century (Sander and others 1983; Crow 1988), no silvicultural guidelines exist for using fire to regenerate oak (Rouse 1986). The purpose of this paper will be to 1) describe the ecology of oak regeneration in regard to fire, and 2) present tentative guidelines for the silvicultural use of fire to regenerate oak.

It must be emphasized that these arc tentative guidelines and must be tested prior to implementation as management recommendations.

THE ECOLOGY OF OAK REGENERATION

Large seed crops are produced by oaks at 2- to lo-year intervals, although there is great variation among species (Sander and others 1983). In the southern Appalachians, acorn yields of greater than 1000 pounds per acre (fresh weight) occasionally occur which allow oak seedlings to become established. Deer and turkeys are major consumers of acorns, although Sciurids, especially chipmunks and flying squirrels, may consume more than half of the oak mast available to wildlife in the southern Appalachians (Johnson and others 1989).

In addition to wildlife predation of acorns, insects also consume large quantities of acorns. Annually about 50 percent of the acorn crop in Ohio is destroyed by the larvae of Curculio weevils, acorn moths, and gall wasps. Other insects attack germinating acorns and oak seedlings. However, recent studies indicate that prescribed burning may reduce populations of oak insect pests (Galford and others 1988). A reduction in insect predation would allow more acorns to be scattered and buried by jays and squirrels, thus enhancing the probability of successful germination, and also encourage subsequent seedling establishment.

Areas of thin litter are preferred by squirrels and blue jays for acorn burial, suggesting that recently burned areas provide conditions conducive to oak establishment (Galford and others 1988). An interesting and important ecological finding is that jays collect and disperse only sound nuts (Dartey-Hill and Johnson 1981), which implies that if these acorns escape predation they will result in well-established first-year seedlings. Seedlings from freshly germinated acorns are unable to emerge through a heavy litter cover (Sander and others 1983). Germination and first-year survival are best when acorns are buried about i-inch deep in the mineral soil (Sander and others 1983).

Species in the oak-pine complex adapted during their evolutionary history to regimes of occasional and frequent fire by developing survival mechanisms which enabled them to withstand intense heat or to regenerate successfully following tire. Martin (1989) suggests that bark thickness may be the single attribute that best characterizes a species's adaptation to fire. While bark thickness is undoubtedly of great importance to the survival of mature trees in regimes of frequent fire, it

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is the ability to continually resprout following top-kill that enables most hardwood species, and especially oak, to regenerate under such conditions.

Although all hardwood species sprout in a regime of annual winter fire, sprouts remain relatively small and inconspicuous because of repeated top-kill by tire (Thor and Nichols 1974; Langdon 198 1; Waldrop and others 1987). Annual summer tires eventually eliminate all hardwood sprouts. Biennial summer tires also gradually eliminate hardwood sprouts, but oak succumbs more slowly than other species (fig. 1). Oaks, in the absence of prolific root sprouters such as sweetgum, would gradually dominate the advance regeneration pool because of the tenacity of their sprouting (Carve1 1 and Tryon 1961; Waldrop and others 1987).

At the turn of the century, summer tires were quite common as farmers burned the land to facilitate grazing. They had learned from early settlers, who in turn had learned from their Indian predecessors, that growing season fires best maintained an open forest with a rich herbaceous layer (Komarek 1974). However, not all areas would bum every year, so hardwood sprouts would have survived in areas where tire occurred at irregular intervals. It is reasonable to assume that, because of their tenacious sprouting ability, oaks would have dominated the advance regeneration pool.

Periodic winter and summer bums at intervals of about 4-7 years allow a vigorous hardwood understory to develop (Langdon 1981; Waldrop and others 1987). However, stems generally remain small enough (< 2 in) to be top-killed by the next fire. Hardwood sprouting is more vigorous following periodic winter bums because of greater carbohydrate reserves (Hodgkins 1958). Thor and Nichols (1974) noted that even with periodic and annual winter burning, oak stems tend to increase at the expense of competing hardwoods. After two periodic winter bums and eight annual winter bums, oak stems comprised 61 and 67 percent of the total stems, compared to 51 percent oak stems on the unburned plots. Swan (1970) has similarly shown that surface tires increase the proportion of oak in a stand even if no seedling establishment occurs, i.e., by persistent sprouting.

A regime of frequent burning over long periods of time would create an open stand, whether burning occurs in pine or hardwood stands. In hardwood stands, long-term burning would tend to eliminate small understory stems outright and would gradually reduce the mid- and overstory canopy through mortality resulting from tire wounds. Increased light reaching the forest floor in these open stands would maintain the vigor of oak advance regeneration. Loftis (1990) demonstrated that elimination of the subcanopy by herbicides

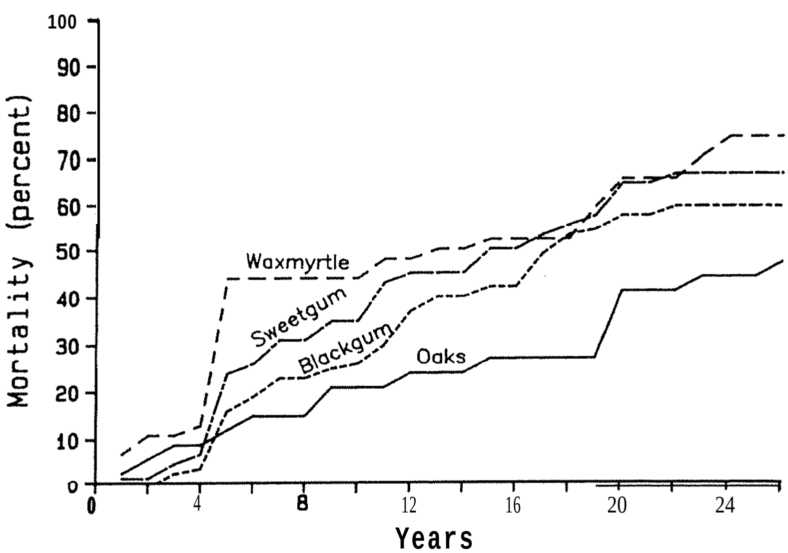

Figure 1. Cumulative mortality of hardwood roots over 26 years of biennial prescribed burning (From Langdon 198 1).

encouraged development of advance regeneration of red oak in mature mixed hardwood stands in the southern Appalachians. Long-term burning would have created stands similar to those created by injecting understory hardwoods with herbicides.

Studies of effects of single fires on composition of mixed stands have produced varied results. McGee (1979) found that single spring and fall bums in small sapling-sized mixed hardwood stands in northern Alabama had little effect on species composition other than to increase relative dominance of red maple. However, a single intense wildfire in a young mixed hardwood stand in West Virginia shifted species composition to a predominately oak stand (Carve11 and Maxey 1969).

Broadcast burning of logging slash in the mountains of South Carolina and Georgia increased the number of oak sprouts and, more importantly, the number of top-killed oak stems (up to 6-in ground diameter) with basal sprouts (Augspurger and others 1987). Severe fires xcrify forest sites by consuming much of the forest floor and perhaps even organic matter in the mineral soil, as well as by exposing the site to greater solar radiation through canopy reduction. Conversion of mesic sites to xeric conditions by intense fires or by a long regime of low intensity fires, along with their tenacious ability to resprout, could explain in large part the ability of oaks to dominate sites where more mesic species normally occur.

The absence of fire for long periods of time has allowed the composition and structure of the southern Appalachian forest to change to a condition where oak species can no longer dominate on better sites. Species that arc intolerant of fire have become established and grown to a size where they, because of thicker bark associated with age, can resist fire damage. Such species as mockemut and pignut hickories, scarlet oak, red maple, and blackgum arc examples of such species that arc oftcn found on sites where fire has been long absent (Harmon 1984; Martin 1989). Suppression of fire has allowed mesic species, both trees and shrubs, to occupy drier sites where fire was once more frequent and oak more dominant. In particular, yellow poplar stands now often reach ridge tops and rhododendron has dramatically increased its area1 extent (Van Lear and Waldrop 1989; Martin 1989). Impenetrable thickets of ericaceous species such as mountain laurel and rhododendron now often dominate midstorics and understories of hardwood stands in the Southern Appalachians and prevent desirable hardwood regeneration from becoming established (Beck 1988).

SILVICULTURAL USE OF FIRE IN OAK REGENERATION

There is no dispute among silviculturists that oak advance regeneration is neccssary before a new oak-dominated stand can be regenerated (Clark and Watt 1971; Sander and others 1983; Loftis 1988; Lorimer 1989). However, while many acknowledge the role that fire may have played in creating the present mature oak stands, no silvicultural guidelines have been developed for using fire to regenerate oak stands.

Based on the history of fire in the southern Appalachians and on documented ecological responses of oaks and associated species to fire discussed earlier, the following scenarios are presented as tentative guidelines for using fire in oak management. Further research will be necessary to test and line tune these suggestions before they can be recommended as silvicultural practices.

To Promote Advance Regeneration

Little (1974) suggested, as did Van Lear and Waldrop (1989), that an extended period of repeated burns prior to harvest may be necessary to improve the status of oak in the advance regeneration pool, especially on better sites. The famous Santee Fire Plot study, although conducted in another physiographic region, showed that annual summer bums for 5 years in a pine stand in the Coastal Plain killed about 40 percent of oak root stocks compared to 55 to PO percent of other woody competitors (Waldrop and others 1987). Biennial summer burning killed hardwood root stocks more slowly but the rate, of mortality for other woody species was still significantly greater than that of oak species. Annual winter burning, while not as effective as summer burning in altering species composition, still tends to xerify the site by consuming litter and reducing shading of top-killed understory species.

Thus, a regime of frequent understory bums, including both summer and winter bums, during a period of 5 to 20 years prior to harvest should promote oak seedling establishment and allow oak seedling-sprouts to dominate the advance regeneration pool (fig. 2). A relatively open stand with few midstory and understory trees would provide sufficient light for the oak advance regeneration to develop into stems of sufficient size to outgrow other species after the overstory is removed. Without frequent fire, all advance regeneration species would respond to the favorable light conditions in an open stand.

Preharvest burning reduces the forest floor, thereby encouraging burial of acorns by squirrels and bluejays. Burning theoretically reduces insect predation of acorns and young oak seedlings. The proposed burning regime should be a mix of summer and winter fires adjusted to maintain the vigor of the oak advance regeneration. There is no research

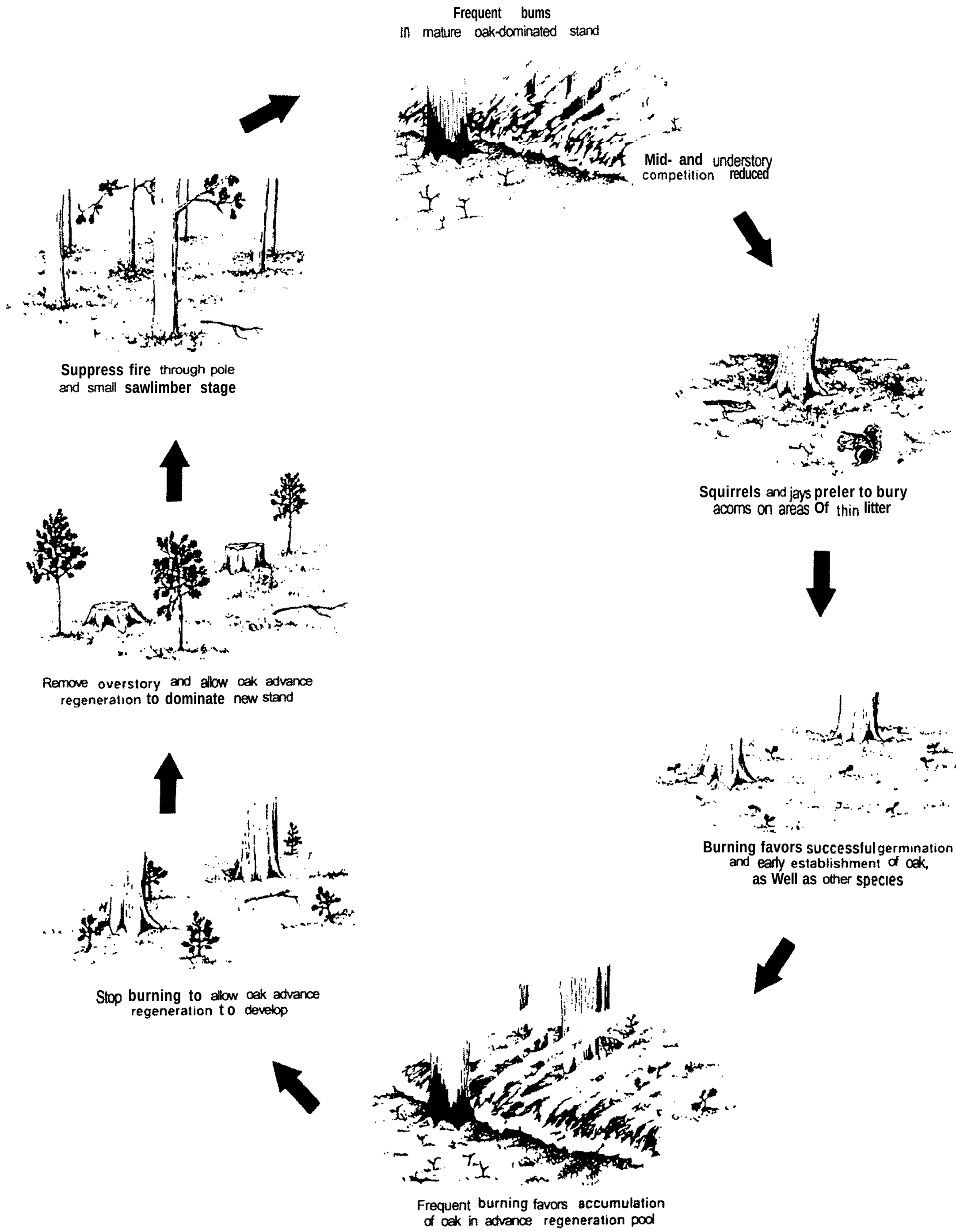

Figure 2. Tentative scenario of using prescribed fire to encourage advance regeneration of oak.

that currently documents what this mix of bums should be. Once an adequate number of oak seedling-sprouts are present and numbers of competing species have been sufficiently reduced, fire should be withheld to allow the oak advance regeneration to attain sufficient size to outgrow other species which germinate or sprout after the mature stand is cut. Sander and others (1983) recommends that 435 advance regeneration oak stems per acre over 4.5 ft tall be present before the ovcrstory is removed.

Fire has been suppressed for so long in the southern Appalachians that it may be necessary to use herbicides to remove midstory trees that have grown too large to be killed by low-intensity fires. Loftis (1988, 1990) has convincingly shown that growth of advance regeneration of northern red oak can be enhanced by herbicidal removal of mid- and understory competitors. A combination of herbicide treatment and frequent fire may be required to secure oak regeneration and allow it to maintain its vigor in mixed hardwood forests which have not been burned for decades. Frequent understory burning will be necessary because singe bums benefit oak regeneration only slightly (Tcuke and Van Lear 1982).

Foresters have long recognized that wildfire during the growing season is a major cause of butt rot in hardwoods, but relatively little information is available concerning the relationship between prescribed fires of lower intensity and stem damage. Wendel and Smith (1986) found that a strip-head fire in the spring in an oak-hickory stand in West Virginia caused a decline in overstory vigor and resulted in death of many trees during the 5 years after burning. However, a low-intensity winter fire *in* a mixed hardwood stand in the southern Appalachians resulted in little or no cambium damage to large crop trces (Sanders and others 1987). Smaller trees did suffer stem damage, but in even-aged management these trees would be used for products not requiring stems of high quality. If not removed, these damaged trees would eventually succumb to disease and be lost from the stand. More research is needed to determine if and when low-intensity fires can be used without excessive damage to stems of large valuable crop trees in mature hardwood stands.

To Increase Quality and Numbers of Oak Stems after Clearcutting

The fell-and-bum technique for regenerating mixed pine-hardwood stands has been used successfully in the southern Appalachians and is fully described in Waldrop and others (1989). Basically, the technique involves felling residual trees left after commercial clearcutting when their crowns arc almost fully leafed out. After curing for l-3 months, the logging debris is broadcast burned with a high intensity tire conducted under conditions that produce little or no soil damage. Planting pine seedlings at low densities among the hardwood coppice produces a mixed pine-hardwood stand.

Broadcast burning following clearcutting of hardwood or mixed pine-hardwood stands promotes better quality oak sprouts by forcing them to develop from the groundline. Over 97 percent of all oak sprouts developing after broadcast burning of logging slash in the southern Appalachians were basal sprouts, versus 71 percent for unburned areas (Augspurger and others 1987). Suppressed buds higher on the stump are apparently destroyed by the intense heat of the fire. Sprouts from the base of the stump will not develop rot as readily as those from higher on the stump and can be grown on longer rotations for more valuable products.

Broadcast burning increases the number of oak sprouts, as well as the number of small oak stumps with at least one basal sprout. Small oak (< 6 in) stems in the understory of mature stands often arc poorly formed and,, unless killed back by fire or some other agent, will not develop into quality stems. However, when top-killed by the intense heat of broadcast bums, sprouts from these fire-killed stems are more likely to develop into sound timber trees than other types of oak regeneration (Roth and Hepting 1943).

Intense tires can sometimes result in the introduction of oak in the succeeding stand. Nowacki (1988) documented cases in northern Wisconsin where clearcutting of old-growth maple-hemlock stands and slash burning resulted in even-aged stands dominated by northern red oak. Lorimer (1989) suggested that these oak stands probably developed from acorns brought into the burned area by birds and animals. The author has made similar observations following an intense wildfire in the mountains of South Carolina.

SUMMARY AND CONCLUSIONS

There is no doubt that oaks in the southern Appalachians are being replaced by other species on better sites where oaks were once dominant. Oaks are definitely favored by some type of disturbance regime. Based on the history of this region and literature concerning responses of oak to fire, it appears that oak replacement is largely the result of a different fire regime from that which existed in the region in previous millennia. In the past, frequent fires allowed oak regeneration to accumulate and develop in the understory of open mature stands at the expense of shade-tolerant, fire-intolerant species. When the overstory of these stands was either completely or partially removed by various agents (wind, insects, wildfire, Indian clearing, harvesting, etc.), conditions were created which allowed advance-regeneration dominated by oak to develop into mature stands.

If oaks are to bc maintained as a dominant overstory species on medium to good quality sites in the southern Appalachians, it seems that foresters will have to either restore tire to some semblance of its historical role as a major environmental factor or develop artificial methods that simulate the effects of tire. If research does not soon discover the secrets of maintaining oaks on these sites, foresters through their tire suppression efforts will have encouraged the demise of oak on these sites, much to the detriment of numerous ecosystem values.

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RESPONSE TYPES TO PRESCRIBED FIRE IN OAK FOREST UNDERSTORY

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Abstract-We examined data collected on the understory of a prescribe-burned upland oak forest at the University of Tennessee Highland Rim Forestry Station from 1965 through 1989. Treatments were annual and periodic (once in live years) burns and no bum. Each was replicated three times.

Species number declined dramatically under the no-bum regime. Tree seedling establishment was inhibited and sprout size decreased in the annual and periodic burns-here the understory aspect is quite open. Tree sprout cover in the periodic bums followed the incidence of fire, it has lowest in the September following each winter burn. Graminoid cover, chiefly little bluestem Schizachyrium_scoparium_, decreased to zero in the no-bum treatment. It was variable in other treatments but decreased irregularly from 1973-I 975 to 1988. In 1989 it established a new high. Composite, legume and other forb cover declined to zero under the no-bum treatment. Composite cover in annual plots peaked in 19734 but decreased irregularly since; legume cover increased irregularly; other forb cover decreased irregularly in annual bum plots. Composite cover in periodic plots oscillated around fire years when cover was generally highest; legume cover peaked in four of the five fire years; other forb cover generally increased in tha year following a fire. Twenty-two response-types occurred.

INTRODUCTION

Studies of the effects of fire on natural or man-fostered systems in central and southeastern United States are chiefly those in the grassland where these communities were long maintained as grazing land (Risser and others 1981) and in southeastern pineland which were also maintained as grazing and timberland (and for naval stores in the past) (Wahlcnberg and others 1939). Summaries of the effects of fire on grasslands can be found in Risser and others (1981), Daubenmire (1968a), Wright and Bailey (1982), Vogl (1974), and Collins and Wallace (1990). Summaries of the effects of fire on conifer forests especially southeastern forests are in chapters in Kozlowski and Ahlgrcn (1974), USDA Forest Service (1971), Wright and Bailey (1982), and Wood (1981). Older literature is summarized in Garren (1943), and Ahlgrcn and Ahlgren (1960). Usc of tire in the central Deciduous Forest has been on the decline since abandonment of open range practices and intensification and specialization in land use (Vogl 1974, Chandler and others 1983). Fire use continues in hardwood and mixed forests as a wildlife management tool (Wood 1981), and to modify understory composition or size class structure (Wade and others 1989, Faulkner and others 1989). Studies on hardwood understory include Paulsell (1957) and DeSelm and others (1973, in press).

This paper is concerned with the consequences of 25 years of annual and periodic prescribed fires on the undcrstory species in oak-dominated vegetation at Highland Rim Forest Expcrimcnt Station, Franklin County, Tennessee. The study contributes to an understanding of the maintenance of graminoid-forb undcrstory and openings in the upland oak forests of this region and adds to our knowledge of response behavior of these species to fire.

For the period 1965-1970 the understory of the bum and control plots was examined (DeSelm and others 1973). In the six treatment years number of species sampled were 13, 23 and 35 taxa in control, periodic and annual bum plots, respectively. With increased burning, tree frequency and tree sprout cover decreased, grass and forb cover increased, and herbaceous vine and shrub cover became elevated on the periodic burns. Eighty-four percent of the species responded positively (by increased cover) to fire treatment by 1970. Tree, litter and soil changes were examined by Nichols (1971) and Thor and Nichols (1973).

THE STUDY AREA

The study area is at 36° 30'N : 86°W at the eastern edge of the Interior Low Plateau Province (the southeastern Highland Rim) in Middle Tennessee. The land surface here undulates and it has loess-derived soils in which a water-movement-inhibiting pan has developed in several series (Fenneman 1938, Fox and others 1958, Love and others 1959). Forest vegetation is of the upland oak swamp, post oak-blackjack, and southern red oak-scarlet oak types (DeSelm and others 1973). Conversion of this vegetation to agricultural land and to loblolly pine plantations is still occurring (Thor and Huffman 1969, USDA Soil Conservation Service 1971, Buckner and others 1986).

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Early surveyors report a few places with no forest, and such modem sites as the May Prairie have a physiognomic and floristic resemblance to midwestem prairie (DeSelm and others 1973). The origin and maintenance of grass vegetation and grass understory has in part stimulated this study.

American Indian use was followed by agriculture and livestock grazing between the late 1700s and early 1940s. Army maneuvers during World War II have been part of this site's history. The local farmers commonly burned off the woods in winter and a railroad line on the edge of the station also storied fires annually (Faulkner 1968, Haywood 1823, DeSelm and others 1973).

METHODS

Data were collected on nine I.8 acre experimental bum plots, 19651989 inclusive. Plots were split among annual, periodic (usually 5 years) and control. A 50-foot tape was stretched three feet above the ground between permanent stakes in each plot. At each one-foot mark a 0.19 inch diameter metal pin was lowered vertically to the ground and each "hit" on each vascular plant was recorded; those plants below three feet in height are herein called understory. Annual burns began in 1963; periodic bums were made in 1964, 1969, 1974, 1979, 1983 and 1988. Late winter bums *were* used to simulate the action of local land owners. Plant nomenclature follows Fcmald (1950).

Data reported here are cover (sum of hits per species x 2) along each 50-foot line. Cover values are averages of the three replications. No frequency or relativized data are reported. Years have been segregated into early 1965-1972, middle 1973-1981, and late 1982-1989 groups. Some taxa occur widely (the wides) across the series of years with somewhat variable-to-steady cover. In other taxa, the cover increases (the increasers), or the cover decreases (the decreasers), and in a few the cover increases (bulges) or decreases (sags) in the middle years. Taxa not present in the 'early years that appear later are termed "invaders." Taxa present in early or early and middle years but which are not present in late years are termed "retreaters." Sporadics, which occur in early, middle, and late years, totaling 22 taxa, were not considered. Some terminology is from Vogl (1974).

RESULTS AND DISCUSSION

General

Taxa seen along the strips over the years totaled 141 spccics: 13 tree, 15 shrub, 2 woody vine, 2 1 graminoid, 22 legumes, 40 composite and 22 other forb taxa (counts excluded unknowns of various categories). This represented 52 percent of the 270 known vascular flora of the Station. Of these, 119 are included in this study. Two taxa occurred only in control plots: Gcntiana villosa and Lisuidambar stvracillua (but in 0.01 acre plots, Liquidambar has been found in the periodic bum). Thus, all but one taxa was at least mildly fire tolerant.

Although several State rare taxa occurred on the Forest (DeSelm 1990) only one occurred on the bum plots; Gymnopogon brevifolius is listed as a species of special concern (Somers and others 1989).

Forty-five taxa occurred in all three treatments, 39 taxa occurred in both bum treatments, 31 occurred in the annual bum only, and 12 occurred in the periodic bum only.

Total cover (sum of woody plant-graminoid-forb) in annual plots peaked in 1973--but by 1987 it decreased 43 percent after which it rose again. In the periodic plots total cover fell 46 percent between 1980 and 1987 after which it rose sharply. These 43 and 46 percent decreases in cover represented temporary increases in bare ground. Total cover in check plots decreased gradually until 1976 (the last year of herb cover), remained more or less steady through 1988, then increased in 1989 (Table 1).

Woody Plants

Cover by woody plants was irregular but more or less constant in annual plots over the year series. In periodic plots, woody plant cover increased irregularly. In check plots woody plant cover decreased through 1988; in 1989 it increased to the level approximating 1975 (Table 1).

Trees

Sum of cover of tree taxa on control plots apparently decreased until about 1979 after which it became variable. Some tree stems grew upward beyond the sampling line and were no longer recorded. Some young trees died under the developing canopy but others have spread onto the sampling line. Nyssa sylvatica, Qucrcus coccinea, Q. falcata, Q. velutina and Vaccinium arborcum were recorded most to all years (Nyssa and Q. coccinea were decrcasers). Qucrcus stellata. Q. marilandica and Carya tomentosa occurred in early or middle years. Liquidambar and Cornus florida occurred in late or middle and late years. Acer rubrum appeared in 1969 and 1989, this apparently represent& disappearance by height growth followed by recent sprouting A summary of response types appears in Tables 2 and 3.

In annual bum plot strips, six taxa occurred widely (Carya, Nyssa, Q. coccinea, Q. stellata, Q. velutina and Sassafras albidum). coccinea and Q. velutina were decreasers, Q. stellata and Sassafras were increasers. Cornus florida, Q. falcata and Vaccinium arboreum occurred in early and/or middle years. Qucrcus lyrata was recorded in five middle and late years through 1983. The last several species, except perhaps Q. lyrata, were fire sensitive.

In periodic plots sum of cover of tree species and cover of several individual tree species was lower in fire years than between tires--they were burned back by the tires. Taxa with decreased cover on two to five fires were Carya, Nyssa, Q. coccinea, Q. falcata. Q. marilandica. Q. stellata. Q. velutina, and Vaccinium Sassafras (present six years) and Q. lyrata (present two years) barely survived these fires. Acer rubrum was seen only recently (1982- 1988). velutina and Vaccinium were increasers, Q. coccinea was a dccreaser, and Q. falcata and Q. stellata increased in middle years (bulge species).

Shrubs

In the control plot strips, shrubs persisted various numbers of years under the developing canopy: Rhus glabra to 1965, Rubus (erect) to 1972, Ceanothus americana and Rhus copallina to 1974, Rubus (dcwbcrry) to 1979, and Salix tristis to 1980. Vaccinium vacillans occurred each year, V. staminium appeared 1989. V. vacillans occurred in control oak-pine plots in southern New Jersey where its cover was reduced by 1950s drought (Stephenson 1965).

Certain shrubs in the annual burn plot strips occurred widely across the years (mostly many years): Ceanothus, Rhododendron nudiflorum, Rhus copallina, R. toxicodendron, Rubus (dewberry and erect), Salix tristis and Vaccinium vacillans. Increase in Rubus cover has been seen in central Wisconsin with fire **(Reich** and others 1990). Vaccinium vacillans and Rhus copallina increased in cover. In southern New Jersey V. vacillans's cover increased with burning frequency (Buell and Cantlon 1953). Rhododendron was a sag species. Ascyrum stragalum, Pyrus melanocarpa, Rhus glabra and Salix humilis persisted only one to three years (1965- 1967). **Ascvrum** hvpericoides and Vaccinium stamineum have been recorded since 1985 and 1988, respectively.

Shrub cover on periodic plot strips included the widely occurring Ceanothus. Rhus copallina, Rubus (dewberry and erect), Salix and Vaccinium vacillans. g s e , Ceanothus decreased in cover while Rubus erect, Salix tristis and Vaccinium vacillans increased in cover. Ascyrum hvpericoides and A. stragalum appeared in plot strips in 1983 and 1988 respectively. Rhus copallina cover peaked in fire years and dropped 50 percent or more in each of the following three to four years. Rosa Carolina and Rhus toxicodendron disappeared after 1967 and 1972, rcspectivcly. Rhus glabra behaved like Acer rubrum in the control plots, it occurred in both early and recent years, Rhus in 1965, 1983 and 1984.

Woody Vines

Woody vines were mainly Smilax glauca--only a few hits were made on Vitis aestivalis. In control plots, Vitis occurred 1965-1968 only. In these plots Smilax was a decreaser, early year covers averaged 5.8, later year covers averaged 1.5 percent. Canopy closure and deer browse affected coverage.

In the annual bum plot strips, Smilax cover decreased slightly. In the periodic bum plot strips cover varied from 2.0 to 20.6 per year. Compared to the previous year, cover increased one tire year, remained constant one tire year and decreased three fire years. In the year following fire, compared to the fire year, cover increased after one fire, and decreased after four fires.

Graminoids

Total graminoid cover in annual plots peaked in 1973, 1978 and 1989. In periodic plots it peaked in 1975, 1980 and 1989 (the year following a fire in each case). Graminoid cover decreased steadily in control plots (Table 1).

Little Bluestem--Schizachyrium scoparium

Control plot strips showed the disappearance of blucstem by 1977. Shade and tree litter are believed to be causes.

Cover of bluestem on annual bum plot strips increased irregularly to 1973, fell, peaked again in 1978, fell irregularly until 1988 and peaked again in 1989. Biomass of Schizachvrium also experienced multiple high and low value years in the 61 year record in Kansas (Gene Towne, personal communication). Although Andropogon gcrardii cover increased with annual burning in Missouri, cover of Schizachyrium increased with alternate year fires (Kucera and Koclling 1964).

Periodic plot strip cover increased irregularly to 1975, dccreascd, increased again to 1980, decreased to 1988, then increased again in 1989. These variations do not match burn years. On two fire years, 1969 and 1988, cover decreased slightly from the year before, and it rose in 1974, 1979, 1983 from the year before. The positive effects of this treatment on the cover of this grass were certainly not dramatic. The high peaks of this grass in both treatments classed this taxon as a midphasc bulge species. The 1989 peak may be part of a new trend.

Table 1. Total cover of **graminoids, forbs,** and **woody plants** by **treatments** and year.

Year	Annual			Periodic			Check		
	Gram.	Forbs	Woody	Gram.	Forbs	Woody	Gram.	Forbs	Woody
1865	32.6	54.6	24.6	35.3	27.5	37.6	57.4	16.6	43.6
1966	46.0	59.2	23.2	44.7	18.1	25.4	57.6	15.6	46.6
1967	64.2	43.7	21.0	32.6	2.6	23.4	37.4	9.6	43.0
1968	63.9	60.3	34.6	36.6	20.6	27.5	34.0	12.6	42.2
196s	58.7	43.4	23.2	33.6	37.9	26.1	14.0	6.4	26.6
1870	56.4	54.8	26.2	34.0	33.4	29.2	6.0	6.0	50.2
1871	63.2	48.6	16.0	38.3	16.2	41.4	6.6	8.4	31.2
1972	73.6	56.9	22.1	45.6	23.7	47.6	3.4	4.0	30.0
1973	**99.6**	66.5	24.3	52.0	16.2	42.1	3.4	4.0	26.0
1974	77.8	64.6	28.1	63.2	41.4	43.3	1.4	1.4	26.4
1975	**69.6**	53.2	32.5	65.0	40.6	63.6	0.0	0.6	21.6
1976	67.7	55.6	37.0	66.6	31.8	66.6	0.6		15.6
1877	51.3	36.7	16.4	26.8	25.2	61.3			14.4
1978	64.4	42.8	20.6	51.6	20.3	61.0			I0.6
197s	70.3	45.2	23.3	60.7	70.6	43.8			7.4
1960	57.5	34.6	18.6	64.0	26.1	**69.2**			13.4
1981	63.1	43.6	24.3	62.4	28.2	66.1			13.6
1862	49.1	33.7	17.5	50.7	21.7	61.4			9.2
1863	51.7	40.2	27.1	64.7	37.0	62.0			12.4
1964	48.0	31.4	24.7	53.6	27.7	61.8			10.6
1985	45.0	39.5	27.0	44.6	20.1	53.6			14.6
1966	44.9	31.4	26.7	36.3	16.5	55.8			15.2
1867	57.0	26.6	19.3	30.6	15.6	52.2			8.6
1996	46.8	46.2	26.0	26.5	56.0	75.6			12.0
196s	94.7	42.6	24.0	164.3	39.9	**89.7**			20.6

Other Graminoids (Gramineae [Poaceae], Cyperaceae)

In control plots, other graminoids were represented by low cover of only seven taxa--no more than five present any one year. They persisted only through 197 1.

Twenty-four other graminoids occurred in annual burn plots. Taxa present almost every year arc Andropogon gerardii and Sorphastrum nutans with 6-10 percent cover per year. A few other graminoids occurred more or less widely: Andropogon gyrans, Carex sp., Eleocharis tenuis, and Panicum dichotomum and P. lanuginosum. above taxa, Andropogon gerardii and Sorghastrum were increasers, and the two Panicum species were decreasers. Aristida dichotoma disappeared after 1966, A. purpurea after 1972, Gymnopogon brevifolius after 1981, Panicum angustifolium after 1978, P. ravenellii after 1975, Scleria pauciflora after 1972, and Setaria geniculata after 1968. globularis occurred only in the middle years of the series. A few taxa appeared late in the series, Agrostis perennans in 1980, Digitaria ischaemum and Microstegium vimineum in 1989, Muhlenbergia tenuiflora in 1984, Panicum laxiflorum in 1977 and P. villosissimum in 1986. The Digitaria and Microstegium are widespread weeds.

Concurrent peaks and valleys of cover in the annual burns occurred in a few taxa some years, but the correspondence in peaks was not impressive and did not argue strongly for response to weather. Peaks and valleys were best expressed in the high cover species Andropogon gerardii and Sorghastrum. These taxa also experienced rises and falls in biomass values on Kansas prairie (Gene Towne, personal communication). The sum of all other graminoid cover peaked in 1972, 1973, 1979, 1986, 1987, and 1989. Sorghastrum contributed greatly to all of these peaks. Andropogon gerardii contributed in 1979, 1987, and 1989.

Other graminoids in periodic bum plot strips numbered 19 species A few taxa occurred 11 or more years, Andropoyon gerardii. Panicum commutatum. P. dichotomum. P. laxiflorum, P. microcarpon and Sorghastrum nutans. A. gerardii was a bulge species, Sorghastrum was an increaser.

Two taxa were seen in the early or early and middle part of the series, Agrostis hycmalis through 1976 and Muhlenbergia tcnuifolia in 1965. A few taxa were seen only later, Aristida curtisii only in 1974, Panicum laxiflorum since 1976, P. sphaerocarpon in 1979, P. villosissimum since 1981, and Rhvnchosoora globularis in 1980.

The effects of the periodic fire on the total cover of all taxa in fire years was variable, some years cover increased, some years it decreases. However, the year following a fire, an increase in cover was achieved, the increase in cover was 1.1 to 9.7 times the cover the year before. Taxa with increased cover were the "fire follower" class of Lemon (1949). The effect is temporary; the second year after a tire, other graminoid cover total decreased.

Forbs

In 1965 and 1966 annual plot total forb cover exceeded woody plant and graminoid cover but after 1966 it decreased to a level intermediate between them. This suggested an early-in-the-treatment (early successional) forb dominant stage as was seen early in some southeastern seres (Quarterman 1957, Oosting 1942). Annual bum forb cover peaked in 1973 but decreased irregularly to 1987--a 60 percent loss of cover. In periodic fire plots total forb cover peaked in 1969, 1974-5, 1979, 1983 and 1988 (each fire year). There was a decrease in total forb cover 1979-1987 of 79 percent of the 1979 value. Forb cover in control plots decreased irregularly through 1975 (Table I).

Composites (Compositae, Asteracee)

Sixteen composite taxa, including unknown categories, occurred in the control burn plots. Occurrences ranged from one to 10 taxa per year. All were eliminated by 1975.

Annual bum composites, expressed as total hits on all taxa, increased to a peak in 1973 and 1974 and decreased irregularly thereafter (but increased slightly in 1989). This decrease in cover was apparent to us and was a cause. of comment. Recent plot photographs showed few composites in most late years compared to earlier years. Numbers of taxa in early years averaged 19.3, in late years 10.0. Taxa showing the above trend with peaks in 1973 or 1974, sometimes with additional peaks, were: Aster dumosus, A. hemisphericus, A. patens, A. undulatus, Coreopsis tripteris and Solidago odora. All but A. hcmisphericus were bulge species.

Several taxa occurred only in the early years: Antennaria plantaginifolia through 1968, Hieracium gronovii through 1971, Sericocarpus linifolius in 1968. A few taxa persisted through the middle years: Helianthus aneustifolius through 1978, H. silphioides through 1977, Solidago bicolor through 1977, S. erecta through 1981, and S. speciosa through 1974. Two taxa only occurred in the middle years: Helianthus strumosus and Scnecio anonymus. Several taxa appeared only in late or middle and late years; these were Ambrosia artimisiifolia seen first in 1987, Erectites hieracifolia seen first in 1982, Eupatorium album seen first in 1975, E. aromaticum seen first in 1973, E. semiserratum seen in 1985, and Solidago canadcnsis seen first in 1973. The Ambrosia, Erectites, Eupatorium album, and S. canadensis were weedy taxa locally.

Helianthus hirsutus and Coreopsis major were dccreasers--in the late years these taxa were present live of 16 possible times. H. mollis appeared to be on a two- to four-year low to high cover cycle. The reasons for this was unknown but its negative response to insect attack and wet weather were noted.

Total hits on composites increased and decreased with bums and between bums in the periodic plot strips. Composite cover generally peaked in fire years and decreased thereafter (although this did not happen during the wet year of 1989 after the 1988 tire).

A few taxa occurred only early in the total year sequence: Aster patens, Antennaria plantaginifolia, Gnaphalium obtusifolium. Solidago nemoralis and Vemonia flaccidifolia. A few other taxa occurred in early and middle or middle years: Aster hemisphericus, Eupatorium sessilifolium, Hieracium gronovii, Senecio anonymus and Solidago speciosa. Some taxa occurred only in the middle and late or late years of the series: Aster simplex, Chrysopsis mariana, Erigeron canadensis, Solidago canadensis, and Solidago erecta. The Erigeron and S. canadcnsis were weeds locally. Chrvsopsis spp. invaded burned longleaf pine stands (Heywood and Bumette 1934).

Sixteen other taxa occurred sparingly to frequently across the year-series. Solidago odora was a decreaser. A few year-frequent taxa peaked during fire years. They were Eupatorium aromaticum (five fires), Solidago odora (four fires), Helianthus silphioidcs and Aster dumosus (two fires), and Helianthus hirsutus, and Eupatorium rotundifolium (one fire each). On the other hand, Coreopsis major cover decreased in fire years (means were 0.52 percent cover in fire years versus 1.7 percent cover during non-fire years). Composite seedling rosettes (unknown Aster, composite, Eupatorium, Helianthus and Solidago) increased in cover the year after the fire years (four of five fires).

Legumes (Leguminosae, Fabaceae, Mimoseae, Caesalpinaceae)
Thirteen legume taxa occur in the control plots at the rate of one to six taxa per year. All were eliminated by 1970.

Twenty one named species, one hybrid and three unknown legume taxa occurred in the annual bum plot strips. Taxa which occurred only in the early years were Amphicarpa bractcata. Dcsmodium virginianum and Psoralea usoralioides. Species which occurred in middle or early and middle years were Dcsmodium paniculatum, Lespedeza capitata and L. virginica. The hybrid L. intermedia x capitata occurred annually in the middle and late years.

Sixteen taxa occurred widely across the year-series; 10 taxa 13 or more years, six taxa occurred only 2- 12 years. Of these wide taxa, Desmodium marilandicum. Lesuedeza intermedia, L. repens, Stvlosanthes biflora and Tephrosia virginiana were increasers. These were part of a general trend of increased legume cover with time; the cover increased 40.8 percent from the early to late year groups. Similar increases in legume importance were rcportcd by Wahlenberg and others (1939). L. procumbens has a low-middle, and Schrankia microohvlla has a high middle year cover.

Sum-of-legume cover and certain species cover suggested cycles of 2-5 years intervals but dates of species peaks usually did not correspond. Response of legumes to periodic bums was various; a general response was that species drop out. The mean number of taxa in early years was 9.8; the mean number in the late years was 20 percent lower. Several legume taxa occurred in 12 or more years across the series. Included were three increasers Cassia nictitans Clitoria mariana and Tcphrosia virginiana, Lespedeza repens a decreaser, and L. intermcdia a bulge species. Taxa present in early or early and middle years that disappeared later were: Amphicarpum bracteatum in 1969, Desmodium ciliare 1977, D. marilandicum 1976, D. viridiflorum 1967, Calactia volubilis 1981, Lesnedeza hirta 1967, Psoralea usoralioides 1968. Apparently only one species invaded, Cassia fasciculata; this has been present since 1975. Two taxa occurred only in the middle years, Desmodium obtusum and D. paniculatum.

Among wide taxa, peaks usually occurred in the periodic bum years. The two highest peaks (1979 and 1988) arc amplified by high cover of Cassia nictitans. An increase in frequency of C. nictitans with burning has been rcportcd (Cushwa and others 1970). Four other taxa had high cover in three to four fire years compared to non-tire years: Clitoria mariana, Lespedeza repens, L. procumbens and Schrankia microphylla. The cover of these taxa decreased in the years after each fire. Lespedeza intermedia peaked in two fire years only. Cover of Lesnedeza vireinica and Tcphrosia increased on three of five fires the year after the fire.

In periodic tire years the cover of Stvlosanthes biflora, which averaged 1.6, fell to zero--it was absent in tire years. it was another species influenced negatively by fire.

Other Forhs
In the control plots, other forb cover averaged low and lasted only until 1975.

In annual plot strips only Pvcnanthemum tenuifolium and unknown forb were present widely across the years. Seven taxa occurred only one year; eight taxa occurred 2-10 years.

Taxa occurring in the early or early and middle years in annual plots were Aureolaria virginica. Galium circaezans, Gerardia tenuiflora, Lobelia inflata, L. puberula, Scutellaria inteerifolia, and Viola saeittata. Three taxa occurred only in middle of the sequence: Gerardia pectinata, Hypericum densiflorum, and Trichostema dichotoma. Late occurring taxa were Houstonia caerulea, Ipomea pandurata, Leehea minor, and Rhexia mariana.

Over the annual bum year-series, the number of taxa declined slightly; the mean number of taxa in early versus late years was 4.8 versus 2.8, respectively. Similarly, the sum of all hits was 13.6 versus 5.0; this cover comparison was heavily influenced by an early-in-the-series peak by unknown forb cover in 1971, and peaks by Pycnanthemum tenuifolium in 1965 and 1968.

Periodic other forb bum plot data, as in the annual strips, contained few species and those that appeared did so for only a few years. *One* species, Hypericum dcnsiflorum, disappeared--it was last seen in 1978. A few taxa occurred only in the middle of the year series: Acalvpha virginica, Convolvulus sepium, Diodia virginica, Gerardia pectinata, Houstonia caerulea, Lechea minor and Lobelia nuberula. e Diodia and Lobelia have been seen two years, the others one year. The only new taxon was Gerardia tenuifolia which appeared in 1983. Mean cover in tire years was 3.0; mean cover in non-tire years was 4.0; the difference suggested a depressing effect of fire on forbs. The year following the tire the mean cover was 5.8--the rise suggests a positive fertilization or release-from-competition effect. These effects can be seen in Potcntilla simplex in which percent cover in fire years was low; in non-tire years it was intcrmediatc; the year after a fire, cover was highest.

General Discussion
The methods used collected minimal annual data although they were favored by nearly exact position replication between years and places. The three-foot maximum height measurement over-emphasized disappearance of understory stems which grew taller than three feet. Early maturing species may not be seen, and late maturing species may have

been over-represented in this constant-date sampling. The wet year of 1989 increased cover of some species groups markedly. Although time of year collection of data has always been a bias in ecological studies, in this summer-autumn flora bias is believed minimal. Animal activity may influence data; some Smilax was browsed and one Rhododendron was lost by burrowing. A peduncle-biting insect inhibited fruit set in Helianthus mollis.

Results expressed as percentage cover simply measured the degree of success achieved by the species in that environment (Daubcnmire 1968b). A parallel expression is "number of years seen," which represents the species response to comparatively stable soil conditions but changing climatic conditions and changing conditions of interspecific competition. Variable responses (variable occurrence, increase, decrease, invade, retreat) are typical behaviors of populations under stress (Grime 1979).

The mechanisms of response to fire, for the woody plants, were related to top death and subsequent growth of sprouts from suppressed or adventitious buds (Barbour and others 1987). Most herbs were hemicryptophytes with terminal bud which, if not injured by fire, provides post-bum growth potential (Daubenmire 1968b). Annual herbs (therophytes) comprise I 1.2 percent of the flora of these plots--their occurrence is more or less equally divided between annual and periodic bum plots. They make up nearly one-fourth (23.5 percent) of invader occurrences in annual and periodic plots--a proportion twice that in the plot flora. They often invade burned grassland (Vogl 1974).

Species can be rated on their response to fire treatment. Only two taxa were exclusive to the control plots--this suggested that other taxa were at least tolerant of the stress of fires of this study and the long history of previous woods fires and accidental railroad fires. Of low tolerance were taxa which were early and early and middle retrcaters, decreaser-retrcaters, and the late and middle and late invaders. Widely occurring taxa make up the rest of the classes. Wide decreasers, wide sag species, and wide taxa with low cover in fire years were more tolerant than previous classes. The most tolerant were wides with peaks in fire years, certain wides with their own cycle of variable cover, those with middle year bulges, and wide increasers. Least tolerant taxa were called pyrophobes; most tolerant taxa here were called pyrophiles. But these terms are absolutes and express only the extremes of a group of classes of fire tolerance suggested above. In fact, these classes may be part of a gradient of responses to a large number of fire intensity/frequencies.

Burns

In both bum treatments, the canopy was open, and overstory trees were few. The 0.5-5.0 inch DBH class was essentially missing (DeSelm and others in press). In late years annual bum plots were grass-forb-woody plant dominated. In late years, periodic plots were woody plant-grass-forb dominated. Due to a flush of Cassia nictitans after the fire of 1988, the order was woody plants-forb-grass. In the wet year of 1989 the order was grass-woody plant-forb. In 1989 periodic plots had 44 percent more cover than annual bum plots. Total species number on periodic plots always increased in fire years and after four fires, and decreased thereafter (see also Collins and Gibson 1990).

Individual species response to the two types of bum treatments were seldom identical; only 20 taxa (14 percent) had the same response to annual and periodic burning. The variety of responses suggested that the sprouting habit of trees and shrubs and the hemicryptophytic life form of most herbs here was not strictly fire adaptations, but were a fire response to adaptive mechanisms evolved under a complex of disturbances including fire, grazing, browsing, and/or drought.

The net response of legumes to fire was positive; there was an increase in cover in both annual and periodic treatments. The species number remained constant in annual burns (although it decreases slightly in periodic bums). A few legume taxa peaked dramatically in periodic fire years and two taxa peaked the year after a periodic fire. The legumes, plus Ceanothus (Bond, 1983), widely occurring in annual and periodic plots, and free-living N-fixers, replace at least some nitrogen volatilized in fires (Chandler and others 1983). Nitrogen losses arc reported in soil in burned grassland (Collins and Wallace 1990).

After 27 years of treatment, eight taxa, all woody, occurred on the strips in control bum plots. This is similar to the 10 woody taxa after 20 years under loblolly pine in coastal South Carolina (Lewis and Harshberger 1976). Eleven herb taxa were present in the pine stand but there were none under oaks here. A more open canopy or physical factors associated with the litter or oak roots (McPherson and Thompson 1972) or allclopathic substances (Rice 1984) may be responsible for the contrasting numbers of taxon under these overstories. Slightly more woody taxa occurred in the oak plots here with fire (13 annual, 16 periodic taxa) than under pine (10 annual, 12 periodic taxa). Under oak 39 (annual) and 34 (periodic) herb taxa occurred; under pine 26 (annual) and 18 (periodic) herbs occurred. Although the species number is lower under pine, the percentage distribution of grass, composite and legume taxa is similar. The lower numbers of taxa in bum plots under pine versus oak may represent differential pre-plot-establishment land use history (as grazing intensity) or some factor as fertility or moisture holding capacity.

The plots described by Paulsell (1957) are floristically and physiognomically similar to our study plots. But his specific results, reported as species frequencies after seven years of treatment bear little similarity to ours.

Species Equilibrium

In the control plots, 42 taxa have retreated (including 10 woody taxa) and only three woody taxa have invaded for a net loss of 39 taxa. The overall loss rate was 1.1 taxa per year. Herbs persisted on control plots only until 1978--they disappeared at a rate of 1.6 taxa per year.

In the periodic burn plots there were 12 invader and 21 retreater taxa over the years for a net loss of nine species. In the annual bum plots there were 18 invaders and 33 rctrcaters for a net loss of 15 species. Eliminating species co-occurrence between annual and periodic plots there were 48 retreaters (nearly two per year) and 26 invading taxa for a net loss of 22 taxa (nearly one per year). These losses include three tree retreaters and one tree invader for a net loss of two tree taxa--hardly suggestive of succession toward forest stability. In all taxa, maximum disturbance (annual bum) has induced maximum species movements but with littlc likelihood of establishment of equilibrium vegetation (Grime 1979, Risscr and others 1981).

In the periodic bums, five of 12 invaders appeared for the first time the year of one of the fivc periodic fires. That one species invading per (periodic) fire compares to the annual bums with 0.72 species per fire invading (18 invaders/25 fires).

Life Form/History

Much has been made of life history/life form as control of response to fire. In this flora very little is known of the details of life history response to stress. The form/family species classification used previously indicated a surge of Hemicryptophyte cover with increased fire as was expected (DeSelm and others 1973). Kccley (1981) has shown how life form/history dctermincs response to fire. In this study many response types have been discussed. Only occasionally do they match with life form/family classes (Tables 2 and 3). The tree form, for example, occurred in 13 response types, shrubs occurred in 10, graminoids occurred in 12, legumes occurred in 15 and composites occurred in 13.

Equally, bum plot trees had 11 responses. Rhizome spreading shrubs had seven responses, other shrub six. Among herbs, annuals had nine responses and occurred only in bum plots. Chamaephytes (two species) had two responses in bum plots. Gcophytes (five species) had three responses in bums and did not occur in check plots. Stolmferons herbs (three species) had live responses in fire plots. Climbing herbs (live species) had six responses in bum plots. Graminoids had 11 responses in tire plots, other forbs had 17 responses. Clearly much more needs to be known about the life history of these taxa to explain this level of variable response to fire. Such knowledge would aid those who seek to manage extensive wildland pastures of the southeastern United States (U.S.D.A. Forest Service 1981).

Table 2. Response types, treatment occurrence, numbers of taxa represented per type and life forms/families of taxa in burn and check plots.[1,2]

Response Type	Treatment[3] Occurrence	No. of Taxa in All Treatments	Life Forms, Families Among Taxa
UK	A, P, C	25	T, Sk, Gr, C, Le
WD	A, P, C	13	T, Sk, WV, Gr, C, Ot
UN	A, P, C	16	T, Sh, WV, Gr, C, Le, Ot
WA	P	6	T, C, Le, Ot
WAO	P	2	T
WAN	P	2	
UP	P	6	Sh, C, Le
WB	A, P	5	Gr, Le
WOB	A	5	C
WO	A	3	C, Le
WPN	P	3	Le
WPD	P	1	Le
WC	A		Le
ER	A, P, C	59	T, Sk. WV, Gr, C, Le, Ot
Em	A, P, C	29	T, Sh, Cr. C, Le, Ot
EMDR	C	3	Sh, Gr
EDR	C	5	Gr
M	A, P	22	T, Gr, C, Le, Ot
MLI	A, P, C	10	T, Gr, C, Le
LI	A, P, c	23	T, Sh, Cr. C, Ot
WSK	A, P	32	T, Sh, Gr, C, Le, Ot
EL	A, P, C	5	T, Sh, WV, Le

[1]See text for species in response types.
[2]See table 3 for key to abbreviations.
[3]A = Annual, P = periodic, C = Check plots.

Table 3. Key to response type abbreviations

A	Low in fire years of periodic burns
B	Bulge, curved year-trend, cover high in middle years
C	Composite
D	Decreaser, cover values decrease with years
E	Early years (1965-1972)
G	Sag, cover decreases in middle years
Gr	Graminoid
I	invaders, taxa found on plots after initial inventory of 1965
K	Constant, cover varies little between years, no trends
L	Late year* (1982-1989)
Le	Legume
M	Middle years (1973-1981)
N	Increaser, cover values increase with years
O	Own cycle, cover with apparent high-low periodicity
Ot	Other forb
P	Peak, cover peaks in periodic burn fire years
S	Scattered, taxa wide but only 1/3 - 2/3 of years represented
Sh	Shrub
T	Tree
W	Wide, occurs in two thirds of years scattered in early, middle, and late years
WV	Woody vine

CONCLUSIONS

General

Species in the same genus or family or life form group behaved both similarly in some cases and dissimilarly in other cases with respect to their long-term response to tire. It is impossible to generalize with any accuracy about any group. For example, in the genus Rhus in periodic plots, R. glabra appeared in early and late years, R. toxicodendron disappeared after 1972 and the cover of R. conallina, which was present throughout, peaked in fire years but was depressed between fires. In annual bums, R. glabra occurred only in early years, R. toxicodendron and R. copallina both occurred widely but the latter was an increaser. Other species in the same family or life form exhibit equally variable responses.

Cover of species on control plots changed as it did in treatment plots. The causes of change were not known in either case. Differences extant between bum (annual and periodic) and control plots were not necessarily due only to treatment effects on burned plots; there may have been equally large chronological developmental changes on control plots induced by or paralleled by canopy closure and litter accumulation.

In addition to widely occurring species--all species which exhibited little cover change, or which were increasers or dccreascrs--there were also other classes. Retreaters were present in early or early and middle years and disappeared thereafter. Invaders appeared in middle or middle and late years but were not present in early years. Sporadics, which occurred in early, middle and late years totaled 10 or fewer, have not been considered in this paper. Of the 22 response types seen among the 141 taxa, 10 types occurred in all three treatments, but a few others were specific to treatment or life-form-family categories. A gradient of response occurred among the species present. Those that responded most positively were called pyrophiles--those that responded least positively (but have been there long enough to see once) were termed pyrophobes. Most taxa occurred between these extremes.

Annual Burns

Some taxa seem to have oscillating cover even in the uniform treatment of annual bums. These wides may have responded to some internal growth cycle (as trees that fruit cyclically) (Fowells 196.5). They may also have responded to annual weather changes (Fritts 1976). For example Towne and Owensby (1989) found annual Kansas prairie biomass correlated with precipitation. Or they may have responded to changes in competition from neighbors whose cover responded as above. Weather regimes seemed to be a likely source of year to year variation in species behavior. Its effects will be considered in a later paper.

A few taxa (as Schizachvrium) peaked in middle years, cover before and after these peaks was generally lower. They were, in early years increasers, in later years decreasers. These may have been the oscilating cover type with their own cycle but with a very long time between peaks.

Taxa which occurred only in middle years may have been a low average cover example of the middle-years-peak species noted above. Or perhaps they should be considered invader-retreaters. They occurred in four form-family groups in all three treatments. With more extensive sampling, these might prove to be middle years peak species noted above. Wide, middle-year-sag species occurred as did those that occur only in early and late years.

Periodic Burns

Composite seedlings established in the year of periodic fires on those plots. Comparable cohorts of grass and legume seedlings were not seen. New taxa were also most likely to invade the periodic plots during bum years.

Six kinds of response types occurred in periodic burns only. Three of these had low cover in tire years, and high cover between tires; there were those with more or less constant cover between peaks and between valleys, those with a middle year-group bulge, and those which were increasers, Three other types have peak cover during fire years. There were those with constant or uniform peaks and valleys across the year-series, those which were increasers, and those which were decreasers.

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IMPLICATIONS FOR LONG-TERM PRAIRIE MANAGEMENT FROM SEASONAL BURNING OF LOESS HILL AND TALLGRASS PRAIRIES

Thomas B. Bragg[1]

Abstract-Data From prescribed bums of northwestern Iowa loess hills and eastern Nebraska tallgrass prairies were used to assess the effects of season of burning and implications for long-term management of grassland ecosystems. Overall forb cover declined most without burning (-22%). Compared to unburned areas, species number on both sites was higher (i-5) with fall burning with the response most pronounced at the loess hills site. The response of species such as false sunflower (Heliopsis helianthoides) suggest that summer and fall burning may do most to encourage seedling germination and establishment. Other species, such as white aster (Aster ericoides) on the tallgrass site and gray goldenrod (Solidapo nemoralis) at the loess hills site, increased in cover with summer or fall burning. Some species showed significant changes irrespective of treatment; in the tallgrass area, porcupine-grass (Stipa spartea) decrease d and flowering spurge (Euphorbia corollata) increased.

The vegetative responses recorded suggest the likely importance of applying some summer and fail burning, in conjunction with the usual spring burning, to the long-term maintenance of diversity in the tallgrass, loess hills, and perhaps other, grassland ecosystems

INTRODUCTION

The effects of fire in prairie ecosystems of the central North America have been extensively studied (for example, see Vogel 1974; Risser and others 1981; Wright and Bailey 1982 j. Generally, studies have determined that fire is a natural component of these grassland ecosystems and that continued fire management is important, whether the objective is to maintain the vitality and diversity of the natural ecosystem or to manage for other, agroeconomic, purposes. More recently, research has begun to reline the understanding of fire's role in grassland ecosystems with an increased focus on fire frequency (the number of years between bums) and fire season (the season during which fire is applied). Grassland fires occurred naturally at various times of the year, including summers (Moore 1972; Bragg 1982), thus the seasonal aspect of fire is important to understanding its rule in the long-term management of this ecosystem. Studies comparing differences in effects of season of bum, however, have largely focused on the northerly, mixed-grass prairies of North and South Dakota. Few community-level studies on this specific aspect of fire ecology have been published for the tallgrass prairie.

Response of Grasses and Grass-Like Species

Many comparative studies on the effect of fire season on prairie vegetation have focused on either Spring or Fall, these being times most often appropriate for the management objective of cattle grazing. Burning during these seasons, particularly in the Spring, has also been carried over into ecological management of grasslands. Studies on the effects of fire often focus on grasses since they are the predominant vegetation type and since they have the greatest effect on forage production. When considered ecosystem-wide, the results of such studies have been found to be similar only in that they all differ considerably depending on season of burning, latitudinal location, and local climatic conditions. These differences on grass and grass-like species are reflected in two general areas of fire effects: productivity and species composition. For the sake of comparison of effects of season of burning, results are discussed separately for northern prairies (e.g. North and South Dakota) and for central (more southerly) prairies (e.g. Nebraska, Kansas, and Oklahoma).

Effect on Productivity

Native grass or grass-lie species, for which studies on productivity have been conducted both under various burning regimes and at different locations, include the warm-season (C,) species blue grams [Bouteloua gracilis (H.B.K.) Lag. ex Griffiths] and cool-season (C,) species such as western wheatgrass (Agropyron smithii Rydb.), needle-and-thread (Stipa comata Trin. & Rupr.), and Kentucky bluegrass (Poa pratensis L.). Also included is threadleaf sedge (Carex tilifolia Nutt.) although the carbon-fixation status of this species does not appear to have been determined.

As with most species, the effect of burning on blue grama differ by location and'with climatic conditions. In studies in South Dakota, for example, spring (April) burning was found to reduce the productivity of blue grama whereas the response to fall burning was variable, increasing production when precipitation was adequate and decreasing it when precipitation was low. In the mixed prairie of North Dakota, however, both spring (May) and fall (October) fires increased blue grama production with spring burning resulting in the

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greater increase (Whisenant and Uresk 1989). For western wheatgrass in South Dakota, productivity increased with Fall burning but either increased or decreased in response to spring burning, again depending on precipitation. The response of this species to fire in the more northern North Dakota prairies, however, was an increase in production with both spring and fall burning (Whisenant and Uresk 1989). As with blue grama, spring burning resulted in the greater response.

Needle-and-thread is one native grass species for which productivity is generally reduced by fire throughout much of its range. In the mixed-grass prairies of both North and South Dakota, productivity of this cool-season grass declined with both spring and fall bums (Dix 1960; Wright and Klemmedson 1965; Engle and Bultsma 1984; Whisenant and Uresk 1989). Gartner et al. (1986), however, did report a greater productivity of this species with both spring and fall bums. In the central, more southerly, Nebraska tallgrass prairie, however, summer mowing (approximating summer burning) resulted in a higher canopy cover of the congeneric porcupine-grass (Stipa spar-tea Trin.) than in areas burned in the spring (Hover and Bragg 1982).

Threadleaf sedge, a species common in the mixed-grass prairies of the Dakotas is particularly informative. It has been reported to be unaffected by spring or fall burning (Schripsema 1977; Whisenant and Uresk 1989) although it is reduced by fall burning in North Dakota (Dix 1960; White and Currie 1983). The general response of this species to burning is similar to that of needle-and-thread, a C_3 species, but unlike that of western wheatgrass, another C_3 species. If this observation is extrapolated to other species, it suggests either that the response to burning may occurs independent of carbon-fixation pathway or that the tire conditions under which the previous studies were conducted are not fully known.

A summary of the effects of fire on grass productivity, then, suggests that the complexity of fire effect studies. Only two common denominators are suggested, first, fires in dry years reduce productivity, and second, not all C, and C, species can be expected to respond similarly to burning.

Effect on Composition.
Another aspect of the effect of season of bum is the response of the community as a whole, which is the principal focus of this study. Despite individual species responses, community level studies in the northern mixed-grass prairie have indicated that season of fire occurrence is not a sufficient factor to substantially alter community composition (Dix 1960) or to alter the C_3/C_4 ratio of the northern mixed-grass prairie (Steutcr 1987).

The C, grasses of the northern mix&prairie appear to be a fire-adapted guild (Steuter 1987). The tendency for cool-season grass species to increase or to bc unaffected in the northern Great Plains, however, is opposite to the response in more southerly tallgrass prairies of Kansas and Oklahoma. In these tallgrass prairies, dominated by warm-season, C_4 species, spring bums more consistently decrease cool-season, C_3 species, including porcupine-grass, Kentucky bluegrass, Canada wild rye (Elymus canadensis L.), and Scribner dichanthelium [Dichanthelium oligosanthes (Schult.) Gould var. scribnerianum (Nash) Gould] (Hensel 1923; Ehrenreich 1959; Hadley and Kieckhefer 1963; Robocker and Miller 1955; Old 1969; Anderson and others 1983). The effect of fire at other seasons, however, has not been widely studied although it has been found that little bluestem (Androooeon scoparius Michx.), is most adversely affected by summer (July) bums (Adams et al. 1982)

Response of Forbs
General results on fire effects indicate that forbs are increasingly adversely affected as burning occurs at increasingly later spring dates. For example, late spring burning in the tallgrass prairies of Kansas reduced all forbs (McMurphy and Anderson 1965; Towne and Owcnsby 1984; Hulbert 1988) compared to earlier dates. However, while these studies reflect a reduction in cover (suggesting also a reduction in productivity), the actual composition of forbs is little effected (Anderson 1965). Similar effects related to season of bum are reflected in the shortgrass prairies of western Kansas. There, forbs are less effected by dormant season (fall/winter) bums than by spring bums which occur after they have initiated growth (Hopkins and others 1948).

One principal study on the response of individual forb species to seasonal effects of burning was conducted by Biondini and others (1989) in northern mixed prairie. In this study, the density of nine forbs was significantly effected by tire season. Species responses relevant to the present study include western ragweed (Ambrosia psilostachya DC.) and white aster (Aster ericoides L.), which were most positively affected by fail bums, blue lettuce (Lactuca oblongifolia Nutt.) most effected by summer burns, and stiff sunflower [Helianthus rigidus (Cass.) Desf.] and wavy-leaf thistle [Cirsium undulatum (Nutt.) Spreng.] most effected by spring burns. Only pasque flower (Anemone patcns L.) had the highest density without burning. In another study in the northern prairie region, Schripsema (1977) recorded increases in species such as silver-leaf scurf-pea (Psoralea argophylla Pursh) with late spring (late May) burning whereas a winter (March) bum had the opposite effect.

In more southerly tallgrass prairies, fall burning increased rigid sunflower (Solidigo rigida L.) (Schwegman and McClain 1985) and leadplant (Amorpha canescens Pursh) (Towne and Owensby 1984) although the greatest increase was among the annual species such as grooved flax (Linum sulcatum Ridd.) and white sweet clover (Melilotus alba Medic. (Schwegman and McClain 1985). Whorled milkwort (Polygala verticilata L.) and grooved flax were best established in spring burned plots (Schwegman and McClain 1985) but late spring (mid-May) burns adversely affected species such as prairie violet (Viola pedatifida G. Don), white-eyed grass (Sisyrinchium campestre Bickn.) and downy gentian (Gentiana puberulenta Pringle); gay-feather (Liatris aspera Michx.) and smooth blue aster (Aster laevis L.), however, ad significantly more leaves with late than with early spring bums (Lovel 1 and others 1983).

The results of previous studies on effects burning suggest that the basic characteristics (e.g. productivity and species composition) of grasslands of different latitudes should respond differently to fire and that the response will be further modified by season of burning and climatic conditions. The objective of this paper is to identify such differences by comparing the results from two grasslands that are similar physiognomically but that differ in both latitude and dominant species. Specifically, the study will compare a Loess Hills prairie of northwestern Iowa and a tallgrass prairie of eastern Nebraska in order to assess similarities in plant responses to fire. Further, the study is designed to assess the possible role of different seasons of burning and their implications for global application in long--term management of grassland ecosystems.

METHODS AND MATERIAL

The study involves unreplicated sites and unreplicated locations within each site. This design was necessitated by a combination of the travel distance, the absence of additional sites to which access could be controlled, and the time required for both fire treatment and field evaluation. Therefore, extrapolation of results to other sites of the same vegetation type can only be used in a speculative manner and then only with caution. However, in those instances where similar responses to burning are noted at each site, the response could be considered to be replicated (e.g. two prairie sites were evaluated) and thus it is more likely to be representative of general trends. In addition to limiting how broadly the results can be inferred, the lack of adequate replication limited the kind of statistical evaluations that could be appropriately applied.

An additional caution to extrapolation of results is necessary due to the absence of any tire treatment at either site for many years, probably decades, prior to the study. The plant communities that were burned, therefore, may not be the same as those that dominated historically when fires reoccurred with some regularity. Studies at these sites are continuing at least through the 1990's in order to assess this possibility.

Study Sites

The Loess Hills study site was located on Five Ridge Prairie (within Sections 20, 21 and 29, Township 91N, Range 48;W) located in northwest Iowa approximately 20 kilometers north of Sioux City. The prairie is managed by the Plymouth County Conservation Board in cooperation with the Iowa Chapter of The Nature Conservancy. Treatment plots were located in the northwest quarter of Section 29 along a southwest facing, 20-26% slope on which native prairie vegetation prevailed. The site was dominated by grass species, particularly little blue-stem and plains muhly [Muhlenbergia cuspidata (Torr.) Rydb.]. The soil of the site is a Hamburg silt loam (Typic Udorthent soil subgroup, Entisol soil order). The Hamburg series consists of excessively drained, calcareous, silty soils formed on loess (Worster and Harvey 1976). Climate of the region is continental with normal daily highs of 30 C in July and a low of minus -14 C in January. Normal annual precipitation (based on 1951 to 1980 data) averages 64 centimeters with 74% occurring during the growing season (April through September). Climatic data are from National Oceanic and Atmospheric Association (1989a).

The tallgrass study site was located on Stolley Prairie, approximately 20 kilometers west of Omaha, Nebraska in Douglas County (NW 1/4 Section 15, Township 15N Range 11 E) Stolley prairie is privately owned, jointly leased for wildlife habitat by the Audubon Society of Omaha and the Papio-Missouri River Natural Resources District, and managed by the Biology Department, University of Nebraska at Omaha. The prairie had been mowed for more than 20 years until haying ceased in 1980 with leasing of the site. Treatment plots were located on a north-facing, 7-l 1% slope, tallgrass prairie dominated by big bluestem and porcupine-grass. The soil is a Marshall silty clay loam (Typic Hapludoll Subgroup, Mollisol Soil Order), a deep, well-drained soil formed in loess (Bartlett 1975). Climate of the region is continental with normal mean highs of 30 C in July and normal mean lows of minus 12 C in January. Normal annual precipitation (based on 195 1 to 1980 data) averages 76 centimeters with 74% occurring during the growing season (April through September). Climatic data are from National Oceanic and Atmospheric Association (1989b).

Treatment

At each study site, treatment plots, approximately 20 by 20 meters in size, were established in a strati&d, complete block design with plots situated at either upper-slope or mid-slope locations. A single, lo-meter (Tallgrass) or 20-meter (Loess Hills) transect was centrally located within each treatment plot and permanently established with two metal rods at each end. Differences in transect length were due to size of the area available for the study; the loess hills prairie was smaller in size due to woody plant invasion from lowland valleys. Along each transect, ten microplots were systematically placed. I was able to evaluate the same microplots each year of the study by extending a meter tape between the rods and using established intervals at each evaluation.

Neither of the study sites had been burned within recent memory. After preliminary data collection in 1981, randomly selected treatment plots at the Tallgrass site were burned in early May, early July, and mid-Scptembcr 1983. With the exception of Fall treatments, all plots were resampled in the Fall of 1983, 1984, and 1986. Fall bum treatment plots were not sampled in 1983 since treatment had not yet been applied: evaluations for 1981 were used to represent pm-bum conditions for this treatment.

At the Loess Hills site, plots were burned in mid-October 1986 (after pretreatment data collection) and in late April and early July 1987; the fall burn was conducted in 1986 (rather than 1987) so that all fire treatment plots would be effected by the same (1987) growing season. Treatment plots were resampled the Fall of 1987, 1988, and 1989. At this site, spring evaluations were also conducted in each sampling year in order to record any species that were not visible in the Fall.

Data Collection

Because of different, site-specific characteristics of the plant canopy cover, microplot size varied for each site. Microplot size was 30 x 50 centimeters For the Tallgrass site and 50 x 100 centimeters for the Locss Hills site. The larger size used in the Loess Hills was needed due to lower total plant canopy cover and more widely spaced plants. The number of microplots to be evaluated was determined From preliminary sampling of each community type from which it was determined that ten microplots incorporated 90% of all species situated along each transect in each site. Microplots were systematically placed along each transect to facilitate relocation in subsequent years.

Within each microplot, canopy cover of each species was recorded as were the general cover categories of "bare soil" (soil devoid of surface litter; litter is dead plant matter that is no longer connected to a living plant) and "forb". Coverage was estimated based on procedures modified from Daubcnmire (1959). Cover categories were 0%, 1-5%, S-25%, 25-50%, 50-75%, 75-95%, 95-99%, and >99%. Because of lack of adequate replicates and for the purpose of this broad scope paper, descriptive statistics (Mean $\pm$ Standard Error) were calculated for all species and used to compare effects of treatment.

RESULTS AND DISCUSSION

Site Differences

Site differences are characterized by data collected prior to the year of fire treatment. In addition to species differences, noteworthy pretreatment differences included significant differences (based on Standard Error) in bare soil (soil not covered with litter) (7% on the loess hills site compared to 1% on the tallgrass site) and in forb cover (38% loess hills; 45% tallgrass) (tables I and 2). In addition, 49 native species were recorded in loess hills, pretreatment microplots compared to 44 species for the tallgrass microplots. After adjusting for species observed at each site, but not necessarily present within microplots, 16 species were identified to be unique to the loess hills with 7 unique to the tallgrass site. This supports a qualitative observation made during field evaluations that the loess hills had higher plant species diversity than the tallgrass site and that this difference may bc due to more niches afforded by the greater surface heterogeneity as reflected in bare soil. The tallgrass site, however, did have an active pocket gopher (Gwmvs bursarius) population that has the potential to profoundly affect the ecosystem (Huntly and Inouye 1988) and is likely to afford some, continuous bare soil niches. Pocket gopher activity was not observed at the loess hills site

Community-Level Responses

Some treatment effects were found to be similar at both sites of which the effect on forb cover and Species Richness (the total number of species) are most noteworthy. The fall following spring and summer treatment, forb cover declined from pre-treatment conditions in all microplots at both sites regardless of whether burned or unburned. The cause for this response is unclear except that precipitation does not appear to be the principal factor; both treatment years were followed by hear average or above-average precipitation (fig. I). While the decline in forb cover occurred at both study sites, it was greatest in unburned plots (-15%) at the tallgrass prairie and second greatest in the unburned plots (-33%) at the loess hills prairie where it was second only to summer burning (- 35%). Three growing seasons following fire treatment (4 seasons for the tallgrass prairie), the unburned plots continued to show the greatest loss of forb cover based on pre-treatment values (-18% for tallgrass; -21% for loess hills) (tables 1 and 2). It should be noted, however, that the decline in forb cover in the absence of fire, reflects only a change in the amount of a species and not necessarily changes in population size.

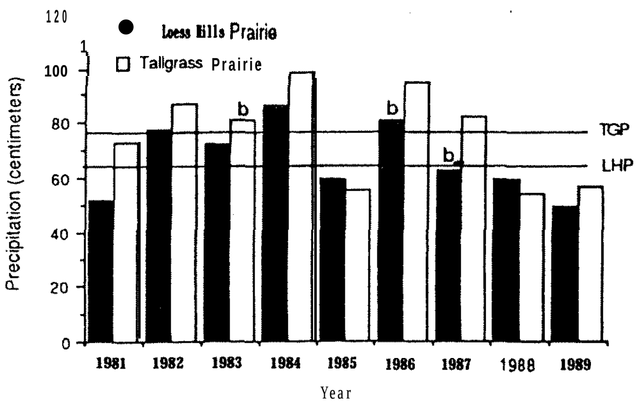


Fig. I.-Precipitation for Sioux City, Iowa [Loess Hills (= LWP) Prairie site] and North Omaha, Nebraska [Tallgrass (TGP) Prairie site] weather stations for the years 1981 through 1989. Horizontal lines represent normal precipitation (National Oceanic and Atmospheric Administration 1989a, 1989b). "b" = bum treatment year (For LHP, 1986 = Fall treatment and 1987 = Spring and Summer treatments)

Table 1. Canopy cover (Mean ± Standard Error) of tallgrass prairie site species with frequency values greater rhsa 50 percent, in either this or the loess hills prairie site. Values represent 20 microplots from two transects per treatment. scientific and common names are from the Crest Plains Flora Association (1986). N.D : no data; tr : <0.5 percent cover.

Species	Evaluation Year	Season of Prescribed Burn: Spring	Summer	Fall	Unburned
GENERAL CATEGORIES					
Number of species	1981	29	32	29	31
	1983	34	24		31
	1984	32	30	30	26
	1986	27	32	29	31
Forb Cover	1981	39±3.6	49±5.2	44±3.8	4915.9
	1983	21±3.0	1422.7		16±2.4
	1984	19±3.8	32±5.3	50+3.6	24±4.1
	1986	33±4.3	43±6.3	33±5.1	28±4.4
Bare Soil	1981		2±1.0	2±1.9	
	1963	6912.5	67±6.8		2...2
	1984	2±1.0	7±3.5	1012.6	8±3.9
	1986	2±1.9	8±3.5	2±1.0	2±1.0
Achilles millefolium (yarrow)	1981	6±2.2	3±2.0	2±1.0	2±1.0
	1983	1±0.8	tr		2±1.0
	1964	1±0.8	7±3.6	1±0.8	tr
	1%.5	1±0.8	3±1.4	tr	tr
canescens (leadplant)	1981	0	2±1.2	2±1.2	tr
	1983	1±0.8	8±4.5		4±2.1
	1984	4+2.1	7±3.6	23±5.2	5±2.7
	1986	43.2	8±2.7	16±4.8	1±0.8
Andropogon gerardii (big bluestem)	1981	78±2.7	n-4.5	81±3.8	5924.8
	1983	91±2.1	74±6.0		77±3.8
	1984	81±2.7	85±5.0	86±1.7	60±6.0
	1986	77±2.9	74±5.6	83±2.6	74±5.3
Andropogon scoparius (Little bluestem)	1981	10±3.8	1±0.8	17±3.8	2±1.0
	1983	1±0.8	tr		1±0.8
	1984	0	0	0	0
	1986	1±0.8	2±1.9	1±0.8	4±2.1
Anemone cylindrica (candle anemone)	1961	tr	1±0.8	0	1±0.8
	1983	tr	tr		9±7.5
	1984	tr	0	0	3±2,0
	1986	0	1±0.8	0	3±2.1
Aster ericoides (whi te ester)	1981	tr	1±0.8	3±1.2	1±0.8
	1983	0	tr		0
	1984	0	3±3.1	2±1.9	0
	1986	1±0.8	6±2.7	9±3.9	0
Bouteloua curtipendula (sideoats grama)	1981	17±3.9	17±4.7	21±3.9	13±2.8
	1983	3±2.0	5±1.5		5±1.5
	1984	10+4.7	2±1.0	1±0.8	2±1.0
	1986	221.9	2±1,0	1±0.8	1±0.8
Carex spp. (sedge)	1981	10+1.6	12±1.4	13±1.2	10±1.6
	1983	19±3.3	15±2.1	-	6±1.7
	'984	16±3.7	18±3.2	13±1.9	10±2.2
	1986	4±1.s	10±3.8	3±1.4	5±2.2
Ceanothuss (New Jersey tea)	1981	11+4.7	5±3.6	1±0.8	1±0.8
	1983	1G5.3	3±3.1		6±4.6
	1984	13+5.9	6±3.6	0	4±3.1
	1986	es.3	4±3.1	0	6±3.6
YZansIs (horse-wed)	1981	0	9	0	0
	1983	0	0		0
	1984	tr	4±1.3	tr	0
	1986	0	tr	1±0.8	2±1.0
Dichanthelium oligosanthes var. scribnerianum (Scribner dichanthelium)	1981	9+1.6	10-1.5	14+0.6	7±1.5
	1983	6±2.0	tr		3±1.2
	1984	10±3.1	2+1.0	4+1.3	1±0.2
	1986	1±0.8	1±0.8	1±0.2	tr
Elymusnsis (Canada wild rye)	1981	5+2.2	1+0.8	3±2.1	4+1.5
	1983	4±1.3	4±2.1		4±1.5
	1984	6±2.3	11±4.1	2+1.0	4±2.1
	19%	1±0.8	3±1.4	2±1.0	2±1.0
Equisetum laevigatum (smooth scouring rush)	1981	3+1.2	0	2±1.0	4+1.5
	1983	1+0.3	0		2±1.0
	1984	652.7	0	1±0.8	5±2.2
	1986	8±2.7	0	2±1.0	4+2.2

Table 1. (continued)

Species	Evaluation Year	Season of Prescribed Burn: spring	Summer	Fall	Unburned
Erigeron strigosus (daisy fleabane)	1981	0	2±1.2	0	1±0.8
	1983	tr	0		0
	1984	tr	1+0.8	3±1.2	0
		0	1±0.8	2±1.0	0
Euphorbia corollata (flowering spurge)	1981	5±1.6	3±1.4	5±1.6	8±2.8
	1983	8±3.2	5±2.2		>±1.6
	1984	6+3.1	2±1.9	4+2.1	10+4.2
	1986	1023.4	9±3.8	8±3.8	1323.2
Heliopsis helianthoides (false sunflower)	1981	0	4±1.5	1±0.8	tr
	1983	tr	2±1.2		2+1.9
	1984	1+0.8	9+4.5	tr	2+1.9
	1986	4±3.1	20±5.4	5±3.2	3±2.0
Kuhnia eupatoroides (false boneset)	1981	1±0.8	0	0	0
	1083	tr	0		1+0.8
	1984	tr	tr	tr	1±0.8
	1986	1±0.8	tr	9	3±2.1
Linum rigidum var. compactum (stiffstem flax)	1981	6±2.3	4±1.5	5±1.6	2±1.0
	1983	0	0		0
	1384	tr	5+1.6	20±4.2	0
	1986	0	lfp.8	0	0
Oxalis dellenii (gray-green wood sorrel)	1981	tr	tr	1±0.2	tr
	1983	3±1.0	|		0
	1984	tr	tr	1±0.2	0
	1986	tr	tr	tr	tr
Phlox pilosa (prairie phlox)	1981	6±1.7	12±2.0	8±1.6	5+1.6
	1983	11±3.4	2±0.8		952.7
	1984	10±3.2	6±1.5	16±3.6	8±2.7
	1986	16±4.0	13±2.4	20+3.4	10±3.1
Physalis virginiana (Virginia ground cherry)	1981	0	0	0	0
	1983	9	3±1.4		2+1 0
	1984	tr	2±1.0	0	-0'
	1986	4±2.6	10.8	1±0.8	2±1.2
Poa pratensis (Kentucky bluegrass)	1981	74±4.0	80±3.9	70±5.1	69±4.1
	1983	58±6.8	11±2.0		81±3.2
	1984	66±4.4	56±4.7	80±2.8	86±2.6
	1986	79±2.8	72±5.0	71±4.1	71±4.2
Rudbeckia (black-eyed susan)	1911	7+2.2	3±1.4	11±2.5	8±2.2
	1983	2+1.0	0		0
	1984	2±1.0	10+4.1	14+3.5	0
	1986	0	3±1.2	2±1.0	0
Sorghastrum nutans (indiangrass)	1011	4±2.1	4±1.5	8±1.7	3±1.4
	1983	3+1.4	6±2.3		5±1.5
	1984	10±3.8	8±2.7	16+3.7	16±4.8
	1986	22±5.4	8+3.1	2ois.0	23±6.0
Sporobolus asper (rough dropseed)	1981	2±1.0	4+1.6	8±1.7	0
	1983	0	-0		1±0.8
	1984	0	1+0.8	0	2±1.9
	1986	0	-0	|	tr
Stipaea (porcupine-grass)	1981	36±7.1	24±6.5	25±7.4	61±7.5
	1983	18±3.6	3±1.4		32 5.6
	1384	28±5.5	16±5,2	8±5.1	43±7.5
	1986	5±2.1	2±1.0	4±2.1	6±2.7
Taraxacum officinale (common dandelion)	1981	1±0.8	5±1.5	1±0.8	1±0.2
	1983	0	1+0.8		0
	1984	0	-0	0	0
	1986	0	tr	0	0
Trifolium pratense (red clover)	1981	36+4.1	43+5.5	35±3.7	42+4.9
	1983	9±2.1	-0		1±0.8
	1984	3±1.3	10+3.5	35±5.5	tr
	1986	tr	4x3.1	0	1±0.8
Viola pedatifida (prairie violet)	1981	1±0.8	1±0.8	1±0.8	tr
	1983	1±0.8	2±1.0		tr
	1984	2±1.0	2±1.9	3±1.2	tr
	1986	2±1.0	1±0.8	3±1.2	1±0.8

Table 2. Canopy cover (Mean ± Standard Error) of loess hills prairie site species with frequency values greater than 50 percent, in either this or the tallgrass prairie site. Values represent 20 microplots from two transects per treatment except for the Unburned plot which represents one transect. Scientific and common names are from the Great Plains Flora Association (1986). tr = 10.5 percent.

Species	Evaluation Year	Season of Prescribed Burn: Spring	Summer	Fall	Unburned
GENERAL CATEGORIES					
Number of Species	1986	38	32	34	24
	1967	35	25	34	20
	1988	33	33	34	20
	1989	33	32	32	20
Forb Cover	1986	32±3.7	34±3.3	39±3.1	47±7.1
	1987	22±2.4	21±2.6	32±2.6	32k. 2
	1988	18±2.6	25±3.3	30±3.5	30±6.6
	1989	21±2.8	28±4.1	33±2.1	2415.4
Bare Soil	1986	4±1.4	9±2.1	8±2.2	7±2.2
	IS87	23±2.8	37±1.8	35±2.5	2±0.2
	1988	11±2.1	40TI.7	30±2.5	10±2.1
	1980	8±2.6	23±2.5	2123.0	1323.4
Andropogon gerardii (big bluestem)	1986	3227.9	9±3.2	16±4.0	20±8.1
	ISS7	36±8.3	7±2.7	15±3.8	23±8.6
	1988	34±7.9	6±2.2	1423.1	19±4.6
	1989	32±6.0	12±3.8	14±3.5	25±6.7
Andropogon scoparius (little bluestem)	1986	66±4.7	83±1.7	84±2.0	78±5.8
	1987	49±5.3	80±2.1	79±2.9	78±5.0
	1988	46±6.0	82±2.6	79±2.7	80±3.1
	1989	36±4.8	80±2.7	73±3.6	78±3.5
Anemone (pasque flower)	1986	19.8	1±0.7	2±1.0	1±0.3
	1987	3±1.3	652.2	k1.2	1±0.3
	1988	2±0.7	3±1.2	3±1.2	co. 3
	1989	1±0.8	2±1.0	2±0.7	110.3
Aster sericeus (silky aster)	1986	5±1.5	10±2.1	11±3.0	27±4.6
	1987	3±1.4	5±1.4	7±1.6	25~4.6
	1988	3±1.2	8±1.6	8±1.6	27±5.8
	1989	3±1.2	7±1.5	6±1.5	23±4.4
Bouteloua curtipendula (sideoats grama)	1986	22±2.4	25±3.0	26±2.6	22±3.5
	1987	23±2.8	20±2.9	25±2.9	26±3.8
	1988	29±3.5	30±3.5	3322.7	33±3.1
	1989	16±3.3	2622.6	24±3.1	24±6.3
Carex spp. (sedge)	1986	4±1.4	1±0.2	1±0.2	tr
	1987	9±2.6	1±0.8	4±1.4	tr
	1988	7±1.5	3±1.2	2±1.0	1±0.3
	1989	2f0.7	4±1.3	120.2	tr
Dalea spp. (prairie clover)	1986	1±0.8	7±2.7	2±1.0	tr
	1987	GO.8	6±2.3	4±1.5	0
	1988	3±2.0	11±3.8	431.5	tr
	1989	2±1.0	1214.0	3±2.0	tr
Dichanthelium oligosanthes var. scribnerianum (Scribner dichanthelium)	1986	tr	tr	tr	0
	1987	tr	tr	1±0.8	tr
	1988	1±0.9	tr	tr	tr
	1989	tr	tr	tr	tr
Dichanthelium wilcoxianum (Wilcox dichanthelium)	198.5	1±0.2	tr	tr	tr
	1987	tr	tr	1±0.8	tr
	1988	tr	0	tr	0
	1989	tr	tr	tr	tr

Table 2. (continued).

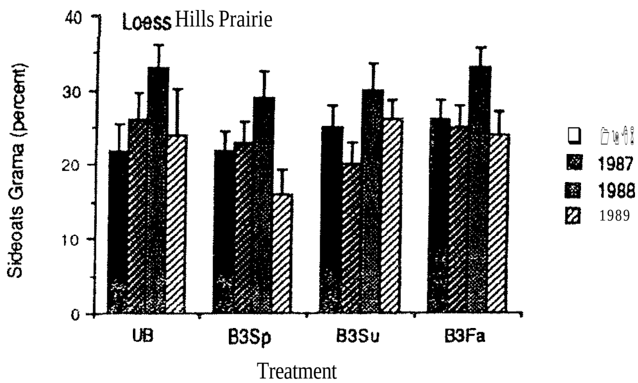
Species	Evaluation Year	Season of Prescribed Burn: Spring	Summer	Fall	Unburned
Echinacea angustifolia (purple coneflower)	1986	tr	3±0.8	1±0.8	3±2.0
	1987	1±0.8	tr	3±1.4	tr
	1988	1±0.8	2±1.0	2±1.0	3±2.0
	1988	tr	1±0.8	2±1.0	2±1.5
Hedeoma hispidum (rough false pennyroyal)	1986	1±0.3	2±0.4	2±0.4	1±0.3
	1987	0	0	0	tr
	1988	tr	0	0	0
	1989	0	tr	0	tr
Linum rigidum var. compactum (stiffstun flex)	1986	tr	2±1.2	1±0.8	tr
	1987	tr	0	tr	0
	1988	tr	tr	1±0.8	0
	1989	0	1±0.2	tr	tr
Lygodesmia juncea (skeletonweed)	1986	tr	4±1.5	6±1.5	6±2.4
	1987	tr	1±0.8	11±2.6	aT2.4
	1988	1±0.8	4±1.4	11±2.6	5±2.0
	1989	tr	2±1.0	5f1.4	7±2.3
Muhlenbergia cuspidata (plains muhly)	1986	8±2.7	6±1.6	4±1.5	tr
	1987	6±2.2	2±1.2	3±1.4	4±2.3
	1988	10±3.1	611.6	4±1.5	tr
	1989	1055.4	5±2.2	4±1.5	tr
Psoralea tenuiflora (wild alfalfa)	1986	tr	tr	1±0.9	tr
	1987	tr	tr	0	tr
	1988	tr	1±0.9	0	0
	1989	tr	tr	0	tr
Rhus glabra (smooth sumac)	1986	18±6.8	2±1.2	7±3.6	7±4.0
	1987	23±6.6	2±1.0	5±2.2	6±3.9
	1988	18±4.6	2±1.0	6±3.2	6-3.9
	1989	15±4.1	1±0.8	4±2.1	3_2.0
Sisyrinchium campestre (white-eyed grass)	1986	5±2.1	6±1.8	7±2.0	6±2.0
	1987	tr	1±0.4	2±0.4	2±1.5
	1988	tr	1±0.4	2±0.9	-0
	1988	1±0.4	1±0.4	tr	0
Solidago nemoralis (gray goldenrod)	1986	9±2.7	9±1.6	7±1.6	7±2.3
	1987	4±1.3	3±1.1	7±1.b	6±2.4
	1988	311.4	1±0.2	8±1.6	1±0.3
	1989	3±1.2	1±0.2	5±1.4	tr
Solidago rigida (rigid goldenrod)	1986	4±1.3	3±1.2	4±1.4	tr
	1987	8±2.2	3±1.2	6±1.b	tr
	1988	6±1.6	1±0.8	6±1.6	2±2.0
	1988	6±1.6	1±0.8	6±1.b	2±1.5
Sorghastrum nutans (indiangrass)	1986	14±4.2	3±2.1	6±2.7	9±5.0
	1987	16±4.3	8±3.1	8±3.6	9±5.0
	1988	21±6.2	7±2.7	12±3.8	13±5.7
	1989	17±5.0	4±2.6	10±3.8	6±2.4
Verbena stricta (hoary vervain)	1986	PO.2	tr	tr	tr
	1987	1±0.8	tr	1±0.8	tr
	1988	tr	0	tr	tr
	1989	0	0	tr	0

One consideration in the changes in species richness and forb cover with treatment is climate of which precipitation is an important component. Precipitation during all post-bum years at the loess hills site and during the second post-bum year at the tallgrass site averaged less than normal (fig. 1). The different responses of species diversity, both positive and negative, to season of burning, suggest that, while a drought year may be a poor time to bum prairies during some season of a year, some other season of the same year may be a reasonable time to bum. These results, if verified by further study, are particularly relevant to prairie management. Woody plant invasion threatens both loess hills (Heineman, 1982) and tallgrass (Bragg and Hulbert 1976) prairie areas. This study suggests that tire, a management tool that controls woody plant invasion, can be applied during appropriate seasons, even those of a drought year, without adversely affecting long-term species diversity. Further, results of this study that show increased diversity with different seasons of bum, suggest that any season of burning is better for maintaining grassland diversity than is fire exclusion. In the present study, this effect appears to be particularly true for the loess hills prairie ecosystem.

While one common effect between sites was a decline in forb cover without burning, another common effect was a short-term decrease in Species Richness with summer burning (-8% for tallgrass species; -7% for loess hills species); sufficient data were not available, however, to test for the statistical significance of this difference. This overall reduction in number with summer burning, however, did not persist beyond the second post-bum growing season (tables 1 and 2). For example, three growing seasons after burning in the loess hills, Summer treatment plots had recovered to pm-bum numbers. By this time, it was the Spring burned microplots that reflected the greatest loss of species (-5); unburned microplots averaged four less and fall burned plots averaged two less species. For the tallgrass prairie site, four growing seasons after treatment, species richness of all but thc Spring treatment (-2 species) was at pre-bum numbers. Thus, for both the loess hills and tke tallgrass prairie ccosystcms, spring treatments, which represent the most widely applied time of fire management, showed the greatest long-term (f-4 year) loss of species;

Individual Species' Responses

The response of individual species provides further insight into the seasonal effect of burning on specific prairie types and on grassland ecosystems in general. Of the several species common to both sites, only big bluestem (Andropogon gerardii Vitman), grass [Sorghastrum nutans (L.) Nash], sideoats grama [Bouteloua curtipendula (Michx.) Torr.), and sedge (Carex spp.) were sufficiently abundant to make comparisons between the tallgrass and loess kills sites. Tke responses of all but sideoats grama were similar between sites (tables 1 and 2). Sideoats grama, however, declined an average of 11 percent with all treatments at the tallgrass site but was maintained at or above prebum amounts both without burning and with all bum treatments except spring burning (fig. 2). The most likely explanation for this difference in



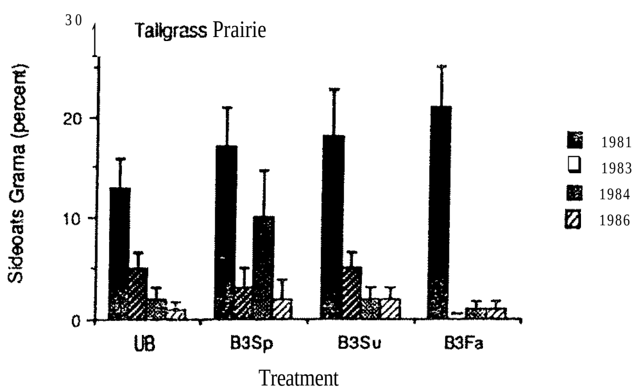
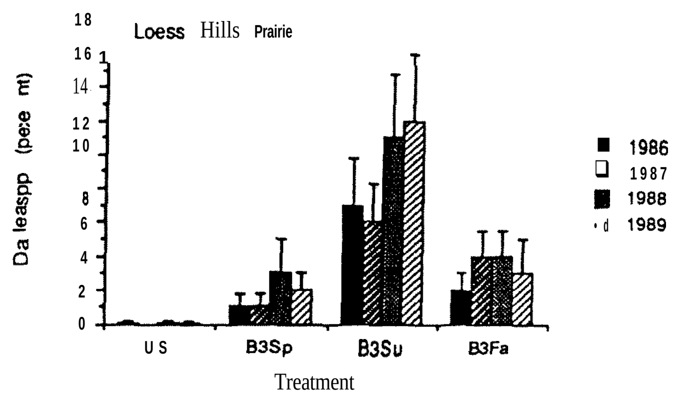


Fig. 2.-Sideoats grama cover for loess kills and tallgrass prairie sites contrasting the site-specific response of this species to burning. Vertical lines represent one Standard Error. B3 = three-year bum schedule, Sp = Spring, Su = Summer, Fa = Fall, UB = Unburned. Tallgrass site bum year = 1983; loess hills burn years = 1986 for Fall treatment and 1987 for Spring and Summer treatments; "-" for Tallgrass Prairie in 1983 indicates "no data".

response between sites is the long history of mowing management prior to the initiation of this study. Such management would have maintained a low canopy thus enabling sideoats grama, a mid-height grass, to persist in an ecosystem otherwise dominated by tall-statured species. With the cessation of mowing at the tallgrass prairie in 1980, canopy cover of the tall-statured component of the ecosystem increased as evidenced, for example, by the 15 percent increase in big bluestem and the 20 percent increase in indiangrass cover in unburned plots (table 1). The decline in mid-height and increase in tall-statured species were amplified by fire's tendency to favor tallgrass species (Ehrenreich 1959; Hadley 1970). At the loess hills site, however, tallgrass species were only a minor component of the ecosystem thus, fire's favoring of tallgrass species did not substantially affect the mid-height grasses such as sideoats grama and little bluestem (table 2). The effect of fire on a species (e.g. sideoats grama), therefore, may not operate directly on that species but rather may operate indirectly by favoring an intermediate species (e.g. indiangrass) that outcompetes the shorter grasses for some limited resource such as light.

Other species, either found only at one site or found in sufficient numbers only at one site, provide yet further insight on long-term management implications for prairies in general. One such insight of particular importance would be any evidence that fire encourages seedling establishment in a prairie ecosystem. Recruitment is one of the most critical facets of long-term prairie management since it ensures a replacement of a species' population, thereby maintaining ecosystem diversity over decades. No studies have been conducted on fire-affected seedling germination and establishment in the loess hills prairie but those in the tallgrass prairie generally show variable results. For example, Glenn-Lewin et al. (1990) found that, in years with adequate precipitation, burning resulted in higher seedling establishment than occurred without burning; one species that showed this effect was Scribncr dichanthelium. In dry years, however, they found that burning reduced seedling establishment. Also noteworthy was that, with adequate precipitation, germination of some species (e.g. Kentucky bluegrass and prairie violet) was particularly high in either the unburned area or in areas burned the previous year.

While the present study did not focus on identification and establishment of seedlings, one might hypothesize that significant increases in canopy cover would be a logical consequence of such fire-initiated establishment. Initially being absent or having low cover (e.g. with seedlings) growth of new plants would be reflected in a significant increases in canopy cover over a few years. Evidence for such an effect of fire was found in the tallgrass prairie with summer burning of false sunflower [Heliopsis helianthoides (L.) Sweet var. scabra (Dun.) Fem.] (fig. 3), summer and fall burning of white aster, and fall burning of leadplant (table 1). In the loess hills, such a response was found for prairie clover (Dalea spp.) with summer and fall burning (fig. 3). Note that all of these seedling-initiation signatures occurred only with summer and fall burning, which are not the normal times for prairie management burning in the tallgrass or loess hills prairie regions. Rohn and Bragg (1989), for example, found that germinability of false sunflower and white prairie clover (Dalea candida Michx. ex Willd) declined with spring burning. These results suggest that, for the long-term maintenance of a diverse community, occasional bums of some portions of an area at times of the year other than spring may be important to ensuring the continuation of a full complement of species.



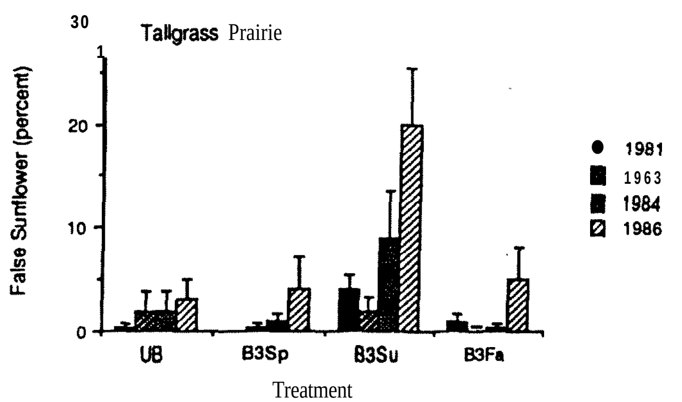


Fig. 3.-Canopy cover of Dalea spp. (prairie clover; loess hills prairie) and false sunflower (tallgrass prairie) showing the type of increase in cover hypothesized to reflect an increase in number of individuals in response to summer burning. Vertical lines represent one Standard Error. B3 = three-year bum schedule, Sp = Spring, Su = Summer, Fa = Fall, UB = Unburned; "-" for Tallgrass Prairie in 1983 indicates "no data". Tallgrass site bum year = 1983; loess hills bum years = 1986 for Fall treatment and 1987 for Spring and Summer treatments.

In addition to possibly encouraging seedling success, fires at different times of the year both in the loess hills and in the tallgrass prairie sites increased the cover of some species. Among the species that showed this response at the tallgrass prairie were white aster, stiffstem flax [Linum rigidum var. compactum (A. Nels.) Rogers] (fig. 4), and black-eyed susan (Rudbeckia hirta L.), all of which increased with summer and fall burning; Stiffstem flax and black-eyed susan declined in cover without burning. At the loess hills site, other species showed a similar response pattern including gray goldenrod (Solidago nemoralis Ait.) with fall burning (fig. 4) and, as at the tallgrass site, stiffstem flax with summer and fall burning. Gray goldenrod, *for* example, declined during the four *years* of the study with all treatments except fall burning. Again, note the apparent importance of summer and fall burning to maximizing the canopy cover these species.

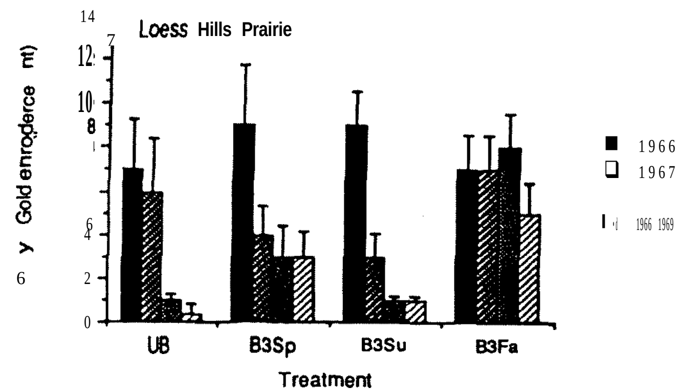
Significant declines with all seasons of *burning* were detected for sideoats grama and rough dropseed [Sporobolus asper (Michx.) Kunth] in the tallgrass prairie and silky aster (Aster sericeus Vent.) in the loess hills prairie. e bluestem declined with summer burning in the loess hills prairie. As has been pointed out above, note the particular role of summer and fall burning in affecting species' responses. Only candle anemone (Anemone cylindrica A. Gray) in the tallgrass prairie site increased significantly without some fire treatment.

Some species showed significant changes irrespective of treatment. For these, it appears that some factor other than fire is important in explaining their response. Species that decreased without regard to treatment included Scribner dichanthelium and porcupine-grass in the tallgrass prairie and white-eyed grass in the loess hills (tables 1 and 2). The one species that increased significantly was flowering spurge (Euphorbia corollata L.).

CONCLUSIONS

When taken in combination, the vegetative responses to fire reported at the tallgrass and loess hills study sites, suggest several considerations. First, the same species may respond differently in different ecosystems. This is a logical conclusion but one that needs to be carefully considered particularly when developing management plans for grasslands even withii the same general geographic area. Second, in order to maintain long-term *(many* decades long) diversity of a grassland ecosystem under relatively static climatic conditions, this study suggests that serious consideration be given to some application of fire management at various times of the year. While further research is clearly needed, data from both the tallgrass and the loess hills grasslands suggest that successful seedling establishment, for example, might require different seasons of burning. Summer and fall burning seem to be times that are most likely to encourage such seedling success for several species. Higher biomass produced by some species with summer and fall bums further suggests the need for a consideration of occasional non-spring bums.

Thus, to maintain vegetative diversity both by seedling establishment and by maximizing species productivity, some areas or portions of areas within a preserve should be burned some time during the growing season. The size of such growing-season bums, however, should not be so extensive as to adversely affect the resident invertebrate population of the area. Such small scale summer bums are probably representative of the natural ecosystem in which sufficient fuel is present to support a fire (Bragg 1982) but where the amount of green matter in the fuel bed would not have supported a high intensity, widespread fire.



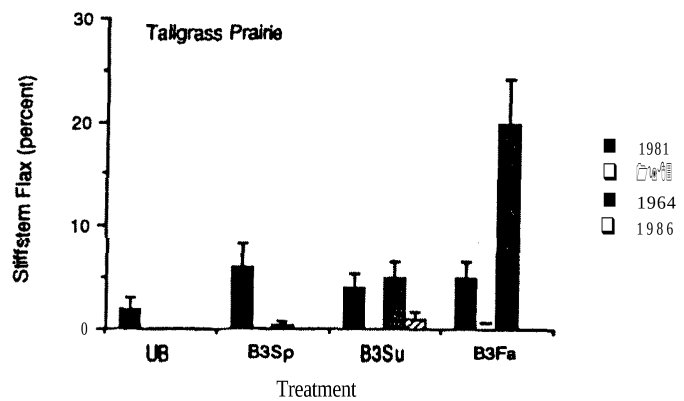


Fig. 4.-Canopy cover of grey goldenrod (loess hills prairie) and stiffstem flax (tallgrass prairie) indicating that their persistence may be dependent on summer or fall bums. Vertical lines represent one Standard Error. B3 = three-year bum schedule, Sp = Spring, Su = Summer, Fa = Fall, UB = Unburned; "-" for Tallgrass Prairie in 1983 indicates "no data". Tallgrass site bum year = 1983; locss hills bum years = 1986 for Fall treatment and 1987 for Spring and Summer treatments.

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FORTY YEARS OF PRESCRIBED BURNING ON THE SANTEE FIRE PLOTS: EFFECTS ON OVERSTORY AND MIDSTORY VEGETATION

Thomas A. Waldrop and F. Thomas Lloyd'

Abstract-Several combinations of season and frequency of burning were applied in Coastal Plain loblolly pine (Pinus taeda L.) stands over a 40-year period. Pine growth was unaffected by treatment. Aboveground portions of small hardwoods (less than 12.5 cm d.b.h.) were killed and replaced by numerous sprouts. With annual summer burning, sprouts were replaced by grasses and forbs. Study results emphasize the resilience of southern forests to low-intensity burning and that frequent burning over a long period is needed to produce significant changes to forest structure and species composition.

INTRODUCTION

It is well established in the literature and in other papers at this symposium that fire has been a major ecological force in the evolution of southern forests. Ecological and meteorological evidence suggest that lightning-caused fires were a major force in creating open pine forests in the Southeast (Komarek 1974). Archeological evidence has established the presence of Paleo-Indians in the region as early as 12,000 years ago (Chapman 1985). The movement of Indian tribes for game and cropland crested variable patterns of fue frequency across the landscape, thus producing a mosaic of vegetation types and stand ages (Buckner 1989). Southeastern forests described by the first white settlers of the 1600's and 1700's were often open pine and hardwood stands with grasses underneath. Early writers suggested these open forests owed their existence to frequent burning (Bartram 1791; Harper 1962; Van Lear and Waldrop 1989). Frequent burning continued through the early 1900's, when fire protection policies of the U.S. Department of Agriculture, Forest Service, and cooperating State Forestry agencies attempted to prevent the use of fue. Prescribed burning for fuel reduction gained acceptance in the 1940's and 1950's, but only after a series of wildfues showed the disastrous consequences of fire exclusion (Pyne 1982). As a result, contemporary forests developed with a dense understory and a larger hardwood component.

It can be difficult to appreciate the important role of fire in shaping the species composition and structure of Southeastern forests. The changes fire causes in plant communities can be slow and depend on fire intensity, the season and frequency of burning, and the number of successive fires used. Opportunities to observe changes in vegetative characteristics over long periods are limited. A long-term study by the Southeastern Forest Experiment Station may give an indication of the ecological role fire once played. The experiment, known as the Santee Fire Plot Study, was established in 1946. Various combinations of season and frequency of burning were maintained for over 40 years. Previous papers have compared the effects of these various fire regimes on pine growth, understory vegetation, and soil properties at specific years during the study. This paper discusses changes to the structure and species composition of the overstory and midstory as they occurred over time and relates those changes to presettlement fire frequency and effects. Changes to understory vegetation after 43 years of burning are presented in another paper in these proceedings (White and others 1991).

DESCRIPTION OF THE STUDY

Study plots are on the Santee Experimental Forest in Berkeley County, SC, and on the Westvaco Woodlands in neighboring Georgetown County. Both areas are on a Pleistocene terrace on the Lower Coastal Plain at 7.5 to 9.0 m above sea level. Soils include a variety of series but are generally described as poorly drained Ultisols of medium to heavy texture (McKee 1982). Soils are considered productive with a site index of 27 to 30 m for loblolly pine at age 50. In 1946, the overstory of both study sites consisted of unmanaged, but well-stocked even-aged stands of loblolly pine. Common midstory species were dogwood (Comus florida L.), hickory (Carya sp.), southern red oak (Quercus falcata Michx.), post oak (Q. stellata Wangenh.), water oak (Q. nigra L.), and willow oak (Q. phellos L.). The Santee stand was 42 years old when the study was initiated, while the Westvaco stand was 36 years old. Both stands resulted from natural regeneration after logging. No evidence of previous burning was observed.

Six treatment plots, 0.1 ha in size, were established in each of five replications. Three replications are on the Santee Experimental Forest and two are on the Westvaco woodlands. Treatments include: (1) periodic winter burning, (2) periodic summer burning, (3) biennial summer burning, (4) annual winter burning, (5) annual summer burning, and (6) an unburned control. All winter burning was done on December 1 or as soon afterward as weather permitted. Summer burning was done on or soon after June 1. Periodic bums were conducted when 25 percent of the understory hardwood stems reached 2.5 cm in diameter at breast height (d.b.h.).

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This prescription resulted in variable burning intervals ranging from 3 to 7 years. Annual burning has not been interrupted since 1946. Biennial summer burning was added to the study in 1951.

To protect the study, burning techniques were selected to ensure low fire intensity. Selection was made at the time of burning based on prevalent fuel and weather conditions. In general, backing fues were used on periodically **burned** plots that had thick underbrush or when hot and dry weather increased the risk of high-intensity fires. Headfires or strip headfires were used on annually burned plots that had little underbrush or when fuels were too moist to support a backing fire.

OVERSTORY PINES

Loblolly pine remained the dominant overstory species in all study plots from 1946 to the present. However, growth rates may have been affected. The Santee Fire Plots were designed to study effects on understory vegetation with little consideration to tree growth. Detailed records of the number and size of trees were not kept throughout the history of the study. Therefore, comparisons of treatment effects on diameter and height growth were conducted through increment core analysis and stem analysis procedures, respectively. **A** more detailed description of these methods was given by Waldrop and others (1987).

Basal area per hectare for each burning treatment throughout the study is shown in figure 1. Since records of tree mortality were not **kept,** figure 1 represents the basal area of only those trees that survived until the time of sampling (1984). Differences in the levels of these **curves** represent differences in numbers and **sizes** of trees in treatment plots in 1984, rather than treatment effects. If burning treatments alter tree growth rates, the effect would be shown as

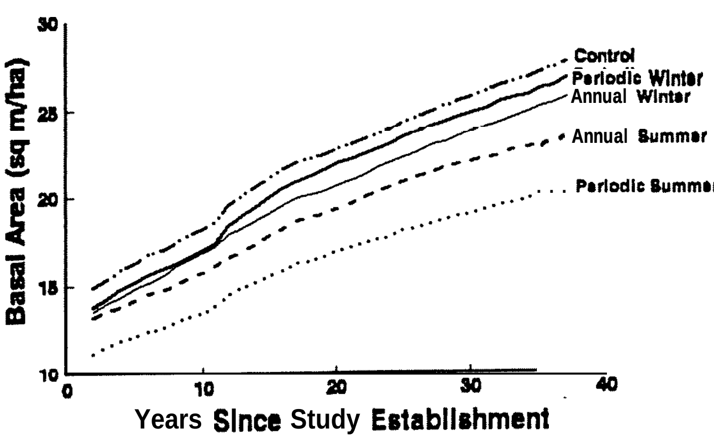


Figure 1 .-Cumulative basal area of trees surviving from 1946 through 1984 by burning treatment.

differences in the slopes of these curves rather than differences in the relative heights. All curves in figure 1 are generally parallel, indicating that burning did not affect diameter growth. Basal area increment during each of four IO-year periods was subjected to analysis of covariance, using measured stand basal area to adjust growth rates for stocking effects. These tests indicated that differences between the slopes of lines were not significant for any period (alpha = 0.05).

Mean tree height for each treatment throughout the lives of these stands is shown in figure 2. **Curves are very close** together, indicating that trees in **various** treatment plots had similar height growth patterns. During the last 30 years, trees in plots burned annually in winter or summer appear to have slightly reduced height growth. These differences were not significant, however, when compared by analysis of variance (alpha = 0.05).

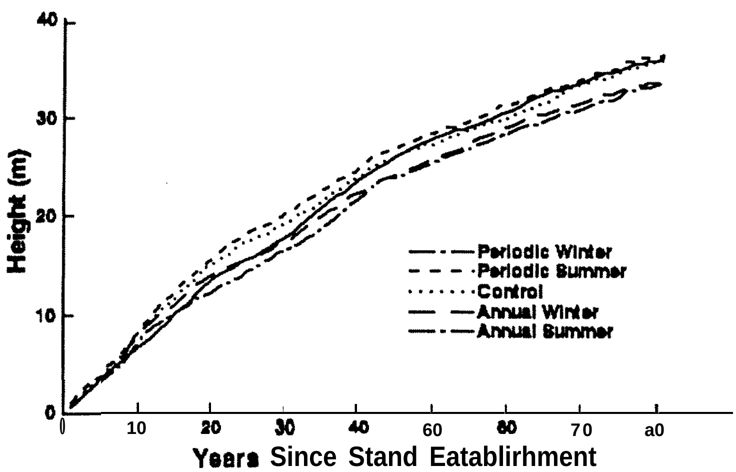


Figure 2.–Mean height of sampled trees by burning treatment from 1905 through 1984.

The lack of differences in diameter and height growth was unexpected. We expected that these low-intensity fires would not **cause** enough crown damage to reduce growth, and that vegetation control and increased soil fertility resulting from prescribed burning would improve growth. However, overstory pines averaged 40 years old at the beginning of the study and were probably too old to respond by the time these site changes reached meaningful levels. Even though McKee (1982) showed increases in phosphorus and calcium availability, no fertilization studies in the Coastal Plain have shown positive responses to these elements in trees of this age. In addition, soil moisture is rarely limiting to pine growth on these poorly drained Coastal Plain sites, even when competing vegetation is not controlled.

MIDSTORY

Diameter Distribution

Species composition of midstory vegetation changed little since study establishment. Dogwood, hickory, and oaks have remained *common on* all treatment plots since 1946. However, repeated measurements of the midstory show that diameter distribution of these hardwoods has been changed by the various combinations of season and frequency of burning. The d.b.h. of all hardwoods in all plots was measured at study establishment (1946), at year 20 (1966), and at year 30 (1976). Later descriptions are unavailable due to severe damage from Hurricane Hugo in September 1989. Stem numbers in each of five diameter classes (< 2.5 cm, 2.6-7.5 *cm,* 7.6-12.5 *cm,* 12.6-17.5 cm, and 17.5+ cm) were used as dependent variables in a split-plot design of an analysis of variance to compare treatment differences over time. Whole-plot effects were those created by burning treatments while the years since study establishment were sub-plot effects. Mean separation was by linear contrast (alpha = 0.05).

At the beginning of the study, unburned control plots appeared to be undisturbed. Every size class of hardwoods from less than 2.5 cm to over 17.5 cm d.b.h. was present (fig. 3A). Diameter distribution followed a reverse-J pattern with numerous stems in small size classes and few stems in larger classes. The number of stems in each size class varied somewhat over time as individual trees grew into larger classes. However, the reverse-J pattern remained.

Hardwood diameter distributions were altered by periodic winter bums and periodic summer bums. For both treatments, the number of stems in the smallest size class (O-2.5 cm) increased significantly between year 0 and year 20 and between year 20 and year 30 (figs. 3B and 3C). Hardwood numbers in the next two classes (2.6-7.5 cm and 7.6-12.5 cm) decreased significantly over the same periods. With periodic summer burning, the smallest size class increased from approximately 11,000 to over 19,000 stems per hectare by year 30. The 2.6- to 7.5-cm size class was most affected, decreasing from over 1,100 to approximately 100 stems per hectare in both periodic treatments. Most changes occurred during the first 20 years, but the changes continued at a reduced rate through year 30.

Hardwoods greater than 12.5 cm d.b.h. were generally unaffected by periodic winter and summer burning (figs. 3B and 3C). At the beginning of the study, these trees were old enough to be protected by thick bark and tall enough that their buds were protected. Most stems less than 12.5 cm d.b.h. were too small to survive burning. However, root systems of these smaller trees survived and produced multiple sprouts, causing the increase in stem numbers in the smallest size class. Bums were frequent enough to prevent the growth of sprouts into a larger size class. Fewer than 10 percent of the trees in the 2.6- to 7.5-cm d.b.h. class survived until year 30. Trees of this intermediate size class are susceptible to top-kill from occasional flareups or hot spots. Since hot spots occur more often during the summer, fewer trees of this size class survived periodic summer bums than periodic winter bums.

Annual winter burning caused changes in the hardwood d.b.h. distribution similar to periodic winter and summer burning. Most stems in the 2.6- to 7.5-cm d.b.h. class were top-killed or girdled during the first few years. Stem numbers in this size class were significantly reduced (from approximately 1,200 per hectare to less than 100) by year 20, with no additional reduction through year 30 (fig. 3D). The number of stems per hectare in the smallest d.b.h. class (O-2.5 cm) increased dramatically over the 30-year period. By year 20, this size class had increased significantly from 16,000 to 21,000 stems per hectare. Between years 20 and 30, that number increased to over 47,000 per hectare. Most of these stems were sprouts less than 1 m tall. Since annual **winter** bums allow sprouts a full growing season to recover from fire, many root systems survived and produced larger numbers of sprouts after each fire. In year 44, White and others (1991) found a slight decrease in the number of stems per hectare in annual winter bum plots and a substantial decrease in cover by woody plants. Even though sprouts are still numerous, these decreases may indicate declining sprout vigor.

Annual summer burning has nearly eliminated woody vegetation in the 0- to 2.5-cm d.b.h. class (fig. 3E). Root systems were probably weakened by burning during the growing season when carbohydrate reserves were low. Burning was frequent enough to kill root systems of all hardwoods less than 7.5 cm d.b.h. during the first 20 *years.* A few hardwood seedlings appeared each spring but did not survive the next fire. As with other treatments, the number of stems between 2.6 and 12.5 cm d.b.h. was significantly reduced by annual summer burning and the majority of the change occurred during the first 20 years. Stem numbers of hardwoods over 12.5 cm d.b.h. were unaffected by annual summer burning.

Root Mortality

Patterns of hardwood rootstock mortality observed during the first few years on the Santee Fii Plots prompted investigators to expand the study. In 1951, biennial *summer* burning was added to provide a comparison with annual summer burning to study root system survival for four hardwood species (Langdon 1981). Individual trees were observed repeatedly to

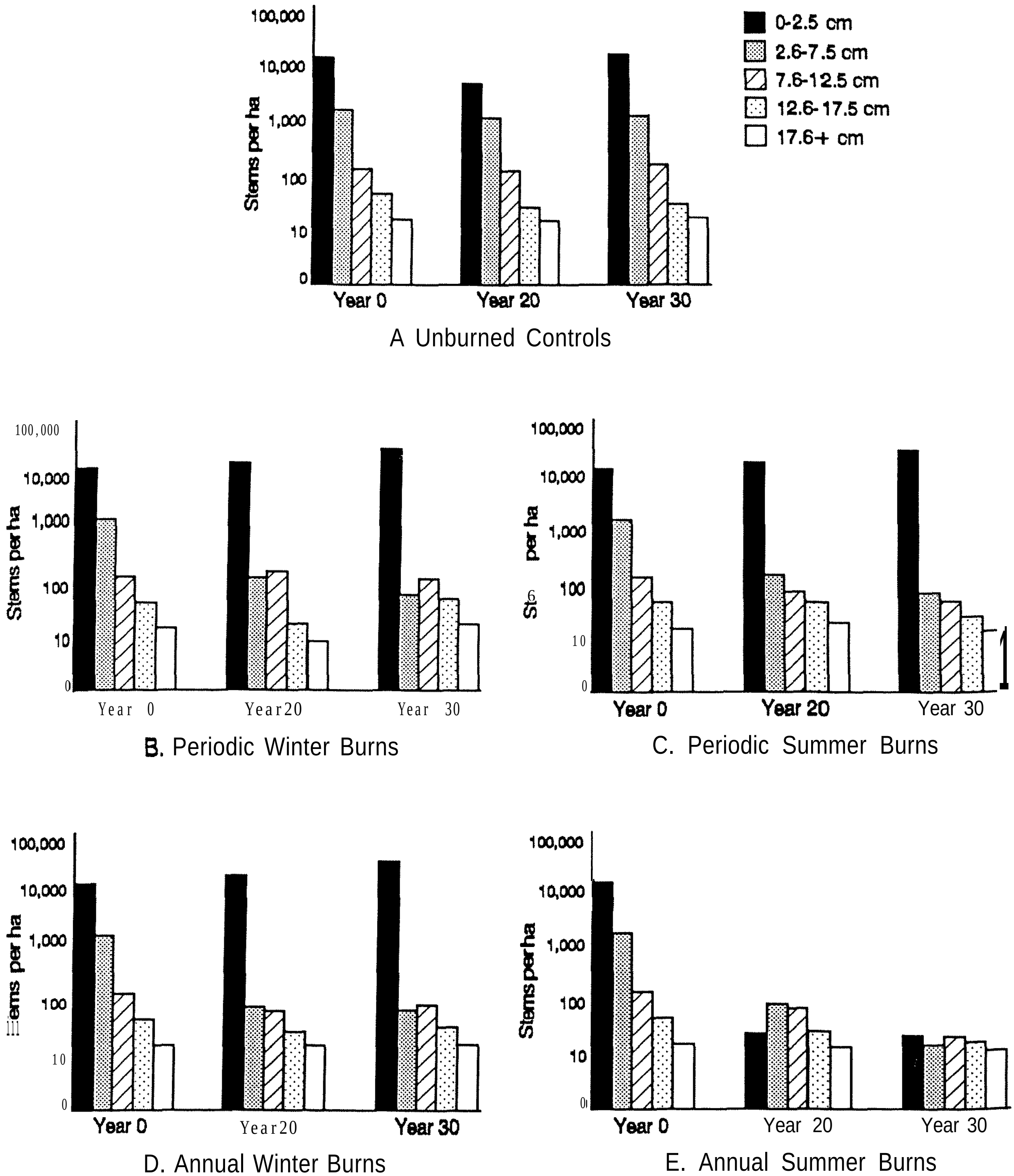
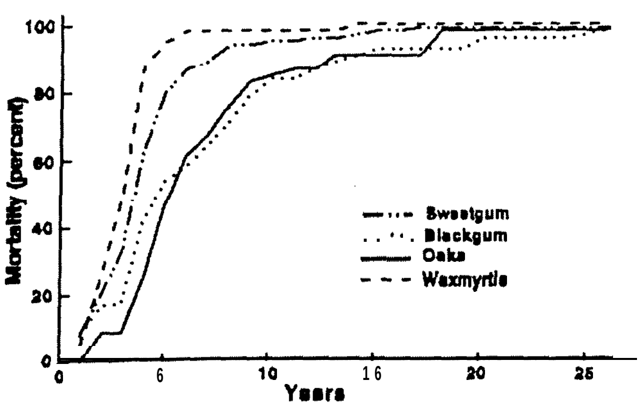


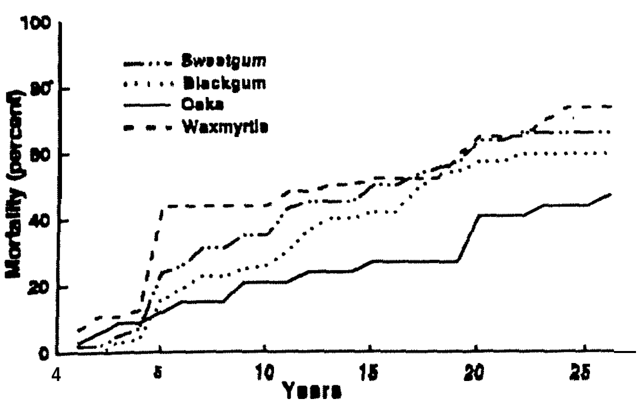
Figure 3.--Diameter distribution of all hardwoods at selected years for (a) unburned control plots, (b) period winter bum plots, (c) periodic summer bum plots, (d) annual winter bum plots, and (e) annual summer bum plots.

determine the number of bums required to kill their root systems. With annual summer burning (fig. 4A), mortality was rapid for sweetgum (Liquidambar styraciflua L.) and waxmyrtle (Myrica cerifera L.), nearing 100 *percent* within 8 years. Oaks and blackgum (Nyssa sylvatica Marsh.) *were* more difficult to kill, requiring approximately 20 years to reach 100 percent mortality. Biennial summer burning (fig. 4B) was less effective in killing root systems of all species tested. After 26 years (13 bums), mortality among the oak species remained less than 50 percent. With biennial burning, root systems have an entire growing season to recover.

Apparently, that time is sufficient for carbohydrate reserves to accumulate enough to allow some resistance to fire.



A. Annual Summer Burn



B. Biennial Summer Burn

Figure 4.-Cumulative mortality of hardwood roots over 26 years of (a) annual summer burning and (b) biennial summer burning (Langdon 1981).

Species Composition

Survival of hardwoods over 12.5 cm d.b.h. was unaffected by burning treatments and, therefore, changes in species composition among larger trees were not observed. The major effect of burning treatments was to kill the aboveground portion of stems smaller than 12.5 cm d.b.h. With most burning treatments, however, root systems survived and sprouted. If burning was stopped or delayed, sprouts would eventually grow into the midstory producing a stand with species similar to unburned controls. Variations among species in plants' abilities to *regenerate* after fire created changes in the species composition of regeneration (fig. *5). In year 44, control* plots were covered mostly by shrubs with some grasses and hardwoods (White and others 1991). Total coverage was increased by periodic winter and summer bums due to increased sprouting of hardwoods and shrubs. Total coverage after annual winter bums was greater than in control plots, but species composition had changed. Burning greatly reduced the shrub component, which was replaced by grasses and forbs. However, numerous hardwood sprouts remained. Annual summer burning was the only treatment which eliminated regeneration of hardwoods. In these plots, the shrubs and hardwoods that were dominant in 1946 were replaced entirely by grasses'and forbs.

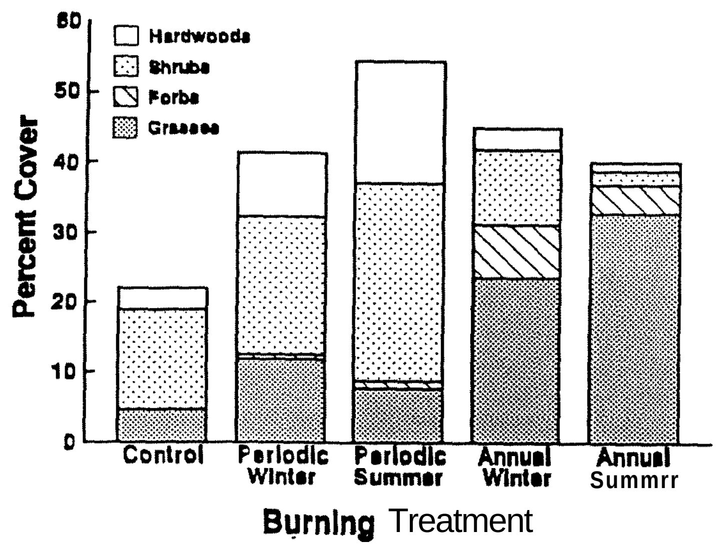


Figure 5.-Percent crown coverage of all understory plants after 44 years of prescribed burning (White and others 1991).

DISCUSSION AND CONCLUSIONS

All tree species on the Santee Fire Plots were well adapted to frequent low-intensity burning. Thick bark and high crowns protected the pines from damage and no growth loss was detected. Hardwoods over 12.5 cm d.b.h. were protected by thick bark and most survived. During the first few years of the study, most hardwoods below 12.5 cm d.b.h. were either top killed or girdled, particularly by summer burning. However, root systems survived and produced multiple sprouts Annual summer burning over a 20-year period was the only treatment that eliminated hardwood sprouts.

The response of tree species to these long-term prescribed burning treatments was considered minimal. Only *one* major trend was observed. Small hardwoods were replaced by large numbers of sprouts during the early years of the study. Later, those sprouts were replaced by grasses and forbs. The gradual change from small hardwoods to grasses and forbs was completed by only the most intensive treatment, annual summer burning. White and others (1991) provide evidence that sprout vigor is decreased by annual winter burning, suggesting that these sprouts may eventually be eliminated. However, a large regeneration Pool of hardwoods still exists after 44 years of treatment. Periodic bums did little to reduce numbers or vigor of hardwood sprouts.

Hardwood sprout survival was affected by the season and frequency of burning (Langdon 1981). Hot summer fires conducted each year when carbohydrate reserves are low produced relatively rapid (20 years) mortality of hardwood rootstocks. Periodic winter, Periodic summer, and annual winter burning allow at least one growing season for sprouts to store carbohydrate reserves in root systems and, therefore, resist mortality. Without annual summer fires, it is questionable whether hardwood sprouts can be eliminated by fue.

This study emphasizes that frequent fires over long periods are needed to create and maintain the open character of pine forests described by early explorers in the Southeast. Periodic burning over 40 years did little to eliminate hardwoods and supported a dense understory shrub layer. Annual winter bums maintain an open understory with vegetation generally leas than 1 m tall. However, that understory includes numerous woody sprouts and a dense hardwood midstory would return if burning was delayed a few years. Of all treatments tested, only annual summer bums produced an open understory with no hardwood regeneration. However, presettlement forests did not support the midstory hardwoods present in study plots. In addition to frequent low-intensity fires, an occasional high-intensity fire or other disturbance would elimiiate large hardwoods.

Although the Santee Fii Plot Study provides information on the frequency and number of fires required to *create* and maintain open pine forests, differences exist between its controlled experimental conditions and the environment of presettlement fires. Annual fires set by Indians were controlled only by weather and geographic barriers. Therefore, fire intensity was probably higher than in the Santee study. Also, large herds of deer (Odocoileus virginianus) browsed the open forests and grasslands. Hotter fires and intense browsing would cause higher mortality rates of hardwood sprouts. The Santee Fire Plots were dominated by loblolly pine, which was much less common than longleaf pine (Pinus palustris Mill.) priomto the 26th century. e loblolly pine seedlings are susceptible to fire, pine regeneration is unlikely to escape the frequent fires on study plots. Seedlings of longleaf pine are resistant to fire during the grass stage. *Prior* to the 20th century, longleaf pine seedlings probably escaped to form the overstory during short gaps in fire frequency or in localized areas where fire intensity was low.

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FORTY YEARS OF PRESCRIBED BURNING ON THE SANTEE FIRE PLOTS: EFFECTS ON UNDERSTORY VEGETATION

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Abstract-The effects of 43 years of repeated prescribed burning on crown cover, species composition, species richness, and diversity in the lower understory strata of the Santee Fire Plots were examined. Five study treatments, installed in 1946, include an unburned control, periodic winter and summer bums, and annual winter and summer bums. Understory cover has not changed in the past 20 years except in the annual winter bum plots where cover of trees ≤ 1.5 m in height declined and grass cover increased. Detrended correspondence analysis indentified four distinct understory plant communitites corresponding to season and frequency of bum. Distribution of understory species across a fire disturbance gradient is discussed in terms of varying plant adaptations to fire. Species richness, when separated into herbaceous and woody species groups, and Shannon's diversity index varied significantly across treatments.

INTRODUCTION

The Santee Fire Plot (SFP) study in the Francis Marion National Forest provides a unique opportunity to examine the response of understory vegetation *to* long-term use of *several* combinations of season and frequency of burning. Several studies have examined the effects of single or repeated prescribed fires on understory vegetation (Abrahamson 1984; Conde and others 1983; Cushwa and others 1966, 1969; DeSelm and others 1974; Fox and Fox 1986; Gilliam and Christensen 1986; Grano 1970; Grelen 1975; Hodgkins 1958; Lemon 1949, 1967), but none of these studies was conducted over a period as long as the period of the SFP study. Prescribed burning in loblolly pine stands on the SFP was initiated in 1946 and continued without interruption until 1989, when the overstory pines were destroyed by Hurricane Hugo.

Previous SFP studies focused on the effect of prescribed fire on understory vegetation (Langdon 1971, 1981; Lewis and Harshbarger 1976; Lotti 1955, 1956; Lotti and others 1960), benefits to wildlife (Lewis and Harshbarger 1976) and soil chemical changes (Wells 1971; McKee 1982). Waldrop and others (1987) summarized the effects of the various treatments on the growth of overstory pines after 40 years. Lewis and Harshbarger (1976) reported the effects of prescribed fire on shrub and herbaceous vegetation in the plots after 20 years. On the basis of information developed by Lewis and Harshbarger (1976), Langdon (1981), Waldrop and others (1987), and Waldrop and Lloyd (1991), the following generalizations can be made regarding the effects of long-term use of prescribed fue *on* understory vegetation in the SFP: (1) the unburned control plots were dominated by several size classes of shrub and hardwood species and contained only small numbers of grasses and virtually no forbs; (2) plots that were burned periodically contained two distinct size classes of understory hardwoods (> 15 cm and < 5 cm d.b.h.) and herbaceous species, most of which were grasses; (3) annual winter and biennial summer bums resulted in large numbers of woody stems < 1 m tall and many grasses and forbs; and (4) annual summer burning virtually eliminated understory woody vegetation, and produced an understory dominated by grasses and forbs.

This paper describes differences among plant communities in the Santee Fire Plots after 43 years of prescribed burning. More specifically, we compare the understory plant communities in the context of plant species composition, species richness, and diversity. We also sought to determine whether there have been any changes *in* understory species composition since year 20 (1967).

METHODS

Site Description

The SFP study was originally designed with three replications on the Santee Experimental Forest in Berkeley County, SC, and two replications on the Westvaco Woodlands in Georgetown, SC. The Westvaco plots were regenerated in 1984 so the present study is confined to the three Santee replications. Study plots are located on the upper terrace of the coastal flatwoods region of the Flatwoods Coastal Plain Province, at an elevation of 9.0 m above sea level (Meyers and others 1986). They contain a variety of soil series, which are generally described as poorly drained Ultisols of medium to heavy texture.

Study Design

The SFP study was initiated in 1946 in 42-year-old naturally regenerated loblolly pine with a we&developed understory of hardwoods (post oak, blackjack oak, southern red oak,

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dogwood, American holly, miscellaneous hickories, sweetgum, and blackgum) and shrubs (bayberry, pepperbush, and gallberry). Initially, five treatments were installed: (1) no-bum control, (2) periodic winter bum, (3) periodic summer bum, (4) annual winter bum and (5) annual summer bum. An additional treatment, biennial summer bum, was installed in 195 1. Because of recent insect-related mortality in some plots of the biennial summer bum, it was not included in this study.

Winter burning was conducted as soon as possible after December 1 of each year when the temperature was 16 °C (60 °F) or higher. Summer burning was conducted after June 1 when the temperature was 32 °C (90 "F) or higher. Burning was conducted only when relative humidity was less than 50 Percent, wind speed was 1 to 7 mi/h and fuel moisture was < 10 percent. Backing fires were used initially; later, head fires (strip and flanking) were used in the annual bum plots. Periodic bums were conducted when 25 percent of the understory stems reached 2.5 cm dbh. The average bum interval for periodic bums was 5 years. More detailed site descriptions can be found in Lotti (1960) and Waldrop and others (1987).

For sampling understory vegetation, a 25- by 25-m sample plot was established within each of the 32- by 32-m treatment plots. Two 25-m line transects were randomly located in each sample plot to determine percent crown cover for the following species groups: grasses, legumes, other herbs, woody vines, shrubs, and trees. The vegetation sampled in this study was the lower understory, which was defined as plants $\leq$ 1.5 m tall or plants having a majority of their crown at or below a height of 1.5 m. Cover was determined along a 25-m line transect by measuring the portion of a crown intersected by the 25-m Line. Where two or more crowns overlapped, the overlapping sections of the lower crown(s) were not included.

Two 0.5- by 2-m subplots were randomly located along each 25-m transect (four subplots per plot) to measure stem density or abundance. All plants were identified to species or genus and the number of plants per species or genus was recorded. In measuring abundance of plants that sprout from roots or rhizomes, no attempt was made to determine whether a clump of stems was associated with just one individual or many. Species not encountered in the four subplots were tallied in two 1- by 25-m subplots, each of which was located adjacent to a 25-m transect. The larger subplots (I- by 25-m) were used primarily to sample relatively uncommon species. Species not encountered in subplots of either size but occurring in a 25- by 25-m sample plot were listed as present but not tallied. The species and density data were used to determine species diversity and richness.

Data Analysis

Analysis of variance was used to test for significant treatment and block effects on species richness and diversity. Mean separation was by Fisher's unprotected LSD test (Statistical Analysis System (SAS) 1987). Species richness is the total number of species in a given area. The Shannon-Weaver index was used as a measure of species diversity and was calculated as:

$$H' = -\sum(p_i \ln p_i)$$

where p_i=proportion of individuals of species i to the total number of individuals of all species (base e logarithms are used here).

Detrended Correspondence Analysis (Gauche 1982; Hill 1979; Hill and Gauche 1980) was used to interpret the variation in vegetation composition among treatments. The technique groups plots or communities based on similarity of species composition and relative abundance. The degree of difference between plots is indicated by standard deviation (S.D.) units. A separation of communities by four S.D. units generally indicates that the two communities have no species in common, while one S.D. unit indicates approximately a 50-percent difference in species composition (Hill 1979; Hill and Gauche 1980).

RESULTS AND DISCUSSION

Changes in Understory Cover Between 1967 and 1989

Lewis and Harshbarger (1976) reported on the status of herbaceous and shrub vegetation after 20 years of prescribed burning on the SFP. We chose to compare percent cover by species group at year 43 with their data to determine whether vegetation changes had occurred since their 1967 study. Only the no-bum, periodic summer, and annual winter treatments were compared, because the interval between burning and sampling was not always the same in both studies.

In the no-bum treatments (fig. la), both shrub and tree cover declined over the 23-year period. Some trees and shrubs formerly in the understory grew into the midstory. Also, midstory hardwoods that were present in 1967 continued to grow, further reducing the amount of light reaching the forest floor.

In the periodic summer bum plots (fig. lb), there were few changes between years 20 and 43. At both times, the understory was dominated by shrubs and trees. A slight increase in total cover (all species) may have been caused by increased sprouting of trees and shrubs (Langdon 1981).

Greater changes were observed in the annual winter bum plots (fig. lc). From year 20 to year 43, tree cover declined and grass cover increased. Little change was observed for the other species groups. Although tree cover declined, the number of hardwood stems (44,700 stems ha-') was similar to

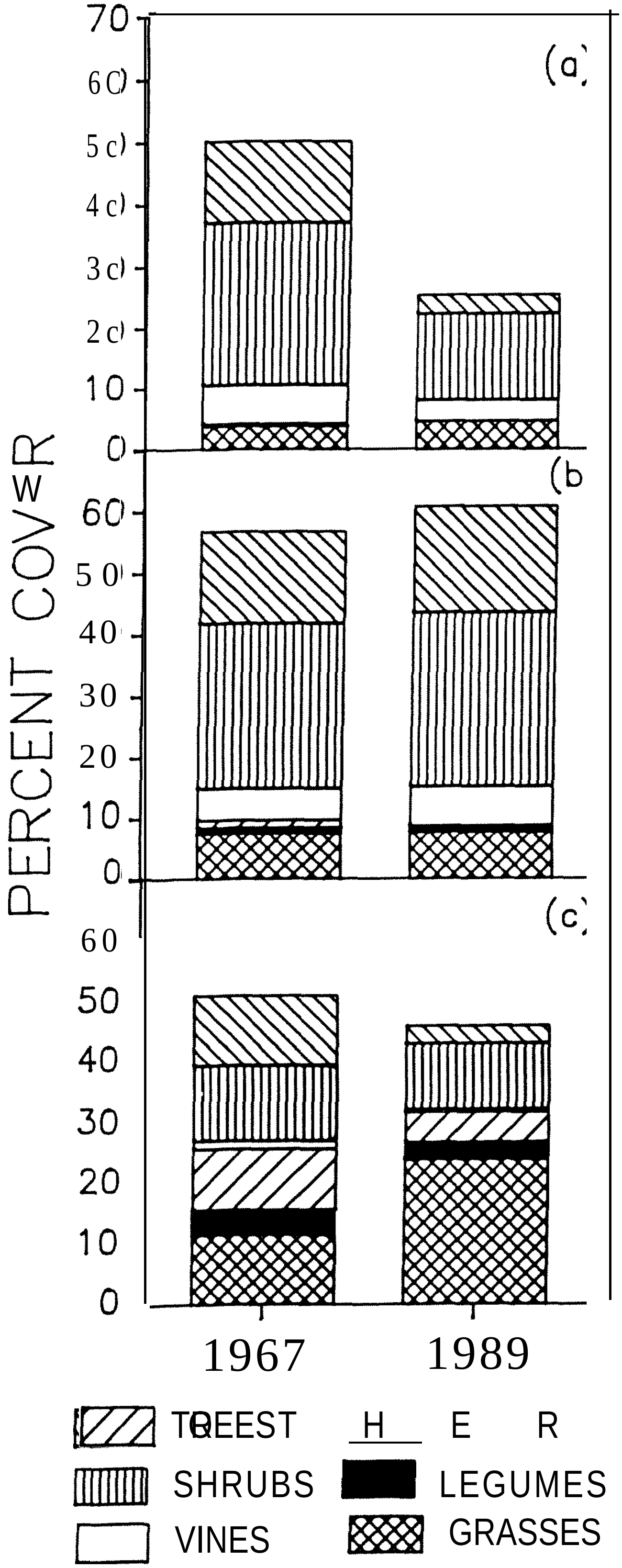

Figure 1--Understory cover by treatment, 1967 and 1989. Treatments: (a) no-bum control, (b) Periodic summer bum, (c) annual winter bum. 1967 data are from Lewis and Harshbarger (1976).

the number reported by Langdon (1981) at year 30 (47,000 stems ha-‘). This pattern suggests that hardwood sprouts are smaller than before and that frequent winter burning may reduce sprout vigor over time. The increased importance of grasses in these plots may be a response to the decline in tree cover or it may have contributed to that decline. While the majority of vegetation changes in annual winter bum plots occurred early in the SFP study, our results indicate that the frequent but relatively mild disturbance associated with this treatment continues to cause changes in vegetation over extended periods of time.

Plant Community Differences

Community Analysis

Detrended correspondence analysis identified four distinct vegetative communities that were associated with season and frequency of burning (fig. 2). Annual summer bums, annual winter bums, Periodic bums, and no-bum controls produced distinctive communities. Differences between treatments were less distinct for the periodically burned plots and the control plots, where woody vegetation predominated. The understory communities produced by periodic winter and summer burning were very similar. The distribution of plots along the X axis leads us to interpret this axis as a tire-mediated disturbance gradient. The relatively large magnitude of difference across treatments (3.5 S.D. units) indicates that beta diversity, or between-community diversity, is high and is affected by season and frequency of burning. Separations along the Y axis are less easily understood, but are interpreted as representing a natural variability gradient. Variability in species composition within a community type decreases as the level of burning increases.

The distribution of species along a fire disturbance gradient reflects the species fire tolerance and competitive vigor. Table 1 is a species synthesis table, as described by Mueller-Dombois and Ellenburg (1974), showing the relative abundance of each species in each treatment plot. This list has been edited to contain only differential species, or those species that demonstrate clear associations for a given treatment or treatments. The 32 species in this table were placed in 5 groups based on their affinity for a given treatment or treatments. Detrended correspondence analysis indicated that the periodic winter and summer burn plots were vegetatively similar and since our sampling of the vegetation took place during the growing season following the burning of the periodic winter plots, only the periodic summer bum treatment is shown in table 1.

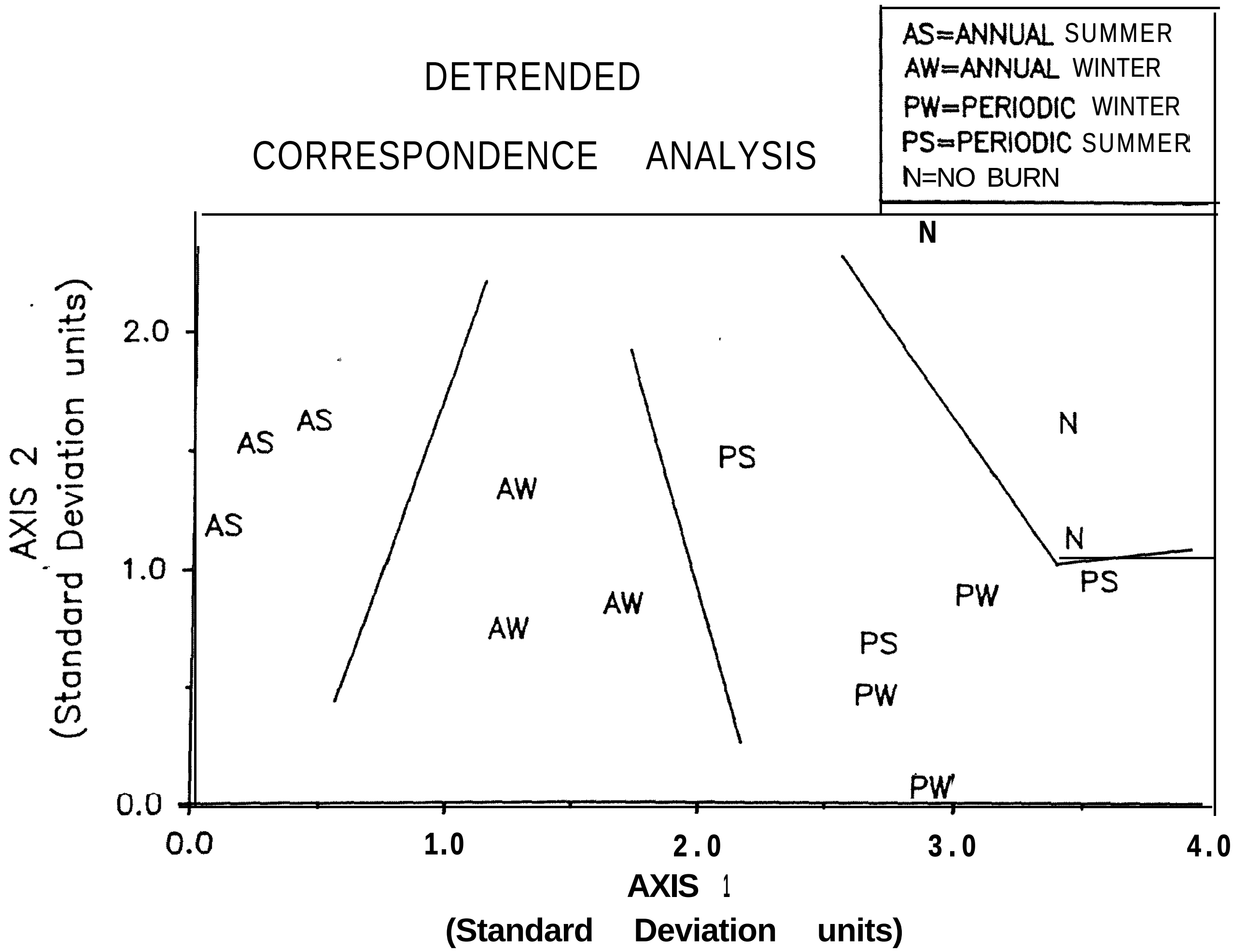

Figure 2-Results of detrended correspondence analysis of all understory plants in all treatment plots. Treatments indicated by the following codes: AS=annual summer bum, AW=annual winter bum, PS=periodic summer bum, PW=periodic winter burn, N=no-burn control. Lines are drawn to show separation between dissimilar groups of plots. See text for explanation of axes.

With few exceptions, groups 1, 2, and 3 are herbaceous plants that have been described as "fire followers" (Lemon 1949, 1967). Many of these plants are also associated with early successional plant communities following non-fire disturbance. Other species, such as the legumes, are known to benefit directly from the effects of fire (Cushwa and others 1969; Martin and Cushwa 1966; Martin and others 1975). The species in group 1 are found almost entirely in the annual summer bum plots. These are generally opportunistic species that lack the competitive vigor to become established in other burned plots, where more vigorous grasses and woody plants predominate. Species in group 2 are most common in the annual winter bum plots, but some of the legume species are also common in the periodic summer or annual summer bum plots. Generally, group 2 species are less tolerant of annual summer burning and do not compete well with the hardier woody vegetation characteristic of the periodically burned plots. The relatively low abundance of legumes in plots that have been burned every summer may result from the lack of full growing seasons in which to partition photosynthate into perennial rootstocks. Plants in group 3 were common in all burned plots but absent in the no-bum control plots, indicating a dependence on frequent disturbance. Four composite species, two grasses (Panicum species and Andropogon virginicus), and three woody plants (Hypericum species, Rubus species, and Rhus coooaliia) comprised this group. Most species in group 3 disperse their seed broadly and compete vigorously for resources and this enables them to become established quickly after fire.

Groups 4 and 5 (table 1) contain all woody plants with the exception of one grass (Uniola laxa) and one perennial (Mitchella repens). Most of the species in this group reproduce vegetatively - but with varying degrees of vigor, as is indicated by the absence of some species from either the annually or periodically burned plots. Group 4 species are relatively abundant in all but the annual summer plots, maintaining their abundance primarily through vegetative reproduction. About half of these species occurred rarely or infrequently in the annual summer plots; however, their occurence in the annual summer plots is probably due to germination from seed that was transported to the plot by

Table l--Species synthesis table showing relative abundance" of each species across treatments (three plots per treatment)

Species[b]	Group	Treatment: Unburned control			Periodic summer			Annual uinter			Annual summer		
Paspalum species	1										9	R	
Polygala lutea											9	R	3
Hypoxis micrantha										R	1	9	7
Rhexia species									R	+	9	R	R
Coreopsis major	2							+	9	1	+		
Cassia nictitans								5		5	9		
Stvlosanthes biflora								4	1	8	4	9	
Galactia macreei								9		1			
Desmodium species							2	+		9	+	R	
Tephrosia hispidula					R		R		R	9	1		
Centrosema viroinianun					R		R	9		9			
Lespedeza species						+	+	+	+	9			
Lobelia nuttallii	3				+				9	R	+	6	4
Aster species					1			1	9	2	1	+	+
Solidaso species					+		R	9	+	4			
											2	1	+
Elaphantopus species					+	+	+	9	+		1	3	+
Panicun species					3	+	1	6	9	3	6	2	3
Andropogon virginicus					+		+		6	9	5	9	3
Hypericun species					+		8		8		+	2	1
Rubus species				1	4	5	5	+	1	5	+	R	
Rhus coppalina			+		+		R	2	R	1	1	R	R
Pinus taeda	4		+	1	8	+	+	R	+	+	7	9	6
Gavlussacia species		+	+	+	+	3	1	+	1	4	R		+
Vaccinium species			1	+	1	1	6	+	3	9	+		+
Uniola laxa		+	+	+	3	+	+	1	R	5			+
Myrica cerifera		1	+	+	9	2	1	4	1		+		
Liquidambar stvraciflua		+	+	+	5	1	+			9			
Smilax species		9	1	1	2	+	R	1	+				
Vitis species		+	1	+	1	1	+	+		+	R		
Quercus species		+	+	+	6	4	3	+		9			+
Gelsemium sempervirens		+	+	+	7	9	1				+		+
Cornus florida	5		3	2	7	8	9				R		
Mitchella repens			4	+	3	9	2						
Persea borbonia		9	+	+			R						
Lvonia lucida			+	9									

[a] Relative abundance indicated as deciles:"+"=1-10 percent of the maximum abundance value for a given species,"1"=11-20 percent; etc. "R" indicates that a species was rare in the vegetation plot (i.e., was present only).
[b] Nomenclature follows Radford and others (1968).

wind or animals. Species in group 5 were relatively intolerant of frequent burning. Comus florida (dogwood) and Mitchella repens (partridge berry) were absent from annual bum plots, while Persea borbonia (redbay) and Lvonia lucida (fetterbush) were absent from both periodic and annual plots. Fetterbush has been previously mentioned as one of several shrubs on the SFP that sprout prolifically after fire (Langdon 1981). Data from other studies (Cypert 1973; Abrahamson 1984) also suggest that this species is tolerant of fire. The absence of this species in year 43 may indicate that the species is intolerant of long-term frequent burning, at least on sites similar to those in the SFP study area.

Species Abundance

Understory species abundance (number of plants 0.1 ha^{-1}) for woody plants is shown in figure 3. Abundance of hardwoods, shrubs, and vines was dramatically reduced by annual summer burning. In the periodic bum plots and the annual winter bum plots, understory hardwood abundance was slightly greater than in unburned controls. Only the annual summer bum plots had lower shrub abundance than the control plots. The large values for shrubs are attributable primarily to the rhizomatous shrubs, Gavlusaccia spp. (Huckleberry) and Vaccinium spp. (blueberry), which sprout prolifically after tire. The greater abundance of all three woody plant groups in periodic winter bum plots was due to the fact that these plots had been burned the winter prior to sampling, which illustrates the immediate response to fire by this predominantly woody understory.

Abundance of grasses, legumes, and other forbs is shown in figure 4. Herbaceous plant abundance increased with increasing fire frequency, and abundance of all three groups was greatest in the annual winter bum plots. The annual winter treatment yielded a substantially higher number of legume stems than all the other treatments. Legume abundance in the annual winter bum plots was higher than values reported from other studies in the South (Buckner and Landers 1979; Cushwa and Jones 1969; Cushwa and others 1970, 1971; Hendricks 1989; Speake and others 1975). Legume abundance in the periodic and the annual summer bum plots was in the range found in the studies cited above, most of which were conducted after single or periodic bums.

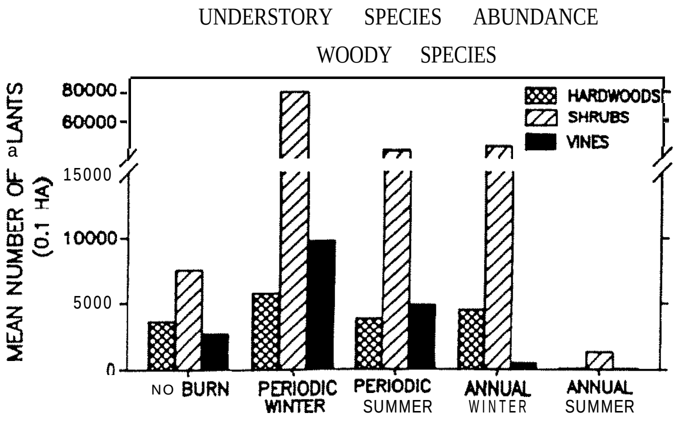

Figure 3--Mean number of stems 0.1 ha^{-1} for understory woody plant groups across all treatments. Note axis scale change.

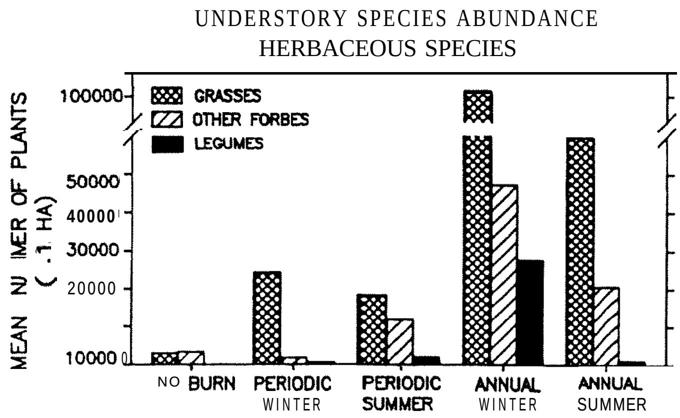

Figure 4--Mean number of stems per 0.1 ha-' for understory herbaceous plant groups across all treatments. Note axis scale change.

Species Richness and Diversity

Understory species richness was not significantly affected by treatment. When species richness was separated into woody and herbaceous categories, treatment effects were significant (fig. 5). Woody species richness was significantly higher for the no-bum and periodic bum treatments than for either of the annual bum treatments. In contrast, herbaceous species richness increased with increasing burning frequency and was significantly higher for the annual winter bum treatment than for the periodic winter and the no-bum treatments.

Shannon diversity, calculated using all understory species, was significantly affected by treatment (table 2). Understory species diversity was significantly higher for the annual winter bum treatment than for the annual summer and pcriodic winter bum treatment but not higher than for the periodic summer and no-bum treatments. It is significant that differences in richness and diversity among treatments were not more distinct. As burning frequency increased, herbaceous species importance increased and there was an associated decline of woody species. This species replacement resulted in relatively small differences in diversity and richness between most treatments. Annual winter burning resulted in higher richness and diversity values because woody biomass was reduced to a level sufficient to allow establishment of herbaceous plants, many of which responded positively to the conditions created by fire.

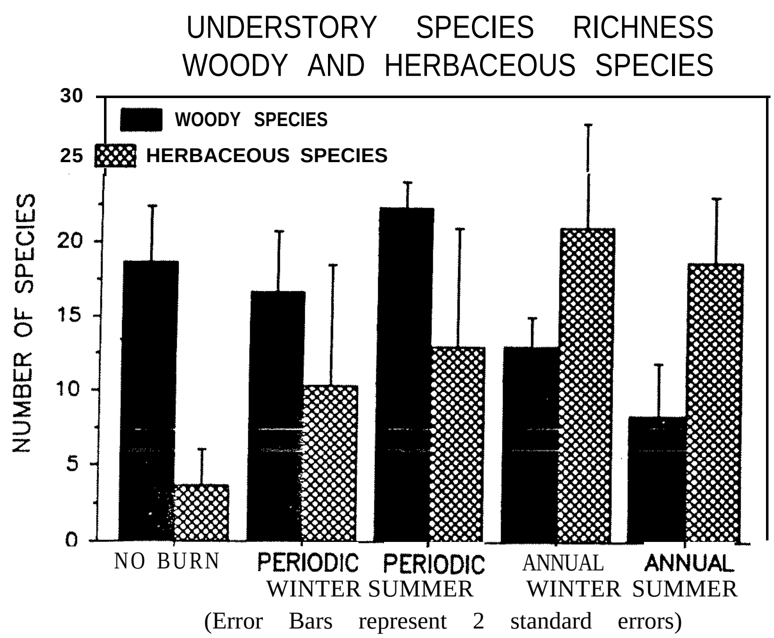

Figure 5--Understory species richness. Error bars represent two standard errors.

Table Z--Shannon diversity indices for understory plant communities from each treatment

Treatment	Diversity Index"
Annual uinter burn	2.40 a
Periodic summer burn	2.28 ab
No burn control	2.07 abc
Annual summer burn	1.88 bc
Periodic winter burn	1.70 c

[a] Means with different letters are significantly different at the 0.05 level.

CONCLUSIONS

While all plants in this southern pine ecosystem are well adapted to fire, it is the fire regime--incorporating intensity, frequency, and season--rather than fire itself, to which plant species are adapted (Gill 1975). Observed differences in species composition of understory plant communities along a fire disturbance gradient were explained by reference to differences in fire tolerance and competitive vigor. Differences in frequency and season of fire produced four distinct plant communities which, when viewed as communities distributed over the landscape, resulted in relatively high beta diversity.

Land managers are faced with increasingly complex problems as the concept of multiple resource management expands to include compositional, structural, and functional biodiversity. Our increased understanding of the "natural" or historical role of fire in shaping forested ecosystems should enable us to better incorporate the use of fue in the management of whole landscapes to accomplish multiple resource objectives.

ACKNOWLEDGMENTS

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FOREST DEVELOPMENT FOLLOWING DISTURBANCES BY FIRE AND BY TIMBER CUTTING FOR CHARCOAL PRODUCTION

Wayne K. Clatterbuck[1]

Abstract-Stand reconstruction techniques and historical documentation were used to analyze present species composition, stand structure, and successional trends on forest lands on the Western Highland Rim of Tennessee. These lands were affected by fire and cutting practices during the late 1800's, when local wood was burned to make charcoal fuel for use at a nearby iron forge. The present two-aged stand structure indicates that there was a discriminatory cutting pattern in which white oak (*Quercus alba* L.) and hickories (*Carya* spp.) were selectively harvested for charcoal. Trees of other species, whatever their size and location, were often left to form the residual stand. Iron forgers apparently favored the hotter burning charcoal of white oak and hickories for producing wrought iron. These results are in contrast to those from other areas, where all tree species were cut and burned to provide charcoal fuel wood for the production of crude pig iron in iron furnaces.

INTRODUCTION

The iron industry flourished on the Western Highland Rim in middle Tennessee during the 19th century. Historical documentation of this industry has focused on the iron-making process, community and social development, and biographies of leading men associated with the industry. Little information is available about the production of charcoal, which was the fuel used to smelt the iron ore and forge wrought iron. Vast timber reserves were necessary to produce sufficient quantities of charcoal as one ton of charcoal was required for each ton of iron produced (Baker 1985). Luther (1977) states that

> "an early chronicler of the industry estimated that to keep a furnace with a 12-ton-per-day iron production going for a year required the cutting of 500 acres of forest, and that to keep one going permanently . . . would require about 16,000 acres (25 square miles) per furnace, allowing 30 years for timber to grow back before the next cutting. In the year 1873 there were 11 furnaces in blast on the Rim, producing iron at the rate of about 50,000 tons per year. In order for all of these furnaces to operate on a 'permanent' basis, then, something on the order of 375 square miles of timber would have been necessary to support them."

Thus, large units of forest land were affected by the charcoal activity. This paper reports on the present species composition, stand structure, and successional trends on forest lands that were affected by (1) cutting during the 1800's for the production of charcoal, (2) fire and grazing during and after the charcoal activities until 1938, and (3) stable State ownership, management, and protection from 1938 to present.

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STUDY AREA

The study was conducted on the 19,887-acre Cheatham Wildlife Management Area (CWMA), which is located 25 miles west of Nashville, TN (36° 12'N, 87° 5'W), on the Western Highland Rim Physiographic Region (Fenneman 1938).

Braun (1950) describes the Western Highland Rim as part of the Western Mesophytic Forest, a transition area between the Mixed Mesophytic Forest Region of the mountains to the east and the Oak-Hickory Forest Region to the west. CWMA is located on the strongly dissected, mature plateau of the Western Highland Rim and consists of narrow to broad ridges, steep dissected side slopes, and V-shaped upland stream valleys (Smalley 1980). Elevations range from 480 to 820 feet. The climate is classified as humid mesothermal (Thomthwaite 1948). Mean annual precipitation is 50 inches and is fairly well distributed throughout the year with slight deficits in late summer and early fall and surpluses during the winter months. Average daily temperature is 15° C with mean temperatures of 2° C in January and 26° C in July (Smalley 1980).

The CWMA was logged and burned in the 19th century during the production of charcoal for a nearby iron forge. Trees were harvested on the ridges and ridge margins where timber was abundant and where the charcoal could easily be transported by wagon downhill to the forge. Most of the land in the broader valleys had been cleared previously for agriculture. Many of the "charcoal hearths" or "fire circles" where the charcoal was produced are still evident on the study area. For more details on charcoal making and iron production processes, see Smith and others (1988) and Ash (1986).

During and after the decline of the iron industry in the 1880's, these cutover areas were periodically, if not annually, burned to promote production of forage for livestock and to retard the advance of woody undergrowth. Fires were also used locally to control snakes and ticks and to expose mast for grazing livestock. Open-range laws were also in effect at that time. Scattered logging for firewood, local building materials, railroad ties, and to clear forest land for agriculture continued, primarily near homesites. The loss of American chestnut (Castanea dentata (Marsh.) Borkh.) to chestnut blight (Endothia parasitica (Murr.) P.J. & H.W. And.) also shaped the composition and structure of the present forest.

In 1938, the State of Tennessee acquired the land for a wildlife management area. The area supports a heavy deer (Odocoileus virginianus L.) population which is intensively managed through controlled hunts. CWMA has been protected from fire and livestock grazing since 1940. Apart from hunting and occasional scattered timber harvesting (mainly in the last 10 *years),* there has been little disturbance by man. Currently, dieback and mortality associated with "oak decline" are present in varying degrees.

Vegetation on two adjacent narrow ridges (A and B) was studied. Ridge A and Ridge B contained 6 and 5 charcoal hearths, respectively. The hearths were located about 200 yards apart along the ridges. The ridges were two miles from the Narrows of the Harpeth River, a historical landmark and the site of the iron forge.

Soils on these ridges and ridge margins are either of the Bodine or Mountview series (North 1981). Both soils are Typic Paleudults and are deep, well-drained, and fine textured. Mountview soils developed in 2 to 3 feet of loess over limestone residuum, while Bodine soils developed in limestone residuum without the loess. Chert fragments are frequent on the surface and throughout the soil mass. Site index (base age 50) for upland oaks is 65 to 70 feet (Schnur 1937). The primary tree species on these ridges are white oak, black oak (Q. velutina Lam.), hickories, flowering dogwood (Comus florida L.), sourwood (Oxvdendrum arboreum L. DC.), and sassafras (Sassafras albidum (Nutt.) Nees.). Other associated species are northern red oak (Q. rubra L.), yellow-poplar (Liriodendron tulipifera L.), white ash (Fraxinus americana L.), scarlet oak (Q. coccinea Muenchh.), and blackgum (Nyssa sylvatica Marsh.).

PROCEDURES

In 1986, a total of 12 one-fifth-acre circular plots were used to sample existing vegetation. Six plots were established on each ridge. Three plots on each ridge had their outer boundary adjacent to a charcoal hearth, while the other three were located in relatively undisturbed areas not affected by charcoal cutting. On each plot, the following data were recorded for each tree with diameters at breast height (4.5 feet) of 5.5 inches and greater: species, azimuth and distance from plot center, I-inch diameter class, and crown class (dominant, codominant, intermediate, or suppressed). Trees with diameters from 1.6 to 5.4 inches were tallied by l-inch diameter class and species. Total heights were measured, and increment cores at DBH were taken from at least three overstory trees on each plot for site productivity assessments, diameter growth profiles, and to determine total age. Point basal area was estimated from plot center with a lo-factor prism. Data from each of the two ridges were pooled by disturbance class (cut for charcoal or not) because the ridges were similar in disturbance history and site quality.

Stem analysis to reconstruct height and diameter growth patterns, to reference fire scars, and to determine age structure was conducted on 12 trees from two plots, one plot from each ridge. Each tree was sectioned at 0.5 feet above the ground and at 4-foot intervals along the bole to the tallest centrally located growing tip. The number of annual rings in each section was subtracted from the tree's total age to determine how old the tree was when its terminal leader was at or near the height of each section. Heights were plotted over corresponding ages to illustrate the height growth pattern of each tree. Diameter growth at 4.5 feet was determined by measuring the annual increment along four Perpendicular radii. Height and diameter data were analyzed using accepted stand reconstruction and graphical procedures (Oliver 1982, Clatterbuck and Hodges 1988). Only height and diameter relationships of individual trees are presented in this paper because the small sample size prevents making statistically testable generalizations.

Historical documentation was used as much as possible to reference forest development. Local newspapers, magazines, and books were searched for relevant information about early iron and charcoal production as were county survey records. The earliest aerial photographs of the study area, which were taken in 1938, were obtained. Local residents were interviewed concerning their recollections of land use events.

RESULTS AND DISCUSSION

Plot Data

Data from the study plots indicated that areas cut for charcoal and the uncut areas had different age structures (table 1). The dominant and codominant trees in the uncut areas were even-aged and averaged 125 years old. White oak, black oak, hickories, and occasional yellow-poplar and blackgum composed the overstory, while dogwood and sourwood made up the midstory. These areas were in the understory reinitiation stage (Oliver 1981): the dominant overstory trees were beginning to decline, allowing a more favorable understory environment for herbaceous and woody vegetation, especially advanced reproduction of tree species.

Table 1.- Stand parameters, based on trees greater than 5.5 inches in diameter, from six sample plots in areas cut for charcoal and from six sample plots in uncut areas.

	DBH (inches)		Age (years)		Density (stems/acre)	
	mean	range	mean	range	mean	range
			AREAS CUT FOR CHARCOAL			
Black oak	22	12-29	130	106-139	24	14-32
White oak	12	6-16	60	48-75	42	30-60
Hickories	9	6-14	58	52-68	10	4-19
Other species"	10	6-21	75	40-135	11	6-24
			UNCUT AREAS			
Black oak	20	a-27	130	110-136	18	a-32
White oak	19					
Hickories	13	6-31 6-22	127 91	115-132 63-125	26 a	17-50 3-16
Other species"	10	6-24	115	55-130	11	6-17

"Includes yellow-poplar, blackgum, scarlet oak, northern red oak, white ash, flowering doguood, and sourwood.

In contrast, the areas cut for charcoal were two-aged, with 60- and 130-year age classes (table 1). More surprising was the species segregation in these stands. One might hypothesize that any tree species that was easy to cut and transport would have been used to make charcoal. However, the charcoal producers were discriminating enough to cut only white oak and hickories, presumably because they judged that these species made the best and hottest burning charcoal for forging iron. Black oak and other species were not cut. Black oaks adjacent to the charcoal hearths have diameters of 20 to 28 inches and many possess fire scars caused by the charcoal activities and subsequent burning for grazing. Many suppressed black oaks were released by the charcoal cutting contributing to their poor, open-grown form. White oak and hickories near the charcoal hearths are 60 to 70 years old. They originated from sprouts or seeds after cutting and burning ceased, and are in the large pole to small sawtimber size classes. Although two-aged, the cut areas were also in the understory reinitiation stage.

The charcoal hearths, with their circular shape, black soil, and absence of overstory vegetation, were conspicuous on the landscape. The soils in these hearths had lost their structure and were nearly sterile as a result of the intense heat associated with charcoal production. The only tree species to colonize these areas were sassafras and dogwood. These trees averaged 32 feet in height, 3 inches in diameter, and 55 years of age.

Stem Analysis

Cumulative height and diameter growth patterns were reconstructed using stem analysis information. Data from trees on the uncut areas are not presented here because these areas exhibited structure and growth patterns typically associated with even-aged stands (Smith 1962). Figure 1 shows the height and diameter growth for the following representative trees on a plot in an area where trees were cut for charcoal: (1) a 132-year-old black oak located 20 feet from the edge of the charcoal hearth, (2) a 132-year-old white oak located 75 feet from the hearth, (3) a 55-year-old sassafras present in the hearth, and (4) a .54-year-old white oak located 30 feet from the hearth. The 132-year-old oaks are residuals left from the charcoal cutting. The older white oak probably was not cut for charcoal because of its distance from the hearth. The younger white oak and sassafras regenerated once the burning and grazing ceased in the 1930's.

Stem analysis supplemented and corroborated the plot data for the area that was cut for charcoal. In 1885, the present 132-year-old oaks were 31 years old, 3 to 4 inches in diameter, and 32 to 40 feet tall (fig. la). By most accounts, charcoal production had stopped by that time (Smith and others 1988). These stems grew slowly and were probably suppressed resulting in spindly form and flat-topped crowns. These trees were probably released when the overstory was cut for charcoal. However, little increase in height occurred between 1885 and 1935 for two possible reasons: (1) the annual burning of the area to enhance grazing and (2) the time necessary for suppressed trees to respond to release. The combination of these two factors is hypothesized to have hindered trees from increasing their crown volumes and altering their crown shapes enough to allow substantial increase in height over this 50-year period.

However, with a slow buildup in crown volume, substantial increases in total height began to occur in the 1930's (fig. la) when these stands were protected from fire and open-range grazing was prohibited. Total height increased 35 percent for each of the older oaks for the 50 years from 1935 to 1985. For the previous SO-year period (1885 to 1935) when fue and grazing was common, total height only increased 18 to 20 percent for both trees.

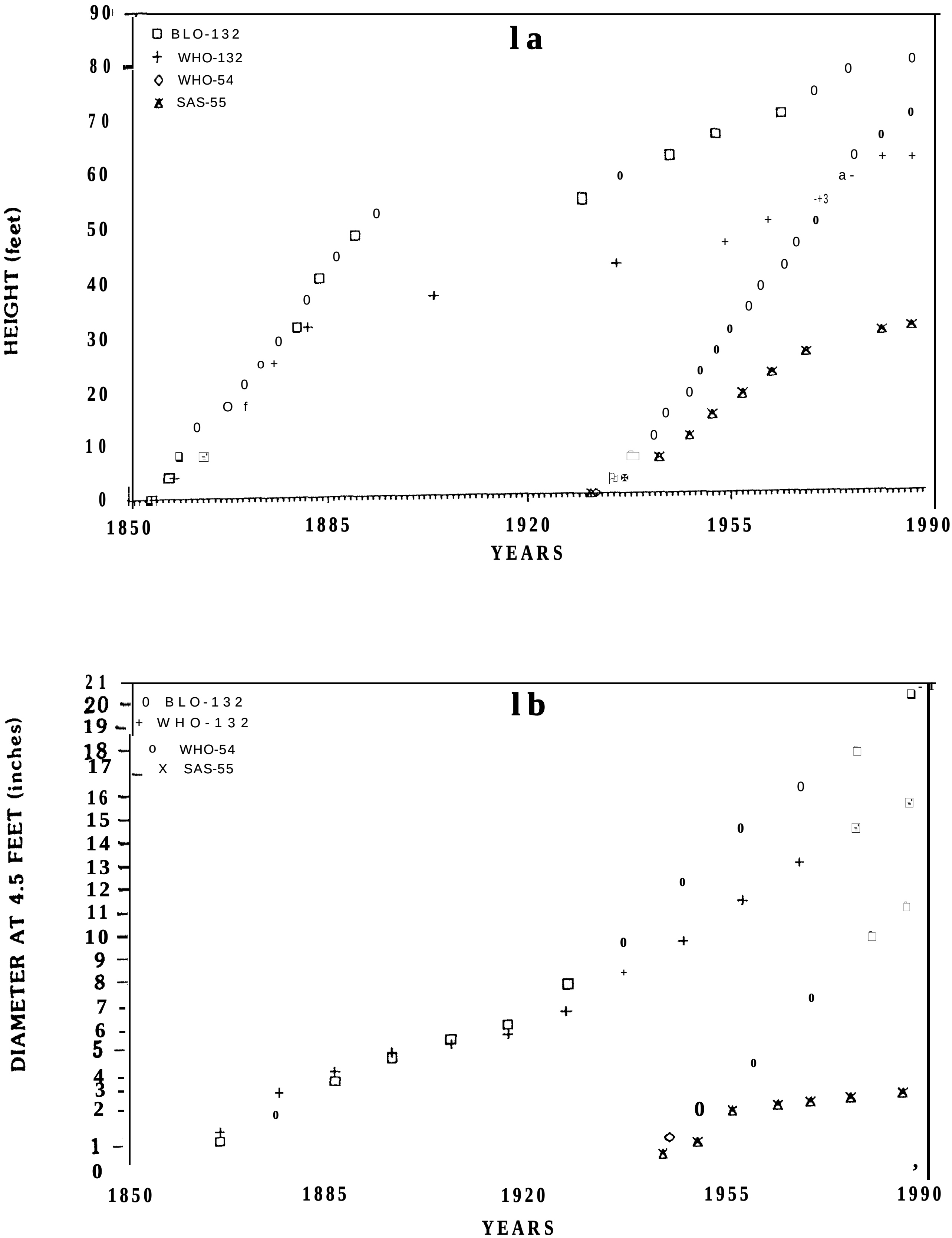

Figure 1.-Cumulative height (la) and diameter (lb) growth patterns of a 132-year-old white oak (WHO-132), a 132-year-old black oak, a 54-year-old white oak (WHO-54), and a %-year-old sassafras (SAS-55) from an area that was cut for charcoal.

Height growth of the 54-year-old white oak was almost linear, increasing in height at an average of 1.3 feet per year. This oak regenerated in a cutover open area that had been burned and grazed for many years. Older trees were not close enough to affect its growth, form, and development. The sassafras was 33 feet tall at age 55; its slow growth is indicative of low soil productivity in the charcoal hearth.

The diameter growth patterns of these stem-analyzed trees are similar to the height growth patterns (fig. lb). Both the older white oak and black oak had diameters of 4 inches in the 1880's when they were released from overstory vegetation, 8 and 10 inches, respectively in 1940 following the grazing and burning, and presently are 16 and 21 inches, respectively. The 54-year-old white oak with its uninhibited growth had a steady diameter growth rate of 2.1 inches per decade. The sassafras in the charcoal hearth was only 3.5 inches in diameter at 55 years of age.

Fire scars were numerous in the basal cross-sections of the older oaks. For both trees, the most severe scarring occurred in 1872, 1894, and 1922. These major fire occurrences probably were localized because they were not referenced in the local literature. However, the abundance of fue scars on these older oaks suggests that burning the forest was common practice in this area prior to 1940.

Historical Documentation

The 1938 aerial photographs of the study area showed a series of patchliie 1- to 3-acre openings along the ridges. Although the charcoal hearths could not be distinguished on the black and white photographs, the centers of the openings were devoid of trees. Isolated individual trees were scattered in these openings; the majority of these were black oaks that were not cut for charcoal and that survived the numerous ground fires.

Long-time local residents verified that fire had been used on a annual basis to "green-up" the herbaceous vegetation for grazing and to control encroachment of woody vegetation into open areas. Both cattle and hogs roamed freely and grazed in the forest until fence laws were passed and enforced. It has been verified that cattle were transported by rail from Texas to the study area during the Dust Bowl years of the 1930's.

American chestnut was a component of these forests and was valued not only for local uses such as firewood, building materials, and mast, but also as a cash product. A manufacturing plant near Nashville, TN purchased chestnut wood in quantity and extracted tannin from it. The tannin was then used to fix coloring in dyes, wine, and beer and to produce an astringent drug. Chestnut logs, whether green, affected with blight, or dead and lying on the forest floor, were used by this industry. The decayed remains of American chestnut logs that are evident in other areas of Tennessee are not present on or in the immediate vicinity of CWMA. The frequent use of fire, chestnut blight, and this specialized industrial use of chestnut logs all influenced stand development in the study area before 1940. American chestnut may have also been used to produce charcoal, but that could not be determined from this study.

Implications

Forest development following charcoal cutting in the study area was unlike forest development in other places where charcoal has been produced and used to fuel iron furnaces. The iron forgers who operated at the Narrows of the Harpeth River used only those species that they judged would produce the best and hottest burning charcoal, primarily white oak and hickories. Other species were intentionally left, and this cutting pattern eventually created a two-aged forest. This unique two-aged species segregation does not occur in areas where all trees, regardless of species, were cut and burned to produce charcoal for iron furnaces. Where charcoal has been produced for use in iron furnaces, larger areas of land, approaching 100 acres, have been cut. The same charcoal hearths were generally used several times, and on many areas the woody even-aged regrowth was cut two or three times (Ash 1986; Smith et al. 1988; Martin 1989). On the study area, the forest was cut once, the charcoal hearths were used once, and a mosaic of 1- to 3-acre cuts resulted.

Oak decline and associated mortality have been prevalent at CWMA for the last decade. Mostly black oaks and scarlet oaks have died, but so have other oak species and hickories. Generally, mortality occurs on the poorer sites -- the drier upper side slopes, ridge margins, and ridge crests. Several stress-related factors including senescence, insect defoliation, disease pathogens, climatic fluctuations (particularly drought), and above average stand densities have been proposed as causes for oak decline and mortality. Although none of these hypotheses has been adequately proven, it is probable that a complex of factors contribute to the mortality (Starkey and Oak 1989). On CWMA, the large, fire scarred, overmature black oaks that survived the charcoal cutting and the subsequent fires and grazing are the trees most susceptible to decline and mortality. The younger, more vigorous oaks, for the most part, have not been affected. Thus, current oak decline and mortality may be attributed at least in part to the older age classes and the species segregation initiated by the charcoal cutting.

Data from this study reflect the ability of oaks and hickories to persist in areas that are grazed and burned repeatedly following timber harvesting. Although burning and grazing usually precludes the establishment of woody vegetation, the rootstocks of oaks and hickories have the ability to resprout repeatedly from suppressed buds at or below ground level.

Thus, periodic burning and associated grazing promotes advanced regeneration and establishment of oaks and hickories and gives them an ecological advantage over their associates (Van Lear and Waldrop 1989). Even though research has not determined the precise combination of season, frequency, and number of bums needed to promote oaks through silvicultural practices, it is evident that the land use events on CWMA have favored the development of an oak-hickory forest.

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NATURAL REVEGETATION OF BURNED AND UNBURNED CLEARCUTS IN WESTERN LARCH FORESTS OF NORTHWEST MONTANA

Raymond C. Shearer and Peter F. Stickney'

Abstract—In 1967 and 1968, seven south- and east-facing units, averaging 4-ha each, in a western larch forest of northwest Montana were (1) clearcut and burned by prescribed fire or wildfire, (2) clearcut and unburned, or (3) uncut and burned by wildfire. More than 20 years of forest succession data from permanent transects show that fire caused a marked change in composition of ail vegetation. Herb cover, mostly tireweed, dominated burned sites through the fifth year. Shrub cover (such as from willow or shiny leaf ceanothus) dominated burned sites from the 6th through the 20th years, but the herb cover changed little during this period. Trees rapidly regenerated burned sites, and height of pioneer species, such as western larch and lodgepole pine, exceeded that of shrubs about 7 years after treatment. But the percentage of conifer cover increased slowly and usually required al least 20 years to equal shrub cover. Without tire, the herb and shrub component remained relatively stable; trees were limited to the smaller, more shade-tolerant uncut conifers. Trees established slowly on unburned sites, and most were shade-tolerant subalpine fir and Engelmann spruce.

INTRODUCTION

Disturbance reinitiates the plant succession cycle. Fire has been the agent of the most extensive disturbances in the Northern Rocky Mountains. Land managers can predict successional pathways on the basis of early responses to prescribed burning. Postfire vegetation is composed of "survivor" and "colonizer" species (Stickney 1982). Survivors are established plants capable of regrowth after fire, and colonizers are new plants that establish from seed on the burned site. Seeds of residual colonizers are already on the site and survive fire either in seedbanks in the ground (Baker 1989) or in tree crowns. Seeds of offsite colonizers disperse onto burned areas, usually from nearby unburned sites.

Stickney (1986) attributes early stages of forest succession after fire to differential development of species present in the initial community. Prebum species composition and severity of burning largely determine what survivor and residual colonizer species will be present. Establishment of offsite colonizers depends on the production and dispersal of seed, mostly from nearby sources, and on favorable site conditions for germination and establishment. Once the initial vegetation is established, successional development usually is limited to changes in species abundance.

Establishment of trees may begin immediately after disturbance, but trees develop more slowly than do some herbs and shrubs. Conifer regeneration in the Northern Rockies continues, sometimes in large numbers, for at least 15 years after burning (Shearer 1989). The faster growing herbs and shrubs dominate the conifers until the trees begin sustained rapid height growth.

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This paper describes differences in natural revegetation of south- and east-facing burned and unburned clearcuts in a western larch (Larix occidentalis) forest. Revegetation of a wildfire-burned uncut stand is also compared.

STUDY AREA

The experimental work was conducted in the Miller Creek Demonstration Forest (MC) in western Montana, at latitude 48" 31' N., longitude 114" 43' W. MC is a research and demonstration area in the Flathead National Forest.

Elevations of the treated units are 1,424 to 1,654 m, and slopes average 24 percent (12 to 37 percent). The local climate is cool and moist; mean annual temperature is 5 "C, and mean annual precipitation is 635 mm. The growing season (May to August) has a high proportion of clear, hot days and only 17 to 30 percent of the yearly precipitation falls during this period (Schmidt and others 1976). Soils have developed in glacial till composed of argillites and quartzites of the Wallace (Belt) Formation and overlain with 13 to 140 mm of loess (DeByle 1981).

Forest cover is of the western larch type (Eyre 1980). Percent conifer composition (based on volume of the uncut forest) was: Engelmann spruce (Picea ennelmannii) 31, Douglas-fir (Pseudotsuea menziesii var. glauca) 31, western larch 26, subalpine fir (Abies lasiocama) 6, and lodgepole pine (Pinus contorta) 6 (Beaufait and others 1977). The predominant potential climax vegetation is classified as the Abies lasiocarpa/Clintonia uniflora (ABLA/CLUN) habitat type (Pfister and others 1977). Three phases are represented: Xerophyllum tenax (XETE) on the drier south- and west-facing slopes, Menziesia ferruginea (MEFE) on the cooler middle and upper east- and north-facing terrain, and Clintonia uniflora (CLUN) on west-, east-, and north-facing slopes on the remaining sites.

Table 1.--Site and fire effects description of four units on south-facing slopes, Miller Creek Demonstration Forest

	Unburned	Prescr. fire May 18. 1968	Prescr. fire Aug. 8. 1967	Wildfire Aug. 23. 1967
SITE				
Elevation (m)	1456	1498	1479	1424
Slope (%)	12	22	21	24
Azimuth (deg.)	210	196	193	189
Dry slope (%)	100	88	91	94
Habitat type	ABLA/CLUN,XETE	ABLA/CLUN,XETE	ABLA/CLUN,XETE	ABLA/CLUN,XETE
FIRE EFFECTS				
Fine fuel red. (%)	N/A	82	74	ca 90
Duff red. (%)	N/A	16	84	100
Unburned duff (cm)	N/A	4.3	0.5	0
Soil exposure (%)	N/A	14	84	100

METHODS

This paper reports on portions of two studies that describe herb, shrub, and tree development on south- and east-facing experimental burning units that were (1) clearcut and burned by prescribed broadcast burning or wildfire, (2) clearcut without burning, or (3) uncut but burned by wildfire. The burning units averaged 4-ha in area.

Three south-facing units were clearcut in 1967; the fourth was not cut (table 1). Two of the south-facing clearcuts were prescribed burned (one in early August 1967, the other in mid-May 1968), and the third was not burned. A wildfire burned the uncut unit in late August 1967. The three east-facing units were clearcut in 1967 (table 2). Two of these units were prescribed burned in early October 1967 and early August 1968; the third unit was not burned.

Successional Development

The postfire development of vegetation was measured annually (most units) on permanent plots located within the 4-ha experimental burning units. The permanent plots within a burning unit were referenced to two 25-m baselines, usually arranged end to end (Stickney 1980). Each baseline served as the base for live contiguous 5 x 5 m plots. Within each S-m plot, three smaller plots were nested to accommodate the sampling of lower/shorter woody plants and herbaceous vegetation. Shrubs and trees were sampled according to height as: (1) 2.5 m and taller on 5 x 5 m plots, (2) height 1.5 to 2.45 m on 3 x 3 m plots, and (3) height 0.5 to 1.45 m on 1.5 x 1.5 m plots. Herbs (irrespective of height) and low woody plants (including shrubs and trees <0.5 m high) were sampled in two 0.5 x 0.5 m plots nested in each 5 x 5 m plot along the baseline. Cover (aerial crown) by plant species was measured to quantify the successional development of shrubs and trees and ocularly estimated for herbaceous and low woody plants.

The total number of conifer seedlings and saplings and the number of plots with at least one conifer seedling or sapling were determined at S-year intervals on 31 to 74 temporary 0.0004-ha circular plots systematically installed throughout each unit. Each of these circular plots was enlarged to

Table 2.--Site and fire effects description of three units on east-facing slopes, Miller Creek Demonstration Forest

	Unburned	Prescr. fire Aug. 7. 1968	Prescr. fire Oct. 2. 1967
SITE			
Elevation (m)	1585	1654	1448
Slope (%)	37		22
Azimuth (deg.)	59	4	63
Hoist slope (%)	92	78	80
Habitat type	ABLA/CLUN,MEFE	ABLA/CLUN,MEFE	ABLA/CLUN,CLUN
FIRE EFFECTS			
Fine fuel red. (%)	N/A	92	44
Duff red. (%)	N/A	60	49
Unburned duff (cm)	N/A	2.5	2.8
Soil exposure (%)	N/A	72	88

0.0013 ha to determine the number of (1) established (at least 30.5 cm tall for larch and lodgepole pine or 15 cm for other species) and (2) plots with one tree present. Also, the height of the tallest conifer of each species was recorded for each plot.

The data presented in this paper were not analyzed statistically.

Severity of Fire Treatment

Assessment of fire severity treatment to vegetation follows the Ryan-Noste Fire Severity Index (Ryan and Noste 1985, p. 232) as the standard. Severity, as defined by Ryan and Noste, differs from "tire intensity" because it incorporates the downward heat pulse to site in addition to the upward heat pulse. Expressed as ground char depth, the downward heat pulse is the critical one so far as understory vegetation is concerned. The postfire manifestations of ground char class are expressed in the depth reduction of the litter/duff layer/mantle. On the experimental burning units being reported here, ground char classes ranged from light for most of the broadcast burned units to moderate for the wildfire and summer burned units.

RESULTS

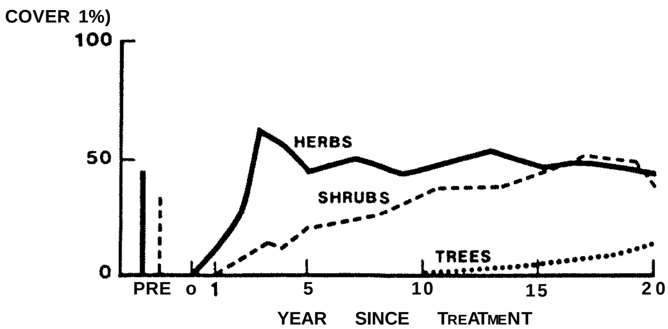
Fire caused changes in composition of all vegetation. The degree of modification varied with severity of the tire treatment as shown by changes on units receiving differing fire treatments on south- and east-facing slopes.

Reforestation of South-Facing Slopes

Overstory trees common in 200- to 250-year-old virgin forests on south-facing slopes at MC include Douglas-fu, western larch, Engelmann spruce, subalpine fir, and less frequently lodgepole pine. The more prominent understory plants include the shrubs huckleberry (Vaccinium globulare), mountain maple (Acer glabrum), and spirea (Suiraea betulifolia); and the herbs arnica (Arnica latifolia), beargrass (Xerophyllum tenax), and prince's pine (Chimanhilia umbellata).ing removes the overstory conifers and, when followed by slash burning, eliminates any onsite seed source for trees.

Clearcut and a spring prescribed fire.
Following clearcutting, a prescribed tire on May 18, 1968 (when the lower half of the duff was still wet from snowmelt and rain) left a continuous, intact, duff mantle as a seedbed and killed the aerial portions of understory herbs and shrubs. Many topkilled herbs and shrubs quickly regrew from root crowns and rootstocks. Forest succession began with the regrowth of an abundant survivor component of arnica and beargrass and the establishment of the offsite colonizers fireweed (Epilobium annustifolium) and bullthistle (Cirsium vulgare). Herbaceous cover developed rapidly. Fireweed quickly established; beargrass regrew less rapidly but more persistently than did tireweed, and was a major component of the herbaceous cover. The herb stage dominated the first 15 years of succession (fig. 1) because shrub development was dependent on the slow recovery of huckleberry and sparse colonization by Scouler's willow (Salix scouleriana). Conifers regenerated slowly because of unfavorable seedbed and harsh site. Regeneration may also have been limited by infrequent good seed crops and the distance from the seed source.

Herbaceous cover and shrub cover were similar (45 to 50 percent) from the 15th through the 20th years. Tree cover developed slowly; increasing to about 10 percent after 20 years (fig. 1). In 1984, 17 growing seasons after treatment, at least one conifer seedling or sapling grew on 79 percent of the plots. There were more than 1,900 total and established trees/ha (fig. 2). Most of these were Douglas-fir, larch, Engelmann spruce, and subalpine fir (table 3). Conifer density was greatest close to the nearby uncut timber; overstocking occurred in patches. There were few trees on a drier slope in the interior of the clearcut. Many of the conifers growing on that slope were exposed to direct sunlight. Most conifers originated from the 1971 cone crop that was rated good for all species.



	PRE	1	5	10	15	20
Survivor :						
XETE (H)	13	3	12	15	18	25
ARLA (H)	14	7	11	1	<1	4
VAGL (S)	29	<1	3	8	16	11
Colonizer :						
EPAN (H)	0	<1	17	22	12	8
SASC (S)	0	0	5	4	5	2

Figure 1. - Early successional development (cover) of major life form groups and prominent species from 1968 through 1988 on a south-facing clearcut, prescribed burned May 18, 1968, Miller Creek Demonstration Forest; ARLA = arnica, EPAN = fireweed, SASC = willow, VAGL = huckleberry, XETE = beargrass; H = herb, S = shrub.

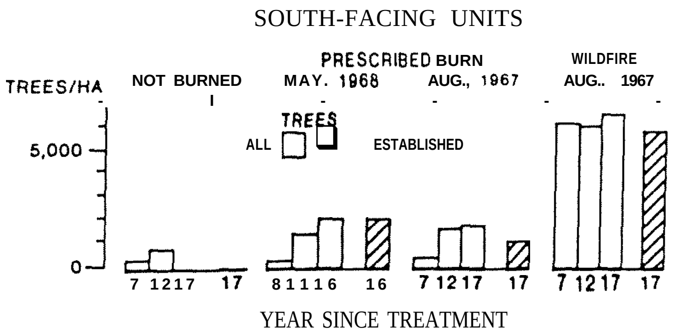


Figure 2. - Average number of all conifer seedlings and saplings per hectare (open bars) by years since treatment and number of established conifers (larch and pine at least 30.5 cm tall, others at least 15 cm) at the most recent measurement (crosshatched bar) on four south-facing units, Miller Creek Demonstration Forest, 1984.

Because all of the important nonconiferous species present in 1988 are traceable to the plant community established in the first postfire year, it may be several decades before seeds from other shrubs or tree species influence succession on this site.

Clearcut and a summer prescribed fire.
In contrast to the spring burning, a prescribed fire on August 8, 1967, when duff moisture was low, consumed most of the duff and as a consequence, killed many plants by burning their aerial portions and lethally heating their roots within the surface 3 cm of soil. A wildfire rebuming the area on August 23 consumed the remaining duff, thereby increasing plant mortality. Consequently, the postfire community resulting from this double-bum treatment contained few survivor plants and revegetation was largely in the form of colonization by pioneer plants. Predominant colonizers were shinyleaf ceanothus (Ccanothus velutinus) originating from seed in a ground-stored seedbank emplaced well prior to the fire and fireweed whose seed dispersed onto the bum in the fall following the fire. A few seeds of conifers dispersed long distances from outside the bum.

Forest succession began with the germination of these seeds and the regrowth of spirea, beargrass, and huckleberry. Early dominance by herbaceous plants, mainly fireweed, was of short duration. Shrub seedlings of ceanothus, germinating profusely from the seedbanks, dominated the site after about 7 years of postfire development. Once the herb and shrub layer provided shade on this south-facing site, more conifer scedlings became established, especially from the good cone crop of 1971 4 years after the fire. Cover estimates are not available for this unit, but shrubs still dominate after 20 years although scattered conifers have overtopped the shrubs. In 1984, most of the 1,500 total and the 970 established trees/ha were Douglas-fir and larch (fig. 2, table 3). Conifers grew on nearly half of the plots.

In spite of an ash seedbed, tree density on most of the clearcut has remained low because there is no onsite seed source. A few trees on a ridge above the unit survived the wildfire and provided some seed for regeneration. High temperature at the soil-air interface and low moisture in the surface 10 cm limited early conifer seedling survival. Lack of moisture and competition with shrubs limited recent survival. The number of established shade-intolerant larch

Table 3.--Percent composition of established conifer regeneration' on south- and east-facing slopes by treatment[b], Miller Creek Demonstration Forest, 1984

Treatment	Tree composition				
	LAOC	PSME	PIEN	ABLA	PICO
		SOUTH-FACING UNITS			
		0			0
CC, PB May 1968	21	41	20	13	3
CC, PB Aug. 1967	32	45	9	10	4
UC, WF Aug. 1967	10	2	1	1	86
	0	EAST-FACING	CLEARCUTS	0	
CC, NB	8	67	33	22	0
CC, PB Aug. 1968		12	57		1
CC, PB Oct. 1967	28	29	21	22	0

'Based on data from 0.0013-ha circular plots that recorded all subsequent natural regeneration >30.5 cm tall for western larch end lodgepole pine and >15.0 cm tall for all other species. ABLA = subalpine fir, LAOC = uestern larch, PICO = lodgepole pine, PIEN = Engelmann spruce, PSHE = Douglas-fir.

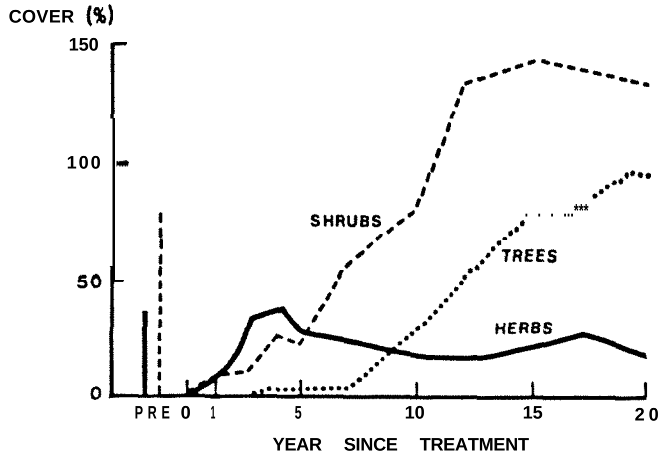
[b]CC = clearcut, UC = Uncut;
PB = prescribed burned, WF = wildfire, NB = not burned

and lodgepole pine decreased by 40 percent from 1979 through 1984, while numbers of shade-tolerant Douglas-fir increased 192 percent.

Clearcut without fire.
A third clearcut was slashed but received no fire treatment. Logging and slashing removed all overstory trees but did not remove the small-diameter understory trees, most of which were subalpine fir. The initial post treatment community was composed almost exclusively of species that were present in the prelogged forest (fireweed was the exception). Alder and menziesia were important shrubs, both before and after logging and site treatment. During the 20 years since disturbance, only a few subalpine fir and Engelmann spruce have regenerated (fig. 2, table 3).

Uncut, summer wildfire.
On August 23, 1967, a wildfire burned a virgin stand of larch, Douglas-fir, and lodgepole pine that was designated for logging later in the summer. The fire killed the overstory trees (except two larch), burned the aerial portion of all other vegetation, and ashed the litter/duff layer to the soil surface. High mortality of huckleberry and beargrass, the major shrub and herb species, resulted from this fire. Because few plants survived the fire, the site was available for colonization by pioneer species. Colonizers in the fist postfire year were fireweed (herb), ceanothus (shrub), and lodgepole pine and larch (tree). Seed sources for these initial colonizers were onsite seedbanks for ceanothus (ground-stored) and lodgepole 'pine and larch (tree crowns) and offsite for fireweed. Forest succession began with the germination and establishment of these tree species coupled with regrowth of surviving spirea, beargrass, and huckleberry. The fast initial growth of herb cover was due mostly to the rapid development of fireweed. Herb cover peaked at 4 years and shrub cover dominated after only 7 years, mainly because spirea and ceanothus grew rapidly (fig. 3). Although most conifers established in the first year at the same time as fireweed and ceanothus, their height did not begin to exceed that of the shrubs until the ninth year. After 20 years, shrub cover still was twice as great as that of conifers (fig. 3). It is expected that increased shading resulting from height growth and crown development of conifers will cause reductions in ceanothus cover. During the winter of 1986-87, low temperature coupled with lower-than-average snow cover killed a large proportion of ceanothus crowns. Some recovery was noted in 1989.

In 1984 (succession year 17), more than 6,400 total and 5,600 established trees/ha, mostly lodgepole pine and western larch, covered the area (fig. 3, table 3). Trees occurred in 97 percent of the plots-the result of seedfall from fire-killed onsite trees.



	PRE 0	1	5	10	15	20
Survivor:						
XETE (H)	23	2	5	10	10	8
SPBE (S)	a	2	11	16	23	26
VAGL (S)	54	0	1	4	5	8
Colonizer:						
EPAN (H)	0	0	16	8	5	3
CEVE (S)	0	<1	<1	55	100	75
PICO (T)		0	2	22	58	82

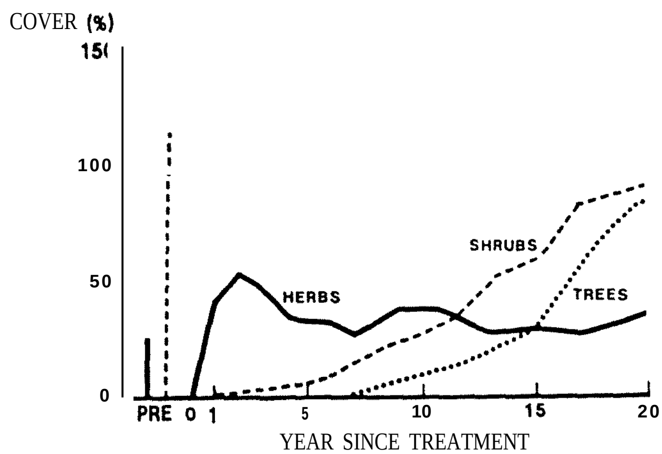
Figure 3. - Early succession development (cover) of major life form groups and prominent species from 1967 through 1987 of a south-facing uncut forest burned by wildfire on August 23, 1967, Miller Creek Demonstration Forest; CEVE = ceanothus, EPAN = fireweed, PICO = lodgepole pine, SPBE = spirea, VAGL = huckleberry, XETE = beargrass; H = herb, S = shrub, T = *tree.*

Reforestation of East-Facing Slopes

Tree species found on the south-facing slopes at MC are also characteristic of virgin forests on more mesic east-facing slopes. The understory shrub and herb species found on the east-facing slopes are typical of moister sites. Prominent shrubs include yew (Taxus brevifolia), menziesia (Menziesia ferruainea), and alder (Alnus sinuata), in addition to huckleberry; important herbs were arnica and oak fern (Gymnocarpium dryopteris). As on south aspects, clearcutting and burning eliminated onsite sources for coniferous seed. Burning decreased or eliminated potential sources for seed in the slash. Where slash is not burned, advance coniferous regeneration survived on the site.

Clearcut and a fall prescribed fire.
After a record-dry summer, a prescribed fire on October 2, 1967 (during the first major storm since late June) burned half of the freshly moistened 5.8 cm duff layer. The fire eliminated western yew, the major understory species, and greatly reduced the *cover* percentages of the *other* principal

understory species, huckleberry and arnica (fig. 4). The reduced duff layer and poor survival of understory shrubs and herbs combined to provide a conducive site for colonization. Five offsite colonizers attained prominence in early succession: fireweed, Scouler's willow, western larch, Douglas-fir, and subalpine fir. As on the south-facing slopes, fireweed showed the most rapid development and attained 47 percent canopy cover by the second year after fire. It remained the most abundant cover species for the duration of the herb stage. Recovery of huckleberry survivors and development of initial colonizer Scouler's willow were primarily responsible for succession to the shrub stage in the 13th year. Shrubs remained the most abundant life form through 1989. Some conifer seeds germinated the first year after the fire but most of them originated as secondary offsite colonizers from the seed crop of 1971, 4 *years* after burning. Because two edges of the unit bordered uncut forest, thousands of conifer seedlings per hectare germinated in 1972. More than 12,600 seedlings/ha, mostly larch and Douglas-fir, survived in 1974 (fig. 5, table 3). Regeneration more than doubled over the next 10 years because of large increases in Douglas-fir, Engelmann spruce, and subalpine fir. During this period the number of shade-intolerant western larch decreased about 22 percent. More than 22,000 conifer seedlings and saplings/ha were counted in 1984, and more than 21,000 of these were considered established. In the 20th year of succession, percentages of cover of trees and of shrubs were high and almost equal, while percentages of herb cover were much less (fig. 4).



Survivor						
ARLA (H)	12	2	1	2	6	9
VAGL (S)	26	1	2	9	21	38
ALSI (S)	7	0	0	0	3	12
TABR (S)	63	0	0	0	0	0
Colonizer :						
EPAN (H)	0	41	30	17	9	12
SASC (S)	0	0	<1	7	14	15
LAOC (T)	0	<1	<1	6	20	44

Figure 4. - Early successional development (cover) of major life form groups and prominent species from 1967 through 1987 of an east-facing clearcut, prescribed burned October 2, 1967, Miller Creek Demonstration Forest; ALSI = alder, ARLA = arnica, EPAN = fireweed, LAOC = western larch, SASC = willow, TABR = yew, VAGL = huckleberry; H = herb, S = shrub, T = tree.

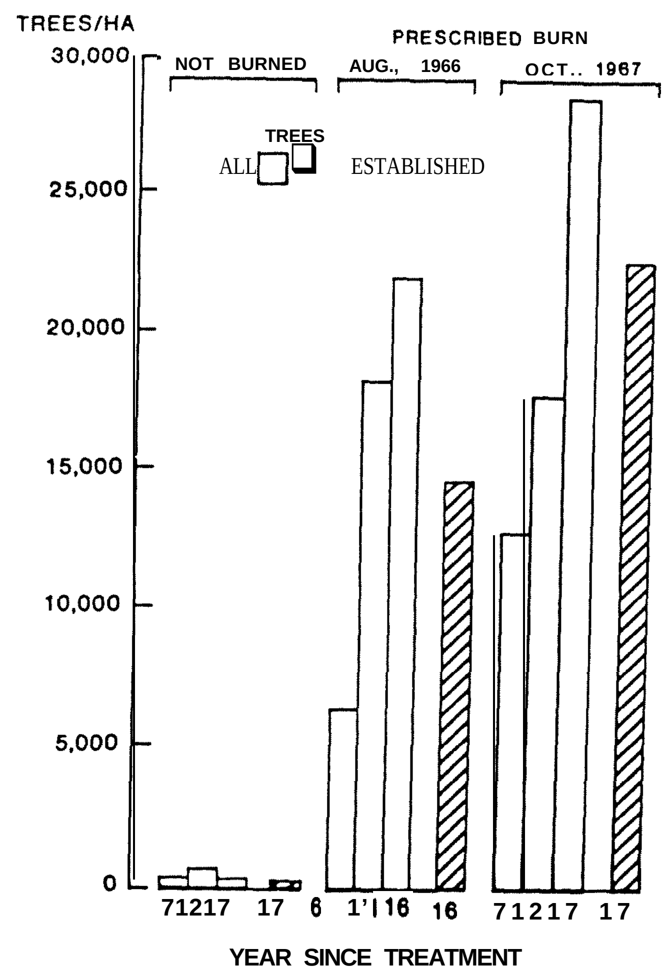
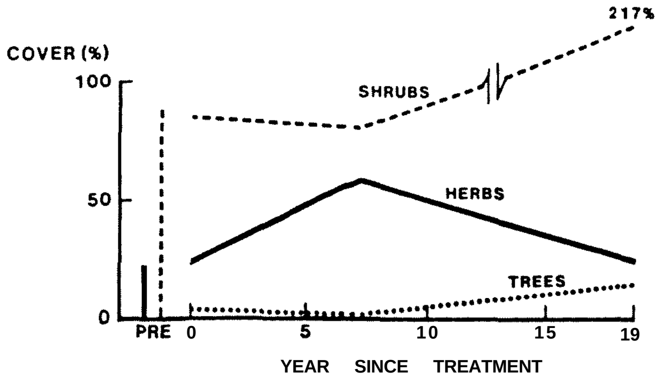


Figure 5. - Average number of all conifer seedlings and saplings per hectare by years (open bars) since treatment and number of established conifers (larch and pine at least 30.5 cm tall, others at least 15 cm) at the most recent measurement (crosshatched bar) on three east-facing units, Miller Creek Demonstration Forest, 1984.

Clearcut and a summer prescribed fire.
After the onsite conifer seed source was removed by clearcutting and slashing, the August 7, 1968, prescribed fire burned the aerial portions of the understory vegetation and removed the upper 60 percent of the duff layer. The fire substantially reduced cover of most of the shade-tolerant understory species, especially huckleberry, alder, and menziesia. Herbs responded quickly, covering 68 percent of the freshly burned site by the second year of succession. Arnica, a rhizomatous herb, attained 19 percent by regrowth; fireweed, which grew from seed from offsite sources, attained 47 percent coverage. The few survivor shrubs regrew slowly in early succession, and cover of herbs was live times as great as cover of shrubs by the sixth year, the last year for which data are available.

Conifers regenerated quickly from seed of offsite trees bordering two sides of the clearcut. Most seedlings resulted from the abundant 1971 seedfall. The moderate or good cone crops of 1974, 1976, and 1980 provided seed for additional seedling establishment. In 1974, 6 years after the fire, there were 6,300 seedlings/ha (fig. 5). This increased to more than 18,100 in 1984 and 21,800 in 1989. About two-thirds of the tree seedlings counted in 1984 were considered established-there were seven times more established seedlings in 1984 as there were in 1979. Of the established conifers, 57 percent were Engelmann spruce, 22 percent suablpine fir, 12 percent Douglas-fir, 8 percent larch, and 1 percent lodgepole pine (table 3). Only the Engelmann spruce were as tall here as they were on the other clearcuts. The few established lodgepole pine were much shorter than elsewhere, but they were 0.6 m taller than the other species. In contrast, height growth of Engelmann spruce was greater than on other burned clearcuts at Miller.

Cleat-cut and no fire.
An cast-facing unit bordering the clearcut prescribed burned on August 7, 1968, provided an unburned contrast. The unmerchantable trees were cut but not removed after clearcutting, but many small subalpine fir and a few other shade-tolerant conifers were. not cut. With no further disturbance, the understory alder, menziesia, arnica, and small conifers constituted the initial community with species composition little changed from the prelogging forest (fig. 6). The major exception, fireweed, colonized sites where removal of overstory trees revealed gaps in the shrub layer. Because shrubs constituted the most important cover group both before and after ***logging, forest*** succession began with an initial shrub stage that has continued for 20 years. Possibly some secondary colonization of aider and menziesia from onsite seed sources occurred during the middle of the first decade. Alder and menziesia have become codominant for the shrub stage. Twenty years after logging, fireweed represents a minor component overall but maintains higher coverage in the openings of the shrub stand. Although adjacent uncut stands supplied much seed, competition for light and moisture is intense and few conifers became established. The current tree component is sparse and consists of subalpine fir that were established before the stand was clearcut and were too small for slashing in 1967 and about 62 Douglas-fir and Engelmann spruce/ha (table 3, fig. 5). A few western larch and lodgepole pine are present, probably growing on small areas disturbed during timber harvest.



Survivor :			
ARLA (H)	5	19	8
VAGL (S)	25	18	20
MEFE (S)	28	23	SO
ALSI (S)	26	34	90
Colonizer :			
EPAN (H)	0	34	8

Figure 6. • Early successional development (cover) of major life form groups and prominent survivor and colonizer species from 1968 through 1987 on an east-facing clearcut that was not burned, Miller Creek Demonstration Forest; ALSI = alder, ARLA = arnica, EPAN = fireweed; MEFE = menziesia, VAGL = huckleberry; H = herb, S = shrub.

DISCUSSION
Maturing forests are inhospitable to the establishment and growth of many native species. Fire renews and rejuvenates aging ecosystems following years of successional changes and the accumulation of duff and litter. The potential for fire to alter succession depends mainly on the composition of the forest community, the onsite seed source, and the severity of the fire. Low-severity fires leave a high survivor component and do little to change species **composition, leaving sites in a** later stage of succession. As fire severity increases, the burned area becomes more favorable to colonizers. Severe fires kill more of the existing vegetation and have the potential to greatly change the course of subsequent succession, that is, set succession back to an earlier stage. Fire similarly affects succession after clearcutting. Without fire or other site modification after timber harvest, conifer regeneration is slow and often excludes shade-intolerant species. The MC study showed response of vegetation to a

wide range of fire characteristics. On south aspects conifer regeneration was successful on an uncut unit burned during the August 1967 wildfire, less so on prescribed burned clearcuts, and unsuccessful on an unburned clearcut. On the east-facing slopes conifer regeneration was successful on prescribed burned clearcuts but was also unsuccessful on an unburned clearcut. Ryan and Noste (1985) show that when plenty of conifer seed is present, regeneration on clearcuts following fire depends on the severity of burning treatment.

Clearcutting followed by prescribed burning can deplete the surface of most of the woody residues. Although conifer regeneration following clearcutting was usually most successful when the duff layer was removed by fire, research has shown that most conifers successfully regenerate through 1.3 cm of duff in the western larch forest type (DeByle 1981). Retention of a shallow duff layer and other organic matter, especially the woody component, protects the soil from intense summer rainfall for the first few years (DeByle 1981) and helps maintain the productivity of the site (Harvey and others 1987).

Many trees that burned during the wildfire on the south-facing uncut unit bore mature cones. Some cones burned and their seeds were lost while other cones were singed and their undamaged seeds dispersed *on* the ash covered surface a few days after the fire. Seeds with unburned wings could disperse farther than seeds with partly burned wings. Overhead shade from dead and surviving trees promoted seedling survival through decreased surface soil temperatures and reduced evapotranspiration. Prompt conifer seedfall permitted establishment before shrub and herb competition for moisture and light became extreme. Following less severe wildfire, more overstory trees survive and serve as a continuing onsite seed source. The availability of onsite seed to affect regeneration depends on the regularity and amount of seed crops and the duration of a receptive seedbed after burning.

Burned seedbed had greater conifer germination and seedling establishment than on unburned sites. Regeneration usually increased where the duff layer was reduced most. Sparse conifer establishment on the two presecribed burned clearcuts on south aspects *resulted* from the harsh site conditions. Without shading, the soil surface dried quickly and temperatures as high as 79 °C were measured in June and July (Shearer 1976). The combination of lethal temperature and rapid drying of surface soil soon after germination often causes high mortality of new seedlings (Shearer 1976). The May prescribed fire left a 4-cm-deep residual duff layer that was unfavorable for seedling survival. Cracks developed in the duff as it warmed and dried during the sunny, dry season following site treatment. In subsequent years, seedlings survived much better in the enlarging cracks than on the surface of duff. The August prescribed fire nearly eliminated all of the duff layer on the other clearcut and left it exposed to extremes in light and temperature. Seedfall was deficient because few trees grew nearby.

The east-facing clearcuts were less influenced by long periods of intense radiation. Each of the clearcuts had one or two sides bordering uncut timber that provided abundant seed for natural regeneration. Both prescribed burned clearcuts regenerated readily and conifers now account for a substantial percentage of the plant cover. Subsequent regeneration failed on the unburned clearcut where no exposed mineral seedbed was left after logging.

Lodgepole pine have serotinous cones, but the other conifers present depend on the current cone crop to provide seeds after late summer or early fall burning. A fair or good cone crop usually provides sufficient seed to regenerate the site the spring after the fire. But for stands that are burned when few *or* no onsite cones are available, regeneration is dependent on offsite seed sources. The number of new seedlings decreases as distance to the trees increases.

If clearcutting occurs near the time of cone maturity, the cones will open and disperse their seeds. Severe slash fires bum much of the duff layer and destroy all or most of the seed, preparing a substrate free of heavy duff and favorable for secd germination. If seeds are present in nearby stands and disseminate into the units, as they did on the east-facing clearcuts that were burned by the summer and fall prescribed fires, prompt regeneration occurs. If seed is unavailable, as in the south-facing clearcut prescribed burned in August 1967, regeneration is delayed. Cutting without slash burning maintains the viable seed, but does not prepare a seedbed conducive to seedling establishment (examples are east- and south-facing clearcuts where fire was excluded).

Exposed charred duff apparently decomposed rapidly. Within a few years, the depth of this layer decreased sufficiently so that significant numbers of conifer seedlings became established. After some early seedling establishment, the abundant seed crop of 1971 dispersed onto the 5-year-old nearly duff-free ground surface and a substantial pulse of new conifer seedlings established.

If these new forests were to bum with a tree-killing fire, the initial postfire community would be composed mostly of survivor species. Principal shrubs would be ceanothus, spirea, and huckleberry; principal herbs would be fireweed and beargrass. Tree species probably would be excluded from the *site* for lack of *an* onsite seed *source.* This condition would be equivalent to the double bum situation in the Northern Rocky Mountains that created large shrubfields as noted by foresters at the turn of the century (Lieberg 1897).

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CHANGES IN WOODY VEGETATION IN FLORIDA DRY PRAIRIE AND WETLANDS DURING A PERIOD OF FIRE EXCLUSION, AND AFTER DRY-GROWING-SEASON FIRE

Jean M. Huffman and S.W. Blanchard[1]

Abstract-South Florida dry prairie and herbaceous wetlands are recognized as tire maintained communities. Aerial photography was used to show how the woody vegetation in Myakka River State Park (Sarasota County, Florida) changed over approximately thirty years of fire exclusion (1939-1968). Rapid increases occurred in Quercus virginiana and Serenoa repens uplands and in forested and shrubby wetland associations. Corresponding decreases occurred in dry prairies and herbaceous wetlands. Present management goals are to maintain and restore tire-dependant plant communities. Drought-condition bums early in the growing season appear to be more effective in reducing woody species cover than traditional dormant-season bums or wet growing-season bums.

INTRODUCTION

Increases in woody vegetation are known to occur in many southeastern Coastal Plain plant communities following fire excusion or when fire frequency is reduced (Heyward 1939; Alexander 1973; Platt and Schwartz 1990; Wade and others 1980).

We mapped vegetation change over a thirty-year period of fire exclusion in two areas within Myakka River State Park, a 11,686 ha. preserve of dry prairie, pine flatwoods, marshes and oak-palm (Quercus virginiana, Q. laurifolia and Sabal palmetto) hammock located in Sarasota and Manatee Counties in Southwest Florida (Fig.1).

Vegetation changes that have occurred following fire exclusion in Florida dry prairies and imbedded wetlands were documented. We also mapped changes resulting from attempts at restoration using the reintroduction of different types of prescribed tires. Restoration efforts over the past several decades suggest that not just fire, but fire at a specific time of year and under specific moisture conditions is critical for restoration of dry prairie habitats invaded by woody species during periods of tire suppression.

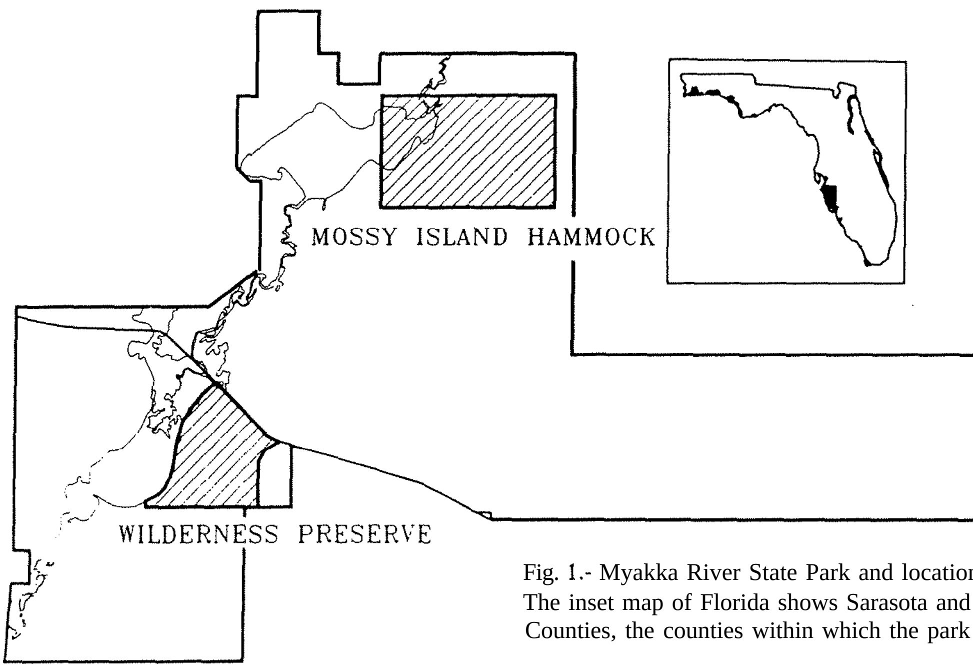

Fig. 1.- Myakka River State Park and location of study sites. The inset map of Florida shows Sarasota and Manatee Counties, the counties within which the park is located.

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STUDY AREA

Climate

The climate of Southwest Florida is characterized by an annual cycle that includes a dry season from October through May and a wet season from June through September (Chen and Gerber 1990). At Myakka River State Park, 61 percent of the average total annual rainfall (144 cm) occurs during the wet season (Fitzgerald 1990). The wet season does not correspond exactly to the growing season, which typically occurs from April through October. Therefore, at the time that growth of plants is initiated in the spring, the dry season has not yet ended.

Fire History

Myakka River State Park lies within an area that has a long history of lightning-ignited fires and fires set by cattlemen. The thunderstorm (lightning) season typically occurs from May through September in south Florida, and approximately 95 percent of total annual thunderstorm days occur during this period (Robbins and Meyers 1989). Fire records from southwest Florida indicate that naturally-ignited tires bum the largest areas at the end of the dry season in early spring (Miller and others 1983). Presumably this was the historic pattern of burning in this region before European settlement. For the last 100 years or more cattlemen and ranchers have burned primarily in the winter, in the beginning and middle of the dry season. These bums usually occur after the passage of a cold front brings rain, when fuel moistures, and often water tables, are high and seasonal wetlands have standing water.

The State Park was established in 1934, a period of strong anti-fire sentiment in the Southeastern United States. During the late 1930's and early 1940's the Civilian Conservation Corps made the fighting of fires at Myakka a high priority. Hundreds of miles of firebreaks were cut throughout the park and fires were extinguished whenever possible. Although many lightning fires were ignited, they were suppressed as quickly as possible. The result was not total fire exclusion but a much reduced fire frequency in most areas. This tire-suppression policy continued in effect through the late 1960's when prescribed burning was accepted as a management tool. Prescribed fire was not regularly used within the park until the late 1970's, when winter-burning was initiated. Spring and summer growing season bums were initiated in the early 1980's (Robert Dye pen. comm.).

Plant Communities

Early accounts of the Myakka region suggest an almost treeless landscape of dry and wet prairies, and scattered pine flatwoods in which closed canopy hardwood forest occurred only as scattered, small "islands" and narrow borders along the river and lake systems. (Townshend 1875; Reid 1843). These are still the major habitat types present in Myakka River State Park today.

A large Portion of the park (6,0000-7,000 ha.) consists of dry prairie, which contains a highly diverse mix of grasses (e.g. Axistida stricta Schizachyrium scoparium, and Sorghastrum secundum) forbs (e.g. Rhynchospora plumosa, Lachnocaulon anceps, Pityopsis nraminifolia, and Carphephorous odoritissima) low shrubs (e.g. Quercus geminata, Vaccinium darrowii, Ilex glabra, and Lyonia fruticosa), and saw palmetto (Serenoa repens). Florida dry prairie is a fire maintained plant community that occurs only in south central and southwest Florida (Davis 1967, Harper 1927). This community has been globally ranked (G2) as threatened by The Nature Conservancy (Florida Natural Areas Inventory 1990). Dry prairie is the native habitat for several species of listed animals including Crested Caracara, Florida Grasshopper Sparrow and Florida Burrowing Owl, all species or subspecies which were common in the 1940's (Van Duyn 1941) but do not regularly occur in Myakka River State Park today.

Hundreds of small wetlands are scattered within dry prairie and flatwoods areas. These wetlands have seasonally fluctuating water levels, typically drying near the end of the dry season. Dominant species include Hypericum fasciculatum, Panicum hemitomon, and Pontederia cordata.

Hammocks are closed canopy forests that are dominated by Quercus vireiniana and Sabal palmetto. They occur along the Myakka River and lakes, and, in smaller patches adjacent to other wetlands. Groundcover is generally lacking or consisting of a sparse cover of herbs.

METHODS

We used Soil Conservation Service and Florida Department of Transportation aerial photos from the 1940's and 1980's to map vegetation in two selected areas within the park. We selected sites that currently have large amounts of dry prairie-hammock edge. The Wilderness Preserve site covers approximately 850 acres and the Mossy Island Hammock site covers approximately 1,500 acres. Site locations are shown in figure 1. In the earlier photographs, boundaries between areas with and those without canopy cover were quite distinct, as most canopied areas had 90 percent or greater tree cover. Boundaries between forested and nonforested communities were very sharp. In photos from the later series, these boundaries were not as clear. Ground-truthing was used for the 1990 series.

Wilderness Preserve areas with greater than 75 percent canopy cover in March 1948 and March 1985 were delineated. The majority of cover consisted of live oaks (Quercus virainiana), laurel oaks (Q. laurifolia), sabal palms (Sabal palmetto), and smaller numbers of South Florida slash pines (Pinus elliottii var. densa). Wetlands with woody cover of red maple (Acer rubrum), buttonbush (Cephalanthus occidentalis), willow (Salix caroliniana), or popash (Fraxinus caroliniana) were also included in the canopy-covered category. Wherever oaks (especially laurel oaks in wetter areas) invaded wetlands, the area was then classified as forested upland.

Mossy Island aerials taken in April 1940 and January 1986 were used both to delineate canopy-covered areas and to distinguish between forested and open wetlands. Species comprising upland and wetland woody cover in Mossy Island were the same as those comprising Wilderness Preserve upland and wetland cover.

Fires that occur during the growing season, after an extended dry period are hereafter referred to as dry growing-season burns. Reductions in the cover of woody vegetation after one dry growing-season bum in the Wilderness Preserve study area, and after two, or in some sections three, dry-growing-season burns in the Mossy lsland Hammock area, are shown using the same mapping methods on aerial photography from November 1990.

Community boundaries were mapped and digitized into a PC ARC/INFO Geographic information System computer database. The digital maps were transformed into state plane coordinates, and areas occupied by the vegetation types were calculated. The aerial photographs were not rectified; because landmarks had changed during the course of 40 years, it was sometimes difficult to locate registration points precisely. Transforming maps lo state plane coordinates helped minimize errors resulting from the use of unrectified aerials. We also used percentage of area covered rather than total acreage to compensate for unrectified aerials.

Field observations were used for descriptions of vegetation composition and change in the mapped areas. Fire and weather records kept at Myakka River State Park were consulted for information on fire conditions.

RESULTS

Before Fire Suppression (1940's Maps)

In the 1940's vegetation cover of the two study areas was predominantly open, with non-woody vegetation the dominant cover type. Open prairie reached to the lake shore in both series (see Figs. 2 and 3). Most wetlands were open, with very few forested or woody wetlands. Hardwood hammocks occurred in small areas closely associated with wetlands.

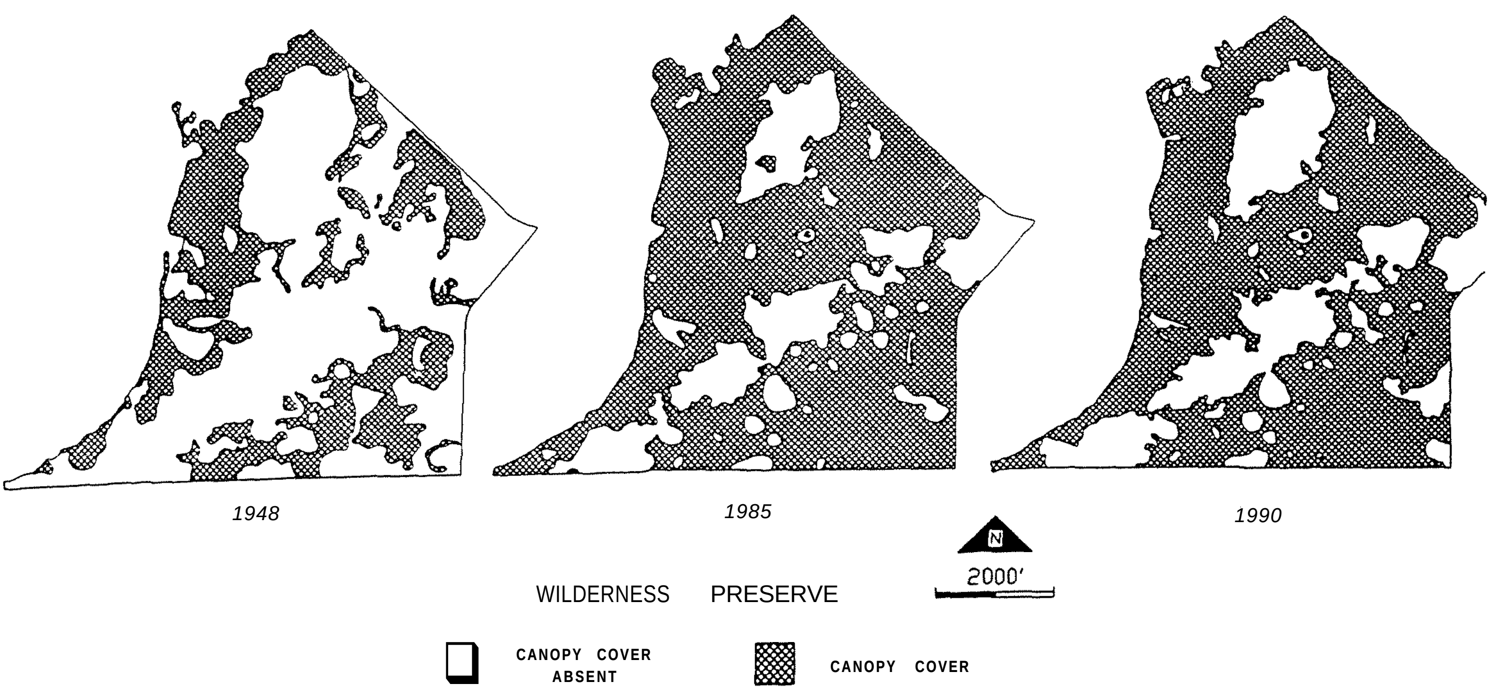
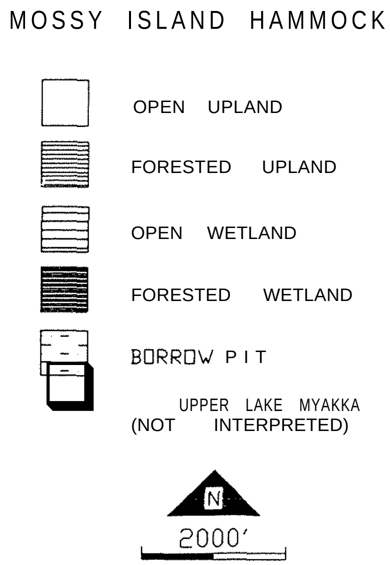


Fig. 2.- Wilderness Preserve canopy cover in 1948 before fire exclusion,. in 1985, after fire exclusion; and in 1990, after one dry-growing season bum. Shaded areas indicate canopy cover. The north, east and southern edges are bounded by roads. The west side is bordered by the marshes of the Lower Lake Myakka.



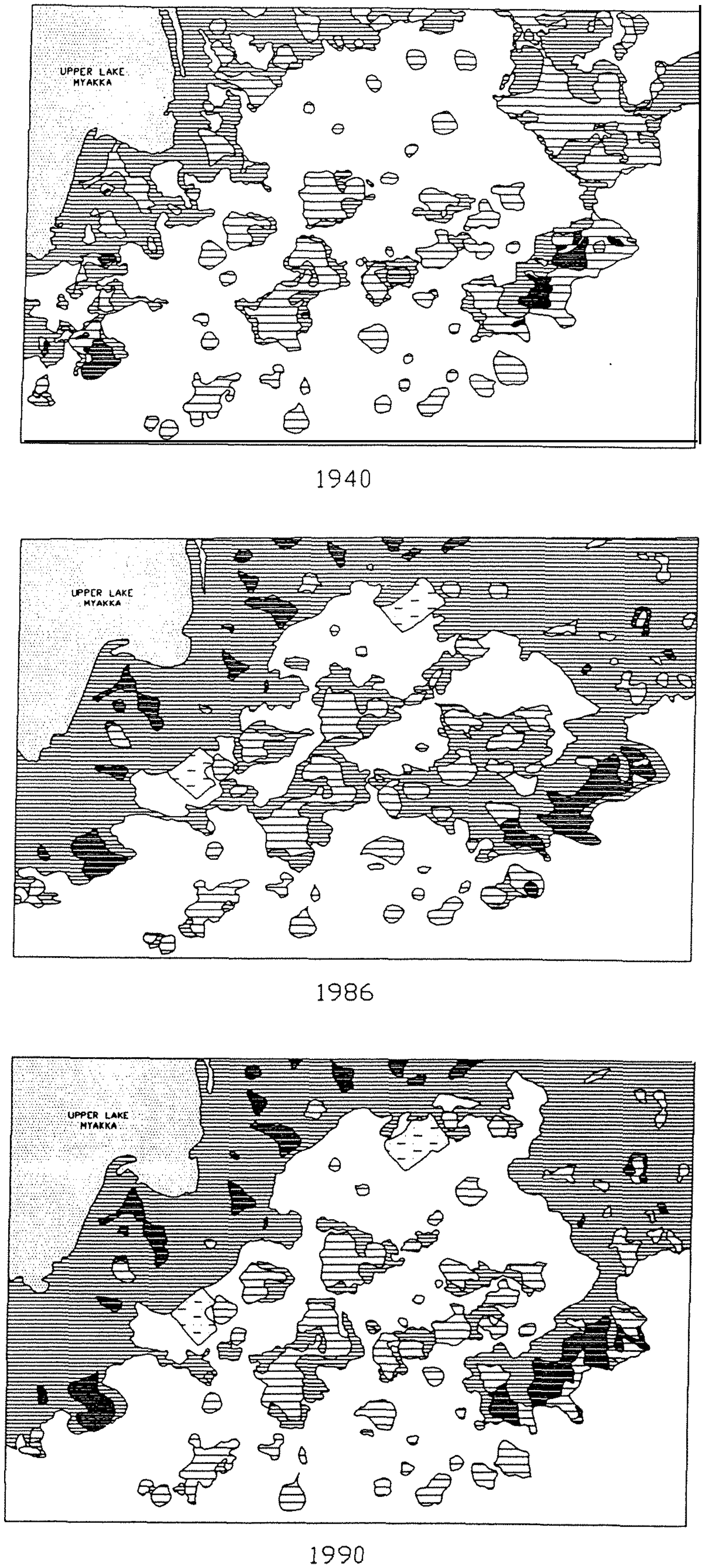


Fig. 3.- Mossy Island Hammock study site vegetation in 1940 before fire exclusion; in 1986 after fire exclusion; and in 1990, after two dry-growing season bums.

After Fire Suppression (1980's)

Uplands - In both sites woody canopy coverage (mostly Quercus virginiana and Q. laurifolia) increased dramatically from 1948 to 1986.

Where oaks were adjacent to dry prairie or pine flatwoods they expanded into these habitats (Figs. 2,3). In the Wilderness Preserve Series (fig. 2) open areas decreased from 66 percent to 26 percent of the total area (fig. 4). In the Mossy Island series (fig. 5) dry prairie decreased from 58 to 39 percent of the total mapped area. Nearly all of this decrease is traceable to the large increases oak stands. It is important to note that oak canopy stands that have expanded into dry prairie since the 1940's are very different from the original oak hammocks. Oak stands that have developed during fire exclusion have a dense palmetto understory (Fig. 6). In contrast, the older, pre-fue-exclusion hammock has an open understory or a sabal palm understory (Fig. 7), with palmettos only along hammock fringes. It is this border fringe of oaks and palmettos that have expanded tremendously during the period of fire suppression.

Wetlands - Nearly half of open wetlands were lost between the 1940's and 1986 (Figs. 2 and 3). In the Mossy Island series in 1940 most wetlands were open and grassy, and wetlands of this type constituted 14 percent of the total area. In 1986 many wetlands were dominated by trees or shrubs and open wetlands constituted only 8 percent of the total area.

Areas of forested wetlands increased from 1 to 4 percent of the total area. This change accounts for 37.5 percent of open wetland loss. Fifty-one percent of open wetland reduction represents shifts to the forested upland category. Construction of a shallow drainage ditch (combined with fire exclusion) contributed to the extensive change from wetland to oak cover in the northeast section of the map. The remaining open wetland reduction is represented by the open wetland area that was converted to shallow borrow pits.

All woody species were mapped as a group, however there were several patterns apparent in woody species increase in wetlands. Open wetlands that are surrounded by hammock are less likely to bum as often as wetlands surrounded by dry prairie. These wetlands were the most likely to change from open herbaceous to woody cover during the period of fire exclusion. This change was especially rapid in floodplain wetlands along the river and lake, and in other wetlands that contained areas of hardwoods in 1940 (compare 1940 and 1986 maps in Fig. 3).

Woody species, including Myrica cerifera (wax myrtle), Fraxinus caroliniana (popash), Salix caroliniana (willow), and Cephalanthus occidentalis (buttonbush), expanded their coverage of formerly grassy wetlands. The most common woody colonizer of small floodplain marshes was popash. This species was not observed, however, to colonize isolated, prairie wetlands. Laurel oaks often moved into the outer zones of wetlands bordered by hammocks. Very little oak cover increase occurred in wetlands that did not have adjacent hammock.

Cover type	Percentage of cover	
	1948	1986
Open uplands and wetlands	66	26
Closed canopy uplands and wetlands	34	74

Fig. 4.-Vegetation cover Wilderness Preserve, Myakka River State Park.

Cover type	Percentage of cover	
	1940	1986
Open upland	58	39
Forested upland	18	39
Open wetland	14	8
Forested wetland	1	4
Borrow pits	0	1
Upper take	9	9

Fig. 5.- Vegetation cover of Mossy Island Hammock, Myakka River State Park.

Fig. 6.- Live oaks with dense saw palmetto understory typical of areas where oak has expanded into dry prairie since the 1940%.

In both floodplain hammock and prairie-bordered ponds wax myrtle sometimes established in the outer zones. Willow, buttonbush or more rarely, maple, colonized or expanded coverage in the deeper, more central wetland zones.

After Reintroduction of Fire (1990 maps)

Prescribed burning was initiated at Myakka River State Park in the late 1960's. Burning in the sixties and seventies halted the expansion of young oaks but resulted in little or no reduction of existing canopy cover. Because fires were conducted in the traditional manner during the dormant-season (winter), at times when fuel moisture and water table levels were. relatively high, fires did not move into the oak-palmetto areas. Fire also did not move into wetlands with increased woody vegetation. Growing-season prescribed bums were initiated in the park in 1983, but it was not until 1986 that a bum moved into oak-palmetto and woody wetland areas.

Mossy Island Hammock. Three bums have occurred in the Mossy Island Hammock area between 1986 and 1990. On May 31, 1986, at the end of a long dry season a fire, resulting from a natural ignition, occurred in the Mossy Island Hammock area. Prior to this fire, the area was last partially burned by a lightning-ignited fire which occurred on August 21, 1985. This fire occurred late in the afternoon and was accompanied by high humidities and rain. Typical of earlier bums it did not result in any significant reduction in oak or woody wetland species cover (Robert Dye pers. comm.). The effects of the May, 1986 bum were quite different from those of previous bums. Many oaks that had invaded dry prairie since 1940 were damaged severely. Some individuals with d.b.h. over 12 inches were killed outright; epicormic and basal resprouting occurred on others. This fire was the first to cause substantial reductions in oak canopy cover.

A second growing-season headfire burned into the same area on June 30, 1988. This fire took place under only moderately dry conditions but also killed many oaks that presumably had already been weakened by the first fire. These bums demonstrated that fre can cause mortality of large oaks when a palmetto understory is present. Slash pine also survived the fire (Figure 8). No oaks in the older hammock areas without palmetto understory were killed. Areas in which oak cover burned corresponded to areas that were dry prairie in 1948 (see figs. 4 and 5).

The Mossy Island Hammock series also shows changes in woody-wetland vegetation, Wetland water levels at the time of these fires were low, especially during the 1988 bum. This allowed fue to sweep across wetlands, reducing woody species cover.

The western portion of the Mossy Island Hammock study area burned once more, on May 11, 1990, with an intense bum. The remainder of the area burned on May 24 and 31, 1990 with a milder bum. The 1990 map of the Mossy Island Hammock series (fig. 3) shows the extent of woody species cover reduction following the 3 bums of June 1986, June 1988 and May 1990.

Wilderness Preserve. The Wilderness Preserve area was burned in May 1983 under high humidities without any significant reduction of oak cover (Robert Dye, park manager, pers. comm.). The first growing-season fire under dry conditions occurred in this area on June 1, 1989. This fire occurred during a period of very low wetland water levels and low humidities and reduced oak canopy coverage in oak-palmetto areas that had established during the period of fire exclusion. The 1990 Wilderness Preserve series map in figure 2 shows the extent of hardwood canopy reduction in 1990 after the dry growing-season fur: of June 1, 1989.

This fire occurred 90 days after the last 1/2 inch rain when nearly all wetlands were without standing water. The fire burned into wax myrtle, willow, popash and buttonbush in areas that had increased in cover since 1948. The fire reduced woody cover in wetlands and resulted in a return of characteristic herbaceous species such as Panicum hemitomom (maidencane) and Polveonum nunctatum (smartweed). No previous prescribed bum had touched these areas.

DISCUSSION

Fire suppression and vegetation change

When fire suppression occurs the boundaries between habitat types change (Myers 1985; Platt and Schwartz 1990). Our data suggests an expansion of a habitat type with elements of dry prairie and hammock but which is actually neither. This oak-savanna, fringing habitat type only, not the original hammock, increased during the thirty-year period of fire exclusion. The dense cover of palmetto still present in the understory of this new habitat type burned under dry-condition prescribed fires and contributed the fuel that resulted in oak mortality. These processes cause this boundary type habitat between dry prairie and oak-palm hammock to be very dynamic, expanding and contracting with varying fire regimes, while true hammock areas remained more stable.

While aerial photography can be used to illustrate oak canopy cover increases and decreases, the changes within dry prairies are more difficult to document. In the absence of fire shrubs and palmetto are known to increase, both in cover and height (Givens and others 1982; Hilmon 1969). These increases may occur at the expense of the herbaceous element of the dry prairie flora. The increase in palmetto and woody shrubs alters fire intensity and behavior, causing less frequent, more

Fig. 7.- Original hammock with live oaks and sabal palms, note absence of saw palmetto understory.

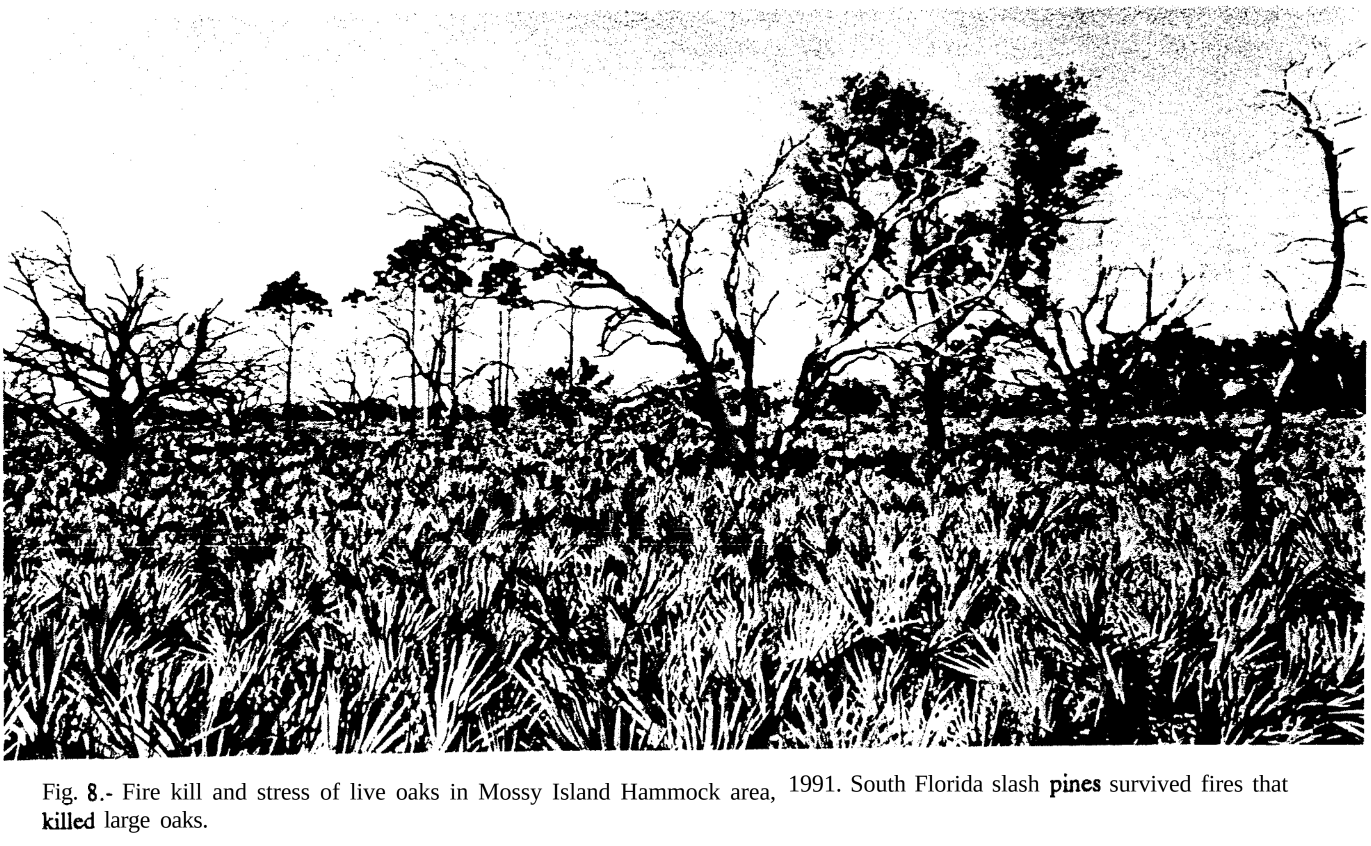


Fig. 8.- Fire kill and stress of live oaks in Mossy Island Hammock area, 1991. South Florida slash pines survived fires that killed large oaks.

intense fires, which may contribute to oak mortality. However the reintroduction of fue, even during the dry growing-season, does not appear to be sufficient to control increased amounts of saw palmetto occurring as a result of altered fire regimes (pers. obs.).

Fire and Hardwood Mortality

Only dry growing-season fires were observed to move into shrub-dominated wetlands. Although other fires occasionally impacted the edge of the oak palmetto zone only dry growing-season fires were observed to move far into this zone and cause mortality in large oaks. These results are similar to those found by (Platt and others 1991), who found that spring fires caused the highest mortality rates for oaks in sandhill habitats in north Florida. They found that fire temperature was not a significant factor in their study of oak mortality but suggested that the phenological state of the vegetation was the most critical factor. We suggest that dry conditions appear to cause added stress to oaks making them even more vulnerable to fire during the growing season.

Management Considerations

Fiie management plans often are implemented in areas that have previously had a history of fur: suppression. When managing a natural area it is important to consider the vegetation changes that occurred during these fire suppression periods.

When reversal of fire exclusion changes is a management goal growing season bums are very useful for obtaining hardwood control. The growing season is recognized as the "natural" fue season in Florida. Growing season bums are known to stimulate flowering of some species and kill invading oaks (Platt and others 1991). Our present study demonstrates the success of dry growing season fires in restoring herbaceous wetlands and reducing oak-palmetto fringe habitat. Many recognize the probable significance of spring drought fires (Robbins and Meyers 1989; Wade and others 1980), however, few managers of natural areas use prescribed fue under spring drought conditions.

Fires occurring under very dry conditions are more difficult to control and more likely to spot for long distances. These considerations must be taken into account but experienced fire managers can conduct successful bums in very dry conditions.

A dry growing-season bum should not be implemented in areas with large fuel accumulations. A fuel reduction bum may be required in these cases and it may be necessary to take special precautions such as removing fuel that has accumulated around the bases of pines before attempting a dry growing-season fire. General sensitivity of pines should be considered. It is important to think about any other possible sensitive elements before conducting bums under very dry conditions.

CONCLUSIONS

In the absence of frequent fire, oaks colonize dry prairie and wetland edges, and hardwood wetland species increase dramatically within wetlands. Where plant communities depend on frequent fire for maintenance, even a few decades of fire exclusion can cause major changes in dominant woody vegetation. The maintenance of open, grassy, dry prairie and wetlands in South Florida is dependent on frequent burning.

The reintroduction of fire after an extended period of fire exclusion, however, does not always reverse the abundance of woody species that have increased as a result of altered fire regimes. Prescribed fire usually does not move into long excluded areas that have oak canopy cover. Even when fire does occur in these areas usually very low or no mortality of well-established hardwoods occurs.

Observations of dry growing-season bums at Myakka indicate that such bums, which usually occur in South Florida at the end of the dry season and the beginning of the lightning season (April and May) may be an important element of habitat management. Conducting all bums under conditions when fuel moisture and water levels are high may be causing significant shifts to occur in vegetation, especially in wetlands. Growing season bums under dry conditions at the end of the dry season, were almost certainly 'part of the presettlement fire regime. At least a periodic bums of this type may be necessary for the long-term maintenance of wetland plant communities.

Dry growing-season bums have also been observed to kill well-established hardwoods -- large oaks in dry prairie and various woody species in wetlands. In areas where hardwood cover has increased as a result of long-term fire exclusion or long intervals between fires, dry growing-season fire therefore is an important component of a fire management plan where. restoration and reversal of fire-exclusion effects is a management goal.

ACKNOWLEDGEMENTS

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LIVESTOCK GRAZING ALTERS SUCCESSION AFTER FIRE IN A COLORADO SUBALPINE FOREST

William L. Baker[1]

Abstract-Plant succession after fires is often considered a relatively predictable process, yet the possibility of alternative outcomes, or multiple stable states, has not been thoroughly studied. The present study makes use of field data collected along a livestock exclosure fence on a site near Pikes Peak, CO, at an elevation of about 3350 m. These data suggest that the west-facing part of the site was once occupied by a subalpine forest dominated by *Picea engelmannii* and that this forest was burned in about A.D. 1867. Livestock grazing discouraged tree regeneration until a major decrease in use in the mid-1940's. On the grazed side of the fence, the forest became dominated by *Pinus aristata*, a tree commonly found on drier, rockier sites like that produced on this moister site by heavy grazing. On the ungrazed side of the fence *Picea engelmannii* appears to be regaining its dominance. This is an example of extrinsically produced alternative outcomes, a successional result that warrants further study.

INTRODUCTION

Plant succession after fires and other natural disturbances has traditionally been viewed as a relatively predictable process ending in reestablishment of the predisturbance climatic climax vegetation. This traditional postfire succession model, which derives from the ideas of Clements (1916), has formed the basis for much research on postfire succession (e.g. Fischer and Bradley 1987). The possibility of relatively permanent alternative outcomes, or multiple stable states (Holling 1973), has been raised (Jameson 1987) but has not been demonstrated with empirical data.

The multiple stable states model was developed to account for the observation that some ecological systems can be moved into alternative states by insect outbreaks, overgrazing, overfishing, pollution, and other disturbances. The essential characteristics of a system with multiple stable states is that the original state is not regained once the disturbance is discontinued (Holling 1973). The existence of multiple stable states can be demonstrated theoretically (e.g. May 1977), but the empirical evidence for their existence has been challenged (Connell and Sousa 1983). Connell and Sousa suggest that all purported examples fail for one or more of three reasons: (1) the physical environment is different in the different alternate states; (2) the alternate states persist only when disturbances are maintained; or (3) the evidence is simply inadequate.

Succession after fires in southern Rocky Mountain subalpine forests is often slow and variable (Stahelin 1943; Veblen and Lorenz 1986). There is some evidence of failure to restore pre-fire tree composition (Veblen and Lorenz 1986), but no clear evidence of multiple stable states. The evidence presented here suggests that livestock grazing following a fire can alter the course of succession, and that the result may be a potentially stable alternative forest.

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STUDY SITE

The study site consists of two hillsides in a subalpine forest about 7 km south-southeast of the summit of Pies Peak, Colorado (figs. 1 and 2). The forest is now dominated by bristlecone pine (*Pinus aristata*), Engelmann spruce (*Picea engelmannii*), and quaking aspen (*Populus tremuloides*). Smaller numbers of limber pine (*Pinus flexilis*) are present. Elevation of the sloping study area ranges from 3,290 to 3,400 m. Treeline is at about 3,650 m.

The study is focused on vegetation on two sides of a fence that prevents livestock that graze on a U.S.D.A. Forest Service allotment south of the fence from entering a protected watershed owned by the City of Colorado Springs (fig. 1). The fence was installed between 1890 and 1902, and the watershed on the north side has not been grazed by livestock since that time (Personal communication, Bennie Baucom, Superintendent, Water System Operations, City of Colorado Springs). The Forest Service lands south of the fence are part of the Deer Park Unit of the Beaver C&H Allotment, which has been a designated cattle allotment since the early 1900's (U.S.D.A. Forest Service, Pikes Peak Ranger District records). Forest Service records of grazing levels in this allotment are spotty, but by the 1950's grazing had been reduced to less than 20 percent of the 1930 level (fig. 3a). Evidence of excessive use was apparent by the 1930's. In 1934, Forest Ranger William Cochran commented in a memo to the Forest Supervisor that "...a very heavy reduction of the present use must be made" (Working Plans, Pies Peak District Office, Colorado Springs). The site was burned by a large forest fire, which is discussed below. The fire burned both sides of the fence and much of the area in figure 2.

METHODS

Six 20- by 50-m (0.1 ha) plots were sampled. Plots 1, 3, and 5 were on the south (grazed) side of the fence, and plots 2, 4, and 6 were on the north (ungrazed) side of the fence.

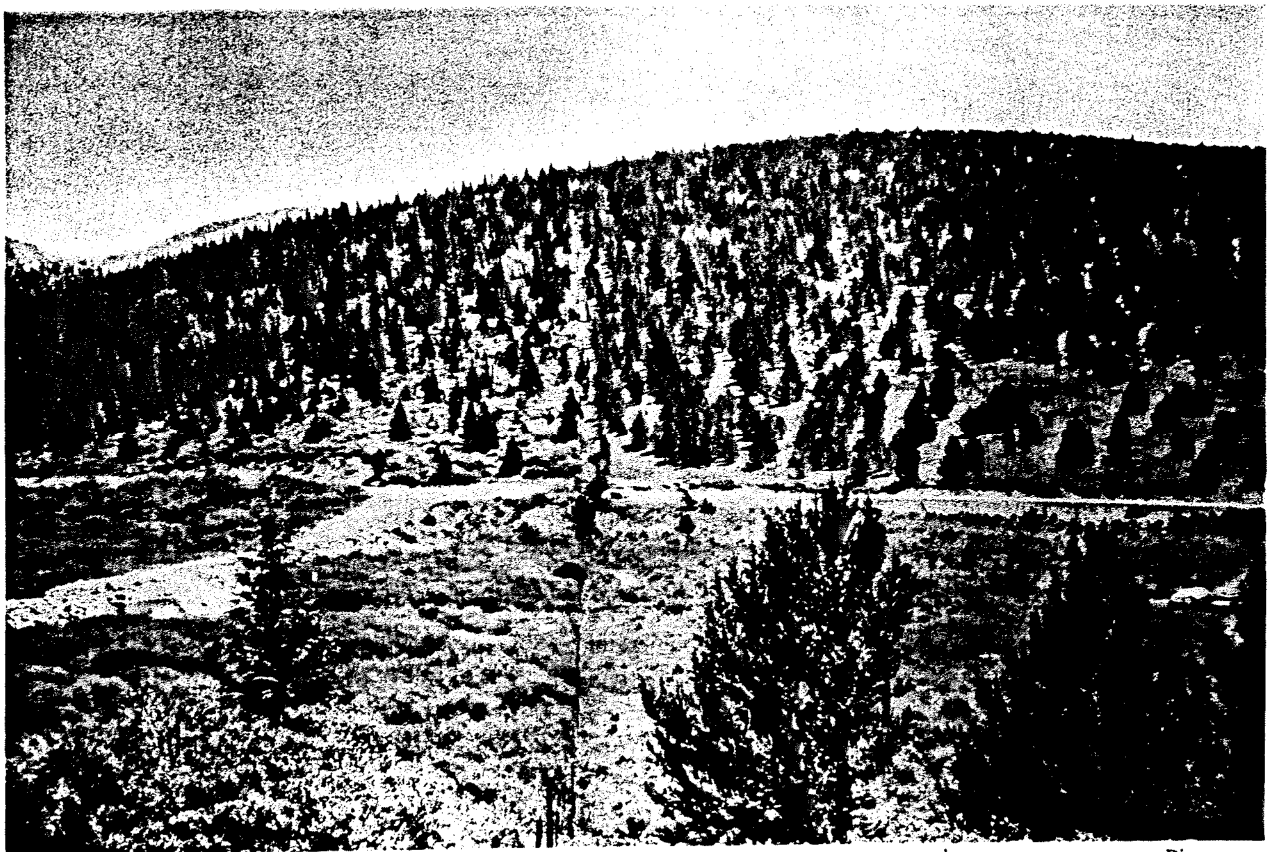
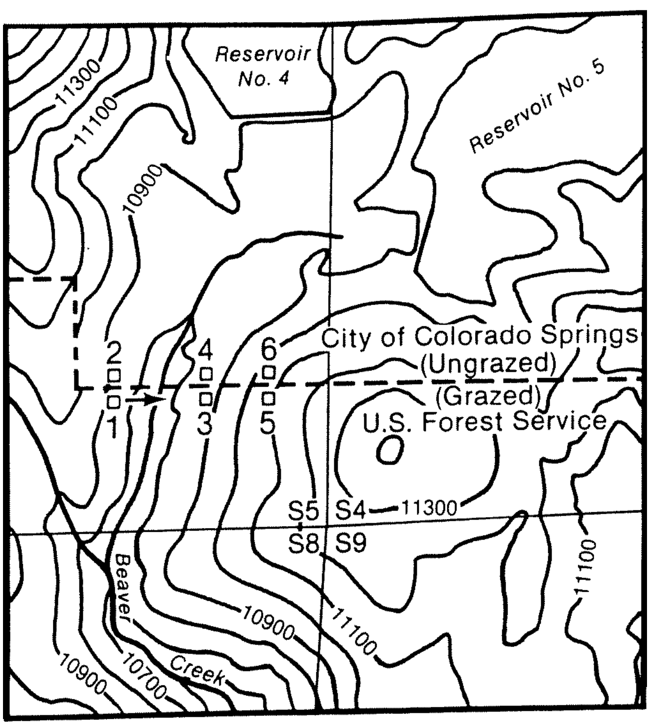


Figure 1.-Looking east along the fenceline at the west-facing part of the site. The ungrazed side, with abundant Picea engelmannii and Salix, is on the left, and the grazed side, with abundant Pinus ~~aristata~~, is on the right. The fire burned across the entire hillside.



LEGEND

- - - - - Fence

□ Sampling Plot

→ Location for Figure 1 Photograph

N

DENVER.

COLORADO

0 .5 Km.

0 .5 Mi.

100' Contour interval

Source: U.S.G.S. Pikes Peak 1:24,000, 1951(Pr 1984).

Figure Z.-Study area map.

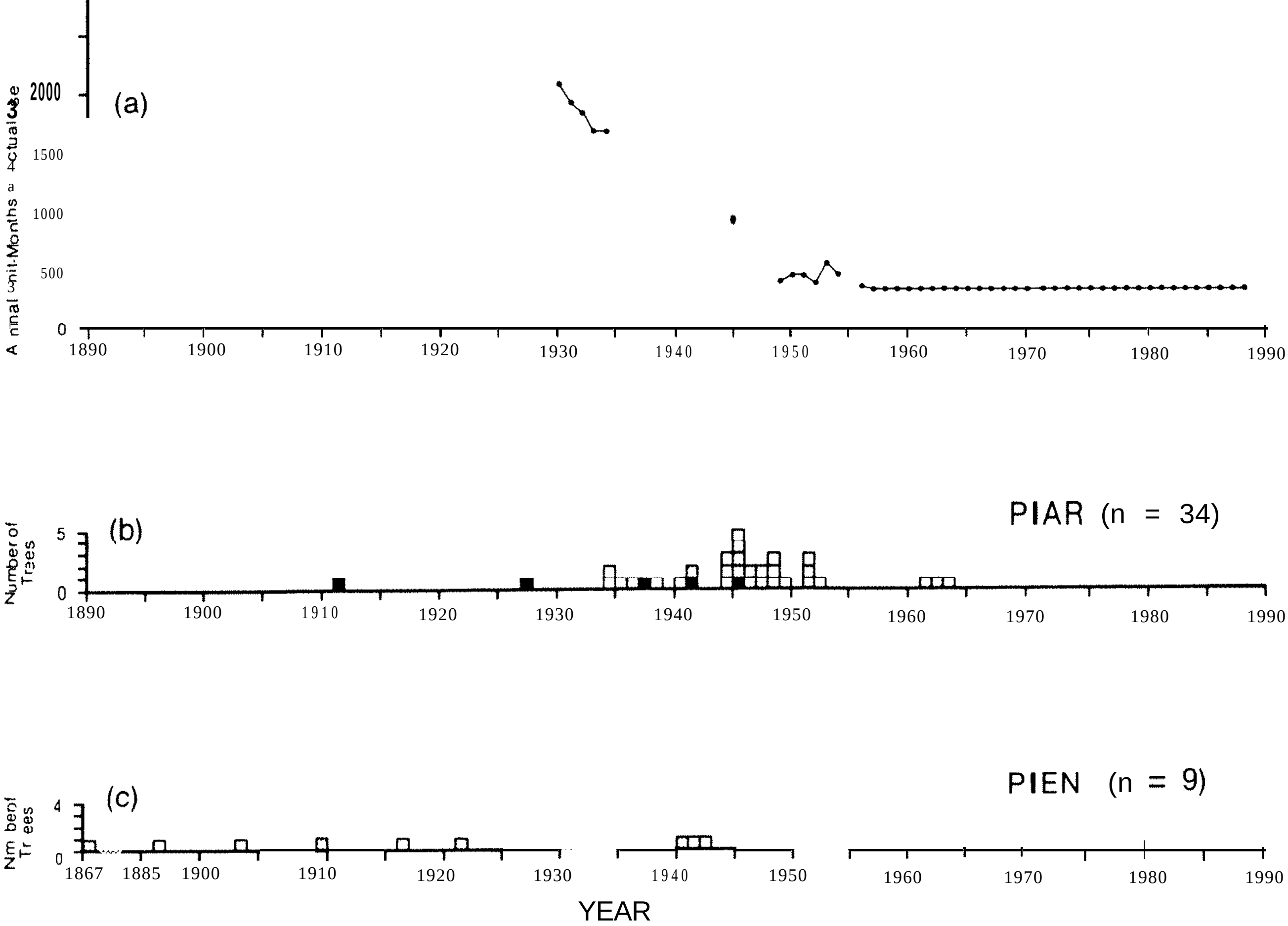


Figure 3.-(a) History of cattle use on the Deer Park Unit of the Beaver C&H Allotment. Data are from U.S. Forest Service-Pies Peak Ranger District records. (b) Dates of establishment of largest Pinus aristata. Solid bars are dates from cores that contained the actual center of the tree. Shaded bars are dates estimated from cores that were close to containing the center (three or fewer years added to estimate the date). Unshaded bars are dates from cores that were an estimated 4-8 years from containing the center. (c) Dates of establishment of largest Picea engelmannii. The shading is as in (b).

Elevation, aspect, and slope were identical in adjoining plots (table 1). In each plot I tallied all trees, including seedlings and saplings, by size class. Seedlings are defined as stems < 5 cm diameter and < 1 m tall. Saplings are < 5 cm diameter and > 1 m tall. The remaining size classes, beginning at 5.0 cm diameter, are all 5 cm wide (e.g. 5 to 9.99 cm).

To determine the composition and structure of the prefire forest, we tallied by size classes the standing dead and downed trees killed by the fire in plots 5 and 6. Standing dead trees were not sufficiently abundant in the other plots. Nearly all the trees killed in the fue could be identified to species, and most stems remained intact, although smaller trees probably were consumed by the fire.

To determine the approximate date when trees became established following the fire, we removed increment cores from the bases of 34 of the largest P. aristata and 9 of the largest P. engelmannii. The cores were sanded, and each core's rings were counted under a stereomicroscope. When the core did not extend to the center of a stem, the number of additional rings needed to reach the center was estimated.

Table 1. --Environmental data for the six fenceline plots.

Plot	Environmental variables		
	Elevation	Aspect	Slope
	Meters	-----Degrees-----	
1	3,307	100	22
2	3,307	100	22
3	3,307	285	20
4	3,307	285	20
5	3,377	270	19
6	3,377	270	19

First the radius of the circle that contained the first ring on the core was estimated. Then the number of rings that might occur within that radius was estimated by multiplying the radius in cm by the average density of rings over the length of the core (rings/cm of core).

The date of the forest fire was determined by cross correlating tree-ring width variations in 14 standing dead burned trees with ring-width variations in the Almagre Mountain master chronology developed by the University of Arizona's Laboratory of Tree-Ring Research (Drew 1974). This chronology is from a site approximately 2 km east of the study area. Tree-ring widths were measured with a stereomicroscope and a computer-assisted Henson incremental measuring machine. Ring-width time series were corrected for growth trends by fitting a negative exponential or straight line to each series. The series were then standardized. INDEX, a program produced by the University of Arizona's Laboratory of Tree-Ring Research (Graybill 1979), was used to perform these computations. Each series, including the Almagre Mountain series, was then pre-whitened by fitting standard autoregressive-moving average (ARMA) time series models. This is necessary to avoid spurious results from cross-correlation (Yamaguchi 1986). The last year of growth present on each burned tree was then determined by floating each time series against the Almagre Mountain chronology and locating the highest cross-correlation coefficient.

RESULTS

The tire probably occurred in A.D. 1867. Many of the last years of growth present on the burned trees, based on the ring-width cross correlations, are near that date. Last years of growth for 14 burned trees were: 1867, 1867, 1866, 1866, 1861, 1861, 1860, 1858, 1856, 1851, 1847, 1840, 1828, and 1815. The last year of growth is not necessarily the fire year, as the fire might have burned into the stem, removing the outer part of the core. Thus the evidence from the burned trees only suggests that the fire occurred in or after 1867. The abundance of dates in the 1860's suggests that the fire occurred within a few years of 1867. The oldest living tree (P. engelmannii) contained 121 rings and an estimated one additional ring to the center, for a pith date of A.D. 1867, suggesting that 1867 was the actual fire year.

The prefire forest on the west-facing part of the site was dominated by P. engelmannii. Stems in diameter classes from about 15 to 25 cm were most numerous, and only a few P. aristata were present (fig. 4). The null hypothesis that the prefire size class distribution for P. engelmannii in plot 5 does not differ from the prefire distribution in plot 6 (across the fence) cannot be rejected at the 0.05 level of significance (chi-square=8.70 and d.f.=6).

Five of eight postfire P. eneelmannii became established between 1886 and 1921, whereas most of the largest P. aristata originated between 1934 and 1952 (Fig. 3c, 3b).

While prefire size class distributions in adjoining plots on opposite sides of the fence had not differed significantly, there were significant differences between postfire size class distributions (fig. 4). In general, P. aristata was much more abundant on the grazed side of the fence, particularly in the west-facing plots (3-6). The null hypothesis, that P. aristata age class distributions were the same on both sides of the fence, was rejected at the 0.05 level of significance (chi-square=51.86 and d.f.=6), for paired plots 1 and 2, but could not be tested for the remaining plots (too many zero entries). Nonetheless, these distributions are completely different (fig. 4). P. aristata is much more common on the grazed side of the fence, and P. eneelmannii was generally much more abundant on the ungrazed side of the fence (fig. 4). The null hypothesis, that P. engelmannii distributions were the same on both sides of the fence, was rejected at the 0.05 level of significance for paired plots 3 and 4 (chi-square= 16.41 and d.f.=5) and paired plots 5 and 6 (chi-square=53.59 and d.f.=6), but could not be tested for paired plots 1 and 2. Populus tremuloides was also more abundant on the ungrazed side of the fence, but was absent from both plots 3 and 4 (fig. 4). The null hypothesis, that P. tremuloides distributions were the same on both sides of the fence, was rejected at the 0.05 level of significance for paired plots 1 and 2 (chi-square=22.36 and d.f.=3).

DISCUSSION

These data suggest that a wildfire in A.D. 1867 burned a west-facing hillside subalpine forest that was dominated by P. engelmannii. P. engelmannii began reestablishing on the burned hillside immediately after the tire, but had not fully reoccupied the site when livestock grazing began. Grazing probably occurred on both sides of the fence until the fence was installed sometime between 1890 and 1902. P. eneelmannii continued to reestablish on both sides of the fence, but at a slower rate on the south side due to livestock use. When the intensity of livestock use was decreased to 20 percent of its former level, in the 1940's, the environment had been modified by the effects of grazing on the vegetation. As a consequence, the site probably had lower cover of Salix and other subalpine plants (as is apparent now-fig. I), and thus greater insolation received at ground level, resulting in an effectively drier site. P. aristata is typically found on drier, more southerly-facing, rockier sites, often in close proximity to moister, more northerly-facing, less rocky P. engelmannii sites (Baker, unpublished data). The grazing may thus have shifted the environment on this site toward one favoring P. aristata establishment. Until grazing was decreased, establishment was not possible. Pearson (1934) has described a similar pattern in which Pinus ponderosa establishment was favored when heavy grazing, which decreased competition from grass, was followed by lighter grazing, that allowed tree seedlings to survive.

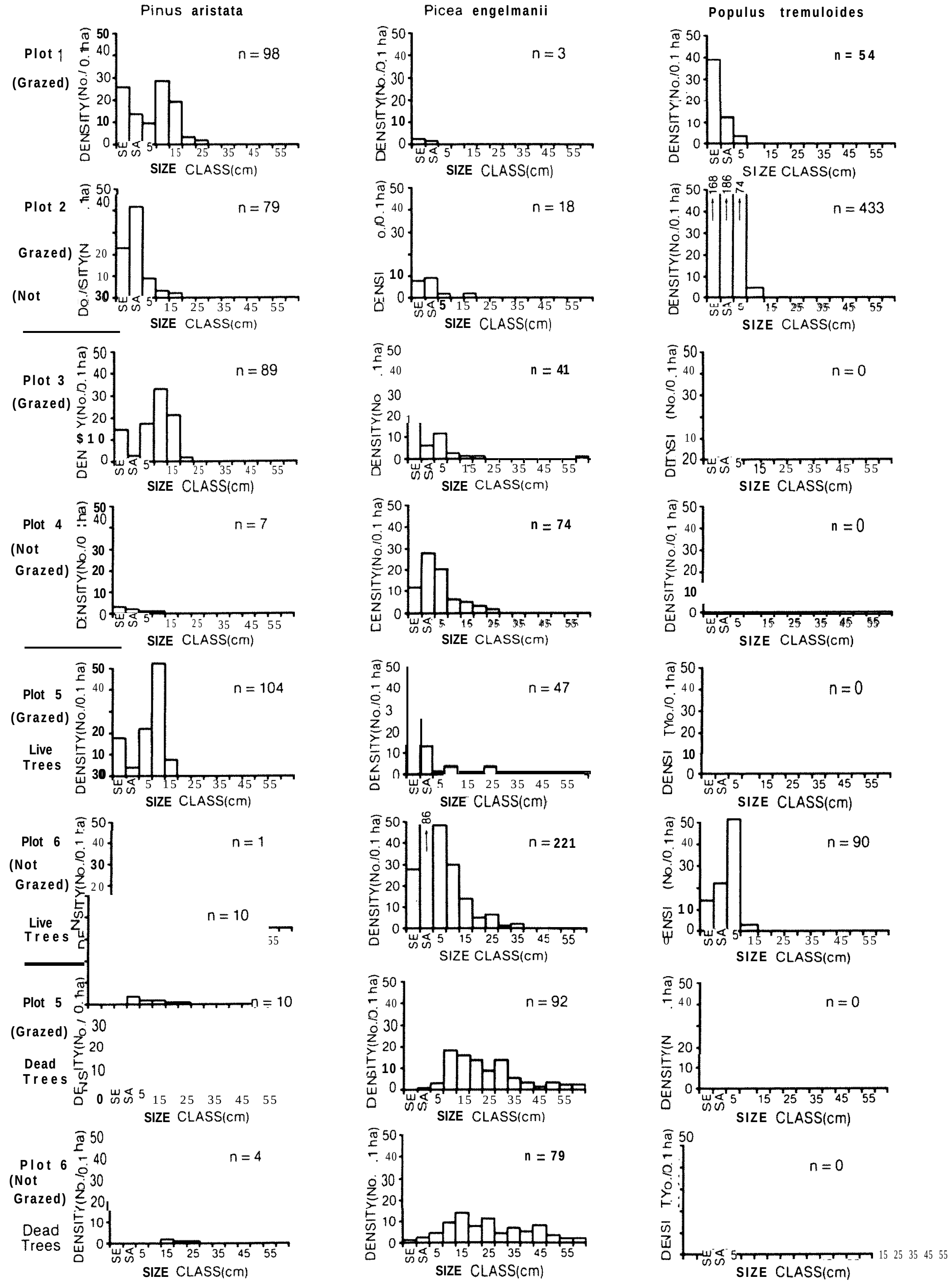


Figure 4.-Size class distributions for all live trees present in the six fenceline plots and for dead and downed trees in plots 5 and 6. Size classes are: SE=Seedlings (< 1 m tall and < 5 cm diameter), SA=Saplings (> 1 m tall and < 5 cm diameter). Other size classes are 5.0 cm wide, with the label indicating the lower limit of the size class.

It is not known whether the P. aristata and P. tremuloides postfire forest on the east-facing part of the site (plots 1 and 2) represents reestablishment of the prefire forest. The prefire composition is unknown. P. aristata is common in the postfire forest on both the grazed and ungrazed sides of the fence, suggesting that on this part of the site grazing is not the primary reason for P. aristata abundance. This part of the site is probably drier, however, because its aspect is somewhat more southerly than that in plots 3 to 6. This drier environment may have favored P. aristata postfire establishment even without grazing.

The pattern of succession on the ungrazed part of the west-facing burned hillside has been as expected. P. engelmannii has returned and has reestablished dominance. In the grazed area on this hillside, however, P. engelmannii has not reestablished dominance. P. aristata has achieved dominance and will likely retain it for some time. The traditional successional model is not appropriate in this case.

Is this, then, an example of multiple stable states? The case described here fails to fulfill Connell and Sousa's (1983) first criterion, as the environment is different on opposite sides of the fence and the difference is the result of an extrinsic influence--overgrazing. The environment is different because Salix is nearly absent from the grazed side of the fence (Fig. 1). This results in a different microenvironment on the two sides of the fence at ground level. The case described here does, however, meet the second criterion of Connell and Sousa, as the alternative state would persist if the disturbance were excluded. Excluding livestock from the south side of the fence could conceivably reverse most of the abiotic effects of grazing and even some of the biotic changes, but the P. aristata population is firmly established and is not likely to die if livestock are removed. Moreover, the coincidence of peak P. aristata invasion with a decline in grazing intensity (fig. 3), suggests that the alternative state of P. aristata dominance is now favored by removal of the disturbance source.

Connell and Sousa also argue that the persistence of the alternative state must be demonstrated through at least one turnover of the population. P. aristata has a maximum lifespan of over 1,500 years (Krebs 1972), and forests containing trees that are 400 or more years old are common (Baker, unpublished data). Connell and Sousa's requirement for persistence is theoretically sound, but neither persistence nor the lack of it can be demonstrated in the near future. The alternative state of P. aristata dominance has already persisted for about 50 years, however, a duration that is significant in terms of forest management. I suggest that a true alternative state should persist following removal of the disturbance agent, but that a variety of inferential evidence of this persistence should be acceptable.

The multiple stable states described by Connell and Sousa (1983), where the alternate state must be maintained intrinsically (by the biota), have yet to be clearly demonstrated to occur in nature. The rigorous criterion of intrinsic maintenance these authors use may be justified from a theoretical standpoint, but it is important not to discourage further study of extrinsically produced and maintained multiple stable states.

Environmental changes accompany most kinds of natural disturbances and human land uses. The possibility of unexpected outcomes following these disturbances and land uses is thus of considerable interest. This possibility has previously been raised in connection with various human influences on vegetation, including effects of tire suppression (Vale 1982), overgrazing (Anderson and Holte 1981; Walker and others 1981; Westoby and others 1989), and clearcut logging (Vale 1988). The failure of trees to regenerate following fires (e.g. Payette and Gagnon 1985) due to climatic change may also produce alternative states. Other potential examples of extrinsic multiple stable states have been reviewed by Holling (1973), Vale (1982), and Connell and Sousa (1983).

In an increasingly managed world in which environmental changes accompany many ordinary human land uses, it is critical to understand how and why persisting alternative states may arise. Livestock grazing following fires may produce unexpected and persistent alternative outcomes.

ACKNOWLEDGMENTS

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CLIMATIC CHANGE AND THE MODELING OF FIRE EFFECTS IN COASTAL SAGE SCRUB AND CHAPARRAL

George P. Malanson and Walter E. Westman[1]

Abstract-Human-inducedclimatic change will affect the processes and rates of species growth and thus the rates of accumulation and the composition of fuel loads. The combination of altered furl loads and altered weather patterns will result in altered fire regimes. Altered tire regimes will in turn affect species growth and community composition. This feedback can be incorporated in computer simulation models of the response of vegetation to a changing climate, but the ways species will grow in climates in which they do not now exist and the way this growth can be translated into fuel loads are not well understood for California shrublands, in which the continual production of basal sprouts allows shrubs to continue life while some branches die, we propose that the apportionment of biomass into live and dead fuel classes is the critical issue in modeling this feedback.

INTRODUCTION

It is becoming increasingly clear that minimizing the risk of wildfires and maximizing natural processes in wildlands are often incompatible goals for land managers (cf. Malanson 1985a). This realization has come with increasing understanding of the history and processes both of wildfire and of ecosystems. If land managers are to resolve these conflicts, it is necessary that the relationship between wildfires and biotic elements of wild ecosystems bc understood. In this paper we discuss how human-induced climatic changes may affect w&&e-vegetation interactions and how some aspects of the wildfire-vegetation relationship might be addressed in future research. We examine computer simulation models of population dynamics in California shrublands in which fire intensity is important, and we consider the way in which ecosystem processes must be translated into fuel dynamics.

The relationship between ecological processes and quantitative estimates of fire intensity has been addressed in a number of ecological studies. We used a calculation of fire intensity to assess the impacts of fire regime on Californian costal sage scrub (Westman and others 1981; Malanson and O'Leary 1982), chaparral (Malanson and O'Leary 1985), and French garrigue (Malanson and Trabaud 1987). WC have also attempted to determine what ecological processes produce fuel loads different enough to result in different fire intensities and rates of spread (Malanson and Butler 1984; Malanson and Trabaud 1988). The incorporation of fire behavior into an iterative computer simulation model of species dynamics is, however, elementary. WC modeled the dynamics of Californian coastal sage scrub over periods of 200 years under a variety of fire regimes, and WC included the effects of altered fire intensity at different fire intervals in this model (Malanson 1983, 1985b). These studies and others like them, have shown that the dynamics of species populations and individual growth affect fuel loads, and that these fuel loads affect fire behavior which in turn affects species dynamics.

CLIMATIC CHANGE

Projected Changes and Responses

If the global climate changes, our present understanding of the feedbacks between ecological processes and fire behavior maybe inadequate for the purposes of managing fire regimes. Projections of general circulation models (GCM's) indicate that the global climate may warm from 1 to 5 °C in the next century due to emissions of radiatively-active trace gases (Schneider 1991). Species responses to a change in climate are not easily predicted. The rate of change in climate at the Pleistocene-Holocene boundary was not as rapid as that projected for the next century, and so analogs from the fossil record may not bc applicable. We know that species can live in less stressful (warmer and moister?) conditions than those occurring in their natural range (Darwin 1859, MacArthur 1972, Woodward 1991); yet we do not know how well species will grow in their present locations if and when the climate changes. The inertia of biogeographic response, which results from the advantage to species already occupying sites, is likely to be important (cf. Cole 1985; Hanson and others 1989). But eventually climatic change will alter relative abundances in given locales and also the production of fuel (e.g., Suffling, this volume). The changes wilt also alter the frequency of the conditions under which fires are ignited and spread (Beer and others 1988). These changes in frequency will not be directly analogous to those that occurred during the Holocene (cf. Clark 1985). Current conceptions of the problems of suppression and prescribed burning are based on the pattern of fuel development and fire weather from the recent past. These concepts may not lit the patterns of the future (cf. van Wagcncr 1987).

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First-Generation Model for Coastal Sage Scrub

Research on the effects of climatic change on Californian coastal sage scrub suggests that altered fire regimes may result in greater changes in species composition than the direct effects of climatic change (Malanson and Westman 1991; Westman and Malanson 1991). We used a computer simulation model that was designed to examine the effects of different fire regimes under a constant climate and altered the fire intensities and ecological functions to include the effects of climatic change. In this model, however, we simulated conditions at a new equilibrium climate, and we did not include a feedback connecting plant growth, fuel, fire, and regeneration. We had previously made observations and estimated fuel loads in Californian coastal sage scrub (Westman and others 1981; Malanson 1985); we altered these estimates as we judged appropriate to reflect climatic changes projected by two different GCM's. These GCM's, the GISS (Hansen and others 1981) and GFDL (Manabe and Wetherald 1980), project warmer temperatures and increased precipitation in southern California. To analyze the effects of altered temperature (Temperature runs), we changed the fuel loads by the same proportion as the change in total foliar cover projected for the species in question in a direct gradient analysis: fuel load was reduced by 13%. To analyze the effects of increased winter precipitation (Composite Moisture Index, CMI runs), we increased the dead fuel linearly from zero at the time of fire up to double that currently found on 40-year-old sites. Albini's FIREMODS program (1976a), which makes use of Rothermel's fire model (1972), was used to calculate new fire intensities on the basis of the new total fuel loads. As an index of fire intensity we used the total heat release (Joules per square meter) calculated by the program (fig. 1). Albini (1976b) recommended total heat release as the best indicator of the effects of tire on vegetation. Because of the complexity of fire behavior and the lack of functions describing the flux of heat from the fire to the regeneration organs of the plants, this index of intensity is best considered to be a surrogate measure of fire effect.

These fire intensities were used to alter the rates of resprouting of the component species for the new climates through previously described response functions based on field observations (Malanson and Westman 1991). Under these fire regimes thee are changes in the relative abundances of the five species involved. However, overall cover changes only when a IO-year fire interval, i.e., a short interval, is assumed; in this case, cover declines throughout the 200-year period simulated. This result indicates that the increases in dead fuel may be the most realistic assumption for most conditions. But this assumption does not address the issue of declining fuel loads under the lo-year-interval fire regime, which indicates that increasing fire intensities are not the only impact. Neither does it address the fuel load changes that would occur if species with different physiognomies were considered. If the area were occupied by increasing cover of grasses or chaparral shrubs, the fuel loads would have to be

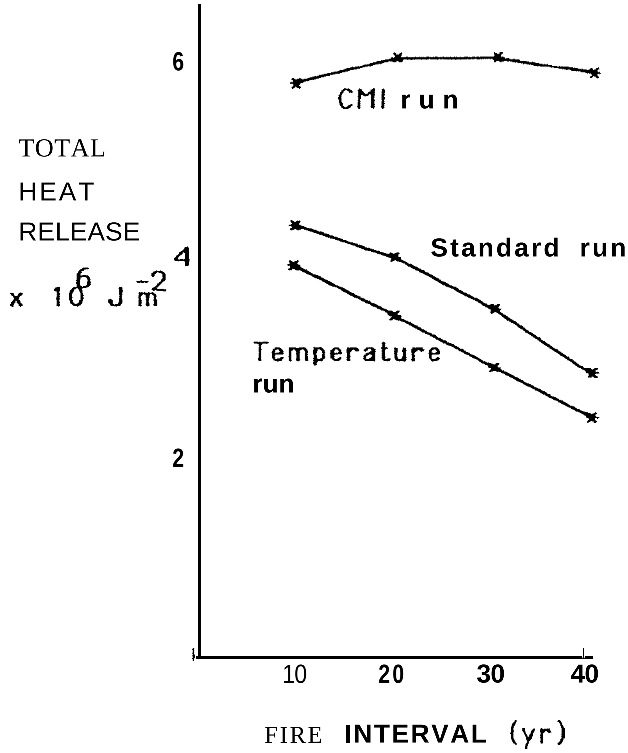

Figure 1. Fire intensities projected for coastal sage scrub under assumptions of altered climate (doubled carbon dioxide levels, GISS model). The Standard run is produced for current observed fuel loads; the Temperature run indicates the results of a decrease in fuel load assumed for an increase in temperature; the CMI (Composite Moisture Index) run indicates the results of an increase in fuel load assumed for an increase in precipitation.

altered in a different ay than to simply assume change in the growth of the coastal sage scrub species. Therefore, it is necessary to include the feedback between species composition and growth in the iterative process of modeling fire in the simulation (e.g., fig 2).

FUEL RESPONSES

The concept that the growth of plants in a simulation should determine the fuel load at the next tire seems straightforward. The fuel load, however, is more complex than a direct measure of foliar cover or of biomass would indicate. First, the distribution of wood in different branch sizes, with different surface to volume ratios and thus different rates of combustion, is known for present shrub species, but both the species and their growth forms may be altered under a different climate. Second, the heat content of the fuel, especially in Mediterranean-type ecosystems where the content of volatile oils is high, may change in anew climate. Third, the spatial distribution of fuels in three dimensions (e.g., grass-shrub proximity or fuel packing) could change with changing grass/shrub biomass ratio. These three factors could result in changes in both fire behavior and intensity, and thus in potential effects on vegetation. For the present, however, wc will discuss one change in the fuel load: the ratio of live to dead biomass.

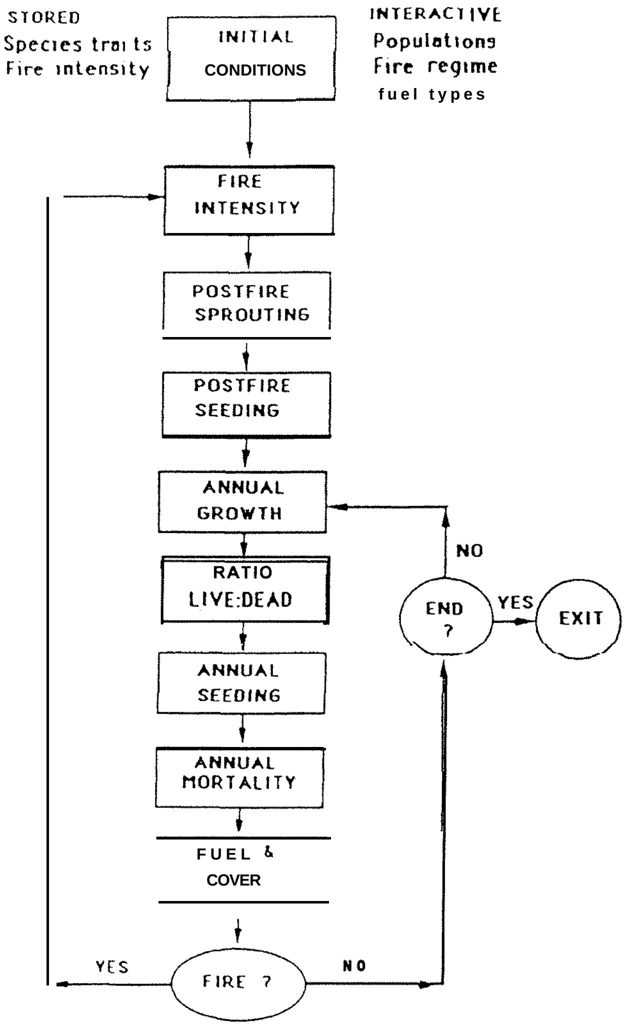

Figure 2. Vegetative response to fire and climate with a feedback to fire intensity through fuel load.

Importance of Dead Fuel

Calculations of fire intensity show that the live:dead ratio is of great importance. Using the fuel loads reported by countryman (1964) for light, medium, and heavy chaparral, we divided these fuels into six categories of live to dead fuel ratios, and entered them into FIREMODS. The results indicate that especially at high fuel loads, when larger stems that are live contribute less to combustion, the proportion of dead fuel greatly affects the estimation of fire intensity (fig. 3). These results are not always consistent with our simulations of tire intensity (total heat release) in coastal sage scrub in an altered climate: our estimates of fire intensity under conditions of increased moisture availability (approximately 6 x 10^6J m^{-2}) may be too high (fig. 1), while our calculations based on Countryman's estimates of fuel load for light chaparral (approximately 2.4 x 10^6J m^{-2}) may be too low (fig. 3).

Dead Fuel in Coastal Sage Scrub and Chaparral

Both chaparral and coastal sage scrub can produce considerable amounts of dead fuel during long fire-free periods. Chaparral has been noted for this feature, which is often referred to as sencsccnce. In coastal sage scrub, we observed that individual shrubs continually produce new basal branches during the fire-free period (Malanson and Westman 1985). Kecley and Keelcy (1988) observed the same characteristic in certain chaparral species. A plant with this trait can replace its dead branches with live ones, and thus can produce new growth without expanding its area. This ability is critical for the continued production of dead standing fuel. In the Mediterranean climate, standing dead fuel does not decompose rapidly, although fuel falling to the ground does not seem to accumulate, since litter loads arc not heavy. This trait of coastal sage and chaparral shrubs, while indicating the importance of recording the change in fuel characteristics through time, may also indicate a pathway for assessing the potential effects of climatic change on standing dead fuel.

Standing dead fuel becomes common in these shrublands after the canopy has closed. This indicates that as the site becomes crowded, and perhaps as nutrient and moisture reserves become more finely divided among individual shrubs and even among the branches of a single shrub, the ratio of production to respiration (Ps/Pr) in and individual branch becomes critical and that branch may then die. Under a

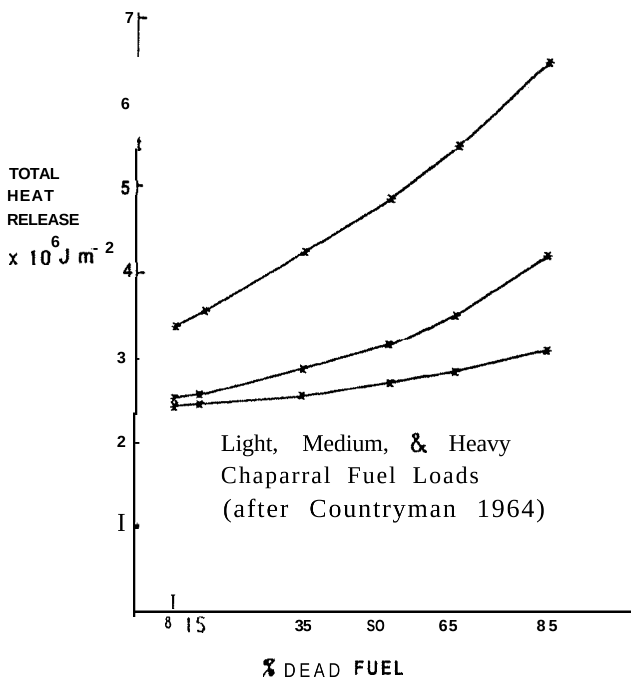

Figure 3. Fire intensities projected for three fuel loads apportioned into six classes of live-todead fuel ratio.

closed canopy, light levels for subcanopy leaves may be insufficient to support a net positive Ps/Pr ratio. This reasoning allows us o link the production of dead fuel to the overall crowding, or foliar cover, and a site--the abundance variable most easily assessed.

Proposed Modeling of Fuel Loads

In our current modeling work for both coastal sage scrub and chaparral, we are incorporating the dynamics of dead fuel. In our previous model of coastal sage scrub (Malanson 1984; Malanson and Westman 1991), and in other models of shrub and forest dynamics (e.g., Botkin and others 1972; van Tongeren and Prentice 1986) growth is limited by crowding. In our previous model, when foliar cover reaches 90 percent, no further growth occurs in any iteration until enough mortality has occurred to reduce cover below this threshold. In other models, and in our own current work, growth is limited much as population growth is limited in the logistic, i.e., exponential growth is increasingly reduced as an upper limit, in our case of total foliar cover, is approached. In order to apportion this growth between live and dead fuels, it is necessary to assume that a proportion of the growth produced during an iteration is in replacement of a branch that has died. We propose to set the upper limit of total foliar cover at 150 percent on a site. When cover exceeds 100 percent, however, an increasing proportion of the decreasing amount of growth is considered to be replacement only (fig. 4). The increase in fuel will proceed as follows: following fire, live growth will begin to fill a site; as the site fills, the rate of growth will slow; once the canopy closes, the rate of growth continues to slow and much of the growth will be recorded as an accumulation of dead fuel; no upper limit for dead fuel is specified a priori. In this function, climate has no direct effect on the quantity of dead fuel, but affects it only indirectly by influencing growth.

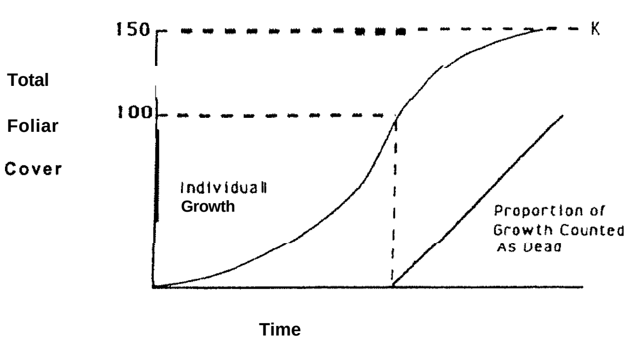

Figure 4. Apportionment of growth into live and dead fuel as overall growth is limited by crowding.

There is another route to modeling dead fuel, however. In earlier models, individuals or proportions of cohorts die: in the forest dynamic models, individual trees die if their rate of growth drops below a threshold; in our earlier simulation, mortality was a function of age. We propose to use the threshold approach in new models of California shrublands. If growth is reduced below a threshold as climate changes, a proportion of the cohort, which is the unit of record in our model, will die and that proportion of the biomass will be added to the dead fuel category. Thus if climates become more harsh, mortality will increase and add to dead fuel, while simultaneously releasing extant shrubs from competition and allowing continued growth.

In our present mode, fuel loads, both in terms of the biomass and the live-to-dead ratio, can then be calculated in a simulation as the growth of species responds to climatic change. Fire behavior simulations that make use of models like FIREMODS require a great deal of computation; it will therefore be best to calculate a matrix of fire intensities for fuel mixtures that vary in live-to-dead ratio, biomass, and the content of chaparral, coastal sage, and grass species (the different physiognomic types vary in their fuel packing and surface to volume ratios). When a hypothetical fire is to occur in hypothetical vegetation, the fire intensity that is appropriate for the projected vegetation can be selected from the matrix. In this way the feedback between fuel and regeneration can be completed.

CONCLUSIONS

These models of species growth and fuel load cannot predict with certainty the abundances of species in climates that do not now exist. They can indicate the general direction and magnitude of changes we might expect. They certainly will help to pinpoint areas in which additional empirical work is needed. While uncertainties do exist, it is probable that the rates at which ecological processes operate in fire environments and the current patterns of fire regime and of species distributions will change with climatic change (cf. Clark 1988). Investments in the planning and implementation of wildland and fire management programs can be more efficient if models of the system in altered climatic conditions are incorporated in the planning process.

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WILDLAND FIRE MANAGEMENT AND LANDSCAPE DIVERSITY IN THE BOREAL FOREST OF NORTHWESTERN ONTARIO DURING AN ERA OF CLIMATIC WARMING

Roger Suffling'

Abstract-A climatic gradient across Northwestern Ontario induces a spatial gradient in tire incidence, with few fires in the Northeastern part and many in the Southwestern part The resultant landscape mosaics exhibit maximum landscape (beta) diversity with intermediate disturbance frequency, as predicted by a theoretical model. This implies that the results of tire suppression on landscape-scale habitat diversity differ qualitatively, depending on previous fire occurrence. Diversity is promoted by fire in tire-free areas, and suppressed by fire where fire occurs frequently. Fire occurrence has fluctuated wildly, however, over periods shorter and longer than the life span of forest trees and, with anticipated anthropogenic global climate warming, fire occurrence may depart from the norms of living memory. Thus the future lightning-fire regime cannot necessarily be regarded as an unmodified feature of the natural environment. Because temporal variation in fire frequency makes estimation of a "natural" tire frequency almost meaningless, wildland fire management policies should not be aimed at maintaining vegetation in a state that is representative of a particular historical time. Policy objectives can be set, however, to retain a minimum area of each ecosystem type, with the minimum defined by reference to historical variation.

INTRODUCTION

Fire management by Europeans in North American forests has proceeded through a number of philosophical phases: from no management, to complete fire suppression, to a mixed model with fire suppression in some areas under some circumstances and fire tolerance or fire setting under others (e.g. Dubé 1977; Elfring 1989; Van Wagner 1990). Change in attitude has been most dramatic in some designated wilderness areas in fire-prone regions where fire is no longer seen as destructive and tends to be viewed as an integral component of the natural environment (e.g. Woods and Day 1977; Houston 1973; Van Wagner and Methven 1980; Romme and Knight 1983; Hemstrom and Franklin 1983; Lopoukhine 1991). In such areas, fires are often categorized as of natural origin and therefore to be left to burn if possible, or of human origin and thus to be suppressed (e.g. Anon 1975; Elfring 1989; Schullery 1989). Not everyone endorses this approach, however, as the aftermath of the 1988 Yellowstone fires has demonstrated (Bonnicksen 1989; Buck 1989). In overtly modified landscapes, fire is usually suppressed, but is also used as a tool for deliberate modification of the landscape (e.g. Rego and others 1988; Amo and Gruell 1986), or for reduction of unnatural fuel accumulation (e.g. Wade and others 1980; Pehl and others 1986; Birk and Bridges 1989).

There is a widespread belief that fire promotes what is variously described as landscape diversity or heterogeneity in both wilderness and overtly modified landscapes (e.g. Wright 1974; Romme and Knight 1982; White 1987; Hannson 1979; Forman and Godron 1986; Loucks 1970 ; Agee and others 1990). The first part of this paper calls into question the universality of this notion. It is hypothesized that fire, whether natural or otherwise, can promote landscape diversity, but can also suppress it in definable circumstances.

These ideas arc of more compelling concern in view of anticipated global climate change. Atmospheric carbon dioxide concentration will probably reach double the pre-industrial revolution level in the next 50 to 100 years (Bolin 1986), thus trapping more heat in the lower atmosphere. Various general circulation models suggest that a doubled CO, concentration will increase global mean equilibrium surface temperature by 1.5 to 5.5°C in this period (Botin and others 1986; Flavin 1989; Anon 1990).

Studies by Van Wagner (1988) and Suffling (1990) confirm the general belief the area of forest burnt in Northern Hemisphere regions is greater during warm summers. Thus climate warming is of direct concern to fire managers, as the fire climate will probably deviate from that of living memory. The second part of the paper addresses possible fire management responses to climate warming and landscape diversity questions in wildland areas.

A MODEL OF LANDSCAPE DIVERSITY

Many landscapes, including continental boreal forests, can be thought of as disturbance mosaics or, in more abstract terms, as populations of ecosystems. Heinselman (1973) introduced this notion when he redefined fire as normal but infrequent in temperate forest landscapes. This led to the attractive notion of Temperate Zone wildland forested landscapes in which "Fire rotation controls the distribution of age classes of stands and the succession within stands. The resulting diversity may represent long range stability, as implied by the

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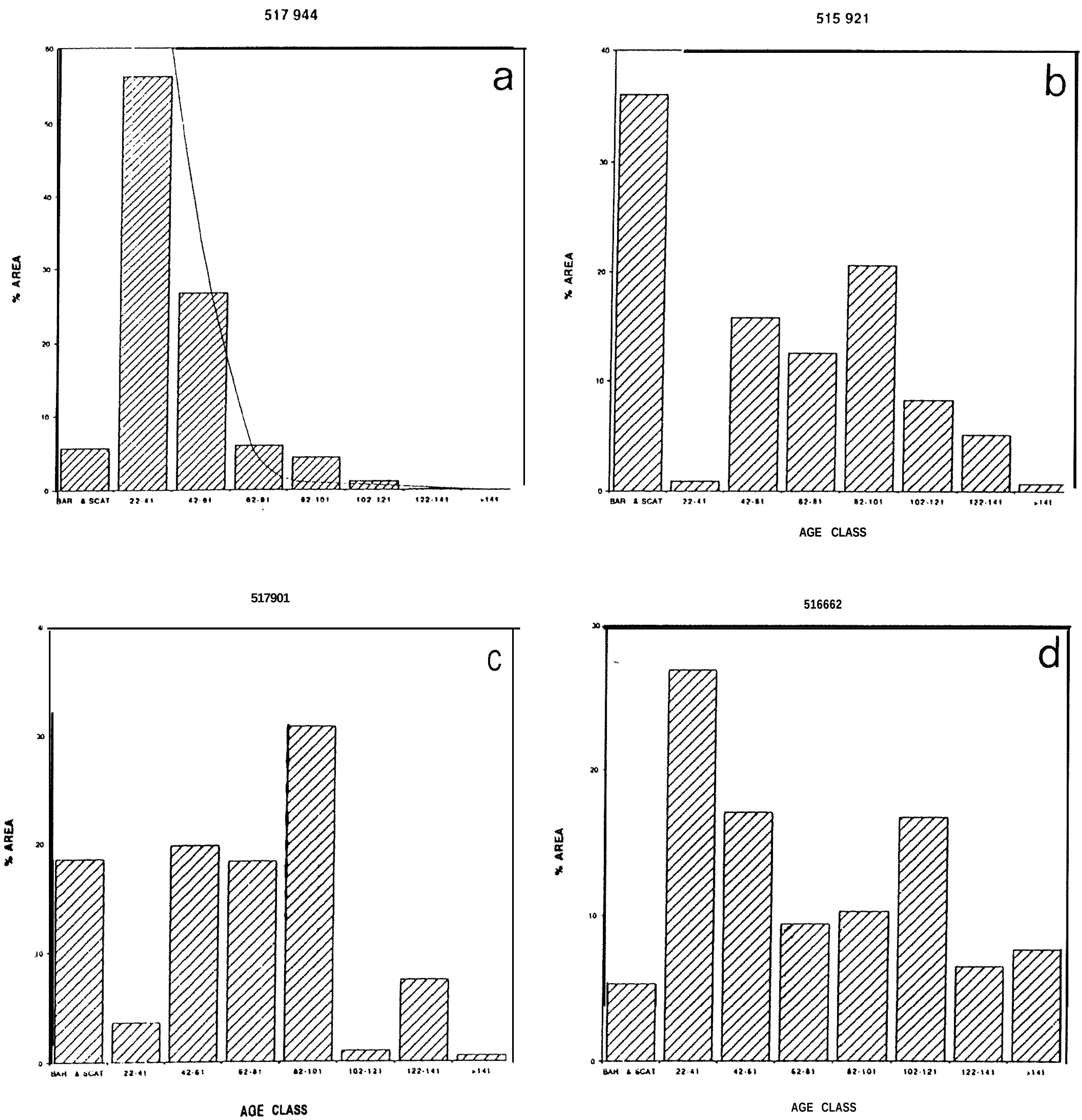


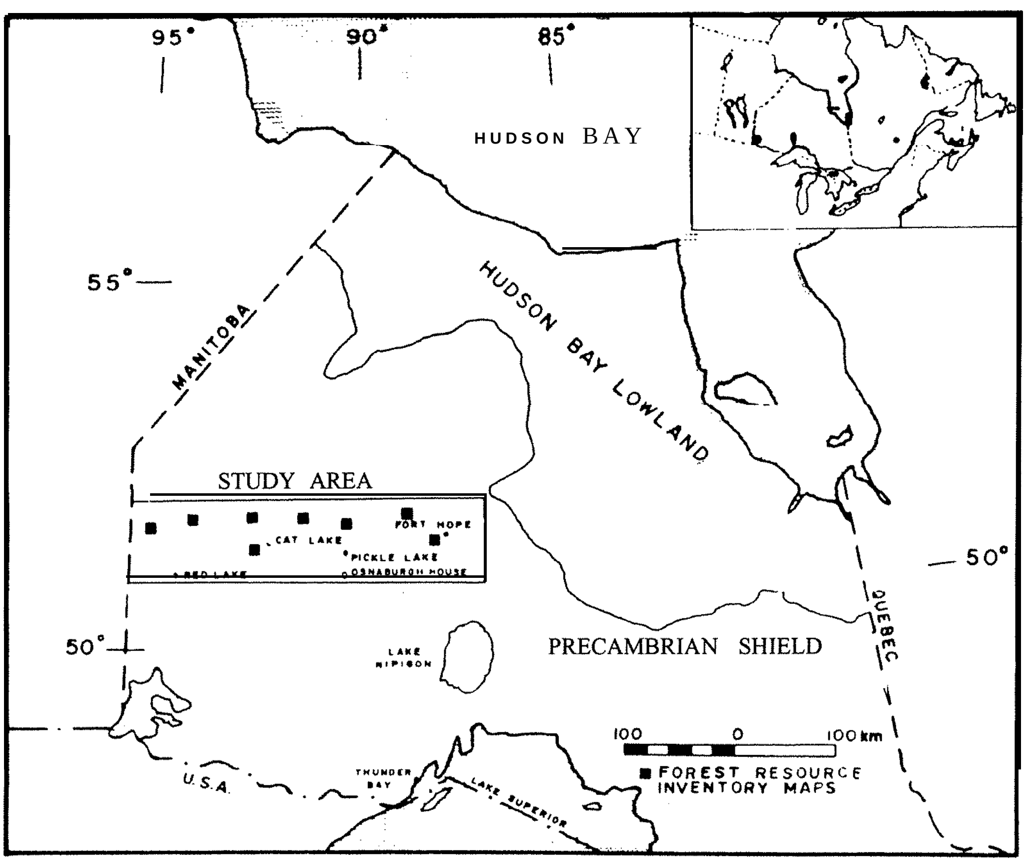
Figure I --Stand age-class distributions for Northwest Ontario for four areas ranging from most fire prone (la) to least fire-prone (Id). The data for the 4 graphs are stand ages of main stands since disturbance recorded on 1: 15840 Ontario Forestry Resource Inventory maps at the 8 locations shown on figure 2. Map titles indicate longitude and latitude (e.g. 516882 = 51.6°N 88.2°W). A negative exponential curve (Van Wagner 1978) is fitted for figure la ($r^2=0.81$), but omitting the "barren and scattered" category which is an amalgam of recently regenerated stands and sparsely treed areas such as rock barrens.

palaeoecological record" (Wright 1974). This theory was given quantitative form by Van Wagner (1978) who showed, for the same Minnesota Great Lakes mixed forest landscape, that the distribution of stand ages followed a negative exponential curve (see figure la for an example of this distribution). The model applies if the chance of disturbance of any stand is equal throughout its life and if the amount of disturbance remains substantially unaltered in the long term. Some subsequent investigations confirmed the model (Yaric 1979; Harmon 1984), but other studies and data did not support it, or applied inconsistently (e.g. Hemstrom and Franklin 1982; Suffling 1983; Tande 1979; Antonovski and Ter-Mikhaelian 1987). This is leading to increasing support for a shifting-state concept of forest landscape. These latter results tend to demonstrate what palaeoecologists have long claimed, that the area1 amount of disturbance fluctuates widely over time, not only in the short term, but also over periods as long as or longer than the life span of individual trees (e.g. MacDonald and others 1991; Romme and Knight 1982; Romme and Despain 1989). Figure 1 shows a typical range of age-class distributions encountered in Northwest Ontario, Canada, where change in fire occurrence over time disrupts the negative exponential pattern, especially where the overall firc return period is long, as in figure ld.

The disturbance mosaic can be used to calculate the landscape diversity, or beta diversity associated with differences between stands in the mosaic (Suffling 1983). This diversity has a richness component (essentially the number of kinds of stand), and an evenness component expressing the relative amount of different kinds of forest (Suffling and others 1988). The two measures are commonly combined in the Shannon equation (Shannon 1948).

Landscape diversity is a function of inherent differences between sites based, for instance, on aspect or drainage. It also depends on the forest age class distribution that has been created by disturbance. Simulation models of stand-age distributions over time predict that landscapes with intermediate frequency of disturbance should have higher landscape diversity than those with very frequent disturbance and those that have almost no disturbance (Suffling and others 1988). This is the case whether fire occurs equally in all age classes or is concentrated on older ones.

The continental boreal forest of Northwest Ontario (fig. 2) was used to test the model (Suffling and others 1988). This huge, more-or-less flat glacial peneplain, exhibits spatial climatic variation that is little affcctcd by altitude, and its



Figurc 2--Thc location of the study area in Northwest Ontario geology is sufficiently uniform that climate variation begins to show consistently across the landscape. The area adjacent to the Hudson Bay Lowland is cooler and more humid than that against the Manitoba border. The amount of fire reflects this climatic variation, grading from a high of over 1 percent of land area burnt per year in the Southwest to almost no fire at all in the Northeast (Fig. 3). Most fires arc stand-replacing crown fires, and the size of the disturbance patch created can vary from less than 1 to over 100,000 ha. There has, thus far, been very little logging in this area.

The fire gradient induces a cline in vcgctation. Forests in the southwest arc generally young and are dominated by fire-adapted jack pine (*Pinus banksiana* Lamb.) and aspen (*Populus tremuloides* Michx.). Those in the northwest arc generally much older and balsam fir (*Abies balsamea* (L.) Mill.) and white spruce (*Piceo glauca* (Mocnch) Voss) are much commoner there (figs. 1 and 4). In the center of the region, a mixture of these forest types prevails (Suffling 1988). Measurements of landscape diversity (fig. 5) confirm the theoretical predictions that diversity will be highest in the center of the area, where frequency of disturbance is intermediate (Suffling and others 1988).

EFFECTS OF PRESCRIBED FJRE AN-D FIRE SUPPRESSION ON LANDSCAPE DIVERSITY

Predictions that the model generates, and the empirical confirmation of its applicability, justify several generalizations concerning fire management. In a landscape with little previous disturbance, prescribed burning will increase landscape diversity by creating patches of immature habitat in the primarily mature mosaic. Conversely, application of more fire in an already frequently burnt landscape will reduce the diversity of the landscape. In a landscape previously experiencing intermediate disturbance that has produced maximal landscape diversity, either tire suppression or increased prescribed or natural fire will reduce the landscape's diversity.

Land managers are thus faced with a problem: Promoting maximal landscape diversity is not necessarily synonymous with keeping an area pristine. By managing for some primeval wilderness condition with a different tire occurrence from the present one, a manager might actually reduce landscape diversity. In reality, howcvcr, many wildland areas have for many years been managed under tire exclusion policies that have climinnted or reduced both lightning tires and aboriginal burning patterns (Barrett and Amo 1982; Lewis 1977), and have tended to result in a bell-shaped distribution of stand age classes (Van Wagner and Methvcn 1980).

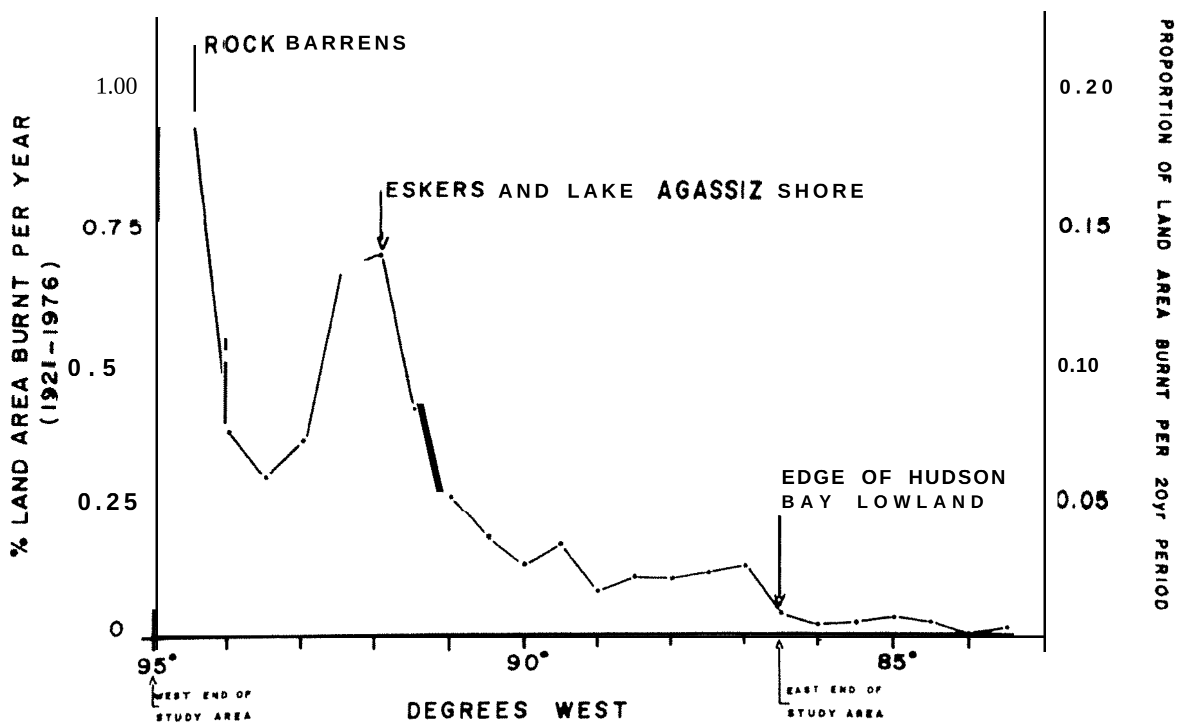


Figure 3--The gradient in fire occurrence across the study area in Northwest Ontario. (After Suffling and others 1988)

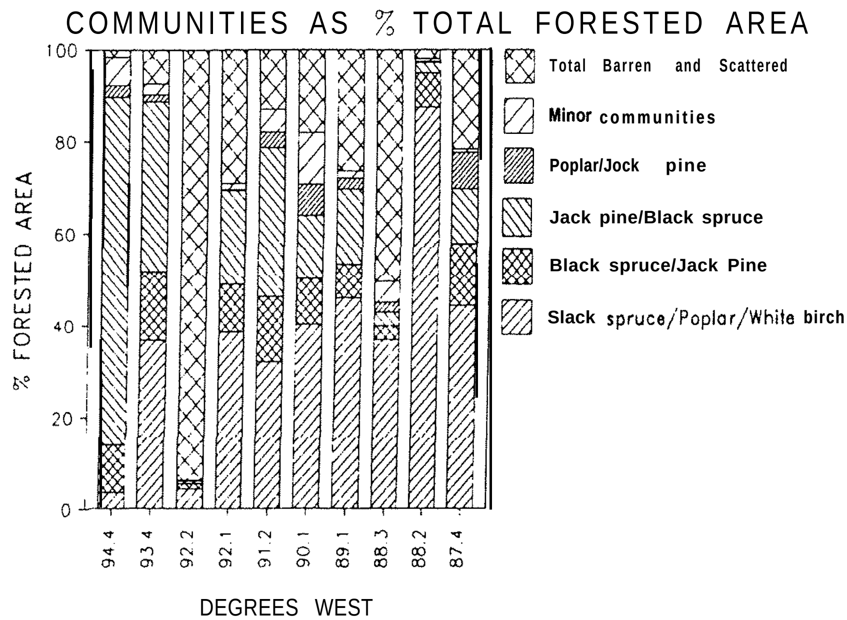


Figure 4--The change in community composition of forested upland sites across the study area in Northwest Ontario.(After Suffling 1988).

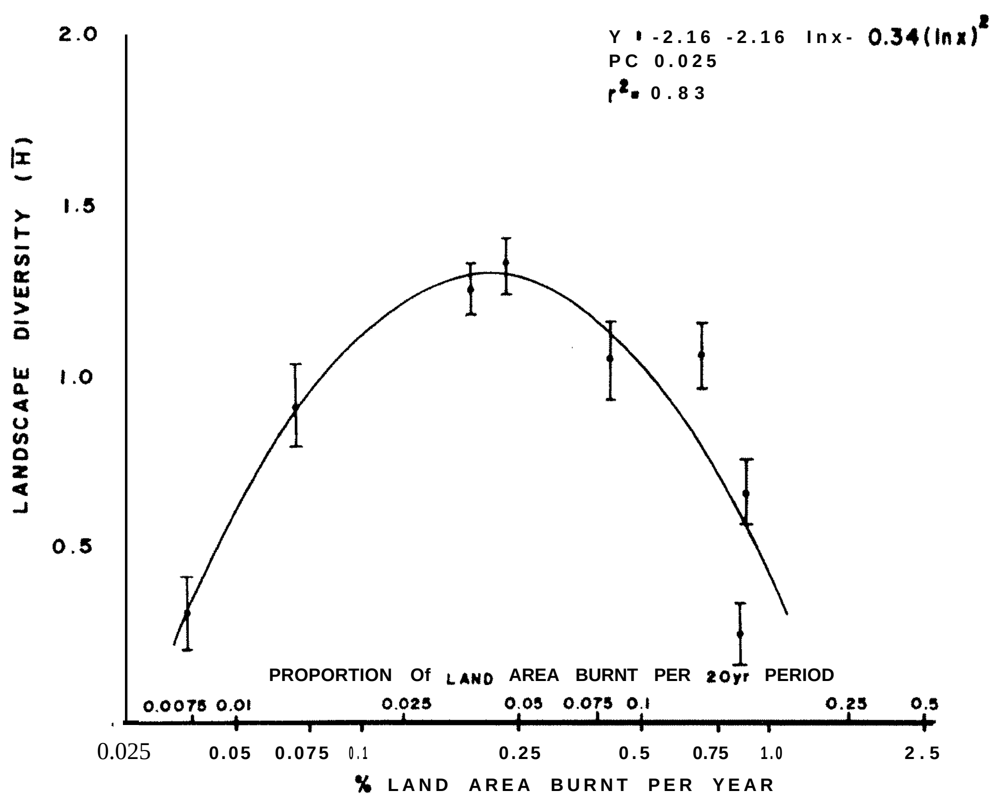


Fig. S--The relationship of landscape diversity (Shannon's H statistic) in the Northwest Ontario study area, to disturbance by forest fires (After Suffling and others 1988).

FIRE MANAGEMENT IN AN ERA OF CLIMATE CHANGE

The policy of allowing natural fires to bum while suppressing human-caused tires (if they can be recognized as suchl) rellcs on a premise that the climate that starts fires or encourages them to spread is an unmodified component of the natural system. Now, however, there are predictions that climate will be anthropogenically warmed. The amount of change is debated and the "control" situation without greenhouse warming is not clearly definable for several reasons: First, natural climatic fluctuation will certainly occur anyway. Second, we presently have much less reliable information about anthropogenic change in precipitation than we do about temperature. Third, regional forecasts from the present generation of general circulation models (GCM's) are not thought to be very accurate, and good regional analogues of GCM's will be 3 to 5 years in the making. Fourth, there is the possibility of deliberately ameliorating anthropogenic climate change, but it is generally agreed that some warming is now inevitable. The "predictions" are thus scenarios. Those who make them are under no illusion that they represent anything other than options or a range of possible futures.

Given these complications, one can only be reasonably sure that there will be more fire in many forest regions than is natural, so that a "let bum" policy will no longer promote a natural tire regime. There will be changes in the relative amounts of different habitats, in landscape diversity, in the kinds and amounts of fire ecotone, and in the spatial relationships and patch sizes of different habitats (Suffling and others 1988; Turner 1989; Turner and others 1989).

The wildland tire manager's first reaction might be to attempt to control fires to an extent that approximates the historical pristine condition (e.g. Hawkes 1980), so as to preserve a "vignette of primitive America" (Leopold and others 1963). The development of landscape-based fire ecology models (e.g. Heinselman 1973; Wright 1974; Van Wagner 1978; Johnson and Van Wagner 1985; Parks and Alig 1988) gave much theoretical support to this philosophy. However, considerable research aimed at defining a pristine condition (either today's, or an earlier era's), has often demonstrated that there has been considerable variation in fire occurrence even the last hundred years (e.g. Romme and Despain 1982; Suffling 1988). Thus, the objective of recreating the pristine has been reinterpreted as not... "trying to hold nature steady but rather maintaining natural dynamics and discouraging anthropogenic deterioration" (Noss 1987). Where one is able, however, to assemble a history that predates the end of the little ice age (1820- 1835 in many areas of North America), the variation between present and the fire past regimes is sufficiently enormous to render unworkable even Noss' interpretation of Leopold's concept.

Fire records from Northwest Ontario demonstrate this clearly, Government records provide a history of tire only since 1926 (fig. 6) and demonstrate a steady diminution of fire until the 1940's. Except in 1961, a disastrous fire year, very little forest was burnt until 1974. Fire areas increased dramatically thereafter in response to some of the warmest and driest summers of this century.

Because of the fierce, stand-replacing nature of fires in this region, it is difficult to establish a reliable quantitative history from fire scar information. Our attempts to establish a regional fire history from charcoal in lake sediments have proved fruitless because varves are not formed in the area's oligotrophic lakes. Fortunately, however, historical information from Hudson's Bay Company fur trade journals for Osnaburgh House spans the period from 1786 to 19 11, and demonstrates a massive outbreak of forest fires in the 1820's, a relatively quiescent period from 1830 to 1860, and then a steadily increasing tire incidence until the turn of the century. While much of this variation was climatically driven, we know that a large proportion of the recorded tires were started by people (Suffling and others in press), and that this activity was intimately bound up with economic, social, and attitudinal changes associated with the fur trade. This information on the temporal distribution of fires tallies well with the stand-age distributions for this area (fig. 1) that show many present-day stands dating from between 1860 and about 1900. (The data for figure 1 largely predate the post-1974 fire outbreak, so this latter outbreak does not show on these figures).

If one wished to manage the fire regime of this Northwest Ontario area, what information base should be used to identify the "natural" condition? The present high fire activity is anomalous if considered in the context of the period of government statistics from 1927 to the present, and would thus require suppression, but the current increase in fires is driven by climatic variation rather than by some change in human-set fires. (We do not yet have the advantage of hindsight, however, and cannot say whether the recent fire outbreak is just a major fluctuation, or represents the beginning of anthropogenic climate warming). Conversely, if one used the stand age distribution to establish a "natural" baseline, one would conclude that fire was virtually absent from the 1920's but was common before that. If one used the Hudson's Bay Company record (which does not allow a quantitative determination of fire frequency), one could use the low fire period at the end of the little ice age or the high fire period of the 1820's (though there is a strong suspicion that numerous large fires of the 1820's resulted from both climatic influence and human activity). Alternatively, one could pick any of the subsequent high or low fire eras.

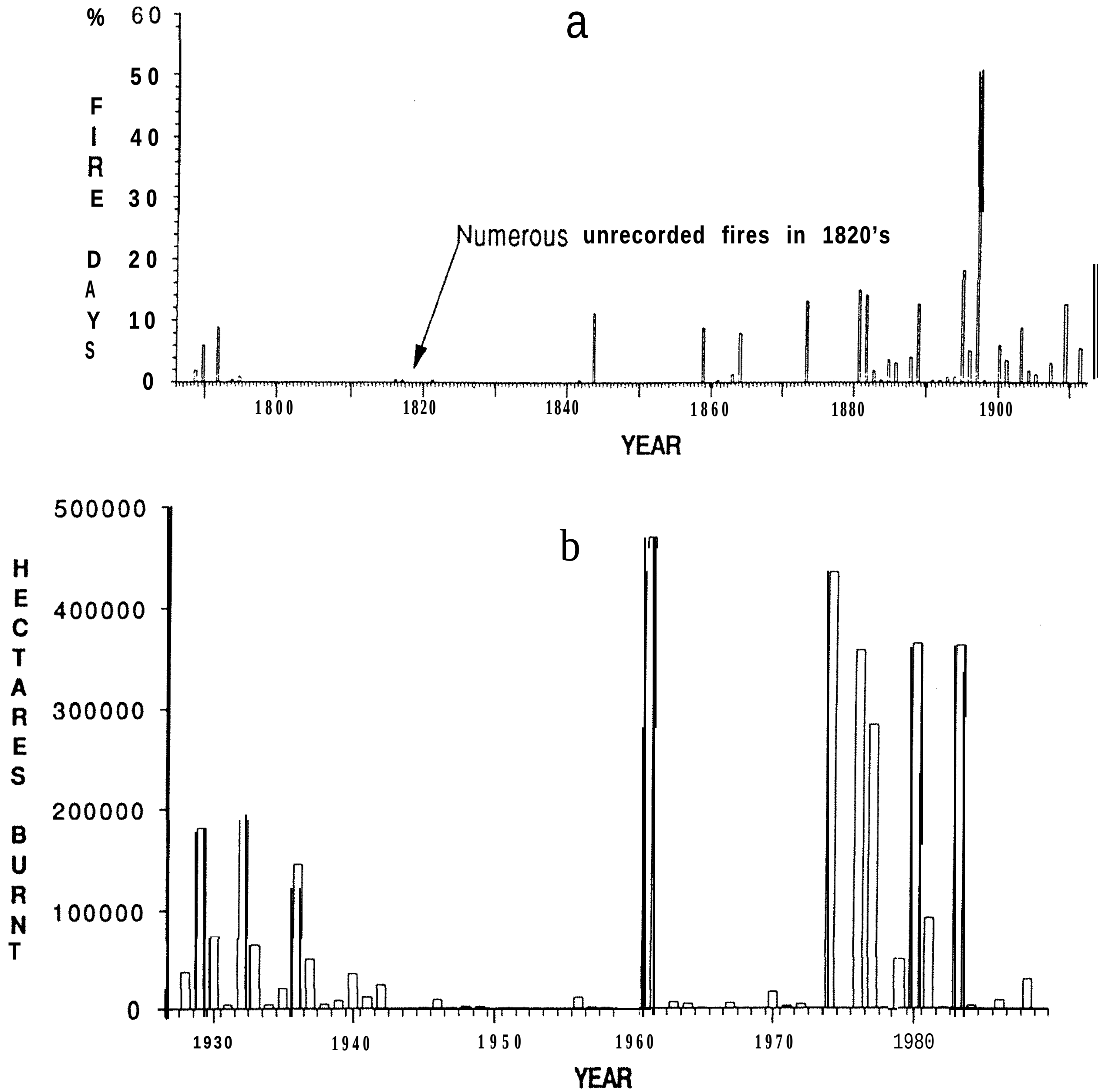


Fig. 6- Historical variation in fire occurrence in Northwest Ontario. a: Fire occurrence as represented by "fire days" (Suffling 1988) at Osnaburgh House. b: Ontario government statistics for area burnt in the Kenora District, 1927-1989 (Ontario Ministry of Natural Resources 1928-1990).

It is possible that the outbreaks of fire in Northwest Ontario follow about a SO-year cycle, with high fire eras in the 1820's, 1870's, and 1980's. There is little information for the early 1920's but oral tradition suggests that fires were frequent at that time. Data from other regions support the possibility that cyclical fire occurrence is commonplace at a landscape scale. Charcoal fragment counts in varved lake sediments in the boreal forest of Wood Buffalo National Park in Canada's Northwest Territories (MacDonald 1990 pers. comm.) imply a 100 to 300 year cycle of fire in the landscape over a 2000 year period. Similarly, the conifer forests of the Yellowstone Plateau in Wyoming have experienced long quiescent periods punctuated by major cyclic fire outbreaks about every 300 to 400 years, as in 1988 (Romme and Knight 1982; Romme and Despain 1989). Current research on the effects of spatial landscape patterns is beginning to explain these temporal variations (Antonovski and Ter-Mikhaelian 1987; Turner 1989).

In none of the cases noted above is there any indication that the major fire outbreaks are merely extremely large events in a stochastic series. In each case, fire occurrence appears to have "flipped" between high and low states without the appearance of an intermediate condition. Thus, adoption of an average fire return period would be arbitrary, and would not mimic nature. Likewise, any attempt to "fix" the landscape adopting a particular fire frequency from a high or low fire period will be unnatural.

How can the fire manager resolve this dilemma? One approach is to identify acceptable limits of variation in the disturbance mosaic over time -an ecosystem supply strategy. For instance, if one decides that it is desirable to retain some mature stands of jack pine over 100 years old, tire management policy can be tailored to protect such stands if their total area falls below a defined limit represented by a certain percentage of potential jack pine site area. Conversely, one might set prescribed bums in potential jack pine areas if the total area of jack pine under 20 years old were to fall below a defined limit. Acceptable limits could be set on the basis of the historical representation of ecosystem types in the landscape, on aesthetic or other cultural values, or on the need to preserve certain ecosystem types for their valued flora or fauna.

CONCLUSION

Fire managers should not assume, a that forest fires (or, for that matter, any other patch disturbance) will increase landscape diversity, or that they will reduce it. The effect of fire on landscape diversity depends on the current status of the landscape mosaic and, thus, on previous disturbance. Because global climate warming will increase forest fire occurrence in the boreal and other biomes, wildland fire managers should no longer assume that the lightning fire regime as non-anthropogenic. The timing and extent of increase in fire, as well as the "control" fire regime that might occur without global climate warming arc currently unknown. One might wish to maintain the status quo in wildland areas in terms of proportions of different ecosystem types. However, these proportions shift constantly over time, even at the landscape scale, in response to natural climate variation and the spatial pattern in the landscape (Antonovski and Ter-Mikhaelian 1987; Turner 1989), which links to endogenous fuel processes. Thus, one must decide what fire is to create and what to protect from tire. This can mean dctermining what minimum area of each ecosystem type should exist in the landscape. Such definitions can bc based on the status quo, on historical variation, or on culturally defined values. Sadly, in an era of climate warming the ethic of leaving nature to continue without human interference becomes illusory.

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HAZEL PISTOL EROSION PLOT STUDY ON THE SISKIYOU NATIONAL FOREST IN SOUTHWEST OREGON

William F. Hansen'

Abstract-In November 1977, small erosion plots were installed on 30, 50 and 70 percent slopes following forest management activities in southwest Oregon. Activities included clearcutting using a skyline yarding system, followed by burning to reduce logging debris and hardwood competition in 1976. Little of the soil was exposed prior to burning because one end of the logs was suspended during the yarding operation. The burning intensity was severe due to the high amounts of logging debris and the relatively dry conditions with low fuel moistures. After burning, mineral soils were exposed on over 75 percent of the area. Rainfall measured 111 and 150 inches after 3 months and 12 months, respectively. Surface runoff and erosion leaving the plots were collected and measured. Some data loss occurred due to pipes plugging or container overflow. During the first 3 months, surface runoff measured from the burned area varied from 27.6 to 33.4 inches on 30 percent plots, 35.0 to 51.6 inches on SO percent plots and 43.9 to 44.3 inches on 70 percent plots. The unburned 70 percent plots had water movement of 22.4 to 34.2 inches. Soil loss (<2mm) was 2.02 to 3.57 tons/acre on 30 percent plots, 2.18 to 5.89 tons/acre on SO percent plots and 4.04 to 18.68 tons/acre on 70 percent slopes. Plots on 70 percent slopes within the clearcut area that were not burned had erosion ranging from 1.26 to 3.09 tons/acre. Surface runoff and erosion figures after one year are also presented. The magnitude of erosion was partly due to wind-driven rains near the Pacific Ocean and the highly erosive siltstone soils of the Dothan Formation. This study was helpful in changing attitudes about the effects of burning and requiring burning prescriptions that protect soils (e.g., by burning under conditions with greater fuel and soil moistures or requiring more fuel removed during the yarding operation), Visual indicators of surface erosion and methods for minimizing or mitigating the effects of prescribed burning are also discussed.

INTRODUCTION

Relatively little information was available on surface erosion quantities following forest practices in southwest Oregon when this study was conducted on the Siskiyou National Forest (SNF) in 1977 and 1978. During that time, the SNF was a leader in developing and implementing technology to reduce environmental impacts from forest practices. Resource values and constraints were extremely high with steep slopes covered with old growth Douglas-fir (Pseudotsuga menziesii) and beautiful streams with some of the most valuable salmon and steelhead habitat in the nation.

Prior to the study, the Siskiyou National Forest had identified many sensitive environmental issues. In response to the critical issues, forest practices were being carefully scrutinized to reduce environmental impacts. Road construction was a primary concern because of its potential effects to the soil, water and fishery resources. Access roads were typically kept near ridges to avoid stream crossings and reduce surface and mass soil movement into streams. Side casting of soils during road construction was minimized or even hauled away in very steep terrain. An aggressive program to provide road surface drainage and to revegetate the bare soils adjacent to roads was also being implemented. Skyline yarding systems, which partially or totally suspend logs on steep slopes or in streamside areas, were being successfully used to reduce the logging impacts associated with conventional ground-based skidders on steep slopes.

Fire management practices were addressed after the major contributors to erosion and stream sedimentation had been identified and were being reduced. The effects of post-logging bums became a concern of watershed specialists during monitoring trips on the SNF. Observations causing concern included loss of surface organic layer, exposure of mineral soil, soil pedestals, fresh silt in streams, and turbid water during storm events. Burning practices and attitudes about burning would be difficult to change without some evidence to back up observations. The challenge to "prove it" or at least "measure it" was a necessary and reasonable request by the unconvinced majority.

The concerns about erosion following Prescribed burns would have been reduced if erosion was not measured under severe conditions. The SNF was an ideal testing ground to measure erosion in the late 1970's, and severe conditions following prescribed bums were not hard to find. The usual objectives of burning were to reduce heavy fuel loading from logging debris and to reduce competing vegetation with the next generation of Douglas-fir seedlings. These objectives were usually accomplished with hot burns, in the summer or early fall. Soil litter and the organic layer were often consumed

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To the untrained eye, overland flow and erosion were not problems because the streams usually carry little sediment. Soil erosion and water quality changes occur rapidly in response to rainfall intensity and duration. These processes are not easy to measure. Small plots were chosen to collect surface runoff and erosion realizing potential problems with variability within an area and impact from the plot edge. Advantages included the low cost, ability to collect and measure all the soil and water leaving each plot and ease with which photos could be used to show plot and sampling details.

DESCRIPTION OF THE STUDY

Location

The study was conducted on the Chetco Ranger District of the SNF in southwest Oregon, (specifically the north side of unit 2 of the Hazel-Pistol timber sale, Township 38S, Range 13W, section 28, NE 1/4 of the SW 1/4, Willamctte Meridian). The study area is about 8 miles from the Pacific Ocean in the Pistol River basin between Brookings and Gold Beach, Oregon. The aspect was generally southwest to southeast.

Background Information

Following clear-cutting using skyline yarding with one-end log suspension, the unit was burned in the fall of 1976 to reduce fuel residue and vegetation competition. Although not measured, over 7.5 percent of the mineral soils were exposed. Unusually dry conditions persisted until the fall of 1977 when the plots were installed.

The coastal landscape is typically deeply dissected by streams with boulders, bedrock or debris, which prevent further channel degradation (downcutting). Adjacent slopes often have high potential for erosion or instability due to soil materials, high rainfall, steep slopes and loss of support from channel erosion. The soils are also extremely complex due to numerous geologic changes. Average annual rainfall is over 100 inches (2.5 m) for much of the National Forest. Dry, hot summers with periodic lightning storms and burning by early natives and settlers have caused many past wildfires. In many cases, the litter layer and organic surface soils are shallow to non-existent. High decomposition rates and erosion are contributing factors. Such harsh site conditions present rcvcgetation and regeneration problems, especially on the south-facing, skeletal (gravelly) soils. The study area selected has some of the most severe conditions on the SNF and in Oregon.

Geology and Soils

The bedrock of the study area consists of bedded layers of moderately hard s&stone, massive to slightly fractured mudstone and sandstone rocks of the Dothan formation. The soils were derived from colluvial and residual material. Soils of the area are primarily thin gravelly loams on slopes over 40 percent and thick silt and clay loams on slopes under 40 percent. Slopes in the unit are locally highly dissected and range from 20 to 90 percent, The soils arc moderately unstable and highly erosive.

Climate

The temperature extremes of cold winters and hot summers in southwest Oregon are moderated somewhat due to the close proximity to the Pacific Ocean. Warm, moist air masses are cooled as they are pushed upward by the coastal mountains. At an elevation of 2600 feet and only 8 miles from the coast, the study area is subject to high wind speeds and precipitation. Winds typically blow from the west to southwest with speeds occasionally exceeding 50 miles per hour. The average annual precipitation for the study area is estimated at over 125 inches and occurs primarily between November and May. Rainfall events are usually long duration with low to moderate intensity. Temperature differences from the coast may be present when the coast is fogged in and the study area is clear. Winter freeze-thaw cycles occur with few snow events.

METHODS

Experimental Design

The sampling methods were designed to test the effectiveness of grass seeding in reducing water movement and surface erosion on an area clearcut and broadcast burned. The experimental design was a 3 X 2 factorial analysis with one replication, or 12 plots total. The factors varied were slope and grass seed. Slopes used were 30, 50 and 70 percent and grass seeding was either 0 or 7 pounds per acre.

Plot Design

The plot boundaries consisted of 2 X 4 lumber with a 2 X 8 for the upper boundary. Each plot was designed to be 112500 of an acre (17.4 square feet) and the plot dimensions varied according to the slope. Each plot was drained into 6 inch fascia gutter scraps along the lower boundary. The gutter with end caps was nailed to the wooden boundary with the gutter lip bent down about 1/2 inch. The wooden boundary with gutter was eased into the correct position on the plot and staked to the ground at several locations outside the plot. The bent gutter lip was pressed into the soil. On the outside of the wooden boundary, a small ditch about 4 inches deep was constructed and filled with concrete to provide a good seal to prevent surface water from entering or leaving. Concrete was also placed by hand above the bent gutter inside the plot to prevent water from bypassing the gutter. A small trench above each plot diverted other runoff away from the plot.

Soil and Water Measurements

Surface water from the plot drained into a 55 gallon drum using 3/4 inch black plastic tubing. However, after continuous clogging problems, 1 1/2 inch black plastic tubing was installed. Plastic tubing fittings were used to go from the gutter into the 55 gallon drum lid, A 500 ml plastic bottle was placed over the tubing outlet to collect the heavier sediment. The lighter sediment was collected in the drum with the water from the plot. The water in the drum was measured, mixed and sampled. The concentration in the sample multiplied by the volume in the drum gave the amount of sediment in the drum. Larger or heavier materials often settled out in the gutter. This material was collcctcd, oven-dried, sieved into soil (< 2mm) or large. particles (> 2mm), weighed separately and added to the estimated sediment from the drum. Large particles included rocks or pebbles, Douglas-fir cones, needles, leaves and other debris. The collection gutters were cleaned out in February 1978 and November 1978, respectively, approximately 3 months and 12 months after installation. Rainfall mcasurcmcnts wcrc madc using a Belfort recording raingage. Under $5,000 was spent to collect this information.

RESULTS AND DISCUSSION

Some adjustments in the data analysis had to be made duc to unforeseen problems. The grass failed to germinate properly within the plots and only scattered depressions had any success. Two plots were accidently located on an unburned portion of the burned unit. Another plot flooded with water and filled with sediment from an ephemeral microchannel which had not been diverted away from the plot. Lost data from plugged tubing and drum overflows from large storm events posed additional problems. The statistical efficiency of the factorial plot design was lost with these problems, but the information collected provided valuable insight to surface erosion and water movement after typical forest practices of the time.

Information was intensively collected on the study plots from November 11, 1977 to February 22, 1978 (104 days).

Fourteen separate rainfall events were identified during this period, ranging from 1.1 to 18.4 inches. Rainfall totaled 11 l inches (2.8m) Average storm intensities were less than 0.25 inches per hour, while peak 2-hour intensity reached 0.82 inches per hour. During the first year, over 143 inches of rainfall was measured and about 7 inches was estimated, for a total of 150 inches (3.8m).

Figure 1 presents data to compare rainfall and runoff by plot slope and treatment for only those dates when collectors did not plug or overtlow. There were a few discrepancies when runoff exceeded rainfall (the raingage opening was level while the plot openings were not) during individual storms. Whether the raingage caught less of the windblown rain, or the plots caught more, is not known. The amount of overland flow from the plots was alarming and provided strong evidence that surface erosion mechanisms existed. The following table presents the measured rainfall and runoff summarized at two points in time over a year. Since the collectors occasionally plugged or overflowed, estimates of runoff are low by approximately 10 percent for all treatments, except the 70B treatment data are 30 percent low.

Time Period	Time (months)	Rainfall (in)	Average Plot Runoff (in) by Treatment 308	SOB	70B	7ou
11/77-2/78	3 4	111	30.7	41.8	44.1	28.3
11177-1 1178	12	150	57.8	71.5	63.8	48.8

(30, 50, 70 = % slopes, B = Burned, U = Unburned, 1 in = 2.54 cm)

Figure 2 presents the measured erosion by treatment for the one year. Due to collection problems, some data was probably lost. The amount lost is believed to be much less than the amount of runoff lost because the gutters were effective sediment traps when pipes clogged. The following table summarizes soil and total erosion after 3 months and one year.

Time Period	Time (months)	Rainfall (in)	Total Erosion (tons/ac) by Treatment 30B	SOB	70B	7ou
11/77-2/78	3.4	111	2.5(2.7)	4.6(5.8)	11.4(20.0)	2.2(3.8)
11/77-1 1178	12	150	3.8(4.3)	6.2(9.1)	12.8(23.2)	2.7(6.0)

(same symbols as previous table, 1 ton/acre ≈ 2240 kg/ha)

Poor record keeping after November 1978 made the data collcctcd after one year questionable. However, visual indications of continued surface water and erosion occurred because vegetative cover was slow to develop. In 1983, seven years after the bum, signs of accelerated erosion of litter, mineral soil and rock fragments on the 70 percent unburned plots were disturbing.

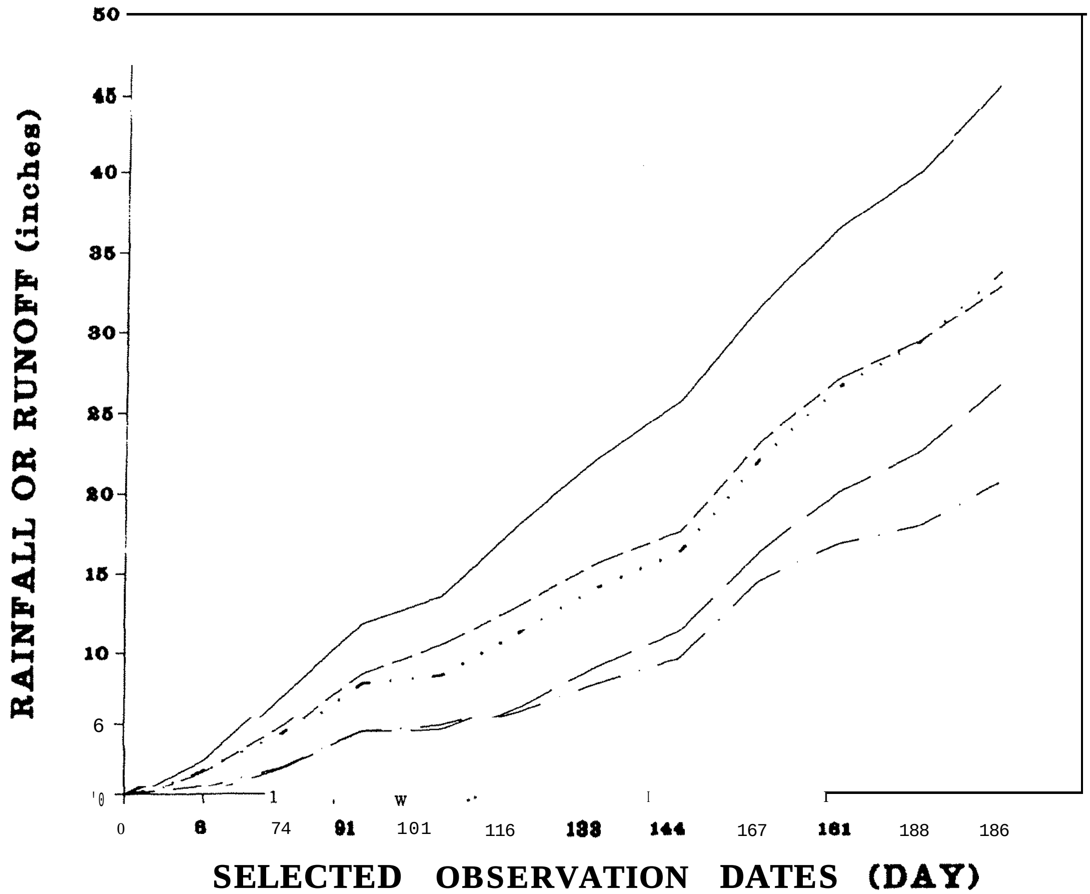
Several types of erosion processes were observed on the area, including raindrop, sheet, and rill erosion. These were detected or inferred by close observations during intense rainfall periods or inspection during the study period on the erosion plots.

Raindrop erosion occurred when large amounts of kinetic energy were expended on the soil surface by falling raindrops. In an undisturbed forest, vegetation and litter absorb this energy. Soil particles exposed during management activities are susceptible to detachment by the raindrop impact. The wind driven rain supplies additional velocity and energy. Raindrop erosion may clog surface pores thereby reducing infiltration. Soil pedestals formed under the protection of pebbles or wood were another indicator of soil remaining in place when shielded from raindrop impact.

Sheet erosion occurred as thin layers of surface materials were gradually removed. This was noticed as a fine root network was eventually exposed on the 30 percent plots. Larger roots and gravel were also exposed on the steeper sites as tines were removed. Soil delivery to the collection device was diffuse and defined water movement was difficult to observe.

Rill erosion was apparent during one heavy rainfall event on plot 6 (70 percent bum slope). Microchannels no more than an inch in cross section developed. Soil was being removed by running water of sufficient volume and velocity to generate

Figure 1
CUMULATIVE RAINFALL AND RUN[redacted]
BY TREATMENT FOR SELECTED DA[redacted]



LEGEND

RAINFALL

RUNOFF 30B

RUNOFF 50B

RUNOFF 70B

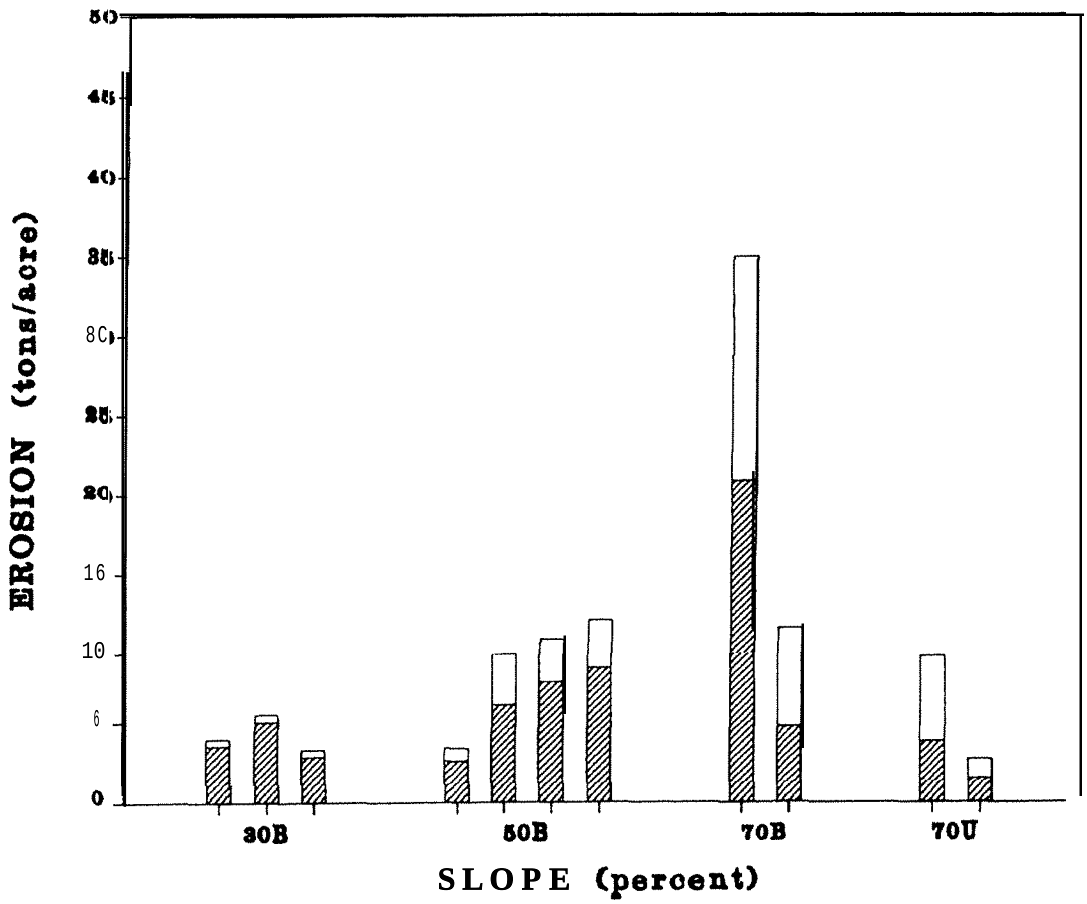
RUNOFF 70U

B - BURNED

U = UNBURNED

30, 50, 70 = PERCENT SLOPE

Figure 2
SOIL EROSION BY SLOPE AND TREATMENT 11/77 TO 11/78



LEGEND

COARSE >2mm — B = BURNED

SOIL <2mm — U = UN BURNED

30, 60, 70 = PERCENT SLOPE

cutting power. As the soil particles eroded away, pebbles and small rocks could be seen and heard tumbling down the slope as they were moved by water and gravity.

Other visual observations of erosion were made outside of the erosion plots, within the harvest unit. Examples of exposed roots could be found on all slope classes. Fine roots were often exposed on 30 percent slopes and larger roots were sometimes exposed on steeper areas. These roots were not tire scarred and were apparently buried at the time of the prescribed bum. Fire scars on trees were occasionally above the soil surface, indicating measurable soil erosion. An increase in surface rock content was noticeable on slopes exceeding 50 percent or more. Soil deposition occurred in surface depressions, above woody debris and in pool areas within the stream channels. Soil protected from the bum under large woody debris or rocks had about 1/2 to 1 inch of litter and organic soil. Small rocks were suspended on soil pedestals. Streams would rapidly change in turbidity and sediment loads in response to rainfall intensity.

Part of the results include management's reaction to information collected on the study area. This study was an eye opener to forest managers, who previously perceived that surface erosion and overland flow effects were negligible following prescribed burning. After some initial deliberations and reactions to change, adjustments were made to strengthen the prescribed burning program.

Prescribed burning plans were adjusted to protect the surface soil and organic layer, including its ability to take up and store water. Burning is accomplished when the duff layer is moist (usually a few days after a soaking rain in the spring). Directional falling of the old growth trees on steep slopes reduced breakage of logs, prevented high debris loads in streams and increased the tree utilization. Required yarding of unutilized material (YUM), is another method to reduce the logging waste and fire intensity.

Burning specialists began to receive additional training in measuring weather, fuel moisture, fuel load, and flame height values to reduce tire impacts to soil resources. Strategies such as helicopter lighting also reduced fire intensity. Monitoring post-burning conditions also help evaluate the bum. When areas are accidently burned too hot, grass seeding with fertilization helped mitigate burning effects. The grass species mixture can help provide immediate cover needs with soil improvement and wildlife benefits.

CONCLUSIONS

This study was undertaken to document the presence or absence of surface runoff and erosion following typical clearcut and prescribed burning practices in southwest Oregon during the late 1970s. Severe conditions were chosen to lest whether surface runoff and erosion were valid concerns. The effects of using grass seeding as possible mitigation was not possible because much of the grass was apparently lost due to erosion. During the intensive 104-day study of the burned area, 111 inches of precipitation occurred, producing surface runoff in excess of 30 to 50 inches and soil erosion from 2 to over 18 tons/acre. In contrast, the steep unburned areas produced substantially less runoff (20 to 30 inches) and soil erosion (1 to 3 tons/acre).

The results of this study convinced forest managers that some adjustments in prescribed burning practices were needed to protect soil, water and tishery resources. Resetting burning objectives to protect these resources was the first step. Methods designed at minimizing potential impacts to both onsite resources, such as soil productivity, and offsite resources such as downstream water quality, fishery habitat and air quality, were included in prescribed burning plans. Practices were implemented to reduce fuel loading through greater utilization and adjust burning intensity to protect soil resources.

Burning is a useful and necessary tool in forest management, but it can cause unacceptable adverse impacts if not properly applied. With adequate planning, timber harvest and burning practices can be adjusted to achieve soil and water resource objectives, with good success at residue abatement and temporary vegetation control. Soil, slope, climatic and historic land use factors should be assessed to help evaluate the erosion potential of an area prior to burning. When burning under conditions with severe erosion potential cannot be avoided, aggressive efforts to revegetate exposed mineral soils are needed.

Despite the limited application of small plot studies, they are helpful in this case to identify and measure site specific processes that arc difficult to measure on a large scale. However, several factors should be considered before applying the results of this study to other conditions. The presence of abundant wind-blown rain, highly crodiblc soils, steep slopes and exposed mineral soil from a combination of forest logging and burning practices were all important contributing factors in the severity of the study results.

ACKNOWLEDGEMENTS

Foremost, I recognize and thank Malcolm Drake, Hydrologic Technician, for his valuable assistance and hard work in the installation and service of the plots. Other USDA-Forest Service employees that I want to recognize include John Millet (retired), encouraged and supported this study; Harvey Timeus, Joe Waller, and Ed Gross, Chetco Ranger District, helped service the erosion plots; Roy Meyer and Mike Amaranthus, described the soils; Bob Ettner, encouraged reporting of these findings; Luis Mundo helped with the computer graphics and Lynda Hansen helped with typing and editing.

THE SIGNIFICANCE OF FIRE IN AN OLIGOTROPHIC FOREST ECOSYSTEM

Frank S. Gilliam[1]

Abstract-Past and present climate conditions have interacted with soil development to result in distinctly oligotrophic (nutrient-poor) conditions in many southeastern U. S. Coastal Plain ecosystems. Fire historically has been an important abiotic component in these systems favoring the dominance of plant species which require fire for successful regeneration and growth. This study examined the role of periodic fire in several components of an oligotrophic lower Coastal Plain pine flatwoods ecosystem. Except for some loss of nitrogen (N) from the forest floor, experimental burns had slight effects on nutrient loss from the system. Fires volatilized an average of 24 kilograms N per hectare. Much of this loss is balanced by annual net (precipitation input minus stream flow output) ecosystem increases in N. Fire increased nutrient availability in the soil, an increase which coincided with increases in the biomass and species diversity of the herbaceous layer. Thus, fire is important in maintaining nutrient availability in these nutrient-poor soils. Evidence presented in this study support the idea that pine flatwoods are especially limited by phosphorus (P) and potassium (K) availability and that fire significantly increases available levels of P and K in the soil. Fire is considered here a characteristic property of the ecosystem, one which integrates all hierarchical levels of organization of the system.

INTRODUCTION

General hypotheses concerning the importance or role of fire in ecosystems appear difficult to make, given the great variety of ecosystem types wherein fire occurs at a sufficient frequency to be considered a component of the system. It is a reasonable hypothesis, however, that a predominant role of fire, regardless of ecosystem type, is to increase or maintain the availability of an essential (usually growth-limiting) resource, either energy (sunlight), nutrients, or water. The specific role of fire would be determined by which resource, or combination of resources, is limiting in a particular ecosystem. For example, in tallgrass prairie, which has nutrient-rich soils, but experiences substantial build-up of plant detritus which intercepts both light and water, fire appears to be important in maintaining availability of energy and water, but not nutrients.

The Coastal Plain of the southeastern United States has long been a region of great interest to fire ecologists, as evidenced by earlier reviews by Wells (1943) and Garren (1943), and more recently by Christensen (1981). This is a region wherein past and present climatic factors have influenced soil development in a way that resulted in oligotrophic (nutrient-poor) conditions (Gilliam 1990). Such conditions have, in turn, favored the dominance of plant species, such as pines, which require fire for successful reproduction and growth. These species, adapted to low soil fertility, produce acidic, low-nutrient detritus, thus maintaining oligotrophic conditions, a scheme that represents co-development of biotic and abiotic components of the ecosystem (Jenny 1980).

The main objective of this study was to examine the effects of fire on several components of a pine flatwoods ecosystem of the lower Coastal Plain of South Carolina. These results were used to address the hypothesis that fire, as an integral part of the system, serves a significant function in increasing nutrient availability. A second objective of this study is to look at the specific role of fire at each hierarchical level of organization of the system (ecosystem, community, and population) to address the contention that fire is "incorporated" (sensu O'Neill and others 1986) at the level of the ecosystem.

In addition to the presentation of previously unpublished data, this paper provides a brief synthesis of several aspects of the Santee Watershed Study. These include studies on the effects of fire on water quality (Richter and others 1982, 1984), precipitation chemistry (Richter and others 1983), soil nutrients (Gilliam and Richter 1985, 1988; Gilliam 1990), and effects of fire on herbaceous layer vegetation (Gilliam and Christensen 1986; Gilliam 1988).

MATERIALS AND METHODS

Study Site

The study was carried out on Watershed 77 (WS77) of the Santee Experimental Forest. This forest is within the Francis Marion National Forest in South Carolina, approximately 50 kilometers north-northwest of Charleston (33°N, 80°W). WS77 is 165 hectares in area and is typical of lower Coastal Plain pine flatwoods ecosystems. Topographic relief of this and other first-order watersheds of the region varies by 5.5 meters. Prior to the start of the study, WS77 had not been burned for **40** years.

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WS77 soils are clayey, mixed, thermic, vertic Aquults of the Bayboro, Bcthera, Carolina, and Wahee series. Although these soils are of mixed minerologies they arc gencrally derived from old and highly-weathered secondary sediments of an alluvial origin and from montmorillonitic deposits of a marine origin. The soils tend to be extremely acidic, infertile, and low in weatherable minerals (Gilliam 1990). Each of the four series are described as very strongly acidic in reaction to at least 130 centimeters (Hatchcll and Henderson 1976).

Vegetation of WS77 is characteristic of Coastal Plain pine flatwoods. The dominant ovcrstory species were pines, loblolly pine (Pinus taeda L.--75 percent of the overstory basal area) and longleaf pine (P. palustris Miller--l7 percent). Other canopy species were sweetgum (Liauidambar stvraciflua L.--4 percent), black gum (Nyssa svlvatica Marshall--3 percent), and shortleaf pine (P. echinata Miller--Z percent). Dominant shrub species included nearly equal mixtures of wax myrtle (Myrica cerifera L.), gallberry (Ilex glabra (L.) Gray), and lowbush blueberry (Vaccinum tenellum Aiton.). The herb layer was dominated by broom sedge (Andropogon virginicus L.), with switch cane (Arundinaria gigantea (Walter) Muhl.) abundant along seeps and stream channels.

The climate for this region is classified as humid mesothermal (Trewartha 1954), with mild winters and warm, moist summers. Mean monthly minimum temperatures for January and July (extreme months) are 4 and 20°C, respectively, whereas mean monthly maximum temperatures are 12 and 32°C. Seasonal patterns of precipitation, stream flow, and cvapotranspiration for WS77 are shown in fig. 1. Precipitation averaged 135 centimeters annually, while stream flow averaged 35 centimeters annually. Precipitation typically exceeded evapotranspiration throughout the year (fig. 1).

Sampling

Precipitation and Stream Flow

Nutrient inputs were estimated from weekly precipitation sampling and chemical analysis. Precipitation was sampled with a network of nine bulk collectors and volume was determined directly using a method described in Thicssen (1911).

Similarly, nutrient outputs were estimated from chemical analysis of weekly stream flow grab samples taken behind the calibrated weir at WS77. Weekly flow volume was calculated from continuous stream height monitoring. Daily flow volume was calculated from these readings by U. S. D. A. Forest Service Computations. All sampling (precipitation and stream flow) was carried out for 6 years.

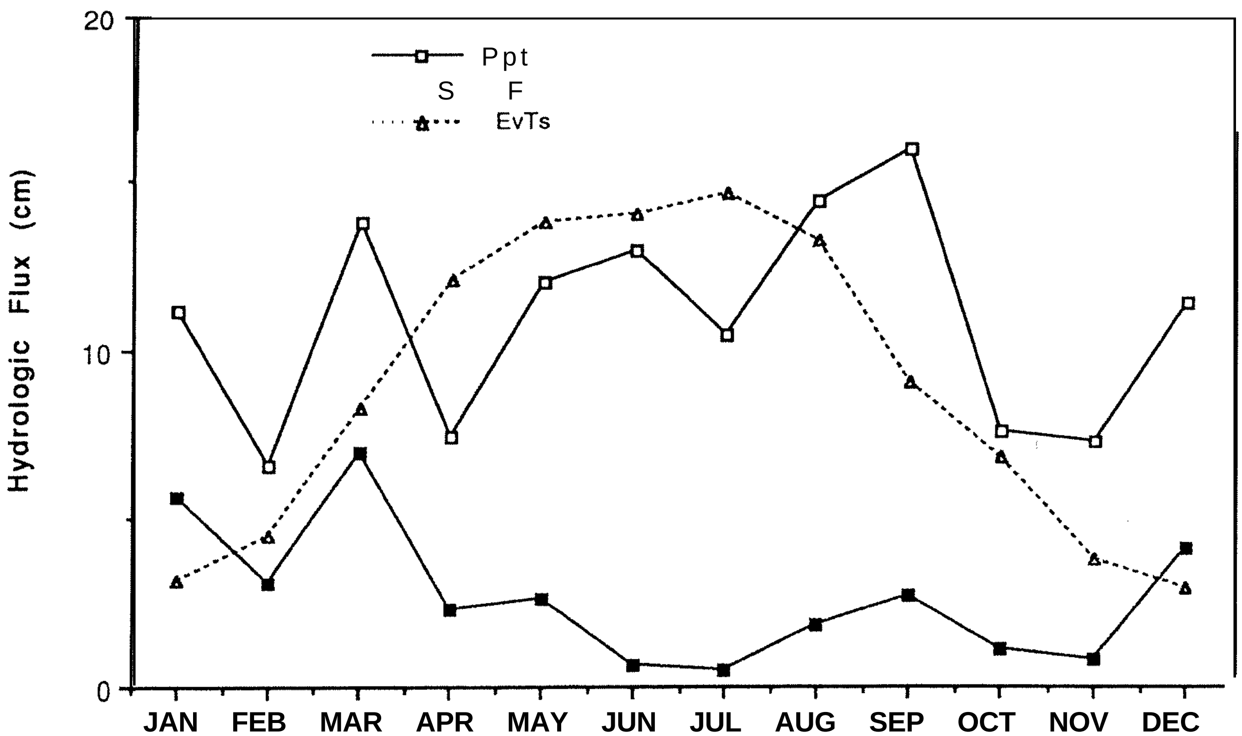

Figure 1 .-Mean monthly fluxes of precipitation (Ppt), stream flow (SF), and evapotranspiration (EvTs) for WS77

Fire Effects

WS77 was divided into 20 compartments of approximately 8 hectares. Fires were administered as summer or winter prescribed fires, largely as backing tires. A total of nine fires administered during this study. See Gilliam and Christensen (I 986) for a complete description of compartments and fire treatments. Briefly, nine compartments receiving either winter-only fires, winter and summer fires, or no fire (control) were chosen randomly from the 20 compartments of the watershed.

Effects of tire were estimated from sampling (usually both before and after the fire) within 10 IO-meter x lo-meter plots in each compartment. Forest floor and mineral soil were sampled both before and after the bum. Forest floor was sampled with a 14.7-centimeter diameter litter cutter; mineral soil was sampled with a 2.0-centimeter diameter soil corer to a depth of 20 centimeters and cores were divided into O-5 centimeters, 5-10 centimeters, and 10-20 centimeters depths. Five subsamples taken randomly within each plot were compositcd for each sample type.

Overstory and shrub layer vegetation were sampled once prior to burning. All stems > 0.6 centimeters diameter (at 1.5 meter in height) within each plot were identified and measured, either for diameter (trees) or canopy cover (shrubs).

The hcrbaceous layer, defined as all vascular plants ≤ 1 meter in height, was sampled in five of the 10 plots in each compartment to determine 1) herb layer cover and biomass, 2) species richness and diversity, and 3) nutrient content. Herb layer cover was estimated non-destructively in two 0.5 meter x lo-meter transects in each of the live sample plots. The transects were subdivided to yield 10 I-square meter subplots. Per cent cover was estimated visually for each species in all subplots. Biomass was estimated by harvesting three separate SO-meter x 0.5meter transects. These transects were subdivided into 7.5 0.5meter x 2-meter subplots.

A separate design was used to determine nutrient concentrations of herb layer vegetation in burned and unburned areas. Ten pairs of sample plots were established in the topographic extremes of WS77, five in upslope areas and live in lowland areas. One plot of each pair was burned and the other was left unburned. Herb layer vegetation was sampled by harvesting all above-ground parts within the two transects as described previously. All herb sampling (cover estimates, biomass harvests, and nutrient analysis harvests) was carricd out in the summer.

Analyses

Precipitation and Stream Flow

Precipitation and stream flow were analyzed for pH with a glass electrode. Metal cations (Na’, K^+, Ca^{++}, Mg’ $^+$) were determined with atomic absorption spectrophotomctry (Isaac and Kerber 1971). Ammonium (NH4’) was determined by isocyanurate colorimctry (Reardon and others 1966), NO_3^- by Cd reduction and azo-dye colorimetry (APHA 1976). PO_4^{3} by molybdenum blue colorimetry (Mehlich 1953), and SO,’ by turbidimetry (Schlesinger and others 1982).

Mineral Soil

Samples of mineral soil were air-dried and ground in a hammer mill to pass a 2-millimeter screen. Measured samples of about 10 grams each were extracted with a dilute double-acid solution at a 1:5 soil/solution ratio according to Mchlich (1953), a method established for acid, clay soils. Extractable elements were determined as described above.

Herb Layer Vegetation

Harvested herb layer material was oven-dried and ground in a Wiley mill. Plant tissue was digested using a H_2SO_4-H_2O_2 method (Lowther 1980) and analyzed for Ca, Mg, K, N, and P as described above.

Data Analysis

Fire effects on soil were tested using t-tests to compare pre-bum soil pH and nutrient cation concentrations and those of post-bum soils. T-tests were also used to test the effects of fire on plant tissue nutrient concentrations by comparing burned and unburned means. In each case the level of significance was $p < 0.05$. Linear regression analysis was used to generate a model relating herb layer cover to biomass The level of significance was $p < 0.01$ (Zar 1974).

RESULTS AND DISCUSSION

Ecosystem-Level Effects of Fire

Although nutrient budgets arc somewhat incomplete in this study, the components studied provide reasonable estimates of total nutrient flux. For example, soil surveys suggest minimal deep seepage loss because of poorly drained throughout WS77 (U.S.D.A. 1980). Denitrification should also be minimal, due to low NO; production in these extremely acidic soils. Finally, N fixation is probably low because of the low frequency of legumes in the forest (Gilliam and Christensen 1986) and because non-symbiotic N fixers are generally rare in acidic forest soils (Alexander 1977). Thus, input/output data may be strongly indicative of the nutrient status of the

Table 1. **Input-output budgets for** cations in precipitation and stream flow for WS77. Data represent averages **from 1976- 1982.**

Input-Output	H^+	Na^+	K^+	Ca^{++}	Mg^{++}	NH_4	NO_3	$SO_4^=$	Cl	$PO_4^{\equiv}$
	keq/ha/yr									
Precipitation	0.54	0.27	0.03	0.26	0.13	0.06	0.12	0.50	0.45	0.01
Stream Flow	0.05	0.49	0.03	0.37	0.22	0.01	0.00	0.51	0.61	0.01
Net (I-O)	+.49	-.22	0	-.11	-.09	+.05	+.12	-.01	-.16	0

ecosystem. Table 1 shows precipitation and stream flow nutrient budgets for the entire 6-year period of the study. Hydrogen ion was greatly conserved by the system, with precipitation H^+ inputs exceeding stream flow outputs by an order of magnitude. Also conserved were NH_4^+ and NO_3^-. Although such patterns are not conclusive, these data suggest that N, commonly limiting in forest ecosystems, may be a limiting nutrient in this forest.

There were net annual outputs of Na^+, Ca^{++}, Mg^{++}, $SO_4^=$, and Cl^- over the study period (table 1). Although many of the soils of this region were derived from highly-weathered sediments of an alluvial origin, net outputs of these ions indicates that, for WS77, further weathering is taking place and that these parent material sediments were largely of a marine origin.

None of the nine fires in the six years of the study had any significant effect on stream flow nutrient output (Richter and others 1982). The main loss of nutrients due to fire was an average volatilization of 24 kilograms N per hectare from the forest floor (Richter, and others 1984). Assuming a tire cycle of 5 to 7 years (Christensen 1981), however, this loss is balanced by annual accumulations of inorganic N (2.4 kilograms per hectare per year--calculated from table 1) and organic N (approximately 2 kilograms per hectare per year--Richter 1980) from precipitation.

Nutrient budgets were balanced for K and P (table 1), suggesting strongly that K and P (in addition to N) may be grow&limiting in these soils. As discussed in Gilliam (1988), this contention is supported further by comparisons of nutrient concentrations in herb layer vegetation from similar and contrasting ecosystems (table 2). Among these sites, including hardwood forests and other conifer forests, K, N, and P concentrations were typically lowest for herb layer vegetation from WS77 (table 2).

Table 2. **Herbaceous Layer nutrient concentrations for various sites.**

Site/Study	K	Ca	Mg	N	P
	%				
Eastern Illinois hardwoods/ Peterson and Rolfe (1982)	3.79	1.17	0.42	2.32	0.36
Northern hardwood forest/ Siccama, and others (1970)	3.18	0.74	0.33	2.38	0.18
Northeast Minnesota/ Grigal and Ohmann (1980)	3.25	2.28	0.50	1.38	0.34
Central New York State/ Bard (1949)	3.01	2.00	----	1.93	0.21
Boreal forest/ Gagnon, and others (1958)	0.51	0.81	0.24	----	0.19
Lower Coastal Plain/ Garten (1978)	0.60	0.85	0.16		0.18
Coastal Plain flatwoods/ Gilliam (1988)	0.84	0.77	0.20	1.19	0.06

Nutrient Availability and Uptake

The effect of fire on extractable soil nutrients was minimal and varied with season of bum (table 3). Summer burns seemed to have little influence on soil nutrients, except for a significant decrease in extractable NH_4^+. For winter burns, howcvcr, there were significant increases in pH and extractable K^+, Ca^{2+}, and NH_4^+. Although data for extractable P are not shown here, increases in extractable P in these soils in response to fire has been demonstrated (Gilliam 1983). Therefore, there is an indication that tire may increase availability of limiting nutrients.

Gilliam and Christensen (1986) summarized the response of herb layer cover and species richness of WS77 to fire. They sampled nine randomly chosen compartments representing six tire treatments, including winter- and summer-burned compartments and unburned control compartments. They found that only (but not all) winter fires had appreciable effects on the herb layer. Thus, it should be stressed that, depending on the ecosystem component being studied, fire effects may be seasonal and highly variable. Furthermore, such variability itself can have great significance on the level of the ecosystem (Christensen 1981). For the purpose of comparison, specific results for a particular winter fire will be presented in this paper.

Tissue nutrient concentrations for herb layer vegetation were significantly ($p < 0.05$) higher in burned plots than unburned plots for K, N, and P (fig. 2). There were no significant differences for Ca and Mg. This pattern suggests that tire may increase the availability of K, N, and P.

The relationship of herb layer cover and harvested biomass for each species in the three harvest transects is shown in fig. 3. This relationship yielded the equation

$$y = -0.03 + 1.81x \qquad (1)$$

where y is herb biomass in grams per square meter and x is herb cover in per cent. The correlation coefficient was 0.94 and was significant at $p < 0.01$. The relationship is based on mean values for individual species. Thus, given the highly significant correlation, equation (1) can be used to estimate biomass for individual species in plots of the burned and unburned compartments. Biomass was summed for all species in each plot to yield total herb layer biomass per plot.

Average cover was significantly ($p < 0.05$) higher in the winter bum plots compared to the control plots (37 percent vs. 16 percent, respectively; table 4). Using equation (1) for each individual species in these plots, this difference translated to a greater than two-fold increase in herb layer above-ground biomass (65 grams per square meter versus 28 grams per square meter).

Table 3. T-test comparisons of pre- vs. Post-bum soils at different depths and seasons of burning.

Summer burn

Depth/Treatment	pH	K^+	Ca^{++}	Mg^{++}	NH_4
	-----	-----	µeq/g	-----	-----
0-5 cm/Pre-burn	4.38	0.7	12.2	5.5	1.1
0-5 cm/Post-burn	4.35	0.7	11.0	5.4	0.6*
					0.7
5-10 cm/Post-burn	4.48	0.3	6.6	4.1	0.2*
					0.4
10-20 cm/Post-burn	4.65	0.2	5.6	4.7	0.1*

* indicates significant difference ($p<0.05$) between pre- and post-burn means

Winter burn

Depth/Treatment	pH	K^+	Ca^{++}	Mg	NH_4
	-----	-----	µeq/g	-----	-----
		0.9		.	0.7
0-5 cm/Pre-burn	4.16	1.1*	4.	25	0.9*
0-5 cm/Post- hum	4.26*		7.3*	3.0	
	4.35				0.3
5-10 cm/Post-burn	4.45*	0.5	3.4	2.2	0.4*
10-20 cm/Pre-burn	4.48	0.4	3.6	2.8	0.3
10-20 cm/Post-burn	4.58	0.3	2.6	2.3	0.4*

* indicates significant difference ($p<0.05$) between pre- and post-burn means

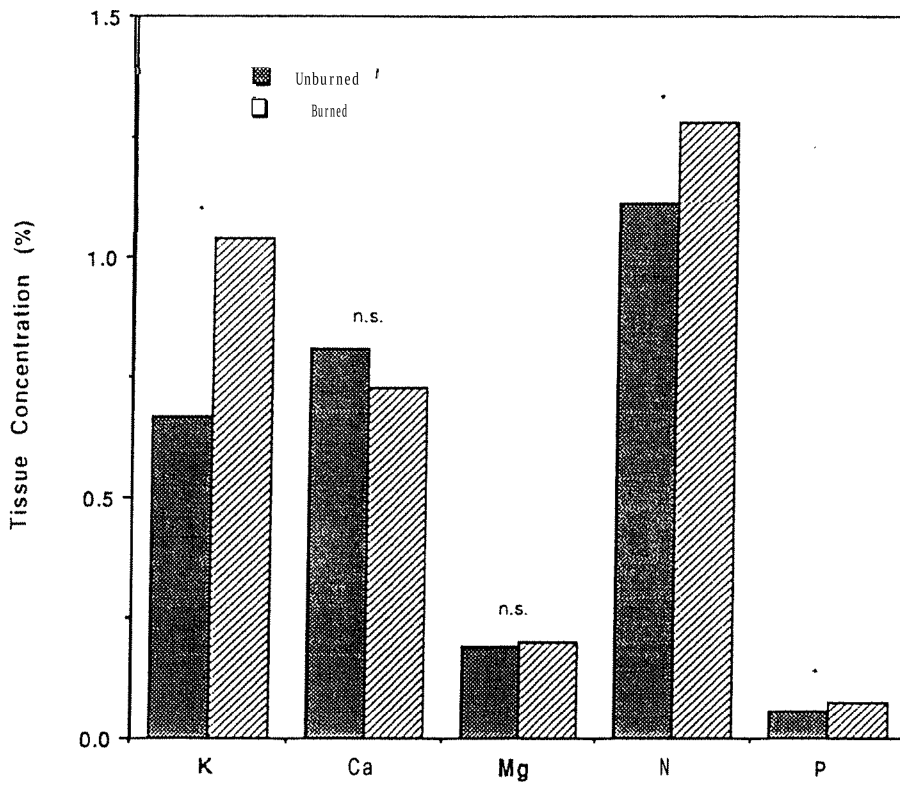

Figure 2.-Nutrient concentrations of burned and unburned plot herb layer vcgctation. *Indicates significant diffcrence between burned and unburned treatments at $p < 0.05$.

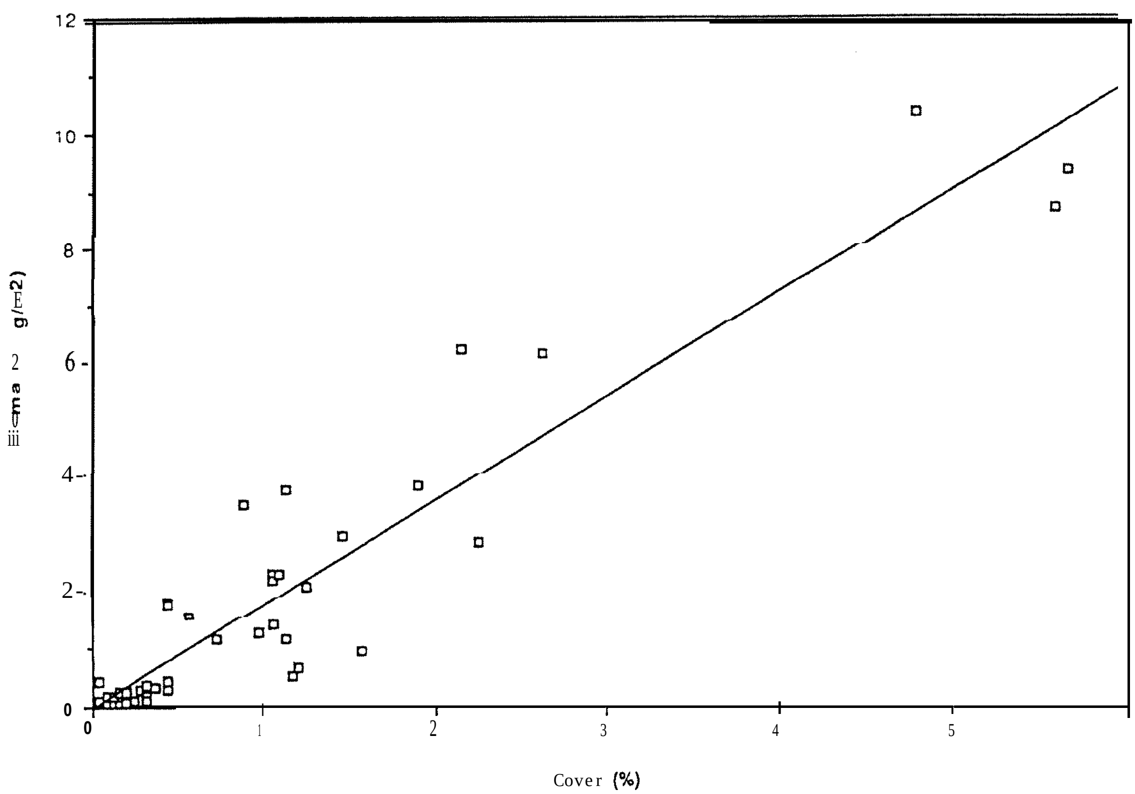

Figure 3.-Relationship of herb cover and harvested herb biomass for WS77. Each point represents average biomass and cover values for individual species. See text for equation.

Table 4. Herbaceous Layer cover, biomass, species richness, Shannon-Weiner diversity, and nutrient content for burned and unburned plots of WS77. Error vatues are one standard error of the mean.

Treatment	Cover (%)	Biomass (g/m^2)	K	Ca	Mg	N	P	Diversity	Richness (spp./plot)
			------	------	kg/ha	------	------		
Control	16, 2±2 7	28 ± 3 4 7	1.9	2.2	0.5	3.1	0.2	1.95±0.15	. ± .
Winter bum	36.7±3.7	65.1±6.3	6.8	5.0	1.3	8.3	0.5	2.50±0.10	29.5±1.9

Herb layer nutrient content was approximated by applying the appropriate nutrient concentration data from fig. 2 to unburned and burned herb layer biomass means in table 4; i.e., "burned" K, N, and P values from fig. 2 were used with "winter bum" biomass from table 4 and "unburned" values in fig. 2 were used with "control" biomass. Since fire did not significantly influence Ca and Mg concentrations, overall mean values from fig. 2 for these nutrients were used with mean biomass values from table 4.

Not surprisingly, using this method, increases in herb layer nutrient content were especially pronounced for K, N, and P. These increases were >3.5-fold, 2.7-fold, and >2.5-fold for K, N, and P, respectively (table 4).

It merits repeating that these degrees of differences, whether for herb cover, biomass, or nutrient content, are not indicative of all fires in this ecosystem, since some fires (especially summer tires) had no appreciable influence on the herb layer. These data, thcrcfor, provide a meaningful comparison representative of the potential effects of fire in this system.

Community-Level Effects of Fire

Although the major emphasis of much of this work has been on ecosystem-level effects of fire, the hcrbaceous layer is also useful in assessing the effects of Iirc on the level of the plant community, especially with respect to effects on species diversity and composition. Herb layer species diversity was measured for each plot in winter bum and control compartments as the Shannon-Weiner Diversity Index (H), using the equation

$$H = -\sum_{i=1}^{n} [p_i \cdot \ln(p_i)] \qquad (2)$$

where p_i is the decimal fraction of individuals of the ith species and a is the total number of species.

Fire significantly increased species diversity of the herb layer for this particular winter bum (table 4), a response typical for other winter tires of WS77 (Gilliam and Christensen 1986). The value of H reflects both numbers of species present as well as their relative importance, measured here as relative cover. Thus, much of the increase in the diversity index was from a significant increase in species richness, from 17 species per plot in control compartments to 30 species per plot in winter bum compartments (table 4).

In addition 'to increasing the numbers of species in burned plots, fire altered species composition as well (table 5). Grass species in particular increased in importance in burned areas. Indeed, for the species listed in table 5, tire did not so much alter which species were important as it altered species cover, on both an absolute and a relative basis.

Table 5. Important species for the herbaceous layer in burned and unburned plots of WS77. Nomenclature follows Radford, and others (1968).

Control		Wmter burn	
Species	Relative Cover (%)	Species	Relative Cover (%)
Lonicera ianonica	16.3	Andropogon virginicus	21.4
Andropogon virginicus	15.2	Liquidambar styraciftua	8.5
Ilex gtabra	12.1	Vaccinum tenellum	5.9
Vaccinum tenellum	8.8	Vitis rotundifolia	5.8
Myrica cerifera	7.6	Vaccinium elliottii	5.4
Liquidambar styraciflua	6.5	Rubus betulifotius	5.1
Rubus betulifolius	4.3	Ilex glabra	4.0
Pinus taeda	2.7	Mvrica cerifera	3.2
Mitchella repens	2.3	Festuca elatior	2.7
Vitis rotundi folia	2.1	Lonicera japonica	2.7

Population-Level Effects of Fire

Fire will affect populations of plant species differentially, depending on the species' life history characteristics and resource requirements. Many species in southeastern Coastal Plain ecosystem not only respond positively to relatively high fire frequencies, but actually are dependent on fire for successful reproduction and growth. A well-documented example of such a fire-dependent species is longleaf pine. There are excellent accounts of the relationship between fire and longleaf pine, the most recent of which focuses on the importance of fire in several aspects of its population dynamics (Platt and others 1988).

Woody species data for WS77 provides an example of the effects of long-term fire exclusion on longleaf pine, since WS77 had not been burned for approximately 40 year prior to the initiation of the study. Figure 4 is a size class frequency distribution comparing longleaf pine to loblolly pine, which is a much less tire-dependent species. The distribution pattern for loblolly pine is typical of a successfully regenerating species, with high frequencies of small stems and attenuating numbers toward larger size classes. In contrast, the pattern for longleaf pine (e.g., extremely low frequencies of small stems) is indicative of greatly suppressed regeneration. Thus, long-term fire exclusion and greatly reduced fire frequencies cause sharp declines in longleaf pine populations.

Conclusions

This Coastal Plain pine flatwoods ecosystem is distinctly oligotrophic and fire, as an integral part of the system, serves a significant role in increasing nutrient availability. It is thus notable that P and K typically increase in availability after fire.

The importance of fire on the plant community level was evident in its effects on the herbaceous layer. Although these effects were variable (especially varying with season of bum), fire can cause substantial increases in species diversity, apparently by altering microenvironments and ultimately increasing resource availability.

Fire also plays a vital role in the life history and population dynamics of several plant species in pine flatwoods systems. Data presented here demonstrate the importance of fire in maintaining successful regeneration of the canopy co-dominant species, longleaf pine.

Thus, tire effects appear to be integrated across all hierarchical levels of organization, from the population to the community to the ecosystem. Fire serves significant functions that are both required and unique at each level.

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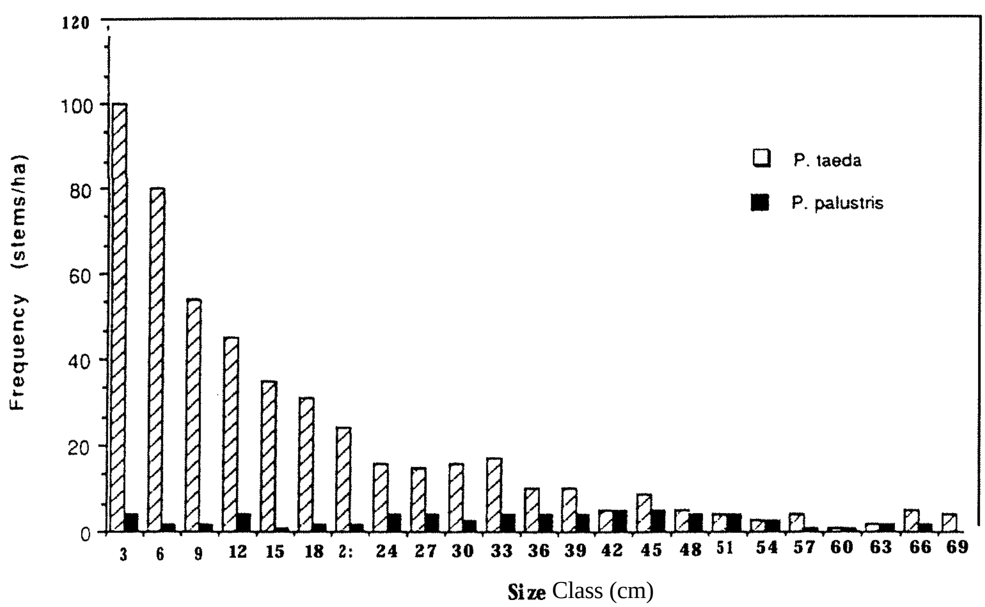

Figure 4.-Size-class distributions for loblolly pine (P. taeda) and longleaf pine (P. palustris) for WS77.

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THE EFFECT OF A HIGH INTENSITY FIRE ON THE PATCH DYNAMICS OF VA MYCORRHIZAE IN PINYON-JUNIPER WOODLANDS

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Abstract—Overall effects of fire on forest ecosystems are complex, ranging from reduction of aboveground biomass to impacts on soil microbial processes. This study reports on the short-term ecological effects of a high intensity fire on the vesicular-arbuscular (VA) mycorrhizae distribution, densily and diversity in pinyon-juniper woodlands. In fall of 1989, 1 hectare of mature pinyon-juniper located near the Grand Canyon, Arizona, was intentionally burned using drip torches. Soil cores were taken from interspaces and beneath canopies of pinyon and juniper during the spring of 1989 and immediately prior to and 96 hours after burning the following fall. In the spring, there were no differences in VA mycorrhizal species richness under pinyon, juniper or interspaces. Glomus fasciculatum and G. aggregatum were the two most frequently observed species. Immediately before the bum, species richness was slightly lower than in spring for each of the three cover types. Following burning, G. fasciculatum, G. deserticola, and G. macrocarpum were the only remaining species in each of the three cover types. Seasonal differences% soil spore densities were found between spring and pre-burned conditions. Spore numbers were significantly lower in interspaces than under canopies. Post-bum spore numbers were significantly reduced under tree canopies (up to X8 percent loss) as compared with the interspaces (47 percent loss). Loss of mycorrhizae was negatively correlated with soil temperature and heating duration, which varied with the amount of litter and duff burned (under tree canopies) and suhcanopy position.

INTRODUCTION

The importance of mycorrhizae in ecosystem function is well documented (Allen 1988; Mosse 1973; Menge and others 1978; Safir and others 1987). Without mycorrhizac many plants show a decreased growth rate or fail to develop beyond germination and Smith 1983; Masse 1973; Powell and Bagyaraj 1984). Studies have shown that this symbiosis is fragile, and that mycorrhizal activity decreases with increasing levels of disturbance (Daft and Nicholson 1974; Habte 1989; Janos 1980; Jasper and others 1989; Klopatck and others 1988; Warner 1983; Williams and Allen 1984). For example, the frequency of vesicular-arbuscular (VA) mycorrhizal propagules decreases from a moderate disturbance such as livestock grazing (Bcthlcnfalvy and Dakcssian 1984a; Rcece and Bonham 1978) to a severe disturbance such as surface mining (Allen and Allen 1980; Gould and Libcrta 1981; Zac and Parkinson 1982).

Klopatck and others (1979) estimated that the pinyon-juniper association is the third most expansive vegetation type in the United States. It covers approximately 32.5 million hectares in the western U.S. and 5.75 million hectares in Arizona (Arnold and others 1964). Pinyon-juniper woodlands are located between arid and semiarid mesic ecosystems. On the xeric end of the scale, juniper trees and desert shrubs coexist, while pinyon and ponderosa pine coexist on more mesic sites. Intermediate between these limits both pinyon pine and species of juniper exist together with interspace areas occupied by shrubs, grasses and other herbaceous cover. Why these trees exist in such diverse environments may be due to their mycorrhizal association. For example, it is known that many arid land shrub species are VA mycorrhizal, as are juniper trees, while all pine species are cctomycorrhizal.

Pinyon-juniper woodlands are managed for multiple use. As a result, both grazing (over 100 years West 1984) and prescribed burning (over 75 years USDA Forest Service) are perturbations that have occurred simultaneously in these woodlands for many years. Natural and prescribed fires impact the spatial mosaic patchwork of both VA mycorrhizae (juniper, interspace grass and shrub) (Klopatck and others 1988) and ectomycorrhizac in forest ecosystems (pine, spruce, and fir) (Mikola and others 1964; Schoenbcrger and Perry 1982). Until recently, little was known about the response of VA mycorrhizal symbionts to fire. Klopatck and others (1988) showed that after a simulated fire, VA mycorrhizal colonization was reduced when burning temperatures exceeded 90° C. Soil water availability at the time of burning also played an important role in VA mycorrhizal survival, with dry soils being more of a detriment than wet soils because of higher resultant temperatures. Gibson and Hetrick

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(1988) found significant reductions of three VA mycorrhizal species following a tire in the tall grass prairie of Kansas. Dhillion and others (1988) stated that colonization levels of VA mycorrhizal fungi in little bluestem roots were significantly reduced on burned sites when compared to unburned but, increased significantly after one growing season. Their results suggest that the response of VA mycorrhizal fungi to fire may be attributed to changes in the host plant rather than the direct effect of fire. Fire temperatures did not reach a level high enough to kill all the plants, thereby leaving a large residual VA mycorrhizal pool in the soil and in plant roots. In fact, they showed that fire actually stimulated plant 'growth, unlike tires in pinyon-juniper woodlands.

In previous work, we determined that fire had a negative impact on VA mycorrhizae by decreasing the number of propagules. We wanted to determine if these results were representative under field conditions. Thus, the objective of this study was to determine how fire effects VA mycorrhizae density, diversity and distribution under field conditions in the pinyon-juniper ecosystem. Results on the effects of tire on ectomycorrhizae arc forthcoming.

MATERIALS AND METHODS

Site Description

The study area is located on the Coconino Plateau of the Colorado Plateau, adjacent to the Grand Canyon National Park on the Kaibab National Forest. Site elevations range from 1875 to 2075 meters. Soils are Lithic and Fluventic Ustochrepts having a sandy loam texture and belong to the Winona-Boysag association (Hendricks 1985). Kaibab limestone, with intrusions of Moenkopi sandstone, are the dominant parent materials; slope is minimal, ranging from O-2 percent. The seasonal regime of cold, wet winters and hot summers with occasional thunderstorms in this region, results in this being the evolutionary center for pinyon-juniper development (Nielson 1987). Annual precipitation of 350 millimeters is bimodally distributed, approximately half occurs as intense thunderstorms from July to September, with the remainder coming as mild winter rains or snows from December to April. Soil moisture deficits exist from March through October. Temperatures are variable, ranging from - 27 to +38° C with an average of 150 days between last and first frost. Permanent weather recording stations are located in Tusayan, Arizona, less than 20 kilometers from the study area.

The area is dominated by pinyon pine (Pinus edulis Engclm.) and Utah juniper (Juniperus osteosperma (Torr.) Little). Several species of grass [blue grama grass (Bouteloua gracilis (H.B.K.) Lag. ex Steud.), squirrel tail (Sitanion hystrix (Nutt.) J.G. Smith), Stipa sp.] and shrub species [snakeweed (Guttierrezia sarothrae (Pursh) Britt and Rusby), rabbitbrush (Chrvthothamnus sp.), cliffrose (Cowania mcxicana var. stansburiana (Torr.) Jepson)] dominate the interspaces.

Experimental Design

From the area described above, we chose approximately 1 hectare of mature pinyon-juniper (250 plus years old) as our study site. The site was divided into quadrants (4 subplots) in which the position and number of each pinyon, juniper, and interspace was mapped. Every tree was marked with brass tags bearing ID numbers. The site was fenced to exclude livestock grazing. On September 11, 1989, we burned the site using hand-held drip torches. All living, downed, and dead fuels were ignited. Burning was conducted by the Kaibab National Forest, Tusayan Ranger District with assistance of the National Park Service, Grand Canyon.

Soil samples were evaluated for VA mycorrhizal spores in the spring of 1989 and immediately before and 96 hours after the September bum. Spring samples were taken to assess seasonal variability. During each sampling period, soil cores were taken from the same three randomly selected pinyon and juniper and interspaces in each of the four quadrants. Soil cores were taken 96 hours after the burn (post-bum) because trees were still burning and smoldering. Soil cores were taken from the base of the tree, mid canopy and at the canopy edge to a depth of 10 centimeters. This yielded 18 cores per quadrant, totaling 72 tree cores (2 tree species X 3 trees per quadrant X 3 samples per tree X 4 quadrants = 72). Four additional soil cores were taken per quadrant from interspaces, for a total of 16 interspace samples. Cores were wrapped in polyurethane and refrigerated at 4" C until processed.

In the laboratory, each sample was sieved (2 millimeters) to remove rocks and allowed to air-dry. From this, 20-gram samples were taken to estimate spore numbers using differential centrifugation (Ianson and Allen 1986). Spores were placed in a petri dish with sterile distilled water and examined under a dissecting microscope at 40X. Spores were divided into live and dead. Viability was determined by placing spores on a microscope slide, those which exuded cytoplasm when crushed were considered viable. Species identification were determined with a compound microscope at 400-1000X. Spore numbers are reported as means with $\pm$ standard errors of the mean. Significant differences ($p < 0.05$) in spore numbers were isolated using Tukcy's honest significant difference measure. Percent loss of mycorrhizae was calculated by subtracting the difference between pre- and post-bum spore numbers and dividing it by the pre-bum spore number.

RESULTS ANDDISCUSSION

Species Richness

Eight species of VA mycorrhizal fungi were recovered from the site (table 1). In spring (May 1989), there were no differences in species richness under pinyon, juniper, or interspaces (fig. I). Glomus fasciculatum and G. aggregatum were the two most frequently observed species with G. macrocarpum being the least dominant. Pinyon pine, although ectomycorrhizai, has been reported to have numerous VA mycorrhizal propagules around its base (Klopatek and Klopatek 1986). This is likely due to: 1) aeolian deposition of spores, and 2) the intermixing of juniper roots with those of pinyon. Wind deposits sand particles under pinyon pine (Barth 1980; Klopatek 1987) and presumably deposits these large spores along with the sand. In addition, on a recent excavation, we found juniper roots intertwined with pine roofs (Klopatek and Klopatek, unpublished). Thus, pinyon is an important repository for VA mycorrhizal propagules. This is in contrast to other pine dominated forests where no VA mycorrhizal spores are found (Kovacic and others 1984).

Table I.--List of species from soi ls taken from under pinyon and juniper canopies and interspaces. No differences in species were found in either of the three cover types. Species are Listed in the order of relative abundance.

Gtomus macrocarpum Tut & Tut
G. occultum (Walker)
G. mosseae (Nicot & Gerd) Gerd. & Trappe
Scutettospora calospora (Nicol. & Gerd.) Walker & Sanders
G. deserticuta Trappe, Bloss & Menge
G. aggregatum Schenck & Smith
G. fascicutatum (Taxter sensu Gerd.) Gerd. & Trappe
Acaulospora laevis Gerd. & Trappe

Species richness varied with season. Immcdiatcly before the fall burn, species richness dropped in each of the three cover types (fig. 1) compared with spring. Species richness was greatest in interspaces covered with grass, followed by pine and juniper soils, respectively. In post-burn samples, no differences in richness were found among the three cover types; but, species richness declined in all post-bum samples (fig, 1). G. fasiculatum, G. desetticula, and G. macrocarpum were the only species that survived the fire. These species arc all thick walled as compared with the other five (table I). In addition, they are commonly found in very arid, alkaline soils (Bethlenfalvay and others 1984; El-Giahmi and others 1976; Pfeiffer and Bloss 1980; Safir 1987) and, therefore, may be more resistant to extreme temperatures.

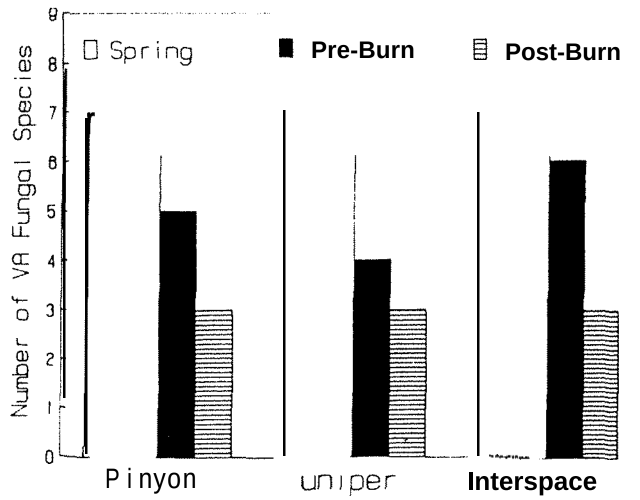

Figure 1 .--Change in the number of VA mycorrhizal species in soils taken from beneath pinyon and juniper and interspaces due to change in season and effect of fire

Spore Densities

Spore density varied under pinyon, juniper, and interspaces during the spring and pre-bum samplings (fig. 2). Overall spore counts under pinyon were not significantly higher in the spring than in pre-bum samples. Significant differences existed ($p < 0.05$) between spring and pre-bum samples in juniper and interspace soils (fig. 2). This decrease between spring and *pre-bum* samples **may** be attributed to a large amount of germination and hyphal activity rather than spore production following the summer rains. There were statistical differences ($p < 0.05$) in spore numbers between juniper and pinyon soils in the spring sampling, and most samples were significantly greater ($p < 0.05$) than interspaces (fig. 2). In general, the pre-bum pattern of spore dispersal exhibited the highest proportion at the base of the trees and decreased outward.

Burning significantly ($p = < 0.05$) decrease the overall number of VA mycorrhizal spores in soils beneath pinyon and juniper canopies (up to 88 percent) and interspaces (up to 47 percent loss) (fig. 2). Following the burn, spore numbers under canopies ranged from a high of sixteen to a low of four per 20 grams of soil. There did not seem to be a pattern of spore distribution and subcanopy position. The substantial losses under canopies was probably due to the direct effects of the soil temperatures. The highest soil temperatures were reached under canopies (up to 315" C at 2 centimeter depth) compared with interspaces (up to 68" C at 2 centimeter depth). The large fuel load, including aboveground material, litter, and duff, in addition to a more complete combustion of these fuels, probably contributed to a more intense bum under the canopies. Smoldering duff and tree stumps maintained high temperatures for several days. Magnitude and duration

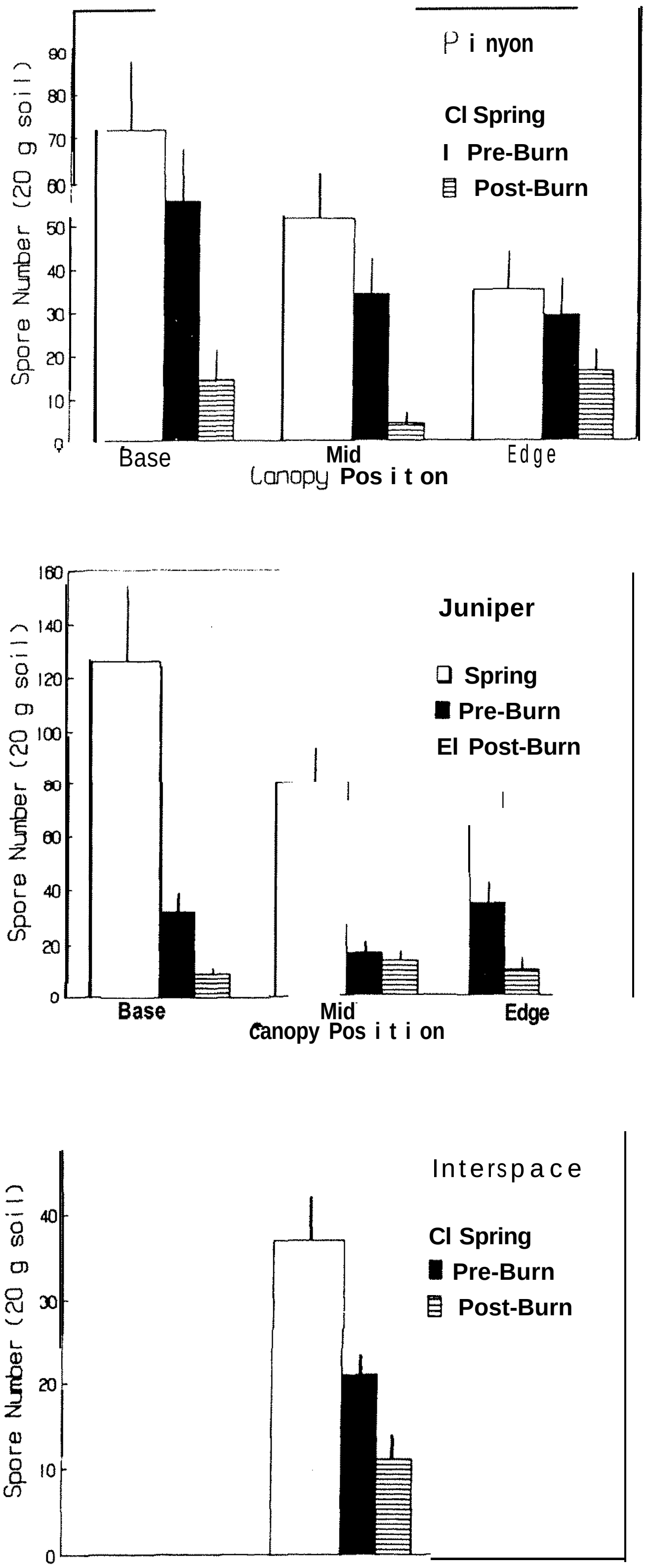

Figure Z.--Number of VA mycorrhizal spores per 20-gram soil sample taken from beneath pinyon and juniper canopies and interspaces. Samples were taken at three locations under tree canopies--base, mid and canopy edge and from grass covered interspaces

arc the two principal factors causing heat injury to plants (Hare 1961) and are also likely to be deleterious to VA mycorrhizal fungi. Smoke resulting from burning of the trees may also have contributed to the loss in mycorrhizae as it has been shown to reduce other microbial activities (e.g., Li and others 1988).

Interspaces had little aboveground vegetation and litter and no duff, which resulted in an overall lower soil temperature. Klopatek and others (1988) showed interspaces were the least affected by a simulated bum compared with canopy microcosms. We observed that tire either swept through the interspaces or did not bum at all. Pruning of grasses does not adversely affect mycorrhizal colonization, but temporarily inhibits sporulation (Powell and Bagyaraj 1984). We anticipate that the burning of grasses will produce the same response. If grasses are killed, and roots are not severely damaged by the fire, we theorize that root pieces will serve as propagules. Tommerup and Abbot (1981) showed that colonized root pieces can remain viable propagules for extended periods in partially dried soils, but they lose viability once moisture levels increase, (Gould and Liberta 198 1; Hall 1979) due to decomposition. Thus, the fire shifted the distribution of spores from under the canopies to the interspaces.

The time required for mycorrhizal populations to recover following tire in pinyon-juniper woodlands is unknown. Janos (1980), MacMahon (1987), and Allen and Allen (1988) suggest that mycorrhizal fungi are essential in ecosystem recovery, facilitating plant establishment by regulating nutrient flow from the soil to the plant. Thus, in order to understand and manage this ecosystem, it is necessary to understand mycorrhizal response to tire and how it affects patch dynamics that lead to a mosaic landscape pattern (i.e., from a canopy dominated mycorrhizal community to a interspace dominated mycorrhizal community). This “patch” pattern of disturbance is unlike a widespread disturbance, such as stripmining (Klopatek and Klopatek 1984). Thus, the natural mosaic configuration of canopy and interspace leads to a significant shift in the “patchwork dynamics” of mycorrhizal distribution following fire.

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FOREST SOIL CHARACTERISTICS FOLLOWING WILDFIRE IN THE SHENANDOAH NATIONAL PARK, VIRGINIA

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Abstract-Forest floor and mineral soil samples were collected and analyzed to determine physical and chemical differences among three bum levels (high, low, and unburned) under a mixed pine forest one year after a mid-July, 1988 wildfire. Total forest floor thickness and weight were significantly different among all three burn levels. Low intensity surface tires consumed the surface Oi-Oc layer of the forest floor leaving the Oa layer relatively intact, whereas high intensity fires resulted in the complete destruction of the forest floor. Total carbon and nitrogen concentration and content were significantly higher in residual Oa material of low bum areas compared to unburned Oa material. Active acidity (pH) in the top 10 cm of mineral soil in high and low bum areas measured 4.6 and was significantly higher than unburned areas with a value of 4.3. Total carbon and nitrogen levels in the surface 10 cm of mineral soil were also higher in low burn areas whereas high bum areas were lower than unburned sites. Mineral soil inorganic nitrogen levels were significantly higher in both high and low bum areas compared to unburned arc as, therehy providing a pulse of available nitrogen for plant uptake

INTRODUCTION

Fire has and will continue to play an important role in affecting biotic and edaphic components of forest ecosystems. The Table Mountain pine (*Pinus pungens* Lamb.) pitch pine (*Pinus rigida* Mill.) forest complex is typically considered a fire-adapted community. In fact, several authors (Sanders and Buckncr 1988; Bardcn and Woods 1976; Zobel 1969) concluded that high intensity fires were necessary to ensure successful regeneration and establishment of Table Mountain pine by (1) opening serotinous cones (Table Mountain pine), (2) inducing basal sprouting (pitch pine), (3) destroying excessive litter and exposing the mineral soil, (4) eliminating dense undcrstory vegetation, and (5) destroying allelopathic substances. However, little is known about the role and impact of variable intensity wildfire on forest floor and mineral soil characteristics upon which these species occur.

High elevation sites supporting mixed pine forests arc generally moisture-limiting and typified by shallow, acidic rocky soils with minimal rooting volume and associated infertile conditions. Low intensity fires (prescribed and wildfire) may actually enhance soil fertility by increasing pH (Gricr 1975; Metz and others 1961; Wells and others 1979), providing an influx of inorganic forma of nutrients and increasing solubility of these nutrients (Alban 1977; Metz et al 1961; Lewis 1974; Covington and Sacket 1986), and volatilizing compounds such as monoterpcnes which arc known to have inhibiting effects on bacteria populations responsible for ammonification and nitrification processes (White 1986a). Conversely, high intensity fires may result in a significant reduction of the total nutrient capital from the site resulting in the further reduction of already poor site quality conditions. However, these losses following high intensity fires may not bc altogether detrimental, since Table Mountain pine is thought to have a low nutrient requirement which may naturally select and promote the maintenance of this species on xeric, poor quality sites.

Much work has been done on the effect of prescribed tires on soil properties, but these fires arc generally of low intensity and results are stated as contrasts between burned and unburned. In contrast, variable intensity wildfires provide comparisons among several intensity levels; however, statistical analyses and inferences from results are limited due to non-random placement and inability to replicate treatments. Nonetheless, wildfires provide a unique study arena because of their natural occurrence and exhibition of several intensity levels. The objectives of our study are to determine forest floor and mineral soil physical and chemical properties following a variable intensity wildfire within a mixed pine forest and to discuss the possible importance of these impacts in relation to existing vegetation.

METHODS

Study Site

On July 11, 1988, the National Park Service (NPS) located a lightning-caused wildfire on Dovel Mountain in the Shenandoah National Park and adjacent private lands. Dove1 Mountain is located approximately 6.5 km northcast of the town of Shenandoah in Page county, Virginia. The mid-July wildfire burned approximately 350 ha before being brought under control and declared extinguished on August 8, 1988.

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The study area is located in the Blue Ridge Physiographic region and is underlain by the Erwin (Antictam) and Hampton Geologic Formations (Allen 1967). Soils are derived from granodioritc, arkosic sandstone, and grcenstone and have not been classified into series but are typically shallow and skeletal, with numerous rock outcroppings. Elevations in the area range from 350 to 800 m and slope inclination ranges from 40 to 65 percent. Mixed pine forests consist prcdominatly of Table Mountain, pitch, and Virginia pine (Pinus virginiana Mill.).

Field Methods

Areas were selected within the burned and adjacent unburned forest so that conditions of uniform species composition, age, and density; slope position, aspect, elevation, and steepness; and soil characteristics were represented. Fire intensity or bum levels were not actually measured, however, this terminology is used to categorize and represent the level of overstory mortality that resulted following fire occurrence. High intensity bum levels represent greater than 75 percent overstory mortality of basal area and crown cover, whereas low intensity bum levels represent less than 25 percent overstory mortality. A combination of crown and surface fires resulted in high intensity bum sites whereas surface fires represented low intensity burn areas. All sampling occurred on backslope positions with southwest-facing slopes.

Forest floor and mineral soil sampling occurred simultaneously during the first full growing season following the fire. Three sites were located within each bum level for a total of 9 sites. At each site, six sampling points were randomly located for a total of 18 forest floor and composite mineral soil samples within each burn level. The forest floor was sampled with a 100-cm^2 template. At each sampling point, a knife was used to cut along the template border and the Oi-Oe layer was removed and bagged. The Oa layer was removed separately and also bagged. Mean depth of the Oi-Oe and Oa layers were determined within each fire intensity level. At each sampling point, three mineral soil samples were taken to a depth of 10 cm, and combined to form one composite sample. Bulk density samples were also taken to the 10-cm depth at each sampling point using the excavation method (Blake and Hartge 1986).

Lab Methods

Forest floor samples were oven-dried at 65°C for 48 hours, and rocks and other non-plant material were removed to determine weight of the Oi-Oe and Oa layers of the forest floor per unit area. Forest floor samples were sieved to remove the mineral soil fraction before being ground in a 2-mm Wiley mill. Ground samples were then remixed with the mineral soil fraction using a sample splitter. Total carbon was determined using a Leco™ high-temperature induction furnace (Nelson and Sommers 1982). Total nitrogen was digested using the micro-Kjeldahl method of Bremner and Mulvaney (1982), followed by analysis of the resultant extracts using a Technicon™ autoanalyzer.

Composite mineral soil samples were air-dried and sieved to separate coarse fragments. Active acidity, measured as pH, was determined using a 2:1 distilled water to soil ratio. Total nitrogen and carbon levels of the surface IO cm of mineral soil were determined using the same procedures as those used for forest floor samples. Inorganic nitrogen was determined by extracting exchangeable NH,-N, NO,-N, and NO,-N, using 2M KC1 extractant, followed by analysis with a Technicon" autoanalyzer.

Statistical Methods

Forest floor and mineral soil variables were subjected to analysis of variance for a completely randomized design followed by Tukey's multiple comparison procedure to determine significant differences at the 0.05 level among fire intensity levels (high, low, and unburned).

RESULTS AND DISCUSSION

Forest Floor Parameters

In forest ecosystems, the major portion of macro-nutrients arc tied up in the surface organic matter. These nutrients are slowly released through the process of microbial-mediated decomposition and mineralization. Under normal oxidation conditions, organic matter provides a slow release, revolving fund of nutrients for plant uptake. Conversely, fire tends to rapidly release these nutrients either by volatilizing lower molecular weight gases (H,O, and N) into the atmosphere or concentrating many basic cations in the residual ash.

Wildfire consumes the forest floor in direct proportion to the intensity of the fire. Mean depth of the Oi-Oe layer in unburned areas was 1.4 cm; however, this layer was totally absent in low bum areas the first year after the tire (fig. 1).

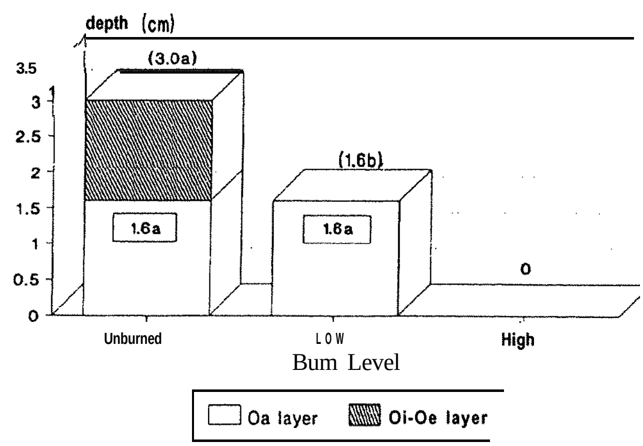

Figure 1 . Forest floor depth one year after fire occurrence. [Values in boxes represent comparisons among Oa layers while values in parentheses represent totals (Oi-Oe + Oa). Means followed by the same letter are not significantly different at the 0.05 level.]

Although the Oi-Oc layer was consumed following low intensity fires, the Oa layer was left relatively intact with a mean depth of 1.6 cm. Mean depth of the Oa layer in unburned areas was also 1.6 cm. Unlike low intensity surface fires, high intensity fires resulted in the complete destruction of the entire forest floor (Oi-Oe + Oa). Therefore, the remaining discussion of forest floor parameters will focus mainly on differences between low and unburned areas.

Similar to trends in depth, forest floor weight also changes in direct proportion to the intensity of the fire. Mean weight of the Oi-Oe layer for unburned sites averaged 24390 kg ha^{-1} while no weights were recorded for low bum areas due to the consumption of this layer during pyrolysis (fig. 2). Unlike the Oi-Oe layer, Oa layer weights were not different between unburned and low bum areas.

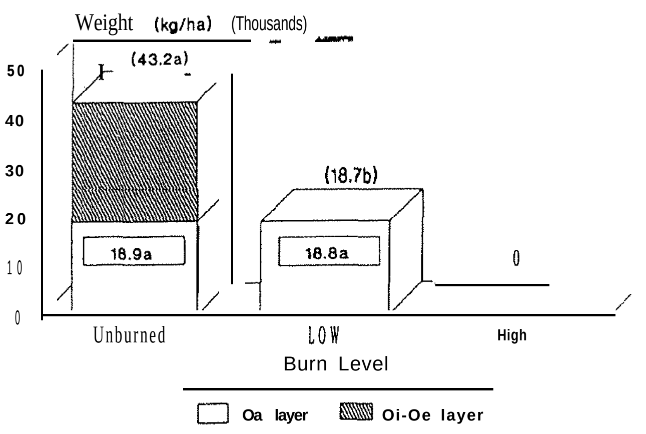

Figure 2 Forest floor weight (kg ha^{-1}) one year after fire occurrence. Means followed by the same letter are not significantly different at the 0.05 level.

Total forest floor depth and weight, represented by combining the Oi-Oe and Oa values, was significantly less on burned areas due to the partial or complete destruction of forest floor material (figs. 1, 2). Assuming that high, low, and unburned areas had similar forest floor depths and weights prior to fire occurrence, low bum values would then represent a 47 percent reduction in depth and a 57 percent reduction in weight while high intensity bum areas represent a 100 percent reduction in depth and weight compared to unburned sites.

Reductions in forest floor depth and weight following burning have been well documented in the literature. Immediately after a periodic winter bum in the south, total forest floor weights were decreased from 26900 kg ha $^{-1}$ to 19600 kg ha $^{-1}$. After 20 years of annual summer bums, the forest floor was reduced to 7800 kg ha $^{-1}$, whereas annual winter burns reduced the forest floor to 14600 kg ha-' (Brendcr and Cooper 1968).

Several studies (Brender and Cooper 1968; Moehring and others 1966; Romancier 1960) have demonstrated that prescribed burning does not result in a significant loss of forest floor material. In fact, a single prescribed bum may remove only a small percentage of the total forest floor depth and weight. Results from this and other related studies indicate that reductions in forest floor material arc directly related to tire intensity.

Total C concentration (%) and content (kg ha^{-1}) were significantly higher in the residual Oa material collected from low bum areas compared to unburned Oa material. Total C concentration of Oa material from low and unburned areas were 61 .O and 52.7 percent, respectively (fig. 3). Since post-tire Oa weights were similar for low and unburned areas, increases in total C concentration also resulted in greater C content (kg ha-') of the Oa layer on low bum sites. Overall, however, total C content for the entire forest floor was 50 percent lower on low bum areas, compared to unburned areas, due to the consumption of the overlying Oi-Oe layer.

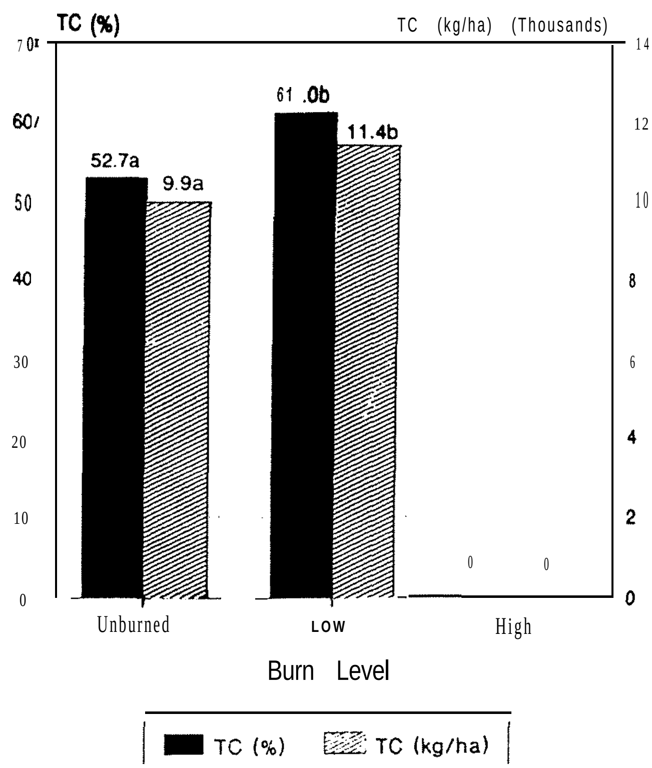

Figure 3 . Total carbon concentration (%) and content (kg ha $^{-1}$) of the Oa layer one year after fire occurrence. Means followed by the same letter are not significantly different at the 0.05 level.

The major elemental components of organic matter include N, 0, H, and C. During normal decay processes of organic matter, some C is converted to CO,, some is incorporated into microbial tissue while the remaining C is converted to more stable humus forms which are higher in total C concentration than the original organic matter (Stevenson 1986). Pyrolysis reactions tend to accelerate this process by volatilizing lighter atomic elements (N,O,H) while converting the original organic matter to more stable humus forms that contain a higher percentage of reduced, elemental C. Likewise, other basic cations (K,Ca,etc.) along with reduced, elemental C, are concentrated in the residual ash following fire. The reduction of C and the concentrating effect of pyrolysis reactions would explain the increase of total C noted in the Oa layer of low bum areas in this study. This reduced form of C does not supply a readily mineralizable source for microbial assimilation and may remain in the soil as fusain for many years (Sopcr 1919; Hansen 1943). Since C is a large chemical constituent of organic matter (approximately 58%), a loss of organic matter following fire will result in a reduction in C content from the site. The more intense the tires, the greater the consumption of the forest floor and subsequent C pool. Organic matter also serves as an important source of N, P, and S, which are also reduced following consumption of the forest floor.

Similar to C trends, total N concentration (ppm) and content (kg ha ¹) were also higher in the residual Oa material of low bum areas compared to unburned Oa material (fig. 4). An increase in total N following low intensity tires may be due, in part, to an increase in inorganic N in the Oa layer. Scveral authors (White and others 1973; Klemmedson and others 1962; Kovacic and others 1986) have found similar trends in inorganic N concentrations one year after prescribed tires. This increase may be attributed to incomplete combustion and volatilization of N with subsequent downward translocation and reprecipitation of N gases in cooler forest floor and mineral soil layers (Klemmcndson and others 1962; Tangren and McMahon 1976; Wells 1971). Substantial amounts of NH4-N are also produced chemically by soil heating and microbially after tire. Unlike NH_4-N, NO,-N is not produced during soil heating, but is formed during subsequent mineralization and nitrification processes. Additionally, White (1986a) found that potential N mineralization and nitrification increased in residual forest floor material following prescribed fires. Jones and Richards (1977) suggested that nitrification processes following fire may not be due to Nitrosomonas or Nitrobacter bacteria but to heterotrophic fungi. Increases in total N may also be correlated with an increase in total C and/or to an increase in N2-fixation following fire. N2-fixation may contribute 10 to 100 kg ha of N annually (Stevenson 1986).

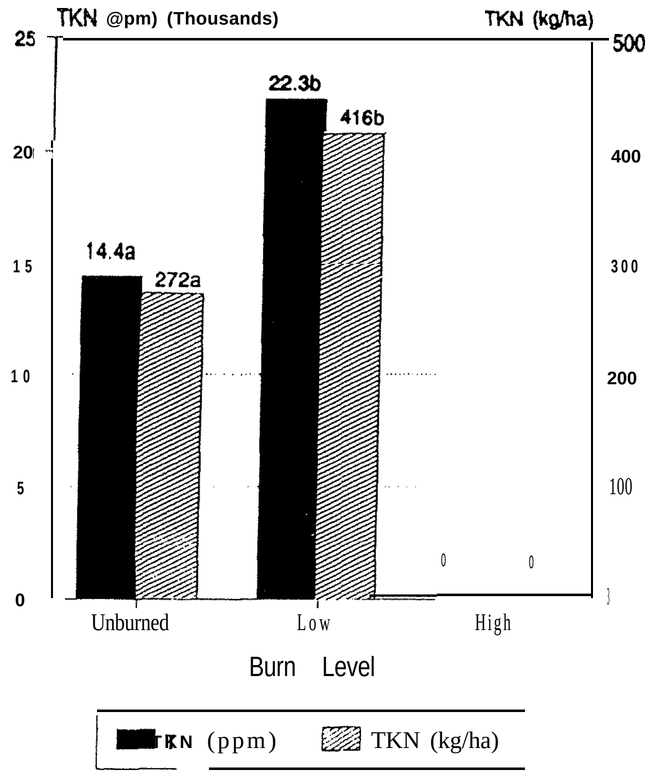

Figure 4 Total nitrogen concentration (%) and content (kg ha ¹) of the Oa layer one year after fire occurrence. Means followed by the same letter are not significantly different at the 0.05 level.

Assuming homogeneity on burned and unburned sites prior to fire occurrence, combined forest floor (Oi-Oe + Oa) C and N values for low burn areas would represent a 50 and 28 percent reduction, respectively, compared to unburned values (figs. 5, 6). These reductions in total C and N content of the residual forest floor in low bum areas is attributed to the consumption of the overlying Oi-Oe layer during pyrolysis.

Generally, low intensity tires, as occurred on some areas in this study, may remove only the Oi-Oe layer, leaving the Oa layer relatively intact. This residual Oa layer serves to protect the underlying mineral soil from erosion and provides a more mineralizable source of nutrients (White 1986a). Conversely, high intensity fires may remove the entire forest floor, thereby exposing the mineral soil to the vagaries of weather and possibly reducing infiltration, water holding capacity, and other associated benefits attributable to the surface organic matter.

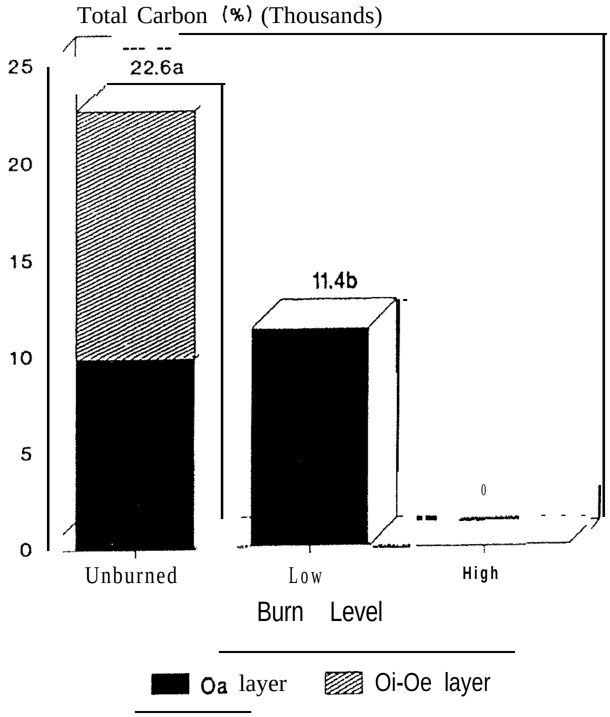

Figure 5 . Total carbon content (kg ha ¹) of the entire forest floor (Oi-Oe + Oa values) one year after fire occurrence. Means followed by the same letter are not significantly different at the 0.05 level.

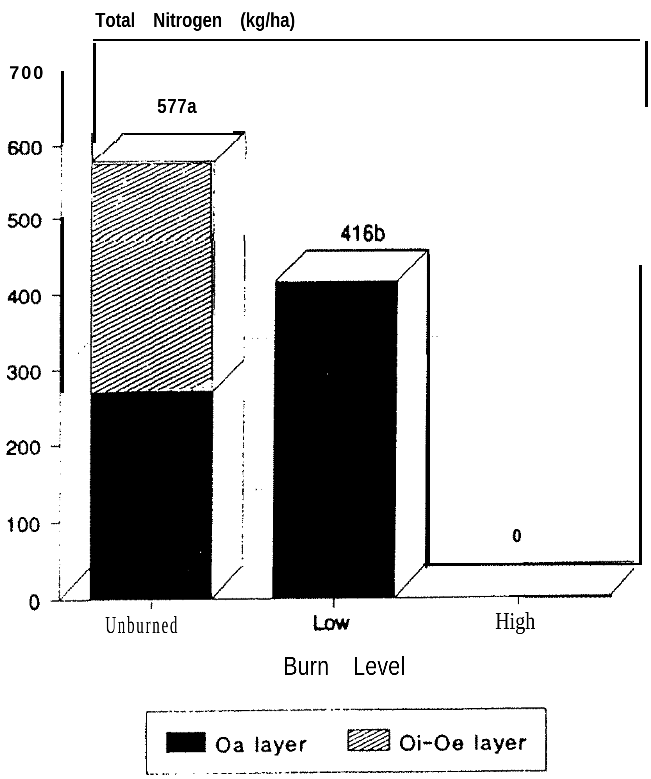

Figure 6 . Total nitrogen content (kg ha") of the entire forest floor (Oi-Oe + Oa values) one year after tire occurrence. Means followed by the same letter arc not significantly different at the 0.05 level.

Mineral Soil Parameters

Fire generally results in a decrease in soil acidity. Soil acidity in the surface 10 cm of mineral soil was 4.3 in unburned areas and significantly higher than low and high bum areas which measured 4.6. This rise in pH following fire is generally attributed to an influx of nutrient-rich ash and a resultant increase in exchangeable cations in the surface mineral soil. The magnitude of change in soil acidity is dependent upon such factors as nutrient concentration of the overlying litter, cation exchange capacity (CEC), buffering capacity and original pH of the soil, and rainfall frequency and amounts (Gricr 1975; Metz and others 1961). McKee (1982) found that soil acidity decreased in the surface (O-8 cm) mineral soil following a combination of different prescribed bum treatments on four coastal plain pine sites. Changes in pH ranged from 0.1 to 0.4 units across the four study areas. Greater changes in pH may occur, however, most studies indicate changes of less than one pH unit following prescribed tires which return to prcbum levels within a few years.

Total C and N levels in the surface 10 cm of mineral soil were higher on low bum sites compared to unburned areas (figs. 7, 8). Conversely, high bum areas had lower total C and N levels than unburned sites. An increase in total C following low intensity fires may be associated with the redistribution and movement of colloidal-sized charred material, high in elemental carbon, downward from the overlying residual ash into the mineral soil by gravity and water (Mctz and others 1961), and/or by isoelectric precipitation of alkali humatcs produced during burning (Viro 1969). A large portion of the mineral soil organic matter (humus) is associated with the forest floor-mineral soil interface and within the top 2-3 centimeters of the mineral soil. Following the removal of the entire forest floor by high intensity fires, soil humus may also be destroyed where temperatures exceed 250°C. These temperatures are easily achieved during high intensity fires where temperatures have been shown to exceed 700°C at the mineral soil surface (Debano and Rice 1971). Another factor which may contribute to a loss of total C from the mineral soil is the physical removal of the surface mineral soil and associated C due to erosion. Once the protective forest floor mantle is removed from the site, such as by high intensity tires, erosion may lead to a direct loss of C and other nutrients from the site. In this study, erosion was not measured, but was observed by the accumulation of mineral soil at downslope positions below the high intensity bum areas. Low and unburned sites did not show any observable signs of surface erosion.

Total N followed similar trends to total C levels. Increases in total N following fire are likely due to an increase in organic and inorganic forms of N. Reasons for these increases in total N following tire have already been discussed earlier in relation to forest floor material.

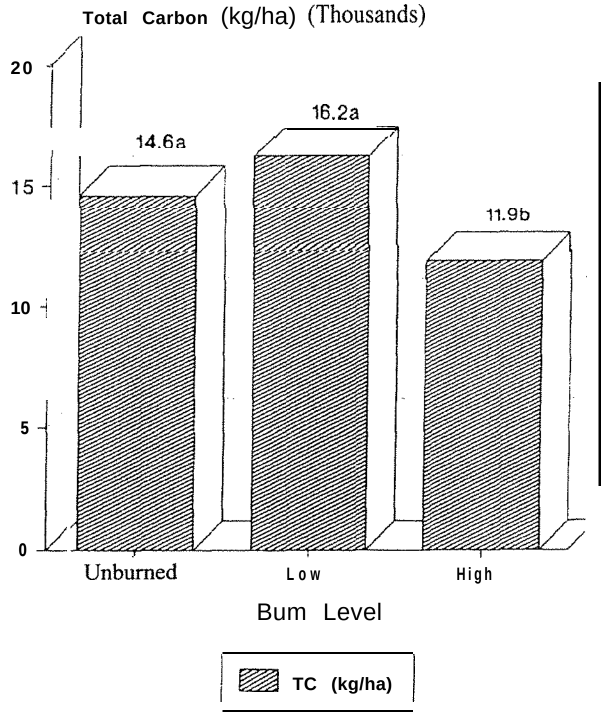

Figure 7 . Total carbon content (kg ha [1]) within the top 10 cm of the mineral soil one year after fire occurrcncc. Means followed by the same letter are not significantly different at the 0.05 level.

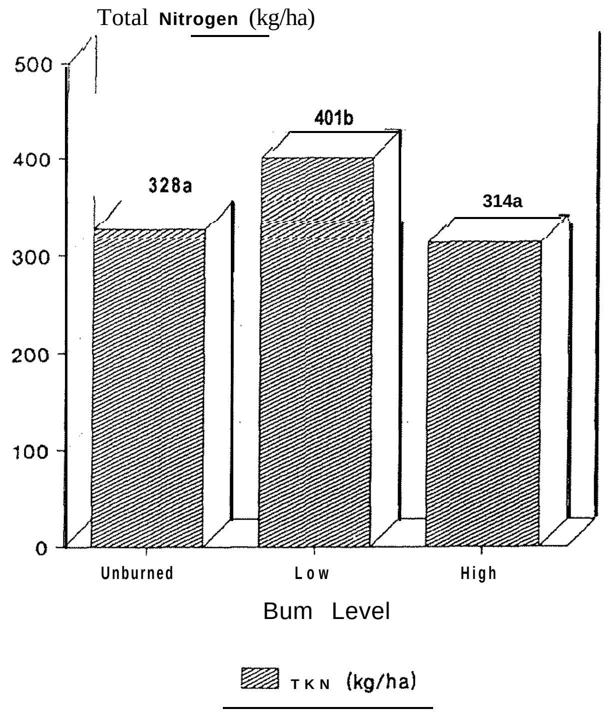

Figure 8 . Total nitrogen content (kg ha") within the top 10 cm of the mineral soil one year after fire occurrence. Means followed by the same letter are not significantly different at the 0.05 level.

Inorganic N levels were signiticantty higher on burned areas compared to unburned areas (fig. 9). Similar results by other authors (Bums 1952; Wells 1971; Alban 1977; White 1986a; Covington and Sackct 1986) have also shown increases in inorganic N following burning which is generally attributed to an increase in mineralization rates (Likens and others 1970; White 1986b; Lodhi and Killingbeck 1980) following a disturbance such as fire or clearcutting. Mineralization (ammonification t nitrification) is the process whereby organic N is converted to plant available inorganic forms through microbial-mediated biochemical transformations and is influenced by factors which affect microbial populations and activities such as pH, temperature, soil moisture, and the presence of compounds such as polyphenolics, tannins, and monotcrpenes. Following tire, increases in soil pH, temperature, and moisture, and the removal of inhibiting compounds favor increased rates of mineralization. An increase in inorganic N may also be associated with the downward translocation and rcprecipitation of N gases at cooler mineral soil depths. Klemmendson and others (1962) showed that burning accelerated N movement from the overlying forest floor into the mineral soil. Although mineralization rates vary over time with fluctuations of the factors which influence this process, the results of our point-in-time sampling of inorganic N reflects the conditions found by other investigators.

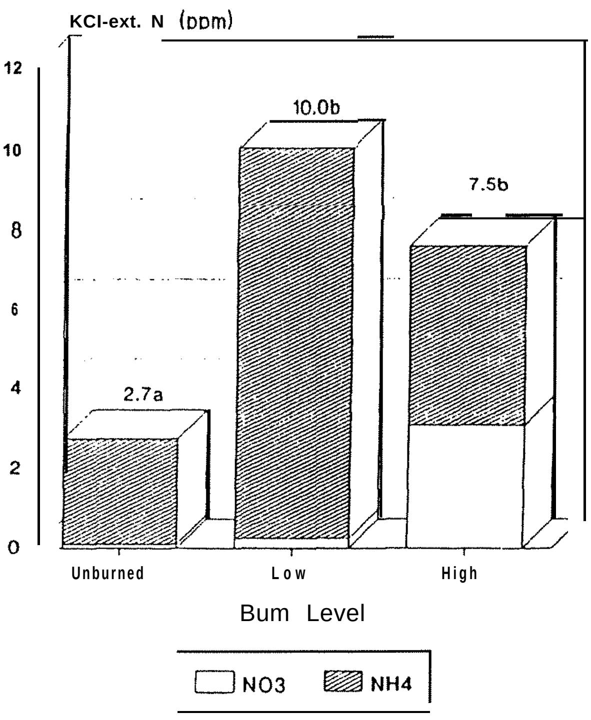

Figure 9 . KCl-extractable nitrogen concentration (ppm) within the top 10 cm of the mineral. soil one year alter fire occurrence. Means followed by the same letter are not significantly different at the 0.05 level.

Most of the increase in inorganic N following low intensity fires is due to NH_4-N, whereas most of the increase in inorganic N following high intensity fires is due to an increase in both NH,-N and NO,-N levels (fig. 9). A lack of NO,-N in low and unburned areas suggests that nitrification processes may be inhibited by some factor. Nitrification processes are severely limited at pH values below 4.5, insufficient soil moisture and cool temperatures, and in the presence of inhibitory compounds such as phenols, tannins, and monoterpenes (Stevenson 1986; White 1986b; Lodhi and Killingbeck 1986). Given the fact that low and high intensity areas had a pH of 4.6, the increase in NH,-N and the lack of NO,-N following low intensity fires suggests that the process of ammonification may be enhanced or less affected than nitrilication in the presence of factors such as monoterpenes, phenols, and tannins in the residual forest floor material. White (in press) suggested that monoterpenes may have a greater inhibitory effect on Nitrosomonas and Nitrobacter populations, responsible for nitrification, than on microorganisms responsible for ammonification. Lodhi and Killingbeck (1980) found that water extracts laden with soluble polyphenolics, tannins, and monoterpenes reduced numbers of Nitrosomonas by 93 percent. Other factors possibly contributing to an increase in the NO,-N to NH,-N ratio, following high intensity fires, is the fact that the overstory canopy was left mainly intact following low intensity fires, whereas complete ovcrstory mortality resulted following high intensity fires. This overstory removal following high intensity fires results in more direct solar radiation and a resultant increase in soil temperature. Sub-surface soil moisture also increases due to removal of transpiring vegetation. Increased soil temperatures and moisture favor rapid nitrilication (Stevenson 1986).

Since N *is* the *most* limiting nutrient on most forest sites and C sources contributing to the "Greenhouse Effect" are important considerations, changes in these pools should be considcrcd. Total C and N pools (forest floor + mineral soil) in low bum areas one year after fire occurrence were 25 and 10 percent lower, rcspcctivcly, than unburned areas. Unlike low intensity bum areas, total C and N pools in high intensity burn areas were 68 and 65 percent lower, respectively, than unburned areas. Low intensity fires resulted in a partial redistribution of C and N from the forest floor into the mineral soil with a slight overall reduction in the combined nutrient capital. High intensity fires, however, resulted in no redistribution of nutrients and a greater loss of combined C and N capital from the site. Although reductions in these pools may bc severe following high intensity fires, losses may not be altogether detrimental when considering the synccology of mixed pine forests on these poor sites. Periodic severe fires may bc necessary to prevent stagnation and to promote successful regeneration and establishment of these species while inhibiting more nutrient- and moisture-demanding hardwood species.

The impacts of fire on soil are highly variable and depend on such factors as tire intensity and duration, weather conditions, and forest floor and mineral soil characteristics at the time of burning. Overall, low intensity fires in this ecosystem seem to have little deleterious effects on soil properties and may, in fact, facilitate increased mineralization rates, thereby releasing a pulse of nutrients to the site. Increases in pH, microbial activity and nitrogen fixation may also occur. Conversely, high intensity fires result in the removal of the protective forest floor and a much greater loss of nutrients from the site. However, a question must be raised in relation to the overall impact of high intensity tires in relation to site quality and the associated vegetation occupying a site. Is the loss of nutrients more significant on better sites where there is a greater buffer capacity and the total loss of nutrients in proportion to the whole is small? Or is the loss more significant on poorer sites where a small loss may represent a large portion of the nutrient capital for that site? Given adequate time without disturbance, forest floor and mineral soil properties tend to increase in both depth and fertility. On minimally developed soils as in this study, where soil depth is shallow and inherent fertility of parent material is low, much of the sites' fertility is derived from and dependent upon the turnover of forest floor material and external inputs. Considering the five major soil-forming factors, and in the absence of disturbance, depth and fertility of the mineral soil should increase, thereby allowing more nutrient-demanding species to invade and compete for site resources.

Tree species occupying better, more productive sites generally have higher nutrient requirements; thercforc, even a small change in nutrient levels may result in a species shift. Convcrscly, tree species occupying poorer sites may have lower minimal nutrient requirements and may actually depend on periodic fire to disrupt the progress towards site quality improvcmcnt. In fact, one of the possible secondary functions for the high content and slow turnover rate of monoterpenes in fresh litter of fire-adapted Pinus species, may be to increase probability of fire occurrence. Terpenes and resins have a heat of combustion of 7720 calories per gram, which is twice that of cellulose (Rothermal 1976). These extractivcs are outgassed early in the pyrolysis process and may contribute three-fourths of the total flame height *and* intensity of the flame zone (Philpot 1969).

It is possible that low intensity fires would not provide the necessary conditions required by Tahle Mountain pine to successfully regenerate and compete, whereas high intensity fires generally result in (1) opening of scrotinous cones; (2) removal of the forest floor providing favorable seed bed conditions; (3) removal of inhibitory compounds such as monoterpenes, tannins, and polyphenolics; and (4) ironically, limiting site quality improvcmcnt by preventing nutrient buildup to *occur.* Thus, a combination of these factors may result in retarding the invasion of more nutrient-demanding competitors and sustaining the endemic Table Mountain pine on nutrient- and moisture-limiting sites.

SUMMARY

Total forest floor thickness and weight were significantly reduced following low and high intensity tires. However, low intensity tires consumed only the Oi-Oe layer, leaving the Oa layer relatively intact, whereas high intensity fires resulted in the complete removal and destruction of the entire forest floor. Total C and N content of the Oa layer was higher on low bum areas compared to unburned Oa material. However, an overall reduction of forest floor total C and N resulted following both low and high intensity tires due to the partial or complete destruction of forest floor material.

Mineral soil pH was significantly higher in both low and high intensity bum areas compared to unburned areas. Additionally, total C and N content of the surface IO cm of mineral soil was also higher on low bum areas compared to unburned areas. However, high bum areas had lower total C and N levels. Inorganic N levels were significantly higher in both low and high intensity bum areas, thereby providing a pulse of available N for plant uptake.

ACKNOWLEDGEMENTS

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THE INTERACTION OF PRESCRIBED FIRE, MONOTERPENES, AND SOIL N-CYCLING PROCESSES IN A STAND OF PONDEROSA PINE (*Pinus ponderosa*).

Carleton S. White[1]

Abstract—Monoterpenes, principal components of turpentine, have been shown to be inversely correlated with N mineralization and nitrification rates in ponderosa pine (*Pinus ponderosa* Dougl. ex Laws.) soil. and are suspected to be allelopathic substances causing germination inhibition or growth regulation. Because monoterpenes are highly flammable, prescribed fire may represent an efficient method of lowering monoterpene concentrations in both organic and mineral soil horizons. Samples of the forest floor and the 0- to 10-cm soil horizon were collected from four separate plots within a pondcrosa pine stand immediately before and after fire treatment. The prescribed tire treatments resulted in a greater proportionate loss of monoterpenes than of forest-floor biomass: loss of 55 percent of the forest-floor mass corresponded to a 99 percent loss of monoterpenes. Forest-floor inorganic N content was doubled following treatment, with all the increase as NH_4^+-N; the mineral soil inorganic N content was unchanged. During incubation for potential N mineralization, only the burned forest-floor samples produced nitrate. Thus, the prescribed fire treatments resulted in potentially favorable changes in organic matter quantity and quality, levels of inorganic N, and potential rates of N-cycling processes.

INTRODUCTION

Management of southwestern pondcrosa pine (*Pinus ponderosa* Dougl. ex Laws.) often includes the usc of prescribed burning to reduce accumulated fuels. Studies have documented increases in the inorganic N content of forest-floor or mineral soil horizons immediately after burning (White 1986a; Kovacic and others1986) or within the first growing season after burning (White 1986a; Ryan and Covington 1986; Covington and Sackett 1984; Covington and Sackett 1986). The increase in inorganic N is accompanied by a decrease in total N within the forest floor (White and others 1973 ; Klemmcdson and others 1962; Kovacic and others 1986) and by an increase in biomass and nutrient content of understory vegetation (Harris and Covington 1983; Vlamis and others 19.55).

White (1986a) conducted research on the effects of prescribed burning on four plots within a ponderosa pine stand located in the Jemcz Mountains of New Mexico. The bum treatments resulted in an immediate increase in the amount of NH_4^+-N in the forest floor. Potential N mineralization and nitrilication, as determined by laboratory incubations, were increased in samples of the forest floor collected within 12 hours of the bum. Nitrogen mineralization and nitrification potentials of the mineral soil were significantly increased in only 1 of 4 plots immediately after the bums; howcvcr, both processes were significantly increased in the mineral soil from all plots 6 months after the bum and remained elevated 10 months after the bum. White suggested that the immediate increase in nitrilication in the forest floor and the subsequent increase in nitrification in the mineral soil could be explained by the loss of volatile inhibitors from the forest floor.

The roles of volatile and water-soluble inhibitors of N mineralization and nitrification in the forest floor of the same pondcrosa pine ecosystem were studied by White (1986b). Water extracts of unburned forest floor inhibited nitrification by 17 percent when applied to actively nitrifying mineral soil from the same pondcrosa pine ecosystem after the bum treatment. Placing vials containing unburned forest floor or selected monoterpenes of ponderosa pine in sealed jars that contained actively nitrifying soil inhibited nitrification by 87.4 percent and 100 percent, respectively, and inhibited N mineralization by 73.3 percent and 67.7 percent, respectively. White (1986b) suggested that organic compounds that are water-soluble and volatile act as inhibitors of N mineralization and nitrilication in this ponderosa pine ecosystem. Inhibition of nitrification was also observed by Lodhi and Killingbeck (1980), who found that water extracts of pondcrosa pine needles applied to soil suspensions reduced numbers of *Nitrosomonas* by 93 percent. They suspected that polyphenolics and condensed tannins were the active compounds inhibiting nitrification. It appears that a number of secondary compounds produced by ponderosa pine could act synergistically to inhibit nitrilication and N-mineralization processes.

Laboratory bioassays (White, in press) showed that monotcrpenes could interact with N-cycling processes through four mechanisms: (a) by reducing net N mineralization; (b) by inhibiting nitrification; (c) by enhancing assimilatory uptake of NO_3^--N; and (d) by stimulating immobilization of inorganic N. The net effect of monoterpene addition on soil

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inorganic N content was to reduce the amount of NO,‘-N relative to NH,+-N, leading to net immobilization of inorganic N at high monoterpenc additions (fig. 1). While (IN PRESS) also showed that monoterpcnc concentrations were highest in the L horizon and declined by an order of magnitude with each descending organic and mineral soil horizon (fig. 2). Prescribed burning has the potential to reduce the total monoterpene content of the soil profile drastically because the organic horizons rich in monoterpenes would be consumed preferentially.

The goal of this study was to identify the changes in monoterpenc and inorganic N concentrations in the forest floor and mineral soil immediately following (within hours) prescribed burning of plots within a ponderosa pine stand, and to identify the change in potential N mineralization and nitrilication characteristics of these horizons.

METHODS

The present study was conducted on the same site where White (1986a, in press) had worked. The site is in north-central New Mexico, within the Jemez National Forest, near Bear Springs (elevation 2225 m). It is located on a small knoll of volcanic ash and pumice, with very uniform A horizon soils. Slope of each plot was less than 7”. The overstory was composed entirely of pondcrosa pine. Scattered seedlings and saplings of pinyon pine (Pinus edulis) and various species of junipers (Juniperus spp.) were present. White originally chose the plots to avoid heavy fuel deposits and to favor areas with approximately equal accumulations of forest floor material and woody debris. Four of the 8 original plots were treated with prescribed burning on 7 November 1983. The 4 remaining plots were used in this study foilowing a study of seasonal dynamics in monoterpene content and potential N mineralization and nitrification characteristics (White, in press).

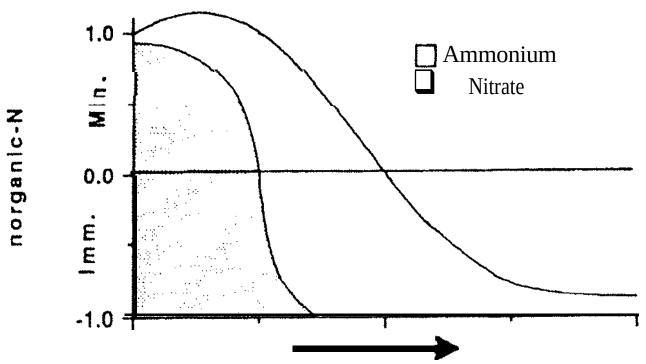

Fig. 1. Generalized response of soil inorganic-N levels to increasing monoterpenc additions. Monoterpcnc additions arc relative values, not actual concentrations (from White in press).

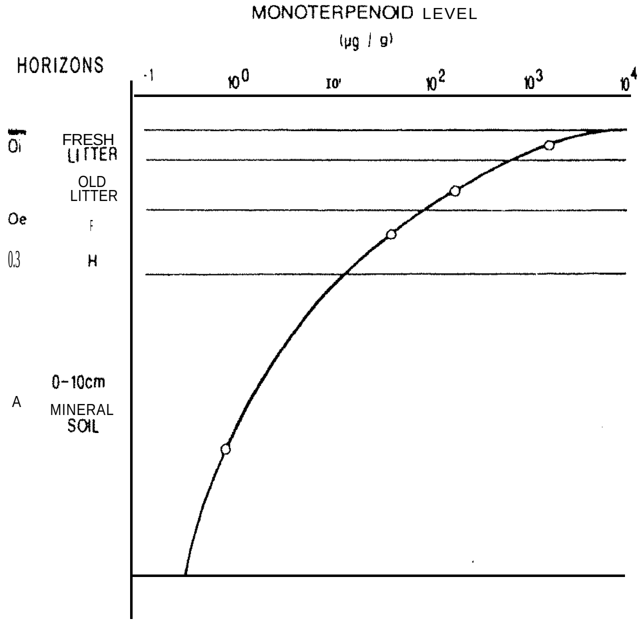

Fig. 2. Mean monoterpenc concentration (n = 8) within the designated soil horizons of a ponderosa pine stand in New Mexico. The fresh litter was collected in October 1987; other horizons were collected in November 1987. Horizons are drawn in approximate proportion to actual depth in the field based upon the mineral soil horizon equal to IO cm.

The plots were burned by U.S.D.A. Forest Service personnel on 27 October 1988. A fire break was scraped around the perimeter of each plot and the associated 5-m buffer zone. The plots were burned by igniting three ships about 5 m apart parallel to the length of the plot. Rates of fire spread ranged from 0.02 to 0.05 $m\ s^{-1}$. On 3 of the 4 plots, postburn samples were collected within one hour of burning. The fourth plot, which had a southern aspect, burned the longest; glowing combustion was observed about 2 hours after ignition. Samples from the fourth plot were taken approximately 3 hours after ignition. The amount of forest floor consumed by the burning was determined as the diffcrcncc bctwccn ash-free mass of the prcburn and postburn samples.

Collections were made from each plot on 27 October 1988 immediately before and after burning. Each plot measured 4m by 9m. A 5-m buffer zone around the perimeter of each plot was also burned. One sample per plot was collected before burning, and one sample along the same line transect was collected after burning. For each sample, all of the forest-floor material (0 horizon) beneath a 0.25 m^2 template was harvested. The template was placed on the forest floor, a knife was used to make cuts around the template border, and the surrounding forest floor was scraped away. Mineral soil was collected to a 10-cm depth with a lo-cm-diameter corer. The soil collection was made at the center of the area from which the forest floor was harvested.

All samples were placed in resealable plastic bags and kept in the dark on ice during transport to the laboratory. In the laboratory, all samples were kept refrigerated at 4 "C. Roots with diameter greater than about I mm were removed by hand sorting. All material larger than 6.4 mm in diameter was removed from all samples by sieving. Needles and other materials too long to lit into the incubation cup were cut into appropriate lengths (usually into halves or thirds). All samples were corrected for ash content by determining weight remaining after ignition at 500°C.

Nitrogen mineralization and nitritication potentials were measured by aerobic incubation. After a portion of each sample had been adjusted to 50 percent of determined water-holding capacity (WHC) by methods described in White and McDonnell (1988), a total of 17 subsamples per horizon were apportioned into 125-ml plastic cups. Each cup received approximately IO g dry-weight (DW) of mineral soil or 3 g DW of forest floor. One subsample of each horizon was immediately extracted with 100 *ml* 2 N KCl for NO_3^--N and NH,+-N analyses, and another subsample was frozen (-5 "C) prior to processing for monoterpene analysis. The remainder of the cups were covered with plastic wrap, sealed with a rubber band, and incubated in the dark at 20 "C. As reported by Jones and Richards (1977), the plastic wrap minimized water loss during incubation, yet exchange of CO, and 0, was sufficient to keep the subsamples aerobic during incubation. Moisture content was monitored by weight loss and replenished as needed

At weekly intervals to 10 weeks, a subsample of each horizon was removed for NO_3^--N and NH,+-N analyses. After extraction with 100 ml 2N KCl for 18-24 hours, the clarified supcrnatant was analyzed for NH,'-N and NO_3^--N+NO_2^--N (NO;-N was never detected) on a Technicon AutoAnalyzer. White (1986a) has described the procedures employed.

After incubation for 1, 2, 4, 7, and IO weeks, a subsample of each horizon was frozen (-5 "C) prior to processing for monoterpcnc analyses. Subsamplcs of mineral soil were transferred to plastic scintillation vials and stored at -80 "C. A mortar and pestle were used to gring the forest floor subsamples separately under liquid nitrogen to break apart the larger material. These subsamplcs were then transferred with liquid nitrogen to a Tecator¿ centrifugal grinder fitted with a LO-mm screen. After grinding, the forest-floor material was transferred to plastic scintillation vials, sealed, and stored at - 80 °C until monoterpene analyses could be performed. Subsamples (ranging in weight from 9 to 10 g for mineral soil and 2 to 3 g for forest floor) were extracted with 10 ml of ether, which contained a known amount of fenchyl acetate for use as an internal standard, in a 50-ml Erlcnmeyer flask that was covered with paraffin film and aluminum foil. After 1 hour extraction at room temperature, the ether was decanted and centrifuged (mineral soils did not require centrifugation).

The clarified supematant was pipetted into a ground-glass-stoppered culture tube, sealed with paraffin film, and refrigerated at 4 "C. A 4-μl portion of the refrigerated ether extract was injected into a Shimadzu GC-9 fitted with a split injector (split ratio was 50: I), a bonded methyl silicone capillary column (25 m in length, 0.25 mm in inside diameter, 0.25 micron in film thickness), and a flame-ionization detector. The injector temperature was 270 °C, flow rate was 4 cc min^{-1}, and initial oven temperature was 60 °C. Oven temperature was increased by 4 °C min^{-1} to 109 "C, then by 40 °C min^{-1} to 250 "C. Individual monotcrpene standards (verified with CC-MS) were added to sample extracts, and monoterpenes were identified by co-chromatography. Peak area was converted to mass of individual monoterpenes by means of calibration curves generated with standards. An average calibration factor was used to convert the peak area of each unknown to relative mass.

The effects of the prescribed tire treatment were determined for each analysis by comparing the 4 pretreatment samples to the 4 post-treatment samples by analysis of variance (control vs. treatment, $n=4$). Ash-free mass lost upon ignition was used as a measure of burn intensity since rates of spread and flame heights were approximately equal for each plot. All concentration data were converted to mass per unit area for seasonal comparisons. Net N mineralization was defined as the increase in the amount of inorganic N (NH,'-N + NO, - N) over the entire 10-week incubation. Net immobilization was defined as the decrease in the amount of inorganic N over the entire 10-week incubation. Relative nitrification was defined as the percent of the total inorganic-N pool comprised by NO<-N at the end of the incubation period. All statistical analyses were performed with SAS-PC ⌐ (Statistical Analysis System, SAS Institute Inc.) or with StatView⌐ (Brainpower, inc.). The a *priori* probability level accepted to be significant was <0.10; howcver, the probability level for each statistical analysis will be given below.

RESULTS

The amount of forest floor remaining after the prescribed burning treatments was significantly less than before treatment ($P<0.003$), but varied from 38 to 80 percent (fig. 3). The lowest amount of mass lost was approximately equal to the mass of the L horizon alone as determined in previous collections (White in press), while the greatest amount of mass lost was approximately equal to the entire L horizon and about half of the combined F and H horizons.

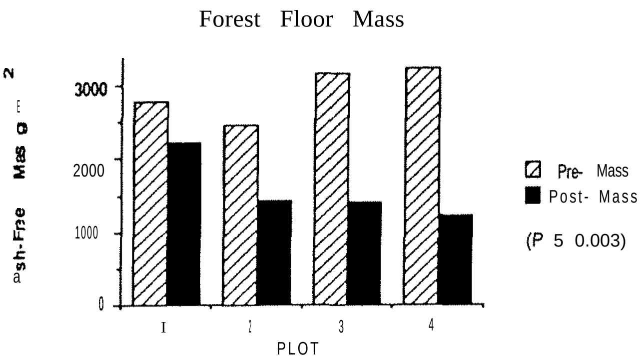

Fig. 3. Ash-free forest floor mass before (Pre-Mass) and after (Post-Mass) prescribed burning. Probability level for the comparison of preburn and postburn mass is shown ($n=4$).

Total monoterpene concentration in the postburn forest floor declined in 3 of the 4 plots and increased in the plot with the smallest loss of mass (fig. 4). When expressed on an area basis, the total amount of monoterpene declined in all plots ($P=0.067$, fig. 5). Monoterpene content of the postburn forest floor was a function of the fraction of original forest-floor mass ($r^2=0.993$, $P < 0.01$; fig. 6). The bum treatments reduced the content of some monoterpcnes to a greater extent than others. Monoterpenes with a double-bonded carbon atom in a terminal position on the molecule (including camphene, b-pincne, sabinene, Iimonene, myrcene, and limoncne oxide; fig. 7) and monocyclic monoterpencs (including p-cymene, a-phellandrene, limonene, and g-terpinene; fig. 8) were reduced to very low or undetectable levels, even in the plot that lost the least amount of forest floor (plot 1).

The bum treatments significantly increased the amount of inorganic N *in* the forest floor ($P=0.012$; fig. 9), with all the increase due to higher NH_4^+-N content and no change in NO_3^--N content. The amount of inorganic N in the mineral soil was unchanged by the bum treatment ($P=0.91$; fig. 10). Measurement of potential N mineralization and nitritication for the prebumed forest floor showed net immobilization of inorganic N and no net nitritication (net decline in inorganic

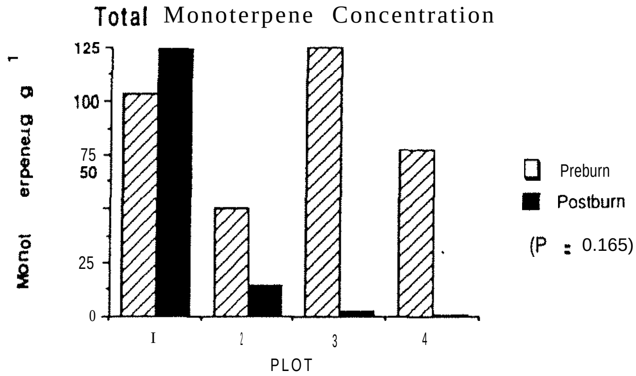

Fig. 4. Total monoterpenc concentration of the 0 horizon before (Preburn) and after (Postbum) prescribed burning. Probability level for the comparison of prebum and postburn concentrations is shown ($n=4$).

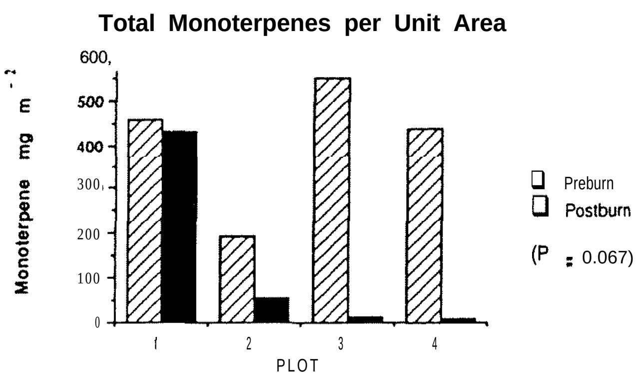

Fig. 5. Total monoterpene content of the combined organic and 0- to IO-cm mineral soil horizons before (Pre-bum) and after (Postbum) prescribed burning. Probability level for the comparison of prebum and postburn content is shown ($n=4$).

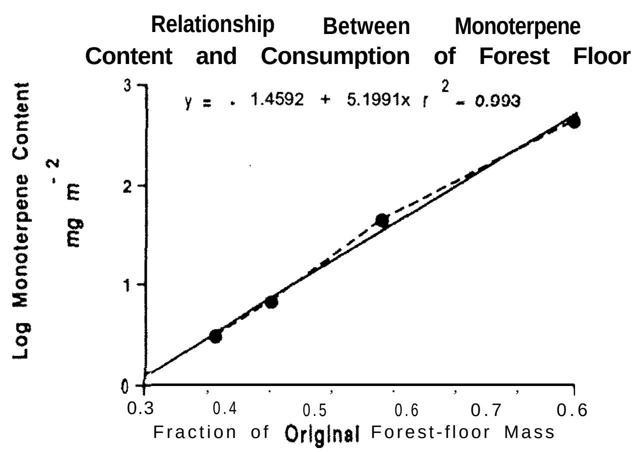

Fig. 6 Relationship between monoterpene content of the combined postburn organic and 0- to 10-cm mineral soil horizons and the amount of consumption (expressed as the remaining fraction of original forest-floor mass) by the prescribed burning.

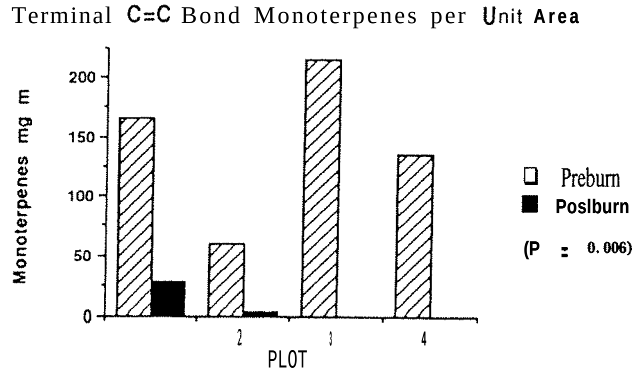

Fig. 7. Content of monoterpencs that contain terminal unsaturated carbon-carbon bonds for the combined organic and 0- to 10-cm mineral soil horizons before (Prebum) and after (Postbum) prescribed burning. Probability level for the comparison of prcbum and postburn content is shown (n=4).

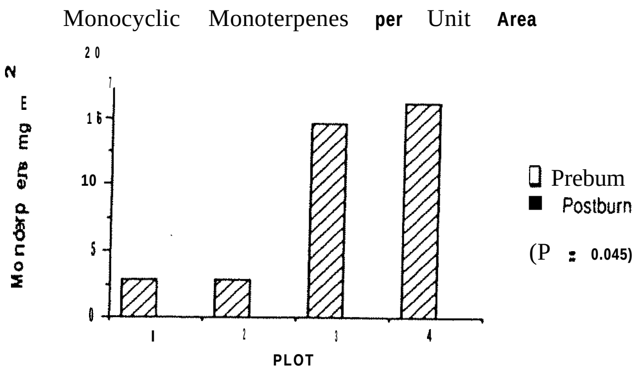

Fig. 8. Content of monocyclic monoterpcnes for the combined organic and 0- to IO-cm mineral soil horizons before (Prcbum) and after (Posthum) prescribed burning. Probability level for the comparison of prebum and postburn content is shown (n =4).

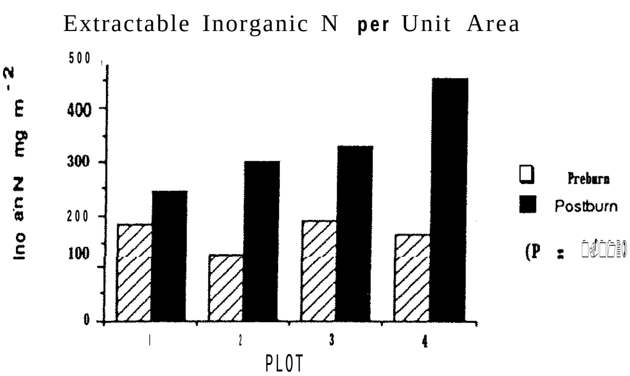

Fig. 9. Extractable inorganic N content of the 0 horizon before (Prebum) and alter (Postbum) prescribed burning. Probability level for the comparison of prebum and postburn content is shown (n =4).

N, shown by comparison of prebum values in figures 9 and 11). Although the burned forest floor showed the same basic pattcrn of net immobilization (difference in postburn values in figures 9 and 11), the potential N mineralization and nitrilication pattern for the burned forest floor substantially deviated from the pattern shown by the pre-burned forest floor in two ways. First, net inorganic N levels in the burned forest floor began to increase rapidly over the final 3 weeks of incubation (fig. 12), whereas the pre-burned forest floor showed very little increase over the same period. Second, forest floor samples from all of the burned plots produced NO;-N during the 10-week incubation, whereas all of samples from the pre-burned plots showed no detectable amount of NO;-N after the IO-week incubation. When the amount of NO;-N present at the end of incubation in the four burned forest floor samples was compared with the amount in the preburned samples (none in all 4 prcbumed), the difference was not statistically significant (P=0.18). In part, the lack of significance was due to the large variation in the amount of NO;-N produced by the burned forest floors. The plot that had the greatest reduction in forest-floor mass (plot 4) produced the most NO_3^--N, but NO_3^--N production was not significantly correlated (P > 0.10) either with forest floor consumption or with the amount of NH,‘-N present in the sample (representing substrate for nitrilication; fig. 13).

DISCUSSION

The plots burned in the present study were the control plots for a previous study (White 1986a) on the effects of prescribed burning in ponderosa pine. In the previous study, there was a greater range in prebum biomass (from 1650 to 3590 g m^{-2} ash-free weight) and in the amount of forest floor consumed by the bum (from 150 to 2070 g m^{-2}). The prescribed burning treatments were conducted under similar conditions (same prescription in both studies) and had similar characteristics, and therefore had very similar effects on soil inorganic N levels. In both studies, the increase in forest-floor inorganic N was proportional to the amount of forest floor consumed, with all the increase as NH,‘-N. This relationship was highly significant in the first bum (r^2=0.97, P<O.Ol; White 1986a), where the range in forest floor consumption was greater, but was not as strong in the current study (r^2=0.78). Burning did not significantly change mineral soil inorganic N levels in either study, except in the plot where consumption of forest floor was greatest (White 1986a). Potential N mineralization and nitrilication in the mineral soil was unchanged in both studies.

Prescribed burning reduced weight of monoterpenes proportionally more than it reduced forest-floor mass. Removal of the upper organic horizon with the highest monoterpene concentration could explain a large portion of the decline, but monoterpene concentrations after burning were even lower than those measured in pm-burned F-H

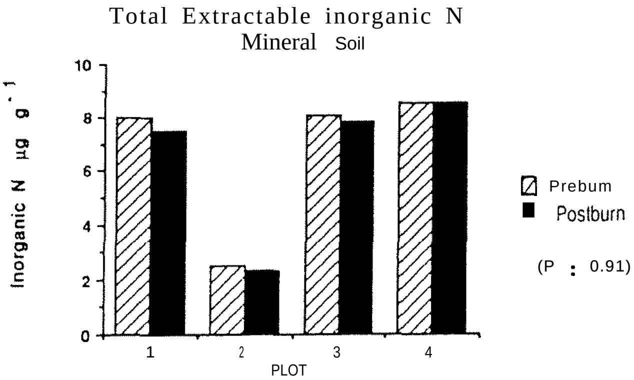

Fig. 10. Extractable inorganic N content of the 0- to IO-cm mineral soil horizon before (Prebum) and after (Postbum) prescribed burning. Probability level for the comparison of prebum and postburn content is shown (n=4).

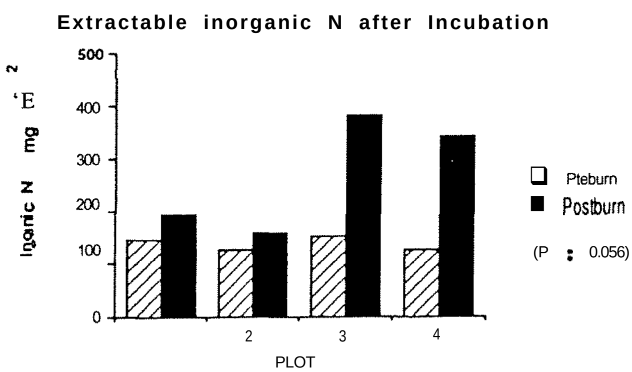

Fig. 11. Extractable inorganic N content of the 0 horizon before (Prebum) and after (Postbum) prescribed burning at the end of IO-week aerobic incubation for potential N mineralization. Probability level for the comparison of preburn and postburn content is shown (n=4).

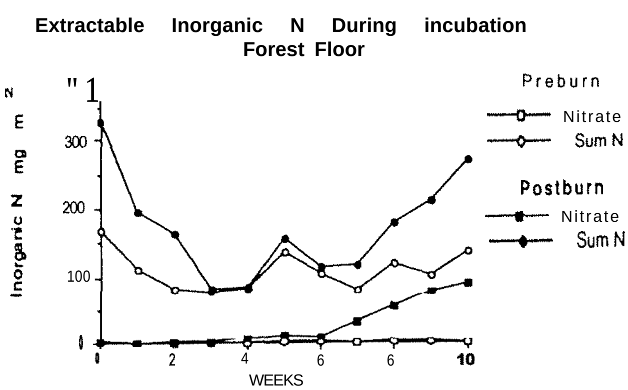

Fig. 12. Mean values (n-4) of extractable nitrate and ammonium + nitrate (Sum N) content of the 0 horizon before (Prcbum) and after (Postbum) prescribed burning at weekly intervals during aerobic incubation at 20 °C.

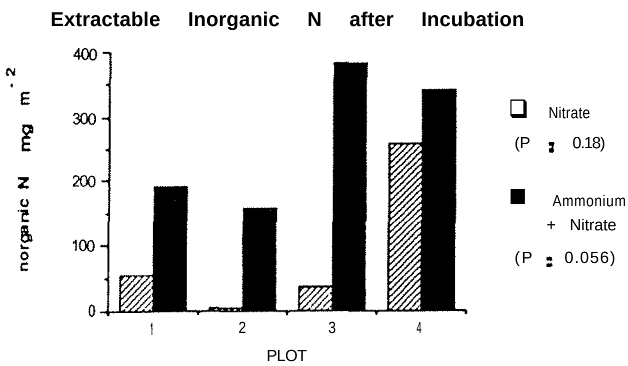

Fig. 13. Extractable nitrate and ammonium + nitrate content of the postburn 0 horizon at the end of IO-week aerobic incubation for potential N mineralization. Probability level for the comparison of prebum and postburn nitrate and ammonium + nitrate content is shown (n=4).

horizons in 4 other collections before the bum (White IN PRESS). The lower-than-expected concentrations suggest that monoterpcnes were volatilized and probably cornbusted. The relationship between monoterpene concentration and residual forest-floor mass (fig. 6) suggests that reduction in forest-floor fuels by 50 percent can reduce monoterpcnes by over 90 percent. Removal of monotcrpenes would reduce the probability of fire and enhance the potential for higher rates of N mineralization and nitrification. White (1986a) observed increased rates of N mineralization in the forest floor and mineral soil in burned plots 10 months after treatment. If burning that reduces forest-floor mass by 50 percent results in increased soil moisture as in other studies in ponderosa pine (Haase 1986, Ryan 1978), field conditions will become more favorable for N mineralization. Thus, bums that consume half of the forest floor could increase site fertility significantly.

ACKNOWLEDGMENTS

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LOSS, RETENTION, AND REPLACEMENT OF NITROGEN ASSOCIATED WITH SITE PREPARATION BURNING IN SOUTHERN PINE-HARDWOOD FORESTS

Lindsay R. Boring, Joseph J. Hendricks, and M. Boyd Edwards[1]

Abstract-High-intensity site preparation burning is a common forest regeneration practice on harvested pine and mixed pine-hardwood sites in the southeastern USA. This practice could result in excessive losses of forest floor organic matter and nitrogen, and could subsequently decrease long-term productivity. In general, intensive burning may result in large losses of forest floor nitrogen, primarily by combustion and convection of N gases. However, past studies may have overestimated combustion losses due to inadequate knowledge of potential gaseous N retention mechanisms. Long-term inputs of N from biological fixation and atmospheric deposition may replace large amounts of N lost from tire, but more information on N_2-fixation processes is needed over time and space. Also, additional studies are needed to determine practical N_2-fixation management applications, such as retention of coarse woody debris and enhancement of N_2-fixing plant populations.

INTRODUCTION

High-intensity site preparation burning is a common and cost-effective regeneration tool on harvested pine and mixed pine-hardwood forest sites in the southeastern United States (Abercrombie and Sims 1986). Few studies have comprehensively examined the immediate and potential long-term impacts of this practice upon site productivity (Van Lear and Johnson 1983; Van Lear and Waldrop 1986). Site preparation effects are a central concern of many forest managers since the technique may alter forest floor organic matter and nutrient reserves that provide long-term site productivity for future stand rotations (Van Lear and others 1983).

Although low-intensity prescribed burning has been shown to have no deleterious effects upon the productivity of loblolly pine forests in the southeastern Coastal Plain (McKee 1982), high intensity site-preparation burning could result in excessive losses of forest floor organic matter and nitrogen. Several studies have provided gross mass-balance estimates of organic nitrogen loss in logging residues up to several hundred kg ha^{-1} from site-preparation burning (Van Lear and Kapeluck 1989). However, few detailed, process-level studies have been conducted to examine forest floor nitrogen losses and the mechanisms of nitrogen loss and retention during and following burning (fig. 1; Jorgensen and Wells 1986; Little and Ohmann 1988; Van Lear and others 1983). In addition, long-term replacement of nitrogen through biological fixation and other inputs has not been adequately examined to assess the potential contributions to the nitrogen balance of intensively-burned sites (Boring and others 1988, Hendricks 1989).

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In this paper, we summarize the state of knowledge concerning key processes regulating N losses, retention, and replacement associated with intense fire in southeastern pine and mixed pine-hardwood forests, as well as in other related ecosystems. We synthesize the research results and perspectives from our work, as well as those from other investigators, and identify key areas for further research.

NITROGEN LOSSES AND RETENTION

There are several mechanisms by which N may be lost during and following intensive burning, including combustion and convection of N gases and particulates, leaching of NO_3^- in the soil solution, denitrification of residual inorganic N, and erosion of organic matter by water and wind (fig. 1). Although each of these mechanisms may pfay a role in the loss of N reserves, past studies suggest that the combustion losses may exceed the others in importance (Christensen 1987; Raison and others 1985; Van Lear and Waldrop 1989).

Intensive burning results in large losses of N by volatilization and the removal of gases and particulates in wind currents (Jorgensen and Wells 1986). The amount of N lost varies and depends primarily on the fire severity, which in turn is determined by the amount and flammability of organic matter, moisture conditions, and the residence time of the peak thermal pulse. Mass balance estimates based upon loss of woody debris and other forest floor fuels have indicated losses may range as high as 300 kg ha“ for severe site-preparation fires in the southeastern Piedmont region (Van Lear and Kapeluck 1989; Wells and others 1979), and up to 1000 kg ha” for intense fires in western Douglas-fir forests (Binkley 1986; Little and Ohmann 1988).

Although a considerable amount of the lost N may have originated from the smaller-sized fuels of residual logging slash and young colonizing vegetation, the potential losses from the original forest floor and soil organic matter may have the greatest impact upon N balance and long-term site

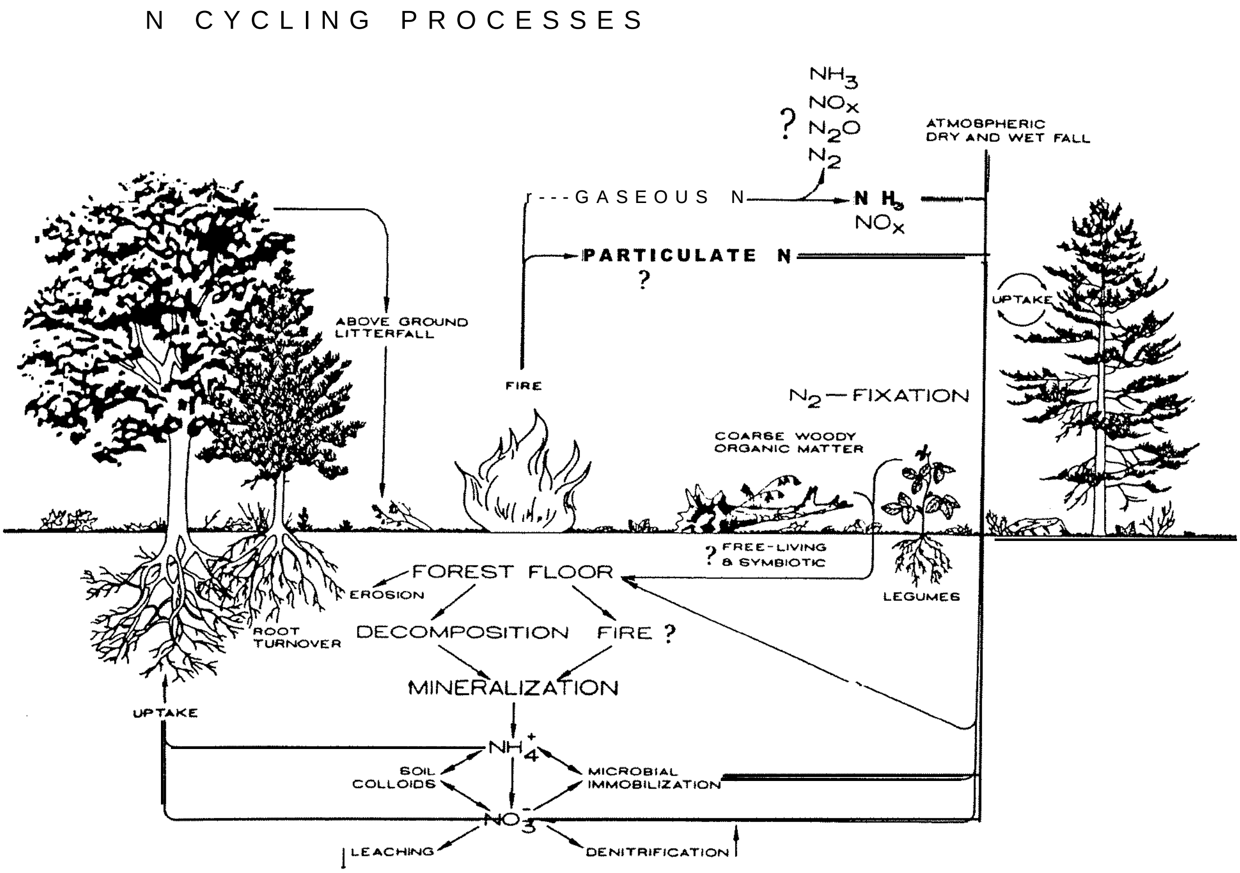

Figure 1. Nitrogen cycling processes and their relationships to losses, retention, and replacement

productivity (Van Lear and Kapeluck 1989). Losses of N from forest floor mass have been estimated to range from 130 to 170 kg ha“ (Van Lear and Kapeluck 1989), depending primarily upon the severity of the bum, with hotter and slower tires causing greater losses (Knight 1966).

Most N loss from tires is associated with gaseous loss rather than particulates (Raison and others 1985). Using controlled ignitions in a muffle furnace, volatile loss of N was shown to begin at approximately 300°C and increase to 60 percent at 700°C. Such controlled experiments formerly assumed that most of the N was lost as N_2, and that it was convected offsite following oxidation (DeBell and Ralston 1970; Knight 1966). However, other workers have shown that ammonia gases may be liberated at temperatures as low as 100°C, and subsequent information has revealed more complex processes regulating these losses (Raison 1979).

Combustion studies with southern pine litter-soil systems have shown a complex scenario of N loss, which includes significant volatilization of NH, and NO, at relatively low temperatures (Lewis 1975). Although we now appreciate the complexity of N volatilization, we still have few detailed case studies that accurately document N losses from southern pine and pine-hardwood forests. We also suggest that there may be retention mechanisms in southern pine forests which have not been examined by early studies (Lewis 1975; Raison 1979).

Some researchers have reported surprisingly variable and low losses of N from burned forest floors of different types, and some of the increases in inorganic N availability have been attributed to rapid increases in nonsymbiotic N fixation or to the occurrence of legumes on the sites (Jorgensen and Hodges 1970, Mroz and others 1980). Although replacement of N from these sources is known to occur, some of these rapid increases in inorganic N availability are not explainable by decomposition and nitrogen fixation since the increases appear immediately (hours to a day) following burning (Raison 1979).

Instead, it is likely that ‘thermal mineralization of organic matter occurs, where inorganic N is released and retained in significant amounts in subsurface layers of the mineral soil. Studies in some soil-plant systems have documented elevated inorganic N immediately following burning as NH_4-N, NO,-N, and possibly as labile organic compounds. This may happen by ammonia hydrolysis and vertical translocation as gases are volatilized by the initial thermal pulse of the tire and are subsequently condensed by cooler temperatures in lower soil mineral horizons (DeBano and others 1976, 1979; Kitur and Frye 1983). Although this process has been examined in North American chaparral and in Australian eucalyptus forests, the mechanism has not been documented in southern pine or pine-hardwood forests.

Biological transformations related to decomposition and nitrification become more important with time after burning, and play dominant roles in the welldocumented but short-term (weeks to months after burning) increases of inorganic N availability in southern pine forests (Christensen 1987; Raison 1979; Schoch and Binkley 1986). The magnitude and duration of such effects may be highly specific to individual forest floor types and bum characteristics, especially as related to initial litter quality and N contents. Mroz and others (1980) found widely variable ammonification, nitrification and immobilization responses among different forest floor types, and concluded that it was difficult to make universal generalizations about microbial processes following burning. Although stimulation of nitrification may be a key burning response in southern forest ecosystems, elevation or reduction of microbial immobilization and gaseous N_2O flux also play significant roles in regulating the availability of NO,-N (Christensen 1987). Nitrification and gaseous nitrogen transformations require further research and it is not clear to what degree N retention after burning may be quantitatively affected by either NO,-N leaching or gaseous losses. Further, these processes likely interact with the recovery rates of microbial and plant nitrogen uptake which may minimize N losses through biomass immobilization.

NITROGEN REPLACEMENT

Although site-preparation burning may cause large losses of forest floor nitrogen reserves, natural sources of replacement may help maintain or improve the productivity and quality of these forest ecosystems. Two major pathways of nitrogen replacement are atmospheric deposition and biological nitrogen fixation (Boring and others 1988) (fig. 1). Atmospheric deposition includes the input of a variety of nitrogen-containing compounds by both dry and wet modes. Atmospheric N_2 may be biologically fixed (and converted to organic N) by symbiotic and nonsymbiotic organisms. A thorough understanding of the magnitudes of these nitrogen inputs and their impacts on nutrient cycling is incomplete for any single ecosystem. Furthermore, knowledge of these fundamental processes is essential to accurately assess the impact of site-preparation burning on the nitrogen balance and long-term productivity of forest ecosystems.

Atmospheric Deposition

A variety of nitrogen forms may be deposited in terrestrial ecosystems from the atmosphere (Boring and others 1988). Some of these, such as dissolved NH,+ and NO_3^-, can be rapidly incorporated into terrestrial nitrogen cycles following wet or dry deposition. Other constituents, such as those in aerosol and gaseous forms, may be transferred directly to vegetation surfaces (Okano and Machida 1989). Many factors that may affect the spatial and temporal patterns of these deposition inputs. One is the proximity to nitrogen sources such as industrial emissions. Also, factors that regulate the transport and transformation of atmospheric nitrogen forms, such as precipitation patterns and meteorological conditions, are important considerations.

Measurements of all of the potential forms of nitrogen deposition are unavailable for any terrestrial ecosystem. However, although conservatively estimated, nitrogen inputs measured in bulk precipitation provide some measure of the relative importance of atmospheric deposition to forests. Estimates of wet deposition inputs to southeastern forest ecosystems may range from 5.1 to 12.4 kg ha^{-1} yr^{-1} (Kelly and Meagher 1986; Richter and others 1983; Riekerk 1983; Swank and Waide 1987; Van Lear and others 1983; Wells and Jorgensen 1975). The upper value of this range was recorded at Walker Branch in eastern Tennessee which is in close proximity to coal-fired power plants (Kelly and Meagher 1986).

Dry deposition may also contribute significant amounts of nitrogen to forest ecosystems. The dryfall contribution to the total NH,+ and NO; deposited in open collectors exceeds 20 percent at Coweeta (Swank and Waide 1987) and exceeds 50 percent at the Walker Branch site (Kelly and Meagher 1986). These figures primarily represent large particulate inputs and are not representative of all nitrogen in aerosols and gases which may originate from combustion processes. Combining both wet and dry N inputs, general estimates for southeastern forest ecosystems may range from 6 to 14 kg ha^{-1} yr^{-1} excluding aerosol and gas fractions which may also be high (fig. 2).

Nitrogen Fixation

In southeastern forest ecosystems, detectable nitrogen-fixation activity may occur in the forest floor and surface mineral soil horizons, coarse woody debris (CWD) on the forest floor, and the, nodules of symbiotic nitrogen-fixing plants. In general, the fixation rates of symbiotic organisms are greatest during the early successional stages of forest development (Boring and Swank 1984; Boring and others 1988). The harvesting of merchantable timber, followed by the felling of residual stems and intense site-preparation burning creates an early successional environment and may therefore promote detectable, if not significant, nitrogen-fixation activity among a variety of organisms.

Forest Floor and Soil

Clearcutting may increase nonsymbiotic nitrogen-fixation in the forest floor (excluding CWD) and soil through the transfer of large carbon pools in the form of dead roots and small fractions of logging debris (Boring and others 1988). At the Coweeta Hydrologic Laboratory, maximum fixation rates of 4-6 kg ha^{-1} yr^{-1} were measured three to five years following clearcutting (Waide and others 1987). These high rates are likely to be short-lived, however, due to the sensitivity of nitrogen-fixing bacterial populations to substrate quality, moisture, temperature, and pH (Jorgensen and Wells 1986;

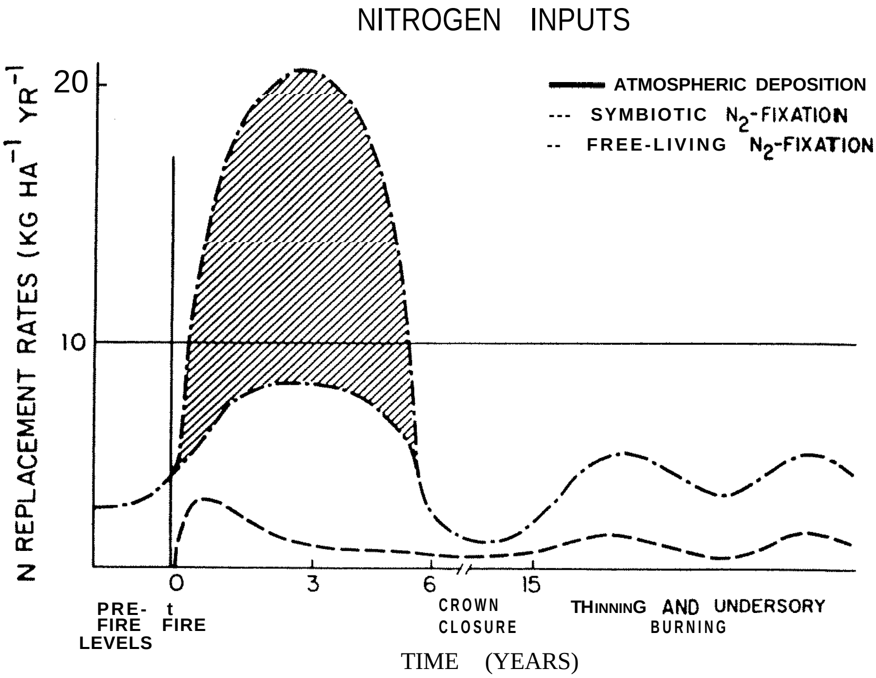

Figure 2. Range of temporal dynamics of nitrogen replacement in relation to forest succession; based upon studies of sites with at least moderate N-fixing plant populations.

Vance and others 1983). Most estimates of nonsymbiotic fixation in southeastern temperate coniferous and deciduous forest are much smaller and range from 0.1 to 3.7 kg ha^{-1} yr^{-1} (Di Stefano and Gholz 1989; Gholz and others 1985; Grant and Binkley 1987; Jorgensen and Wells 1986; Van Lear and others 1983).

There appears to be a paucity of information concerning the effect of intense site-preparation burning on nonsymbiotic forest floor and soil nitrogen fixation. However, research on the effects of less intense understory burning on this form of fixation have been contradictory. Maggs and Hewett (1986) reported a three-fold increase in nitrogen-fixation activity following understory burning, whereas Vance and others (1983) and DiStefano and Gholz (1989) reported little or no response to burning. Even given these conflicting results, it is generally assumed that such low nonsymbiotic fixation activity in the forest floor and soil will not make a significant contribution to the replacement of nitrogen following intense site-preparation burning.

Coarse Woody Debris

Silvicultural clearcutting of southeastern forests may deposit from 15 to 50 tons ha^{-1} of non-harvestable coarse woody debris (CWD) to a site (Sanders and Van Lear 1988). The majority of this CWD remains on the site following burning since even the most intense fires generally consume stems only up to 10 centimeters in diameter. This CWD may play a significant nitrogen replacement role by serving as a carbon substrate for a highly diverse group of nonsymbiotic nitrogen-fixing organisms, as well as for wood-eating insects that form symbiotic relationships with nitrogen-fixing gut bacteria (Dawson 1983; Roskoski 1980; Silvester and others 1982).

Research conducted in old-growth Douglas-fir stands of the Pacific Northwest has revealed that nonsymbiotic fixation rates in large masses of well-decomposed wood by such organisms as bacteria, fungi, blue-green algae, and lichens may reach 1.4 kg ha-' yr^{-1} (Silvester and others 1982). In a northern hardwood forest at the Hubbard Brook Experimental Forest, comparable annual nitrogen fixation rates in decaying wood ranged from 0 to 2 kg ha^{-1} yr^{-1}, with the rates being directly correlated to the standing crop of decaying wood (Roskoski 1980, 1981). Although the nitrogen replacement role of CWD in southeastern forest ecosystems is less well understood, Todd and others (1975, 1978) reported values of 1.22 to 1.66 kg ha^{-1} yr^{-1} for an oak hickory forest at Coweeta and DiStefano and Gholz (1989) measured high specific rates in low masses of decomposing wood of slash pine plantations in North Florida.

It is important to note that these area estimates of non-symbiotic nitrogen fixation are a function of CWD biomass as well as actual rates of fixation activity. Also, Silvester and others (1982) suggested that fixation rates generally increase as the CWD, which may remain on the site for several years, becomes more highly decomposed. Therefore, nonsymbiotic fixers (even with low specific activity) in the abundant CWD of southeastern clearcut sites may Potentially contribute significant amounts of nitrogen during the decomposition process of the debris.

As stated earlier, CWD serves as a substrate for wood-eating fauna such as termites, bark beetles, and cockroaches, which may contain nitrogen-fixing bacteria in their gut (Dawson 1983; French and others 1976; Potrikus and Breznak 1977). Among these organisms, the process of nitrogen fixation has been best established for temperate and tropical termites

(Bentley 1984; Prestwich and others 1980). Due to numerous difficulties preventing accurate measurements, nitrogen fixation values for termites have not been related to nitrogen-balance estimates, although their contribution could be signiiicanl on some forest sites.

Higher Plants

Symbiotic nitrogen-fixing plants commonly occur on sites where nitrogen is limiting, early successional areas, or sites subject to substantial N loss, such as from intense and/or frequent fires (Boring and others 1988). In southeastern forest ecosystems, fire-adapted native herbaceous legumes generally thrive during the first few years following clearcutting and intense site-preparation burning (Cushwa and others 1969; Czuhai and Cushwa 1968).

Legume populations are typically largest during early stages of stand development for a number of reasons. The seed of several legumes have their highest germination rates following scarification with moist heat at temperatures approaching 80" C (Cushwa and others 1968, 1970). Also, it is believed that these seeds remain viable in the litter layer and soil for prolonged periods of time. Finally, these fire-adapted, early successional species thrive under the high light environment of pine stands prior to crown closure (Brunswig and Johnson 1972; Cushwa and others 1971).

Nitrogen fixation rates of these herbaceous legumes are predicted to be highest during this early stage of stand development (fig. 2). The absence of a developed overstory and the reduced competition from non-fire adapted species should result in larger quantities of photosynthate available for the energetically expensive nitrogen fixation process. Hendricks (1989) assessed the *nitrogen* fiiation activity of three dominant legume species (Desmodium viridiflorum, Lespedeza hirta, and L. procumbens) in a cleared and burned area of a Georgia Piedmont pine forest. In the early to middle part of the growing season, D. viridiflorum and L. procumbens exhibited *specific acetylene* reduction activity per nodule biomass comparable to those of black locust, Robinia pseudoacacia (Boring and Swank 1984), and greater than those of many actinorhizal nitrogen-fixing species, although total nodule biomass was considerably lower (Binkley 1981; McNabb and Geist 1979; Tripp and others 1979). This nodule activity, however, declined substantially during the remainder of the hot and dry growing season, and the third species L. hirta was rarely observed to nodulate.

The total amount of nitrogen fixed by herbaceous legumes on these Georgia Piedmont sites was roughly estimated to range from <OS kg N ha" yr^{-1} for areas with small legume populations (500 - 700 individuals ha") to 7 - 9 kg N ha^{-1} yr^{-1} for areas with relatively large populations (20,000 - 30,000 individuals ha^{-1}; Hendricks 1989). These values represent broad ranges of estimates and illustrate the potential importance that factors controlling the spatial and temporal variation of legume populations may have on forest floor and soil nitrogen reserves. Sites with larger legume populations than ours may have substantially more nitrogen fixation.

The spatial variation of herbaceous legumes depends upon many factors including the fire-history of the site. Table 1 gives the results of a legume population survey for two similar Georgia Piedmont sites that were cleared and burned following southern pine beetle infestation. These sites, which were sampled two years following the intensive bum, differed primarily in their fire history as one had no previous burning and the other had received periodic winter understory bums since 1962. The site that had been managed under a burning regime had a substantially higher diversity and density of

Table 1. Density (#/ha) of native and naturalized herbaceoup legume species on cleared and burned sites in the Georgia Piedmont. Site 1 had no previous burning history, uhereas site 2 has been managed under a 4 year burning regime since 1962

SPECIES	DENSITY (#/ha) SITE 1	SITE 2
Cassia nictitans	12	3,429
Centrosema virginianum	12	381
Crotalaria sagittalis		143
Desmodium		
cil i-are		238
laevigatum	32	1,048
marilandicum	74	2,333
nuttallii		1,238
rotundifolium		381
tenuifolium		333
viridiflorum	28	3,952
Other spp.	3	334
Lespedeza		
bicolor'		619
cuneata'	42	667
hirta		3,333
nuttallii		1,048
procunbens	380	6,905
virginica	6	2,333
Other spp.		476
Tephrosia		667
TOTAL	731	29,858

legumes due primarily to increases in species of Desmodium and Lesoedeza (table 1). Although these data only represent two sites, other reports also underscore the importance of regular burning to generate a cycle of high seed germination and plant establishment rates (Cushwa and others 1966, 1969, 1970; Czuhai and Cushwa 1968; Devet and Hopkins 1967; Speake 1966).

Brunswig and Johnson (1972) studied the temporal variation of legume populations in southeastern pine plantations during the first seven years following intense site-preparation. The results indicated that annuals, primarily C. nictitans, were the predominant legumes in one-year old plantations. By the third year, perennial legumes (primarily species of Desmodium and Lespedeza) were more abundant than annuals. In older stands, the annuals were essentially eliminated and the perennials decreased substantially due to crown closure. However, in latter stages of stand development when light gaps appear and understory burning is commonly initiated, legumes may still influence nitrogen availability via the temporal dispersal of a viable seed bank.

Approximately 300 native legumes occur in the southeastern United States (Wilbur 1963). A majority of these are herbaceous legumes that are tolerant of acidic soils, shading, and litter accumulation on the forest floor, and commonly occur in pine and mixed pine-hardwood forest ecosystems. Although the value of these legumes to wildlife has long been recognized, their nitrogen accretion role as well as their contribution to functional biodiversity are just beginning to be recognized in southeastern forest ecosystems.

SUMMARY AND RESEARCH NEEDS

Our present knowledge of the impacts of intensive fire upon ecosystem nitrogen cycling processes and long-term productivity in southern forests is incomplete, and many key questions require additional research. We generally know that large amounts of forest floor nitrogen are lost with severe bums, and that long-term replacement sources may be potentially significant but variable. However, in the past we may have overestimated N volatilization on some sites by not examining potential retention processes in humus and mineral soil horizons below the litter. Simultaneously we have not examined the potential for additional and smaller short-term gaseous losses via N_2O flux from residual inorganic N following burning, or examined the interactions of immobilization by microbial or plant biomass with nitritication and gaseous N flux (Matson and Vitousek 1987).

Our interpretation of intensive fire effects upon long-term N balance is uncertain until we better understand how adequately nitrogen fixation processes replace N losses over time and space. We need to better understand how to more effectively manage nitrogen fixation inputs. More effective management may require retention of coarse woody debris, increased populations of nitrogen-fixing plants, and modification of environmental factors that control their fixation rates (e.g. P fertilization). A more complete understanding of these processes and their potential management need to be integrated into a whole-ecosystem perspective, possibly with the use of simulation models, to better interpret their impacts upon long-term N balance and forest productivity over several stand rotations.

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FIRE EFFECTS ON NUTRIENT POOLS OF WOODLAND FLOOR MATERIALS AND SOILS IN A PINYON-JUNIPER ECOSYSTEM

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Abstract-The total pools of carbon and nitrogen in the litter, duff, and soil compartments were quantified in a mature pinyon-juniper woodland. One hectare of the woodland was burned in a fire similar to a severe wildfire, where almost all aboveground vegetation was killed. Soil temperatures reached as high as 325” C at 5 cm below the soil surface. The total nutrient pools were again quantified after fire indicating significant losses of both C and N from forest floor material. The litter loss of C was as high as 92 percent and N loss was as high as 88 percent. Loss of C from the duff layer was estimated from 78-80 percent, while nitrogen loss was 75 percent. Soil N displayed a slight, significant increase under canopy, but not in the interspaces. The greatest initial effect was the reduction of the C/N ratios favoring mineralization. The total amount of C and N lost from the woodland was 12.6 Mg ha” C and 167 kg ha^{-1} N.

INTRODUCTION

Pinyon-juniper woodlands occupy a significant expanse of the semiarid United States (Klopatek and others 1979). They are characterized by extreme variability in climate, soil, and topography that produces units of canopy-covered and non-canopy covered (interspace) patches. Therefore, a seemingly uniform disturbance, such as fire, to this patchy mosaic ecosystem type may result in complex, non-uniform responses. Despite the expansiveness of the pinyon-juniper woodlands and their susceptibility to fire, studies of fire effects in pinyon-juniper woodlands have only been recent and limited (DeBano and others 1987; DeBano and Klopatek 1988; Gifford 1981; Klopatek and others 1988).

Studies on the effects of fire upon soil nutrients, and specifically soil N, have produced conflicting results. Some investigators have reported no significant changes in total soil N contents (Covington and Sackett 1986; Jurgensen and others 1981; Kovacic and others 1986; White 1986). Waldrop and others (1987) found no changes in either soil organic matter or total N after 30 years of various prescribed burning treatments in loblolly pine.

Stock and Lewis (1986) reported significant increases in N that were due to leaching of ash material in fynbos (chaparral) ecosystems, whereas DeBano and others (1979) reported a significant loss of total N from laboratory burnings of chaparral soils. Fuller and others (1955) described that loss of soil N was proportional to the fire intensity in ponderosa pine forests with a concomitant decrease in C/N ratios. Grier (1975) noted significant nutrient losses from an intense fire on the eastern slope of the Cascade Mountains of Washington and estimated a loss of 855 kg ha^{-1} of N. Covington and Sackett (1986) found a 37% reduction in forest floor material following a low-intensity prescribed fire in ponderosa pine forest with an additional 20% reduction 7 months after burning, but no significant soil N loss.

In a comprehensive study of the effects of prescribed burning, Wells (1971) noted that while periodic bums had caused significant losses of forest floor material immediately after the bum, there seemed to be a tendency for the system to regain its organic matter (both C and N) over time and approach the control condition. Additionally, he found a small increase in available P, but the increase may be short lived in calcareous soils (DeBano and Klopatek 1988).

Christensen (1973) found greater NH, availability and nit&cation rates following burning in chaparral, and Jurgensen and others (1981) found that broadcast burning caused a minor net loss of N (approximately 100 kg ha”), but resulted in greatly enhanced soil N, nitrification, and base cation availability from a clearcut site in Montana. They concluded that no significant long-term losses of N occurred. Schoch and Binkley (1986) found that prescribed burning increased decomposition rates and N availability (as indexed by incubation) in loblolly pine stands in North Carolina. Similar results of increased levels of NH, and NO, have been reported in ponderosa pine (Covington and Sackett 1986; Kovacic and others 1986; White 1986) and Douglas-fir ecosystems (Jurgensen and others 1981). Increases in NH_4 are presumably the result of physicochemical reactions caused by the elevated temperature (Kovacic and others 1986). Subsequent, post-fire increases in NO, are attributed to increased nitrification rates, that are possibly enhanced by the reduction of allelochemical compounds (White 1986).

In summary, the results of fire on C and N losses from shrubland and coniferous forest ecosystems are mixed, but appear related to fire intensity. With this as a background, we attempted to document changes in the total pools of C and N in a pinyon-juniper woodland resulting from fire. The objective of the study was to determine if the patchy nature of

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this ecosystem was reflected in the fire effects on its nutrient pools. Initially, we conducted our experiment using microcosms and determined that there might be a substantial loss of nutrients from the ecosystem (Klopatek and others 1990). Here, we report on the effects of the subsequent field study. While a complete inventory of the pools of both C and N was undertaken in this ecosystem, only the effects of fire on the forest floor and soil components are reported on in this paper.

METHODS

In September, 1989 a bum was conducted of a l-ha section of mature pinyon-juniper woodland in the Kaibab National Forest in northcentral Arizona. The meteorological conditions at the beginning of the bum were: a wind speed of 6.0 m sec.‘, temperature of 27” C, and a relative humidity of 18 percent. Moisture content of the large downed fuel and litter was 9 and 6 percent, respectively. Fire temperatures were monitored at the litter surface and the 0, 2, 5, and 10 cm depth of soil by a series of chromel-alumel thermocouples connected to a data logger. Burning was conducted by U.S. Forest Service personnel from the Tusayan Ranger District and Grand Canyon National Park staff. The bum and a nearby control site were fenced with barbed wire to prevent disturbance from local Livestock. Prior to the bum, an inventory was conducted of the biomass of all trees, shrubs and herbaceous species on the site using previously developed allometric relationships. The site contained 340 pinyons and 167 juniper ha” (ail of which were marked with permanent brass id tags) with a tree cover of 4286 m^2 ha-‘. Aboveground tree biomass totaled 135.2 mt ha”. Tree-ring analysis revealed that many of the trees were > 350 years old without any fire scars and, combined with replacement patterns, indicated a substantial time since intense burning. Soils were primarily sandy loams with Ph values ranging from 7.3 to 8.1. The site was part of the Dillman mature site described elsewhere in this volume (C.C. Klopatek and others 1990).

The site was divided into four relatively equal-sized quadrants. Samples were taken of litter and duff accumulations from under the mid-canopy of four randomly selected trees of each species in each quadrant using 2.5 cm x 25 cm quadrats. Five quadrats were sampled for litter in the interspaces of each quadrant. There was no duff in the interspaces. These measurements were combined with identical measurements taken from the nearby control site to yield a statistical population (n = 32) used to calculate regression coefficients of total C and N relative to the cross-sectional area of the bole and canopy cover.

Subsamples of litter and duff were taken back to the laboratory for analysis of organic C and total N. Additionally, soil samples were taken from the O-10 and 10-20 cm depth along with the litter and duff samples, yielding 16 soil samples from each depth and in each cover type. Sampling was repeated from 24 to 48 hrs post-bum when the canopy sites had cooled down. Although a qualitative separation of litter and duff ash were possible for estimation of loss by ignition, samples were combined for nutrient analysis.

Litter, duff, and ash samples were analyzed for C content by ashing in a muffle furnace at 550” C for 8 hrs with the oxidized portion multiplied by 0.58 to yield organic C; soil organic C was analyzed by the Walkley-Black Method (Page and others 1982). All material was analyzed for N by digesting the material following methods of Raveh and Avnimelech (1979) and measuring the digest with a Wescan ammonia analyzer. Statistical analyses were conducted using simple linear regressions for calculating duff and litter biomass per tree and AOV procedures for measurements of before and after burning differences (SAS 1985).

RESULTS

To understand the effects of fire *on* the nutrient pools in the landscape it was necessary to quantify the partitioning of nutrients relative to the patch mosaic of the woodland. We multiplied the total amount of pre-bum C and N in the litter, duff and soil (0 - 20 cm depth) per unit area by the area occupied by each cover type to obtain the total woodland floor and soil nutrient pools. Figures 1a,b show that, despite the fact that the interspace and its associated vegetation covered 57.1 percent of the area compared to 42.9 percent by the trees (28.4 pinyon, 14.5 juniper), the trees partitioned the majority of the resources under their canopies, accumulating nearly 68 percent of the total C and 70 percent of the total N.

There existed significant differences between the pre-bum organic C concentrations of litter, duff, and soil material of all three cover types (table 1). Total N concentrations of pinyon, juniper, and interspace litter were not different. The C:N ratios of pinyon and juniper litter (83 and 77, respectively) were similar but differed with that of the interspace litter (120). Higher C:N ratios for the interspace litter compared to under the canopies is typical of mature pinyon-juniper ecosystems in Arizona and may represent greater competition for available N in the interspaces (Klopatek 1987). Total N of litter and duff were not statistically different under pinyon but were under juniper; both differed from their respective underlying soil.

Peak fire temperatures experienced under the canopy were as high as 374, 305 and 260” C at the 2, 5, and 10 cm soil depths and 68” C at the 2 cm depth of the interspaces. The effect of fire was most noticeable on the litter and duff pools of the tree patches. Figures 2 and 3 represent the amount of C and N per unit area by cover type. Organic C was reduced an average of 92 and 80 percent in pinyon litter and duff, and 91 and 78 percent of the juniper litter and duff, respectively (fig. 2a,b). The resulting post-fire concentrations of organic C did not differ between types (table 1).

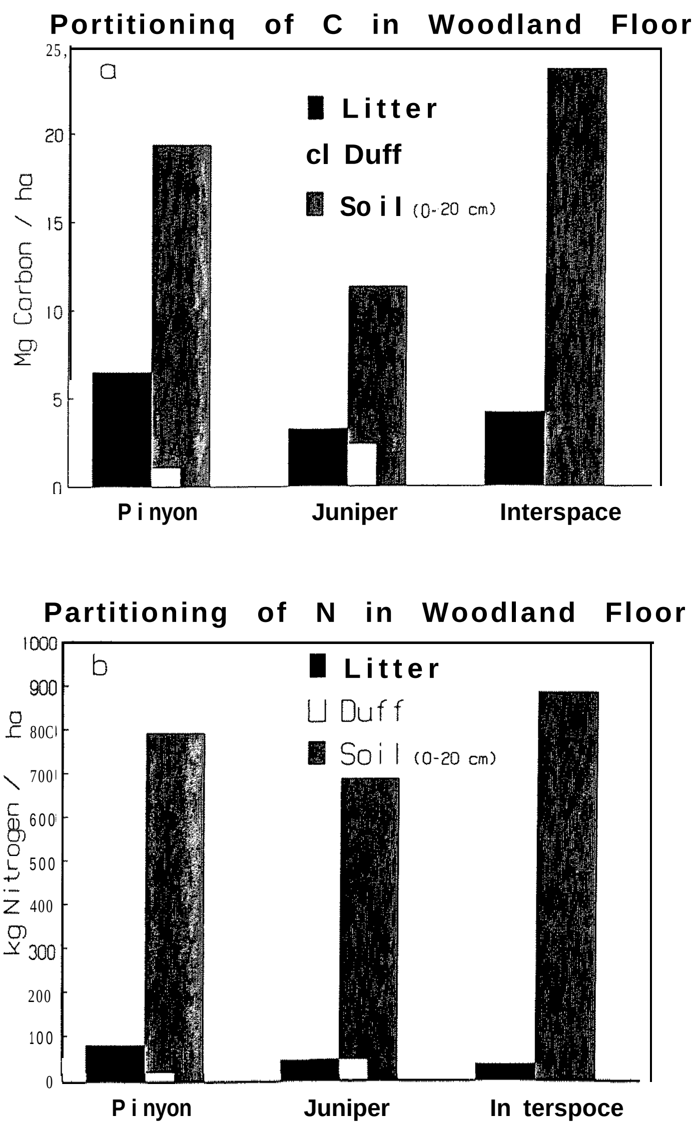

Figure l.-Total organic carbon (a) and nitrogen (b) in a pinyon-juniper woodland floor located in northern Arizona.

Nitrogen exhibited similar results (fig. 3a,b), but the relative concentration of total N did not decrease as much as C (table 1). If corrections are made for weight loss due to combustion, loss of N from the pinyon litter and duff compartments was 87 and 75 percent, and 88 and 75 percent of the juniper litter and duff, respectively. The loss of C and N closely parallel the results reported by Raison and others (1985). An important result of the fue is the change in the C:N ratios of the ash as compared to the litter and duff (table 1), being significantly reduced for both pinyon and juniper.

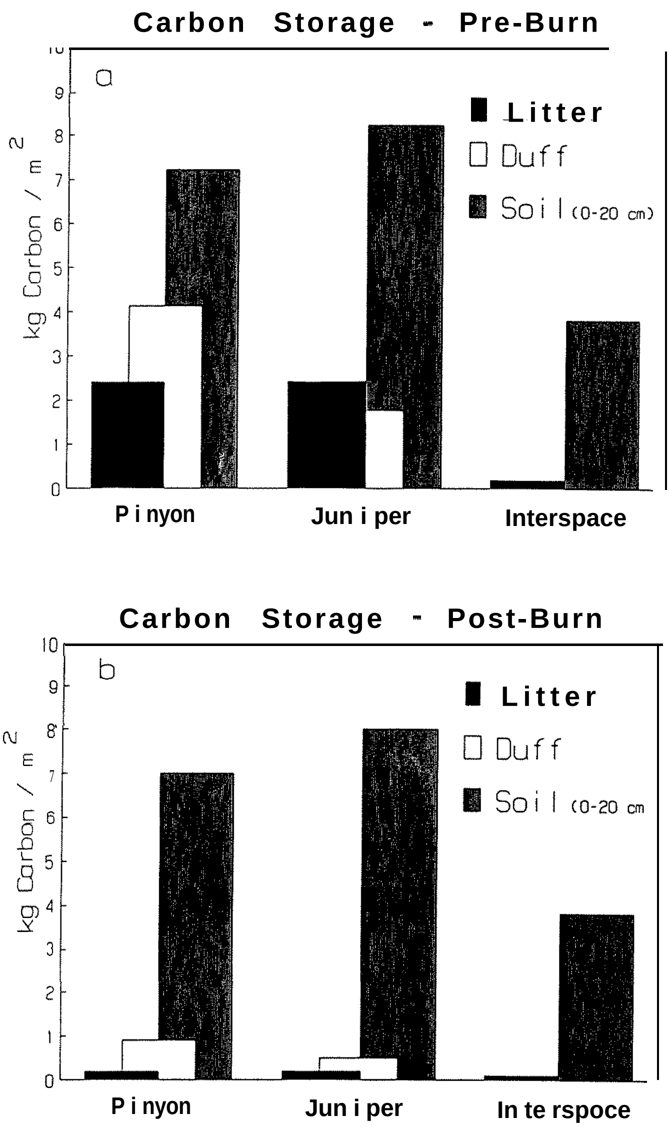

Figure 2.--Carbon storage pre- (a) and post- (b) bum in the woodland floor of a pinyon-juniper ecosystem in northern Arizona.

The effect of burning on total N and organic C were less pronounced in the underlying soil compared to the overlaying duff and litter. Both pinyon and juniper soils at the O-10 cm depth displayed significant increases in concentrations of total N (table 1) although no significant differences in organic C. Decreases in the soils at the 10-20 cm depth were not significant. The interspace soils followed a similar pattern, but differences were not significant. Increases in both C and N in underlying soils are consistent with reports that organic materials are translocated downward in the soil during burning and are wicked up from lower depths (DeBano and others 1976). Soil C:N ratios exhibited no significance before and after differences.

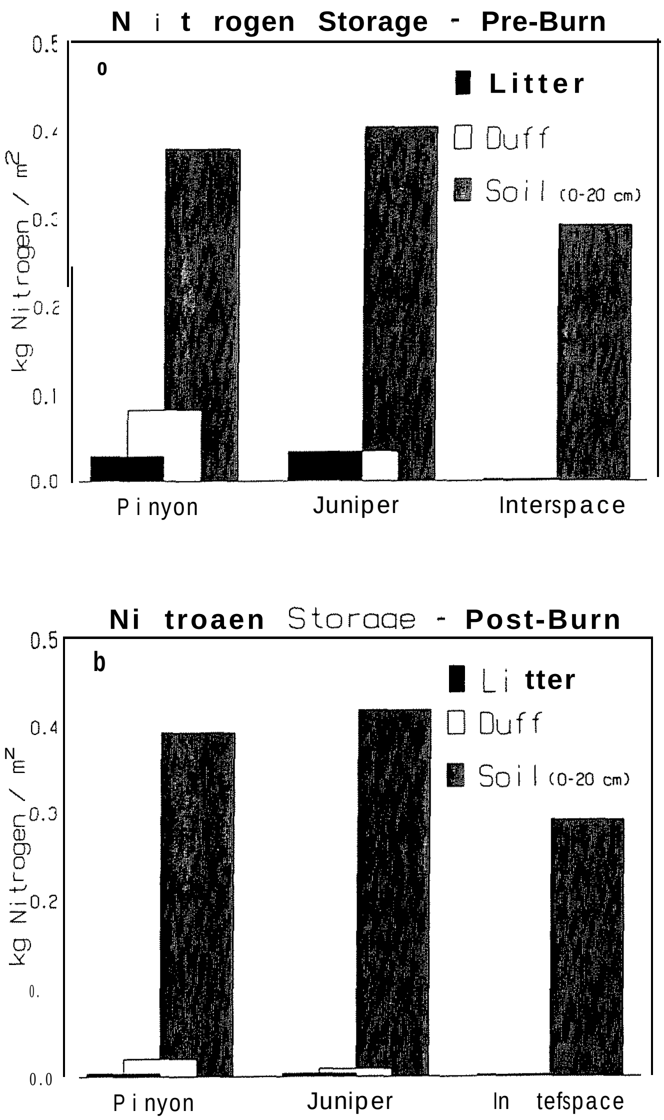

Figure 3.--Nitrogen storage pre- (a) and post- (b) bum in the woodland floor of a pinyon-juniper ecosystem in northern Arizona.

DISCUSSION

The effect of burning a pinyon-juniper woodland had surprisingly similar results to our microcosm experiment in which simulated bums conducted over both pinyon and juniper soils led to a large loss of C and N [as well as 50 percent of the P] (DeBano and Klopatek 1988; Klopatek and others 1990). Loss of N agreed with that reported by Raison and others (1985) in that nitrogenous compounds are usually lost proportionally to the amount of C oxidized.

The effect of burning on soil N has presented an interesting paradox that has been vigorously debated in the literature for years (e.g., Knight 1966; McKee 1982). As stated previously, some authors have reported insignificant total N increases, while others reported significant increases in soil N that were caused by the leaching of the above ash material. WC found significant losses of N from the woodland floor. When the loss of C and N are applied to the spatial distribution of pinyon-juniper-interspace mosaic, the losses total nearly 12.6 Mg ha" of C and 167 kg ha" N. Although large amounts of C and N were lost from the litter and duff in this study, the changes in total N of the combined soil depths were minimal. However, the soils at the surrounding field sites that had been previously (10-90 years) burned all had significantly less soil N than the unburned (Klopatek 1987). Our findings do not indicate why the loss of N occurs, although we hypothesize it may be due to the loss of C. This is a result of a combination of reduced C/N ratios of the ash (compared to the litter and duff) and the increase in decomposable substrates in dead root material. It should be mentioned that our reported organic C concentrations include the reduced charcoal that probably is not available for microbial utilization. The newly available root material will lead to a initial immobilization of soil N, but a release after decomposition has occurred. Our preliminary findings indicate an increase in CO, release from the forest floor of the bum site when compared to the nearby control site, indicating increased decomposition similar to that reported by Shoch and Binkley (1986).

Pinyon-juniper woodlands in Arizona are often situated between fire adapted ecosystems: chaparral and grasslands on the xeric end and ponderosa pine on the more mesic end. Clearly, pinyon-juniper is not a fire adapted system as many of its trees have low-lying branches that act as ladder fuels resulting in the trees “torching” or crown fires. This torching adds considerable heat to the system, effectively consumes all litter and duff, and may sterilize the upper soil layers. The result is an initial greater loss of nutrients and provides a positive feedback mechanism to accelerate subsequent losses.

Table 1.--Organic carbon, total nitrogen and carbon:nitrogen ratios of pinyon, juniper, and interspace Litter and duff material before and after burning

	Carbon g kg^{-1}		Nitrogen g kg^{-1}		C/N	
	Pre-burn	Post-burn	Pre-burn	Post-burn	Pre-burn	Post-burn
Piyon						
Litter (ash)	432a,w	41b,x[1]	5.2a,x	2.1b,x[1]	83.1a,x	19.5b,x
Duff (ash)	231a,x	41b,x	4.6a,x	2.1b,x	50.2a,y	19.5b,x
Soil O-10 cm	40a,y	43a,x	1.9b,y	2.1a,x	21.1a,z	20.5a,x
IO-20 cm	27a,z	28a,y	1.6a,y	1 .Sa,y	16.9a,z	18,7a,x
Juniper						
Litter (ash)	352a,x	47b,x	4.6a,x	1.8b,x	76.5a,y	26.lb,y
Duff (ash)	169a,y	47b,x	3.2a,x	1.8b,x	52.8a,y	26. lb,y
Soil O-10 cm	39a,z	40a,x	1.8b,y	2.3a,x	21.7a,z	17.4a,y
10-20 cm	33a,z	31a,y	1.7a,y	1.5a,x	19.4a,z	20.7a,y
Interspace						
titter (ash)	445a,x	34b,x	3.7a,x	0.9b,x	120.2a,x	37.2b,x
Soil O-10 cm	13a,y	14a,y	1 .Oa,y	I.Oa,x	13,4a,y	14.1a,y
10-20 cm	16a,y	16a,y	1.3a,y	1.3a,x	12.3a,y	13.3a,y

[1]Post-burn litter and duff ash values represent composites due to the difficulty in separating ash material for chemical analysis.

Common letters (a,b) following values indicate no significant differences ($p < 0.05$) between pre- and post-burn for the same element, while common letters (x,y,z) indicate no significant differences within columns for the same cover type.

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EFFECTS OF FELL-AND-BURN SITE PREPARATION ON WILDLIFE HABITAT AND SMALL MAMMALS IN THE UPPER SOUTHEASTERN PIEDMONT

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Abstract-The fell-and-bum site preparation technique is an effective means of regenerating low-quality hardwood stands in the Southern Appalachian Mountains to more productive pine-hardwood mixtures. This technique offers a number of advantages over conversion to pine monoculture. These include: lower cost, increased vegetation diversity within the stand, improved aesthetics, and continued mast production. However, the technique has not been fully tested in the Piedmont and other regions. This study reports the early successional effects of several variations of the fell-and-bum technique on small mammal communities and wildlife habitat in the Upper Southeastern Piedmont. Burning was increased forage production and species richness of vegetation. Winter felling of residual stems was more effective than spring felling in stimulating forage production and increasing species richness of vegetation.

INTRODUCTION

The fell-and-bum site preparation technique has proven successful as an inexpensive means to regenerate low-quality stands to more productive pine-hardwood mixtures in the Southern Appalachian Mountains (Phillips and Abercrombie 1987). Complete descriptions of the technique are given by Abercrombie and Sims (1986), Phillips and Abercrombie (1987), and Van Lear and Waldrop (1988). Briefly, the technique involves a commercial clearcut followed by a spring felling of residual stems (> 2 m in height) and a summer broadcast bum, after which pines are planted on a 3 m by 3 m (10 by 10 feet) or wider spacing. It is anticipated that the technique will produce results in the Upper Southeastern Piedmont similar to those observed in the mountains. However, differences in climate, soils, topography, and rainfall may make refinements to the technique necessary (Waldrop and others 1989). This method could become an attractive alternative to pine monoculture management for nonindustrial private forest landowners, who control approximately 80 percent of the commercial forested land in the Piedmont.

Benefits to wildlife have not been documented. However, it has been supposed that use of fell-and-bum methods would benefit certain game species, but their has been little consideration of effects on small mammals, insects, and herpetofauna in treated stands. For these reasons it is important to determine the effects of the technique on all components of the natural community before promoting its use in the Piedmont.

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METHODS

Study Area

Study areas were located in the Upper Piedmont Plateau region of western South Carolina, on the Clemson University Experimental Forest in Pickens and Oconee Counties. Soils were sandy loams of the Cecil and Pacolet series. Annual temperature and precipitation average 15.5” C and 148 cm, respectively. During 1989, mean annual temperature was 0.8” C below normal, and mean annual precipitation was 23 cm above normal (NOAA 1989).

Prior to harvest, stand ages ranged from 4.5 to 55 years. Site indexes for shortleaf pine (Pinus echinata Mill.) at base age 50 years averaged 18 m (range 15 to 20 m). Stands consisted primarily of low-quality hardwoods dominated by upland oaks (Quercus spp.) and small numbers of shortleaf pine, loblolly pine (P. taeda L.), and Virginia pine (P. virginia Mill.). Basal area averaged 8.6 m^2/ha. Aspects ranged from 180 to 230 degrees, and slope. averaged 13 .5 percent (range 10.0 to 20.0 percent).

Treatments

Each of three replications was divided into five 0.8 ha treatment areas. Each treatment area contained 5 to 7 sample plots, 0.1 ha in size. Treatments included clearcutting followed by winter-felling with and without summer burning; spring-felling with and without summer burning; and an unharvested control.

Habitat Analysis

Procedures for habitat analysis were modified from United States Fish and Wildlife Service Habitat Suitability Index (HSI) models (Mengak 1987, Mengak and others 1989, and Sanders 1985). Prior to the harvest, 10 to 20 0.04 ha vegetation plots were established along a transect within each treatment area. Plots were spaced at varying distances along the transect to best utilize the available area, avoid overlap, and maintain a southerly aspect. Permanent small-mammal trapping stations were also established at each plot center. Sampling was conducted during three sampling periods in 1989. Periods were chosen to evaluate the habitat at the lowest level of vegetation production (Jan. 1 to Mar. 31), the peak of production (May 1 to July 31), and the end of the growing season (Sept. 1 to Nov. 31). Ground cover estimates (by species) were obtained using a 35 mm ocular tube (James and Shugart 1970). Estimates were made at l-m intervals along two 10-m transects within each sample plot, Transects were centered on the trap station.

Aboveground forage biomass was determined by clipping the current year's growth of all plants in a 1 m by 1 m plot randomly located within each 0.04-ha plot to a height of 1.5 m; forage was weighed in the field. Clipped material was separated into three categories: woody, forbs, and grasses. Moisture content of forage biomass was determined in the laboratory after drying in a forced-air oven at 60 degrees C° for 72 hours.

Trapping

Trapping took place during the three periods when vegetation was sampled. Traplines were prebaited with peanut butter for 5 nights with the traps closed and then sampling was conducted for 5 consecutive nights during each trapping period. Four trap types were used in each treatment area: Victor rat traps, Victor mouse traps, Museum special traps, and pitfalls with drift fences. All traps were rebaited each day during the prebaiting and trapping periods. Trapping design was identical for all periods.

One snap trap of each type was placed within 2 m of each randomly located trap station. Trapping stations were marked with l-m sections of rebar in order to establish permanent trap locations. Pitfalls were randomly located on each site by overlaying a grid on the site map and using a random number table to determine their coordinates. A modification of the trap design described by Williams and Braun (1983) was used. Each drift fence consisted of three 5 m by 51 cm legs of aluminum flashing that met at a common point centered on the pitfall, with 120 degrees between each pair of legs. Flashing was set in a ditch 8 to 10 cm deep. These ditches were then packed with soil and the fences supported with wooden stakes. At the center of the fences, a 19-1 plastic bucket was buried flush with the ground. Buckets were kept one-third to one-half full of water to drown captured animals, and were covered with a lid when not in use. All traps were checked daily during trapping periods.

Vegetation and trapping data were used to calculate Shannon diversity (H ′), evenness (J), and species richness (S) for each trapping period. Shannon diversity was calculated as $H' = -\Sigma P_i (\ln P_i)$ where (P_i) is the proportion of the ith species in the population (Shannon and Weaver 1949). This function measures the uncertainty in predicting the identity of any randomly selected individual based on the total number of species in the sample (S) and the number of individuals (N), or the proportion of that species to the whole sample (P_i) for each species represented in the sample. (J) is a measure of the evenness of the distribution of individuals within the species present, and is calculated as $J = H'/(\ln S)$ (Pielou, 1977).

Insects were collected in ten randomly located 600-ml pitfalls on each site. Traps were used for biomass collection, since terrestrial insects are more susceptible to capture in pitfalls, their numbers would be overestimated if individuals were singled out for identification (Southwood 1978). Traps were kept one-third to one-half full of equal parts of water and ethylene glycol, to keep the insects flexible. Traps were emptied daily during the 5-day period when small mammal trapping took place. All insects were identified to family and weighed for biomass.

All habitat and trapping data were summarized for each site and treatment type. Analysis of variance was used to test for differences between treatments, blocks, and collection periods. Differences were tested for significance at the 0.05 level.

RESULTS AND DISCUSSION

Vegetation

Biomass. Total forage biomass was greatest on winter-felled sites and on burned sites (Fig. 1). All treatments that included felling produced more forage biomass than the unharvested control plots. Total woody biomass, which was highly variable, did not differ significantly among treatments. Woody biomass production varied within the treatments depending on the species of woody vegetation present on the site. This variation within treatments masked any between-treatment differences that might have been developing. Forb production was greatest on burned sites, particularly with winter felling. This forb response resulted from removal of the litter layer which improved seed germination conditions. Grass production was significantly higher on the winter-felled no-bum sites than all other treatments as a result of sprouting from pre-existing rootstocks beneath the little layer. The controls had significantly lower grass production than all other treatments as a result of the heavy litter layer and almost complete shading of the forest floor. Increases in grass and forb coverage at the expense of woody vegetation are common after burning and have been documented by (Langdon 1981, Waldrop and others 1987, Van Lear and Waldrop 1989).

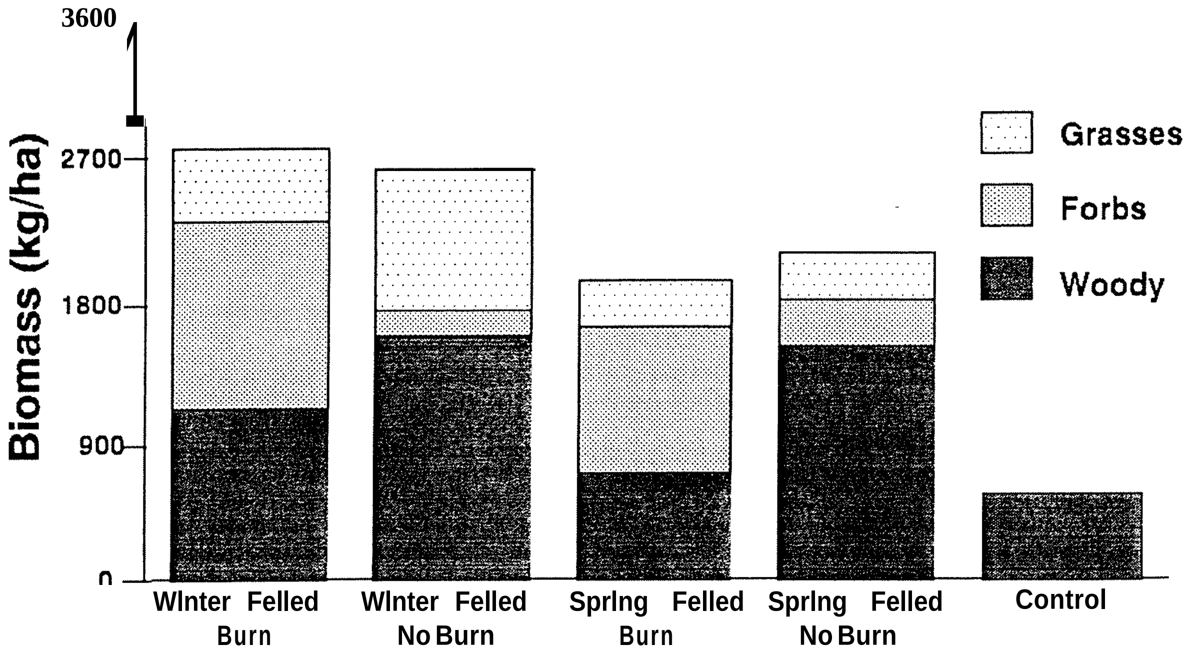

Figure 1. Forage biomass production by species group and treatment on fell-and-bum site-prepared areas in the Upper Southeastern Piedmont.

During the first sampling period after the bum, considerable browsing occurred on blackgum (Nyssa sylvatica Marsh.), American holly (Ilex opaca Ait.), sassafras (Sassafras albidum (Nutt.) Ness.), and smilax (Smilax spp.) seedlings and sprouts. During the second sampling period, utilization of the same species continued. Pokeweed (Phytolacca americana L.) however emerged as the most heavily browsed species. Pokeweed was often browsed to the point of being stunted. During the final sampling period, utilization of woody browse seemed to decline, probably due to the lignification of the woody tissue. While utilization of pokeweed and smilax remained steady. Because summer rainfall was higher than normal forbs on these sites remained succulent into October and November, when they would normally have hardened and been abandoned as preferred browse species.

Diversity, Richness, and **Evenness.** Diversity of vegetation (H ') increased slightly as a result of winter felling, but there were no significant differences in H ' between treatments. Species differences did occur among treatments as a result of burning, but these differences did not significantly affect H '. Burning favored grasses and forbs while unburned areas were dominated by sprouts of trees and shrubs.

Species richness of vegetation (S) was significantly higher on burned areas than in unburned areas and unharvested controls (fig. 2). Among the burning treatments, winter felling produced significantly greater species richness values'. Due to the absence of leaves on the slash, winter-felled sites typically did not bum as evenly or completely as spring-felled sites (Geisinger and others 1989). Therefore a mosaic of burned and unburned microsites was created, with each one capable of supporting a different complement of species.

Vegetative evenness (J) did not differ significantly among treatments. However, J was slightly higher on winter-felled no-bum sites due to increased grass production on those sites.

Small Mammals

Diversity, Richness, and Evenness. Diversity of small mammals (H') showed no significant differences among treatments until the third sampling period (Sept. 1 to Nov. 31) (table 1). At that time, small mammal abundance on all site-prepared areas declined with the winter decline of vegetative browse and ground cover. This pattern agrees with the finding of Briese and Smith (1974) that small mammals shift the centers of their ranges throughout the year to take advantage of the distributional change in food and cover. H ' was greater on the unharvested controls than on the treated sites during the third period as a result of the fall mast crop and the greater cover afforded by the undisturbed little layer.

Species richness (S) values were low in the first sampling period and there were no significant differences in S among treatments (table 2). In the second sampling period, winter-felled, burned sites had significantly higher S values than other sites. However, in the third sampling period, species richness was significantly lower on winter-felled burned sites as a result of the early senescence of the forb species that dominated those sites. Both food and cover declined much earlier on winter-felled burned sites than on those where grasses or woody vegetation were more dominant. Evenness (J) values did not differ significantly among treatments in any period.

'See Evans (1990, unpublished thesis) for a full species listing.

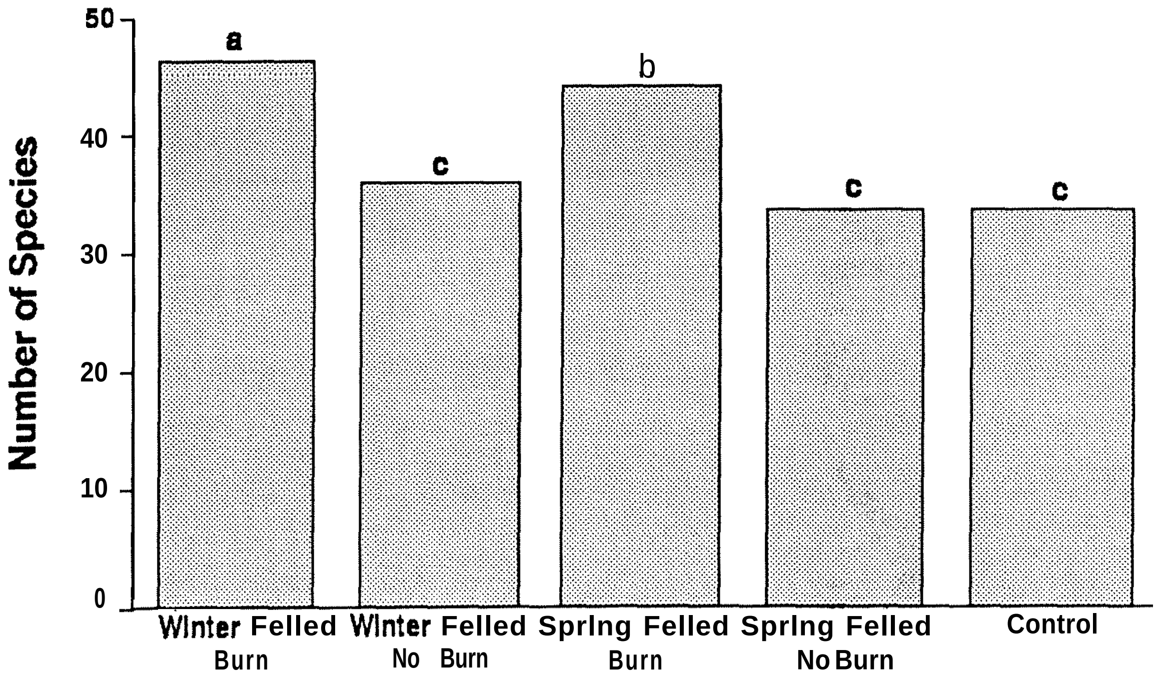

Figure 2. Plant species richness on fell-and-bum site-prepared areas in the Upper Southeastern Piedmont (columns with thc same letter were not significantly different at the 0.05 level using Duncan's Multiple Range Test).

Table 1. Small-mammal diversity (H´) on felled-and-burned study sites in the Upper Southeastern Piedmont, 1989.

Treatment	Period		
	Jan. 1-Mar. 31	May I-July 31	Sept. 1-Nov. 31
Uinter-Fell, Burn	0.0a	0.261a	0.092a
Winter-Fell, No-burn	0.1a	0.173a	0.360ab
Spring-Fell, Burn	0.0a	0.235a	0.409ab
Spring-Fell, No burn	0.0a	0.191a	0.192a
Control	0.0a	0.000a	0.519b

-Values folloued by the same letter uithin a column were not significantly different at the 0.05 level using Duncan's Multiple Range Test.

Table 2. Species richness (S) of small mammals on felled-and-burned study sites in the Upper Southeastern Piedmont 1989.

Treatment	Period		
	Jan. 1-Mar. 31	May 1-July 31	Sept. 1-Nov. 31
Uinter-Fell, Burn	0.333a	2.667a	1.333a
Winter-Fell, No-burn	1.333a	1.667ab	2.667ab
Spring-Fell, Burn	0.667a	2.000ab	3.333b
Spring-Fell, No burn	1.000a	2.000ab	2.000ab
Control	0.667a	0.333b	3.333b

-Values followed by the same letter uithin a column uere not significantly different at the 0.05 level using Duncan's Multiple Range Test.

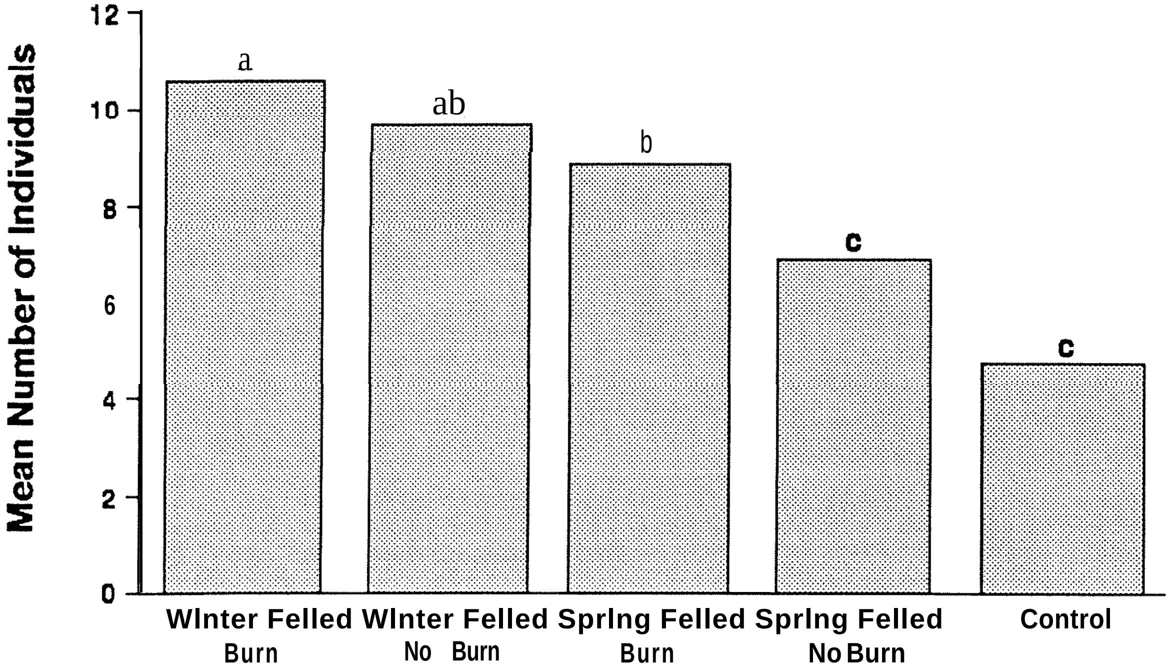

Figure 3. Number of individual small mammals on fell-and-bum site-prepared areas in the Upper Southeastern Piedmont (columns with the same letter were **not** significantly different at the 0.05 level).

Number of individuals. Both winter-felling and burning resulted in higher numbers of small mammals (N) utilizing an area (fig. 3). This increase is probably a response to the increase in available forage (vegetation biomass) that resulted from this treatment combination. Not all small mammal species increased in numbers in response to disturbance. Species that were trapped most often, such as white-footed mice (Peromvscus leucopus), were those that are best adapted to an early successional environment (table 3). This finding agrees with a number of other studies that show an increase in Peromvscus spp. following fire (Ahlgren 1966; Krefting and Ahlgren 1974; Hingtgen and Clark 1984). The increase in Peromvscus spp. was most pronounced during the first two sampling periods and was no longer evident by the third sampling period. By that time the habitat was sufficiently developed to support a larger number of species with more varied food habits and cover requirements.

Table 3. Relative abundance and total number of individual animals (N) captured, by species, on all fell-and-burn study sites in the Upper southeastern Piedmont 1989.

SPECIES		N	ABUNDANCE (PCT)
MAMMAL			
white-footed mouse	(Peromyscus leucopus)	97	63.8
golden mouse	(Ochrotomys nuttallii)	13	8.5
hwse mouse	(Mus musculus)	10	6.6
eastern cottontail rabbit	(Sylvilagus floridanus)	7	4.6
cotton rat	(Sigmodon hispidus)	6	3.9
cotton mouse	(P. gossypinus)	2	1.2
eastern chipmunk	(Tamias striatus)	1	0.7
least shrew	(Cryptotis parva)	1	0.7
southeastern shrew	(Sorex longirostris)	1	0.7
black rat	(Rattus rattus)		0.7
	TOTAL=		
BIRDS			
mourning dove,	(Zenaida macroura)	1	0.7
	TOTAL=	1	0.7
HERPETOFAUNA			
American toed	(Bufo americanus)	3	2.0
Woodhouse's toad	(B. woodhousei)	2	1.2
eastern box turtle,	(Terapene carolina)	2	1.2
southern leopard frog	(Rana sphencocephala)	1	0.7
eastern narrow-mouthed toad	(Gastrophyrne carolinensis)	1	0.7
ring-necked snake	(Diadophus punctatus)	1	0.7
	TOTAL=		
OTHER			
brown grand-daddy longlegs,	(Phalangium opilio)	1	0.7
Carolina Locust,	(Dissostiera carolina)	1	0.7
	TOTAL=	2	1.4
	GRAND TOTAL=	152	100.0

1-indicates an incidental capture in a snap trap

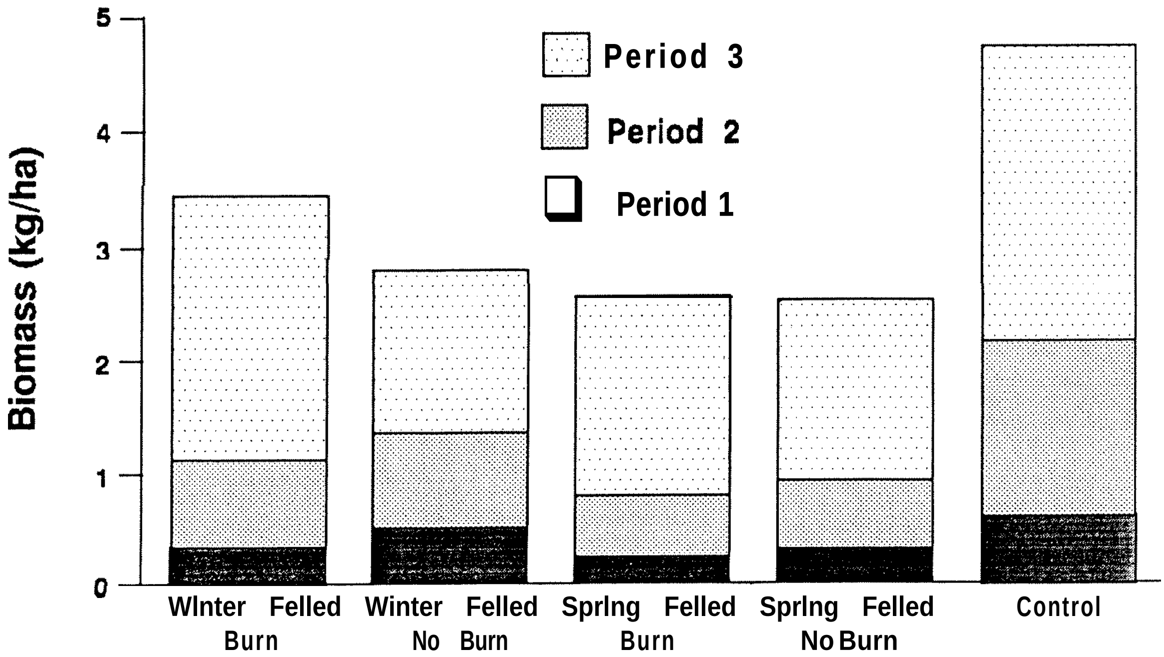

Figure 4. Insect biomass production on fell-and-bum site-prepared areas in the Upper Southeastern Piedmont, 1989.

Insects

Biomass. Total insect biomass was decreased temporarily by all site preparation treatments (fig. 4). Control sites averaged 4.7 kg/ha, while treated sites averaged 2.8 kg/ha. Insect biomass did not decrease as dramatically on winter-felled sites as on other sites, probably because fire intensities were lower on the winter-felled sites. Insect biomass was significantly higher on control sites than on other sites during the first sampling period. Both winter-felled and control sites were significantly higher than other sites during the second sampling period. Recovery of insect biomass production was rapid and there were no longer significant differences by the third sampling period.

SUMMARY AND CONCLUSIONS

Vegetation biomass production was greater for all site preparation treatments than for the control. Burned plots supported richer, more productive plant communities and higher numbers of small mammals than did unburned plots. Winter-felling and burning yielded richer, more productive plant communities and higher numbers of small mammals than spring felling and burning. As vegetation biomass production declined in the fall, small-mammal numbers became highly variable within treatments. Insect biomass production was reduced by all site preparation treatments due to disturbance of the litter layer. However, this decrease in production lasted less than 1 year.

This study indicates that the fell-and-bum site preparation technique, as it is practiced in the Southern Appalachian Mountains, can be used in the Upper Piedmont without adversely affecting forage production for wildlife habitat. If felling of residual stems is conducted in the spring, as is recommended in the Southern Appalachian Mountains, site preparation bums can significantly reduce fuel loads and provide uniform Planting conditions (Sanders and Van Lear 1987; Geisinger and others 1989). Bums conducted after winter felling are less uniform (Geisinger and others 1989) and leave more of the slash and logs that provide cover and foraging sites for small mammals. More complete bums also result in a more homogenous habitat than the mosaic of burned and unburned microsites found on winter-felled areas.

The fell-and-bum technique is a relatively inexpensive method to regenerate pine-hardwood mixtures but its application in the Piedmont requires additional study. Effects on wildlife, water quality, and soil as well as on stand regeneration and development are currently being studied. As Van Lear and Kapeluck (1989) have shown, burning prescriptions on Piedmont sites must be modified if erosion is to be controlled. Species composition and soil characteristics of Piedmont sites are different from those of mountain sites, and it may be necessary to modify fell-and-bum techniques because of those differences. Finally, this study addressed only the early successional habitat changes that resulted from this technique. The impact of this type of site preparation on wildlife as the stands continue to develop is yet to be determined.

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EFFECTS OF FIRE AND TIMBER HARVEST ON VEGETATION AND CERVID USE ON OAK-PINE SITES IN OKLAHOMA OUACHITA MOUNTAINS

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Abstract-This study compared vegetation response and cervid browse use within the Ouachita Mountains following an array of timber harvest and fire regimes. Nine treatments were replicated l-3 times in a completely randomized design on 23 (1.2-I .6 ha) units. The treatments were a no treatment control, a winter rough reduction bum treatment, a clearcut, mechanical site preparation and summer bum treatment, a harvest pine (Pinus echinata) only and annual bum treatment, and 5 harvest pine and thin hardwood treatments with no bum, 4, 3, 2, and 1 year winter-bum intervals. Pine timber was harvested in June and July of 1984 and hardwoods selectively thinned to a basal area of 9 m^2/ha in appropriate treatments. Clearcut treatments were sheared, raked, and windrowed in spring 1985 followed by a summer site preparation bum, and then contour ripped prior to planting in early April 1986. Winter bums were strip headfires applied in mid- to late winter in 1985 through 1988. Little bluestem (Schizachvrium scoparium) and big bluestem (Andropogon gerardi) dominated harvested and winter burned (retrogressed) treatments. Plant frequency and percent ground cover of these 2 species increased on sites burned more frequently. The clearcut and summer burned sites were initially dominated by forbs and panicums (Dicanthelium spp. and Panicum spp.). Then as forbs declined, little bluestem increased in frequency and percent ground cover. Plant species richness was significantly ($P < 0.05$) increased by timber harvest and fire. Among harvested sites, frequency of burning had no significant effect on plant species diversity or plant species evenness. Longer bum intervals or no burning on retrogressed sites allowed woody browse species used by white-tailed deer (Odocoileus virginianus) and possibly elk (Cervus elaphus) to increase. c y o f browse use by cervids in 1988 was greatest on the harvest, thin, 3-year bum interval; harvest, thin, no-bum; harvest, thin, 2-year bum interval; and clearcut treatments. The harvest, thin, no-bum and clearcut treatments also provided screening and bedding cover for cervids, in contrast to other treatments. Winter prescribed fire at 1- or 'L-year intervals favored legumes and created habitat conditions favorable for bobwhite quail (Colinus virginianus). Timber management strategies that create a mosaic of retrogressed burned and unburned sites, and regeneration clearcuts with adequate provisions for hard mast production should provide management flexibility to meet habitat needs of most game species.

INTRODUCTION

The oak (Ouercus spp.)-shortleaf pine forest is the most extensive forest type in the eastern United States (Lotan and others 1978), and is widely considered to be a fire subclimax association (Oosting 1956). In spite of the type's prevalence and importance, there has been insufficient research on forest management and fire ecology in the oak-shortleaf pine type and specifically in the Ouachita Highlands (Lotan and others 1978).

Segelquist and Pennington (1968) documented the lack of an adequate understory forage base for deer in the Ouachita Mountains of Oklahoma. Winter mortality of deer has been related to mast failure and may be compounded by the lack of an evergreen winter browse (Segelquist and Pennington 1968; Segelquist and others 1969, 1972). Forage production in late summer and early fall may be of critical importance in the advent of mast shortfall (Fenwood and others 1984).

In 1977, the Oklahoma Department of Wildlife Conservation began using timber harvest and prescribed fire to improve habitat conditions for deer and elk on the Pushmataha Wildlife Management Area. Harvested settings were maintained in early stages of secondary succession with prescribed fire (site retrogression). Although the effects of forest management and fire on wildlife have often been studied, little research has dealt with manipulation of forested ecosystems for the purpose of benefiting wildlife (Ripley 1980).

My objective was to compare site retrogression through timber harvest and periodic prescribed fire, with regeneration clearcutting and understory rough reduction bums. Changes in plant species richness, diversity, evenness, composition, percent ground cover and browse utilization by cervids were used as measures of treatment effects.

STUDY AREA

The 29.1-ha study area was located within the Forest Habitat Research Area (FHRA) on the 7,395 ha Pushmataha Wildlife Management Area near Clayton, Oklahoma. The Pushmataha Wildlife Management Area lies along the western edge of the Ouachita Highland Province. Study area soils belong to the Camasaw-Pirum-Clebit association with areas of rock

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outcrop. The soils developed from cherty shales and resistant sandstones and were thin and drought prone. The climate was semihumid to humid with hot summers and mild winters (Bain and Watterson 1979).

Prior to acquisition from 1946 to 1954, Pushmataha Wildlife Management Area was grazed, selectively harvested, and frequently burned (Oklahoma Department of Wildlife Conservation 1972). The FHRA was protected from further logging, grazing and fire from acquisition until 1984 (R. Robinson, Area Biologist, Oklahoma Department of Wildlife Conservation, personal communication). From 1983 to 1988 the Pushmataha Wildlife Management Area supported an average deer population of 540 $\pm$ 40 (SE) and 9 $\pm$ 1 (SE) elk (Masters 1991).

Post oak (Q. stellata), shortleaf pine, and to a lesser extent, blackjack oak (Q. marilandica), and mockemut hickory (Carya tomentosa) dominated the overstory. Common woody understory species included tree sparkleberry (Vaccinium arboreum), Virginia creeper (Parthenocissus quinouefolia), greenbriar (Smilax spp.), and grape (Vitis spp.). o n herbaceous plants included little bluestem, panicums, and sedges (Carex spp.) (Masters 1991).

METHODS

Cultural Treatments

Beginning in summer 1984, 9 treatments were applied to 23, 1.2-1.6 ha contiguous, rectangular units on the FHRA in a completely randomized experimental design (Chambers and Brown 1983). Treatments, burning sequence, treatment code, and number of replications (n) were:

(1) no treatment (control) (n = 3);
(2) rough-reduction winter prescribed bum at 4-year interval, 1985 (RRB) (n=3);
(3) clearcut and summer site prep bum, 1985 (CCSP) (n=3);
(4) harvest pine timber only, winter prescribed bum, l-year interval, 1985 to 1988 (H-NT-l) (n=3);
(5) harvest pine timber, thin hardwoods, no bum (H-T) (n=1);
(6) harvest pine timber, thin hardwoods, winter prescribed bum at 4-year interval, 1985 (H-T-4) (n=3);
(7) harvest pine timber, thin hardwoods, winter prescribed bum at 3-year interval, 1985, 1988 (H-T-3) (n=2);
(8) harvest pine timber, thin hardwoods, winter prescribed bum at Z-year interval, 1985, 1987 (H-T-Z) (n=3); and
(9) harvest pine timber, thin hardwoods, winter prescribed bum at l-year interval, 1985 to 1988 (H-T-l) (n=2).

In appropriate treatments merchantable pine timber was harvested and hardwoods selectively thinned by single stem injection using 2-4 D, to a basal area of 9 m^2/ha (includes stems > 5 cm diameter at 1.4 m height), in summer 1984. Prescribed bums using strip headfires were conducted in winter 1985 and in succeeding years at appropriate intervals. Several replications were dropped because they were either burned out of sequence by spotovers or not burned because of rain. The clearcut site-prep treatment included shearing, raking and windrowing of logging debris with a site-prep bum conducted during summer 1985. After contour ripping, genetically improved loblolly pine (P. taeda) was planted on a 2.1 m by 2.4 m spacing, in early April, 1986.

Vegetation Sampling

Understory, midstory and overstory vegetation was sampled using nested quadrats (1 m by 1 m and 4 m by 4 m) (Oosting 1956:47-50, 62). On each treatment unit, 10 permanent plots were established at 19.8 m intervals on two randomly located lines perpendicular to the contour. In order to avoid bias caused by influences from adjacent treatment units, I did not sample within 19.8 m of any edge (Oosting 1956; Mueller-Dombois and Ellenberg 1974). Data collected included plant species density, frequency, percent ground cover, and utilization.

Utilization was categorized based on proportion of current annual growth (CAG) browsed. The categories were none, trace--less than 25 percent, moderate--25 to 50 percent, and heavy--greater than 50 percent utilization.

Overstory and midstory vegetation were categorized by vertical strata and crown position relative to stand canopy structure. Strata designations were 0 to 1 m, 1 to 3 m, and greater than 3 m. Strata greater than 3 m were categorized by position relative to stand canopy structure and were suppressed, intermediate, codominant, and dominant canopy position (Smith 1962). On harvested treatments strata designation of residual trees was based on prior stand structure. No tree or shrub regrowth was greater than 3 m. Overstory vegetation was further quantified using the variable radius plot method (Avery 1964). Basal areas were taken using a 10 basal area factor prism with plot center at the center of each 4 m by 4 m plot. Vegetation sampling and browse use determinations were conducted in September and October of each year because this was a critical period of the year for deer (Fenwood and others 1984). A baseline survey was conducted in 1983.

Data Analysis

Species diversity, evenness, richness, density, and frequency were calculated from vegetation samples (Ludwig and Reynolds 1988). A modification of Krueger's (1972) preference index (RPI) combined across years and treatments was used to rank plant species used by deer and elk. Analysis

Table 1.--Average percent cover for major species groups 1983-88. Timber harvest was applied in summer 1984 and prescribed burns were conducted in 1985 and following.[1]

VEGETATIVE GROUP	TREATMENT[2] CONT	RRB	H-NT-1	H-T	H-T-4	H-1-3	H-1-2	H-1-1	CCSP
YEAR=83									
GRASSES	3	6			1				7
FORBS	2	2		7	11	t8	4	.	3
LEGUMES	1	1		1		1	2	.	1
VINES	1	<1		<1	<1	11	<1		<1
SHRUB 0-1M	6	13		6				.	6
SHRUB 1-3M	1	3		0	1		<1	.	2
TREE MID[3]	14a	10a		4b	2b	1b	4b	.	4b
YEAR=84									
GRASSES	9	10		14	14	8	5	.	7
FORBS	5	6		7	1	1	2		2
LEGUMES	2	2		<1	1	1	2	:	2
			.	<1	<1	1	1		<1
SHRUB 0-1M	10	22		16	11	8	9	:	7
SHRUB 1-3M	3	5		1	2	<1	4		1
TREE MID	23a	24a	.	9ab	1b	7b	3b	:	9ab
YEAR=85									
GRASSES	7b	6b	.	23a	22a	20a	14ab	.	5b
FORBS	2	2	.	25a	17b	11b	14b	.	3
LEGUMES	1	1	.	2	4	1	1	.	1
VINES	7	<1	.	0	<1	17	19		<1
SHRUB 0-1M		9	.	20	13			:	4
SHRUB 1-3M	5a	3ab	.	1b	<1b	0b	0b		<1b
TREE MID	24a	17a	.	1b	2b	0b	2b	:	<1b
YEAR=86									
GRASSES	7b	8b	30a	20ab	29a	27a	20ab	28a	17ab
FORBS	2c	5bc	13bc	16bc	18b	17b	19b	18b	37a
LEGUMES	2	2	4	4	5	6	7	9	3
VINES		<1	<1	<1	1		1	1	<1
SHRUB 0-1M	9	19	10	27	20	12	19	14	9
SHRUB 1-3M	6a	1bc	1bc	3bc	2bc	2bc	4ab	<1bc	oc
TREE MID	22a	10b	3bc	4bc	2c	oc	1c	oc	oc
YEAR=87									
GRASSES	4c	7c	34a	19b	2Sab	21b	21b	3Sa	25ab
FORBS	1	2	11abc	10bc	9bc	7c	17ab	13abc	20a
LEGUMES	1b	1b	6ab	8a	4ab	3ab	9a	Sab	4ab
VINES	1	<1	<1	<1	1	3a	1	1	<1
SHRUB 0-1M	6c	14abc	10bc	29a	21abc	24ab	22abc	8bc	13abc
SHRUB 1-3M	4	2	2	12	6	5	3	2	3
TREE MID	22a	10b	4c	2c	<1c	2c	1c	oc	oc
YEAR=88									
GRASSES	3d	6d	31ab	11cd	25ab	32ab	20cb	37a	28ab
FORES	1b	2b	7ab	4b	5b	7ab	8ab	6ab	13a
LEGUMES	1c	1c	5abc	2bc	3abc	5abc	9a	8ab	5abc
VINES	1	<1	<1	<1	1	1	<1	1	<1
SHRUB 0-1M	7c	15abc	10bc	29a	26ab	28a	23ab	10bc	17abc
SHRUB 1-3M	4b	2b	2b	24a	11b	2b	9b	1b	12b
TREE MID	16a	9bc	2c	12ab	1c	oc	<1c	oc	oc

[1] Row means with the same Letter are not significantly different ($P < 0.05$).

[2] CONT = control, no treatment; RR8 = rough reduction burn in winter, at 4 year intervals; H-NT-1 = harvest pine timber, no thinning of hardwoods, winter prescribed burn at 1 year intervals; H-T = harvest pine timber, thin hardwoods; H-T-4 = harvest pine timber, thin hardwoods, winter prescribed burn at 4 year intervals; H-T-3 = harvest pine timber, thin hardwoods, winter prescribed burn at 3 year intervals; H-T-2 = harvest pine timber, thin hardwoods, winter prescribed burn at 2 year intervals; H-T-1 = harvest pine timber, thin hardwoods, winter prescribed burn at 1 year intervals; CCSP = clearcut, windrow logging slash, summer site prep burn, rip.

[3] TREE MID = Suppressed trees > 3 m height in the midstory, but not extending into the upper canopy layer.

was performed using PC-SAS (SAS Institute 1985, 1987) and SPDIVERS.BAS (Ludwig and Reynolds 1988). Statistical analysis of treatments was by one-way analysis of variance (ANOVA) for unequal sample size. To determine treatment preference by cervids, browse utilization frequency for all sample plots was summed by unit (replication), ranked and analyzed by ANOVA, the equivalent of the Kruskal-Wallis nonparametric procedure (SAS Institute 1985). Mean ranks were separated by Duncan's Multiple Range Test (Steele and Torrie 1980).

RESULTS

Vegetation Response

Pretreatment vegetation sampling in 1983 indicated higher ($\underline{P}$ < 0.05) percent cover of midstory trees in control and RRB replicates than on other treatment units (table 1). Values for utilization and other descriptors of vegetation did not differ among units prior to application of treatments. The only significant differences found between control and RRB treatments in succeeding years were in percent cover of suppressed trees. Rough reduction burning reduced ($\underline{P}$ < 0.05) percent cover in that stratum by 1986-88 (table 1).

Understory response varied after initial timber harvest and thinning of residual hardwoods. In 1984, species diversity increased, and evenness declined ($\underline{P}$ < 0.05) on all harvested and thinned treatments compared to the control and RRB treatments. Species evenness provided an adequate measure of shrub response only in 1984. Species richness of herbs and shrubs immediately after timber harvest was unchanged (figs. 1 and 2).

In 1985 and 1986 after all burning and timber harvest treatments had been applied species richness of herbaceous and shrub vegetation on treated areas were significantly ($\underline{P}$ < 0.05) higher (figs. 1 and 2). Grass cover on treated areas was dominated by little bluestem, big bluestem and panicums. The predominant forbs were horseweed (Conyza canadensis), white snakeroot (Eupatorium rugosum), and fireweed (Erechtities hieracifolia). Shrub response on harvested and burned treatments was composed of primarily winged sumac (Rhus cooallina), dewberry (Rubus spp.), and post oak sprouts. Only legume and vine categories showed no difference in cover among treatments (table 1).

In 1986, values of most vegetational characteristics of CCSP areas did not differ significantly from comesponding values for areas that were harvested and burned (table 1). However species composition and shrub species richness differed by treatment ($\underline{P}$ < 0.05) (fig. 2). Panicums and little bluestem were respective grass dominants on CCSP and all the harvested, thinned and burned treatments. Crabgrass

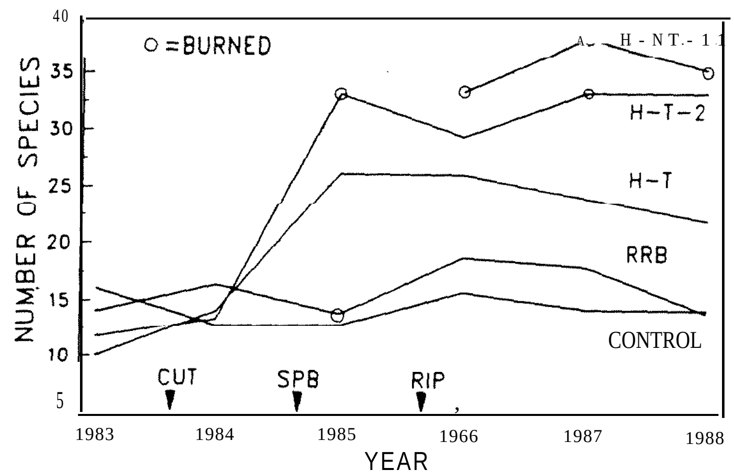

Figure 1.-Mean species richness of herbaceous plants 1983-88. For clarity of presentation some burned treatments were not depicted. Those not depicted were intermediate in response.

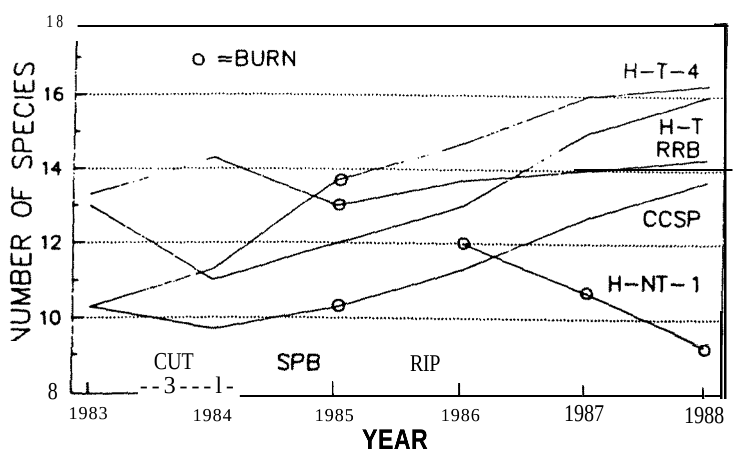

Figure 2.-Mean species richness of shrubs 1983-88. For clarity of presentation some burned treatments were not depicted. Those not depicted were intermediate in response.

(Digitaria violescens) was a significant component of the grass response of the CCSP treatment and occurred infrequently on other treatments. Broomsedge bluestem (Andropogon virginianus) also occurred more frequently on CCSP treatments than on other treatments. Forb response was greatest during 1986 on the CCSP treated areas and was significantly higher for this treatment than for others ($P <$ 0.05) (table 1).

By 1988, percent cover of all plant groups, except vines, differed among treatments (table 1). For herbaceous plants, species richness and evenness differed significantly among treatments. Species richness and diversity differed significantly among treatments for shrubs (fig. 2). Timber harvest and prescribed fire decreased herbaceous species evenness but increased shrub and herb richness and shrub diversity (figs. 1 and 2). Bluestems and panicums were dominant grasses on harvested and burned treatments. In the CCSP treatment areas, the grasses were mainly comprised of panicums and to a lesser extent little bluestem. Broomsedge bluestem occurred more frequently on this treatment than others.

Dominant shrub species on harvested and burned sites included winged sumac, dewberry, post oak sprouts, tree sparkleberry, and winged elm (Ulmus alata). e H - T treatment, sumac was not prevalent, but the above species and shortleaf pine seedlings and saplings were prominent. Dewberry, post oak, coralberry (Symphoricarpos orbiculatus) and loblolly pine were primary shrub constituents on CCSP treatments.

Abundance of preferred forbs increased ($P <$ 0.05) (fig. 3) after timber harvest and prescribed fire then declined as grass cover increased (table 1). Preferred browse increased ($P <$ 0.05) in all except RRB, control and annual burned treatments (fig. 4). Percent cover of preferred browse in the annual burned treatments were not different from percent cover of preferred browse on the control or RRB sites. By 1988 legumes, preferred forbs and preferred browse responded differentially by treatment (figs. 3-5). More frequent burning intervals favored legumes and preferred forbs while less frequent intervals favored shrubs (figs. 4 and 5).

Utilization

Cervids utilized 74 species of plants and 17 plant groups identified to genera. Forbs of 31 species, and additional plants identified only as members of 7 genera were used. Thirteen species of legumes and additional legumes identified to 1 genus (Desmodium spp.) were used. Utilization occurred on 29 species and an additional 7 genera of woody browse. Grass-likes utilized included panicums, sedges, and little bluestem. Rankings of relative preference revealed that woody browse was used more than forbs (table 2).

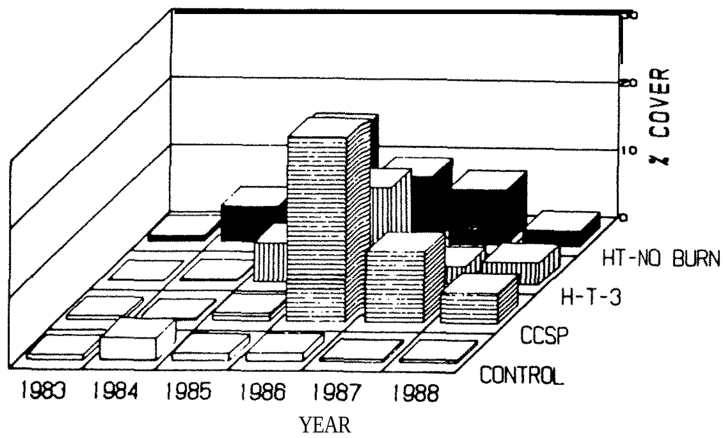

Figure 3.-Percent cover of preferred forbs 1983-88. For clarity of presentation some burned treatments were not depicted. Those not depicted were intermediate in response.

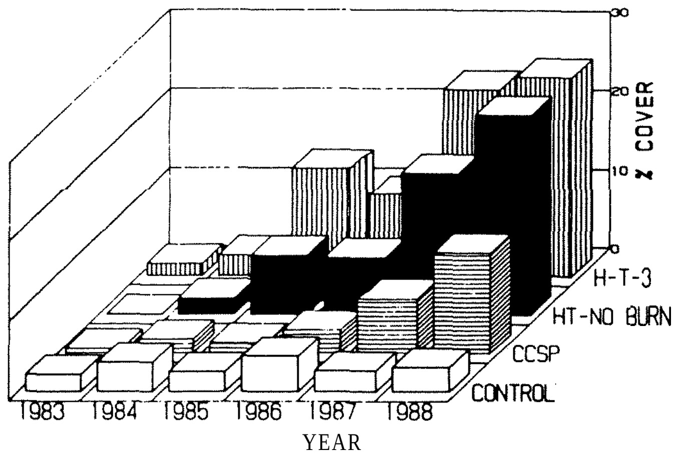

Figure 4.--Percent cover of preferred browse 1983-88. For clarity of presentation some burned treatments were not depicted. Those not depicted were intermediate in response.

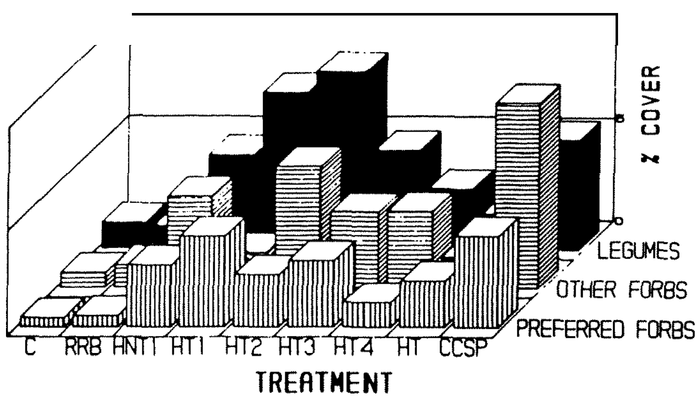

Figure S.--Percent cover of preferred forbs, other forbs, and legumes after all bum intervals had been completed in 1988.

Table Z.--Rankings of preferred cervid food plants based on a summed preference index for all years and treatments.

Browse	Index	Forbs	Index
Smilax spp.	808	Lespedeza spp.	322
Ulmus alata	745	Aster patens	311
Amelanchier arborea	448	Solidago ulmifolia	201
Vitis spp.	422	Honarda fistulosa	135
Vaccinium spp.	201	Phytolacca americana	128
Hypericum spp.	195	Conyza canadensis	120
Rhus glabra	167	Solanum carolinense	116
Rhus copallina	159	Aster spp.	106
Nyssa sylvatica	103		
Rubus spp.	103		

Cervid Use

Mean ranks of cervid frequency of utilization on replicates was significantly different ($\underline{P} < 0.001$) among treatments (table 3). Annual bum and RRB treatments had significantly lower frequency of utilization than other treatments.

DISCUSSION

Vegetation response varied by species. Some plant species increased *on retrogressed* sites and others decreased from pretreatment levels. Species evenness was highest on the control and pretreatment, which indicated that herbaceous species were equally abundant. Site perturbation, caused some plant species, particularly tallgrasses, to become more abundant relative to other species.

Community progression on harvested and burned sites was similar to that reported by Hebb (1971) for clearcutting. Successional stages after harvest were; (1) disturbed site with pretreatment understory ground cover; (2) profusion of grasses and annual forbs; (3) increase in perennial forbs and grasses, decrease in annuals and increase in shrubs; and (4) in the absence of periodic prescribed fire, increases in shrubs and grasses and declines in forbs.

Chronosequences of vegetation on retrogressed sites subjected to fire at varying frequencies was similar to response of burned mesic tallgrass prairie (Anderson and Brown 1986). Longer fire intervals allowed woody species to increase (Bragg and Hulbert 1976; Petranka and McPherson 1979). Summer site-prep bums and ripping associated with the CCSP treatment caused a lag in plant community progression. Species composition was different under this treatment *regime* with forbs dominating the year following the summer site prep bum. As grasses increased, panicums were the primary dominant followed by little bluestem. The broomsedge bluestem component was higher on the CCSP treatment than others. The summer site prep burn apparently set back bluestems and allowed cool season grasses (panicums) and sedges to increase. Shrub species richness and percent cover were slower to increase on CCSP than retrogressed and winter burned sites (table 1 and fig. 2).

Rough-reduction bums caused smaller increases in herbaceous cover and species richness than they have in other cases (Oosting 1944, Lewis and Harshbarger 1976). However, oak-pine forest in the Ouachita Mountains do not have well

Table 3.--Mean ranks of 1988 cervid utilization frequency by treatment on the Pushmataha Forest Habitat Research Area.[1]

TREATMENT"								
H-T-3	H-T	H-I-2	CCSP	CONT	H-T-4	H-T-I	H-NT-1	RRB
19.8	18.5	17.2	17.2	13.2	9.8	8.5	7.3	2.3

[1] Means underscored with the same line are not significantly different ($\underline{P} < 0.05$).

[2] H-I-3 = harvest pine timber, thin hardwoods, uinter prescribed burn at 3 year intervals; H-T = harvest pine timber, thin harduoods; H-T-2 = harvest pine timber, thin hardwoods, winter prescribed burn at 2 year intervats; CCSP = clearcut, windrow logging slash, summer site prep burn, rip; CONT = control, no treatment; H-T-4 = harvest pine timber, thin hardwoods, uinter prescribed burn at 4 year intervals; H-T-1 = harvest pine timber, thin hardwoods, winter prescribed burn at 1 year intervals; H-NT-1 = harvest pine timber only, winter prescribed burn at 1 year intervals; RRB = rough reduction burn in uinter, at 4 year intervals.

developed midstories. Herbaceous species will increase as repeated fire eliminates smaller diameter overstory hardwoods and as pines assume dominance (Lewis and Harshbarger 1976).

Browse use on a treated area was probably related to percent cover of preferred browse and shrub species richness (table 3 and figs. 2 and 4). Woody browse is the major component of deer diets in all months except May (Jenks and others 1990). However when hard mast is available in fall and winter it comprises the major portion of deer diets (Fenwood and others 1985). Presence of preferred forbs, panicums, and sedges on a treatment probably affected use because of the selective foraging nature of deer (Vangilder and others 1982).

Screening, bedding, or escape cover may be important because deer were flushed frequently out of beds only in the H-T and CCSP treatments. The shrub component on H-T and CCSP treatments in the O-l m and l-3 m categories was primarily pine saplings. Pines probably provided a more dense horizontal cover but this parameter was not measured in this study. The presence of cover on HT and CCSP treatments may have increased use on these areas. Deer use increases on recent clearcuts but is limited to 100 m from cover on large clearcuts (Tomm and others 1981). As pine stands develop in height on regeneration areas, deer use of the central portion of the stand will increase. All portions of large (128-276 ha) 4-S year old pine stands were used in a southeast Oklahoma study (Melchoirs and others 1985).

CONCLUSIONS

The primary values associated with controlled burning and overstory removal were increased species richness and availability of preferred food items for deer, elk and possibly bobwhite quail. By varying frequency of fire, managers can shift plant communities to benefit target game species. Winter prescribed fire at 1- or 2-year intervals favored legumes and created habitat conditions favorable for bobwhite quail. Less frequent burning or no burning allowed woody browse species prefered by deer to increase on retrogressed sites (Landers 1987). A prescribed burning rotation at 2- or 3 year intervals on retrogressed sites will allow growth of important deer and quail foods. Clearcutting and site preparation provide benefits in terms of food and cover for deer. However benefits from regenerated clearcuts last only until canopy closure. Site retrogression without burning provides an important cover component to deer.

Timber management strategies that create a mosaic of retrogressed burned and unburned sites, and regeneration clearcuts with adequate provisions for hard mast production should provide management flexibility to meet habitat needs of most game species. The effects of growing season bums on maintaining retrogressed sites for forage production should be evaluated in a similar manner. The long-term effects of prescribed fue on vegetation response and site quality in mountainous terrain should also be evaluated.

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FIRE MANAGEMENT

COST-EFFECTIVE WILDERNESS FIRE MANAGEMENT: A CASE STUDY IN SOUTHERN CALIFORNIA

Christian A. Childers and Douglas D. Piirto[1]

Abstract-Federal wilderness fire management policies have been scrutinizedsince the catastrophic fires in the Greater Yellowstone Area in 1988. While wilderness tire management objectives are still aimed at recreating natural fire regimes, all USDA Forest Service fire management programs must be cost-effective. Since current Forest Service economic analyses do not fully represent the value of fire to wilderness, a cost-effectiveness analysis was developed to compare wilderness fire management options. The analytical procedure is briefly reviewed, illustrated through a southern California case study and case study results arc discussed. These results suggest that containment of some tires may be more cost-effective than current control-oriented practices.

Federal wilderness fire management policies have been scrutinized since the catastrophic fires in the Greater Yellowstone Area in 1988. Catastrophic,in this context, is a tire of any size that results in excessive resource damage, excessive suppression costs, excessive damage to private inholdings, or loss of life (Savcland 1986). No lives were lost in Yellowstonc and many have argued the bcnctits, rather than damages, of these fires to the Yellowstone ecosystems, but private lands were damaged and suppression costs were excessive (US Senate 1988). While wilderness fire management objectives are still aimed at recreating natural tire regimes, all Forest Service fire management programs must be cost-effective. If these objectives were difficult to implement in Yellowstone, they will be *even more so in* southern California, where chaparral covered wilderness areas are often surrounded by high valued private property and improvements. The Forest Service's range of options to meet these objectives include the use of appropriate suppression responses and prescribed tire.

Prescribed tires can take two forms: prescribed natural fires and management ignited prescribed fires (USDA Forest Service 1989). All prescribed fires are monitored and managed through the use of detailed bum plans (USDA Forest Service 1989). Theoretically, the only difference between the two forms of prescribed tire is the source of the ignition, but the timing of the fires is also often different. Prescribed natural fires are naturally occurring unplanned ignitions usually caused by infrequent summer or fall lightning storms. Management ignited prescribed *fires* are ignited by Forest Service personnel on their own time schedule when burning conditions and resource availabilities are optimal (usually late fall, winter, or spring in southern California).

Any fire not classified as a prescribed tire is a wildfire and must receive an appropriate suppression response. These responses range from intensive suppression efforts aimed at keeping the tire as small as possible (a control response) to containment or confinement responses. Containment means surrounding a fire with minimal control lines and utilizing natural barriers to stop its spread. Confinement means limiting a *fire's* spread to a predetermined area principally using natural barriers, preconstructed barriers, or environmental conditions (USDA Forest Service 1989).

A cost-effectiveness analysis has been developed to compare these options for wilderness fire management programs (Childers and Piirto 1989). In this analysis, approximating the average annual burned area of the natural fire regime is defined as the objective, fire gaming is used to develop representative fire costs and sizes, and decision trees are used to develop expected annual cost and burned area values for a range of fire management alternatives. This paper briefly reviews the analytical procedure, illustrates the procedure through a southern California case study (two contiguous wilderness areas on Los Padres National Forest, Santa Barbara, CA.), and discusses the case study results.

THE STUDY AREA

Our case study area comprises 23 1,500 acres of the Dick Smith and San Rafael Wilderness Areas on Los Padres National Forest (fig. 1). The vegetation of this area is predominantly chaparral brush species, including chamise (Adenostoma fasciculatum), assorted ceanothus and manzanita species (Ceanothus spp. and Arctostaphylos spp.), two types of scrub oak (Ouercus dumosa and O. turbinella) and several other pyrophytic shrubs. The chaparral intergrades with coast live oak [Ouercus aerifolia) in some riparian areas, big cone Douglas fir (Psuedotsuga macrocarpa) and digger pine (Pinus sabiniana) on some north slopes, and a variety of other pines at higher elevations. Fire is a natural component of all of these ecosystems.

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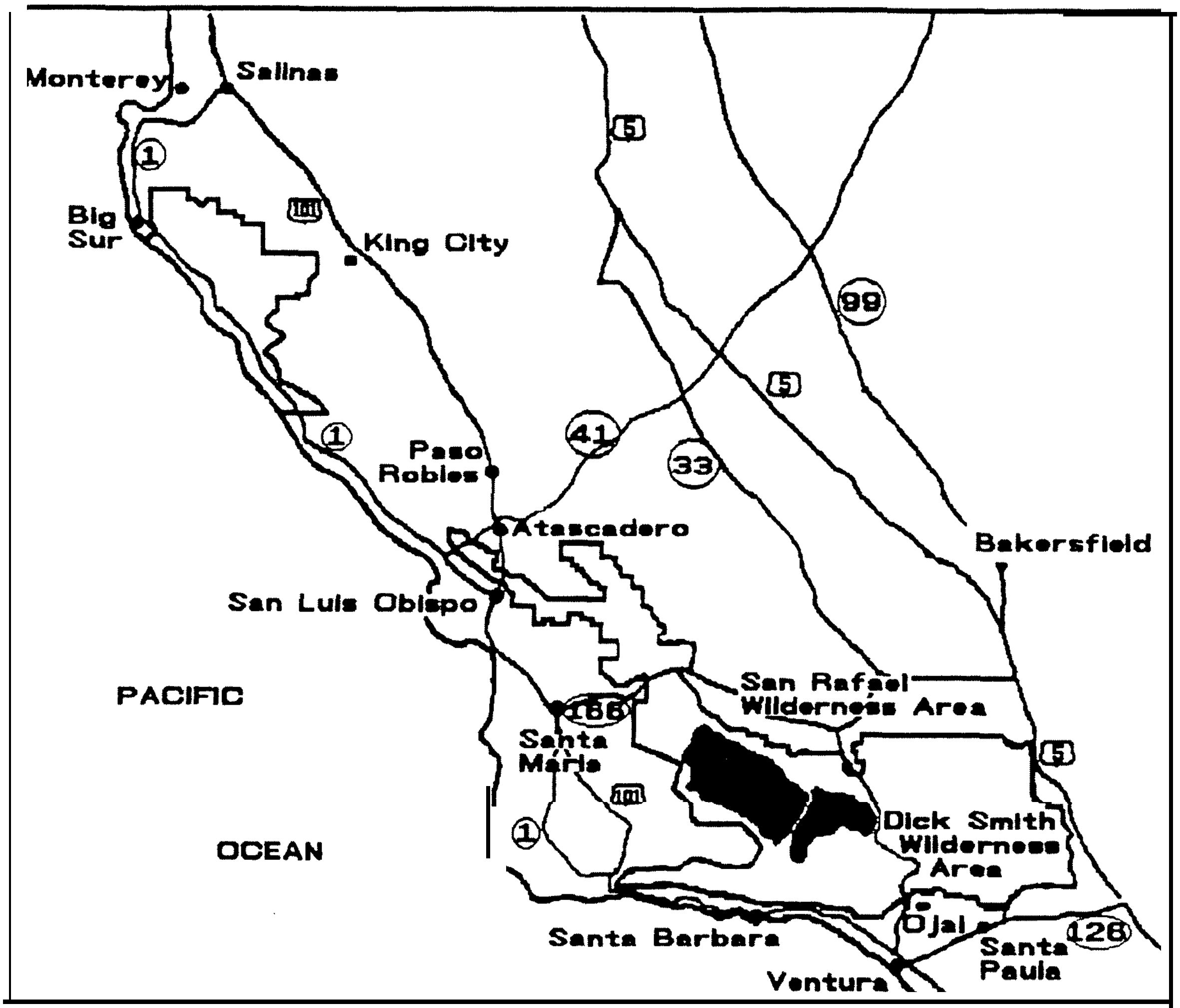

Figure l--Los Padres National Forest, with the Dick Smith and San Rafael Wilderness Areas highlighted

COST-EFFECTIVENESS ANALYSIS

Most Forest Service economic analyses use cost-benefit models. For example, economic analysis of forest level fire management programs is based on the Cost Plus Net Value Change (C + NVC) model (USDA Forest Service 1987). C + NVC computes the sum of program costs and the quantifiable (in monetary terms) effects of fire on resource values. To be efficient, these cost-benefit analyses must include the effects of fire on all relevant resources. C + NVC models currently include fire effect values for many primary forest resources such as timber, minerals, and forage, and many wilderness outputs such as water, fish and wildlife (measured in numbers of visits by hunters and fishermen), and recreational use (USDA Forest Service 1987). Fire's effects on these resources can be and usually is much different than its effects on a wilderness ecosystem. Since the primary economic value of wilderness remains undefined, fire's effects on wilderness also remain undefined. A cost-benefit analysis which does not include all of the relevant costs and benefits will be incomplete, and often misleading (Williams 1973). Therefore, analyses based solely on C + NVC models are inadequate for wilderness fire management planning.

Saveland (1986) avoided this C + NVC problem in a cost-effectiveness comparison of tire management options for the Frank Church-River of No Return Wilderness Area. In his Analysis, the costs of each alternative were the expected annual suppression costs. "Effectiveness" was the approximation of the average "natural" annual burned ares based on what fire history studies revealed. Saveland (1986) justified this well: Plant communities require a certain amount of fire, just as they require a certain amount of precipitation. ..Altering the average annual burned area would be like altering the average annual rainfall. Though Saveland's analysis involved a different fire regime and setting, his definitions and much of his methodology are appropriate for southern California's chaparral.

Cost-effectiveness analysis (CEA), in its truest form, compares the costs of different alternatives, where each alternative will meet the desired objectives, or have the same effects. A CEA has five key elements: the objective; the alternatives; the costs; the model; and a criterion for ranking the alternatives (Quade 1967).

The Objective

The most important, and often the most difficult, step in CEA is a clear definition of the goals or the objectives. Public policy usually includes several goals or objectives and these are often conflicting (Quade 1982). Forest Service Policy is no exception. The Forest Service Manual (USDA Forest Service 1986) defines two objectives *for* wilderness *fire* management:

1. (to) permit lightning caused fires to play, as nearly as possible, their natural ecological role in wilderness;

2. (to) reduce to an acceptable level, the risks and consequences of wildfire within wilderness or escaping from wilderness.

The value of fire playing its natural ecological role is currently unquantifiable in monetary terms; thus, it is not included in Forest Service economic evaluations. The consequences of fire are more straight forward. They include resource and property damage and suppression costs. Risk, while also difficult to quantify monetarily, is the probability of a fire resulting in excessive resource damages or suppression costs. Current Los Padres National Forest tire management plans stress the second objective (reducing the risks and consequences); proposed wildfire responses are suppression intensive (control and contain strategies) and no wilderness prescribed fires have been planned. The Forest's current wilderness *fire* management objective might be to respond to and suppress each ignition at minimal cost, regardless of annual burned area. If we are interested in allowing lightning fire to play its natural role, this must be included in the analysis. Our redefined objective might then be to recreate the natural *fire* regime at minimal cost.

To further define this objective, we need to look at the natural fire return interval. By defining the maximum time interval between fires, we can determine the minimal average annual burned area required to recreate the natural fire regime. Research suggests that the area's chaparral historically burned every 30 years (Byrne 1979, Minnich 1983). Los Padres National Forest fire records (1911-1987) suggest that the chaparral bums every 45 years (USDA Forest Service 1988). Forty-five years probably represents the maximum fire return interval since these records were taken while all fires were being actively suppressed. Using the 45-year return interval, an average of over 5,000 acres of the 23 1,500-acre study area would have to bum annually. It is important to note that this 5000-acre average is a long-term objective, not an annual goal. In some years, 20,000-30,000 acres might bum while in other *years* no prescribed fires will be implemented (just as lightning strikes frequently in some years, while no lightning activity occurs in other years).

The Alternatives

Four alternatives were chosen for the Los Padres CEA.

1. Alternative 1 is the Forest Service's past policy: Control all wildfires regardless of cause, and attempt to meet annual burned area objectives through prescribed burning.

2. Alternative 2 is the fire management strategy proposed in the Los Padres' Land Management Plan: Contain all fires which occur under low intensity and control all moderate to high intensity tires, while pursuing an active prescribed burning program.

3. Alternative 3 (the Confinement Alternative): Confine all low intensity starts, contain moderate to high intensity starts, and control only the starts which occur under extreme tire weather conditions (augmented by prescribed burning as needed).

4. Alternative 4 (the Prescribed Natural Fire Alternative): The same as Alternative 3, with the addition of an approved plan for prescribed natural fire management.

The Costs

Only the relevant variable costs should be included in a CEA (Quade 1982).

Fixed costs--those that remain the same for each alternative--should not be included. For this analysis, fixed costs include fire suppression equipment, suppression manning levels, and fire management Personnel,, because these forestwide resource level requirements are based on over 100 fires a year and an average of less than two ignitions occur annually in the ease study area. The variable *costs* that must be considered are annual suppression costs, prescribed fire costs, and NVCs for fires originating in the study area.

The 'Model

A model is a simplified representation of the real world which includes all of the relevant features (Quade 1967). Decision *trees* can be used to evaluate alternative fire management programs in the face of uncertainties about future fire occurrences, weather, behavior, and sizes (Hirsch and others 1981). Decision trees are used to develop expected values. Expected values are probability weighted averages of all possible outcomes. Expected values are not predictions of actual future costs due to the many variables involved in wildland tires; they provide relative values for comparison. For our analysis, decision tree probabilities were derived from fire history records. The range of cost and burned area values were developed through fire gaming since no historic or comparable fire history records were available for containment, confinement, or prescribed natural fire responses (Childers and Piirto 1989).

Alternative	Representative Location	calm	Weather Pattern	Strategy	Gamed Cost	Expected Value Cost	Gamed s i	Expected Value Size	Gamed Fire NVG	Expected Value NVG
Alternative 4	R.L. 1 0.296	Lightning a385	A(.441)	CF(.75)	$3,095	$117	4.0	a2	$44	$2
				Rx(.25)	$3,689	$46	4.0	0.1	$44	El
			B(.118)	CF(.75)	$6,530	$66	450.0	4.5	$1,976	$20
				Rx(.25)	$6,941	$23	450.0	1.5	$1,976	s7
			C(.147)	CA	$51,730	$867	265.0	4.4	$5,569	$93
			D(.294)	CA	$41,403	$1,387	390.0	13.1	$6,825	$229
		Person 0.615	A(.100)	CF	$3,095	$56	40	a1	$44	$1
			B(.200)	CF	$6,530	$238	450.0	164	$1,976	$72
			C(.200)	CA	$51,730	$1,883	265.0	9.6	$5,569	$203
			D(.500)	CA	$41,403	$3,769	390.0	35.5	$6,825	$621
	RL2 0.25	Lightning 0.933	A(.441)	CF(.75)	$2,887	$223	3.0	a2	($138)	($11)
				Rx(.25)	$47,814	$1,230	740.0	19.0	($20,736)	($533)
			B(.118)	CF(.75)	5163,384	$3,373	1,950.0	48.3	($24,608)	($508)
				Rx(.25)	$182,254	$1,254	1,965.0	13.5	($25,236)	($174)
			C(.147)	CA	$93,335	$3,200	780.0	267	($19,585)	($672)
			D(.294)	CA	3527,336	S4.416	4,200.0	35.2	$240,938	$16,522
		Person 0.067	A(.100)	CF	S2887	$5	3.0	0.0	($138)	so
			B(.200)	CF	$163,384	$547	1,950.0	6.5	($24,608)	($82)
			C(.200)	CA	$93,335	$313	788.0	26	($19,585)	($66)
			D(.500)	CA	$527,336	$36,162	4,200.0	288.0	$240,938	$2,018
	RL3 a364	Lightning 0.938	A(.441)	CF(.75)	$2,525	$285	0.1	0.0	($3)	$0
				Rx(.25)	$4,821	$181	0.1	0.0	($3)	$0
			B(.118)	CF(.75)	$401	S12	0.1	0.0	($3)	$0
				Rx(.25)	$110,546	$1,113	833.0	8.4	($16,435)	($166)
			C(.147)	CF(.75)	$17,807	$670	40.0	1.5	($1,315)	($50)
				Rx(.25)	$88,639	$1,112	835.0	10.5	($20,544)	($258)
			D(.294)	CA	$910,362	$91,383	2,600.0	261.0	$4,847	$487
		Person 0.062	A(.100)	CF	$2,525	$6	a1	0.0	($3)	$0
			B(.200)	CF	$401	$2	a1	0.0	($3)	$0
			C(.200)	CA	$17,807	$80	40.0	a2	($1,315)	($6)
			D(.500)	CA	$910,362	$10,273	2,600.0	29.3	$4,847	$55
	R.L. 4 0.091	Lightning 1.000	A(.441)	CF(.75)	S3.475	$105	5.0	0.2	($116)	($3)
				Rx(.25)	$48,227	$484	748.0	7.4	($20,734)	($208)
			B(.118)	CF(.75)	$167,088	$1,346	1,955.0	15.7	($24,553)	($198)
				Rx(.25)	$183,371	$492	1,970.0	5.3	($25,181)	($68)
			C(.147)	CA	$98,496	El.318	800.0	10.7	($18,904)	($253)
			D(.294)	CA	5973,519	$26,046	2,800.0	74.9	$60,638	$1,622
				Expected Values:		$194,083		9425		$18,697
				Expected Annual Values:		$341,586		1,658.8		$32,907

Figure Z-The decision tree for Alternative 4

A decision tree must be completed for each alternative, using the same probabilities but with different suppression responses and thus different cost and burned area values. The probabilities for each branch of the trees were calculated from the 25-year (I 963-87) fire history of the San Rafael and Dick Smith Wilderness Areas (Childers and Piirto 1989). The decision tree for Alternative 4 of the Los Padres study (fig. 2) illustrates the values and probabilities which were developed for our CEA. Alternative 4's decision tree is presented since it is the most complex decision tree (this is the only alternativc in which strategy is not solcly based on weather pattern).

Fire gaming is the prediction of reprcsentative fire sizes by fire management professionals. Predictions are based on the interactions of estimated tire behavior conditions and given suppression force responses (Harrod and Smith 1983). Our garners included the fire management personnel from the Forest Supervisor's Office and from each of the three ranger districts responsible for the case study area. The "games" consisted of first mapping an overlay of the free-burning fire spread (without any suppression efforts) for a series of time periods. Four weather patterns were mapped at each location and these "fires" were then controlled, contained, confined and managed as prescribed natural tires to develop the cost and burned area values needed to fill in each decision tree. Net Resource Value Changes (NVCs) were calculated using the Forest's 1988 NVC values based on acreage burned by intensity level in each watershed (Childcrs 1991).

Management ignited prescribed fire costs were subjectively estimated at $50 per acre by the garners and by the Santa Barbara Ranger District's Fuels Management Staff. This is more expensive than most recent prescribed fires adjacent to the case study wilderness areas, but initial wilderness prescribed tires will probably be expensive due to the age and continuity of the fuelbeds, remoteness of the fires, and limitations on control lines and the use of mechanized equipment in wildcmess.

A Criterion

The criterion for ranking alternatives depends on the agency's goals and objectives. Many different rankings are possible. For this analysis, we defined our objective as the recreation of the natural fire regime at minimal cost. Given current budgetary constraints, minimizing costs regardless of burned area might be the agency's actual objective. The sources of proposed expenditures (i.e., forest fire fighting funds vs. program or budgeted dollars) might be important considerations. Risk is also a concern. Finally, the ignition source and timing of the tires might be important to prescribed tire planners. Therefore, all of this information must be provided.

RESULTS

Four weather patterns were gamed at each of four fire locations: the first set at representative fire location (RL) 1, the second at RL 2, the third at RL 3, and the fourth set under double ignition conditions (two tires occurring simultaneously) using RLs 2 and 4 (Childers 1991). The results of these games are presented in table 1. These values

Table 1--Final size and cost figures for gamed fires.

	CONTROL		CONTAIN		CONFINE		Rx Natural Fire	
	Size (acres)	cost (8)	Size (acres)	cost ($)	Size (acres)	cost ($)	Size (acres)	cost ($)
Representative Fire Game 1								
Ueather Pattern A	0.5	7,693	0.5	5,113	4.0	3,095	4.0	3,689
Weather Pattern B	2.0	7,900	2.0	4,722	450.0	6,530	450.0	6,941
Weather Pattern C	120.0	84,592	265.0	51,730	(not gamed)		(not gamed)	
Ueather Pattern D	40.0	36,989	390.0	41,403	(not gamed)		(not gamed)	
Representative Fire Game 2								
Weather Pattern A	0.3	3,129	0.3	2,756	3.0	2,887	740.0	47,814
Weather Pattern B	70.0	40,498	780.0	47,792	1,950.0	163,384	1,965.0	182,254
Ueather Pattern C	145.0	86,604	780.0	93,335	(not gamed)		(not gamed)	
Weather Pattern D	1,090.0	366,894	4,200.0	527,336	(not gamed)		(not gamed)	
Representative Fire Game 3					0.1			
Ueather Pattern A	0.1	8,415	0.1	4,427	0.1	2,525	0.1	4,821
Weather Pattern B	0.1	7,541	0.1	4,896		401	833.0	110,546
Weather Pattern C	5.0	18,249	10.0	9,029	40.0	17,807	835.0	88,639
Weather Pattern D	500.0	370,193	2,600.0	910,362	(not gamed)		(not gamed)	
Representative fire Game 4	0.5							
Weather Pattern A	75.0	3,275	0.5	2,903	5.0	3,475	740.0	48,227
Weather Pattern B	[illegible]	44,518	785.0	61,549	1,955.0	167,088	1,970.0	183,371
Ueather Pattern C	310.0	136,861	800.0	98,496	(not gamed)		(not gamed)	
Weather Pattern D	2.260.0	851,674	2.800.0	973.519	(not gamed)		(not gamed)	

were then run through the appropriate alternatives' decision trees (as per Childers and Piirto 1989) and expected values for average annual suppression costs, burned area, and NVCs were calculated for each decision tree. These results are presented in table 2.

Table 3 includes a breakdown of annual suppression costs and acreage into prescribed fire and forest fire fighting (FFF) costs. Table 3 also illustrates the prescribed bum acreage and costs that would be required to meet our 5,000-acre average annual burned area objective under each alternative. All cost values are presented in 1988 dollars.

DISCUSSION

One of the most obvious observations from the decision tree results (table 2) and the total cost of implementing each alternative (table 3) is that alternatives 1 and 2 are very similar, as are alternatives 3 and 4. This can be attributed to the similarity of the containment and control responses and the confinement and prescribed natural fire responses as they were used on many of the gamed fires. One gamer concluded that they were still "fighting" the fires, even under the prescribed natural fire responses. For example, the actual dispatch cards of initial attack resources were used to determine who would respond to each fire under both containment and control; thus, many of the same resources were used on both of these strategies. The run cards were heavily modified for confinement and prescribed natural fire responses, but the objectives of these two were often similar. Once these strategies have been implemented, familiarity with appropriate suppression responses and pre-approved prescribed fire bum plans should lead to greater differences in their results. Despite the similarities, these results do provide some valuable information for the decisionmaker.

Table 2--Average annual wildfire and prescribed natural fire cost, cost and San Rafael Wilderness Areas highlighted per acre managed, average annual burned area, and average annual cost per area burned for four alternative fire management programs for the Dick Figure 2--The decision tree for Alternative 4 Smith and San Rafael Uilderness Areas

	Average annual cost	Cost per acre managed	Average annual burned area (acres)	Average annual cost per burned acre
Historical			5000+	
Alternative 1	9197,611	$0.85	394.8	$500.53
Alternative 2	9195,474	SO.84	447.2	$437.11
Alternative 3	8334,773	$1.45	1,580.0	$211.88
Alternative 4	8341,586	$1.48	1,658.8	$205.92

Table 3--Breakdown of total average annual suppression/management costs and burned areas by source

	ALTERNATIVE 1	2	3	4
Wildfire Acreage:	394.8	447.2	1,580.0	1,543.2
Rx Natural Fire Acreage:	0.0	0.0	0.0	115.6
Mgt Ign Rx Fire Acreage:	4,605.2	4,552.8	3,420.0	3,341.2
F.F.F. costs:	$197,611	$195,474	$334,773	$331,140
Rx Natural Fire Costs:	0	0	0	$10,446
Mgt Ign Rx Fire Costs:	$230,260	$227,640	$171,000	$167,060
Total Annual Costs:	$427,871	$423,114	$505,773	$508,646

If the agency's goal was simply to respond to and suppress or manage each ignition at minimal cost, regardless of annual burned area, alternative 2 would be the most cost-effective. This result is due to the cost-saving advantages of containment over control on most lower intensity tires and the expensive outcomes that can result from trying to confine or manage fires in the decadent fuelbeds.

If, however, the goal is to recreate the natural fire regime (i.e., to meet the 5,000-acre average annual burned area), the decision might be a little more involved. Alternative 2 would still be the least expensive, but alternatives 3 and 4 would require much less program or budgeted dollars to accomplish the objective and result in much more of the acreage burning under natural conditions (natural ignition sources and during the natural fire season).

Containment or confinement strategies can only be used when they are less expensive than controlling a given fire (USDA Forest Service 1989). Table 1 shows that containment cost less than control 56 percent of the times it was used and that confinement cost less or about the ***same*** as control 78 percent of the times it was used. This suggests that containment and confinement are both feasible and cost-effective for our case study ares.

Risk is incorporated into the analysis through the probability of a fire resulting in excessive resource damages or suppression costs (e.g., fire 4D, which cost over $850,000 to suppress regardless of the strategy used). However, none of the confinement or prescribed natural fire responses resulted in a catastrophic fire, and it could be argued that $953,000 (the most expensive gamed fire) is **not** really catastrophic when compared to historic fires like the 1966 Wellman Fire. The Wellman Fire burned 93,600 acres of the case study area and cost over $6.2 million (in 1988 dollars) to suppress. But, since the Wellman Fire occurred under extreme site-specific weather conditions, it would receive a control response under any alternative; and, since it became catastrophic despite control efforts (the only possible response in 1966) it could happen again under any alternative. The risk of another catastrophic fire might seem greater under alternatives 3 and 4, since fires are allowed to get larger, but this is only the short term risk factor. These alternatives would allow ***more*** acres to bum under natural conditions, resulting in cleaner bums than management ignited off-season fires and larger breaks in the decadent fuelbeds, which should help to limit the size of future fires.

SUMMARY

Developing cost-effective wilderness fire management programs is a dilemma faced by many Forest Service land managers. Wilderness fire management is a requirement, but the value of fire in wilderness remains undefinable in monetary terms so it is excluded from most Forest Service economic analyses. Therefore, cost-effectiveness analysis, using the recreation of the natural fire regime as the objective, can provide important economic information. Decision trees help us predict future fire occurrence potentials, and intensive gaming efforts help us estimate fire sizes and costs associated with the implementation of appropriate suppression responses and prescribed natural fires. Case study results suggest that appropriate suppression responses could provide cost-effective alternatives to current control-oriented practices. Through this extensive and thorough cost-effectiveness analysis we can, hopefully, avoid some of the costly mistakes of past experiences in wilderness fire management.

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ADAPTIVE FIRE POLICY

James M. Saveland[1]

Abstract-Adaptive resource management is a continuous learning process in which current knowledge always leads to further experimentation and discovery. Adaptive management evolves by learning from mistakes. Designing adaptive management strategies involves four tasks. First, the problem must be defined and bounded. There is growing recognition of the need to define and bound problems at the landscape level. Second, existing knowledge must be readily accessible so that errors can be detected and used as a basis for further learning. The current information structure supporting fire management was designed to support the 10 a.m. policy and is inadequate to support current policy. Expert systems and other recent developments in artificial intelligence can provide the necessary means to develop an accessible repository of current knowledge. Third, the inherent uncertainty and risk surrounding possible future outcomes must be displayed.- Bayesian decision analysis can be used to deal with uncertainty and risk. Fourth, balanced policies must be designed. These must provide for resource production and protection while creating opportunities to develop better understanding. Signal detection theory and receiver operating characteristic curve analysis provide tools to help design balanced policy. These concepts are illustrated by applying them to the problems surrounding wilderness fire management and the need for long-range tire danger information.

INTRODUCTION - SEEKING A BALANCE

The need to balance competing and often conflicting objectives is a problem whenever policy is being made. In resource management, there is often the need to balance utilization with preservation. The disputes about wilderness designation and forestry activities in spotted owl and red-cockaded woodpecker habitats are controversies in search of a balance point. Several aspects of fire management require a balance. In wilderness fire management, the role of fire in perpetuating disturbance regimes in near-natural landscapes must be balanced with the necessity of protecting resources that would be damaged by fire. In smoke management the use of prescribed fire must be balanced with minimizing the nuisance of smoke. During periods of high fire danger, shutting down the woods to protect them must be balanced with the need to keep the woods open for people who earn their livelihood there. At the interface between wildland and urban areas, it is necessary to balance the threat of wildfire and the costs of risk-reduction measures. How should government regulatory agencies go about determining the balance point? And how can they describe their search for balance and its results to affected parties?

ADAPTIVE RESOURCE MANAGEMENT

Adnptivc resource management (Clark 1989, Holling 1978, Saveland 1989, Thomas and others 1990, Walters 1986) recognizes the fact that the knowledge we base our decisions on is forever incomplete and almost always shrouded in uncertainty. Management is a continual learning process that evolves by learning from mistakes. Several authors have expressed the importance of learning from failure. "You have to accelerate the failure rate to accelerate the success rate" (Peters 1987). "Intelligent needs to be tolerated.

Multitudes of bad ideas need to be floated and freely discussed, in order to harvest a single good one" (Toffler 1990). "The willingness to risk failure is an essential component of most successful initiatives. The unwillingness to face the risks of failure--or an excessive zeal to avoid all risks--is, in the end, an acceptance of mediocrity and an abdication of leadership" (Shapiro 1990).

Designing adaptive policy involves four tasks. First, the management problems must be defined and bounded, often in terms of objectives and constraints. There is an increasing awareness of the need to define resource problems from a landscape perspective (Forman and Godron 1986, Naveh and Lieberman 1984). With the proliferation of geographic information systems, the importance of defining and bounding problems at the landscape level will become even more apparent.

Second, existing knowledge must be readily accessible so that errors can be detected and used as a basis for further learning. Walters (1986) used models to represent existing knowledge. The field of artificial intelligence, especially knowledge-based systems, provide additional capability to capture knowledge (Saveland 1990).

Current fire information systems are inadequate. Most, if not all, fire information systems were designed to support the 10 a.m. policy and do not adequately deal with the complexities of modem fire management. Fire occurrence reports track the efficiency of the suppression effort. When policy was changed to allow prescribed natural fires, only half of the fire ***occurrence report*** form ***for*** the Forest Service had to be filled

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out for these fires. These reports provide almost no useful historical information for managing wilderness and park tire management programs. In addition, adequate cost data is severely lacking, preventing useful economic analysis. Structure and site characteristics in the wildland urban interface are not recorded, preventing analysis of structure losses. The national weather data library is known for its missing and questionable data. Currently, there are no links between national fire occurrence databases and fire weather databases. Entrepreneurial fire managers have been able to download the data into relational databases to conduct analysis. In addition, there are plans to convert the national databases into a relational form. Forest Service fire occurrence data resides in Fort Collins while Park Service data resides in Boise in different formats, further complicating the sharing of data and historical analysis. Prescribed bum plans exist in paper copy or as a word processing document on a computer, the vast historical information largely inaccessible, tucked away in personal file cabinets. The collapse of wilderness and park fire management during the summer of 1988 was not so much a failure of policy as a reflection of an outdated information system's inadequacy to support fire management decisions in today's complex world. Information needs analysis have been conducted recently and the situation is rapidly changing for the better. In addition the coming explosion of GIS technology, with the shortage of spatial data, will improve the situation dramatically.

Third, uncertainty and its propagation through time in relation to management actions must be addressed. Fire managers all too often live in a fairytale world of deterministic models that ignore uncertainty. Bayesian decision analysis offers one means of coming to terms with the inherent uncertainty and risk.

Fourth, balanced policies must be designed. These must provide for continuing resource production and protection while simultaneously probing for more knowledge and untested opportunity. Signal detection theory provides one mechanism to help design balanced policies.

WILDERNESS FIRE MANAGEMENT - AN EXAMPLE

Signal detection theory (Egan 1975, Saveland and Neuenschwander 1990, Swets and Pickett 1982, Wilson 1987) divides a decision problem into three parts: state of nature, response, and outcome (fig. 1). State of nature refers to presence or absence of a signal at the time a person makes a response. The signal is either present or absent. Responses are alternative actions decision makers must choose between. Decision makers can control their response, but have no control over the state of nature. They can respond by saying that they detect signals or that they do not. The point where a person switches between responding yes and responding no is the threshold of evidence. If the signal strength is greater than the threshold of evidence, the response is yes. If signal strength does not reach the threshold of evidence, decision makers will not detect the signal and the response will be no. The threshold of evidence can be varied. As the threshold of evidence is increased, a person is more likely to say no, thus reducing the number of false alarms, but increasing the number of misses. As the threshold of evidence is decreased, a person is more likely to say yes, thus reducing the number of misses and increasing the number of false alarms. This inherent trade-off between misses and false alarms provides the opportunity to find a balance point. A response combined with a state of nature results in an outcome for which the decision maker has some level of utility. One of the strengths of decision theory is that it separates the decision from the outcome.

Response	**State of Nature**	
	Signal Present s	**Noise** n
Yes Y	**HIT** $P(Y\|s)$	**FALSE ALARM** $P(Y\|n)$
No N	**MISS** $P(N\|s)$	**CORRECT REJECTION** $P(N\|n)$

Figure 1 .-The signal detection paradigm.

Response	State of Nature: Undesirable Fire	State of Nature: Desirable Fire
Initial Attack	HIT	FALSE ALARM (????)
Do not Initial Attack	MISS (Yellowstone '88)	CORRECT REJECTION

Figure 2.--Signal detection for wilderness fire.

The wilderness fire decision can be divided into two responses that combine with two states of nature to produce four possible outcomes (fig. 2). The decision maker could choose to suppress a tire that, had it been allowed to bum, would have eventually exceeded acceptable conditions (i.e. become a wildfire). This hit is a desirable outcome because money has been saved by putting the fire out when it was small. Second, the decision maker could choose to let such a fire bum, in which case it would have to be put out later. This miss is an undesirable outcome because the costs of putting out a fire increase exponentially as the fire's size increases.

Third, the decision maker could choose to put out a fire that, had it been allowed to bum, would not have exceeded acceptable conditions (i.e. would have stayed within prescription). This false alarm is an undesirable outcome because an opportunity to allow fire to play its natural role has been missed. Fuel management benefits are not realized, firefighters are exposed to unnecessary risk of injury, and unnecessary costs associated with the suppression effort are incurred. Perhaps most important, nothing is learned. There is no increase in knowledge. Although this block and the hit block can be discussed conceptually, they are counterfactuals, and there is no way to determine these blocks in reality.

Finally, the decision maker might choose to let a fire bum, and this fire would stay within prescription. This correct rejection is another desirable outcome. Fire is allowed to play its natural role in maintaining various ecosystems, benefits associated with fuel management are realized, and the costs of fire suppression are saved.

Thus, the strategy for wilderness fire management is to allow as many non-problem-causing fires to bum as possible. For fires that are expected to cause problems, quick suppression while the fire is small is necessary to minimize costs and damages.

Long-range assessments of fire danger are key factors when managers have to decide whether to suppress specific wilderness fires. The fire danger prediction task can also be put into a signal detection framework (fig. 3). When

Response	State of Nature: High Danger	State of Nature: Low Danger
Predict High	HIT	FALSE ALARM
Predict Low	MISS	CORRECT REJECTION

Figure 3.--Signal detection for long-range forecasting.

lightning ignites fires early in the season, there must be an assessment of what fire danger conditions are likely to evolve later in the season.

An analytical procedure called the receiver operating characteristic (ROC) curve is an inherent part of signal detection theory. The ROC curve is a plot of the percentage of hits on the Y axis against the percentage of false alarms on the X axis (fig. 4). An ROC curve summarizes the set of 2 x 2 matrices (fig. 3) that result when the threshold of evidence is varied continuously, from its largest possible value down to its smallest possible value. The upper left-hand comer, where the percentage of hits equals one and the percentage of false alarms equals zero, represents perfect performance. The positive diagonal, where the percentage of hits equals the percentage of false alarms, is what would be expected based on pure chance.

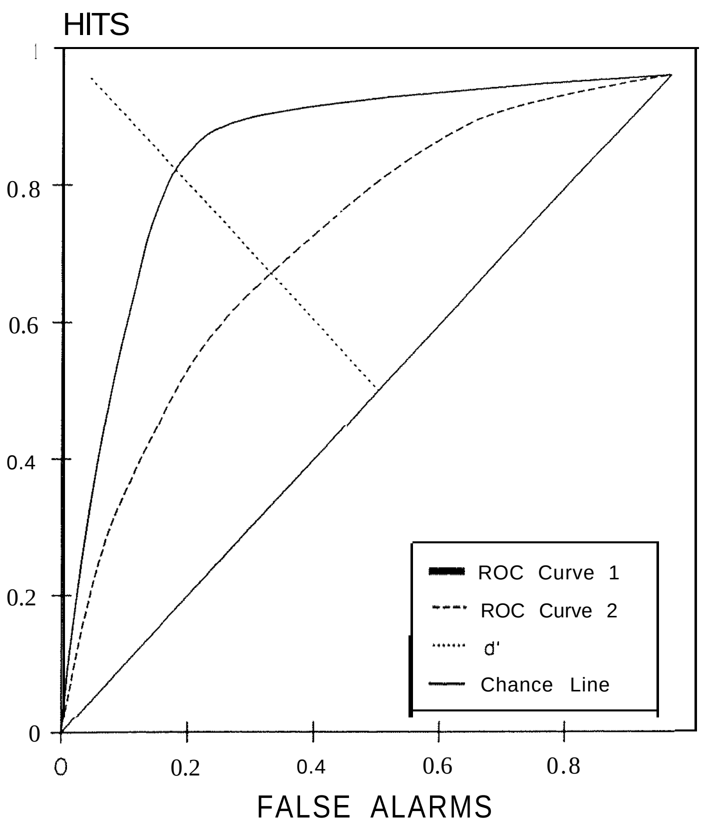
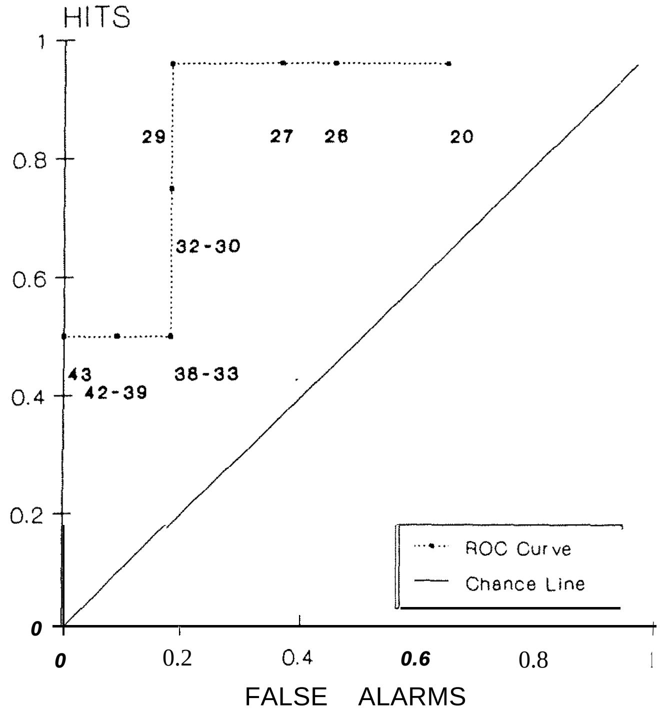


Figure 4.--Receiver operating characteristic (ROC) curves.

Various strategies can be used to select an appropriate threshold of evidence. One such strategy, minimax, attempts to minimize false alarms while maximizing hits.

The ROC curve has four important properties which correspond to the four tasks required to implement adaptive resource management. First, ROC analysis requires that the problem be defined explicitly. In this case, it is necessary to say just what constitutes high fire danger and what does not. In the example to follow, fire danger is defined in terms of the energy release component (ERC) of the national fire danger rating system. If the ERC at a certain date early in the fire season exceeds a threshold (predict high fire danger) and the ERC exceeds a critical value later in the fire season (late-season fire danger is high), the result is a hit. If the ERC early in the fire season exceeds a threshold but the ERC does not exceed the critical value later on, the result is a false alarm. Miss and correct rejection can be defined in a similar manner. The threshold varies to display the possible trade-offs. The critical value is site specific. The manager can select a critical value based on past experience. For example, noting that tires start to spread rapidly on north slopes, develop into crown tires, and become uncontrollable at a certain value, would be a suitable critical value. An explicit definition of fire danger and fire severity will enhance communication between fire staff and line officer decision makers, and between the line officer and the public. Second, the ROC curve displays skill prediction, or how much confidence to place in the prediction. A point near the chance line does not warrant much confidence, while a point close to the upper left-hand comer is reliable. The area under the curve is a measure of skill prediction and can be compared to chance. Skill prediction can also be considered a measure of our current state of knowledge. As more knowledge is obtained prediction systems should improve, and this improvement should result in new ROC curves that get progressively closer to the upper left-hand comer, which represents perfect prediction. Third, the ROC curve expresses the inherent uncertainty of the predictions in terms of Bayesian probability. Each point on the curve corresponds to percentages of hits, false alarms, misses, and correct rejections on a scale of zero to one. Fourth, the ROC curve displays the possible trade-offs between misses and false alarms as the threshold of evidence varies. A high percentage of hits is often possible only when there is a high percentage of false alarms. To reduce the number of false alarms often implies an increase in the number of misses. Selecting an operating point on the ROC curve is selecting a balance point.

Figure 5 is an ROC curve developed for the Westfork Ranger District weather station. The Westfork weather station collects data used by those who make decisions about prescribed natural fires in a portion of the Selway-Bitterroot Wilderness. Fire danger prediction is explicitly defined by a threshold ERC early in the fire season and a critical ERC later on in the season. A critical ERC value of 52 was chosen. During the period from 1973 to 1987, the ERC reached 52 in four of the fifteen years (1973, 1977, 1978, and 1979). Thus in 73 percent of the years, the ERC does not exceed 52 (low danger years), while 27 percent of the years, the ERC exceeds 52 (high danger years). The ROC curve displays percentages of hits and false alarms for threshold ERC values from 20 to 43. The probability that the ERC exceeds 29 on July 10 given that the ERC exceeds the critical



1988 ERC 41
Critical ERC = 52
Data: 1973-1987

Figure 5.--Long-range ERC forecast for Westfork R.D. on July 10.

value of52 later on in the fire season (hit) is 1.0. The probability that the ERC exceeds 29 on July 10 given that the ERC does not exceed the critical value of 52 later on in the tire season (false alarm) is 0.18. it follows that the probability of a miss at that point on the ROC curve is 0 and the probability of a correct rejection .82. Skill prediction is high. The area under the ROC curve is 0.91. If it were important to minimize the number of false alarms, the threshold of evidence could be increased to 43. This would reduce the number of false alarms by 18 percent, but would increase the number of misses by 50 percent. Saveland (1989) presents a similar analysis for Yellowstone National Park.

CONCLUSIONS

Most resource management controversies require seeking a balance between competing, conflicting objectives. Finding a balance is an integral part of adaptive resource management. Implementing adaptive policy involves four steps: defining and bounding the problem, representing current knowledge, representing the uncertainty surrounding our predictions of the future, and designing balanced policies that provide for resource production and protection while permitting experimentation aimed at increasing knowledge. Receiver operating characteristic curve analysis can assist adaptive resource management. ROC forces explicit definitions, represents current knowledge through skill prediction and readily displays uncertainty and possible tradeoffs.

Adaptive resource management points out the limits of our current knowledge and the importance of increasing our knowledge of the structure and function of natural resources. In fact, knowledge can be considered a resource. Surely our policies should promote the acquisition of ***new knowledge.***

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PRESCRIBED FIRE AND VISUAL RESOURCES IN SEQUOIA NATIONAL PARK

Kerry J. Dawson and Steven E. Greco[1]

Abstract-The management goals at Sequoia National Park are to restore the tire climax ecosystems of the giant sequoia-mixed conifer forests to more natural conditions through the reintroduction of tire after many years of tire suppression. Objectives of prescribed tire must address the need for mitigation in "special management areas" (SMAs) that are under heavy impact from human use. The sensitive treatment of scenic **resources in these SMAs can augment natural diversity if the structure of "naturalness" is given priority over uniformity of fuel load reduction. Management actions should seek to: (1) mimic natural fire patterns** whenever possible; (2) avoid **artificial** infrastructure as bum unit determinants; and (3) conserve and enhance scenic resources in areas threatened by intensive human use. Visual resources were inventoried and management objectives recommended.

INTRODUCTION

Prescription fire began in the Giant Forest of Sequoia National Park in 1979. Since then several bums have been conducted. The management objectives of these bums have been primarily to reduce hazardous fuel accumulations and to restore the forest to a more natural ecosystem while sustaining populations of giant sequoias (*Sequoiadendron giganteum*) (NPS 1987a). The overall bum pattern on the forested landscape was originally designed to prevent or minimize the potential risk of a catastrophic fire sweeping over the Giant Forest plateau. In an effort to accomplish these objectives, park resource managers were charged with a variety of sometimes conflicting objectives An independent review was commissioned by then Director of the National Parks Service Western Region, Mr. Chapman in 1986.

The independent review of the giant sequoia-mixed conifer prescribed burning program of Sequoia and Kings Canyon National Parks by the Christensen Panel resulted in a report (Christensen and others 1987). Among many recommendations were instructions to explicitly address aesthetic concerns within the park's "Showcase" areas. The Sequoia Natural Resources Management Division has since changed the term "Showcase" to Special Management Areas. The Panel Report specifically recommended consultation with landscape architects in the development of bum plans with special emphasis on the SMAs.

Special Management Areas are located in the most heavily visited portions of the park. The three primary sources of visual impact within these areas that must be mitigated are the reintroduction of firc, visitor overuse, and overgrown thickets of non-fire climax species. The Sequoia and Kings Canyon Vegetation Management Plan (NPS 1987b) notes that SMAs are designated "where maintenance of natural processes is guided more by scenic concerns."

High visitation via roads and trails are a significant anthropogenic impact within an ecosystem that has management goals for 'naturalness'. The challenge of maintaining a natural aesthetic for this type of visitation is made compelling by the fact that roads and trails concentrate human impacts and have human facilities associated with them (food vendors, parking lots, restrooms, etc.). Current management goals of 'naturalness' are further complicated by historic cultural values that have developed over the past one hundred years since the establishment of the park. The named trees and logs have become 'cultural objects' along trails and roads, such as the General Sherman Tree and other named trees, groves, logs, and stumps in the Giant Forest. These areas of heavy visitation and subsequent substantial human impact must be managed more intensively and thus are termed SMAs.

As stated in the Panel Report (Christensen and others 1987), SMAs should not be seen as "static museums," created through "scene" management, but rather as a part of dynamic ecosystems, sensitively managed to preserve scenic and ecological resources. The Prescribed Fire Management Program (1987a) notes that the intention of management in these areas is not to apply a method of "greenscreening", whereby dramatically different appearing landscapes exist behind SMAs. Instead, these areas should be burned as more sensitive units with special attention given to specific goals and objectives for visual quality, environmental enhancement, and interpretation, as complemented by associated resource objectives.

The National Park Service Act of 1916 declared that "the fundamental purpose of [a National Park] is to conserve the scenery and, the natural and historic objects and the wildlife therein and to provide for enjoyment of the same in such a manner and by such a means as will leave them unimpaired for the enjoyment of future generations." Interpretation of this mandate has clearly demanded a sophisticated level of

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management since the release of the Leopold Panel Report (Leopold and others 1963). The relationship between aesthetics, scenery, and natural process is a complex natural and cultural issue that continues to evolve and will do so through ongoing multidisciplinary research. Visual resources are a prime asset in our National Parks and they must be conserved and managed sensitively.

Preservation and restoration of natural ecosystems and their processes is important to maintain the dynamic character which ultimately formed the giant sequoia-mixed conifer forests prior to intensive human occupaion (Parsons and Nichols 1985). In the Giant Forest, aesthetic and ecological goals need not conflict, but should seek to complement each other as much as possible. It can be achieved by utilizing the recommendations from recent aesthetic research in Sequoia National Park (Dawson and Greco 1987). Most importantly, management should seek to mitigate the effects of past fire suppression and mimic natural tire patterns while educating park visitors about tire ecology.

Historically, the giant sequoia-mixed conifer ecosystem experienced frequent, low intensity fires (Kilgore 1987) which structured the forest prior to human interference. The effects of past management actions in suppressing all natural lightning fires, for some seventy-five years (possibly representing many natural cycles), has resulted in an altered forest structure and high ground fuel accumulation in many areas. The forest structure has been changed to favor shade tolerant fir and incense cedar (Harvey 1985; Kilgore 1985; Bonnicksen and Stone 1982; Bonnicksen 1975) while unnaturally high fuel accumulation risks increased mortality of giant sequoias and understory species during a fire. Past prescribed fires have resulted in what many environmental groups see as unnatural due to inadequate mitigative measures and procedures. Prescription fires are now designed to mitigate these effects through "cool bums" meant to restore natural conditions. The overall concern in SMAs is to have the forest "look" like a low intensity natural bum has moved through the forest even though the fuel load may have the potential for a high intensity fire; and environmental degredation through intensive use may not have the potential for recovery without active mitigation.

RESEARCH PROCEDURES AND METHODOLOGY

The procedures applied in this research were determined by the specific needs of management and recommendations from the Panel Report (Christensen and others 1987). They are (1) to delineate the viewshed boundaries of the SMAs, (2) to inventory and conduct an analysis on the visual resources within the SMAs, (3) to recommend ecologically acceptable visual resource management goals and objectives, and (4) to recommend management treatments to fulfill the visual quality goals and objectives.

The research consisted of an inventory of visual resource elements, formulation of goals and objectives, and development of a set of guidelines for the treatment of fire effects on the character of the landscape and on the character of individual giant sequoia features. The methodology developed for assessing the visual resources at Sequoia National Park can be applied to all roadways and trails within the park. The process model (fig. 1) graphically depicts the recommended methodology for SMA visual resource planning.

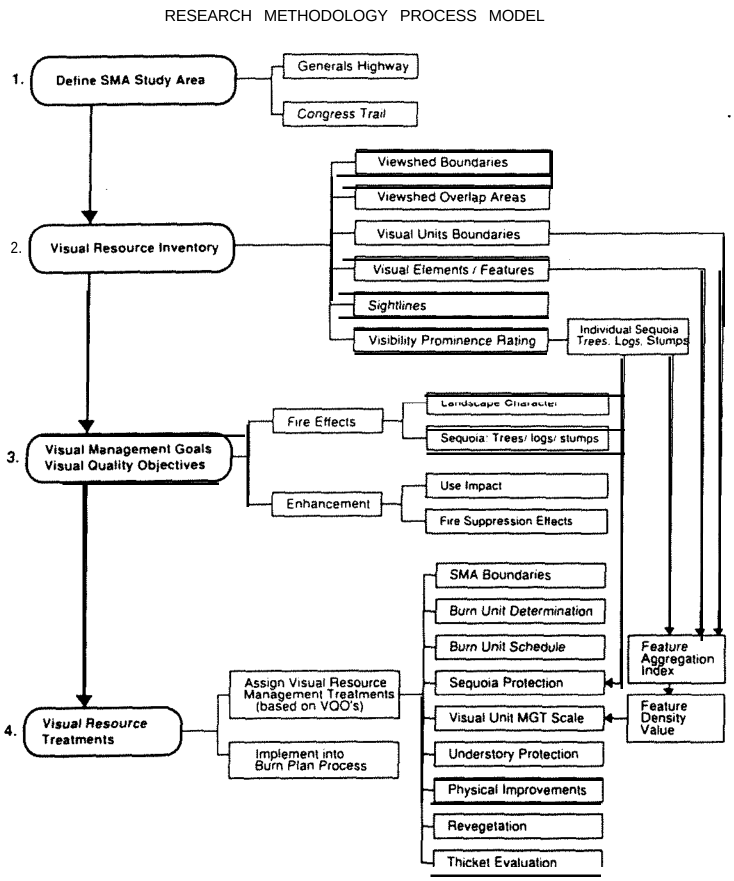

Figure 1 .--Visual resource research methodology and planning approach.

SMA BOUNDARY DELINEATION

The study areas within the SMAs are defined in terms of their respective viewshed boundaries. A viewshed, or visual corridor, is a routed (by road or trail), physically bounded area of landscape that is visible to an observer (Litton 1979). A viewshed delineates the dimensions of the "seen" environment in terms of visual penetration. The viewshed boundary is formed from the dynamic composition of viewing points on a continuum (i.e. a road or trail). The viewing points are representative of a number of observer positions accounting for several viewing orientations (Litton 1973, 1968).

VISUAL RESOURCES INVENTORY AND ANALYSIS

An inventory of visual resources is a descriptive field survey that identifies the seen areas, and physically locates visual and perceptual elements within the selected SMA study areas. It consists of several parts including viewshed delineation, areas of viewshed overlap, visual unit delineation, identification of special features and visual element subunits, determination of giant sequoia visibility through a visual prominence rating, and the location of impacted views due to fire suppression. An inventory was surveyed and compiled for each study area SMA.

The goal of the feature analysis is to provide park managers with a tool to assess the relative difficulty of achieving the visual quality objectives. The Management Scale provides an indexed classification for each visual unit to indicate pre-bum planning intensity and (bum) labor requirements that will be necessary for any given bum unit. For example, in an area with many visual features (i.e., giant sequoias, logs, etc.) the Management Scale value could be rated as class "1" and an area with few visual features could rate as a class "4" value. Hence, if a bum unit contains several class "1" values, then more labor will be required to mitigate excessive fire effects. Formulation of the Visual Unit Management Scale consisted of five steps: a tabulation of features per visual unit; a feature aggregation index calculation; determination of visual unit acreage; a feature density value calculation; and an indexed classification of those values into the Visual Unit Management Scale values.

SMA VISUAL MANAGEMENT GOALS AND VISUAL QUALITY OBJECTIVES

Fire management planning in SMAs requires the development of clear goals and specific objectives as a critical step in the prescribed fire planning process (Fischer 1985; Bancroft and others 1983). Clear exposition of goals and objectives is necessary to evaluate the effectiveness of management actions. Management goals should be broad in scope and attainable through specific objectives that address issues within each goal. The three central issues for visual quality goals and objectives are (1) fire effects on the character of the landscape, (2) fire effects on individual giant sequoias, and (3) enhancement of currently affected visual resources.

Fire Effects on Landscape Character

The giant sequoia-mixed conifer forests have evolved in context of frequent fire return intervals and low fire intensities (Kilgore 1987; Van Wagtendonk 1985). Less frequent, more extensive and intense events, though, have also played an important role in this ecosystem. Kilgore and Taylor (1979) found through tree ring analysis that historical fires near the Giant Forest area were frequently small in size and generally confined to a single slope or drainage. They also report that fires ranged in size between 0.001 ha to 16 ha. In the same study area, Harvey and others (1980) confirm the small nature of these bums, suggesting they were about 10 ha.

In the Redwood Mountain area, the Kilgore and Taylor study (1979) also found tire return intervals on west-facing slopes to be about every 9 years, and on east-facing slopes to be about every 16 years. They also report mean fire-free intervals of 5 years on dry ridges of ponderosa pine and 15-18 years in moist sites of white fir. The average maximum fire-free interval was found to be 14-28 years. Their data also reveals that some clusters of giant sequoias have escaped fire for up to 39 years. Some areas may have escaped fire for a hundred or more years.

Restorative SMA prescription fires should be planned within an appropriate temporal and spatial framework. The juxtaposition of prescribed bums can greatly enhance or detract from the visual and ecological diversity of the forest. The goal should not be to create bums that result in large scale areas of an early successional stage. Rather, management bums should concentrate on maintaining, or creating, successional diversity throughout the forest (Harvey and others 1980). Fire should be introduced on a gradual spatial and temporal basis to restore the forest to a more natural state. Although reducing fuel accumulations is important, it is not necessary that this be the immediate objective of an SMA bum. Small-scale bums should be designed to maintain ecological and visual diversity over appropriate time scales. Planning should incorporate available site-specific fire history research.

To preserve successional and visual diversity, management plans should include small-scale bums, random juxtaposition of bums (a variety of bum contrasts), selected retention of understory vegetation, and limiting the number of bum units treated each year. Planned variation in future bum unit boundaries will also help maintain an ecologically and visually diverse park environment. To increase visual diversity and maintain a sense of ecological continuity along travel corridors, bum unit boundaries should cross roads and trails in some areas and remain adjacent to them in others. If roads and trails are always used as boundaries, one side will always appear different than the other. Human infrastructure should be avoided or limited as bum unit determinants. Because it could lead to a confused perception of the forest to some visitors and contribute to a less naturalistic aesthetic. Extended long-range plans, or areas in need of a second prescribed bum, should include variation in the boundaries of the first prescribed bum, or possibly the relocation of trails during this planning process. It is not recommended that the same boundaries be used for future bums. The return of fire should also be variable, both spatially and temporally. Variation is another very important aspect of visual and ecological diversity, as pointed out in the Christensen Report (1987).

Treatments of designated SMA bum units should be "cooler" prescriptions as noted in the Grant Tree SMA plan (NPS 1980a). Taylor and Daniel (1985) confirm that fire intensity correlates with scenic quality and recreational acceptability in ponderosa pine forests. They found that in comparison to unburned areas, low intensity fires produced improved scenic quality ratings after 3-5 years, but that high intensity tires "seriously declined" in scenic quality ratings after the same time period.

Efforts to provide a high value interpretive program are essential to educate the public about tire ecology and the aesthetic implications of fire ecology in the Giant Forest SMAs. The program is important because visitors are barraged with fire danger signs as they approach the park. McCool and Stankey (1986) found that visitors who were confused and uncertain about the effects of prescribed tire were afraid that it could be "detrimental" and negatively impact the park, but that visitor center exhibits and guided tours help engender an understanding and appreciation of the dynamic processes of forest succession and fire ecology. Roadside and trailside interpretive displays in appropriate locations, with descriptive graphics facilitate this objective. The Hazelwood Nature Trail is an excellent example. Hammit (1979) indicated that the value of interpretive displays located in visually preferred areas can be more rewarding and more likely remembered. Proper placement of displays in the environment appears to aid in the memory process of park visitors.

Fire Effects on Individual Giant Sequoia Trees, Logs and Stumps

Visual features in the Giant Forest are highlighted by the grandeur and presence of a high density of giant sequoias. As a result of this density and the park's design, visitor appreciation of the giant sequoias has rendered many of them as unique natural/cultural objects in the landscape. Hammit (1979) reports that the most remembered scenes by visitors are characterized by visually distinct features. It appears there is a strong correlation between familiarity and preference of scenery. Familiarity is highest in both most preferred and least preferred scenes, indicating that visitors are affected by both positive and negative features observed in landscape experiences.

Since the giant sequoias are a primary visual resource in the Giant Forest, the most visually prominent trees should receive the greatest scenic mitigative measures to retain a natural visual character following restoration bums. Maintaining high scenic and recreational values in the Giant Forest requires sensitive visual resource planning of fire effects and a strong interpretive program to effectively communicate fire ecology to the public. It was recommended that a management goal for the visual quality of distinct foreground features receive judicious burning around the bases of the SMA giant sequoias. The foreground trees have the dual distinction of being most impacted by intense human use and are also visually vulnerable.

Protecting all visible trees from intensive fire effects is not desirable. For visitors to gain a sense of appreciation for a wide range of tire effects, some of the less visibly prominent trees could provide an opportunity for such diversity. It is not intended that foreground trees should be protected at the expense of background giant sequoias. Rather, foreground sequoias should receive more sensitive treatment due to their proximity to high human use pressures and park infrastructure. Intense human use proximate to these trees has resulted in decreased duff cover, soil compaction, increased erosion, and lack of understory regeneration. Many of these trees are under unnatural stress. Background trees receive wilderness standards for giant sequoia management.

To gain better insight and understanding of visitor sensitivity to singeing and charring on highly visible giant sequoias, a special study would have to be conducted. A study has been completed (Quinn 1989) of visitor perceptions of recent prescribed fire management in Sequoia National Park and generally, visitors were not adverse toward fire scars. However, no research was conducted on reaction to singeing versus charring in recent bum units within the park.

The last issue regarding protection of individual giant sequoias is the maintenance of ecological and visual/cultural values associated with horizontal features in the forest landscape experience. The preservation of a select number of highly visible sequoia logs (in addition to named logs) along trails and roadways has been strongly recommended by some groups (Fontaine 1985). The interpretive value of these logs stems from the direct "involvement" the public has with these elements. The tactile experience of touching and passing under these logs can engender a strong appreciation for the grandeur of the giant sequoias. They also demonstrate the dynamic nature of succession in the giant sequoia-mixed conifer ecosystem. Hammit (1979) suggests that prolonged contact with such features increases familiarity. It was recommended that a balanced number of strategically located logs be protected from intense prescribed bums.

Currently Affected Visual Resources

Scenic resources are currently impacted by (1) intensive recreational use, and (2) the structural changes of vegetation in the giant sequoia-mixed conifer forest. The first is due to the effects of visitor overuse and the lack of facilities to accommodate the use volume. The second impact results from tire suppression which promotes the growth of shade tolerant conifer thickets (non-fire climax species) that limit the visibility of numerous giant sequoias within the viewshed. Management goals to alleviate both of these impacts would enhance the overall experience of the park.

Many high visitation areas such as the Congress Trail, General Sherman Tree, and Hazelwood Nature Trail suffer from severe overuse. Strategic signage in these areas is essential to better guide foot traffic (trampling) in these areas which has caused the disintegration of duff and subsequent erosion of surface soil. As a result, dusty or muddy visitor environments have inadvertently created biological and visual resource problems. Problems include erosion around the bases of sequoias exposing fibrous roots, erosion and decay of asphalted edges in parking areas and on trails, and a lack of understory vegetative *cover* due to trampling and soil compaction. Means to reduce these effects focus primarily on redirecting foot traffic in and around facilities and reducing trampling around the trees.

The second issue concerning enhancement of affected visual resources is the extensive growth of shade tolerant conifer thickets (non-tire climax species) resulting from tire suppression and disturbances due to road, trail, and facility construction (NPS 1980b; Bonnicksen 1985). In the absence of regular fire disturbance cycles, these thickets have *grown* unchecked by natural process, thus hindering the ability of the giant sequoia to reproduce successfully and also blocking both historic views and potentially valuable views of the giant sequoias in the Giant Forest SMAs. In addition to these problems, the thickets also represent future fuel load and fuel ladder problems. The visual resource goal should be to conserve scenery which enhances visitor experience within the SMAs through active management of the thickets. The means to achieve this goal is the limited strategic removal of these "overrepresented aggregation types" (Bonnicksen 1985; Cotton and McBride 1987).

VISUAL RESOURCE TREATMENTS

The recommended treatments consists of a Landscape Management Plan and a set of guidelines for visuat resource management in the SMAs. Visual resource treatments are management actions designed to fulfill management goals and visual quality objectives. A photographic monitoring program is also recommended.

Landscape Management Plan

The SMA Landscape Management Plan identifies proposed bum units, planning units, past prescribed bums, bum exclusion areas and thicket problem areas. The bum units have been designed in accordance with the visual quality objectives to maintain a diverse visual character within the SMA study areas. Sections requiring additional research studies are classified as "planning units" and "SMA planning units" on the plan. Small areas of cultural value that are recommended for exclusion from prescribed fire are also indicated on the plan. Additionally, thickets that block views of giant sequoias, and thickets that present future visual resource problems are identified for treatment. Finally, measures to protect visually prominent giant sequoias are based upon the visual prominence ratings are shown on the Visual Resource Inventory maps.

Protection of visual elements is also meant to preserve pockets of mature understory vegetation in addition to giant sequoia protection. These pockets are ecologically important because intensive human use interferes with regeneration and colonization sources which are needed to avoid further damage and are needed as vegetative use buffers. These, too, are identified on the Visual Resource Inventory Maps. The analysis of visual features within the visual units provides a guide for resource managers to evaluate planning for labor requirements when planning bum units. A feature "density" value was generated for each visual unit and broken down into management intensity classes.

Bum Unit Design and Schedule

Bum units *were* designed based on the Fire Effects Guidelines for SMA Landscape Character. Natural boundaries for the SMA bum units are preferred to man-made boundaries in the design. It is recognized that it is essential to use roads and hiking trails in many cases due to economic constraints. However, alternatives to their use should be used where Possible, such as streams, drainages, ridges, old fire lines, meadows, rock outcrops, and new fire limes.

The bum units in a maintenance fire regime should be varied from previous prescribed bums. It is not recommended that the same bum unit boundaries be used more than once if they are unnatural boundaries (trails or roads). Using the same boundaries runs an ecological and visual risk of creating an unnatural mosaic of forest succession. The maintenance bum *regime* units should concentrate on natural fire breaks that travel across trails instead of being bound by them.

Timing of the bum units is a very important aspect of planning. The bum units have been designed to restore the Congress Trail and the SMA section of the Generals Highway to more natural conditions. Following the restoration bum regime, a long-term maintenance fire regime should be formulated for the Giant Forest. It is recommended that this regime be based on area-specific fire history research.

A computer geographic information system (GIS) would greatly enhance the analysis and planning of the bum units in the Giant Forest because it is a very useful tool for evaluating large spatial data sets and many variables.

Guidelines for Thicket Problem Areas

The visual quality objectives regarding enhancement are designed to increase the visibility of giant sequoias affected by extensive thicket growth throughout SMA viewsheds. These thickets are blocking numerous potentially valuable views of giant sequoias (fig. 2). Management for a natural aesthetic and increased visual penetration into the forest within the SMAs warrants judicious mechanical thinning of some of these thickets (Bonnicksen and Stone 1982; Christensen 1987; Cotton and McBride 1987).

Figure 2.--Thickets of mixed conifers are encroaching on the views of giant sequoias due to the disturbance of road construction.

The thickets were mapped on the SMA Landscape Management Plan in two ways. Existing "blocked" views were mapped, and visually "encroaching" thickets are also shown. The encroaching thickets did not present a visual problem at the time the field work was conducted, but will cause visual penetration problems in the near future. They should be monitored photographically and evaluated for mechanical thinning. It was recommended that this be incorporated into the park's Vegetation Management Plan for the development zone (NPS 1987b).

Guidelines for Giant Sequoia Fire Effects Mitigation

As discussed in the visual quality objectives, it is the visually prominent trees which are impacted most by human use pressures. Park infrastructure, such as trails, roads, signs, restrooms, etc., are proximate to the visually prominent trees. The most valuable scenic resources are also the most visually prominent trees. Mitigative measures to protect these trees are critical in terms of ecological, scenic, and park infrastructure resources The objective is not to leave these trees unburned, but to mitigate fire effects. Trees impacted by intensive human use are under stress and unsuppressed fire risks unnatural mortality. The four categories of giant sequoia protection (mitigation measures) are illustrated in figure 3 and include: (1) scorch exclusion, (2) minimal scorch, (3) limited scorch, and (4) unsuppressed scorch (within standard management tree protection guidelines). These relate directly to visual proximity as well as distance from human impact (Dawson and Greco 1987).

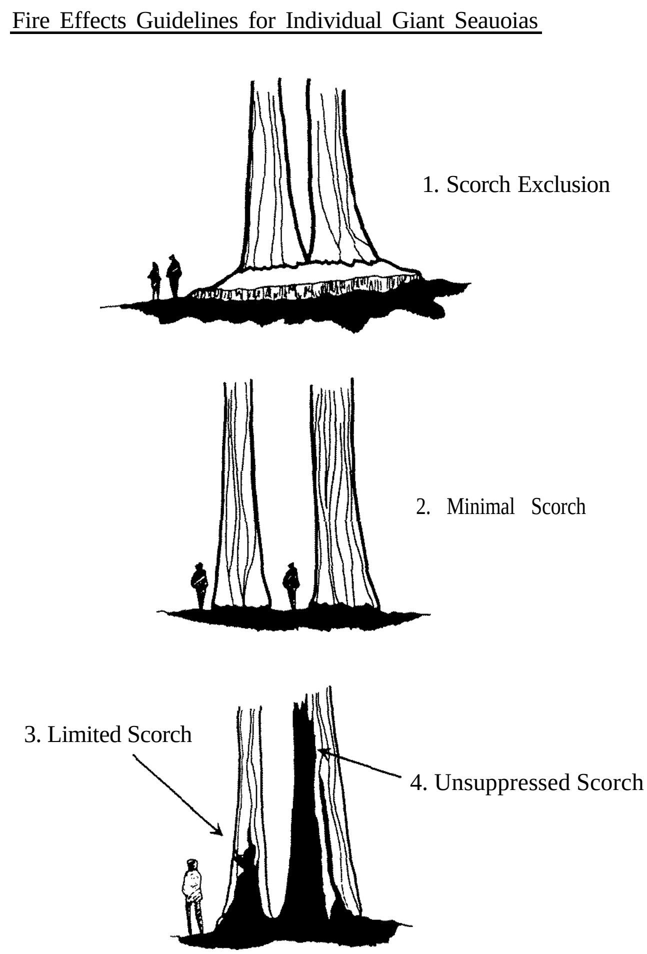

Figure 3.--SMA mitigation measures for giant sequoias

To understand properly the descriptions of the four categories of giant sequoia protection, definitions of scorching, singeing and charring are necessary. In this study, "scorching" is the singeing or charring of sequoia bark. "Singeing" is bark ignition to a depth under one half an inch (< 1/2"). "Charring" is defined as bark ignition to a depth over one half an inch (> 112"). The question of singeing is not an intense aesthetic issue because park visitors seem to accept some fire damage to sequoias (Quinn 1989). However, reaction to varying levels of charring is undetermined and can impair the scenic quality of giant sequoias for longer time periods if the trees are under stress. Therefore, it was recommended that scorch and char guidelines be established in addition to current tree preparation standards (pre-fire) and firing techniques. It should be remembered that the guidelines apply only during the restoration prescribed fire phase.

Guidelines for Understory Protection

Planned retention of understory vegetation pockets is recommended in the SMA bum units. They offer opportunities to maintain visual and ecological diversity while increasing the probability of regeneration by providing colonization sources. Often, these pockets grow among rock outcrops and probably have escaped fire for longer periods under more natural ecosystem conditions. Historically, natural bums have undoubtedly missed many areas creating a mosaic of vegetation characteristic of the sequoia-mixed conifer ecosystem. The most obvious pockets for retention would be growing among rocks that could be supplemented with fire lines to lengthen their presence.

For aesthetics, these groups of plants provide a visual focus, diversity of elements, and demonstrate the scale between visitors and the large-scale giant sequoias and older conifers. Some good examples in Giant Forest are the native dogwoods (*Cornus nuttallii*) and Sierra chinquapin (*Castanopsis sempervirens*), and greenleaf manzanita (*Arctostaphylos patula*). Although some are adapted to fire and resprout after a fire, their rate of growth is slow. Their visual qualities and interpretive qualities could be diminished for many years.

DISCUSSION

There has been concern on the part of National Park Service scientists about some of the research recommendations on visual resources (Dawson and Greco 1987). An interdisciplinary group of staff from Sequoia National Park representing science, administrative management, visitor interpretation, fire management, and resource management met and forwarded comments. The following discussion presents these views as well as further discussion on the visual resource research.

NPS and Understory Issues

The NPS group does not favor "the deliberate retention of mature groups of understory plants, since prescribed fire tends to leave mosaics of burned and unburned areas, and the recovery of the understory plants in post-tire succession is an important part of the story of the forest" (NPS 1988).

At several prescribed bums in the Giant Forest, the visual resource research team observed that fire was applied homogeneously within the bum units. Fire management staff frequently bum areas completely and uniformly, and if fue bypassed any fuel loads, the fire technicians returned moments later to fire that area. This does not mimic natural fire patterns and as a result, pockets of understory plants rarely survive. The practice of multiple-spot firing after the fire has moved through should be modified to rely on this technique only in situations where absolutely necessary (greater than 1000-hour class fuels). Kilgore (1985) supported this concept by pointing out that increased uniformity and lessened mosaic pattern is unnatural.

Litton (1988) has written to Sequoia National Park that "In addition to modifying fuel concentrations, both down material and standing live trees, related to dominant specimens, I further urge protective measure for certain visually significant understory - ground floor components. Several obvious examples of these subordinate features are snags, fallen big trees and mature, tree-form dogwoods; these and others contribute significantly to experiencing a rich landscape, are signs of time and succession, and represent considerably more than fuel needing to be burned. "

Litton further added, "Brewer, King, and Muir confirm and give emphasis to other contemporary accounts that the Sierra Nevada forest were [sic] impressive for their [sic] openness and for the large scale of mature trees. At the same time, these three early observers note the diversity of what they saw in the various forest and woodland species, their associations, regeneration and some of the ground plane and understory characteristics. Brewer notes **species** or type distribution in space and elevation, the combinations of the mixed conifers - some with Big Trees, the array of ages and sizes in Big Trees, [and] the significance of fallen Big Trees in appreciating their size and age. King emphasizes the impact of contrasts found in the association of Big Trees and Sugar Pine and White Fir as well as the experience of the spatial quality found in the open forest. Muir comments on openness, on spatial distribution, on the smooth floor, but also points to the contrast of underbrush with Big Tree bark and speaks in considerable detail about Big Tree regeneration. Diversity, then, appears to be an historic clue about the historic forest in addition to the frequently stated perception of openness."

NPS and Visibility Issues

The NPS group "was unanimously opposed to allowing changes in appearance due to fire only in the medium and low visibility trees, while retaining foreground trees in their present unburned state... in general, all trees regardless of [visibility] rating will be prepared and burned according to current standards..." (NPS 1988).

In the visual resource recommendations, scorch exclusion does not mean "unburned". More importantly, it will be very difficult to treat focal point trees, such as the General Sherman Tree, with prescribed fire. These trces are surrounded by trails, fences, facilities, and/or roads and are also subject to intensive visitor use and abuse. Most foreground trees in special management areas are stressed by pavement, soil compaction and altered topography. As one moves farther from view corridors, this type of impact (direct human disturbance) is lessened. It is evident that there is an ecological relationship between aesthetics and the built environment and treating giant sequoias in the foreground more sensitively than those further away actually rccognizes the impact of these conditions.

NPS and Downed Log Issues

The NPS Group agreed "that logs identified by inlerpretation as having cultural or interpretive value will be protected from fire. However, no effort should be made to preserve logs as horizontal elements, since these logs are important sources for seedbeds, which are an important part of the forest story. In addition, the SMA bum units are small, and it is not likely the loss of logs will produce an impact on the visual resources of the area as a whole" (NPS 1988).

The Yellowstone fires document that horizontal elements (logs) are increased by fire, not decreased, regardless of fire intensity (Ekey 1989; Guth 1989; Simpson 1989). Although it is difficult to compare Yellowstone and Sequoia, logs are universally important ecologically and visually for the maintenance of habitat diversity. It is important to avoid the homogeneous bum coverage typical of hot fires in unnatural fuel accumulations. While totally burnt logs can play a role in sequoia regeneration, firing techniques which attempt to bum all logs does not recognize that some logs also play an important rote in the nutrient cycling of the forest by acting as nutrient reservoirs and reducing soil erosion following a fire. If the fire bums a log as it moves through, this seems acceptable. The problem is when fire crews return to spot-bum a log that the fire has by-passed.

NPS and Thinning Issues

The NPS group "agreed that existing vistas of the Sherman, Grant, and McKinley trees should be preserved. The group was opposed to pre-bum thinning of trees which obstruct sequoias as well as to the suggestion that trees killed by the fire should be cut out" (NPS 1988).

In discussing visual resources, the thickets are diminishing the scenic value of the park from roads and trails. Many of these thickets are less than fifty years old and exist as a result of managed fire exclusion and site disturbance, such as road construction. This abundant growth impacts scenic resources and ecological processes. Kilgore (1987) states that "removing fuel from the intermediate layer between between surface and crown fuels greatly reduces the potential for high intensity surface fires that could lead to crown fires." Under a more natural fire cycle, crown fires are a relatively rare event in the giant sequoia-mixed conifer ecosystem and would be an unnatural and unfortunate consequence of the fuel load build-up due to past fire suppression.

The Christensen Report (1987) indicates approval of 'judicious pre-bum cutting of understory trees...where ignition of such trees might have a negative effect on stand appearance and/or when their removal would enhance the visual effect of adjacent specimen trees."

CONCLUSION

Past human interference with the ecosystem of the giant sequoia-mixed conifer forests has impacted the visual and ecological resources in Sequoia National Park. These impacts have been augmented by concentrated visitor pressure in the areas of the park with roads, trails, and built facilities. Special management areas have been established to address these complex management problems of balancing cultural and natural ecosystem interests.

The detailed visual resource database and mitigation guidelines developed for the Prescribed Fire Management Program were designed to provide park resource managers with new tools to achieve more natural fire effects for the landscape and giant sequoia visual resources. There were forty-four separate treatments recommended with roughly half of the recommendations ***known*** to be implemented (Dawson and Greco 1987). It is pleasing and appreciated that support was so forthcoming from the National Park Service for over half of the treatments. This paper has attempted to explore the complexities of the remainder. However, creating favorable conditions for the perpetuation of the giant sequoia is supported and current management policies using prescribed fire management are improving continuously. The visual resource research has strived to present ecologically acceptable solutions to problems of culture in the context of a natural environment and the role of fire in the giant sequoia-mixed conifer ecosystem which support this continued improvement.

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GIS APPLICATIONS TO THE INDIRECT EFFECTS OF FOREST FIRES IN MOUNTAINOUS TERRAIN

David R. Butler, Stephen J. Walsh, and George P. Malanson[1]

Abstract-Snow-avalanche paths and landslides are common geomorphic features in Glacier National Park (GNP), Montana, and represent hazards to human occupancy and utilization of the park. Forest fires have been spatially extensive there, and it is well documented that areas subjected to forest fires become increasingly susceptible to avalanching and landsliding.

The locations of all snow avalanche paths and landslides in east- central GNP have been mapped on topographic maps and verified. The avalanche paths have been digitized and entered into a geographic information system (GIS) using ARC/INFO. Digital elevation models and Landsat Thematic Mapper digital data were processed to create elevation, slope angle and aspect, and landcover GIS overlays. Merging of overlays illustrates areas of maximum erosion potential by snow avalanching and by landsliding in the event of a forest fire. Post-fire vegetational succession can be accommodated into the GIS to illustrate areas of high, medium, and low hazard from avalanching and landsliding.

INTRODUCTION

The western cordillera of North America experiences hundreds of thousands of snow avalanches and numerous landslides annually. Most snow avalanches follow well defined topographic indentations on the mountainous slopes (Butler, 1989). These snow-avalanche paths (fig. 1) are easily mapped at a variety of scales, so that hazard zones resulting from snow avalanching may be easily delineated (Butler 1979, 1986b, 1989; Butler and Malanson 1985; Walsh and others 1989). Mass movements of earth and rock material, or landslides for the sake of convenience, are also common in the cordillera. Steep terrain, seismic triggers, and unusual precipitation and snowmelt events produce widespread landsliding in the area (Butler and others 1986).

It has been well documented that forest fires geomorphically destabilize a burned area, making it more susceptible to erosion by both snow avalanching (Beals 1910; Munger 1911; Winterbottom 1974; Harris 1986) and landsliding (Swanson 1981; Morris and Moses 1987; Parrett 1987). The removal of forest cover particularly affects areas prone to snow avalanching. Most starting zones for snow avalanches are on fairly steep slopes of 30-45° (fig. 1). The forest cover in this environment provides a significant stabilizing influence on the snowpack, reducing the avalanche hazard. If a forest fire removes this stabilizing influence (fig. 2). expansion of the area of snow movement is likely to occur. This in turn can provide more frequent and larger, and therefore more dangerous, snow avalanches on the low- angle slopes near valley bottoms where roads, railroads, tourist facilities, and communities are likely to be concentrated (Munger 1911; Winterbottom 1974).

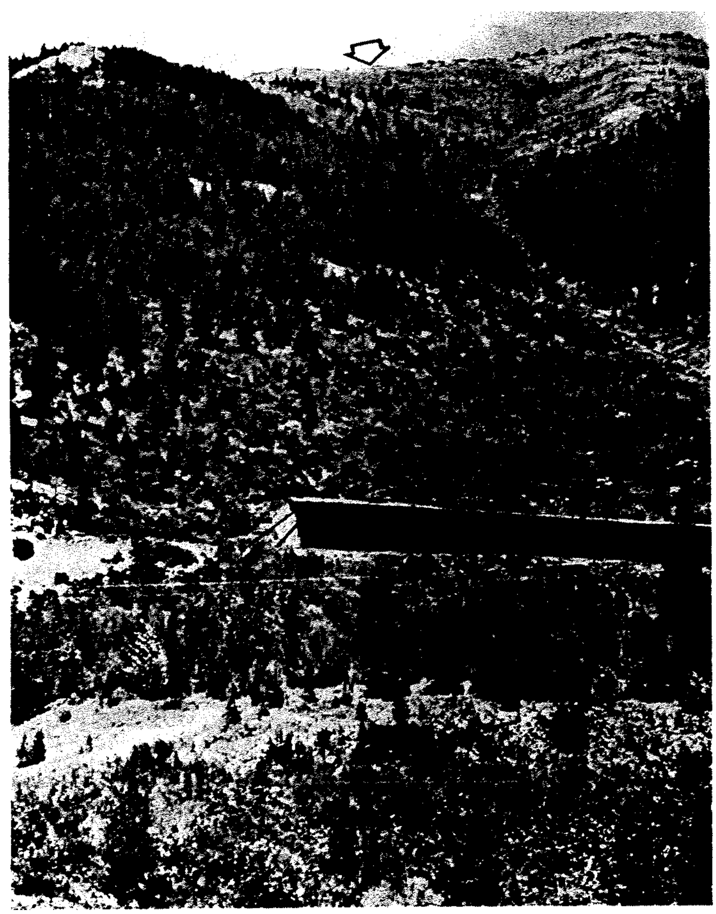

Figure 1. Typical snow avalanche path, southern Glacier National Park, Montana. Arrow points to location of figure 2. Note how lateral boundary of avalanche path exceeds the protective capacity of the snowshed, a result of destabilizing forest fires during 1910-1919. Photo by D.R. Butler.

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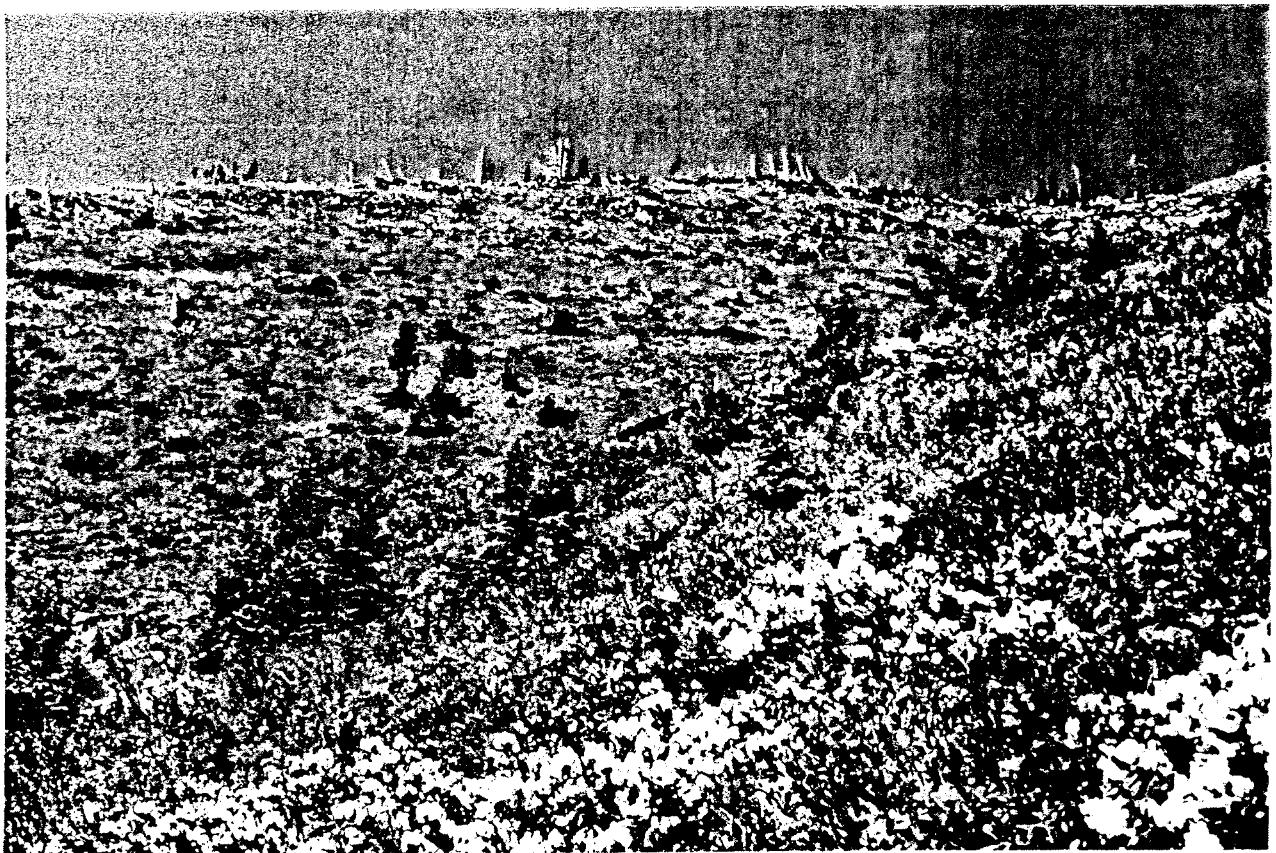


Figure 2. Burned-over starting zone of the Shed Seven avalanche path shown in figure 1. Dead snags are especially visible along the skyline. Photo by D.R. Butler.

Areas already susceptible to landsliding are also likely, in the event of forest fire, to experience reactivation of previously stabilized landslide deposits (Swanson 1981), and accelerated erosion on the surface of these deposits by running water will also occur (Morris and Moses 1987). Burned areas will also generate landslides whereas adjacent unburned areas do not (Parrett 1987). It is, therefore, of paramount importance in mountainous areas where tourism and multiple-use forestry form the economic base, to know where expansion of snow-avalanche paths, reactivation of landslides, and accelerated erosion is likely to occur in a post-bum scenario.

This study describes how a Geographic Information System (GIS) may be used to map and study snow-avalanche paths and landslides, and in turn how information on areas and year of burning by forest fires and the level of plant revegetation can be incorporated into the GIS. This allows the delineation of areas of potential expansion of snow avalanching and landsliding, as well as areas affected by less hazardous but geomorphically and environmentally significant accelerated surface erosion. This information can be used by forest and park management personnel who need to critically examine areas of sensitive habitat, or who may be in charge of hazard analysis in areas of heavy transportation and tourism. In addition, such information can be used to evaluate sediment movement and concentrations within hydrologic systems as a consequence of forest fires and snow avalanching or landsliding and their spatial/temporal distributions.

THE STUDY AREA

Snow avalanching and landsliding are common geomorphic occurrences in the Rocky Mountains of northwestern Montana. Forest fires of varying intensity and extent have burned broad areas susceptible to both avalanches and landslides. One area particularly susceptible to both avalanching and landsliding, a result of a set of unique topographic and geologic conditions, is Glacier National Park, Montana (Butler 1979; Butler and others 1986). This park, created by act of Congress in 1910, preserves approximately one million acres of wilderness which has never been logged.

Glacier National Park contains a mosaic of vegetational types dependent on such factors as elevation, slope aspect, position west or east of the Continental Divide which bisects the park, and fire history. Many historical tires have burned portions of the park before and since 1910 (see, for example, Beals 1910; McLaughlin 1978; Holterman 1985; Finklin 1986; Larson 1987). Until recently, it has been the policy of Glacier Park management to vigorously suppress all forest fires, whether natural or human- caused (Wakimoto 1984). The 1980s saw a shift in policy, with movement toward a management plan that would allow natural fire to play a role in the park ecosystem in designated areas.

Along the southern boundary of Glacier Park, several widespread forest fires occurred during the period 1910-1919 (Payne 1919). There, numerous snow-avalanche paths impinge onto the tracks of the Burlington Northern Railroad,

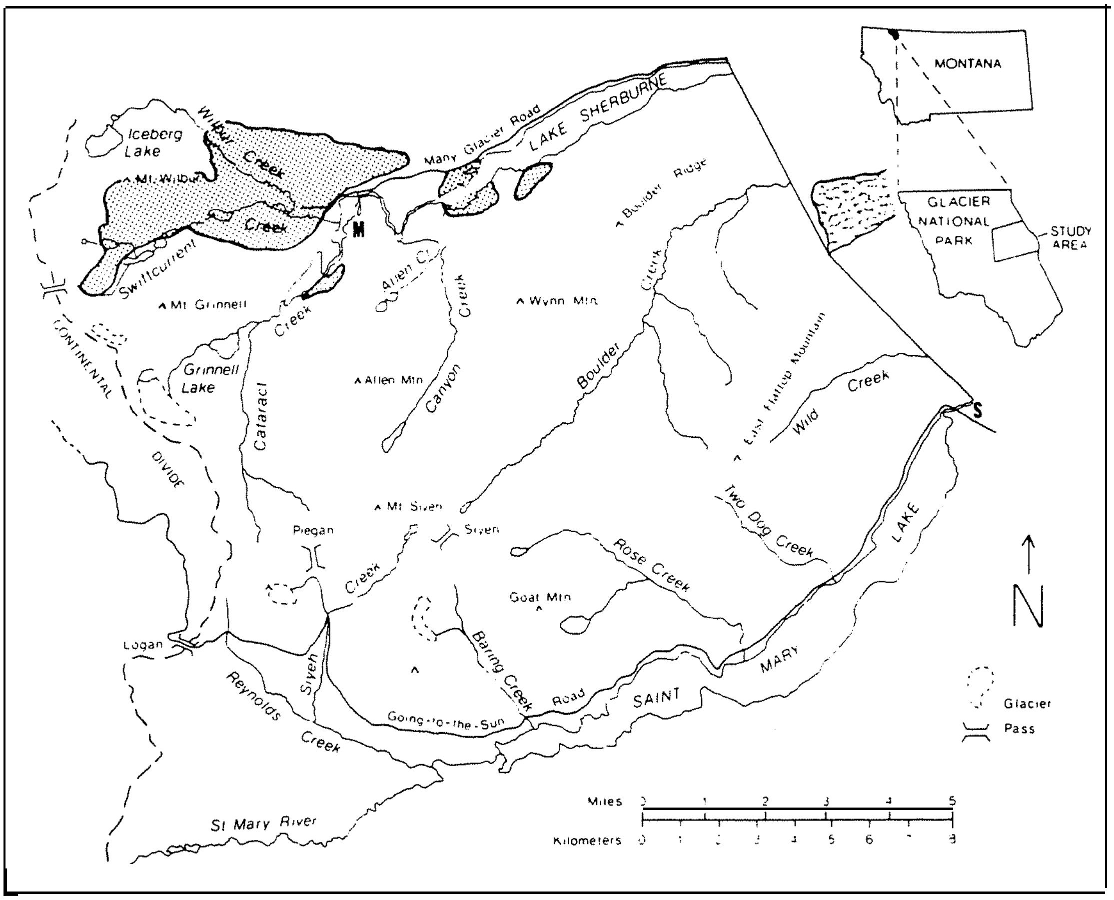


Figure 3. Specific study area for snow-avalanche terrain modeling. Dot pattern shows area of 1936 Swiftcurrent Valley tire which came from across the Continental Divide; stipled pattern shows the Napi Point fire of 1984 (fig. 5)

which form the park boundary. Snowsheds were built in 1910-1914 to protect the rail line from burial by avalanching; however, the burning of large areas of forest in and near avalanche starting zones (fig. 2) expanded the geographic extent of unstable snow so that avalanches cover broader swaths of railroad track than were originally covered with protective sheds (Butler and Malanson 1985).

For this study, we chose to examine a section of Glacier National Park east of the Continental Divide, which contains over 100 snow- avalanche paths and dozens of landslide deposits, and has been subjected to forest fires in 1936 and 1984 (fig. 3). The area chosen is one of the most heavily visited portions of the park, and has several roads and backcountry trails which allow access to field sites.

METHODS

Landslide locations (fig. 4), particularly with reference to landslides occurring in burned parts of the study area (fig. 5a, b), were mapped on the basis of aerial photointerpretation and fieldwork (Oelfke and Butler 1985). Landslide type and slope aspect have also been categorized for these deposits. No other data have yet been calculated for the landslide deposits in the study area. Preliminary examination of this mapping and categorization reveals that generally north-facing, and therefore moister, landslide deposits of the slump/earthflow variety would be most likely to be reactivated in case of forest fire. However, because this portion of the research is still continuing, we devote the remainder of the paper to the analysis of snow- avalanche path location.

Because avalanches tend to occur in spatially-distinct locations, we used a GIS to delineate path location and analyze the spatial characteristics of sites subject to avalanching. We wished to determine why snow-avalanche paths are located where they are in the study area, so that we could then develop a cartographic model which illustrates areas of highest probability for areas of new snow avalanching in the event of a forest fire removing the vegetational cover. It was therefore necessary to map the locations of all avalanche paths within the study area shown in figure 3.

Aerial photointerpretation and field reconnaissance confirmed the location of 121 snow-avalanche paths within the study area. Field work in 1987 and 1988 revealed that Little change had occurred in the outer boundaries, or numbers, of avalanche paths since 1966 when aerial photography was acquired. Minor extension of the longitudinal boundaries of some paths occurred as the result of a major high-magnitude avalanche episode in February, 1979 (Butler and Malanson 1985; Butler 1986a).

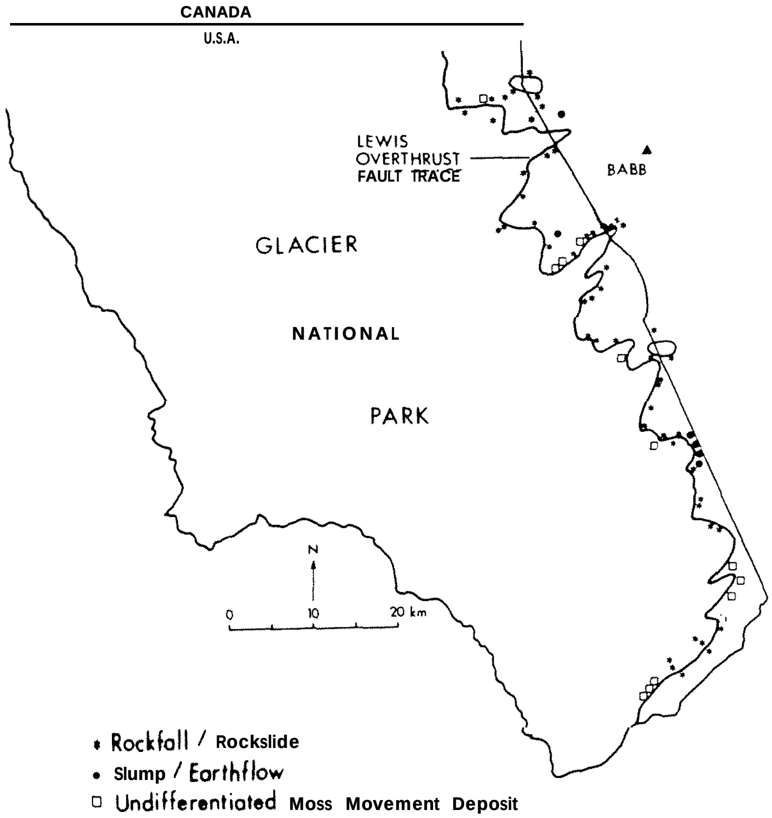


Figure 4. Landslide types and locations in eastern Glacier National Park. Compare to inset map, figure 3, for location of specific study area.

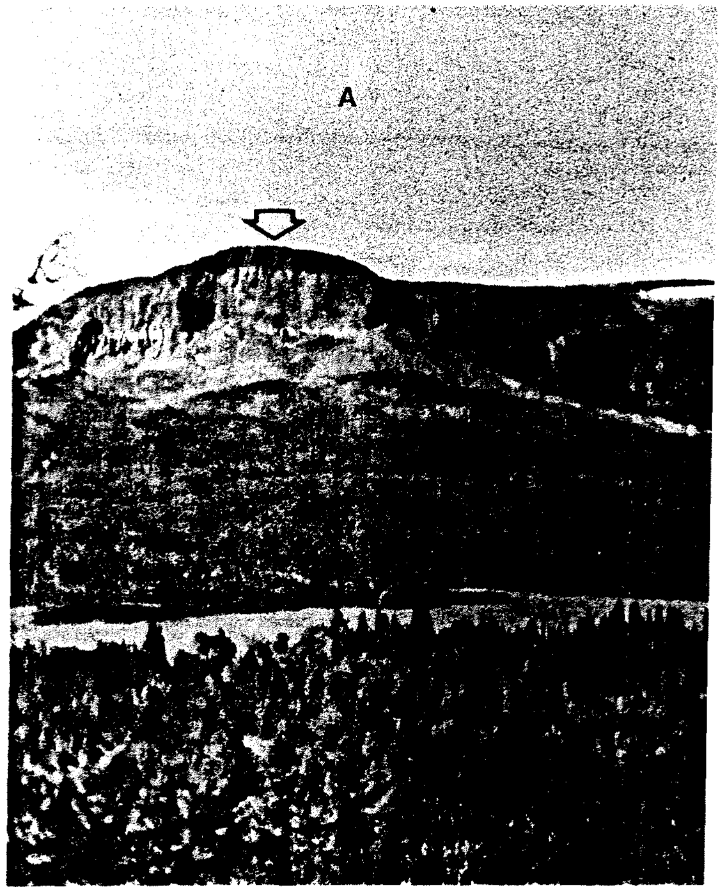
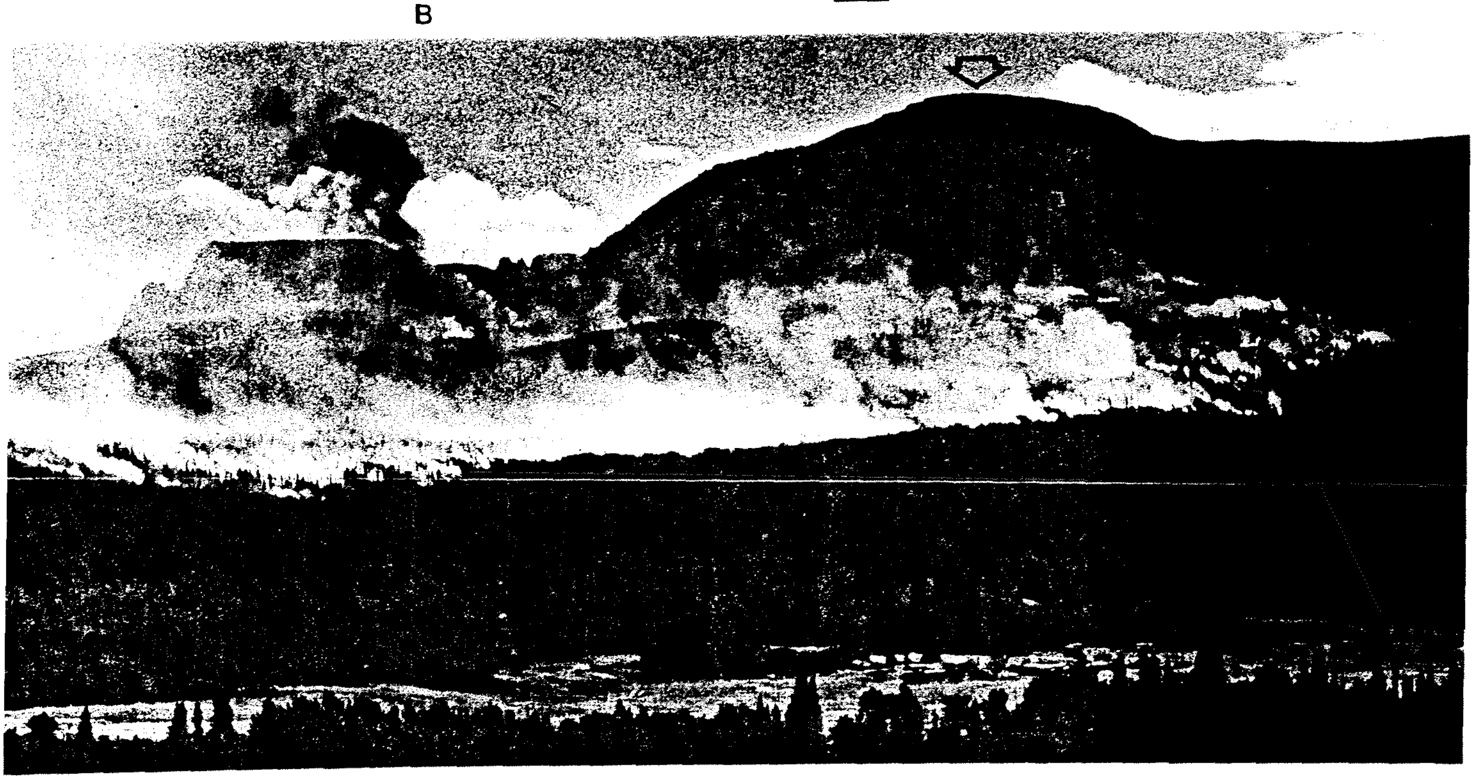


Figure 5. a. Napi Point (arrow), with rockfall avalanche and landslide deposits draping its north-facing base. Photo by D.R. Butler. b. Napi Point (arrow) fire of August, 1984, where forest fire burned and destabilized a broad area of landslide deposits. Photo by Brian Kennedy, Hungry Horse News, Columbia Falls, MT.

B



The location and areal extent of each snow-avalanche path were plotted on 1:24,000-scale topographic maps. Morphometric data were collected for each path from the topographic maps, aerial photographs, and field observations. The morphometric variables were entered into an INFO table within the ARC/INFO GIS for defining the character of each path from a geographic and geomorphic perspective. A GIS thematic overlay of path location was produced and merged with overlays developed for hydrography, geologic structure, lithology, topographic orientation, and land- cover type. Details of the development of these overlays may be found in Walsh and others (1990).

Most of the GIS overlays were compiled by direct digitization and transformation of mapped information for precise co-registration with other thematic overlays. Land-cover type and structural lineaments, however, were characterized through the digital analysis of Landsat Thematic Mapper (TM) data (fig. 6). Terrain orientation was characterized by a U.S. Geological Survey digital elevation model of the study area (fig. 7).

The land-cover GIS overlay was produced through an unsupervised classification of a 6 August 1988 TM scene (see Walsh and others, 1989, for details). Ground control information for approximately one-half of the avalanche paths was acquired during the summers of 1987 and 1988 to aid in cluster-labeling of the land-cover classification. Field data from other avalanche paths within the park (Malanson and Butler 1984a, 1984b, 1986; Butler 1985) revealed broadly similar vegetational types with similar spectral signatures. Cover-type classes used for this study were water, snow and ice, bare rock, lodgepole pine (Pinus contorta) forest, spruce/fir forest (primarily Picea engelmannii and Abies lasiocarpa), mixed herbaceous, and mixed shrubs.

Figure 6. Landsat TM principal components image of study area, with darker shades representing coniferous forest, lakes are black, alpine tundra is grey, and light tones representing fire successional lodgepole pine in Swiftcurrent Valley, and herbaceous plants and shrubs on avalanche paths and landslide deposits.

ELEVATION

CONTOUR INTERVAL
50m

SOURCE:
USGS 1:250,000 DEM

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Figure 7. Digital elevation model of the study area.

Spatial proximity to geologic structural elements and hydrologic features (also controlled largely by structural elements) on the landscape was an important influence on the geographic distribution of snow-avalanche paths within the study area (Butler and Walsh 1990). Measurement of the distance of paths from sills, dikes, faults and lineaments, and rivers and streams was carried out within the ARC/INFO environment through the generation of buffers. Buffers are spatial zones of user-defied diameter that indicate distance from a specified target phenomenon. Distance measures of each path to the selected landscape feature were calculated and added to the path morphometric database. A separate thematic overlay of buffers surrounding sills, dikes, faults and lineaments, and rivers and streams was added to the GIS for integration with the other coverages (fig. 8).

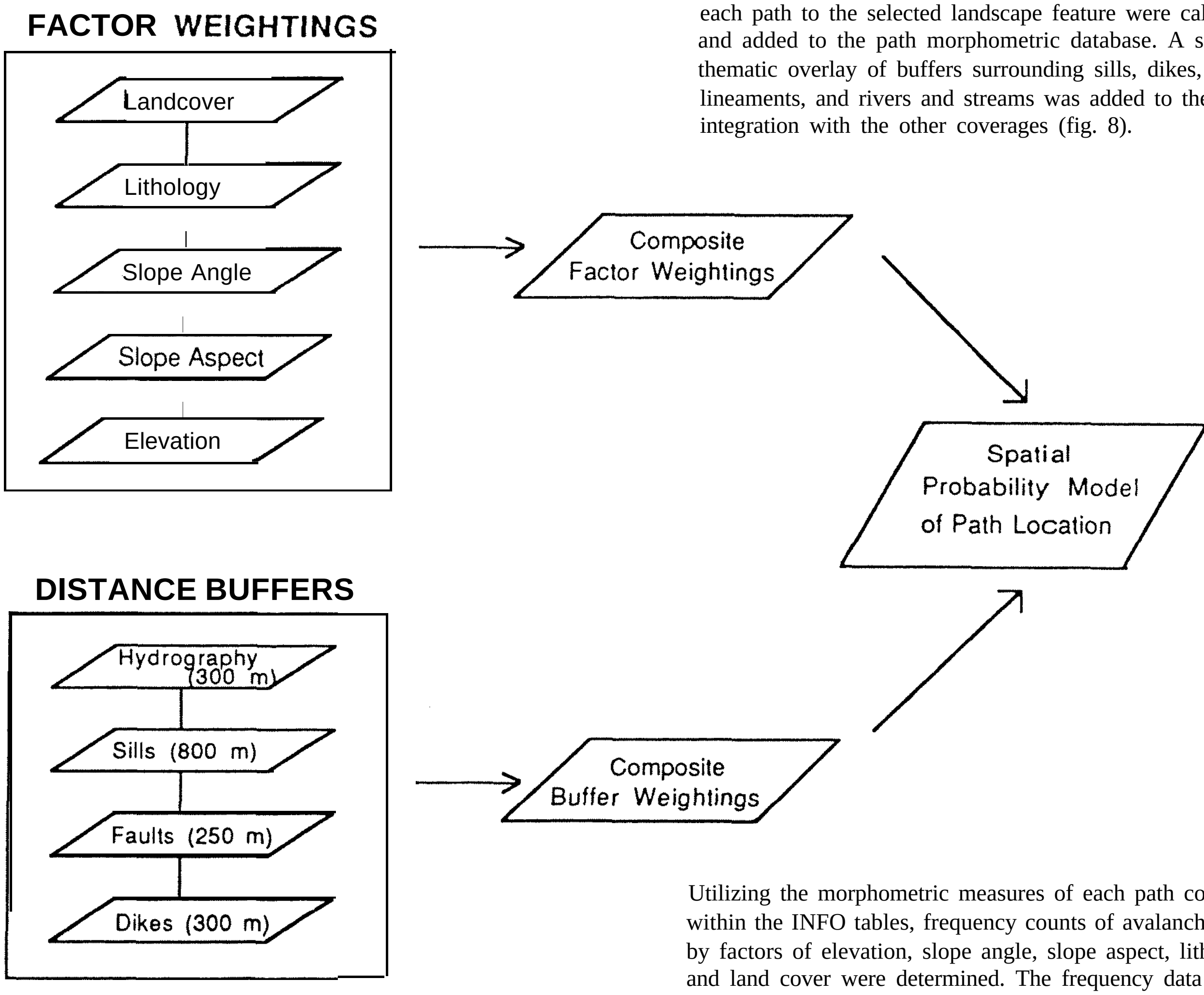

Figure 8. Schematic diagram of the units of the geographic information system used to model the spatial probability of snow avalanche paths in the study area.

Utilizing the morphometric measures of each path contained within the INFO tables, frequency counts of avalanche paths by factors of elevation, slope angle, slope aspect, lithology, and land cover were determined. The frequency data for each overlay were then normalized by weighted area measures (details of the weighting procedure may be found in Walsh and others 1990). Normalized frequency weightings also were derived for the proximal measures associated with spatial buffering. Buffers were weighted by the percent of all snow-avalanche paths occurring within the various buffer distances from the target feature.

RESULTS AND DJSCUSSION

The end result of the GIS analysis, shown in figure 8, was a spatial probability map illustrating where, based on the factors and buffers analyzed, snow avalanches are most likely to develop in the event of removal of forest vegetation by a forest fire (fig. 9). When a forest fire occurs within the study area, the boundaries of the fire can be rapidly digitized and entered into the GIS, and merged with the spatial probability data displayed in figure 9. Forest fire extent and plant regeneration can be assessed through high spatial, spectral, and temporal resolution satellites, such as Landsat and SPOT. Walsh and others (1981) reported on the role of Landsat satellite data for delineating and assessing forest disturbances and levels of forest regeneration. Remotely sensed measures of plant productivity with time can be assessed through use of vegetation indices and merged into the GIS as distinct multi-temporal landcover coverages.

Assessment of the information in the GIS follows. Did the fire bum an area that is likely to become more prone to snow avalanching? If the burned area coincides with terrain categorized as high probability, the answer is yes. Park managers can almost certainly expect currently-existing path margins to expand, as occurred in the southern portion of the park during the 1910-1919 period described earlier, and new paths will probably also develop in those areas marked on figure 9 as high probability areas. If the burned area coincides with the area of medium avalanche probability, expansion of areas of pre-existing avalanching may be likely, but it is questionable if ***new*** avalanche-prone areas will become established. Little concern for expansion of avalanching need be given if the fire occurred in the regions of low avalanche probability.

Temporal data could also be added into the GIS in order to examine the effects of time passed since a fire. For example, the portion of the study area burned in 1936 currently supports a successional lodgepole pine forest assemblage. This provides stability and anchorage for snow on the slopes, but during the first several years after the fire more unstable snow would have existed. High, medium, and low hazard likelihood categories could be added to the GIS based on time since fire: high hazard/spatial probability during the first year after fire; medium hazard/spatial probability during the early successional stages prior to conifer establishment (S-10 years in Glacier Park); and low hazard/spatial probability once the forest has reestablished itself and stabilizes the snowpack. However, once some areas are opened to snow avalanching, it is likely that avalanching will continue there and that succession to coniferous forest will be indefinitely retarded; the still-bare upper reaches of the burned and expanded avalanche paths along the southern margins of the park are mute testament to the disruptive longevity of fire in avalanche-prone terrain.

SPATIAL PROBABILITY OF PATH LOCATION

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Figure 9. Composite probability of where new avalanche paths should develop in the event of removal of forest vegetation by a forest fire.

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GIS APPLICATIONS IN FIRE MANAGEMENT AND RESEARCH

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Abstract—In 1985, Yosemite began using a geographic information system as a for tire management and research. The system has been used to compare historic fire incidence over a range of topography and vegetation types. Parkwide fuel inventories and prescribed bum units have also been depicted to predict fire behavior and effects. Research applications have included a lightning strike incidence analysis and a fire regime analysis hased on climate, vegetation, fuels, and topography. Current projects include developing a 1-hour-timelag fuel moisture theme by coupling the GIS with the BEHAVE fire behavior system to predict the behavior and spread of large fires.

When Yosemite National Park was established in 1890, the enabling legislation specified that all "timber, mineral deposits, natural curiosities or wonders in the area be preserved from injury and retained in their natural condition." Since then, the natural resources of the park have changed. Some of this change has resulted from past fire suppression policies which were intended lo prescrve the timber from injury, but which have significantly altered vegclalion composition and have allowed fuels to accumulalc.

In 1970, a program of prescribed burning was initiated to mitigate these conditions; and in 1972, lightning fires were allowed to burn in much of the park under a specific set of prescriptions (van Wagtendonk 1978). These programs were initiated based on the results of numerous research studies and extensive analyses of field data. To aid in theses analyses, a geographic information system (GIS) was installed in 1985 (van Wagtendonk and Graber 199 1).

In the past, resource information was scattered in files and publications, and on maps of varying scale and accuracy. The advent of relatively inexpensive microprocessors, high-resolution graphics, and large mass-storage devices has made lhc use of computers for entering, storing, retricving, and analyzing resource information a practical technology. Such a system is being used to help Park Service personnel make informed fire management decisions, monitor tong-term fire effects, and research complex fire relationships.

Data used to develop the various data themes were obtained from several sources. Digital data for elevation, slope, and aspect were obtained from the U.S.G.S. Mapped data from the park were used for fire management zones, vegetation type, fuel model, and past fire occurrence. Lightning strike data were obtained from the Automated Lightning Detection System operated by the Bureau of Land Management al the Boise Interagency Fire Center. Digital satellite imagery was used to reline the vegetation data.

The park's vegetation and fuels data are being surveyed in the field. The systematic surveys now being conducted mark the beginning of long-term monitoring of park resources. They will provide baseline data that will be used to verify classifications of remotely sensed imagery.

The software currently in use is the Geographical Resources Analysis Support System (GRASS) developed and supported by the U.S. Army Corps of Engineers (Westervelt 1988). 11 is a raster-based system with vector and image analysis capability and employs the UNIX operating system. GRASS was selected because it is in the public domain and runs on a computer with an open architecture.

FIRE MANAGEMENT APPLICATIONS

The first use of the GIS was to evaluate the rote fire has played in Yosemite's ecosystems (van Wagtendonk 1986). Fire has been an important factor in these systems for thousands of years and is of considerable scientific and political interest.

Fire records dating back to 1930 were reviewed and the point of ignition and areal extent of each lightning fire were digitized. These were than compared to information from the other themes. This analysis showed that fire occurrence and size varied significantly with vegetation type, elevation zone, topographic position, and drainage basin. Table 1 shows the distribution of lightning tires by vegetation type. West-facing slopes received 67 percent of the fires greater than 50 acres in size. In addition, fires were significantly smaller during the 12-year period before the prescribed natural fire management program was implemented in 1972 than during the following 12 years.

The GIS was also used to develop a fuel model map for the park. Fuel models are generalizations of actual fuel parameters and are used to predict fire behavior (Albini 1976). The vegetation and slope themes were combined with field surveys to assign a fuel model to each area of burnable vegetation.

The GIS depicts fire management zones which divide the park into units where different fire strategies are employed. These units include the routine suppression zone where all fires are put out regardless of origin, the conditional zone where

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Table I. Distribution of Lightning fires by vegetation type, Yosemite National Park, 1930-1983. The number of fires per milt ion a- per year is shown in the Last column.

Vegetation Type	Area of Park		Lightening Fires		
	Acres	%	Number	%	#/m/y
Chaparral Woodland	31,975	3.1	38	1.9	22.0
Lower Mixed Conifer	146,935	7.6	563	27.7	70.5
upper Mixed conifer	108,669	15.3	513	25.4	87.3
Red Fir	63.951	24.3	327	16.2	S4.7
Lodgepole Pine	232,202	21.5	467	23.1	37.2
Subalpine	59,383	8.6	36	I. 8	II.2
Alpine	110,391	9.5	38	3.6	12.2
Miscellaneous	7,613	10.2	41	.3	14.6
Total	761.319	loo. 0	2.023	100.0	49.1

lightning fires are allowed to bum before and after the fire season based under specific conditions, and the prescribed natural fire zone where lightning fires are allowed to bum at any time based on prescribed conditions. Thcsc zones were revise following the Greater Yellowstone Area tires in 1988: buffers were established between adjacent lands managed for different objectives. The GIS made this easy.

There are prescribed bum units within the suppression and conditional zones. Park personnel set fires in thcsc units to meet specific management objectives. Maps of thcsc prescribed bum units are also on the system and arc linked to data bases that include information on previous burns and bum schedules. Prior to burning, GIS maps of each unit arc prepared to show topography, fuel models, and resources of special concern such as archwlogical sites or endangered species habitat. These maps am used to plan bums and to predict fire behavior and effects.

FIRE RESEARCH APPLICATIONS

A fire regime analysis based on climate, vegetation, fuels, and topography is underway. Climatic data have been collected at a network of weather stations, and climate themes arc being created by extrapolating these data to the rest of the park by means of topographic variables, temperature lapse rates, and solar radiation equations. These themes, in conjunction with fire incidence, are being used to develop relationships with fire regime parameters such as fire frequency, intensity, and size.

Data on lightning strikes have been analyzed to detect spatial patterns and predict fire occurrence. Lightning strikes Were significantly correlated with elevation but not by slope and aspect. Since vegetation is strongly related with elevation, vegetation also showed a significant effect on lightning strike occurrence (table 2). Although the vegetation types in Table 2 arc slightly different than those in Table 1, a comparison

Table 2. Percent of area and number of Lightning strikes by vegetation type, Yosemite National Park, 1985-1989.

vegetation	%	1985	1986	1987	1988	1989	Total
Chaparral	3.1	27	12	21	27	32	III
Ponderosa	7.6	73	25	49	67	77	291
White Fir	15.3	193	58	114	130	168	663
Red Fir	24.3	378	98	151	259	333	1219
Lodgepole	21.5	301	135	130	292	380	1238
Whitebark	8.6	123	59	71	115	183	551
Alpine	9.5	115	8.5	53	161	266	614
Barren	10.1	116	58	66	117	182	539
Total	100.0	1326	530	655	1166	1555	5234

shows that the greater number of strikes in the lodgepole pine, subalpine, and alpine types did not result in a proportionaly larger number of fires. In those types burning and fuel conditions are not condusive to fire ignition and spread.

The locational accuracy of the strike detection system is reported to be approximately one mile (Krider and others 1980). Additional analyses will be performed in which the data will be adjusted to compensate for this error. The GIS will draw a circle with a l-mile radius around each strike and then select a random point within the circle.

The GIS' most important application in fire management and research will be the prediction of growth of large fires. Initial steps have been taken to develop a map of fuel moisture based on a given set of weather conditions along with topographic, vegetation, and fuel variables (Andrews 1986). Elevation is used to adjust temperature, while slope and aspect adjust for diffcrcnccs in solar radiation. Fuel model and vegetation type determine how much shading occurs. The GIS combined these five themes into over 14,000 unique catcgorics. When these were linked to the SITE module in BEHAVE (Andrews 1986), a fuel moisture value for each category was calculated.

Once fuel moisture is determined, fire behavior predictions can be made by linking the GIS directly to a large fire growth simulator. Bcvins and Andrcws (1989) are currently working on such a simulator; it operates in GRASS and combines the effects of moisture, wind, and topography on fire behavior. Outputs from the simulator will be residence time, flame length, and rate of spread. These could be displayed as maps, as could the area burned by time increments.

Predictions of tire growth will be invaluable to the manager who has to make a decision about a fire today based on the fire's expected location a month from now. These decisions will be easier to make if information about predicted future fire behavior and effects is available. For instance, a decision to suppress a fire because of smoke problems could be avoided if information about fire spread, fuel accumulation, and smoke dispersal were available. This may bc available when all of the various predictive models are fully developed and linked with a GIS that contains current resource information.

CONCLUSION

The GIS in Yosemite has already proven to be a useful tool in fire management and research. Fire operations have become more efficient, and the role tire plays in park ecosystems is better understood. Future applications in rca-time situations will increase the utility of this system. Such applications will include tire planning, suppression operations, and post-fire rehabilitation efforts. GIS technology promises to make increasingly accurate information more accessible to decision makers and researchers; thus make possible more effective protection of our park's valuable resources.

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FIREMAP

George L. Ball and D. Phillip Guertin'

Abstract—FIREMAP is a model for simulating surface fire spread through heterogeneous fuels and over non-uniform terrain. The model was constructed using PROMAP, a language which allows dynamic spatial models to be constructed using raster GIS data bases. The GIS system is used to construct the necessary input data of fuel types, moistures, slope, wind speed and wind direction. The model has been tested against a set of conditions under which specified fire perimeter shapes should be expected. The results of the tests indicate that the fire shapes developed using data representing actual field conditions arc reasonable. A model such as FIREMAP has application for prescribed burns, evaluating fire effects, and fire in the wildland/urban interface.

INTRODUCTION

Predicting the spread of fire is equally important for the use of fire as a management tool as it is in fire suppression. The difficulty of predicting the direction and speed with which a fire will move is compounded when the fire is burning in a rugged terrain of mixed fuel types. The development of a usable model that could generate and display the possible path of a fire using data from an actual site in a simple and straight forward manner would be extremely beneficial.

The BEHAVE system (Andrews 1986) gave fire personnel a tool which could be used to calculate various fire characteristics based on the Rothermel equations (Rothermel 1973). BEHAVE allowed the fire behavior analyst to use a set of measured conditions pertaining to the fire area and predict the rate of spread of the fire as well as several other factors. The rate of spread is calculated for the maximum direction of spread which is usually determined by the slope of the terrain and the wind direction. Additionally, adjustments arc available to give the rates of spread for the flanking and backing fires. It is impractical to try to calculate all possible directions and try to chart the fire over any distance for extended periods of time. As the fire perimeter becomes larger, changes in the variables start to increase so quickly in complex terrain that it becomes no longer possible to try to predict the entire fire perimeter.

Attempts to model fire spread using computers has been an ongoing project in many locations. In Australia, Green (1983) and Green and others (1983) have developed a model of fire spread for bush fires (grassland and shrub vegetation types). Although the model seems to provide a reasonable fire shape it has certain disadvantages. First, the use of an ignition template predetermines what fire shape will be generated (in this case an ellipse). Second, is the assumption that fire spread is by the shortest path from the ignition point. This ignores the fact that as the fire grows the influence of the ignition source diminishes to zero.

A computer program written by Ecnigenburg (1987) provides a method for calculating the fire direction and rate of spread. It is only applicable on small plots and requires the use of pre-marked locations to gather data about the fire. This makes it impractical for use in most situations.

Cohen and others (1989) have created a computer based simulation of fire that is used to test the strategies of fire management by deploying simulated fire fighting equipment. Although they indicate that the environment they are using is derived from Yellowstone National Park, there is no indication as to how the fire characteristics arc calculated and what algorithms are used to spread the fire. The use of this program is not to predict the spread of fire but the management of fire. Again, this program is not practical for use in fire spread prediction.

A model called FIREMAP was conceived as a method of predicting the spread of a surface fire through heterogeneous fuels and over non-uniform terrain by linking to a GIS data base (Vasconcelos 1988; Vasconcelos and others 1990). Vasconcelos used the Map Analysis Package (Tomlin, 1986) which is a GIS available on IBM PCs and clones. The FIREMAP model was applied to data collected during a fire in Ivins Canyon, located in the Spotted Mountains in east-central Arizona. The correspondence between the actual fire and the model was encouraging enough to pursue the further development of the model.

Although the model produced results that mimicked the actual fire, there was concern about the accuracy of the model due to the underlying algorithms employed by the MAP program. The basic premise of the Spread operator in the MAP GIS is to move uphill. Although this is a reasonable assumption for fire, it ignores the fact that fire spread is a result of local changes in the neighborhood of the flame front. What was needed was an algorithm that would display the same characteristics as an actual fire and still maintain the

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integration with the GIS data base. If such a model could be constructed it would have to be verified against some known fire shapes.

Three important papers concerning fire shape proved to be relevant to the FIREMAP problem. The papers by Peet (1967), Van Wagner (1969) and Anderson (1983) present a comparison of fire shapes under wind driven conditions. The wind speed ranged from low (2-4 miles per hour) up to moderate (12-15 miles per hour). The corresponding shapes range from ovoid to elongated ellipse. These shapes arc the result of fitting mathematical shapes to approximate the actual observed fire shapes. A test of the FIREMAP model would be to run it using a set of conditions under which the shape of the fire would be predictable. The remainder of this paper will discuss the development of the second version of FIREMAP and give examples of the results of the tests against predicted fire shape.

PROMAP

The original FIREMAP mode! was hampered by the deficiencies of the GIS program that was used to implement it. An analysis of what would be required to use GIS data bases to model processes such as fire led Ball (1990a, 1990b) to author PROMAP. PROMAP is a simulation language based on raster GIS. It overcomes the limitations of traditional GIS programs by using real numbers and the algorithms used in the operators arc designed for iterative operations. These are essential components in dynamic simulation. The basic premise of PROMAP is the principle of cellular automata (Wolfram 1984; Couclelis 1985, 1987; Gimblctt 1989, Casti 1989; Ball 199Oc).

Cellular automata theory is based on the premise that processes can be described by the influences of neighbors. In this case the neighbors are adjacent cells in the data base. PROMAP make use of this idea in the action of many of its operators. This allows the development of models that respond to neighborhood influences. The spread of fire is dependent on what it encounters in the environment as it moves across the landscape. By utilizing cellular automata theory, the spread of fire can be related to the progression of the flame front from one cell to the next. Therefore, what is required is an algorithm which incorporates the transitions from one cell to another into neighborhood effects.

The implementation of FIREMAP in the second version operates from what would normally be considered the neighboring cell. In this manner the algorithm can scan the surrounding area and determine if there is more than one potential direction from which fire might spread into the cell.

Every cell is considered to be homogeneous as to fuel type, slope and other variables. Consequently, the fire characteristics of each cell can be validly computed using Rothcrmel's approach (Rothermel, 1972; Andrcws, 1986). Each cell, however, can have its own characteristics. The direction of maximum spread in one neighboring cell may be away from the ccl I the algorithm is currently occupying.

Another neighbor may have a direction of maximum spread directly toward the current cell. In this case the fire would spread from the second cell because the fire will spread faster from it then from the first cell. In this manner the model accounts for the differences in neighborhoods as the fire progresses across the landscape.

Testing the Model

To test the function of FIREMAP we established a set of criteria which would allow us to predict the shape of a fire. This set of criteria describes what wc call the Zero State Conditions. Under zero state, we assume that the area of the fire is uniform as to fuel, zero percent slope, zero wind, and all other factors held constant. Under these conditions a fire started as a point source would bum in a circular pattern as in figure 1.

If we relax the condition of zero wind and allow the fire to be wind driven, then the shape of the fire should begin to approximate the shapes predicted by Peet, Van Wagner, and Anderson. With a 4 mile per hour wind the shape of the fire created by the model is seen in figure 2. The shape is not as elliptical as the mathematical formulation because of the square grid cell on which the model is running. The overall shape, however does show the expected heading fire with reasonable flanking and backing fires. Figure 3 shows the result of a wind shift during the simulation.

In figure 4 the right half of the simulation has been made using random fuel moistures in the range of 0-20%. The overall shape of the simulation compared to the uniform moisture simulation shows a more realistic tire pattern.

The ability of the FIREMAP simulation to produce shapes corresponding to the expected mathematical shapes is encouraging. The next step will be to compare the simulation to actual fire shapes using controlled burns.

FUTURE IMPLICATIONS

The capabilities found in FIREMAP show the potential for the development of a complete fire management tool. Two areas can be used as examples of how spatial dynamic models of this type could be used for fire management.

Fire Effects

The capability of FZREMAP *to* produce a realistic simulation of surface fire spread can be extended to post-fire effects. Since the intermediate calculations of the fire equations provide information concerning fire characteristics, such as fire intensity, the spread map can be altered to depict a map of those characteristics. For example, using the map of fire intensity, a model can be generated that would show what percentage of certain type of vegetation would be killed (Kunzmann and others 1990). Once this type of information is available, the next step is to consider what vegetation changes will occur over time.

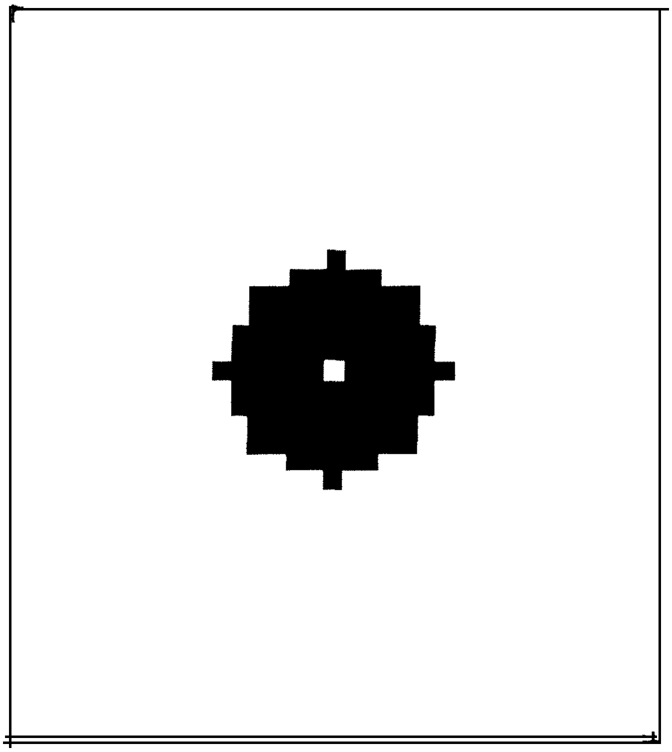

Figure 1--Fire spread simulation with zero wind speed. Unless otherwise noted, all simulations used a fuel model 9, 2% fuel moisture, 100% live woody moisture, zero slope, and wind direction of zero degrees. Duration of the bum is 400 minutes with cell sizes of 50 feet.

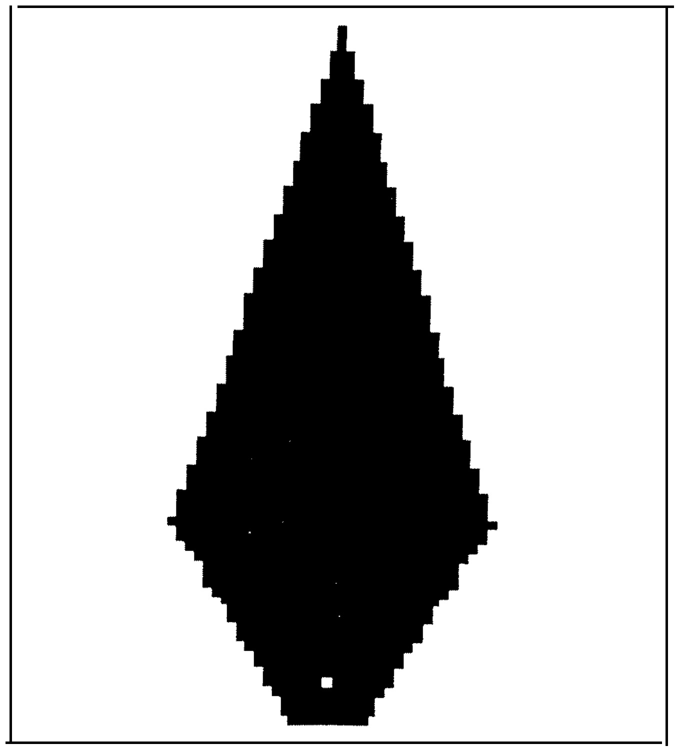

Figure ?--Fire spread simulation with a wind speed of 4 miles per hour. All other conditions arc the same as in figure 1.

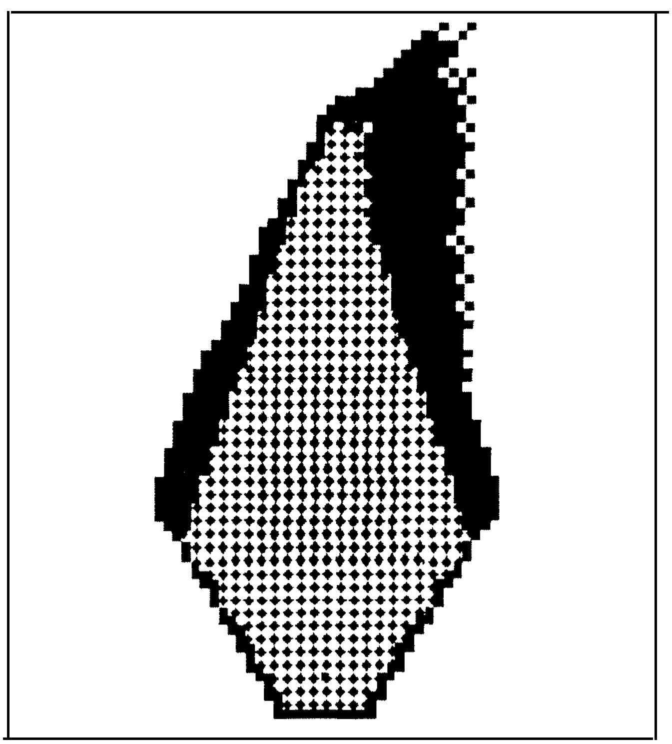

Figure 3--Fire spread simulation with a wind speed of 4 milcs pcr hour. After 400 minutes the wind was shiftcd to 45 dcgrees for an additional 100 minutes. All other conditions arc the same as in figure 1.

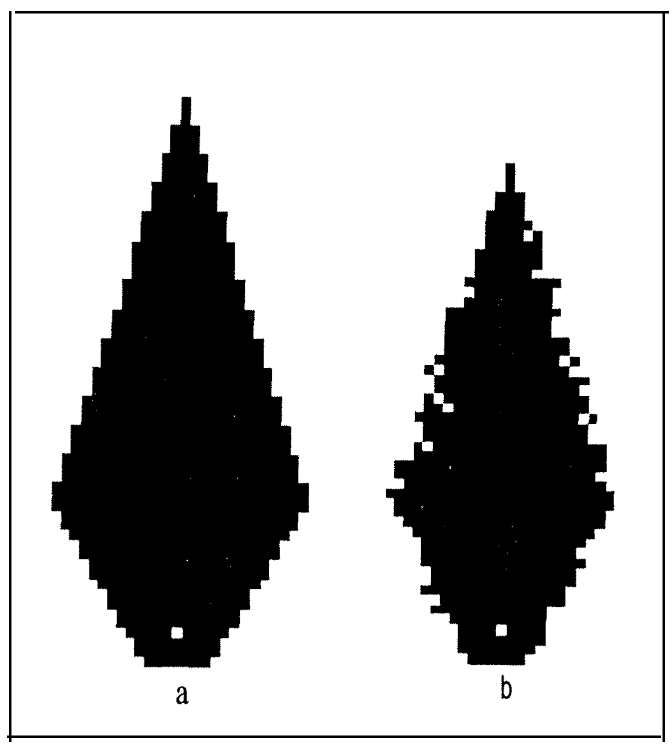

Figure 4--Fire spread simulation with 4 mile per hour wind for 480 minutes. The conditions for part a are the same as in figure 2. In part b the fuel moistures arc randomly distributed across the area with values from 0% to 20%. All other conditions are the same as in part a.

Risk Management

Even at the level of the current capabilities of FIREMAP, the application of this type of model to risk management is cvidcnt. The ability of the fire analyst to anticipate flame lengths, intensity and direction of spread of a fire would provide better utilization of effort and reduce the possible loss of life and property when managing fires.

In areas of the urban/rural interface, the use of FIREMAP could provide information on potential property loss of forcst areas managed for fuel reduction versus areas that are not managed. This could have a significant effect on insurance rates and on gaining acceptance by the public for prescribed bum policies.

CONCLUSION

The ability of the FIREMAP model to simulate the spread of surface fire under specific conditions indicates that the use of this technology can provide better management tools. Further work will need to be done on improving the mathematical descriptions of tire for use in spatial dynamic models. As the models become more sophisticated and are verified by field tests, their application for fire management, fire ecology and related areas is readily apparent

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THE ART OF LONG-RANGE FIRE WEATHER FORECASTING

Francis M. Fujioka[1]

Abstract-On the heels of the Yellowstone fires of 1988, a Cabinet-level tire management review team recommended research "to improve the ability to predict severe tire behavior, conduct long-term weather forecasting, and identify past abnormal events." In a 1989 report, a Forest Service task force identified a high priority need for long-range weather forcccasting, in support of prescribed fire management.

Research on long-range fire weather forecasting started in 1986 is described, particularly a newly developed system for forecasting fire weather elements in a monthly timeframe. Some fundamental differences hctween current and future forecasts arc envisioned, not only in terms of content, but also in preparation and review. Both forecaster and user must come to grips with the quality of forecast information, which has implications for both long-range forecasting and long-range planning.

INTRODUCTION

The Yellowstone fires of 1988 have left scorch marks, not only among the Park's surviving trees, but also in the annals of United States fire managernenl policy. The fire event was scrutinized at the highest level of federal government, when the Sccrctarics of Agriculture and Interior convened a fire management team to review national fire management policies and practices. Among the recommendations in the team's report[2] was the need for research "to improve the ability to predict severe fire behavior, conduct long-term weather forecasting, and identify past abnormal events."

On a separate but related issue, a Forest Service task force established to review prescribed fire management policy rccornrncndcd in 1989 that high priority be given to research on long-range weather forecasting, to support prescribed natural fire management. Fire planners need to determine, as far ahead as reasonably possible, the buildup, duration, and termination of weather-inciuccd fire potential. The assignment is hardly trivial; even in our most common experiences, each of us can probably recall an errant forecast, at that not even a long-range forecast.

Some reflection is warranted on expectations for long-range fire weather forecasts. This paper describes the goals, progress, and prospects of a program for long-range fit-c weather forecasting research, currently being conducted at the Pacific Southwest Research Station (PSW), in Riverside, California. The first section outlines the objectives of the research program, including a paradigm for the application of weather forecast information, irrespective of the forecast horizon (i.e., seasonal, extended-, medium-, or short-range). Rcscarch results obtained to date arc then dcscribcd. Finally, some conjectures arc made on the nature of fire weather forecasts of the future.

[1]Pacific Southwest Research Station, Forest Service, U.S. Department of Agriculture, Riverside, CA.

[2]Report on Fire Management Policy, USDA/USDI, December 14, 198X. Unpuhlishcd report available at the USDA Forest Serv., Fire and Aviation Mgmt., Washington, D.C. 20090.

FIRE WEATHER RESEARCH OBJECTIVES

The research on long-range fire weather forecasting at PSW was inspired by the information needs articulated by fire managers and fire scientists in 1985 (Rios, 1989). The group recognized that fire management planning required weather forecast information at lead times that varied from hours to months (Table 1). In this paper, seasonal refers to a 90-day forecast, extended-range *to* a 30-day forecast, medium to the 3-14 day range, and short-range to forecasts of less than 3 days. Strategic planning, therefore, requires seasonal and cxtcndcd-range forecasts, preparedness planning requires medium-range forecasts, and implementation planning utilizes short-range forecasts. Particular effort was focused on the concept of a 30-day fire potential forecast for national level planning within the Boise Interagency Fire Center. Eventually, the Intelligence Section at BIFC developed a process for creating 30-day categorical forecasts of fire potential for the contiguous United States.

The purpose of the fire weather forecasting research at PSW is to develop fire weather forecast products identified (Table I), but not generally available. In 1987, research was initiated to develop models for a monthly forecast of mean afternoon dry-bulb temperature, dewpoint temperature, windspccd, and precipitation frequency for the U.S. (precipitation frequency is defined as the number of days in the month that precipitation exceeds 0.1 inch). In 1988, research began on medium-range forecasts of the daily variations in these variables, over a period of (nominally) 10 days. In 1989, work commenced on the feasibility of forecasting seasonal fire climate, particularly in relation to El Nino and La Nina events (see Philander, 1989, for a good description of these events).

An equally important goal of the research program is to evaluate uncertainties in the forecasts. It is well-known that, by extending the forecast further into the future, the quality of the forecast can be seriously degraded. If the probabilistic character of the forecast uncertainties is described, the user can assess the risk inherent in using the forecast information in the decision process.

Table 1--The Lead times for weather forecast information vary, depending on the management decision under consideration.

Decision	Strategic: Seasonal. 4x/yr	Strategic: 30-15 days 2x/mo	Preparedness: 14-6 days 2x/wk	Preparedness: 5-3 days Daily	Implementation: 3-1 days 2x/day	Implementation: < 1 day 2x/day
Staffing and support (BIFC		X		X		
Severity fund requests	X	X				
National resource mgmt identify locate	X	X	X			
Alert			X			
Staging Level I			X	X		
Staging Level II					X	
Deployment					X	X

Using Forecast Uncertainty Information

In its report, the Forest Service Task Force on Prescribed Fire Management Criteria recommended a risk assessment procedure comparable to decision analysis[3]. Decision analysis provides a means of evaluating the merits of decision alternatives, weighted by information uncertainty (e.g., weather forecasts), and the values at risk (fig. 1). An example by Seaver and others (1983) shows how uncertainty information is used.

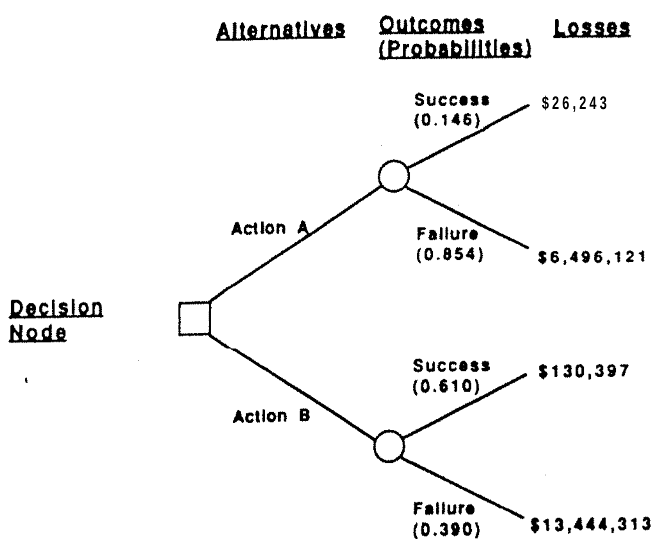

Figure 1--Decision tree from Seaver and others (1983) illustrates how decision analysis integrates information on the uncertainties and the values of decisions and their outcomes.

[3]Report of the Task Force on Prescribed Fire Management Criteria, Forest Service, 1989. Unpublished report available at the USDA Forest Serv., Fire and Aviation Mgmt., Washington, D.C. 20090.

In an escaped tire situation analysis, a tire manager considers whether to suppress a tire at (A) 7 hectares, or at (B) 40 hectares. The loss is much less at 7 hectares, but the probability of successfully implementing A is also small. The losses and probabilities can be depicted on a decision tree.

Suppose we decide, as a rule, to pursue that course of action which results in the smallest expected loss. The information in the decision tree can be used to calculate the expected loss for each decision:

Expected loss for A = probability of success of A * loss with success of A
+ probability of failure of A * loss with failure of A
= 0.146*$26,243 +0.854*$6,496,121
= $5,551,519

Expected loss for B = probability of success of B * loss with success of B
+ probability of failure of B * loss with failure of B
=0.610*$130,397+0.390*$13,444,313
= $5,322,824

The above analysis suggests that the alternative to suppress at 40 hectares is optimal. Although a smaller loss would be incurred by implementing A successfully, the probabilities indicate that, in the long run, choosing A over B is more costly. Recently, Brown and Murphy (1988) demonstrated how decision analysis can be used to assess the economic value of weather forecasts in wildfire management. Their decision tree was more complicated than the one in the previous example, but the principle behind the analysis remains the same: consider the consequences of the decisions, and the probabilities of their outcomes, before deciding on an appropriate response.

Effect of Uncertainties in the Probabilities

The preceding example assumed that the probabilities were known; the expected losses calculated in that context are truc expected values. In reality, the probabilities can only be estimatcd, which introduces another level of uncertainty in the decision process. Suppose, therefore, that the probabilities arc random variables, and that the values used in the computations are estimates. The resultant expected losses thercforc are also random variables, subject to uncertainty. Where the calculated expectations in the preceding example were represented by points on the real line, the expected loss values calculated with the probability estimates are themselves only estimates. If the random variable representing the probability estimates can be described by a probability model, the calculated expected losses may be viewed in the context of confidence intervals. Hence, where the decision previously considered only which of the expected values was the larger, now the overlap, if any, of the confidence intervals might also bc considered. In other words, the loss of certainty in the probability information clouds the expected loss calculations. If the smeared results (intervals) are still distinct from one another, the decision rule can be applied unambiguously. If, on the other hand, the intervals overlap, then the decision rule is compromised. The degree of overlap may be used as a basis for evaluating the acceptability of forecast uncertainties.

Prcsently, fire weather forecasts are not expressed in probabilities. Nor do they usually extend beyond a 5-day period. The research at PSW is aimed at providing the means to do both. The next generation of fire weather forecasts will extend the forecast horizon to 30 days.

EXTENDED-RANGE FIRE WEATHER FORECAST

The 30-day tire weather forecast was developed primarily in response to the national level strategic planning needs at the Boise Interagency Fire Center. The fire weather forecast will be integrated with information on crop moisture, the Palmer Drought Severity Index, and various fire danger and fire bchavior indicators, to produce a forecast of 30-day fire potential for *the* continental U.S. The systcm that presently produces the fire potential map uses 30day and 90-day predictions of temperature and precipitation from the National Weather Scrvicc (fig. 2).

The newly dcveloped 30-day forecast models were tailored to firc weather needs (Klcin and Whistler, 1989). They focus on mean afternoon conditions, when the fire weather threat is usually the grcatcst, and, they add the elements of relative humidity and windspeed. Specifically, the models predict the departures from the monthly afternoon (approximately 1300 Local Standard Time) means of the following variables:

- Dry-bulb temperature
- Dcwpoint tcmpcraturc
- Windspced

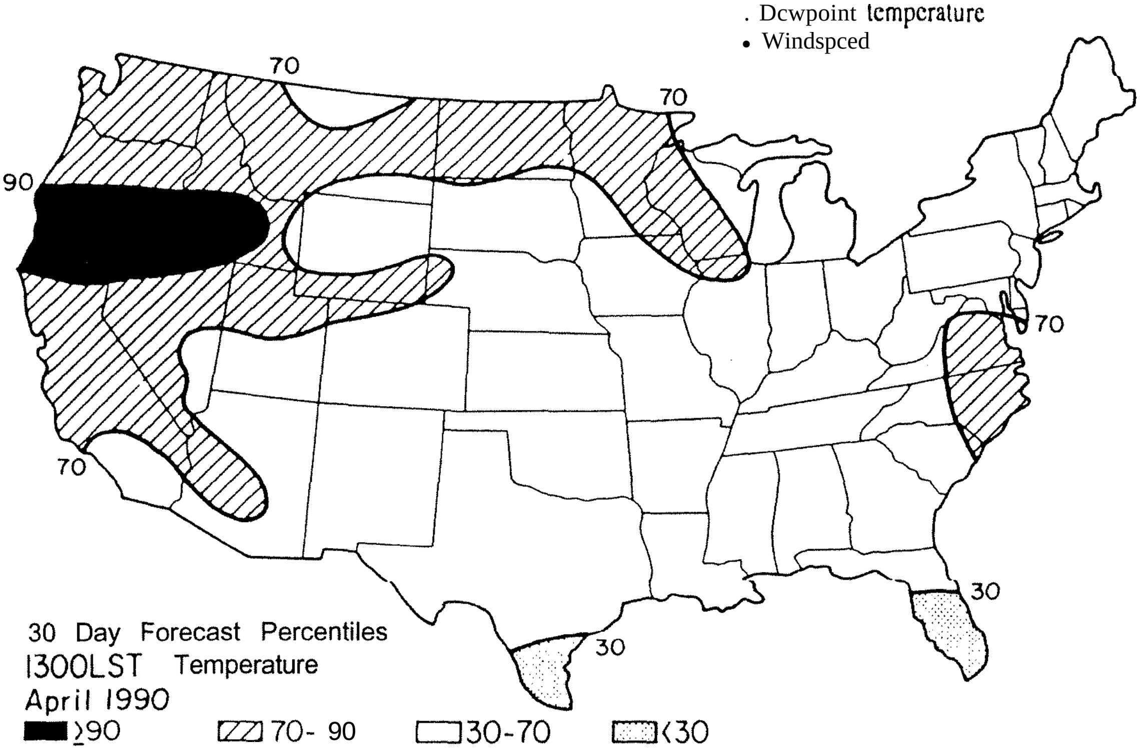

Figure 2--An cxtcndcd-rangc forecast of the 30-day mean tcmpcraturc percentiles for the contiguous U.S. Thc pcrccntilcs rcprcscnt thc proportion of thc climatological database that the forccast value cxcecds.

A model was also developed to predict the number of days in a month that precipitation would exceed 0.10 inch. This is complemented by existing National Weather Scrvicc models for monthly precipitation amount.

The models arc driven by the forecast monthly mean prcssurc-hcights at the 700 millibar level (nominally 10,000 ft), which is prepared by the National Weather Service twice monthly. The assumption underlying the models is that excursions of a variable from its monthly mean, as observed at a given location, arc associated with large-scale atmospheric pressure patterns over the earth's surface; moreover, the excursions can be predicted statistically from information about the expected pressure-heights at a few points, and current conditions at the location of interest. In fact, the models are derived from regression analysis:

$$(x,t+1) = a_o \Delta f(x,t) + \sum_{i=1}^{k_x} a_i \Delta p_i$$

in which $\hat{\Delta f}(x,t+1)$ is the forecast surface weather anomaly for location x and month $t+1$, $\Delta f(x,t)$ is the observed surface weather anomaly for month t at location x, and $\{\Delta p_i\}$ are forecast 700 mb height anomalies at designated gridpoints for month $t+1$. The $\{a_i\}$ are regression constants. The constant a_o may bc 0, which then implies that current weather conditions at the location are not strongly correlated with next month's weather. Regression models have been developed for 127 stations across the contiguous U.S.

The forecasts are interpolated to an orthogonal grid, and contoured. The forecast at each station is also compared to the empirical distribution of the variable, and expressed as a percentage of the observed monthly means in the station record that the forecast value equals or cxcecds. This transformation converts the forecast from its original units to a dimensionless number ranging from 0 to 100. Consequently, diffcrcnt variables may be forecast in different regions of the country, transformed to a percentage value by the process described above, and represented concurrently with othcr forecasts on a map. McCutchan and Main (1989) found that no single variable is best correlated with fire activity in the U.S., and that the forecast of different variables in diffcrcnt parts of the country might in fact be prefcrable.

LONG-RANGE FORECAST TOOLS OF THE FUTURE

Medium-range Fire Weather Forecasts

Research is presently focused on the development of medium-range tire weather forecasts, which implement computer models of global weather processes. Currently, the National Mctcorological Center gcncratcs a daily medium-range forecast of weather conditions out to 10 days, but the distribution of this product is limited (Petersen and Stackpole, 1989). The forecast can describe daily variations of weather, which is the minimum temporal resolution required for National Fire-Danger Rating System (NFDRS) calculations.

Spatial resolution, however, is limiting, for local fire weather applications. The minimum grid interval is approximately 85 km, but the terrain model used to define the surface boundary removes terrain features of less than 5 degrees; the maximum height depicted in the Rocky Mountains is 2600 m. Therefore, the model cannot be expected to describe terrain-induced windflows that complicate fire behavior. Moreover, sea and lake breezes are not forecast accurately, thus reducing the ability to predict moisture variations in areas influenced by large water bodies. A typical way of enhancing the spatial resolution is through statistical models. For example, weather variations at fire weather stations can be modeled as a function of corresponding medium-range model output. Presently, however, the database for such studies are somewhat limited, particularly for the current version of the medium-range forecast model. An objective of the current PSW research effort is to develop forecasts of the daily variations in fire weather over a 10-day period.

Seasonal Fire Weather Forecasts

The objective of the seasonal fire weather forecast is to estimate the weather-induced tire potential in a 90-day timeframe. This forecast will be a prediction of the mean weather conditions over the 90-day time period that begins approximately from the day that the forecast is issued. Like the 30-day forecast, the seasonal forecast will key on variations in large-scale, slowly changing weather patterns.

The experience of the National Weather Service shows that 90-day forecasts of temperature are moderately skillful in winter and summer, but precipitation forecasts do not fare well (Wagner, 1989). The forccastibility of anomalous precipitation may be enhanced during periods that global circulation systems exhibit strong anomalies, e.g. the El Nino and Southern Oscillation events, which are characterized by unusually low sea-surface temperatures in the Pacific Ocean, and unusually strong negative pressure gradient between Tahiti and Darwin, Australia (Chu, 1989). But the question of forccastibility cannot be answered without considering the intended use of the forecast.

A study of thc correlation between El Nino and fire activity (Simard and others, 1985) showed that fire activity tended to decrease in the Southeast during El Nino years, but no relationship was apparent in most of the other states in the contiguous U.S. The correlation in the southeastern states was attributed to anomalously high precipitation experienced by those states in El Nino years, which generally decreased fire potential.

SUMMARY

The not-too-distant future will see the addition of long-range fire weather forecast guidance to the tools of the fire manager. These will include a forecast of daily variations of fire weather out to 10 days (nominally), a forecast of the 30-day means of fire weather variables, and a seasonal forecast of the 90-day means of temperature and precipitation effects. Statistics of forecast reliability will also be provided, so that the manager can assess the credibility of the forecast information, which can vary over time, space, and with particular climatic conditions. Equally important to the decision process arc the values at risk, which can be integrated with the forecast information in a decision analytic model. At best, the models can provide guidance; decisions must still bc made by decisionmakers. The forecasting system visualized here will provide the user with the tools, the process, and the information necessary to deal with long-range fire weather forecasts and their uncertainties.

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CULTURAL

PRIVATE, NON-INDUSTRIAL FOREST OWNER'S PERCEPTIONS OF CONTROLLED BURNING INFLUENCING FOREST MANAGEMENT

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Abstract—Perceptions of controlled burning[2] by private, non- industrial forest (PNIF) owners provide insight into their forest management behavior. Detailed personal interviews of randomly selected forest owners in the Wiregrass region of Alabama were conducted to determine relationships between these perceptions, ownership objectives, and forest management activities. Over two-thirds of those interviewed felt controlled burning was a useful forest management practice on their land although only 25.3 Percent were presently using controlled burning. Both positive and negative perceptions of controlled burning are presented. Emphasis is made on the relationship of these perceptions to forest management behavior by PNIF owners and the subsequent importance to professional foresters who work with this public.

INTRODUCTION

There are numerous and conflicting perceptions of fire in the context of forest management. While professional foresters perceive tire as a useful tool in manipulating forest conditions, fire use by private, non-industrial forest (PN I F) owners likewise has an environmental and historic logic. Trained fire management personnel may at times criticize or question reasons given by these owners for burning; however, the act itself is a planned, deliberate application of knowledge to meet a determined goal. Forester's perceptions of PNIF owners may not encompass this knowledge and may therefore limit communication between the two groups. In order to improve communication and coopetative management efforts between foresters and PMF owners, it is important to discover the perceptions that these individuals have of forest management activities -- in this case, the use of controlled burning.

What are PNIF owner's attitudes regarding fire as a forest management activity? How do they become aware of its utility? Where do they look for information and assistance in its application? What arc their reasons for using controlled burning on their lands? And can wc determine whether those owners with positive perceptions of controlled burning arc more active managers (that *is,* using more activities) than owners with negative perceptions of controlled burning?

With the advent of the professional forester and the growth of industrial forestry in the South, the practice of burning was seen as a major obstruction to implementing scientific forest management (Schiff 1962, Riebold 1971). Frequent burning by livestock owners, turpentiners, and other woods residents prevented foresters from applying their knowledge in forest management (Pyne 1981). To the contrary, foresters found they spent the majority of their time working to control woods fires. It is doubtful that foresters of the early 1900s in the South felt fire was a useful tool in forest management, primarily because of their lack of control over its application. John Shea undertook a "psycho-social" investigation of woods- burners in and around Bankhead National Forest in Alabama in the late 1930s, and generalized his findings to the conditions and behaviors of other fire users across the southeast (Shea 1940). Shea presented the typical southern woodsburner as backward, uncaring, and irresponsible. Perhaps the most significant conclusion drawn from his research was that: .m:l "Southern 'woods burning' is a human problem and should be tackled in a scientific and human way." (Shea 1940) The significance lies in the recognition of human behavior as the problem source and in the proposition that systematic investigation was needed to determine the nature of that behavior and to identify possible means to modify that behavior.

Historically, PNIF's have been considered under-managed resources, producing much less than their capacity of ecological, economic, and social benefits (Burger and Teer 1981, Dutrow 1986, Rogers and others 1988). A perceived lack of PNIF owner awareness and understanding of forest management technology concerns professional foresters (Sedjo and Ostermeier 1978) as a cause of under-management (Black 1983). These problems have justified research on forest management technology transfer based upon communication between professional foresters and PNIF owners. A better understanding of communication elements, patterns, and needs between professional foresters and PNIF owners could contribute to the effectiveness of forest management efforts by both *groups.*

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[2]The term "controlled burning" is used in this paper as it reflects PMF owner's perceptions of what fire in the woods should bc, and was the term used by participants in this study. No difference is implied hy the author between the terms "prescribed burning" and "controlled burning"; rather, the terms tend to illustrate the differences in paradigms of professional foresters and PMF owners, respectively.

Research on agricultural technology transfer has typically employed diffusion and adoption models, which provide theoretical generalizations about how technology information is communicated to landowners and how they decide to use new technologies (Rogers 1983). M uth and Hendee (1980) demonstrated that such models could also be useful in explaining the adoption of forest management technologies. Key elements in a diffusion model are an innovation, its adoption by individuals, and its diffusion through a social system over time as a result of communication. An innovation is an idea or object that is perceived as new; its adoption is affected by a variety of characteristics. An adoption decision is a five-step learning process. First, an individual becomes aware of the innovation; if his interest is aroused, the individual will then seek additional information about the innovation. As his knowledge accumulates, the individual evaluates the merits of the innovation relative to current or alternative conditions. In the fourth step, the individual conducts a trial or experiment with the innovation, from which he eventually decides to either adopt, reject, or continue to gather information concerning the innovation. Information about technological innovations is often disseminated by technical practitioners, change agents, professionals, and educators (McNaul 1972, Vickcrs 1974, Grunig 1980, Rogers 1983, Rogers and others 1988). The practicing professional forester who assists PNIF owners has a dual communication role: that of helping owners become aware of new forest management technologies as well as providing detailed information in later stages of the adoption process. A major goal of extension forestry programs is to provide information and education opportunities for the improvement of PNIF management (Gould 1975).

Adoption models suggest that practitioners who cultivate an understanding of their client audience, including that audicncc's technical knowledge and decision patterns, arc more effective in educating landowners and changing their behavior (Rogers 1983). This is consistent with interpersonal communication models described in communication texts (Fazio and Gilbert 1981). The goal of this study was to help foresters understand PNIF owners and their use of controlled burning; more specific objectives are presented below.

STUDY OBJECTIVES

The problem addressed by this study was to identify how private, non-industrial forest owners become aware, gather information, and conduct evaluations and trials in determining whether to adopt a particular forest management activity. Specifically, the study was developed to: (I) assess the background experiences of Alabama PNIF owners with controlled burning; (2) identify patterns by which Alabama PNIF owners make decisions about undertaking controlled burning; (3) identify PNIF owner characteristics useful in delineating adopter categories as described by Rogers (1983); (4) identify the use and influence of information and assistance sources in stages of adoption and diffusion behavior.

STUDYAREA

This study was conducted in the eight southeastern counties of Alabama that comprise the majority of the Wiregrass region of the state. These counties cover approximately 3.72 million acres, of which 2.34 million acres are forested. In these eight counties there are approximately 20,310 PNIF owners of between 5 and 500 acres who control 44.5 percent of the area's forestland.

METHODOLOGY

A standardized interview instrument was designed and pre-tested to obtain: (1) information on PNIF ownership objectives; (2) information on PNIF forest management practices; (3) indications of owners' opinions on these practices; (4) indications of information sources used by owners; and (5) information on controlled burning use by PNIF owners.

The sampling procedure used was a stratified random sample of PNIF owners in the eight counties who owned more than 5 acres but less than 500 acres of forest land. The sample size for the study was calculated from the total population following Blalock (1982), resulting in a sample of 145 individuals. The sample was stratified across the region by the ratio of the number of owners in each county to the total for the study area, giving the number of interviews needed for each county. The number of interviews planned for each county was increased by an expansion factor of 1.86. This buffer eliminated the need for additional selection of names from county records in order to rcptace names of persons who were deceased, no longer owned land, declined interviews, or otherwise could not be contacted to schedule an interview. A random sample of names was drawn from county land records in the tax assessor's office of the respective county courthouses. Each name drawn was checked against the criteria of ownership (PNIF versus public or industrial) and size (greater than 5 but less than 500 acres). For the latter criterion, Alabama Forestry Commission records were used to determine that individuals did not own more than 500 acres of land in the county. In addition, individuals living outside the state of Alabama or at a distance greater than 60 miles from the boundary of the study area within the state were rejected from the sample. Appointments for interviews were made either through telephone contact or by locating individuals using rural post office routes. All interviews were conducted in person by the principle investigator. Interviews were primarily conducted in individuals' homes, although a number of interviews were arranged at businesses, restaurants, or in farm fields to accommodate participants' schedules.

A bound copy of the interview instrument was offered to each respondent so they might better follow the questions of the interviewer. The interviewer recorded all answers given by participants on his copy of the instrument. These answers included comments in open-ended questions as well as remarks made throughout any portion of the interview session. Completion of the survey instrument required an average of 23 minutes. Responses from the completed interview questionnaires were coded for analysis using the Statistical Package for the Social Sciences (SPSSx) (Nit and others 1983). Chi-square tests were used to determine whether statistically significant associations existed between groups within the sample. The lambda statistic was also used to determine whether independent variables associated with PNIF owners were of any use in predicting particular dependent variables of PNIF owners and their actions. Open-ended question responses were organized and coded using content analysis. A measure of PNIF owner attitudes toward forest management activities was determined through a Likert scale. The Likert method is based on the assumption that a scoring of the responses to items describing a particular variable provides a reasonably good measure of the respondent's attitude towards the variable (Babbie 1983). The use of the scale also prevents purposive response bias or manipulation in that the respondent does not know what variable is being measured. Factor analyses were performed on Likert scale statements to identify attitudinal themes of participants. Cronbach's alpha was used to determine the correlation and homogeneity of participant responses in order develop attitudinal rankings.

RESULTS

The private, non-industrial forest owners of between 5 and 500 acres of forestland in the Wiregrass Region of Alabama were predominantly male, resident owners of approximately 75 acres of forestland. Most of the non-resident owners (64.9 percent) lived within ten miles of their forestland. The majority of individuals lived on farms or in rural, non-farm locations. While the average age of these PNIF owners was between 55 and 60, the ages ranged from 29 to 99 years. One- third of the individuals had less than a high school education, another third had completed high school, and the remainder had junior college to graduate degrees. The occupations of study participants tanged from teachers, construction workers, and ministers to government employees, housewives, and barbers. Among those PNIF owners who were retired (35.9 percent), the majority had been businessmen or farmers.

Within the study area, 71.7 percent of PNIF owners felt controlled burning was a useful forest management activity on their land. These owners commented that controlled burning helped clear understory brush and trash trees, helped fire-proof their land and kept the pines growing free. These individuals had ownerships that ranged from 25 to 500 acres and displayed a broad range of occupations and educational backgrounds. The primary ownership objectives of those individuals who perceived controlled burning as useful ranged from the production of income from the sale of timber or providing a homesite to their land providing a heritage for future generations or shelter for wildlife (Table 1). The remainder of the PNIF owners cited the following primary ownership objectives: land as a future investment (9.6 percent); family recreation area (6.7 percent); hunting area (5.8 percent); firewood production area (3.8 percent); to preserve natural beauty (1.9 percent); and other reasons (4.8 percent).

Table 1. Primary forest ownership objectives of PNIF owners in the Uiregrass Region of Alabama who perceive controlled burning as a useful forest management activity on their forestland (N=104).

Primary Objective	Frequency	Percentage
Income from sale of timber	22	21.2
Homesite	22	21.2
Heritage for future generations	15	14.4
Shelter for wildlife	11	10.6
Future investment	10	9.6
Family recreation	7	6.7
Hunting	6	5.8
Firewood production	4	3.8
Natural beauty	2	1.9
Other reasons	5	4.8
Total	104	100.0

The primary reasons given by the 38.3 percent of PNIF owners who felt that controlled burning was not a useful forest management activity on their forestland were that burning "Destroys small trees." or "Damages timber." (31.7 percent); "[It] runs-off wildlife, especially songbirds." (15.1 percent); and "Mine is hardwood forest, so it doesn't tit." (12.2 percent). Demographic characteristics indicated that these PNIF owners tended to have smaller individual ownerships, somewhat lower education levels, and were more often non-resident owners. However, none of these demographic characteristics were significantly different at the 0.10 level from those of persons who felt controlled burning was a useful forest management activity. Further, the lambda statistic indicates that these individual characteristics arc of little value in predicting the controlled burning utility perception of PNIF owners in the Wiregrass region.

The primary ownership objectives of 41 PNIF owners who felt controlled burning was not a useful forest management activity on their land ranged from their land providing a homesite or a heritage for future generations to providing shelter for wildlife or an area for family recreation (Table 2). Other primary ownership objectives cited by this group were: land as a future investment (9.8 percent); to have an area to cut firewood (4.9 percent); to preserve natural beauty (2.4 percent); and other reasons (4.9 percent). No one in this group identified having an area to hunt as their primary ownership objective.

Table 2. Primary forest ownership objectives of PNIF owners in the Uiregrass Region of Alabama uho do not perceive controlled burning as a useful forest management activity on their forestland (N=41).

Primary Objective	Frequency	Percentage
Income from sale of timber	5	12.2
Heritage for future generations	7	11.1
Homesite	8	19.5
Shelter for wildlife	6	14.6
Future investment	4	9.8
Family recreation	6	14.6
Hunting	0	0.0
Firewood production	2	4.9
Natural beauty	1	2.4
Other reasons	2	4.9
Total	4	loo. 0

Of the 104 individuals interviewed who perceived controlled burning as a useful forest management activity on their land, 35 percent cited "personal experience or observation" as their initial source of awareness of controlled burning utility. These persons described having first observed controlled burning in some fortuitous manner, such as noticing smoke, following it to its source, and then observing the actions of the fire and the individuals tending it. A number of these owners recalled returning to the burn area at some later date, ranging from a few days or weeks to two or more years, and making personal assessments of the impacts of the bums. Almost twice as many study participants who stated that controlled burning was useful cited personal experience or observation as their initial awareness source compared to the number who cited either mass media (ncwspapcr or magazine articles, primarily) or Alabama Forestry Commission personnel as their initial awareness source (Table 3).

Table 3. Sources of initial awareness of controlled burning utility cited by PNIF owners uho perceived controlled burning useful on their forestland in the Uiregrass Region of Alabama (N=119)[1].

Source	Frequency	Percentage
Personal experience or observation	3.5	29.4
Mass media	20	16.8
Alabama Forestry Commission	19	16.0
Friend or neighbor	12	10.1
Industry forester	II	9.2
Others (five - e s)	22	18.5
Total	119	100.0

[1]N is greater than the 104 individuals uho perceived controlled burning useful because some owners could not confidently identify a single source of initial awareness of controlled burning utility.

Responses to Likert statements on controlled burning grouped study participants as having positive, neutral, or negative attitude scores concerning controlled burning. These attitude scores were developed independent of the questions regarding the perceived utility of controlled burning. Study participants with positive attitude scores toward controlled burning as a forest management activity tended to cite Alabama Forestry Commission foresters as their initial awareness source twice as often as owners with neutral to negative controlled burning attitude scores. Individuals with neutral attitude scores concerning controlled burning as a forest management practice tended to cite mass media or personal experience or observation as their initial awareness sources.

Private, non-industrial forest owners were asked whether they knew of any neighbor's or aquaintance's use of controlled burning in the management of their forrestland. This line of questioning was intended to investigate possible vicarious experiences with controlled burning through others' use of the practice. Almost half (47.5 percent) of all individuals interviewed stated that they were aware of controlled burning use by neighbors or aquaintanccs. Over 82 percent of this group of owners also perceived controlled burning as a useful forest management activity on their forestland. A generalization from this could be that owners using or predisposed toward a particular forest management practice, such as controlled burning, arc more likely to be attuned to other's use of the same or similar activities.

Approximately two-thirds (65.8 percent) of those persons who were aware of controlled burning use by neighbors or aquaintanccs felt positively influenced by the controlled burning cxpcricnces of others (Table 4). This number was equivalent to 3 1 .O percent of the total number of individuals interviewed. These persons commented that they saw burning as an effective means for "clearing understory brush", "eliminating trash trees", promoting "better food for deer and turkey", and "helping pine trees grow better"

Of those individuals aware of others' use of controlled burning, 26.3 percent felt negatively influenced by neighbors' or aquaintances' experiences with controlled burning. Eighty percent of these persons stated that they felt the iirc was too hot or damaging, or said that the fire had escaped, burning unintended areas or someone else's land.

Table 4. Private, non-industrial forest owner's stated direction of influence from neighbor's or aquaintance's use of controlled burning (N=69).

Influence	Frequency	Percentage
Positive	45	65.8
Neutral	6	7.9
Negative	18	26.3
Total	69	100.0

All study participants were asked to whom they would most likely go for information if they had specific questions about the use of controlled burning on their forestland. This question was asked regardless of the individual's perception of the utility of controlled burning. Alabama Forestry Commission foresters were cited significantly more often than any other controlled burning information source regardless of

Table 5. Controlled burning information sources identified by PNIF owners of the Alabama Uiregrass Region by attitudes of owners toward controlled burning (N=145).

Sources	Attitudes			
	Negative	Neutral	Positive	
AFC forester	30	29	20	79
	62.5	47.5	55.6	54.5
industry forester	5	5	3	13
	10.4	6.2	a.3	9.0
Other'	13	27	13	53
	27.1	44.3	36.1	36.5
	48	61	36	145
	100.0	100.0	100.0	100.0

Chi-square: 3.451, d.f. = 4 Significant at alpha = 0.10
'Other: ASCS personnel, Cooperative Extension Agents, friends or fellow Landowners, and forestry consultants.

owner's attitudes of controlled burning (Table 5). Those PNIF owners with negative attitudes of controlled burning expressed a distrust of industry foresters as information sources for controlled burning and cited county Cooperative Extension Service agents three times as often as their preferred information source.

When landowners were asked to whom they would most likely go for information if they had a specific question on forest management, there was a slightly greater range of sources given (Table 6). Again, Alabama Forestry Commission foresters were the most frequently cited information source. There were no significant differences in the proportions of forest management information sources cited when compared with owner's attitudes of controlled burning

Private, non-industrial forest owners in the Wircgrass Region of Alabama who perceived controlled burning as a useful forest management activity on their forestland conducted or contracted a variety of forest management activities (Table 7).

The crosstabulation indicates that PMF owners were more likely to conduct the following forest management activities when they considered controlled burning useful than those who did not perceive it useful: planting trees, establishing wildlife food plots, selling timber, using herbicides, preparing a forest inventory, using a written contract to sell timber, and constructing a road on their forestland Individual PNIF owner.3 who perceived controlled burning useful undertook 35 percent more forest management activities than those owners who did not feel controlled burning was a useful forest management activity. While the Chi-square statistic indicates a significant relationship at the 0.10 level between perceived utility of controlled burning and the number forest management activities undertaken by PNIF owners, the number of crosstabulation cells with fewer than five cases is too large to accept Chi-square as a valid test. However, the lambda statistic indicates that knowledge of PNIF owner's perceptions of controlled burning is a significant factor in predicting their use of other forest management aclivitics.

Table 6. Forest management information sources identified by PNIF owners of the Alabama Uiregrass Region by attitudes of owners toward controlled burning (N=145).

Sources	Attitudes			
	Negative	Neutral	Positive	
AFC forester	21	21	10	52
	[illegible]	34.4	27.6	35.9
Industry forester	16.7	5	3	16
		6.2	6.3	11.0
Other'	19	35	23	77
	39.5	57.4	63.9	53.1
	48	61	36	145
	100.0	100.0	100.0	100.0

Chi-square: 6.255, d.f. = 4 Significant at alpha = 0.10

'Other: ASCS personnel, Cooperative Extension Agents, friends or fellow landowners, and forestry consultants.

Table 7. Forest management activities conducted by PNIF Owners in tbe Alabama Uiregrass region by perception of controlled burning as a useful forest management activity on their forestland (N=226).

Activity	Useful.	Not Useful	
Planting trees	33	6	39
	18.3	14	17.2
Wildlife plots	38	30.4	52
	21.1		23.0
			56
Selling timber	43.3	36.4	24.8
Herbicide applications	14	4	18
	7.8	8.7	8.0
Forest inventory	14	2	16
	7.8	4.4	7.1
sales contract	26	4	30
	14.5	8.7	13.3
Road construction	13	2	15
	7.2	4.4	6.6
Total	180	46	226
	100.0	109.0	160.9

Chi-square: 4.7479', d.f. = 6 Significant at alpha = 0.10 Lambda: 0.2860

'The number of cells in the crosstabulation with expected frequencies of less than five is greater than 20 percent. This condition does not allow statistically valid Chi-square tests.

SUMMARY

While studies of fire impacts began in the early 1900s, there appears to have been a hesitancy to study man's behavior in using fire. This investigation indicates that a significant proportion of PNIF owners in the Wiregrass Region of Alabama see controlled burning as a useful forest managcmcnt activity in meeting a variety of ownership objectives. Further, nearly one-third of the PNIF owners first become aware of controlled burning utility through some personal experience rather than any media transmitted message. Regardless of PNIF owner's attitudes and perceptions of controlled burning, the Alabama Forestry Commission is the most frequently cited formal information source of controlled burning. Those owners who perceive controlled bunting as a useful forest **management tool** tend to be more active forest managers relative to those owners who do not feel controlled burning is useful.

CONCLUSIONS

Controlled burning is a very visible forest management activity whose impacts can be observed in a short time period relative to many other activities. It is also suited to use on a variety of size scales, which is important when considering the range of ownership sizes or individual stand sizes found on PNIF lands. State forestry agencies can utilize their standing as controlled burning information sources as an opening to promote other forest managcmcnt activities by PNIF owners who use controlled burning. Neighboring forest owners could similarly be encouraged to undertake more forest management activities through observation of other's activities. Controlled burning can therefore serve professional foresters as a communication tool by promoting increased contacts with PNIF owners. Through a better understanding of adoption and diffusion behaviors of private, non-industrial forest owners, professional foresters can become more proactive in their forest management information and assistance efforts with this public. Controlled burning users can be developed as informal change agents and secondary information sources by state forestry agencies, thus utilizing interpersonal networks within PNIF owner publics. This diffusion network approach by professional foresters would increase the effective targetting of information desired by PNIF owners for their specific management objectives. A consequence of this would be that PNIF lands would produce more of their potential ecological, economic, and social benefits.

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SOCIAL IMPACT OF COMPUTERS IN URBAN FIRE FIGHTING

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Abstract-Computers are rapidly expanding into the urban fire safety area. This paper presents some social implications caused by the use of computers for fire safety databases, arson prediction programs, and fire simulation programs. In regards to the new technological advances this paper raises questions concerning the pros and cons of how computers are being used, who is responsible for the technology, who decides how the technology is used, how does the new technology affect insurance rates, how does it affect the builders and owners, and who is liable for personal damage indirectly caused by the new technology.

INTRODUCTION

With rapid advances in computers and software, new technology is being employed in the area of fire safety. This paper discusses the social impact of this new technology, addressing social issues such as how computers are being used, who is responsible for the technology, who decides how the technology is used, how does the new technology affect insurance rates, how does it affect the builders and owners, and who is liable for personal damage indirectly caused by the new technology.

Current Technology

The use of computers in the fight against urban fires has included diverse areas such as resource allocation, planning, dispatching, training, inventory management, etc. This paper concentrates on the implications of computer usage in three main areas; computer databases, models to predict arson, and models to predict fire movement contained in structures. Computer databases are being designed to provide the fire fighters with more information about the area in which the fire is contained; the models to predict arson are being developed to stop the loss of property and lives which occurs each year due to deliberate fires; the models to predict fire movement are being designed to help risk managers determine the safety level of their structures.

Computer Databases

Probably the widest application of computers to the fire safety field has been in the area of databases. The National Fire Information Retrieval System (NFIRS) has a database for hazardous materials (HAZMAT) which contains information like the chemical name, its flashpoint, upper and lower explosive points, water miscibility, etc. Another database maintained by the Insurance Services Office (ISO) contains details on commercial and public buildings throughout the country. The National institute of Standards and Technology (NIST) Center for Fire Research (CFR) has developed a computerized card catalog (FIREDOC) containing localized information about buildings and materials which can be retrieved by local fire departments (Watts 1987).

In Leominster, Massachusetts, the fire department uses software entitled CAMEO to store information about the location and nature of hazardous materials in the industrial areas of the city. The database contains information such as the properties of the hazardous material, health hazards, first aid measures, protective clothing, etc. While approaching the scene, fire fighters can be informed about the material by the dispatcher (Bisol 1989).

In Phoenix. Arizona, the fire department has gone one step further. They work with a sophisticated Computer Aided Dispatch (CAD) and Mobile Digital Terminal (MDT) system to increase their ability to more efficiently respond to the needs of the public. The system supports over 1.4 million people in Phoenix. Glendale, and Tempe. Each emergency vehicle is quipped with a computer system which can communicate to the main computers at the fire station. MDT enables them to display emergency data on their screen as they rush to the scene of the incident, sometimes even helping them to find the exact location of the emergency (Sawyer 1984). The Phoenix Occupancy Activity Reporting System contains information from building inspections, providing them with information such as the owner, number of occupants (calling out elderly or handicapped persons), fire code violations, hazardous materials, floor plans, etc. which enables the arriving fire chief to better position the fire apparatus to contain the fire (Sawyer 1984).

With CAD and MDT, communications have improved tremendously; messages go more quickly. For example, when a fire box alarm was triggered, the dispatcher previously searched the fire box catalog manually to determine the street address and nearest fire station. With CAD, the computer system handles this operation at a much higher speed. Furthermore, in a major fire up to sixteen pieces of equipment need to be dispatched. Using radio communications this would take ninety-six messages to dispatch the equipment and put it back on line. With the new system, all communication is done by computer (Sawyer 1984). Dispatch operators can now keep up with the calls during peak periods. Equipment is being put to more efficient use. Record keeping and data collection for

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management reports has been automated freeing the individuals lo focus on other areas. The safety of the public and fire personnel has been improved due to increased knowledge of the structure and its contents (people and material). The overall success of the system is attrihutcd to the fact that the development team members worked very closely with the fire fighters. The fire lighters' experience enabled the developers to better understand what was required of the system to help the fire fighter; continuous feedback has helped to constantly improve Ihc system. Using the computer has become common place for the new generation of fire fighters in Phoenix (Sawyer 1984).

Arson Prediction Programs

Each year arson causes thousands of deaths and injuries nationwide as well as property loss in the billions of dollars (Cook 1985). Arson-prevention workers, backed by insurance agencies, city officials, and community organizations, have applied computer technology to the fight against arson and have found that they can reduce the losses. The idea behind the effort is that certain economic, structural, and demographic characteristics of a building and its neighborhood could he connected to arson rates. Generally, owners who have a building occupied by low income minority individuals, with long term leases, at very little rent, have a higher incidence rate of using arson as a means of clearing the building for renovation.

Arson prediction models were constructed so arson-prevention workers could use the results of the model to target specific buildings which arc at high risk for arson. By knowing where arson is most likely to occur, the arson-prevention workers can focus their efforts (fire marshal monitoring to tenant organizing) more cflicienlly on preventing arson in that area. The computer has enabled arson-prevention workers to do what they could not do before; "store and analyze the large data sets necessary to pinpoint the buildings most likely to bc torched" (Cook 1985). Using databases from various agencics, which arc available to the general public, the arson prevention workers are able to combine information about a particular building; this information is then used to determine the possibility this building will be victim to arson. Some problems were experienced due to the dissimilar manner in which the various organizations stored data in their database. For example, the fire department identified fires by addresses which were not always accurate; Ihc housing departments used block and lot numbers.

In 1979, the Boston-based Urban Educational System Inc. developed a computer program to identify buildings that arc prone to arson. The variables used by the program include fire history, code violations, and tenant complaints. The program enabled the Attorney General at that time, Francis X. Bellotti, to prosecute arson-for-profil operations with unprecedented success. The program has helped the prosecution through better data collection of prior fire history, code violations, etc. (Anon 1979).

Using the work performed in Boston as a starting point, workers in arson prediction in New York City created a program called the Arson Risk Prediction Index (ARPI). APRI is mainly a formula which takes into account variables, and weights associated with those variables, to produce a probability that the building will be victim to arson in the next year. Variables include building type, building location, vacancy rate, fire history, serious code violations, number of apartments, owner history, etc. (Dillenbeck 1985). The computer program was not only employed to predict which buildings were at high risk to arson but the arson-prevention workers were also able to discover patterns among owners and problem buildings. This capability enables arson-prevention workers to discover building owners who make a practice of burning their buildings for profit.

The experience gained by the New York City arson-prevention workers has shown that although the program can help narrow the scope of building which are prone to arson, the model is not perfect. Many of the buildings which the program put at high risk did not bum. In fact, when the formula was applied city-wide only 40 percent of the buildings selected were burned. However, when the formula was modified for specific neighborhoods, it was accurate 80 percent of the time. Such results indicate that this information might bc best used to identify the buildings where minimal prevention methods, such as sending warning notices to the occupants, should be employed (Cook 1985).

Fire Simulation Programs

Before the use of computers, engineers approached fire safety risks by looking at the codes. Codes arc standards of practice developed by consensus among engineers and other tire experts. The codes were based on very little actual calculation of fire hazard; therefore, construction based on these codes could not be accurately evaluated as far as risk during a fire. Most observations of fire movcmcnt were done by investigating accidental fire or from experiments conducted by various organizations. Fire safety engineers in lhcsc organizations ignited a lot of materials and watched how they burned. This was costly and time consuming especially when trying different configurations of material. In 1971 a group of scientists, observing that fire is rigidly bound by physical laws, concluded that fire is a rational physical phenomena which follows a logical system (Fitzgerald 1983). Based on this premise, fire safety engineers started to develop models to predict the spread of fire through a building.

There are three aspects to analyze during a fire; fire movement, smoke movement, and people movement. The goal of a computer simulation is lo analyze these movements to identify the acceptable time interval between when an occupant is aware of a tire and when the exits arc blocked because of fire or smoke, thus, providing a tool to "understand, evaluate, and describe the expected performance and reliability of the fire safety systems" (Fitzgerald 1987).

Fire simulations can be modelled either as zones or networks. Zone models work by dividing the structure into several zones, defined by room boundaries. "Advantages of zone models arc that they provide fast, reasonably accurate, modular code that runs on a personal computer (PC), they arc fairly simple to update and extend, and can bc used by non-experts. The disadvantages arc that the model is two-dimensional, only works with simple room geometries, and uses empirical knowledge, which introduces further limits on its applications" (Anon 1989).

There are two major advantages to providing fire simulation programs which run on PCs. First, most organizations can afford a PC; therefore, they will be more inclined to use the software. Second, PCs arc portable and can go whcrcvcr the user needs to take it. For example, fire engineers could carry the computer around as they examine the structure and input information immcdiatcly. The PCs could be installed on fire trucks and tied into the main system so the fire chief can obtain information on the structure and run analyses of the spread of the fire.

A disadvantage to zone models is the usc of empirical knowledge throughout the simulation. Input data to the program is generally provided by a fire engineer who has inspected the structure. These values arc the only data used by the fire simulation program to produce the results. The assignment of these values arc purely expert guesses based on common training given the fire engineers. A current problem is the inconsistency of the numbers produced by the fire safety engineers; unique buildings require unique judgemcnt which varies from one person to another. The credibility of the numbers can be increased by providing the analysts with a "decision support system" or "engineering support system" which will help them produce the more consistent numbers for the input data. The challenge to creating tbcsc expert systems is programming the "judgcmcnt" of the cnginecrs.

Fire safety engineers analyze fire movement by looking at the various input data; for example, room contents and barrier strengths, to determine when these factors come into play and how much they influence the outcome. A graphical, or neural, network of inputs (contents) and outcomes (fire propagation) could be produced. The next generation neural network-based fire simulation program will be able to learn how fire spreads by observing constructed fires and learning the properties of fire movement much like an experienced fire safely cnginccr.

SOCIAL IMPACT

Some scientists believe that true answers to scientific problems lie in an analytical equation; numeric answers are only approximations. Today computer simulations arc dealing with problems with so many variables that the "true visualization requires a numerical solution; the analytical equation is only an approximation" (Koshland, 1985). The fire simulation programs take into account so many variables (i.e. room fuel, barrier strength, etc.), and there are many it does not take into account (i.e. explosions, wind directions, etc.) that the model may bc entirely different from the actual outcome. The arson prediction program has shown that arson patterns cannot be truly summarized in a mathematical formula.

"When decisions are based on the results of computer models, it is extremely important that we can trust the model's hardware, software, arid input data" (Perrolle 1987). All three technologies are negatively affected by bad input data. In the fire simulation program, all the values for the input data arc provided by the fire engineers. Therefore, interpretation may be different among various fire engineers resulting in the same structure producing different results based on the input data. Additionally, these results may all bc different from what actually happens when the structure is on tire due to the absence of a critical factor in the program. For example, the fire simulation model relies on the fire suppression systems (e.g. sprinklers) working perfectly; however, in reality, complex systems have a tendency to malfunction and break down. If the fire suppression systems fail in an actual fire, the results of the model arc incorrect. In the arson prediction program, diverse databases resulted in information not being matched with the correct structure; therefore, the predictions for arson in that structure were incorrect. In the computer databases, mistakes can happen at many stages. The data collected may contain errors; the data may be entered incorrectly; the joining of diverse databases may be done incorrectly. All these factors cart contribute to an erroneous database which could result in mishaps at the scene of the fire. Another factor impacting portable computers, such as those used by fire fighters in Phoenix, is the failure of the hardware. On site, equipment will be exposed to excessive heat, smoke, waler, and chemicals. Equipment failure in these harsh conditions must bc prevented. Back at the station, down-time on the mainframe should be minimized since it would impact the flow of information to the on-site fire fighters.

All these factors contribute to the possibility that the programs can be very inaccurate or the systems can fail at critical times. When the new technology does not perform as expected the public, builders, occupants, fire fighters, insurance agencies, etc. are all affected The following sections describe effects that computer technology has had on thcsc groups of individuals.

The Public

As society's structures become larger and more complex the fear of an extensive fire where massive human lives are lost causes the public to develop what Perrow called dread risk. "Dread risk is associated with

- lack of control over the activity,
- fatal consequences if there were a mishap of some sort,

- high catastrophic potential,
- sanctions of dread,
- inequitable distribution of risks and benefits, and
- the belief that risks are increasing and not easily reducible" (Perrow 1984).

People perish in buildings year after year because Row combustible materials used in construction and furnishings. Incidents occur weekly which show that the current codes arc inadequate. The public, spurred on by such incidents as the MGM Grand Hotel fire on 21 November 1980 which killed 84 people and injured 679, wants to bc better protected from accidental and deliberate fires, especially in high-rise buildings, nursing homes, and hospitals.

The public's concern have caused the introduction of stricter fire codes as well as the desire to test building designs for fire safety before construction starts. Now, the current problem has shifted into the other direction; the public trusts that the new technology will solve the problems Caused by other modern conveniences, such as high-rises, condominiums, etc.

The public has come to depend on technology to provide a comfortable standard of living, yet the public does not consider the health, safety, and environmental consequences that accompany this new technology. Additionally, along with more accurate techniques for measuring risk comes increased public expectations for safety. The public expects that builders will implement all safety measures outlined by the engineers using the computer technology (Zuckcrman 1989).

The fire prediction programs and other uses of computers in fire safety have increased the level of protection that the general public has in fires but has not yet eliminated the possibility of major fires and accidents.

Software Engineers

"Public awareness of the risks of modem technology spawned vast amounts of new regulations and laws helping to create a litigious society, a society in which juries arc inclined to award large settlements to an individual no matter who is at fault" (Zuckcrman 1989). A current issue being debated in the courts is "If the simulation is wrong, and because of this deaths and injuries occur, who is at fault; the dcvelopcrs, the people who supplied the input, the users?" Perrolle points out that "If computer software is considered 'goods' under the law, software suppliers could be held liable for damages caused by program errors" (Perrolle 1987).

If the company that produces or uses fire technology gets sued they will most likely discontinue development and use of thcsc products. Many companies decide not to pursue ideas because of the threat of liability partly due to the lawyers who promote the idea that whenever there is an accident someone must pay. The technological advances in the fire lighting area have been, and will continue to bc, slowed by the courts

Another issue concerns how much the company that produces thcsc prediction programs should charge for the technology. In an ideal society cvcryonc would have equal access to the programs, not just those who can afford them. Volunteer fire departments, which arc funded entirely by local citizens, would not be able to afford the hardware and software costs to install these systems. However, if they were made available to the general public at a low cost, there is no means to prevent the malicious misuse of the technology.

Fire Safety Engineers

The intent of the fire safety cnginccrs is to better protect the public caught in a fire and the fire lighters combatting the inferno. Armed with bcflcr analytical techniques, fire engineers can tell architects and designers how to create structures that will contain fires. If fire safety engineers arc constantly coming up with new methods to contain fires and using the prediction programs to prove that building codes should bc stricter, old buildings can never meet current code requirements. Additionally, the time required to change fire codes exceeds the rate at which new standards arc being dcvclopcd so fire codes cannot be kept up to date. Thcrcforc, when a case appears before a judge, he must decide what code standards the case will be based on.

Additionally, the prediction programs arc having a negative effect on the fire safety profession. Previously, the codes were set by consensus among the fire safety engineers. Now, by running the fire simulation program against structures which have recently burned, fire safety engineers can determine where better codes could have reduced the damage done by the fire. Therefore, the codes arc being set by the results of the fire simulation program and not by the fire safety engineers. This may result in fire safely engineers viewing the simulation programs and expert systems as decreasing their status because this new technology is taking away their judgement.

The attitude of the fire safety cnginccrs towards the new technology may severely impact the work being done in the expert system area. The new technology needs to be introduced into their profession as a set of tools to enhance their job not replace them. Even if the fire safety engineers accept the technology and assist in successfully completing the expert system work, the next generation of fire safety engineers must be careful to scrutinize the computer programs to catch mistakes; the fire safety engineers still need to be instructed in the older methods so they can make judgement calls in situations where the expert system cannot make a decision. A future problem facing fire safety engineers is the work load resulting from pressure to examine more and more structures. As more pressure is put on the fire engineers to examine more buildings, the chance that they overlook an important factor increases. However, the "decision or engineering support systems" will help to reduce the number of mistakes made during inspections.

Fire Fighters

Today, tire lighters are seeing the slow emergence of computers into their field. In Phoenix, the success of the new computer system can be directly related to the input supplied by the fire lighters during the design stage of the system. In a profession which is in dire needs of new recruits, members are hoping that the glamour of the computer technology and the promise of increased safety as a benefit of the technology will entice younger people to consider the profession. However, the new technology presents the fire fighters with other issues. The next generation fire fighter must not only understand the older methods but must also be computer literate causing problems finding qualified individuals. The younger fire lighters, dependent on the new computer system, need to still be taught the old system in case of system failure. The old techniques cannot he totally abandoned since lives may be threatened when the computer system is unavailable.

New issues concerning responsibility are being presented to the next generation of fire fighters. For instance, let's assume the building owner moves something into the building after the data for that structure was collected or the simulation program erroneously predicts the path of fire propagation. Both of these errors may cause the fire fighters to combat the fire differently which could result in fire burning out of control in a situation when it would normally be contained quickly. In incidents such as these, who is responsible for loss of life and property, the person who developed the computer system, the person who collected the input data, the person who entered the data, the fire chief who believed the output? Prior history has shown that the fire chief would most likely be blamed; in fact, the computer simulations might cause incidents of second guessing the fire chiefs decisions by proving that if mother step had been taken the fire would have been controlled. The fire fighters need to be aware that the technology is not flawless and they should not rely on it as their sole source of decision-making criteria. They need to find a balance between what their experience tells them and what the computer tells them to combat the fires.

Builders and Owners

Arson prediction models, and the databases used by them, arc considered "an invasion of privacy" to most building owners. On the other hand, the occupants are glad to see measures taken to protect their homes. Since most owners arc the ones who burn their buildings for profit, they feel their privacy is being invaded by this technology which enables insurance agencies, who do not want owners to collect money by burning their buildings, to track their prior history. Knowing they are being tracked may deter them from burning their own buildings.

Fire simulation models are beneficial in assessing designs and developing more flexible and cost effective fire safety practices. The United Kingdom's Safety and Reliability Directorate (SRD) uses the model routinely in safety assessments; the U.S. Coast Guard uses fire simulation models to improve ship designs in an effect to better protect their crews. Although, the program can be used to detect a serious problem at the design stage instead of the building stage, where it may be too late to make any changes, the strength lies in analyzing current buildings and making improvements to those buildings. By examining structures at the design stage, the fire simulation program enables one design to be compared to another to point out the weaknesses in the performance of some fire prevention systems. However, the program does not force the builder to implement the appropriate design. Most building owners, armed with the programs, can make decisions to determine whether they should risk lives (currently valued at $250K/person) during a fire or spend extra money at construction time to install better fire detection and suppression systems.

If the builders and owners decide to put in a better fire detection and suppression system, any additional money spent on these systems can, and probably will be, passed on to the public in higher costs, fees, and taxes. The builders and owners need to determine just how much the public is willing to spend to be better protected in buildings such as hospitals, schools, nursing homes, high-rises, etc.

Corporate Risk Managers

A corporate risk manager's job is to decide whether to buy insurance or protect the corporate risk through some other means. The corporation views the purchase of insurance as risk. Most companies don't have risk management; they handle it by buying 100 percent insurance. With the introduction of new technology, many companies wishing to decrease money spent on insurance will create new positions for the risk manager or make existing positions more prevalent.

By using decision trees, expert systems, databases, and prediction programs, the modem high-tech risk manager has the ability to determine how much is at risk in the corporation due to a fire. They can dctcrminc what they feel is acceptable risk and modify their existing buildings appropriately or purchase more insurance.

Insurance Agencies

Most insurance companies do not have technological engineers; they rely on history to provide them with the information necessary to determine rates. Faced with major losses due to structural fires, insurance agencies will bring computer technology into their office to help protect their interests. They arc being forced to hire engineers on staff to inspect buildings being insured and set up computer systems which allow them to tap into the same information and databases usecl by the arson prediction program dcvclopcrs

Insurance agcncics arc faced with some tough ethical issues. The insurance agency is a business which must bc profitable; thcrcfore, most insurance companies will not insure a building which the programs state arc either likcly targets for arson or will have massive damage if started on fire. The insurance companies would probably cvcn review older policies to determine if any should bc cancelled for these same reasons. Furthermore, **the** technology will **be** used to justify **higher** insurance rates. The insurance company can simply point to the output of the program to "prove" the building is at high risk.

Armcd with this ncw information and the compute power to process it, the insurance agencies can build a "Bad Building and Owner" database. This database could bc uscd as basis to rcfuse insurance to the individuals who own buildings. If a building/owner is incorrectly placed on this list, mcasures nced to bc put in place to notify peoplc when information about thcmsclves or their building is in the database and how they can gct incorrect information corrected.

CONCLUSION

This paper has focused on the social impact of the new technology being applied to fire safety engineering. At times the new technology seems to have more of a negative impact on society; howcver, thcre are cases wcre the new technology has saved livcs. In Phoenix, the ncw computer system was rcccived positively by all who used it and the rcsult was more cfficicnt response to the needs of the public. In New York City, the arson prediction program helped to dccrease the amount of arson which took place in the Flatbush and Crown Heights communities. The U.S. Coast Guard has increased the safety of thcir **crew** by using fire simulation programs to improve thc fire safety lcvel of their ships.

The intentions of the dcvclopcrs of this ncw technology are rooted in helping the general public to be bcttcr protcctcd from fires. As new technology appears, the developers and sellers must be carcful to cnsure that thc systems arc not blindly trusted or put to the usc for which they were not intcndcd.

Additionally, the public nccds to made aware that technology is neutral; it's how technology is managed that cause problems. Thcrcfore, it's not so much thc information and output of the new technology but what is done with this information that is thc potential problem.

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FIRE’S IMPORTANCE IN SOUTH CENTRAL U.S. FORESTS: DISTRIBUTION OF FIRE EVIDENCE

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Abstract-Evidence of past fire occurrence is estimated to occur on 22.4 million acres, or 26 pcrccnt, of the 87.2 million acres of forests in Alabama, Arkansas, southeast Louisiana, Mississippi, east Oklahoma, Tennessee, and east Texas. Data are drawn from a systematic survey of fire evidence conductcd in conjunction with recent inventories of private and public forested areas in the South Central United States. Some 8.9 million acres that arc estimated to contain evidence of fire are nonstocked or consist of sapling or seedling stands; X.2 million acres consist of sawtimber stands, and 5.3 million acres consist of polctimbcr stands. Fire evidence commonly occurs in forests of the Gulf Coastal Plain and the Ouachita Highlands--in areas dominated by pine and oak-pine forest types and by National Forest and forest industry ownership. Fire evidence is relatively rare in the Mississippi Alluvial Plain, the Boston Mountains of Arkansas, and most of Tennessee--areas dominated by bottomland or upland hardwood forest types and hy a mix of ownerships. Seventy-five percent of the acreage showing evidence of having burned during the past 10 years is associated with wood, livestock, or wildlife production, or with vegetation management; 3 pcrccnt is associated with a natural disturbance. No causal agent has been identified with fires occurring on 22 percent of the acreage.

Comparison with other estimates of annual average firc frequency by State and by potential causal agent suggests that tire frequency estimates based on evidence observations from forest surveys arc crediblc. Given the widespread extent and distribution of fire evidence presented in this report, onc implication is that any changes in fire regulations will have important consequences for forestry in this region. Bccauss survey estimates arc linked with location, forest stand, and tree characteristics, forest survey fire data should prove useful for exploring the relationship of past tire occurrence to regional air quality and wildfirc danger. With the addition of measurements from a subsample of plots, forest survey fire data could be used to assess fire’s impact on the production of water, livestock forage, wildlife habitat, and timber for multi-county and larger areas.

INTRODUCTION

Rccurrcnt fire is essential to many pine forest ecosystems in the southern United States, but there have been few surveys of fire occurrencc over broad geographic areas. State and federal firc control agencies use information about fire occurrcncc to estimate regional smoke hazard and to establish priorities for fire protection in forested areas. Forest resource analysts and othcrs usc such information to assess timber resource conditions and firc-usc practices in selcctcd areas. Fire also plays a role in livestock forage production, wildlife browsc production, carbon storage, leaf litter biomass accumulation, and maintenance of water quality. Models of these multiple values therefore should include fire as a variable that influences the current and projected **status** of forest resources.

In thc South, fire occurrence--whcthcr planned or unplanncd--is not uniformly documcntcd. Wildfires occurring on Southcrn U.S. National Forest land arc recorded by numbcr, causal agent, and total acres affcctcd (USDA Forest Service 1988a). For private and other public land under the protcction of statc forest firc control agcncics, similar wildfire statistics are noted in a separate rcport (USDA Forest Service 1988b). State forest fire control agency wildfire records have been compiled Southwide by county for the periods 1956-1965 (Doolittlc 1969) and 1966-1975 (Doolittlc 1977). Wildfire records for areas not protected by fire control agencies arc not available. Neither are there more recent compilations or more detailed wildfire statistics.

Information on the occurrence of prescribed (i.e. planned) fires across the South is scant. The use of tire as a vegetation managcmcnt tool in southern pine stands has increased during the last 50 years (Williams 1985). Fire is used to prepare a site for stand regeneration, to dispose of logging slash, to reduce hardwood competition, and to limit wildfire hazards. For National Forest land, prescribed fire statistics are listed by total acres and management objective (USDA Forest Service 1988a). Comparable estimates for other public landholdings and private land areas arc not available.

In some fire districts, local fire control personnel know only incidentally of owners with large landholdings where prescribed firc is used, as permission to bum is not required in all States. Many State agencies must rely on “ballpark” estimates gathered from these district personnel, from allied natural resource agencies, from coopcrating forest industries, and from self-administered questionnaires completed by persons in districts that require bum permits. Annual

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statewide prescribed fire acreage estimates are compiled from these data sources (e.g. Ashley 1985, Miles 1985, Moody 1985). However, reliability of these estimates is uncertain (Wade 1985).

Since available fire occurrence data are incomplete at best and anecdotal at worst, they are difficult to compare at multi-county and multi-state levels of resolution. Regional distribution patterns and detailed fire occurrence data by stand-size class, forest type, and acres affected on nonindustrial private ownership are not reported and rarely are collected.

In this report, data about fire evidence and about related measures are presented and discussed. These data were collected during a systematic survey of fire evidence in South Central U.S. forests. Areas included in this survey were: Alabama, Arkansas, southeast Louisiana, Mississippi, east Oklahoma, Tennessee, and east Texas. Data on evidence of fire occurrence in the past decade are presented together with data on evidence of other activities in the past decade so that potential sources of fire, or causal agents, can be at least tentatively identified. These data, and the conclusions drawn from them, are examined for consistency with average annual fire occurrence information obtained from other sources.

METHODS

The Southern Forest Experiment Station's Forest Inventory and Analysis (FIA) Unit conducts an inventory sampling program that assesses the current status of and trends in private and public forest resources. Permanent l-acre plots are located at the intersections of perpendicular grid lines spaced at 3-mile intervals throughout the South Central States. Detailed field observations are obtained in some 17,000 plots on land classified as forest (i.e., at least 1-acre in size, 120 feet in width, and capable of producing crops of industrial wood). When combined with ground-truth of photointerpretation for additional areas, observations from plot samples are expanded statistically to estimate all forest resources in a county, state, or region. Field observations are updated about every 8 to 10 years. Periodic analytical assessments and tabular summaries report the current status and trends in species, number of trees, forested area, and timber productivity. Data are reported by forest type, stand-size class, ownership class, and other characteristics. Further details, including definitions and criteria for classifying forest characteristics, are available in State Resource Bulletins (e.g., McWilliams and Lord 1988, Rudis 1988).

Evidence of past fire occurrence is recorded as present or absent, and consists of physical evidence of bum scars on trees and other objects, reduced litter depth, and other vegetational indicators. Fire evidence also is recorded by age class of the most recent occurrence. For the 1981 survey of Alabama, age class was defined as: recent (1 or 2 years), within 3 years, or 3 years or older. In subsequent surveys, categories have been fine-tuned to limit overlapping ages and establish an open-ended highest age category. The updated categories are recent (1 or 2 years), 3 years to previous survey, and older than previous survey.

Fire evidence observations have been recorded only once for the most recent surveys of each region: Alabama (AL) 1982, southeast Louisiana (LA) 1984[2], east Texas (TX) 1986, east Oklahoma (OK) 1986, Mississippi (MS) 1987, Arkansas (AR) 1988, and Tennessee (TN) 1989. Statistical inferences regarding differences in fire occurrence by forest characteristics are based on chi-square analysis of category frequencies, with significance of chi-square values established *a priori* at the 5-percent probability level.

Evidence of other activities originating since the previous survey are used to suggest whether fires were planned or unplanned. Timber production is recorded if evidence suggests such activity occurred since the last survey. These include timber management activities--site preparation and timber stand improvement-and timber harvest activities-clearcutting and partial cutting. Similarly, evidence of livestock use, game management, and nontimber cutting or clearing, and miscellaneous artifacts associated with human use, are noted as well. (Since age categories used in the 1982 survey did not specify age beyond 3 years, some of the evidence coded then may relate to fires that occurred prior to the previous survey. To avoid confounding by time period, detailed analysis of causal agents includes only data for surveys conducted after 1982.)

For the purposes of this report, fires are classified as "prescribed" where fire evidence occurs in conjunction with evidence of production (timber management or harvest activities, livestock use, game or nongame wildlife management) or miscellaneous activities associated with cutting or clearing (woody debris from noncommercial wood harvest, maintenance of right-of-way). Fires are classified as "wildfire" where fire evidence occurs together with evidence of natural disturbance or salvage operations. "Other agents" is applied to fires in plots with fire evidence and no evidence of production, miscellaneous activities, or wildfire. Detailed observation codes by category are available from the senior author upon request. These and other category and observation codes can be found in Forest Inventory and Analysis (FIA) field manuals (Quick 1980, FIA 1989)

[2] Budgetary constraints in 1983 limited the tally of fire evidence and nontimber activities. A statewide tally is planned for 1991 in Louisiana.

RESULTS

Firc evidence is cstimatcd lo occur on 22.4 million acres, or 26 pcrccnt of the 87.2 million acres of timberland surveyed. Firc cvidcncc and survey year for each state arc as follows: Alabama, 1982, 7.1 million acres (33 pcrccnt of the timbcrland); Arkansas, 1988, 3.4 million acres (20 pcrccnt); southcast Louisiana, 1984, 0.8 million acres (43 percent); Mississippi, 1987, 4.8 million acres (28 pcrccnl); east Oklahoma, 1986, 2.1 million acres (43 percent); Tenncssec, 1989, 1.7 million acres (13 pcrccnt); and cast Texas, 1986, 2.6 million acres (22 pcrccnt). For surveys conducted after 1982, 73 percent of the evidence is associated with fires that occurred since each State was surveyed (approximately 10 years previously).

Estimates of fire evidence in forested areas by county provide regional summaries within and among States (fig. 1). Counties with fire evidence in forests are concentrated chiefly in the Gulf Coastal Plain (south Alabama, Mississippi, and Texas) and the Ouachita Highlands (southcast Oklahoma and west central Arkansas). Fire evidence in forests is relatively rare in the Mississippi Alluvial Plain, the Boston Mountains of Arkansas, and most of Tennessee.

Forest Characteristics

Unless otherwise noted, all differences in fire evidence frequency by forest characteristics are significant, with chi-square values at probabilities less than 0.001.

Area estimates by forest type and stand-size class arc summarized in table 1. Forty percent of the 22.4 million acres estimated to have fire evidence is composed of nonstockcd, sapling, or seedling stands, 37 percent is composed of sawtimber stands, and 24 percent is composed of polctimbcr stands. Fire evidence occurs in all forest types. It occurs in 54 percent of pine plantations, 3.5 percent of natural pine stands, 30 pcrccnt of oak-pint stands, 20 percent of upland hardwood stands, and 6 pcrccnt of bottomland hardwood stands. Within stand size classes fire evidence occurs in 3.5 pcrccnt of nonstockcd, sapling, or seedling stands, 22 percent of poletimber stands, and 22 percent of sawtimbcr stands. Percent of timberland area with fire evidence by forest type and stand-size class is shown in figure 2.

In pine plantations, the proportion of timberland area with fire evidcncc is 57 percent in nonstocked, sapling, or seedling stands, and declines to 48 percent in sawtimbcr stands, a relatively small but significant difference. In natural pine stands, the proportion of area with tire cvidcnce, 35 percent, is not statistically significant by stand-size class. The majority of pine plantation area is in sapling-seedling stand-size class and the majority of natural pine area is in sawtimber stand-size class. Results suggest that fire occurrence remains higher in pine plantations relative to other forest types throughout the life of these stands.

Fire evidcncc occurs more frequently in plantations than in natural pine stands, regardless of stand-size class. Fire evidence is less common in upland hardwoods and bottomland hardwoods than in pine stands. Fire evidence declines as stands mature.

Fire evidence occurs on 41 percent of forest industry land, 28 pcrccnt of public land, 19 percent of farmer-owned land, and 21 percent of nonindustrial private land (table 2). Forest acreage with fire evidence is concentrated in sapling-seedling stand-size class and in forest industry and public ownership. Public land with fire evidcncc is primarily in the sawtimber

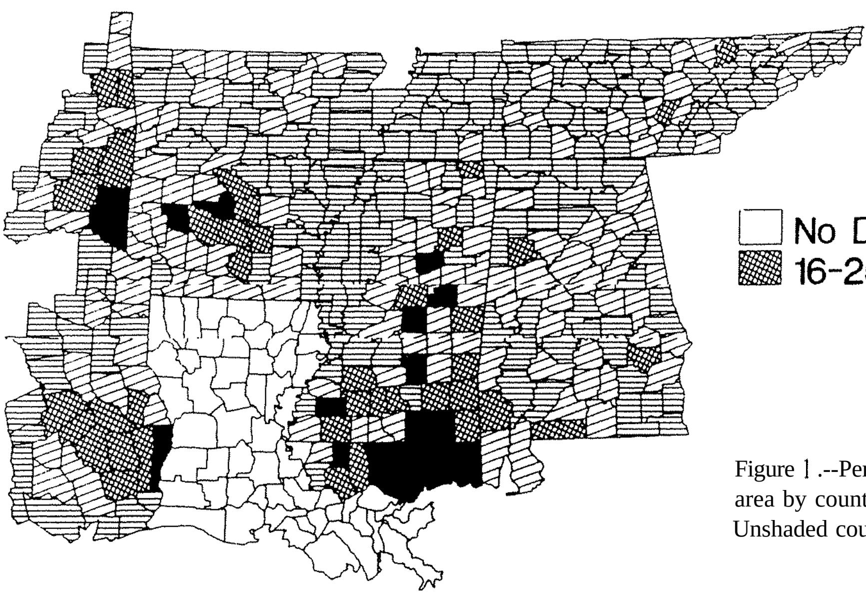

Figure 1.--Percent timberland area with fire evidence per land area by county, South Central United States, 1982-1989. Unshaded counties arc scheduled lo be surveyed in 1991.

Table 1.--Timberland acres with and without fire evidence by forest type and stand-size class.”

Fire evidence and stand-size class	ALL types	Planted pine	Natural pine	Oak-pine	Upland hardwood”	Bottomland hardwood
	Million acres					
ALL stand sizes						
With fire evidence	22.43	3.51	6.00	4.83	7.25	0.75
No fire evidence	64.19	2.99	10.94	11.08	28.95	10.82
Total	81.22	6.56	16.94	15.92	34.22	11.57
Sapling-seedling”						
With fire evidence	8.92	1.98	1.03	2.42	3.25	.21
No fire evidence	16.45	1.49	2.00	3.27	8.14	1.58
Total	25.37	3.48	3.02	5.69	Ii.40	1.78
Poletimber						
With fire evidence	5.30	1.06	1.27	.91	1.88	.19
No fire evidence	19.16	.91	2.38	3.22	0.09	2.59
Total	24.48	1.96	3.64	4.13	1.96	2.78
Sawtimber						
Uith fire evidence	8.20	.53	3.70	1.49	2.12	.36
No fire evidence	29.16	.59	6.57	4.62	10.74	6.66
Total	31.31	1.12	10.27	6.11	12.86	7.01

[a] Rows and columns may not sum to totals due to rounding
[b] Includes 67,000 acres classified as nontyped.
[c] Includes 341,000 acres classified as nonstocked.

stand-six class and in natural pine and oak-pine forest types. Half of the forest industry land with fire cvidcnce is in nonstockcd, sapling, or seedling stand-six class. Forest industry land with fire cvidcnce is relativcly cvenly distributed ovcr planted pine, natural pine, oak-pine, and upland hardwood forest types. Farmer-owned land with fire evidence is evenly distributed by stand-six class; acres are primarily in upland hardwood forest type. Nonindustrial private land with fire cvidcnce is evenly distributed by stand-six class; acres arc concentrated in upland hardwoods, natural pine, and oak-pint stands.

Location

If one is to correctly interprct regional patterns of fire occurrence in forested areas, one needs to consider the arrangcment of forests and adjacent nonforcst areas. The forested urban-wildland interface, i.e., the forested land adjacent to urban areas, is of considerable interest in fire scicncc. Memphis, Houston, Little Rock, Mobile, and Birmingham represent the major urban centers in the South Central Slates survey area. Howcvcr, the Present sampling schemc is inadcquate to categorize urban influences; few sampled plots occur in this area.

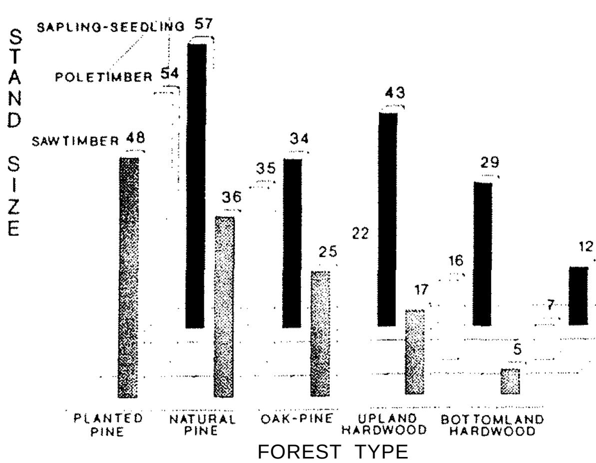

Figure 2.--Percent timberland area with fire evidence by forest type and stand-size class.

Table 2.--Forested acres with fire evidence by forest type, stand-size class, and ownership class[a]

Forest type and stand-size class[b]	Ownership class: All owners	Public land	Forest industry, incl. leased	Farmer	other private
	. . . . Million acres				
All forest types					
Sapling-seedling	8.92	0.57	4.15	1.24	2.95
Poletimber	5.30	.49	1.74	.96	2.11
Sawtimber	6.20	1.51	2.66	1.56	3.19
Total	22.43	2.57	7.66	3.70	8.25
Planted pine					
Sapling-seedling	1.98	.07	1.58	.08	.25
Poletimber	1.66	.03	.82	.05	.16
Sawtimber	.53	.08	.21	.06	.17
Total	3.57	.18	2.61	.19	.58
Natural pine					
Sapling-seedling	1.03	.13	.32	.18	.40
Poletimber	1.27	.18	.47	.16	.45
Sawtimber	3.70	.86	1.69	.45	1.30
Total	6.66	1.18	1.88	.79	2.18
Oak-pine					
Sapling-seedling	2.42	.18	1.20	.25	.79
Poletimber	.91	.13	.21	.16	.42
Sawtimber	1.49	.31	.36	.29	.53
Total	4.63	.62	1.76	.71	1.74
Upland hardwood[c]					
Sapling-seedling	3.25	.18	1.66	.68	1.46
Poletimber	1.88	.14	.21	.53	1.01
Sawtimber	2.12	.23	.24	.60	1.05
Total	7.25	.54	1.44	1.82	3.46
Bottomland hardwood					
Sapling-seedling	.21	.01	.06	.04	.10
Poletimber	.19	.01	.04	.06	.07
Sawtimber	.36	.03	.10	.09	.14
Total	.75	.05	.20	.20	.31

[a] Rows and columns may not sum to totals due to rounding.

[b] Nonstocked areas (89,000 acres) are included in sapling-seedling stand-size class.

[c] Includes 26,666 acres classified as nontyped.

The largest non-forested areas in the South Central States include the extensive farmland acreage along floodplains of the Mississippi and Arkansas Rivers, the Blackbelt Prairie crescent that stretches from north central Mississippi to central Alabama, and the Central Basin of central Tennessee and north Alabama (fig. 3).

Areas dominated by southern pine forest types contain the largest concentration of forested plots with fire evidence. Clusters of plots with fire evidence, particularly those of relatively recent origin (triangles, fig. 3), indicate areas where fire has played an important regional role in forest ecosystem dynamics. Although detailed geostatistics are needed to verify the significance of spatial patterns, the density of forest industry landholdings and pine-dominated public timberland (Rosson and Doolittle 1987) appears directly related to the density of fire evidence.

Hots with evidence that fire occurred since the previous survey (triangles, fig. 3) are to be distinguished from plots with evidence of older fires (circles, fig. 3). A visual inspection of patterns suggests that, on average, areas with historical fire evidence also contain more recent fire evidence.

A careful examination of patterns formed by plots with older fire evidence can suggest additional hypotheses. Geostatistical analysis, coupled with a geographic information system and knowledge of historical fire occurrences, should be helpful in further investigation of pattern differences.

Causal Agents

Approximately 314 of the acres with evidence of fire since the previous survey (excludes the 1982 Alabama survey) also contains evidence of activities associated with production of timber, wildlife, or livestock, or with miscellaneous forms of vegetation management. Known wildfires are noted on 3 percent of the acres. Fire evidence not clearly associated with a causal agent occurs on 22 percent of the acres (table 3). Figure 4 shows percentage of acres with fire evidence by causal agent and ownership class. Timber production dominates on forest industry land (81 percent), and animal production is highest on farmer-owned land (30 percent). Timber production dominates in sapling-seedling stands, and is high for both poletimber and sawtimber stand-size classes (fig. 5). It is notable that the percentage of sapling-seedling acres in the "other" causal agents category is significantly smaller than the percentages of poletimber or sawtimber acres in that category.

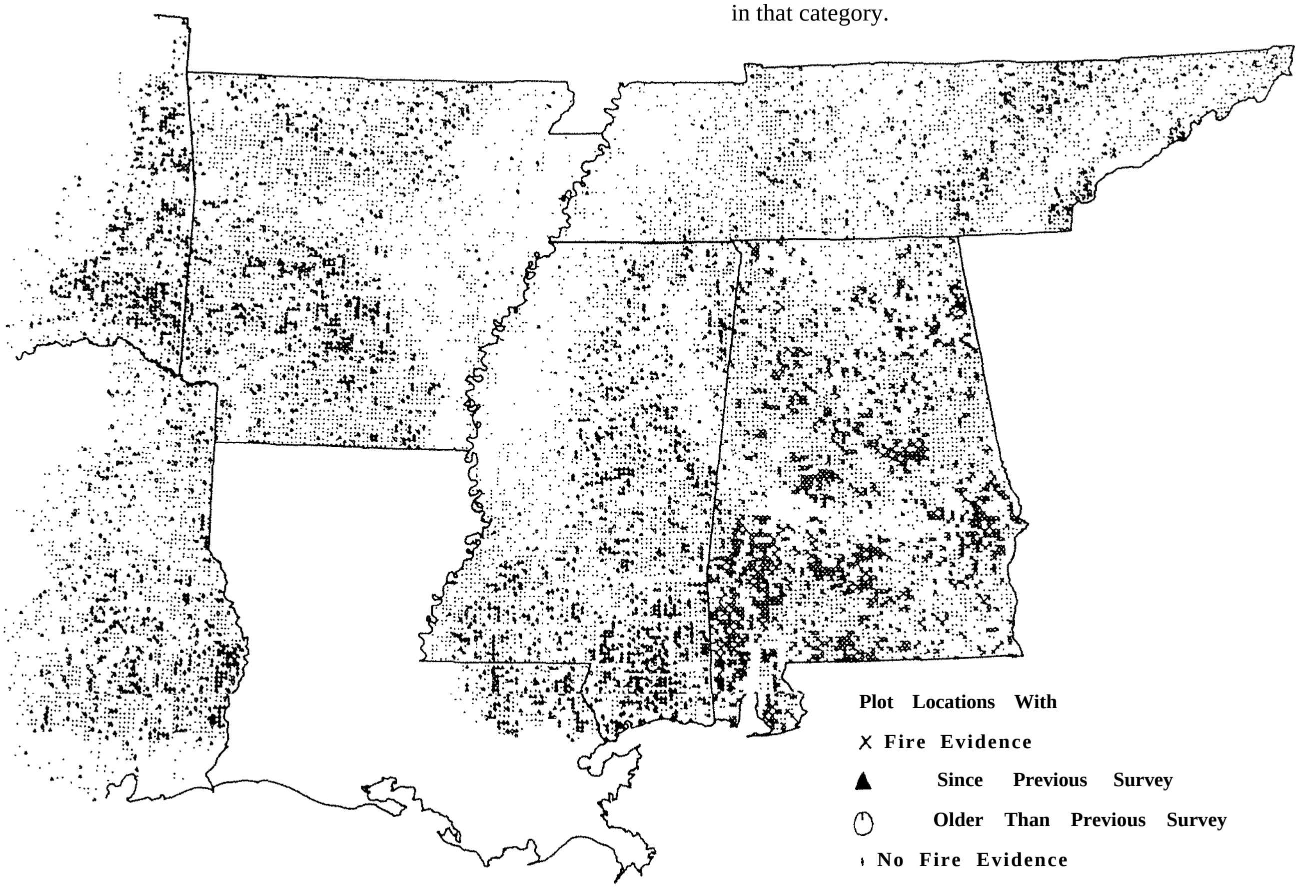

Figure 3.--Timberland area surveyed with and without fire evidence

Table 3. --Area with fire evidence by stand-size class, ownership class, and potential causal agent.[a]

Stand-size class and potential causal agent	ALL owners	Public land	Forest industry, incl. leased	Farmer	other private
	Ownership class				
	. . . . Million acres				
ALL sizes					
Timber production	5.74	0.83	2.95	0.46	1.50
Timber and Livestock or wildlife production	1.67	.14	.97	.18	.37
Livestock or wildlife production	.68	.06	.13	.26	.22
Miscellaneous cutting or clearing	.30	.06	.08	.06	.09
Other	2.44	.41	.41	.42	1.20
Natural disturbances	.39	.05	.08	.09	.17
Total	11.22	1.56	4.63	1.47	3.56
Sapling-seedling[b]					
Timber production	2.68	.21	1.66	.21	.66
Timber and Livestock or wildlife production	.91	.06	.62	.08	.15
Livestock or wildlife production	.24	.01	.05	.09	.08
Miscellaneous cutting or clearing	.07	.02	.02	.02	.01
Other	.54	.06	.08	.12	.28
Natural disturbances	.19		.04	.05	.10
Total	4.93	.36	2.76	.57	1.30
Poletimber					
Timber production	.93	.09	.43	.10	.31
Timber and Livestock or wildlife production	.24	.01	.14	.02	.06
Livestock or wildlife production	.17	.01	.05	.07	.05
Miscellaneous cutting or clearing	.06		.03	.01	.02
other	.74	.13	.16	.10	.34
Natural disturbance	.07	.02	.01	.01	.02
Total	2.20	.25	.83	.31	.80
Sawtimber					
Timber production	1.83	.54	.62	.15	.53
Timber and Livestock or wildlife production	.52	.07	.22	.08	.15
Livestock or wildlife production	.27	.04	.03	.11	.09
Miscellaneous cutting or clearing	.17	.05	.04	.03	.06
other	1.16	.22	.17	.19	.58
Natural disturbance	.14	.03	.03	.03	.05
Total	4.69	.95	1.10	.59	1.46

[a] Excludes Alabama (7.1 million acres) and other surveyed states with fire evidence older than the prior survey (4.1 million acres).
[b] Includes 53,666 acres classified as nonstocked.

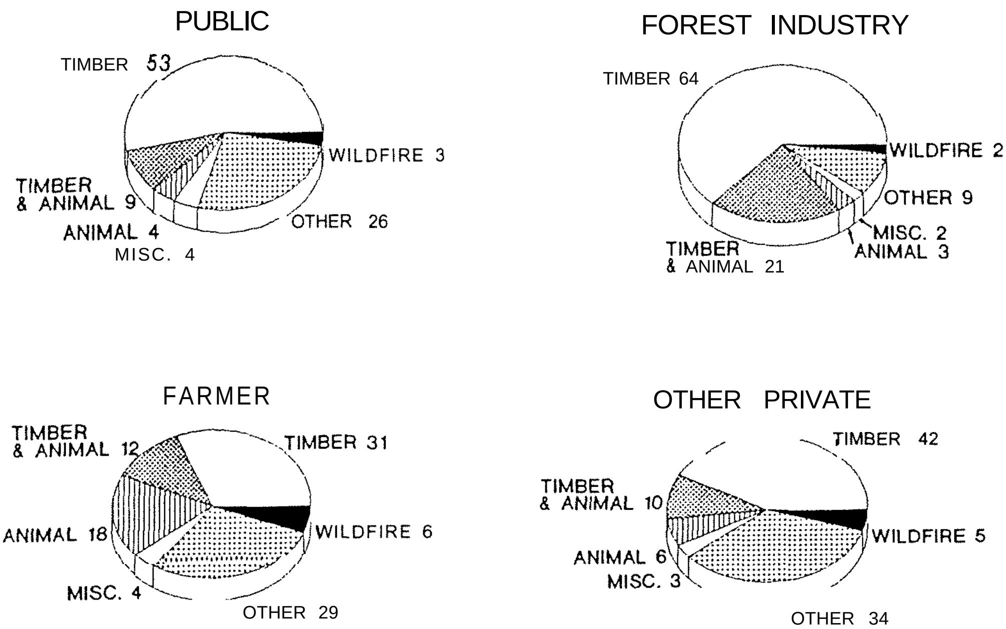

Figure 4.--Percent timberland area with fire evidence since the previous survey by ownership class and potential causal agent (excludes Alabama),

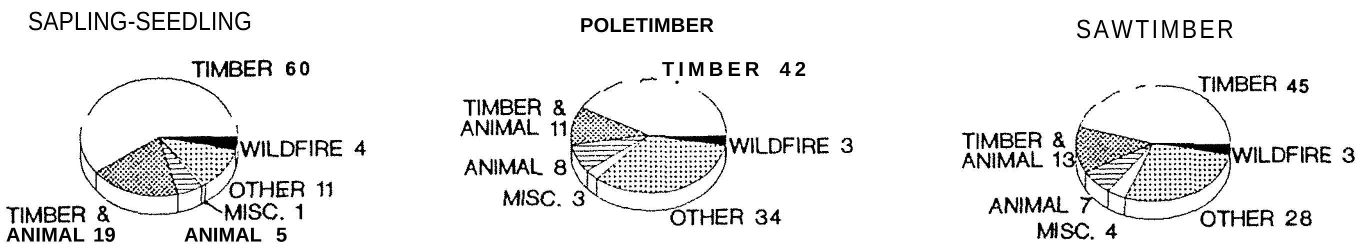

Figure 5.--Percent timberland area with fire evidence since the previous survey by stand-size class and potential causal agent (excludes Alabama).

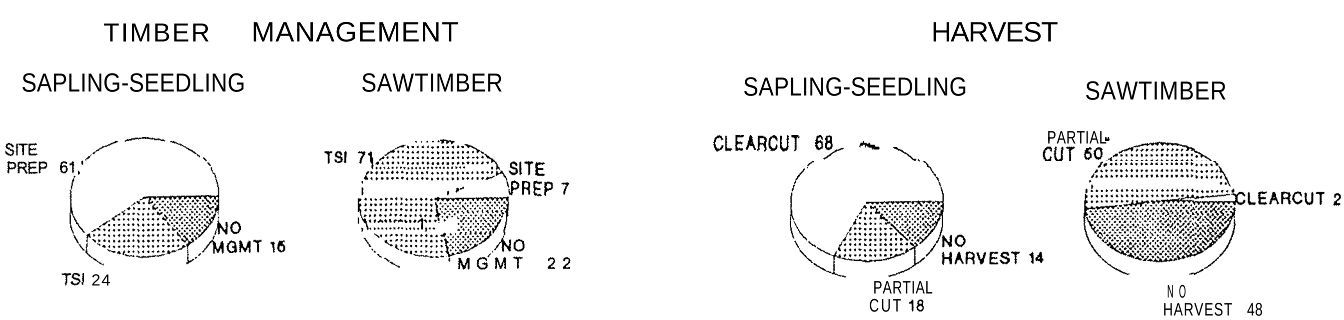

Figure 6.--Percent timberland area with fire evidence and timber production activity by type of timber management and harvest activity (excludes Alabama).

Figure 6 contrasts the occurrence of fire evidence in sapling-seedling stands and in sawtimber stands with timber production by management and harvest activities. Timber management includes site preparation (SITE PREP) and timber stand improvement (TSI); no management (NO MGMT) refers to harvest with no management. Timber harvest includes partial cut and clearcut; no harvest refers to timber management with no harvest activity. A majority of the fire evidence occurring with evidence of timber management is associated with site preparation in sapling-seedling stands and with timber stand improvement in sawtimber stands. In sapling-seedling stands, the majority of fire evidence associated with timber harvest is associated with clearcut operations. In sawtimber stands, most fire evidence associated with timber harvest is associated with partial cut operations.

Estimates of average annual tire occurrence by State, causal agent, and data sour-cc arc presented in table 4. It is worth noting that the causal agent estimates derived from forest survey data arc generally consistent with the estimates based on data from other sources. There is 95 percent confidence that forest survey total acres are reliable within the intervals noted.

DISCUSSION

Forest survey plots arc: categorized simply as having or not having tire evidence, and no interpretation is made regarding fire source. While this limits analysis of the data, the extensive and systematic sampling conducted during forest surveys yield descriptive regional information of heuristic importance. There also will be opportunities to assess trends in fire evidence when the area under study is resurveyed in the near future.

Total acreage with fire evidence from forest surveys represents an estimate of the cumulative acreage burned. No separate estimate is made for the acreage burned more than once between surveys. The forest survey estimate is based on a sample and has a corresponding error associated with the sampling process. In general, sampling error increases as the area considered decreases. Ninety-five percent confidence intervals arc generally within 5 percent of the acreage estimates for data aggregated over multi-county and larger areas.

Since most plots arc surveyed about every 10 years, we feel confident that observations of fire scars, vegetative growth since the last fire, management activities, and other disturbances can be observed readily. However, we feel that analysis of fire evidence by causal agent should be interpreted with caution. For example, fire is attributed to "other" agents less often for forest industry-owned land and in sapling-seedling stands than stands belonging to other categories (figs. 4 and 5). Survey results suggest at least two hypotheses: (1) fire evidence is associated with forest management activities in younger, sapling-seedling stands and on forest-industry land, rather than in older stands or on land in other ownerships; (2) fire evidence occurring in poletimber and sawtimber stands and associated with "other" agents is caused by low-intensity surface wildfires that do little damage tu trees, whereas tire evidence observed in sapling-seedling stands is associated with damage that is readily attributed to wildfire. An extension of the second hypothesis is that because forest industries employ prescribed fire to a greater degree than other landowners, fires caused by "other" agents may have less opportunity to occur.

Although we suggest that co-occurring evidence on sampled plots provides clues to the origin of fire evidence, we recognize the weakness of the assumption. Additional field measurements are needed to test this assumption to quantify fire impacts on forest ecosystems, and to estimate wildfire potential. Development of models that relate the type, intensity, and frequency of fires to fire evidence, e.g., litter depth on plots with and without fire evidence, and the addition of field measures to FIA plots that quantify potential surface moisture and the amount of live and dead materials would be useful in this regard.

Comparison With Other Estimates

We have already said that "ballpark" estimates of fire occurrence can be inaccurate. Nevertheless, the only available estimates of regional fire occurrence arc "ballpark' ones. Are forest survey estimates consistent with existing regional estimates of annual fire occurrence?

Except in the cast: of Alabama, FIA estimates of acreage arc from 13 to 43 percent higher than estimates from other sources. There are many reasons why estimates from other sources are lower. FIA estimates represent averages of cumulative acreage with fire evidence, include prescribed fire as well as wildfire, and represent estimates for public, forest industry, and nonindustrial private lands. Estimates from other sources often represent acreage burned in a single year or averages over a few years; such figures also include prescribed fire and wildfire estimates for different reporting periods and landowner groups. Information from other sources is not as likely to come from all forested areas, particularly in the case of prescribed fires not associated with timber production activities.

Table 4.--Average annual fire occurrence by state, causal agent, and source of data.

State and causal agent	Forest Survey[a]				Other sources[b]		
	Years	Percent	Thousand acres	(+,- 2 S.E.)	Thousand acres	Percent	Years
Alabama[c]	1973-82						
Prescribed fire		44	318.1	(15.4)	518.1	68	1975,84[d]
other agents-wildfire		56	497.3	(17.5)	238.9	32	1988
Total			725.4	(23.3)	757.0		
Arkansas	1979-86						
Prescribed fire		77	191.2	(7.1)	182.6	84	1975,88[e]
Other agents-wildfire		23	58.2	(3.9)	34.9	16	1988
Total			249.4	(8.1)	217.5		
Southeast Louisiana	1975-84						
Prescribed fire		74	53.9	(5.4)	57.6	90	1975,84'
Other agents-wildfire		26	18.7	(3.2)	6.6	10	1988"
Total			72.6	(6.3)	64.2		
Mississippi	1978-87						
Prescribed fire		79	333.2	(14.0)	233.5	68	1975,84[h]
Other agents-wildfire		21	91.1	(7.3)	108.9	32	1988
Total			424.3	(15.8)	342.4		
Fast Oklahoma	1977-86						
Prescribed fire		60	80.5	(8.5)	55.0	59	1977-86'
Other agents-wildfire		40	53.7	(6.9)	39.0	41	1977-86'
Total			134.2	(II. 0)	94.0		
Tennessee	1981-89						
Prescribed fire		51	46.5	(4.6)	26.9	35	1975,84'
Other agents-witdfire		49	45.1	(4.6)	51.0	65	1988
Total			91.6	(6.5)	77.9		
Fast Texas	1977-86						
Prescribed fire		83	160.3	(8.0)	123.8	79	1975,84[k]
Other agents-wildfire		17	33.7	(3.7)	52.9	21	1988
Total			194.0	(8.8)	176.7		

[a] E Lapsed time in years: AL=9.8, AR=9.5, LA=9.1, MS=9.5, OK=9.8, TN=9.0, TX=10.4.

[b] 1988 wildfire estimates are from USDA-FS (1988b). For the entire. South, average annual acres burned by wildfire has changed little in the past decade (USDA-FS 1988a). 1975 prescribed fire estimates are from Johansen and McNab (1982).

[c] May include fire evidence older than previous survey.

[d] Average of 157,320 acres (1975) and 878,970 acres (1984) (Moody 1985).

[e] Far 1988: 35,206 acres National Forest (USDA-FS 1988a), ad 15,000 acres by the State Forestry office (Garner Barnum, pers. comm.). For 1975: on private Land, 132.350 acres in 1975. More recent data on prescribed fire we on forest industry Land are not available (Garner Barnum, pers. comm.)

Statewide average of 462,420 acres (1975) and 450,090 acres (1984) (Miles 1985) adjusted for the portion of forested area surveyed.

[g] Statewide estimate adjusted for the forested area surveyed.

[h] Average of 167,050 acres (1975) and 300,000 acres (1984) (Miles 1985).

10-year average, 1977-1986, based on state and forest industry records (Kurt Atkinson, Oklahoma Forestry Division, pers. comm.).

Average of 31,700 acres (1975) and 22,100 - (1984) (Ashley 1985).

[k] Average of 47,600 acres (1975) and 200,090 acres (1984) (Mikes 1985).

IMPLICATIONS AND CONCLUSIONS

What can we conclude from the data prescntcd? Thcrc arc three major conclusions: (1) fire evidence is pcrvasivc in the South Central States, (2) fire evidence is concentrated in pine--growing parts of the South Central States, (3) observations of evidence of fire occurrcncc from forest surveys can be used as a basis for credible estimates of past fire occurrence.

Projections of forest acres in South Central States by forest type suggest a continuing increase in pine plantation acrcagc and a corresponding decrease in natural pin e acreage, a trend that has continued since 1970 (Birdscy and McWilliams 1986). If cvidcnce of past fire occurrcncc is an indication of future trends, a* increase in pine plantation acreage will be associated with increased fire frequency. Any existing or proposed policies for regulating fire to reduce smoke hazard or increase protection from wildfire will have widespread consequences for forestry in this region.

Forest survey fire cvidcnce data can be used as a basis for studying the potential for air quality degradation and tire danger at the regional level. When combined with wildfire and weather statistics, forest survey tire evidence data can be used to establish regional forest protection priorities. The data prcscntcd can be used as a basis for assessing fire occurrence in studies of water, livestock forage, wildlife habitat, and timber production in multi-county and larger areas. That forest industries and other ownership groups in selected regions have considerable acrcagc with tire evidence suggests that fire plays an important role in these areas. The regional extent of this influence is documented in this report and should be considered when discussing forest management policy and the future condition of forest ecosystems in the South Central States.

The extent and importance of fire's effects on South Central States' forests cannont be fully elucidated from forest survey data without additional information. Because there exists a wide array of forest characteristics, including previous land use and ownership data, testing inferences regarding prescribed fire and wildfire origins on a subsample of plots should prove fruitful. Linkage of forest survey fire evidence data with a suitable geographic information system also can provide regional modelers with supplementary data for use in assessing other values (e.g., water quality and soil erosion) affected by fires.

Continued monitoring of fire evidence on forest survey plots can supply analysts with information about trends in past fire occurrence. Additional measures needed to identify causal agents could he developed for a carefully selected subsample of plots and then modeled for all plots. Such a method would be especially useful for monitoring trends in fire use as a management tool and evaluating tire's effectiveness in increasing timber productivity and other multiple-value forest resources,

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A SITE-SPECIFIC APPROACH FOR ASSESSING THE FIRE RISK TO STRUCTURES AT THE WILDLAND/URBAN INTERFACE

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Abstract-The essence of the wildlandiurban interface fire problem is the loss of homes. The problem is not new, hut is becoming increasingly important as more homes with inadequate adherence to safety codes are built at the wildlandiurban interface. Current regulatory codes are inflexible. Specifications for building and site characteristics cannot be adjusted to accommodate homeowner values. USDA Forest Service Fire Research is developing a wildlandiurban fire interface ignition assessment model as an alternative to current tire safety codes. This model is based on an analytical (rather than statistical) assessment of structural characteristics, site characteristics, and fire severity conditions. The model will be capable of assessing ignition risk for individual structures, and thus will be capable of accommodating homeowner preferences as they affect tire safety.

INTRODUCTION

More than 1,400 homes were damaged or destroyed during wildland fires in Florida, North Carolina, California, and other States during 1985 (Laughlin and Page 1987). This created national interest in what has come to be called the wildlandlurban interface fire problem. Interest in the problem continues to grow as the number of people who live in or adjacent to wildland areas increases (Davis 1990).

Although new emphasis has been placed on the problem of structure loss and damage associated with wildland fires, the problem is an old new. During the last 30 years. frequent conflagrations in California have resulted in losses of structures, primarily homes. After major California fires, reports that identified the fire problem and provided guidance for mitigation were generated (California Dcpartmcnt of Conservation 1971; California Department of Forestry 1980; County Supervisors Association of California 1965; Howard and others 1973; Moore 1981; Radtke 1983). Generally, these reports were commissioned by State and local government agencies. With some exceptions (Dell [n.d.]; Radtke 1982), the target audiences were public officials and fire professionals. Many of these wildlandiurban fire reports were comprchcnsive, providing recommendations, including technical specifications, for urban planning, fire suppression capabilities, vegetation managcmcnt, and building construction. However, despite the production of these reports, the wildlandiurban interface fire problem has continued with little abatement.

Little attention has been given to the social aspects of the wildlandiurban interface fire problem, and this may be one reason why the problem persists. The technical aspects of this fire problem such as building codes and suppression improvements, have dominated discussions about the subject. However, the social aspects of the wildland/urban fire problem gained attention at wildlandiurban intcrfacc workshops conducted during 1986 and 1987 (Laughlin and Page 1987; USDA Forest Service 1987). The participants concluded that a solution to the fire problem must recognize and accommodate homeowner values and motivations.

The social aspects of the problem have not yet been addressed in the practical arena. Current fire standards are generally cmbodied in zoning and building codes (specification codes). These specification codes regulate minimum allowable building and site characteristics. Examples of specification codes include requirements for street width, the number of structures per area, vegetation clearance, roof material, and screening of vent openings. Standards such as these (California Department of Forestry 1980; County Supervisors Association of California 1965; Moore 1981) recommend minimum building and site characteristics for improving structure survival. Where these recommendations are implcmcntcd, structure survival is increased -- but they are generally not implemented at the wildlandlurban interface.

Specification codes cannot make allowances for the diversity of social values. Generally, specification codes arc not flexible in responding to homeowner values and motivations; trade-offs cannot be made to achieve a fire-safe condition. Specification codes arc implcmcntcd or they are not. As a result, specification codes connote uncompromising compliance with government imposed regulations. Because many of the property owners who live in wildlandlurban intcrfacc areas move there to escape urban regulation (Bradshaw 1988), there is great resistance to fire safety regulations that restrict building and site characteristics. This suggests the need for a regulatory approach that can make allowances for the diversity of social values while it identifies measures for reducing the fire risk.

This article examines the wildlandiurban interface fire problem in terms of structure ignition and survival. It also describes a current USDA Forest Service effort to develop a flexible method for assessing the relative risk of structure ignition.

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THE WILDLAND/URBAN INTERFACE FIRE PROBLEM--STRUCTURE SURVIVAL

The essence of the wildlandiurban interface fire problem is the damage to homes during wildland fires. Any attempt lo analyze the problem should recognize the various factors that influence the survivability of homes during wildland fires.

Structure survivability is the probability that a building will not suffer major structural damage during a fire. A structure's survivability depends on the structure's resistance lo ignition and on the ability to suppress any ignitions that might occur. Thus, theoretically, for a given likelihood of structure survival, a variety of ignition resistance levels can be balanced by compensating suppression capabilities. This introduces the idea of trade-offs--in this case, ignition resistance for suppression capabilities, and vice versa.

Structure survival can bc examined in greater detail. The ignition aspect of structure survival can be further defined in terms of the structural fire performance, the fire exposure, and the fire scvcrity conditions. Similarly, the suppression aspect can be defined in terms of suppression availability, safe access lo the structure, and fire scvcrity conditions. *It* is helpful to consider each of these factors in greater detail.

fire performance: During a wildlandiurban interface fire, ignition sources, i.e., firebrands, impinging flames, convective heating, and radiative heating, arc initially external to the structure. Structural fire performance is the susceptibility to ignition and the degree of subsequent fire involvement Structural fire performance depends on the physical characteristics of the structure. For example, a concrete structure with no window openings would resist an external ignition more than a structure with large window openings and an exterior covered with wooden shingles. Also, if an ignition occurs, the rate of fire involvement would very likely he greater for the wooden structure. The concrete structure has a higher fire performance than does the wooden structure. These simple examples represent the cxtremes of a broad spectrum of physical characteristics that dctcrminc structural fire performance.

Fire exposure: The fire exposure of a structure is defined here as the external sources (burning materials excluding flaming brands) of radiative and convective heating, and the site characteristics that influence the amount of heat transferred to the structure. For example, burning trees, shrubs, and wood piles at various distances from the structure dctcrminc how much the exterior is heated When there is burning material downslope from the structure, the potential for convective heating is increased. In terms of fire cxposurc, only those burning materials close enough to the structure to influence an ignition arc considered as heat sources. The area containing such materials can be termed the fire exposure zone. In this way, the concept of fire cxposure can be used to distinguish wildland vegetative fuels (outside the fire exposure zone) from vegetation adjacent lo a structure. Because a fire exposure zone is defined by the characteristics of the fire and the fire performance of the struclurc, fire cxposurc is not limited by property lines. Thus, the delineation of a site-specific fire exposure zone aids in identifying all of the flammable materials relevant to structure survivability.

Fire suppression availability: Availability of protection during a fire is an important aspect of structure survival. Structure ignitions, if cxtinguishcd, can occur without lhc loss of the structure. The quantity of fire suppression staff and cquipmcnt, together with training, cxpcricncc, and response times, dctcrmincs fire suppression availability. This factor is generally considered the domain of fire suppression organizations, but community residents arc also a part of the fire suppression availability factor. Residents arc often the source of fire reports and augment organized suppression forces by working to protect individual properties.

Access: Asscssibility is critical for the utilization of available suppression resources, and thus to structure protection during a iirc. Access is the ability of fire suppression forces, including residents, to locate, reach, and safely remain at a structure and continue suppression efforts.

Fire severity conditions: This refers to the conditions that affect the flammability of fuels, flame tilt, spread rates, and aerially transported burning brands. Fire severity conditions are determined primarily by on-site weather and topography. Fire scvcrity conditions influence the degree of fire exposure and the cffcctivcncss of structure protection.

Structure survivability is an expression of the interactions of the above-&fined factors. Spccilication codes seldom take these factor interactions into account, and therefore present relatively rigid formulas for providing fire-safe environments. Thr failure to account for these factor interactions prevents the incorporation of social values into fire safety measures. The alternative to specifying minimum characteristics (specification codes) to produce a given level of structure survival is to make usc of a model that incorporates factor interactions.

The essence of structure survival, and thus of the wildlandiurban fire interface problem, is ignition. If ignitions do not occur, then structures survive. Although structure survival involves the interaction of all five factors discussed, just three of them--structural fire performance, fire exposure, and fire severity conditions-determine the potential for ignition. For a given level of fire severity, the ignition potential dctcrmincs the level of fire suppression availability and accessibility necessary to produce a given level of structure survival. As structural fire performance decreases and fire exposure increases (increasing the ignition potential), the accessibility and suppression availability must increase if a structure is to survive. The wildland/urban fire interface problem would virtually disappear if structures did not ignite; therefore, the emphasis of a wildland/urban interface fire risk assessment should be on structure ignition.

IGNITION RISK ASSESSMENT MODEL

A structure ignition risk assessment model is now being developed by the USDA Forest Service. This cooperative effort involves the Forest Products Laboratory, Madison, WI, the Riverside Fire Laboratory, Riverside, CA, and the Southern Forest Fire Laboratory, Macon, GA. The product will be a broadly applicable method for assessing the relative risk of individual homes to external ignitions from wildland fires. The prototype model is expected to be completed in 1991.

This model approaches interface home losses in a new way. Recent models by Abt and others (1987) in the United States and Wilson (1988) in Australia have used statistically derived relationships, based on specific fires, to describe characteristics related to potential fire incidence and structure survival. Our model uses analytical relationships to describe characteristics related to the potential for structure ignitions. This approach has the following advantages:

- An analytical approach is not limited by specific event data and its interpretation,
- An analytical approach, based on physical relationships can easily incorporate future gains in understanding to fill current gaps,
- An analytical approach can incorporate the interactions of the various factors affecting the wildland/urban interface fire problem,
- The modeling of interactions provides a means for analyzing mixes of factors, and thus a means for analyzing trade-offs in meeting tire safety requirements.

Our wildland/urban fire interface model assesses the risk of potential ignitions rather than potential structure survival. As noted previously, structure survival depends on both the ignition factors and the suppression factors. Thus, an assessment of potential structure survival would require an assessment of the suppression factors. However, many of these factors (access, suppression availability, and tire severity) are very hard to quantify. For example, the resident's presence at the home during the fire can be critical to the home's survival. But it may not be possible to reliably assess the likelihood that a homeowner will be at home at an unspecified time, especially in a situation complicated by emergency access limitations and evacuation policies. (The statistical models previously cited also do not account for the suppression factors, although structure survival is ostensibly the product of the Australian method.)

The Ignition Risk Rating System

The Ignition Risk Rating System borrows some of its underlying philosophy from the National Fire Danger Rating System (NFDRS) (Deeming and others 1978) The Risk Rating System is based on physical principles so that new understanding can be easily incorporated into it. Where gaps in knowledge exist, personal expertise estimates the effects of the physical processes. As with the NFDRS, the ratings are not incident-specific, but rate the potential fire situation. Therefore, a worst case approach is taken in acquiring data and making computations. Because it is not possible to make precise evaluations of ignition occurrences, the risk ratings are placed in ordinal categories.

Rating risks of potential structure ignitions requires that structure characteristics, site characteristics, and fire severity conditions be described and analyzed to produce assessments. The Ignition Risk Rating System does this in three stages. In their computational order, these stages are the fire source module, the heat transfer module, and the structure ignition module.

The fire **source** module describes the site and fuels around the structure and transforms that information into descriptions of the potential flaming sources affecting the structure. The fire severity conditions are locally identified using National Fire Danger Rating System (NFDRS) burning index cumulative frequency percentiles. The percentiles are computed from historical fire weather data. The flaming source descriptions are based on an on-site fuel inventory in conjunction with the potential fire behavior identified by the NFDRS calculations. The potential flaming sources are described by their flame length, flame zone depth, flaming width, flaming duration, and distance from the structure. A subjective assessment of the structure's relative firebrand **exposure** is made on the basis of the fuel sources adjacent to the structure and on nearby wildlands. For example, a structure is considered less exposed to ignition by firebrands from grass fuels than by a conifer stand with heavy understory fuels. These descriptions are then used by the other modules.

The heat transfer module uses flame source and homesite information from the flame source module to estimate the radiative and convective heating of the building exterior. Due to the impossibility of knowing the specific characteristics of a future incident, the worst case configurations for the heat transfer are used. For example, all fuels are considered to be burning at the same time, the radiative distance is considered to be the closest distance to the fuel, and the view angle between the structure and the flames is assumed to be the angle that produces the greatest heat transfer. Convective heating occurs if the convection column intercepts the structure, but potential cooling from the wind is not considered.

The structure ignition module calculates the System's ignition risk rating. The module uses descriptions of the building's exterior structural charactcristics along with the heat transfer information and the assessed fircbrand exposure to compute the ignition risk. Exterior materials arc generally described by type (wood, stucco, glass, etc.) and exposed surface area. The ignition risk assessment is largely based on data derived from laboratory fire tests conducted on exterior building materials (roof materials, siding, windows, etc.). The module produces the ignition risk rating based on the relative availability of energy for an ignition. Relative risk ratings fall into four classes: low, moderate, high, and extreme. To facilitate a consistent assessment of the ignition potential, the classes are defined in such a way that a structure in the next higher class has twice as much ignition potential as a structure in the one below.

System Benefits

The Systcm will provide property owners and suppression organizations with a guide to assessing, and thereby reducing if necessary, the potential ignition risks to homes. The System will providc the following:

- A flexible means of integrating social values and tire performance requirements--one that will encourage greatcr acceptance of actions nccessary to dccrease the risk of structure ignition,
- A mcans for evaluating a mix of conditions of the components that contribute to structure ignition, thus allowing for site-specific and property-owner-specific actions that meet minimum requirements for ignition risk-making informed tradeoffs,
- A means for informing and educating propcrty-ownvers about thc rclativc risk to their homes,
- A means for informing suppression agencies of thc rclativc fire risk to homes, leading to more informed suppression planning.

SUMMARY

The wildland/urban interface fire problem, i.e., the loss of homes during wildland fires, is not new. The problem persists, not because there are no fire safety guidelines, but because guidelines are not fully implemented. Until recently, homeowner values and motives were not recognized as important in achieving fire safe home sites. The current guidelines used for the wildland/urban fire interface arc fixed, discrete specifications that cannot incorporate variations in homeowner values while maintaining a given level of fire safety.

A USDA Forest Service cooperative research effort is currently developing a wildland/urban interface structure ignition risk assessment model. This physically based analytical model is being designed to account for the intcractions of the factors that contribute to structure ignitions. Thus, the model may be used to identify a variety of fire safcty measures that result in required risk reductions and also accommodate specific homeowner desires.

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PERCEPTION OF FIRE DANGER AND WILDLAND/URBAN POLICIES AFTER WILDFIRE

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Abstract-Quantitative analysis conducted after the May 1985 Palm Coast fire in Florida identified several residential characteristics that influenced vulnerability to wildfire. As a followup to that analysis, homeowners were surveyed to determine their perception of fire danger and to determine their views on alternative mitigation measures they have undertaken as individuals and their view of alternative government mitigation policies. The survey indicates that homeowners perceive wildfire as a serious threat to their safety and property. Homeowners were receptive to a wide variety of government policy options, including restrictive planning, zoning, and building requirements. Older homeowners were more likely to have taken mitigation measures and were more receptive to government intervention.

INTRODUCTION

Palm Coast is a 42,000.acre planned unit development situated in the coastal plain flatwoods along the east coast of Florida, Prior to its development, the land was actively managed for pine timber production. Timhcr management included the periodic use of prescribed fires to control fuel buildup.

Typical of other large developments that started in Florida in the 1960's, Palm Coast was primarily a lot-sales venture. Individuals purchased land using a long-term payment plan with the intention of building at some future date. All roads, underground utilities, and drainage systems were installed within the first 10 years of construction. Except for a densely developed core, homes have been built sporadically throughout the development. Prescribed fires were no longer used after construction began.

In May 1985, a devastating wildfire burned through Palm Coast, destroying 100 homes and damaging 200 others. Lack of brush clearance, fire intensity (due to abundant fuels), type of soffit vent, and type of construction were shown to have been associated with increased fire losses (Abt and others 1987).

In 1988, the Federal Emcrgcncy Management Agency and the United States Forest Service jointly funded a followup study to determine homeowner perception of the wildfire threat and attitudes toward various mitigation strategies. Additional objectives were to determine how mitigation could be incorporated into the land-use planning process and to determine the vulnerability of recently built homes. The purpose of this paper is to summarize the residents' perception of wildfire danger, the mitigation measures taken, and attitudes toward various mitigation policies.

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METHODOLOGY

The study was conducted jointly by the Florida Division of Forestry (DOF) and the University of Florida. A mail survey was designed to gather information about the residents and their perceptions about wildfire. The survey included a cover letter signed by the local district forester and fire chief Two survey areas were selected. One was an area burned in the 1985 fire and the other was a nearby area of similar housing density that was not damaged by the fire. All of the homes in the two sections were surveyed. The survey was hand-delivered by DOF personnel and returned via business reply mail. Three weeks after the surveys were distributed a follow-up letter was mailed. Approximately 276 questionnaires were distributed. There were 124 usable questionnaires returned for a 45-percent response rate. A followup survey of 40 nonrespondents (20 in each section) revealed that many of the houses were vacant. The few followup surveys completed by nonrespondents did not reveal any significant differences with respondents.

The survey questions reported here fit into four categories, (1) demographic information, (2) perceived threat of wildfire, (3) mitigation measures taken, and (4) attitude toward mitigation options. Simple summary statistics arc reported below. Chi-square statistics were used to test for association between the demographic variables and resident acceptance of mitigation measures. For many of the subjective opinion questions a l-to-6 scale was used. The lower (l-to-3) end of the scale was used to represent lack of a threat or agreement with the question (3.5 represents neutral).

RESULTS

Initial tests for differences between the two sections of the development revealed that they differed only in the number of respondents who had personally experienced wildfire. The results reported here are the combined responses from both sections.

Demographics

The section of Palm Coast surveyed is predominantly a retirement community. Sixty percent of the homes have one or more family members over 6.5. Only 25 percent of the homes have children under the age of 18. The average age of the respondents was 56. On average the respondent had lived in the current home for 3.6 years, and 75 percent planned to continue living in the community for at least 5 years. Most respondents (55 percent) had no more than a high school education, though 25 percent had at least a bachelor's degree. The median income was approximately $30,000, and the average market value of the homes was $98,000. Though these statistics give an overview of the community, their primary value was as an explanatory variable in determining attitudes toward mitigation. These analyses are described below.

Perceived Threat of Wildfire

Two aspects of wildfire threat were surveyed. The first was the perceived danger of a wildfire to the home and family if it were to occur. The second was the perceived probability of a wildfire occurring. The questions regarding the perceived threat from wildfire were put into the context of other threats facing the community.

The first table below shows how the respondents viewed wildfire compared to a wide spectrum of common problems. Wildfire was considered the most serious threat facing the community, even 4 years after the major fire. This is especially important given that 69 percent of the respondents had been in their current home for less than 4 years (after the wildfire). The perceived threat of wildfire to the home and family was measured on a 1 to 6 (1 =no threat, 6=extreme threat) scale. The perceived threat of hurricanes and tornadoes was also collected. Wildfire was given an average score of 4.8 as a threat to the home and a score of 4.5 as a threat to the family. Wildfire was rated higher in both categories than either hurricanes or tornadoes.

Perceived problems facing the Palm Coast community:

	Mean score
Economic problems (unemployment, inflation, etc.)	2.98
Drought	3.99
Crime rates	3.59
Illegal drugs	4.04
Damage or injury from hurricanes	2.49
Damage or injury from tornadoes	2.34
Damage or injury from wildfire	5.09
Exposure to radon	2.18
Water supply	3.07

[2] 1 = no problem, 6 = most serious problem.

The above questions asked about the threat of a wildfire if it were to occur. There are at least two other factors that will determine whether a resident should take action to mitigate the damage. The first is the perceived probability of wildfire occurrence and the second is the perceived effectiveness of mitigation.

Residents were asked to give their best estimate of the chances of a serious wildfire in their community in the next 10 years. The average estimate was 57 percent, which was higher than either tornado or hurricane. Residents were also asked to rate on a 1 to 6 scale the control they had over wildfire damage (1 =no control, 6=complete control). The average response was 3.0 which was higher than either tornados or hurricanes. Given the high probability and threat associated with wildfire and the high percentage of new homes, one might expect that vulnerability to wildfire would be important in the choice of a home. When rated on a 1 to 6 scale (1 =not important, 6=extremely important), however, vulnerability to wildfire only rated 2.5 in choice of current home.

The perceived threat of wildfire to the home was associated with education (.04 significance level), where residents with education beyond an undergraduate degree felt less threatened. Residents whose insurance would cover all or most of the damage from a wildfire also felt less of a threat (.08 significance level). The perceived threat of wildfire to the community was related to age, personal wildfire experience, and income. Residents over 65 and residents who had experienced wildfire were far more likely to rate wildfire an extreme threat (.03 and < .01 significance levels respectively). Middle income ($40K-$50K) residents rated wildfire as less of a community threat than others (.02 significance level). New residents (< 4 years) were far more likely to have considered wildfire vulnerability in their home choice. Those who had experienced wildfire, however, felt they had less control over wildfire damage.

Mitigation Measures Taken

Sixty-seven percent of the respondents reported that they had taken some sort of precaution against wildfire. The probability of having taken safety measure was positively related to whether the resident or a close friend had experienced wildfire (.03 and .01 significane levels respectively). The older residents (> 45) were more likely to have taken measures (.002 significance level). Residents without children at home, which were probably the older residents, were also more likely to have taken precautions (.03 significance level) as were homeowners (versus renters) and residents who planned to stay in the community for at least five years (.003 and .04 significance levels).

Ninety-three percent of these safety measures were taken after the 1985 wildfire. Most measures taken cost less than $100 (65 percent). The most common measure was tree or brush removal (53 percent). Others removed mulch (14 percent) or purchased fire safety equipment (25 percent, including water hoses, pool pumps, sprinklers, extinguishers, etc.).

Attitudes Toward Mitigation Policies

Resident attitudes about various mitigation measures and government policies were examined. A wide variety of measures were examined from passive voluntary measures to restrictive governmental intervention. As the tabulations below show, the residents generally favored any and all mitigation measures mentioned. Even controversial measures such as mandatory brush clearance and community-wide control burning were considered acceptable.

Attitude toward government mitigation policies:

Government agencies should:	Mean score'
Provide financial assistance to victims	2.3
Prohibit building in hazardous areas .	2.2
Impose stricter building codes in hazardous areas	1.8
Provide information to homeowners in hazardous areas	1.5
Conduct research on ways to reduce damages	1.8
impose stricter zoning requirements in hazardous areas	1.9
Impose stricter planning requirements in hazardous areas	1.8
Provide non-financial assistance to victims . .	2.7

'1 = no problem, 6 = most serious problem

Attitude toward mitigation measures:

	Mean Score'
Voluntary brush clearance	2.0
Mandatory brush clearance	2.3
Mandatory home site survey for fire hazard	2.4
Community-wide control burning	1.9
Stricter building codes	2.0
Stricter planning requirements for developers	1.8
Increased wildland firefighting resources	1.7
Increased structural firefighting resources	1.9

'1 = no problem, 6 = most serious problem

Though there was general acceptance of government policies, older and permanent residents tended to accept restrictive policies, while highly educated residents tended to be less open to government intervention. For example, residents who planned to stay in the community for the next 5 years were three times more likely to favor government prohibition of building in hazardous areas (.005 significance level). Older residents (> 45) were four times more likely to favor strict planning (.034 significance level), Residents with education beyond an undergraduate degree were far more likely to oppose the imposition of codes, zoning, or planning restrictions (.00, .04, and .OO significance levels respectively). Lower income families (< $30K) were more likely to oppose nonfinancial support (.03 significance level) while those who had experienced wildfire were much more likely to favor non-financial support (.05 significance level).

There were only two resident characteristics that were significantly related to specific mitigation measures. Residents who planned to live in the community for at least 5 more years found a mandatory home survey more acceptable (.04 significance level), while homeowners were much more open to stricter codes than were renters (.04 significance level).

SUMMARY

Five years after the Palm Coast fire, residents consider wildfire the major threat to their community. Many residents have taken precautionary measures, though wildfire vulnerability was not important in home choice. Residents over 65 were more likely to consider wildfire an extreme threat and were more likely to have implemented mitigation measures. They were also supportive of all policies considered. Residents with an education beyond a bachelors degree, however, were less tolerant of possible government intervention.

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FIRE HISTORY

FIRE HISTORY AND EFFECTS ON VEGETATION IN THREE BIOGEOCLIMATIC ZONES OF BRITISH COLUMBIA

John Pa-minter'

Abstract-The different fire regimes present in the province of British Columbia are well-illustrated by the Coastal Western Hemlock, Boreal White and Black Spruce, and Ponderosa Pine Biogeoclimatic Zones. Fires are variable in type, intensity, severity, effects, size, and interval. In the first two biogeoclimatic zones, stand replacement and/or partial stand replacement fires occur. Surface tires that maintain open forests with grassy understories are the historic norm in the Ponderosa Pine Zone.

Fire histories and fire effects have been determined for all three zones. In the Coastal Western Hemlock Zone the emphasis has been on fire history, stand establishment, and prescribed burning. Ecosystem-specific guides describing the type of tire suitable for silvicultural site preparation are widely available Post-fire vegetation successional pathways have been developed for the Boreal White and Black Spruce Zone and are used to predict probable fire outcomes.

In recent times, the adverse implications of fire exclusion have been recognized in ecosystems such as those found in the Ponderosa Pine Zone. The reintroduction of fire through prescribed burning serves to address the problems and meet several resource management objectives.

INTRODUCTION

British Columbia is a large and ecologically diverse province. It covers a total area of 94,900,000 ha, of which 52,200,000 ha arc forested (Farley 1979). The vegetation has been classified into fourteen biogeoclimatic zones which range from, but are not limited to, coastal rainforests, southern grasslands, northern boreal forests, and alpine tundra (Pojar 1983; Pojar and others 1987).

The fire history and tire ecology of these biogeoclimatic zones have been studied to varying degrees, depending on location, the management agency, and context.

OBJECTIVES

The objective of this paper is to describe the fire history, and general fire ecology, of three biogeoclimatic zones. For several of the fourteen zones little information exists and in others the focus has often been on just one subject area, such as prescribed fire effects. By choosing three zones to examine, a range of conditions within the province can bc illustrated.

An historical perspective is provided since much of the older, as well as some of the more recent, fire history work done in the province has received little exposure to date.

COASTAL WESTERN HEMLOCK ZONE

This zone occupies low to middle elevations, mostly west of the coastal mountains, along the entire B.C. coast, on Vancouver Island and the Queen Charlotte Islands (fig. 1). It is the wettest zone and has cool summers and mild winters. Mean annual precipitation is from 1,500 to 4,400 mm (Pojar and Klinka 1983).

The major tree species present are western hemlock (*Tsuga heterophylla*) and western redcedar (*Thuja plicata*). Douglas-fir (*Pseudotsuga menziesii*) is present in the drier subzones; amabilis fir (*Abies amabilis*) and yellow-cedar (*Chamaecyparis nootkatensis*) are common only in the wetter subzones. Present, but restricted to certain habitats, are grand fir (*Abies grandis*), western white pine (*Pinus monticola*), Sitka spruce (*Piceo sitchensis*), bigleaf maple (Acer macrophyllum), red alder (*Alnus rubra*), black cottonwood (*Populus balsamifera* ssp. *trichocarpa*), and shore pine (*Pinus conforta Var. contorta*).

Fire History

Evidence of fire primarily takes the form of charcoal layers in the soil, fire-scarred trees (primarily Douglas-fir), charred bark, and even-aged stands of shade-intolerant species. The fire regime consists of long interval severe crown and surface fires which result in total stand replacement. Lower severity surface and crown tires with partial stand replacement are also common, often within the larger more severe bums.

Historically, large areas burned when lightning-caused fires occurred during extended regional summer droughts. Physiography plays a role in tire history as well, with differing fire incidence being a function of elevation, topography, and aspect (Schmidt 1960).

The fire history of the Coastal Western Hemlock Zone in British Columbia was first studied in the early part of this

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century. The situation is complicated by more than 1.50 years of human disturbance. Removal of the original forests and their replacement by second-growth plantations, in combination with wildfires and prescribed bums, has made the determination of historical tire frequencies difficult or impossible in many localities.

Rigg and Richardson (1938) noted the presence of charred remains in many bogs in northwest Washington and southwestern British Columbia but did not indicate how long tire had been a factor in the regional landscape. In lake sediments from the adjacent western hemlock zone of Washington state, Cwynar (1987) dated Douglas-fir and western hemlock pollen mixed with charcoal fragments to 11,000 years before present. Climatic change has occurred since then, and with it the tire regime and relative abundance of the two conifer species.

Fire intervals likely range from 150 (Martin and others 1976) to 3.50 years or more but may not be truly cyclic (Agee 1981). Fahnestock and Agee (1983) estimated the tire cycle for western hemlock forests of western Washington to be 598 years. The value for western redcedar forests was 3,116 years but the authors considered that figure to be unexpectedly long. Agee (1990) calculated a fire cycle of 400 years for the cedar/spruce/hemlock type in Oregon.

Dendrochronology has revealed fires as early as 670 A.D. in Douglas-fir forests on northern Vancouver Island (Schmidt 1970). The even-aged character of these stands, coupled with charcoal in the soil and charred bark on veteran trees, denoted a tire disturbance history. A second disturbance by tire may result in a stand with two major age classes. Significant fire years on the British Columbia coast are indicated for approximately 1100, 1210, 1410, 1560, 1610, 1660, 1740, and 1820 (Schmidt 1970).

Howe (1915) found that Douglas-fir forests in this zone generally originate after fire. Stand establishment dates suggest fires occurred in 1454, 1489, 1558, 1814, and 1844. Scars on large old trees recorded tires in 1598, 1684, and 1844.

Eis (1962) also considered the oldest tree layer to be indicative of a tire year, with younger and lower layers recording subsequent and less severe disturbances. Based on stand ages, tires are indicated for the lower mainland coast around 1540, 1550, 1660, 1690, 1770, 1790, 1840, 1860, and 1890.

Correlations of tire dates within and between the data for different localities in Schmidt (1970), Howe (191 5), and Eis (1962) indicate more prominent fire years for the lower British Columbia coast as 1489, 15.58, 1660, 1684, 1690, 1820, and 1890. Mathcwes (1973) found a charcoal-rich lake sediment layer which might very well coincide with a

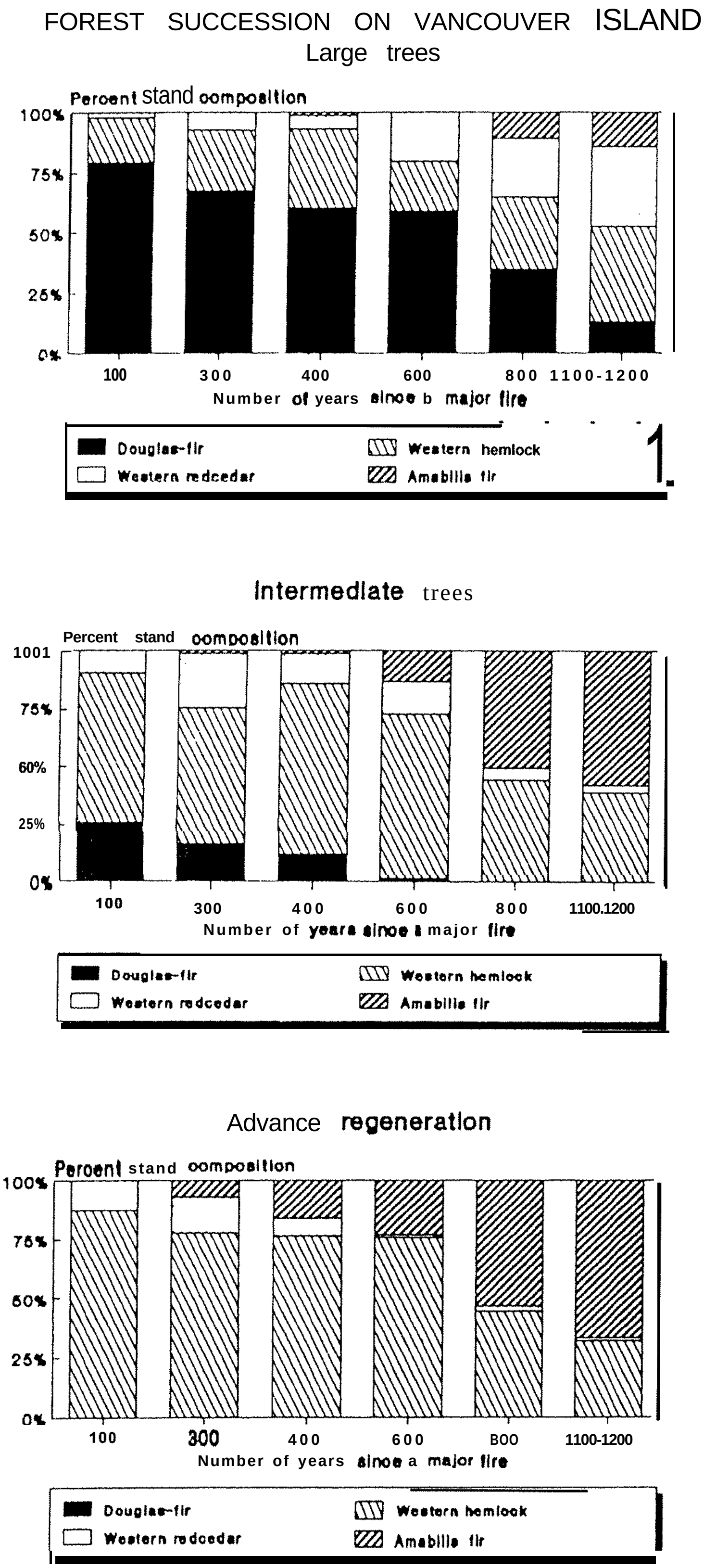

Figure 1. Location of biogeoclimatic zones.

significant (800,000 ha) regional fire event in the mid-1600's described by Schmidt (1957) and/or a local fire dated to around 1660 by Eis (1962).

On the north coast, other disturbance agents such as wind and landslides may be more important than tire. Harris and Farr (1974) noted that while there arc significant areas of even-aged stands on the Alaska panhandle which apparently owe their origin to fire, the bulk of the forests there are uneven-aged. Age class structure suggests that extensive fires burned in 1664, 1734, and between 1804 and 1824. Since 1900 few fires have burned more than 40 ha.

Fire Effects and Succession

At a large scale, Schmidt (1960) felt the lower incidence of forest fires on the north coast was responsible for limiting the range of Douglas-fir. Western redcedar continues north for 480 km beyond the range of Douglas-fir, and western hemlock for 1,100 km. Douglas-fir is capable of growing outside its range but this has been prevented as, without fire, the species cannot colonize new habitat.

On a site basis, post-tire vegetation succession in the Coastal Western Hemlock Zone has been of interest for quite some time. A pioneering study concluded that repeated burning was destroying the seed trees and young growth (Howe 1915). On one-half of the land examined, which had been logged and burned over between 1884 and 1914, the reproduction of new forests was considered inadequate. This was attributed to an increase in the frequency of widespread fires from every 86 years for the period from 15'54 to 1814, to 27 years for the period from 1814 to 1894, to 5 years between 1894 and 1914 (Howe 1915).

Howe (1915) observed that most young Douglas-fir forests were establishing on areas which had been burned or logged and burned. He found Douglas-fir established best on moderately-burned areas and likely required fire to clear away the slash and lesser vegetation. Western hemlock reproduction was encouraged by light surface fires beneath Douglas-fir stands as the moss layer, which usually develops after tire, conserves moisture and is an ideal medium for seedling germination (Howe 19 15).

When established together, western hemlock seedlings may initially outnumber those of Douglas-fir, but the latter species is more robust and has the advantage on exposed post-tire sites with thin duff layers (Agee 1990). Understory vegetation primarily consists of species which sprout from underground rhizomes or from the bud collar (Agee 1981).

By 20 years post-fire most of the snags have fallen and succession has led to a highly diverse shrub stage which supports much wildlife (Agee 1981 and 1990). Shrub and herb cover later decline as the tree canopy develops and closes. Small openings in the canopy resulting from windthrow, snowbreak, insect kill, or disease will be occupied by the more shade-tolerant conifer species, with Douglas-fir limited to the dominant trees which established soon after the tire. If western hemlock did not become established with Douglas-fir during the initial post-tire succession phase, and form part of the main canopy, it may appear after 50 or 100 years have passed (Franklin and Dymess 1973).

After several centuries Douglas-fir density continues to decrease, while the lower crown classes become occupied by western hemlock, western redcedar, and amabilis fir (Agee 1981; Munger 1940; Schmidt 1970; Spies and Franklin 1988). Understory regeneration of western hemlock and amabilis fir may be limited by a dense western hemlock canopy until it starts to break up after 300 to 400 years (Stewart 1986). Decline of the Douglas-fir component may begin at 500 years, or be delayed until after 1,000 (Franklin and Spies 1984).

Munger (1940) and Schmidt (1957) described the later stages of development of these stands when Douglas-fir gives way to the shade-tolerant species. Schmidt's data from 142 plots on Vancouver Island showed initial post-fire recolonization by Douglas-fir, western hemlock, and western redcedar (fig. 2). Amabilis fir is less capable of invading the burned site and depends upon the offspring of its initial colonists to extend itself. As time since fire increases, the species composition shifts away from Douglas-fir and western redcedar to western hemlock and amabilis fir. The long-term absence of fire results in the loss of Douglas-fir.

It has been known for some time that most of the Douglas-Iir forests in the Coastal Western Hemlock Zone are seral to the climax western hemlock and western redcedar (Benedict 1915; Eis 1962; Franklin and others 1981; Howe 1915; Judd 1915; Munger 1940; Wright and Bailey 1982) and long believed that post-fire Douglas-fir forests arc generally even-aged (Franklin and Waring 1979; Judd 1915; Munger 1940).

Indeed, Munger's data showed the Douglas-firs on a 65 ha plot to be nearly all within 25 years of 590 years of age. The western hemlocks of that stand, by contrast, ranged in age from 130 to 525 years. Recent work has found that the range in age of the dominant old-growth Douglas-fir trees may surpass 200 years, indicating a lengthy site reoccupation period (Franklin and Waring 1979; Franklin and others 1979). This may be due to a lack of seed source (necessitating gradual recolonization), vegetative competition (delaying tree establishment), or multiple disturbances (partial rcbums removing portions of the first stand) (Franklin and Waring 1979).

Resource Management Implications

Prescribed burning to remove logging slash and produce conditions amenable to the establishment of Douglas-fir, the preferred species, was considered a necessity (Howe 1915) and has been carried out since the early part of this century.

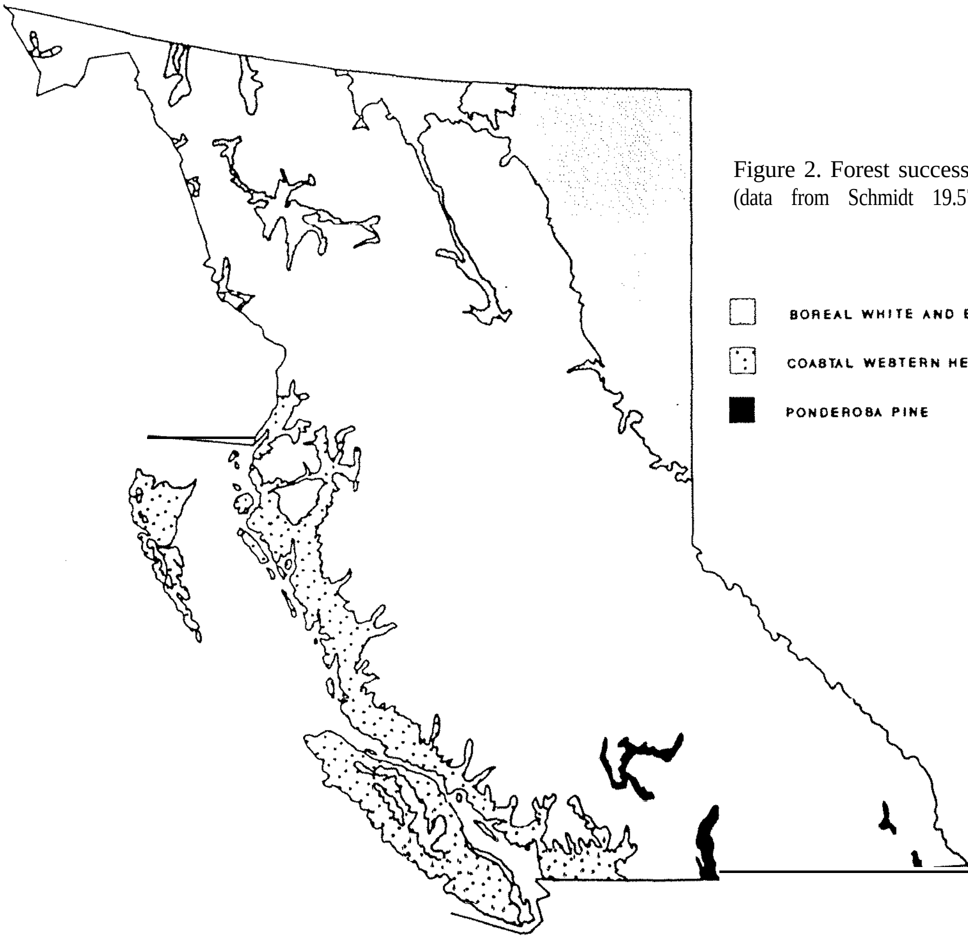

Figure 2. Forest succession on Vancouver Island (data from Schmidt 19.57).

Although compulsory slash disposal was not imposed on logging operators when the Forest Branch was created in 1911, such actions were encouraged. Without prescribed burning after harvesting, it was expected that western hemlock would predominate (Benedict 1915). In northern parts of the zone, where soil conditions were different, slash disposal by burning was not r-commended.

Fire effects in the Coastal Western Hemlock Zone have generally been studied in the context of using prescribed fire for site preparation. Considerable research has been carried out and continues by provincial and federal government agencies, local universities, and forest companies (Beese 1986; Cm-ran 1988; Peterson 1989).

Ecosystem-specific guides describe the proper use of prescribed tire (Haeussler and others 1984; Klinka and others 1984). Recommendations arc made as to whether prescribed fire should be applied to a site and if so, what particular fire effects are suitable.

Concern is growing over the need to preserve islands of old growth timber generally comprised of western hemlock, western redcedar, Douglas-fir and sometimes Sitka spruce (Spies and Franklin 1988). With long interval fires which can grow to significant size, careful planning will be required to preserve these old-growth islands. The need to consider the role of natural tire in their life cycle is obvious.

BOREAL WHITE AND BLACK SPRUCE ZONE

The Boreal White and Black Spruce Zone occupies the low to middle elevations of the northern portion of the province, primarily in the northeast comer (Annas 1983). It has a northern continental climate with long, very cold winters and therefore a short growing season. Discontinuous permafrost exists in some locales.

The Boreal White and Black Spruce Zone is floristically very diverse. Major tree species include white spruce (*Picea glauca*), black spruce (*Picea mariana*), lodgepole pine (*Pinus contorta* var. *latifolia*), trembling aspen (*Populus tremuloides*), paper birch (*Betula papyrifera*), Alaska paper birch (*Betula neoalaskana*), tamarack (*Larix laricina*), and subalpine fir (*Abies lasiocarpa*). Balsam poplar (*Populus balsamifera* spp. *balsamifera*) and white spruce occupy alluvial floodplains.

Fire History

Physical evidence of fire in the boreal forest is found in the form of charcoal deposits in the soil profile; fire-scarred trees and charred snags; as well as in the mosaic character of the forest stands, their age structure, and the morphological and reproductive characteristics of the plant species present (Rowe and Scotter 1973). The forest mosaic may take on the appearance of a patchwork quilt, with each patch being sharply differentiated by abrupt changes in tree species composition and crown heights. Within each component the stand structure and composition may be remarkably uniform (Dix and Swan 1971).

The fire regime consists of short to long interval crown fires and severe surface fires in combination. Stand replacement tires are the norm and surface fires alone are rare, occurring along the edges of crown fires where fire behavior was moderated. Ground fires can be persistent in deep organic soils.

Hazardous tire weather conditions are encouraged by lengthy summer days which result in long diurnal drying periods as well as by extended periods of low precipitation and humidity. In northern British Columbia very large lightning-caused fires have occurred decadally and reached 180,000 ha. Man-caused tires have sometimes surpassed this size during the past 50 years.

Heinselman (1981) summarized fire regimes for the boreal forest. Presettlement fue cycles ranged from 50 to 200 years for types shared by the Boreal White and Black Spruce Zone of northern British Columbia. Fire cycles there differ between forest types, but likely average 140 years.

Since most of the landscape in this zone is subject to fire, stand age class distribution data have been used to model fire cycles and the effects of fire suppression. For northwestern and northeastern British Columbia, fire cycles were 125 and 202 years respectively (Smith 1981). However, a regional tire cycle is a composite, with the components having their own shorter or longer cycles as a function of site type (Rowe 1983).

Although much evidence exists for aboriginal burning and an increased fue frequency due to the ***presence*** of European man in the boreal forest (Lutz 1959), specific references to northern British Columbia are rare. House (1909) told of a hunting expedition in northwestern British Columbia during which his native guide set several grass fires in order to approach a herd of caribou under the cover of smoke.

Lewis's (1982) native informants in northern Alberta described the use of prescribed tire for creating and maintaining meadows, manipulating riparian vegetation, clearing campsites and trails, removing windfalls, and creating firewood. It is likely that such practices were carried out in the boreal forest of neighboring British Columbia, influencing the fire regime there as well.

Lutz (1959) documented fires in the boreal forest caused by European man's escaped campfires as well as his deliberate firing of the forest in order to create supplies of dry fuelwood, signal other parties, drive moose, and promote the growth of forage for domestic stock. Fire was also used to remove local forest cover, thus thawing the permafrost and permitting excavation for minerals (Dawson 1888).

Fire Effects and Succession

The natural role of fire in the boreal forest of North America has been described in detail (Foote 1983; Kayll 1968; Kelsall and others 1977; Lutz 1955; Rowe 1983; Rowe and Scotter 1973; Viereck and Schandelmeier 1980). With stand-replacing crown fires and tree species which are fire sensitive and usually killed by fire, forest regeneration depends on "on-site" or "off-site" adaptations.

On-site adaptations to fue include cone serotiny (lodgepole pine), semi-serotiny (black spruce), root suckering (trembling aspen), and root collar sprouting (paper birches). Species such as white spruce, tamarack, and subalpine fir require live survivors off-site but near enough to provide seed to the burned area. The hardwoods produce prolific amounts of light seed which, if carried by the wind to the burned site, may result in a post-fire hardwood component as well.

Lesser vegetation also possess post-fire reproductive strategies, such as sprouting from roots, rhizomes, and stems, or the production of many light wind-disseminated seeds (Rowe 1983). Some, such as high bush cranberry ***(Viburnum edule)*** and pink corydalis ***(Corydalis sempervirens)*** have seeds that germinate following stimulation by fire (Viereck and Schandelmeier 1980).

Natural regeneration of tree species after fires in the Boreal White and Black Spruce Zone is quite prompt; many forest stands show an age range of only 10 to 15 years, sometimes up to 20. A wave of tree establishment often takes place in the first 5 to 7 ***years. However,*** where initial restocking levels are low and seed source, seedbed, and vegetative competition conditions continue to be favorable, recruitment may continue for several decades (Parminter 1983).

Post-fire changes in floristic composition may be minor, with succession becoming an exercise in changing structure and species dominance. For instance, black spruce - sphagnum moss and black spruce - lodgepole pine types generally exhibit little change in composition as time since fire increases. Even though identifiable stages exist (Foote 1983), vegetation cycling by fire rather than orderly replacement of species through succession is common (Methven and others 1975; Viereck 1983).

In other situations, the proportion of the post-fire stand made up by early successional species, such as trembling aspen, the paper birches, and lodgepole pine, may increase over the pre-fire values for those species. Indeed, they may not have been present at all. Other species, such as white spruce, black spruce, and subalpine fir, will be less represented on a proportional basis until later in the life of the stand.

Although trembling aspen and lodgepole pine overtop white spruce during most of the stand's lifetime, the latter species survives and eventually replaces the former in the absence of fire. In most cases all three species become established simultaneously post-fire (Parminter 1983). A similar situation holds for lodgepole pine and black spruce, and lodgepole pine and subalpine fir mixes. Notably, if fire returns before the more shade-tolerant species (the spruces and subalpine fir) are sexually mature, the more tire-adapted early successional species (trembling aspen, paper birches, and lodgepole pine) will have the upper hand and dominate the post-fire site.

Resource Management Implications

Fire suppression costs in the Boreal White and Black Spruce Zone can be high, the annual area burned large, and the economic value of much of the timber resource relatively low. Therefore, fire effects and post-fire vegetation response are important when making decisions on appropriate wildfire suppression response (as well as in planning for prescribed burning).

Post-fire vegetation development depends on many factors such as the type of pre-tire vegetation present and its state of development; the season of fire occurrence; fire behavior and intensity; the depth of bum; fire size; the nature of the off-site vegetation; physical site characteristics; and post-fire environmental conditions. In spite of all of these variables, the most likely course of post-fire succession can be anticipated.

Post-fire vegetation succession models act as predictive tools to aid in resource management decision-making. After 17 cover types in the Boreal White and Black Spruce Zone were delineated following the approach of Hansen and others (1973), a modification of Kessell and Fischer's (1981) methodology was used to show the multiple pathways of post-fire succession. The possible post-tire outcomes, and the further development of the cover types with and without tire, have been detailed (Pat-minter 1983).

Considerable prescribed burning is carried out to enhance domestic range and wildlife habitat (primarily for large ungulates). Conversion of coniferous to mixedwood stands or of mixedwood to shrub- and herb-dominated types occurs as prescribed burning shortens the fire frequency.

PONDEROSA PINE ZONE

This zone is found at lower elevations in some of the main valleys of the southern third of the central interior. It extends south into the Pacific Northwest states.

Climatically it is the driest, and in summer the warmest, biogeoclimatic zone in the province (Mitchell and Erickson 1983). It is classed as semi-arid continental. The summers are warm and the mean annual precipitation ranges from 200 to 300 mm. Moisture deficits occur during the growing season.

Ponderosa pine (*Pinus* is the predominant tree species, and often forms open park-like stands with an understory of bluebunch wheatgrass (*Agropyron spicatum*). Douglas-fir and trembling aspen occur on moister sites, and western larch (*Larix occidentalis*) is rarer. Grasslands are mixed with the forest cover throughout the zone.

Fire History

Evidence of fire is found as charcoal layers in the soil, charred bark, and fire-scarred trees (primarily ponderosa pine and Douglas-fir). The fire regime most often consists of frequent light surface fires and, rarely, long interval crown fires. The role of fire is to maintain the stand, keeping the understory relatively open and the ground free of excessive woody fuel buildup (Agee 1990). The rarer crown fires in this type open up gaps within the stand which then fill in with a new age class of ponderosa pines.

Charcoal deposited in lake sediments from local and regional grassland and forest fires indicated a fire history going back 300 years or more in the southern part of the Ponderosa Pine Zone (Cawker 1983). Many fires have been lightning-caused A proportion can be attributed to aboriginal prescribed burning, which Barrett and Amo (1982) and Gruell (1985) found to have increased fire frequency in ponderosa pine habitats of the interior western states.

Fire history studies in North America have revealed fire frequencies of about 6 to 15 years for the types of ponderosa pine stands found in British Columbia (Wright 1978; Wright and Bailey 1982). One of the earliest local investigations was that of Melrose (1923). In the southeastern portion of the province he dated 13 fire events between 1729 and 1908, for a fire frequency of 13.7 years.

Dorey (1979), working just above the Montana border, constructed a fire chronology based on 14 fire-scarred trees which indicated a history of surface fires between 1813 and 1940. The fire frequency works out to 6.3 years overall.

Cartwright's (1983) data show a fire frequency of 9 years for a mixed ponderosa pine - Douglas-fir forest near Kamloops, British Columbia. Low (1988) found values of 7.2 and 10.5 years for two areas in the same vicinity.

Fire Effects and Succession

Windfall, insect attacks, mortality, and frequent fire have historically maintained open ponderosa pine and mixed ponderosa pine - Douglas-fir stands (Wright 1978). Tree seedlings established after fire begin the development of an even-aged group. An uneven-aged forest results from this pattern.

Under a natural regime most of the understory grasses, forbs, and shrubs are maintained and enhanced by fire. Low severity fires favor resprouting, while germination from seed is favored after high severity tires (Saveland and Bunting 1988). However, mortality of fire-sensitive understory species, as well as conifers, does take place.

Fuel removal by grazing of domestic stock and fire suppression has significantly affected tire frequencies, either lengthening them substantially or removing the influence of tire altogether. Dorey (1979) found the fire frequency to have been 5.9 years prior to, and 9.6 years after 1911. No fires took place since 1940. Low (1988) found *an* area with a tire frequency of 10.5 years between 1672 and 1900 which had not experienced fire since 1901. Another with a fire frequency of 7.2 years between 1774 and 1933 had not had a fire since 1934.

Early accounts of the ponderosa pine forests described them as being fairly open and interspersed with large areas of grassland. Crown tires seldom occurred in these types (Whitford and Craig 1918). With fire exclusion, such stands become dense, many younger trees establish in the understory, and tree growth stagnates (Amo 1988). Total fuel loading becomes higher and the amount of ladder fuels increases dramatically, increasing the possibility of crown tires (Steele and others 1986). Douglas-fir often invades the lower canopy and becomes predominant, where before it had been absent or minor (Amo 1988; Dorey 1979). Understory vegetation becomes depleted due to the denser tree canopy.

Resource Management Implications

Historical sentiment against underbuming these ponderosa pine forests related to the death of young seedlings, loss of timber production, scarring of trees, and depletion of soil nutrients (Melrose 1923). The opinion of the Forest Branch was that a continued role for surface tires in these stands would be an economic misfortune (Forest Branch 1923).

The ecological consequences of the exclusion of fire from these stands have been recognized in the past two decades. Prescribed burning is now carried out in many pondcrosa pine and mixed ponderosa pine • Douglas-fir forests to maintain and enhance domestic range and wildlife habitat. Other reasons include fuels reduction (especially in the urban/wildland interface zone) and visual resource management. Prescribed tire may be combined with mechanical treatments, such as spacing, thinning, pruning, or selection logging.

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ANTHROPOGENIC FIRE AND TROPICAL DEFORESTATION: A LITERATURE PERSPECTIVE

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Abstract—Anthropogenic fire is an important agent of tropical destruction. Fire has been the primary tool of rural and technologically primitive people for managing and manipulating their natural environment For more than a century observers and researchers have recorded and documented the process of deforestation by fire throughout the tropics. Using the cumulative written work of these investigators, we describe how human-caused fire in the tropics has contributed to deforestation. Human use of fire cannot be eliminated, nor is its elimination necessarily desireable; however, it must be managed. Education of fire users must form the core of any fire management program. Until an effective fire management policy can be implemented, anthropogenic fire will continue to contribute to tropical deforestation.

INTRODUCTION

Fire in nature is an awesome force. In human hands it has become our most powerful tool for altering the face of the earth (Sauer 19.58). Today, as it has been since mankind first learned to use it, fire is the primary tool of rural and technologically primitive people for manipulating and managing their natural environment (Stewart 1956).

Unfortunately, fire has become one of the most important agents in the worldwide destruction of tropical forests, and 98 percent of tropical fires arc anthropogenic, started by some intentional or unintentional human action (Batchelder 1967). Anthropogenic fire has been identified as the most serious threat to Venezuela's forests (Budowski 19.5 1; Camero-Zamorra 1952). In India human-caused fire has been responsible for the loss of much of the subcontinent's forests (Troup 1926). On the Indonesian island of Java periodic, long-continued human firing of vegetation has been the greatest menace to its forests (Shuitcmaker 1950 in Bartlett 19.55). Through repeated burning the tropical forests around the world have been reduced in area and often replaced by savannas and grasslands (dc la Rue 1958; Batchelder and Hirt 1966; Walter 1971).

For more than a century observers and researchers have recorded and documented tropical man's often careless and negligent use of fire. Using the cumulative written work of many of these scientists, we have put together a general scenario, based upon a myriad of specific cases, about how fire has contributed to forest destruction in the tropics.

Batchcldcr and Hirt (1966) wrote:

> because the number of interrelationships among fire, man, and [the] environment are nearly infinite; no one condition or set of conditions can be assumed to be dominant for all parts of the tropics.

We understand and heartily endorse this observation. As a result of the pantropical view we have taken, our generalizations are not necessarily appropriate to all tropical ecosystems, macrosites, and microsites. The information we arc providing, while valid in general, must be checked against site specific conditions. Undoubtedly, tropical forest land managers and policy makers have a great and present need for extensive and intensive research about fire behavior and fire effects in their forests.

TROPICAL FOREST LOCATION

Tropical forests are located in lowland elevations -- generally below 1300 meters -- of the large, global belt around the equator, primarily between the Tropics of Cancer and Capricorn. Where tropical climatic conditions extend beyond the north and south tropical latitudes, so do the tropical forests. The forests are concentrated in Africa and the Americas with more than half of the closed forests located in South and Central America. The forests of the Amazon River drainage account for the bulk of the New World's closed forest. In contrast the vast majority of open tropical forests are located in Africa (See table 1).

TROPICAL DEFORESTATION

Estimates of tropical forest area range from 15 million to 19 million square kilometers -- approximately 42 percent of all tropical land and 13 percent of the earth's land surface. Closed forests -- those with a continuous canopy -- account for about 9 to 1 I million square kilometers (UNESCO 1978). Deforestation rates for the closed forests arc largely hypothetical and vary with the estimator and the definition of deforestation used. Stated loss rate figures run from 100,000 to 245,000 square kilometers per year (Myers 1981).

Geographically, the losses are pantropical. The pressure on the forests is greatest at the forest edge where they are most accessible. This edge can be a broad ecotonal belt or an abrupt boundary.

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Table I. Global. location of tropical forests by percent of total tropical forest area (U.S. Interagency Task Force 1980).

Type of Forest (pct)	Location
All Tropical Forests (1.9 billion hectares)	
42 percent	South and Central America
37 percent	Africa
21 percent	Asia/Australia/Oceania
Closed Tropical Forests (1.1 billion hectares)	
SO percent	South and Central America
30 percent	Africa
20 percent	Asia/Australia/Oceania
Open Tropical Forests (0.8 billion hectares)	
64 percent	Africa, Asia, Australia, Central and South America

Primary dcforcstation activities are industrial logging, fuclwood gathering, and agricultural clearing. Of these thrcc, dcstruction associated with the forest fanner is the most important; and thc forest farmcr's primary tool is fire(Nichols 1901; Budowski 1956; Dcnevan 1978). Some overlap in dcforcstation activities does exist. For example forest farmcrs often move into loggcd areas to establish their farm plots.

In using thc tcrm forest fanner we mcan the traditional shifting agriculturalist as wcll as the modem "pioneer' farmcr-settlcr. Development, greatcr rural population densities, and the dcmands of a market economy have in many places changed subsistence shifting agriculture so much that ecologically it is not much differcnt than the farming practices of the pioneer settlcrs.

FLAMMABILITY OF TROPICAL FOREST TYPES

In the tropics the climatic regulators of vegetation type are the amount of rainfall during the wet season and the duration and intensity of the dry season (Walter 1971). Three general forest types are relevant to our look at anthropogcnic fire and dcforcstation. In order of most rainfall and shortest dry season they arc: the tropical rain forest, the semi-deciduous forest, and the deciduous forest (Mueller-Dombois 1981). Deciduous forests can bc further divided into moist and dry forests (Walter 1973). The moist deciduous forests receive more rainfall than the dry deciduous forests and grow on soils with greater water-holding capacity. Rain, semi-deciduous, and moist deciduous forests are closed forest types. Dry deciduous forests are open types.

The three closed forest types arc fire independent ecosystems (Vogl 1977). Successful natural ignitions are rarely sustained. Low light levels on the floor of the closed forest during the growing periods prevent growth of significant amounts of hcrbaccous vegetation. High temperatures and moisturc levels promote rapid decay of litterfall. Furthcrmorc, low fuel loading and large fuel size, hiyh moisture content, and wide fuel spacings, combined with the high ignition temperature typical of most tropical forest fuels, mitigate against prolonged, continuous combustion in the undisturbed closed forest (Batchelder and Hirt 1966; Trollope 1980). Open forests do have herbaceous and shrub understories because light penetrates to the forest floor.

Once the undisturbed closed forest is cleared, though, xeric undcrstory and overstory species become established on the site and at the forest edge. This secondary vegetation is more likcly to sustain combustion if ignitcd. After fire has invaded the forest once, the forest remains in constant danger from human sct fires (Spurway 1937; Aubreville 1947; Budowski 1966).

Flammability of Tropical Rain Forests

Thc tropical rain forest is evergreen or mostly evergreen. Though there is no true dry season, there can be a relatively dry period in this forest type. Even in such a "dry" period precipitation avcrages at least 100 millimeters per month. Mean annual temperature hovcrs around 24 degrees Celsius. Decomposition rates are rapid, so there is only a thin litter laycr on thc forest floor at any given time. primarily due to its high moisture level, the undisturbed tropical rain forest is almost fireproof. Once the forest is opened, though, it bccomcs susceptible to burning. Dead and live vegetation in thc opening and at the forest edge quickly dry upon exposure to the intense tropical sun, thus becoming more flammable. Under thesc circumstances the tropical rain forest will burn.

Flammability of Semi-deciduous and Deciduous Forests

The semi-deciduous and deciduous forests are seasonal forests because they exist in the alternating climates of wet summers and dry winters. The number of dry months varies from roughly three to six in the semi-deciduous forest, six to eight in the moist deciduous forest, and eight to tcn in the dry deciduous forest (Walter 1971). Mean temperatures range bctwccn 20 and 28 degrees Celsius.

Thc alternating wet and dry seasons is conducive to fire occurrence. During the wet season there is substantial plant growth which cures in the dry season. The deciduous plants not only dry out, but also shed their leaves. Nevertheless, like the rain forest, the semi-deciduous forest ordinarily does not bum easily. If this forest is opened to the sun's insolation, however, its vegetation dries, becomes more flammable, and, therefore, more likely to bum if exposed to an ignition source.

The deciduous forest is more susceptible to burning in its unaltered state (Walter 1971, 1973). As the dry period grows longer and more scvcrc, greater numbers of deciduous tree spccics and fewer evergreen spccics occupy the forest while total tree density decrcascs. The deciduous forest, especially the open forest, has greater fuel accumulations due to abundant litterfall and the curing of grasses and other low understory vcgctation during the dry season. Exposed to the tropical heat, thcsc fuels quickly dessicate. Under these circumstances there arc sufficient fuels of small size, low moisture content, and close spatial arrangement to carry a surface fire when fire enters the forest. As the dry period grows longer and more intense, the potential fuels become more flammable and the fire danger is magnified.

The fuel complex and fire potential of tropical forests arc altered by human activity that removes the vcgctation, thus creating openings in the forests. After the initial clearing fuel loading increases in all size classes, scattered evenly over the site. After a variable period of ***exposure*** to solar heating thcsc fuels dry sufficiently to burn. In the following ***years*** grasses and other hcrbaccous vegetation, as well as woody spccics, occupy the site as pioneers in secondary succession. This vcgctation, adapted to drier, open environments, grows more or less evenly over the site and dries rapidly in the absence of precipitation. These early successional conditions provide a fuel complex that will bum without cutting and extended drying (Vogl 1969; Walter 1971). If disturbed forest sites are rcpcatcdly burned, highly flammable, quickly rcgcncrating grasses such as Imperata species rapidly dominate the site; creating an easily burned, self-replacing fuel complex.

THE PROCESS OF DEFORESTATION BY FIRE

Anthropogcnic fire is concentrated in the open and ecotonal areas of the tropics where humans live and work. Annually, sometimes more often, and usually at the end of the dry season, residents set their fires. Generally, these are surface tires which are carried by ground litter and herbaceous and shrub vegetation. Fire intensities are usually low due to low available fuel loadings, high fuel moisture and relative humidity, and discontinuous fuel spacing. Fire fronts are typically shallow and narrow. Areas burned arc normally small and patchy (Batchelder and Hirt 1966).

These tires pose two major problems important to deforestation. Foremost, with some exceptions, few people make any attempt to control their fires. They simply rely on the low flammability of surrounding green vegetation to contain the fires. This lack of concern about fire control is all too common throughout the tropics.

The second problem is a result of the first. Too many fires escape the intended burning area. Cook (1909) observed that tires were usually allowed to spread wherever fuels would carry them. As the population density of forest farmers increases, abandoned and productive agricultural plots remain in close proximity (Dcncvan 1978). Abandoned plots arc frequently composed of xcric secondary vegetation that readily bums during the dry season. Fires intentionally set in the productive plots accidentally bum into the nearby abandoned plots. These escaped, uncontrolled, human set wildfires eventually spread into the adjoining forest. This scenario accounted for nearly 100,000 hectares of wildfires in Mexico's eastern Yucatan Peninsula during the summer of 1989 (National Fire Protection Association 1989; Garrett 1989).

The actual process of deforestation by fire has been dcscrihcd by several authors from Cook in 1931 to Mueller-Dombois in 1981. Fire originating in adjacent open areas burns to the forest border. Depending on the intensity of the ***fire,*** density of the vcgctntion at the forest edge, fuel loading, and forest moisture conditions, fire may or may not penetrate the forest. Under normal circumstances fire will not enter a rain forest, but fires can burn from several meters to 1 or 2 kilometers into the scm-deciduous and deciduous forests. Once at or within the forest cdgc, fire intensity lessens as available fuel dccrcascs and relative humidity increases. Fire damage is usually minimal. Hcrbaccous growth, coppice stumps, low bushes, suckers, and seedlings are killed and varying amounts of duff and litter arc removed by the fire. Saplings and some fire sensitive species may be killed. Also, larger trees may be scorched or scarred around the butt.

The killing of undergrowth and trees in the burned forest area opens more of the forest to direct sunlight. Grasses quickly establish and rapidly grow in the sunlit ***areas. In*** forests which adjoin annually burned savannas or which surround annually burned openings, the invading undcrstory vcgctation provides the fuel that will allow the next season's fires to spread farther into the forest. Furthermore, opening of the forest edge to greater sunlight alters the edge microclimate to a drier type which also may contribute to increased intensity of the next fire.

LONG-TERM EFFECTS OF UNCONTROLLED FIRE USE

The forest vegetation shields the soil and the site from the drying effects of the sun and wind. Repeated fires open forests and expose the forest soils by removing living vegetation and litter. Soil surface tcmperaturos rise and relative humidity decreases in response to direct solar exposure. Addition of new organic matter is reduced. Given these conditions, the closed nutrient cycling system of the tropical forest is damaged. Movement of essential nutrients to the forest vegetation is interrupted (Richards 1951). Wind and solar insolation dessicate the cxposcd soil and contrihutc to increased evapotranspiration which further reduces soil moisture. Microorganism populations shrink as organic matter content causes reduced nitrogen fixation and nutrient mineralization. Soil impoverishment is the result (Camero-Zamorra 19.51).

The combined effect of repeated firing, insolation, and torrential rains is a breakdown in soil structure. Under these forces the soil disaggregates and compacts (Pittier 1939; Jha and others 1980). Soil density increases and porosity decreases leading to reduced soil moisture holding capacity. In the oxisols, ultisols, inceptisols, and the red earths typical of the humid and seasonal tropics a hardpan may develop if cxposcd to repeated wetting and drying (de la Rue 1958; Donahue and others 1977). Once exposed to the elements, erosion of the fragile topsoil becomes a serious problem.

With progressive opening of the forest microclimate warms and dries, soil moisture and fertility decline, and less demanding woody and herbaceous plants adapted to drier conditions become established (Budowski 1956). Forest vegetation that survives the repeated fires or degraded site conditions lingers on singly or as relict groves and gallery forests (Batchelder and Hirt 1966). Forest regeneration that overcomes the poorer site conditions is either killed during regular burning or suppressed by the invading vegetation which is more fire-resistant (Innes 1971; Vogl 1977). Eventually, even the most fire-resistant woody species are eliminated. At this point the forest site is totally degraded and deforestation is complete. Continual firing will prevent the return of forest growth to formerly forested sites.

MANAGEMENT OF ANTHROPOGENIC FIRE

The challenge is to halt haphazard fire use without prohibiting rational use of fire for legitimate agricultural and non-agricultural land management. However, since fire is the primary land clearing tool of most agriculturalists, implementation of a rational fire management program will be a difficult and sensitive task. Human use of fire can never be successfully eliminated, nor is its prohibition necessarily desirable. A fire management program, while serving to preserve and protect tropical forests and human welfare, must respect the basic rural cultural foundations on which burning rests. It must also allow for inevitable and unavoidable economic use of forests and grasslands.

Governments of countries with tropical forests have become more aware of the forest fire problems they face and of the need to protect their forest resources. We suggest that the first task in dealing with the fire problem should be the formulation of a national fire policy to provide a framework for further actions. Every nation has a unique fire situation, and each nation's fire policy should reflect that uniqueness. In every case, though, the all-important human dimensions of fire must be addressed. Human-caused fires are preventable, but fire policy that threatens traditional land uses will ultimately fail.

To bring anthropogenic fire under control the affected people must understand and support management programs. Education is essential to impart understanding and to change the attitudes of fire users toward fire and the environment. Hand in hand with an educational effort, an agricultural or forestry extension program could be established to instruct users about the correct application of fire and its positive and negative effects on the land and its vegetation. The negligent and indifferent fire habits so common in the tropics must be reformed for a program of managed fire to be successful.

Education is a long-term solution to a pressing problem; nevertheless, the best of intentions, the finest policy statement, and the most modern science and technology will avail us little if the attitudes of fire users remain unaltered. Mr. Helmut Haufe, FAO Regional Forestry Officer in Latin America, stated that anthropogenic wildfires in too many cases are "due to the lack of a proper information and instruction system" (pers. comm. 1981). Mr. Haufe's statement is a significant endorsement for an extensive and vigorous fire education program.

Other actions that may help manage the anthropogenic fire problem are more appropriately considered under the headings of agricultural and rural development, but bear mentioning here. These actions include providing alternatives to traditional agricultural practices and incentives to take up the alternatives. The already mentioned extension programs could help develop and promote alternative methods and technologies. Land use, tax, rural development, and internal colonization policies can also he adjusted to reduce the motives for negligent and abusive fire use.

CONCLUSION

In 1967 Batchelder wrote that the "use of fire in the tropical world is no longer in a stage of 'ecological climax' wherein a stable, harmonious relationship to the environment exists." In fact over the past hundred, if not several hundred years, the careless and repeated use of tire has resulted in ecological disturbances which have steadily forced the retreat of tropical forests worldwide. Natural succession, which normally heals ecological wounds and returns ecosystems to their prcdisturbance states, has been halted by the frequent and often devastating nature of anthropogenic tire. Insofar as tropical forest regeneration is concerned, continuous anthropogenic fire disturbances are unnatural and, therefore, beyond the adaptive and recuperative powers of forest ecosystems. Repeated burning leads to replacement of the original vegetation by a series of seral communities more easily burned until a tire disclimax community is finally cstablishcd.

In a paper delivered to the West Indian Agricultural Confcrcnce in 1901 Nichols called for an immediate end to uncontrolled burning in the tropics. Ninety years later a great variety of voices still echo his call. Since 1901 efforts to reduce the frequency and effects of anthropogenic fire in the tropics have failed. The burning continues. Effective tire management policies, that are strong on education must be developed and implemented. Without them increasing population, development, and colonization pressures within the tropical forest regions will assure the unabated cultural use of fire in its present destructive form.

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WILDFIRE IN THE PALEOZOIC: PRELIMINARY RESULTS OF A CASE STUDY ON THE FIRE ECOLOGY OF A PENNSYLVANIAN FLOODPLAIN FOREST, JOGGINS, NOVA SCOTIA, CANADA

Nan Crystal Arens[1]

Abstract-Sediments of the Joggins Section, Cumberland Basin, western Nova Scotia accumulated in lowland forests approximately 300 million years ago. The 4,000-meter-thick sedimentary sequence includes channel and overbank sandstones interbedded with stacked immature paleosols developed in thick mudstone horizons. Fusain (charcoal) is common in all sedimentary facies present at Joggins. Fusain occurs as isolated macroscopic clasts and also as recognizable layers of clasts and particles within the mudstone/paleosol facies. These layers are interpreted as tire-event horizons. In one example, an ancient tree, rooted below a tire horizon, is preserved with its charred periderm intact. Palynological analysis shows that there is an increase in taxonomic diversity immediately following this tire. Although arborescent lycophytes remain dominant in the palynoflora, cordaitean and medullosan gymnosperms enter the postfire community. These groups decline later as prefire vegetation is reestablished. Numbers of fern-spore species increased following fire, although ferns did not become more abundant. Studies of fire-related floristic patterns preserved in the fossil record can test ecological generalizations and theory derived from present day ecosystems and may help ecologists understand the role of changing fire regimes in long term vegetational change.

INTRODUCTION

The classic image of Carboniferous lowland ecosystems is a static and steamy tropical wetland. Groves of arborcsccnt lycophytcs with their understory of tree ferns and ptcridosperms (seed-bearing plants with fern-like foliage) stand in ever-wet soils, while along the riverbanks clumps of sphcnopsids flourish. Stutzcr and Not (1940) wrote: "It is difficult to believe that such a thing [wildfire] happened in view of the moist condition of the Carboniferous forests. Judging from the plant associations that grew there, fires could not have spread rapidly in a swamp forest." Howcvcr, Izlar (1984) reported that portions of the Okefcnokcc Swamp-Marsh Complex bum every 25 to 30 years, and that this fire regime maintains the floral composition and hctcrogcneity characteristic of the ecosystem. As our understanding of wetland ecosystem ecology develops, we arc called to rcevatuatc our view that ancient swamp and floodplain ecosystems were static. We must reexamine the role of fire in these systems.

For this reason, I have begun a study of community dynamics and fire ecology in the Joggins Section (Middle Pennsylvanian) of western Nova Scotia, Canada. This paper reports on the initial study testing the feasibility of detailed ecological reconstruction of this ancient ecosystem. In this paper, I will: (1) show that ecological-scale resolution of the fossil record is possible in this stratigraphic section; (2) establish that wildfire was a significant factor in this ancient ecosystem; and (3) demonstrate that wildfire may have, in part, controlled the distribution of some floodplain plants.

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GEOLOGIC SETTING

The Joggins section is located in the Cumberland depositional basin of western Nova Scotia, Canada (fig. 1). Approximately 4,000 meters of sediment arc well exposed in a continuous cliff outcrop along the eastern shore of Chignecto Bay. The outcrop face is approximately perpendicular to strike; bedding dips range from 15" to 20" south. Sediments are Middle Pennsylvanian (approximately 305 m.y.b.p.), and have been biostratigraphically datcd as latest Wcstphalian A through earliest Westphalian C using miospores (G. Dolby, unpublished data). Joggins sediments are well correlated with other terrestrial and marine deposits in the North American midcontinent, the Appalachians, and Europe (Phillips and others 1985).

The 4,000 meters of sediment exposed at Joggins record 2 to 5 million years of history (Harland and others 1989). The inferred rapid sedimentation rate suggests that decade-scale or liner stratigraphic resolution may be possible at Joggins. Such line time-scale stratigraphic resolution is essential for the study of community dynamics and responses to disturbance in an ancient ecosystem. Without adequate time-stratigraphic resolution the record of ecological-scale processes will be obscured by the homogenizing effect of slow sedimentation.

The Joggins stratigraphic sequence consists of channel, levee, and overbank sandstones, floodplain mudstones, and thin coals probably attributable to floodplain ponds. Fluvial tics were deposited in an anastomosing river system that drained the Cobequid Highlands to the present-day southeast (Rust and others 1984). Thin, organic-rich lacushine limestones also occur in the lower portions of the section. I have observed

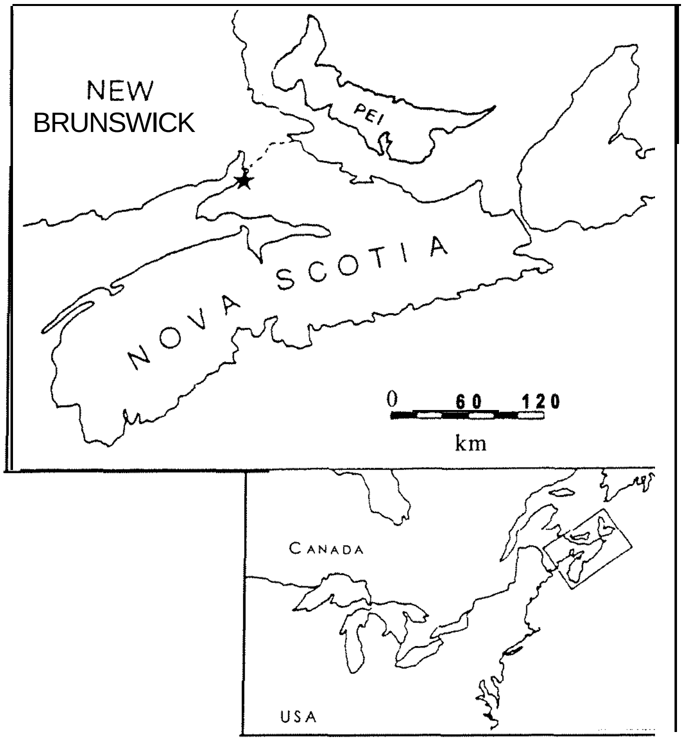

Figure 1 .--Location of the Joggins Section area.

no cvidcncc of marine facies or marine influence on tcrrcstrial facies at Joggins. Within the Joggins Section scdimcnts tend to redden and sandstones to coarsen up-section. Rust and others (1984) concluded that these trends indicated a tectonically mediated steepening of the alluvial plain, which lowered relative water table and increased the amount and grain-size of transported sediment.

For this study, the clastic floodplain mudstonc/palcosol facics arc of particular interest. Mudstone horizons vary from 1 to 20 meters in thickness and are intcrbcddcd with crevasse-splay and channel sands. In lower portions of the section, mudstones and paleosols constitute 70 to 80 pcrccnt of the stratigraphic section. Color of mudstoncs varies from gray-green lo mottled to gray-red or red up section, corresponding to the infcrrcd water table gradient. Despite this gcncral trend, there is substantial color variation on a finer stratigraphic scale within the scction. Mudstones are commonly very poorly-bedded to structureless with moderate to well-developed immature palcosols. Paleosols are characterized by root traces and localized root casts, sidcritc rhizoconcretions, and locally developed soil structures. Recognizable soil horizons are commonly absent or poorly developed; paleosols gcncratly have uniform texture vertically. Upright trccs arc commonly prcscrved as partial sandstone molds within mudslonc facies. These trunks may be partially tilled or cast in mud, commonly with a rind of vitrinous coal, which is the prcscrvcd remains of the lycophytc trcc's thick pcridcrm.

Together, these observations suggcst that floodplain mudstones wcre dcpositcd in frcqucnt (gcologically speaking) flooding events that emplaced small amounts (ccntimctcrs) of fine-graincd sediment, rather than in catastrophic events depositing tens or hundreds of centimeters of sediment. Under this depositional regime, clastic swamp trccs would not be substantially disrupted by sedimentation events, but, continual redefinition of soil surface by sediment input would cause successive readjustment of soil profile and lead to the observed lack of well-developed horizons. This model fits well with observations of modem tropical rivers such as the Brahmaputra in India and Bangladesh (Coleman 1969) and the Orinoco in Venezuela (Armando Torres, 1991, personal communication).

During deposition of Joggins scdiments, the Pennsylvanian Maritimes Basin was near-equatorial and had a tropical to subtropical climate (Rowley and others 1985, Galticr and others 1986). Ziegler and others (1981) reconstructed this region at about 1.5" north latitude and in a zone of Easterlies. Peat-forming ecosystems at Joggins and in the Springhill coal field to the east are interpreted as groundwater-influcnccd (rhcotrophic), rather than raised, bog systems (Caldcr and others in prcss). Groundwater-fed systems occur in the East African rift, where availability of moisture varies seasonally (Cecil and others 1988) If rhcotrophic bogs occurred in Nova Scotia during the Pennsylvanian, it is possible that they, too, formed under conditions of seasonally varying moisture availability. However, Cecil and others intcrprctcd the Westphalian B as equitably and adequately wet, based on the rclative abundance of ombotrophic versus rheotrophic bogs in the central Appalachians. Conversely, Phillips and Peppers (1984) interpret the Westphaiian B as drier and more seasonal than the preceding and succeeding epochs based on swamps in the North American midcontinent. Rowley and others (1985) reconciled these interpretations by suggesting that an increasingly monsoonal climate coupled with the orographic effect of the developing Appalachian highlands to the southwest of the Canadian Maritimes Basin created regionally different rainfall patterns across the Euramerican coal province. In the Maritimcs Cumbcrland Basin, higher paleolatitude and the presence of highlands to the east could further intensify such a regionally seasonal climate.

The Joggins fluvial sandstones provide direct scdimenlological evidence for fluctuating discharge that suggests seasonal rainfall. Multistoried sandstones, graded pebbly sandstones, scour fills, conformable mud drapes, and fan shcctflow in ovcrbank deposits all point to variability in stream discharge, which is consistent with the monsoonal interpretation. However, it is impossible to determine from scdimentological evidence alone whether flood events occurred with seasonal or decade-scale frequency. By cithcr intcrprctation, though, the ancient landscape at Joggins experienced periods when evapotranspiralion exceeded prccipilalion, thus allowing fuels to dry sufficiently to permit wildfire. In this respect, the climate-induced fire regime may have been quite similar to that in the Okefenokec swamp (Izlar 1984) or the Orinoco floodplain.

EVIDENCE OF WILDFIRE ON THE JOGGINS FLOODPLAIN

The presence of fire in the Joggins ecosystem is inferred from the presence of fusain (charcoal) throughout Joggins sediments. Fusain occurs at Joggins in several sedimentary contexts. In coals, fusain occurs as discrete layers that are laterally continuous for meters on the scale of the outcrop. Within these horizons, fusain may occur as discrete, macroscopic clasts or in a mechanically ground, powdery form. Within the mudstone/paleosol facies, fusain also occurs as clasts distributed in discrete, sharp-bounded horizons that can be traced laterally for meters along the exposed outcrop. Clasts occur in a matrix of fmely-ground charred material with varying amounts of clay. Fusain clasts varying from a few millimeters to 50 centimeters (fig. 2 a-c) are also common in the sandstones and siltstones associated with channel facies. The largest fusinized logs show a perpendicular surface fracture pattern consistent with charring by fire. In the reddest sandstone facies, charred logs arc common in mud drapes of channel fill and in point-bar deposits.

The origin of fusain and related materials described from coal macerals has been the subject of much debate between workers who favor a pyrolitic origin for this material and others who believe fusain is produced by some unknown slow oxidization process. Scott (1989) reviewed the evidence and arguments and concluded that most fusain found in the Paleozoic and Tertiary rock record is the direct result of surface burning of vegetation or other surficial organic material. Ting (1982) agreed: "Fusinite and scmifusinitc arc derived primarily from woody tissues charred or partially charred during swamp fire. Once charred, the fusain progenitor--charcoal--becomes extremely stable and inert to any chemical and biochemical attack and is thus well preserved. Some coal beds may contain 20-25 percent fusinite, occurring in numerous fusain bands that suggest frequent swamp fires during peat accumulation." Cope and Chaloner (1985) added that wildfire was an important ecological factor since the evolution of a land flora in the Silurian and Devonian. In accordance with these conclusions, I adopt the charcoal interpretation of fusain and will refer to fusain as fossil charcoal.

METHODS FOR POLLEN ANALYSIS

The samples analyzed were collected stratigraphically above and below a horizon of fusain clasts, mechanically disaggrcgated charcoal, and clay associated with the preserved stump of an ancient tree (fig. 2 d-e). The stump, which is located about 150 meters south of McCarren Creek (Rust and others 1984), is preserved in mudstone/paleosol capped by an ovcrbank sand body. The locality is in the lower portion of the section measured by Rust and others (1984) but is not noted in their published stratigraphic section, probably because cliff-face erosion had not yet exposed the stump at the time of their field work. When collections for this study were made (August, 1989), the stump was badly eroded, but periderm material preserved as fusain rather than vitrinite was clearly visible and the outline of the enlarged base of the tree was easily traceable to a well-defined and laterally continuous horizon of fossil charcoal.

Based on the abundance of centimeter-sized fusain clasts within the mudstone/paleosol facies, this horizon is interpreted as a fire event horizon that records a single wildfire in an ancient stand. This conclusion is based on analogy with studies of Recent sediments, which show that even in lakes, where in situ deposition is less likely than on the floodplain environment, peaks in charcoal abundance can be correlated with single fire events within the drainage basin and charcoal clast size can be related to transport distance (Clark 1988, 1990).

A mudstone sample from below the charcoal horizon sampled prefire vegetation. Stratigraphically successive samples were taken at l-centimeter intervals above the fusain layer. Samples were processed according to standard palynological technique (e.g. Traverse 1988). Rock was degraded in concentrated hydrofluoric acid, unwanted organic material was oxidized with HCI and bleach, clay was removed by heavy liquid separation with ZnCl. Strew slides were made with glycerin jelly. Each slide was scanned systematically and each palynomorph encountered was recorded by genus. Species-level diagnosis was made only for the most abundant spore genus, Lycospora, the spore of several arborescent lycophytes. In each sample, 400 palynomorphs were counted and taxa relative abundances were calculated. Following the count, each slide was scanned for additional rare forms. Relative abundance is plotted by stratigraphic position to yield standard pollen diagrams (figs. 3 and 4).

When interpreting a palynological analysis, one must keep in mind several caveats. First, differing quantities of pollen and spores are produced by different taxa. Wind pollinated plants, for example, produce prodigious amounts of pollen and spores while partially or wholly entomophilous plants will be relatively underrepresented in the dispersed pollen and spore record. This caveat is traditionally reconciled by admitting that palynomorph relative abundance cannot be translated directly into quantitative stand measures such as standing biomass or DBH. In the modem pollen record, differential spore production can be a significant confounding factor. However, entomophily was probably less important in the Carbonifcrous than among modem angiosperms. Consequently, one might expect pollen and spore relative abundance to be a better proxy for individual plant relative abundance in these ancient forests. It is also possible to develop conversion factors that will allow pollen and spore abundance to better approximate other measures of stand composition. This approach has been successfully applied to Pleistocene and Rcccnt stands (Davis and Goodlett 1960).

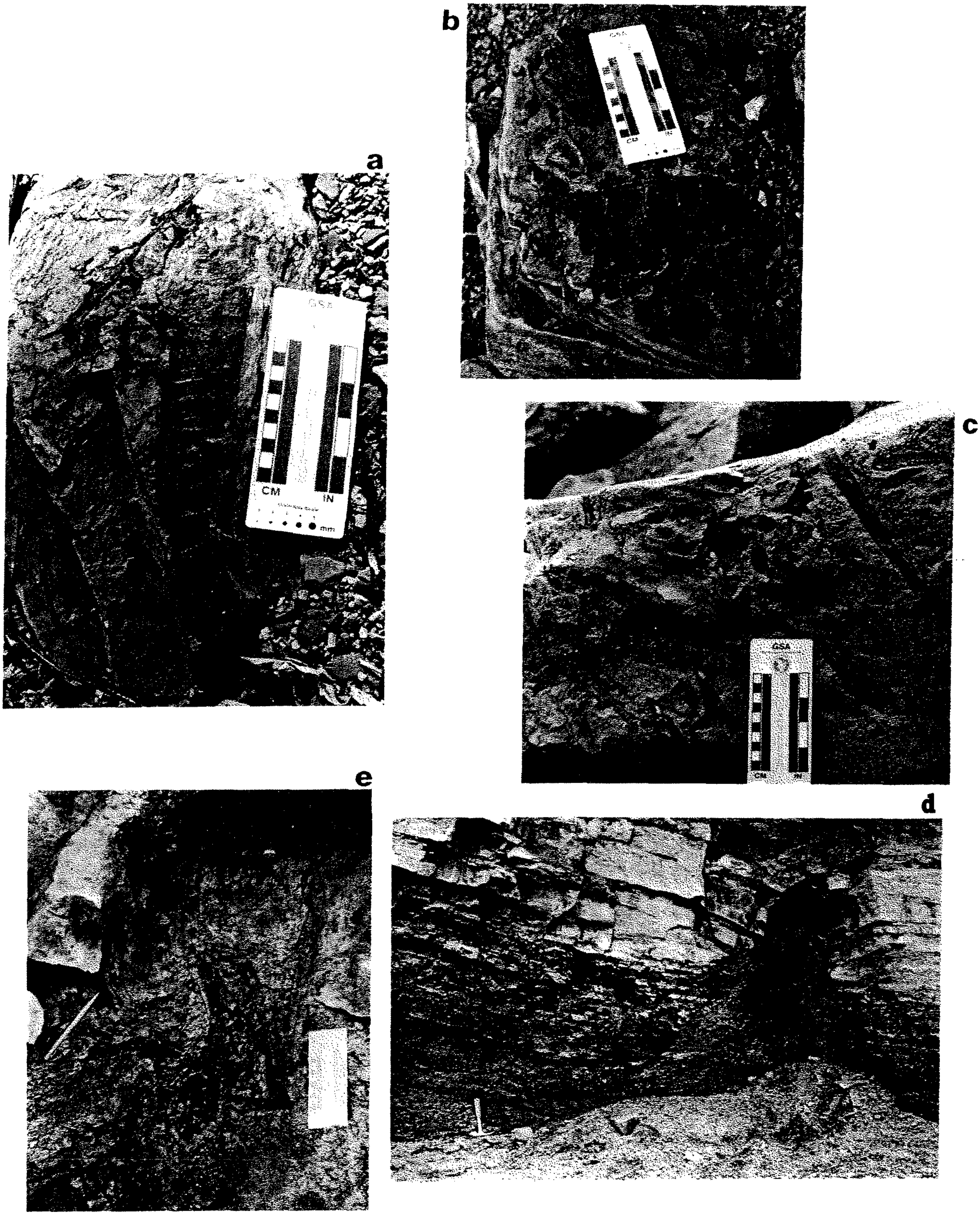

Figure Z.--Physical evidence for fire in Joggins Section rocks. (a-b) Compressions of fusinized (burned) wood in fine-grained sandstone associated with a channel deposit, Spicer's Cove. Note perpendicular fracture pattern characteristic of burned wood. (c) Charcoal fragments and impressions of unburned plant axes in channel sandstone. Arrows indicate charcoal fragments. (d-c) Tree cast with burned peridenn. (d) Extensively eroded outline of tree stump showing outline of tree base extending to dark, charcoal-rich horizon (at hammer shank). Palynological samples taken one meter to the left of hammer position. (e) Close-up of burned trunk. Pen and arrow indicate fusinized material.

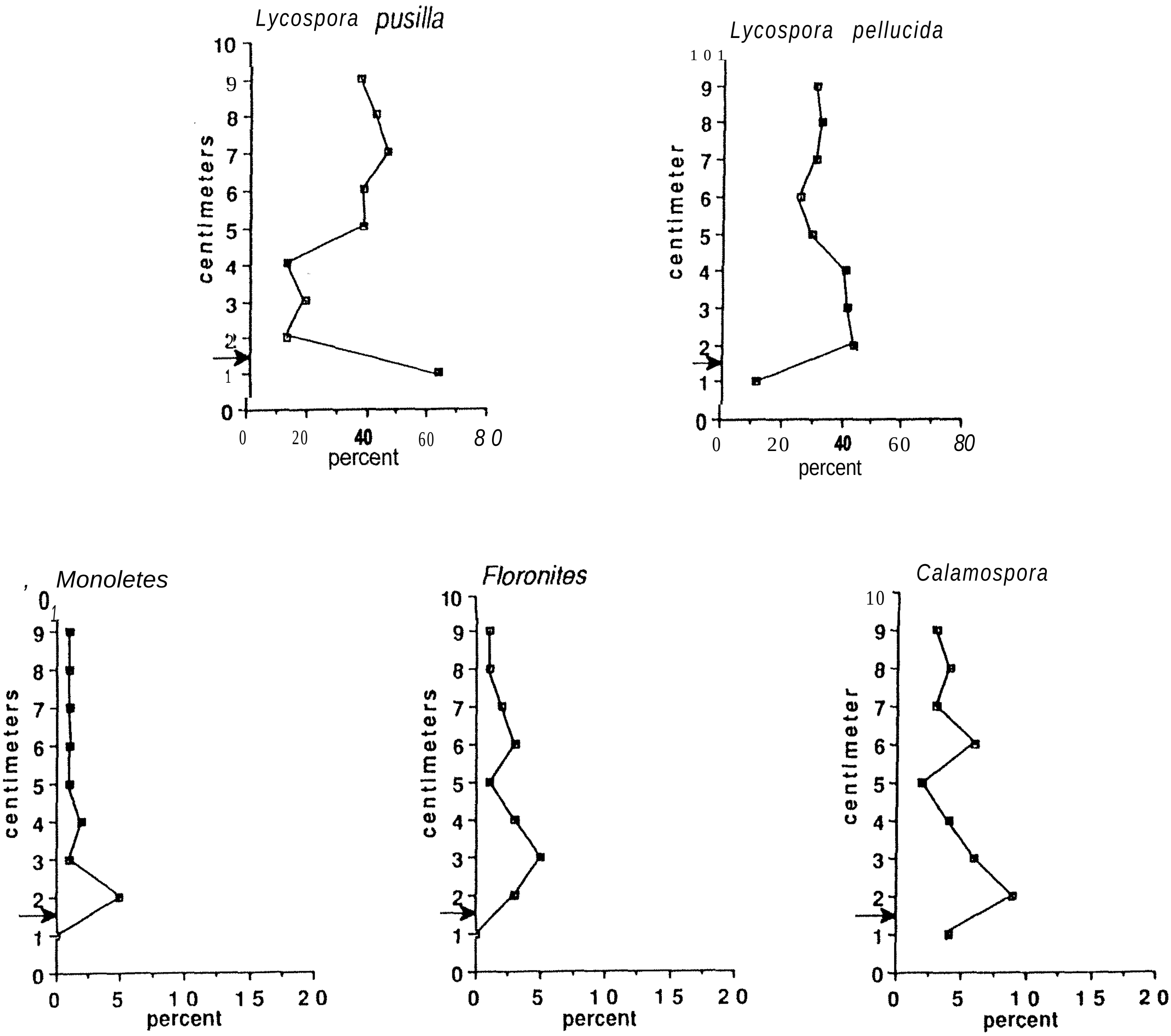

Figure 3.--Relative abundances of several palynomorph taxa. Arrows indicate stratigraphic position of fusain horizon.

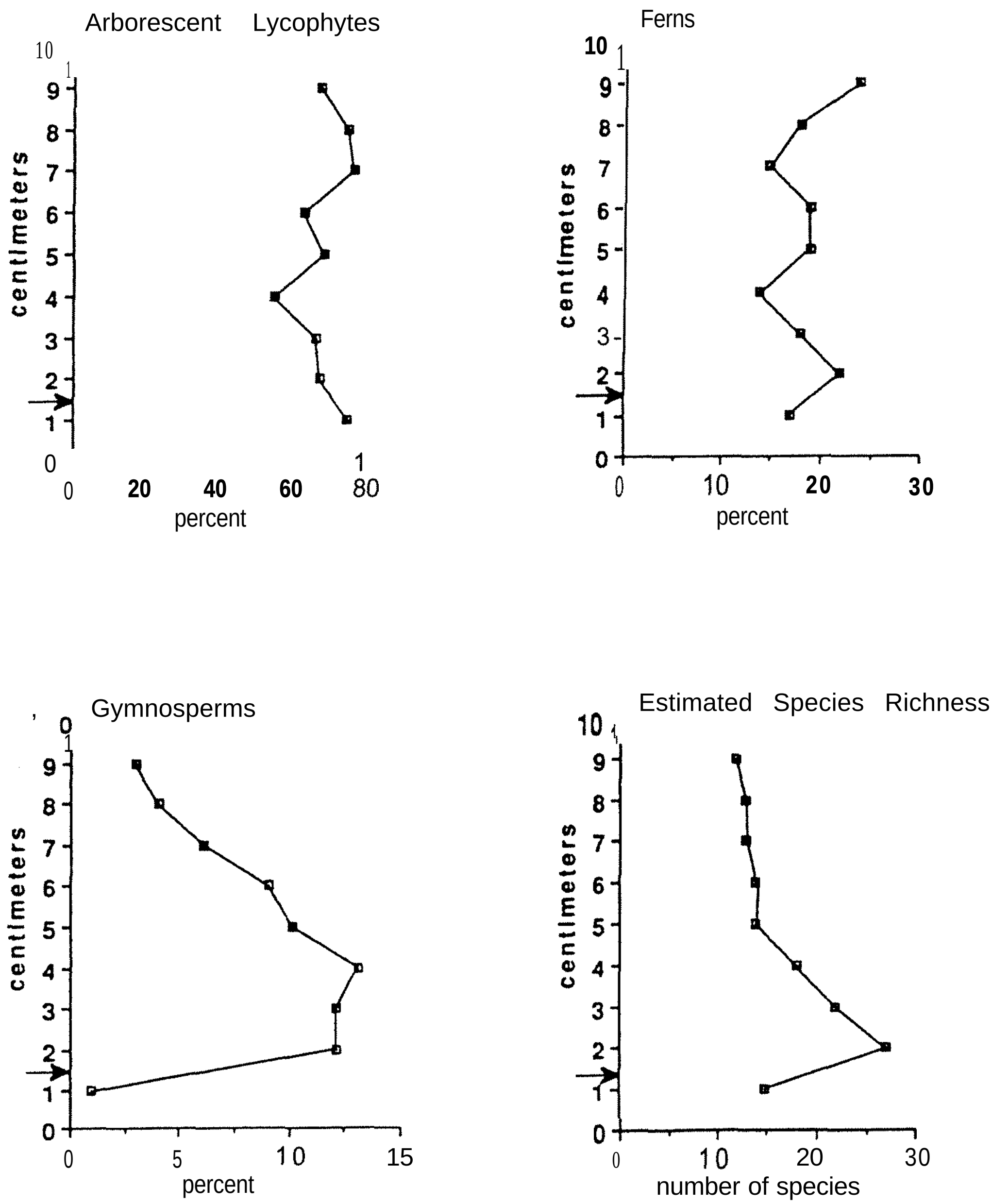

Figure 4.--Relative abundances of major plant groups and estimated species richness. Arrows indicate stratigraphic position of fusain horizon.

However, such conversions are only now being developed for the Carboniferous (Willard 1986). Future interpretation of these communities will incorporate this more quantitative approach to stand composition; however, that is beyond the scope of this preliminary study. Second, changes in pollen and spore abundances do not necessarily mean a change in stand composition. Some modern plants are stimulated to reproduce by disturbance, thus enriching their representation in the dispersed pollen and spore record. At this stage of the study, this influence cannot be discounted, but at the scale of stratigraphic resolution applied in this paper, pollen and spores are regarded as good indicators of standing vegetation (Davis and Goodlett 1960). Clearly, one must tailor questions asked of the pollen record to the time-stratigraphic, taxonomic, and quantitative resolution appropriate to the data.

RESULTS

The spore species Lycospora pusilla, produced by some arborescent lycophytes, was most abundant (63 percent) in the prefire vegetation (fig. 3). The other arborcsccnt lycophyte spore present (Lvcosoora pcllucida) represented only 10 percent of the palynoflora. Lycospora is produced by a number of species in the stem genus Lepidophloios. Other palynomorphs including Calamosuora, the spore of the sphenopsid Calamites, and a complex of ferns made up approximately 10 and 15 percent of the palynoflora respectively. The pollen of gymnosperm taxa including Monoletes, the pollen of mcdullosan pteridospcrms, and Florinites, pollen of the conifer relative Cordaitcs, were not present in the prefire palynoflora. A simple count of palynomorph taxa suggests that about 15 species were present. This estimate of species richness (fig. 4) is tentative and pending more complete taxonomic study of the palynoflora. Note also that, in some cases, a single spore and pollen taxon was produced by more than one mcgafossil taxon, thus under-representing species richness.

The sample collected stratigraphically immediately above the fire horizon shows several important changes in the palynoflora. Abundance of Lvcospora pusilla declined dramatically from 63 to 13 percent. Conversely, L. oellucida increases from 1 l to 43 percent. The increased relative abundance of L. pellucida is not entirely a relic of decline in abundance of L. pusilla; L. pellucida becomes absolutely more common in an average microscope field on slides of comparable palynomorph density. Fern spores and Calamospora become more common above the charcoal horizon. Medullosan and cordaitean gymnosperms also appear to enter the community following tire. The medullosan seed fern pollen, Monoletcs, reaches its greatest abundance (6 percent) immediately above the tire horizon. Estimated species richness also reaches maximum (approximately 27 species) in the sample immediately above the tire horizon. This maximum includes several added rare or uncommon fern and hcrbaceous lycophytc taxa.

In stratigraphically successive samples above the fire horizon, several trends emerge. First, L. pusilla increases in relative abundance while L. pellucida declines until these taxa returned to approximately prefire levels. However, L. pusilla did not reclaim its prefire abundance within the sample interval. Likewise, L. oellucida declined in abundance throughout the sampling interval, but retained a higher postfire abundance (25 to 30 percent) throughout the sampling interval. In contrast with this gradual dominance exchange, Monolctes declined rapidly to a background abundance of 2 or 3 percent immediately after its abundance peak. Florinitcs, the pollen of cordaitean gymnospcrms, increased in abundance more gradually to a maximum abundance in the second centimeter above the fossil charcoal horizon. Following this maximum, Florinites declined gradually to a background level similar to that of mcdullosan pollcn. Spccics richness followed a similar trend of steady decline to the prefire level in the fourth centimeter above the fire horizon. This decline is driven mainly by the loss of the rare and uncommon fern and herbaceous lycophyte taxa that appear immediately above the fire horizon.

DISCUSSION

Rapid and relatively continuous (perhaps seasonal) sedimentation is a prerequisite for ecological-scale resolution of the fossil record. The Joggins Section appears to provide such conditions; however, absolute sedimentation rates are difficult or impossible to ascertain. Consequently, one must look to the vegetation for confirmation of time scale. If the Joggins strata can be interpreted at the ecological scale of resolution, one should be able to detect vegetation change stratigraphically above sedimentologically-inferred disturbances such as tires. Similarly, ecological resolution should show gradual reestablishment of plant abundances similar to those seen before the disturbance. Clearly, these predictions are realized in this case. Figure 4 shows that arborescent lycophytes suffer a 20-percent decline in abundance following fire. Note that arborescent lycophytes with small, wind-dispersed microspores are likely to be over-represented in the postfire palynoflora due to spores transported from nearby, unburned stands into the gap. The lycophyte decrease in postfire samples is largely counterbalanced by an increase in gymnosperm+initially by medullosan pteridosperms and subsequently by cordaiteans. Medullosans were characterized by large fronds displayed on an unbranched axis and minimal investment in low-density wood. Both of these features (minimal branching and low-density wood) are characteristic of modem colonizer trees in the tropics (Ashton 1978, Bazzzz and Pickett 1980, White 1983). Cordaiteans have a more substantial structural investment in greater volumes of dense wood. These structural observations support the conclusion that medullosans, with their "inexpensive" construction, filled the colonizer role in these ancient communities, with the more structurally "expensive" cordaiteans following in a mid-successional phase. Thus, the observed vegetation response, particularly the increase in gymnospcrms, after fire establishes that ecological-scale changes are being observed, return to prefire abundances in stratigraphically subsequent samples reinforces the conclusion. These biological observations coupled with the inferred rapid sedimentation rate support an ecological-scale interpretation of the Joggins section. Thus, while I cannot conclude whether the sampled interval records decades or centuries in absolute time, the observed vegetation changes probably occurred within several generations of the plants involved--clearly an ecological-scale time frame.

If one accepts Scott's (1989) conclusion that fusain in coals and clastic sediments records wildfire, then it is clear from the abundance of fusain in the Joggins sediments that fire was not only present but common in the Joggins swamp ccosystcm. The influence of periodic tire on vegetation thcn becomes intcresting. In a study of early Middle Carboniferous peat-forming mires, Phillips and others (1985) reported a correlation between an increased abundance of cordaitean gymnospcrms and higher fusain percentages. In the upper "Pottsville" (Westphalian A/B correlativcs) coals of northeastern Tcnncssce and eastern Kentucky, Cordaites makes up 33 and 36.7 percent biovolume in coal balls with 9 and 7.4 percent fusain respectively. In the samc region, two correlative coals with lower cordaitean abundances (1.7 and 14.3 percent) contain lower proportions of fusain (0.8 and 2.6 percent respectively). A similar pattern was observed in the Upper Foot Scam, Lancashire, England, the Bouxharmont Scam in Belgium, and the Katharine Seam in the Ruhr, West Germany. Phillips and others (1985) conclude that this correlation reflects the cordaites' preference for drier habitats, which would be more prone to fire or other diagcnctic oxidation.

The association of cordaitean gymnospcrms with fusain is also observed in the Joggins fire horizon. In this case, howcvcr, detailcd stratigraphic sampling shows that the cordaitcs were not simply growing in drier, fire-prone areas; rather, they were present in the community only after fire and could not maintain a significant presence without further disturbance This suggests that fire (or disturbance in general) was an important factor in controlling the distribution of this taxon in the floodplain community.

A clear fusainlabundance relationship for ptcridosperms is not present in the data of Phillips and others (1985) from localities in the Euramerican Carbonifcrous. Medullosan pollen, Monotetcs, is commonly not reportcd in standard palynological analysis because the large grains (100 to 200 micrometers) are eliminated from preparations by standard sieve techniques. Also, if the distribution of medullosan pteridosperms is patchy and ephemeral as hypothesized in this paper, its pollen might be easily missed in a grab-sampling regime. Consequently, absence of Monoletes from these reports (Phillips and others 1985) is not convincing evidence that pteridosperms were absent at those localities. At Joggins, however, the pollen of mcdullosan pteridospcrms, like that of Cordaites, enters the community and has an abundance peak immediately above the fire horizon. Again, these data suggest that the distribution of Monoletcs-producing pteridosperms was influenced by fire. Arens (manuscript in preparation) notes that some of the medullosan pteridospcrms show morphologic and distributional characteristics consistent with their interpretation as colonizers of disturbed habitats, while others appear to have been understory plants. While colonizer and understory medutlosans cannot be distinguished by their pollen, the restriction of medullosans to immediately above the charcoal horizon at Joggins supports the conclusion that some medullosans functioned as colonizing plants that required disturbance (in this case, fire) and were unable to maintain their presence (or at least to reproduce) on a site without subsequent disturbance. This conclusion can easily bc reconciled with Phillips and others (1985); mcdullosan pteridosperms were an ecologically and taxonomically diverse group, and summary data that include taxa from both colonizer and understory medullosan guilds would obscure a correlation between medullosan ccotypes and fusain.

Pollen analysis across the firc horizon clearly shows that there is a vegetative response to disturbance. Two important trends emerge. First, the lycophytes--dominant taxa before the fire--suffer a major decline after fire, but gradually recover prefire dominance. Second, the lycophytcs were replaced in the postfire community largely by gymnospcrms, primarily ptcridospcrms and cordaiteans. However, these taxa apparently could not replace themselves on the site and eventually became locally extinct (Noble and Slatyer, 1980). Thcrcfore, tycophytcs and gymnosperms may also be tilling different ecological guilds in the floodplain communities. The gymnosperms may represent key components of an early successional community that colonized habitat opened by fire; the lycophytes, then, constitute a later successional community that established in the shade of the colonizers and cventually succeeded to dominance in the area. Noble and Slatyer's (1980) modet predicts that if tire occurs again during the gymnosperm stage of succession, mcdullosans and cordaiteans will continue to exist on the site. However, in the absence of fire (or some other disturbance), the gymnospcrms will become locally extinct and lycophytcs will reassert community dominance. This latter prediction was confirmed in this sequence. The ephemeral distribution of early successional gymnosperms, particularly pteridospcrms, is supported by observations in many Pennsylvanian-age lowland environments (Phillips 1981).

Species richness trends associated with disturbed ecosystems are equivocal. In the tropical rain forests of Uganda (Eggcling 1947) and old fields of Nigeria (Jones 1956) species richness is low during the colonization stage, peaks in mid-succession, and declines in late succession. This is inconsistent with the diversity peak observed immediately after disturbance in the Joggins sequence. One possible interpretation is that the initial, low-diversity phase was too brief to be resolved or was not recorded in this sequence. However, in the Yellowstone National Park forest ecosystem (Dale Taylor, 1990, personal communication), the Australian grassland (Burrows, in preparation), and chaparral (Zedler 1977) species richness is greatest immediately following fire, as in the Joggins floodplain community. In these cases, species richness in enhanced by the presence of several codominant species and a variety of less common species, much as in the Joggins sequence. This pattern is promoted by microenvironmental variation generated by heterogeneity in the distribution of plant resources such as light, moisture, and nutrients in the disturbed site (Bazzaz and Sipe, 1987). Analyses of more Joggins fire horizons will undoubtedly clarify the pattern of species richness following disturbance on the ancient floodplain, and permit more definitive interpretation.

FUTURE WORK AT JOGGINS

The rescarch described here represents the first stage of a larger study of the Joggins plant communities. Conclusions prescnted arc, therefore, more accurately described as hypotheses to be tested. The Joggins record presents an opportunity to use fine-scale palynological analyses Lo interpret the effects of disturbance on ancient plant communities. With analysis of more stratigraphic sequences spanning charcoal horizons, one may ask: (1) Is there a generalizable trajectory of vegetation replacement following disturbance in the Middle Pennsylvanian moist floodplain forests? (2) Does a similar pattern occur in contemporaneous peat-forming ecosystems? (3) Does this observed pattern change with sedimentologically-inferred differences in soil moisture? (4) What trends in diversity are observed within and between successional guilds in clastic floodplains and in peat-forming mires? Such an approach, emphasizing dynamic patterns within communities over time scales of several plant generation, offers the opportunity to test ideas of community stability and coherence suggested for the Carboniferous (DiMichele and others 1985, Phillips and others 1985). It also offers a new and different system to test similar ideas derived from and argued about by ecologists studying modem plant communities.

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F'IREHISTORYANDF'IREECOLOGY IN THE COSTA RICAN PARAMOS

Sally P. Horn'

Abstract-The high peaks of the Cordillcra de Talamanca in southern Costa Rica extend above timberline and support bamboo- and shrub-dominated páramo vegetation. The charcoal stratigraphy of sediment cores from glacial lakes reveals that fires set by people or lightning have occurred in the highlands for thousands of years. Historical sources and field evidence document numerous páramo fires since the mid-century. During the past 40 years, fire recurrence intervals at specific sites have ranged from 6 to about 30 years.

Patterns of postfire vegetation development support the initial floristics model of succession. The dominant bamboo (Chusquea [=Swallenochloa] subtessellata) resprouts vigorously following burning, as do associated ericaceous shrubs. Hypericum spp. suffer high mortality and recolonize by seed. Slow rates of growth and colonization by both woody and herbaceous species result in the persistence of bare patches of ground for a decade or more following burning.

INTRODUCTION

In a recent review of tropical alpine plant ecology, Smith and Young (1987) noted that fire is common on most tropical mountaintops. But the authors found few references on fire Frequencies or fire history in tropical highlands, or on the impact of burning on vegetation, soil nutrients, and hydrology.

Since 1984 I have been working to fill this gap for the high mountains of southern Costa Rica. This paper summarizes the results of *my* research -- and that of U.S. and Costa Rican colleagues -- on firc history and fire ecology in the bamboo- and shrub-dominated priramos found above timberline in the Cordillcra de Talamanca.

ENVIRONMENT AND VEGETATION

The uplifted granitic batholith that forms the backbone of the Cordillcra dc Talamanca is mantled by Tertiary volcanic and scdimcntary rocks, which outcrop along with granodioritcs and othcr intrusive rocks on the high peaks (Wcyl 1957). About a dozen areas along the crest of the range reach above timberline, and support small to extensive stands of ncotropical páramo vegetation (fig. 1). Many of these areas arc quite rcmotc and remain poorly known botanically.

Ecological research has focused on the priramos surrounding Ccrro Chinip6 (3819 m), the highest peak in Costa Rica, and Ccrro Bucnavista (3491 m). Glaciers occupied the upper valleys of the Chinip6 massif several times during the Plcistoccne, leaving behind a picturesque ice-carved landscape dotted by some thirty glacial lakes. The extensive (> 5000 ha) Chirripó páramo is protected within Chirripó National Park, which was cstablishcd in 1975. Access to this remote area is provided by rough trails that lead out of settlements on the lower foothills of the Cordillera de Talamanca.

The smaller (c. 1000 ha), unglaciated Buenavista páramo straddles the crest of the Cordillera de Talamanca along the Inter-American Highway route. There are no settlements within the páramo, but Cerro Buenavista is festooned with broadcasting towers, and jeep trails and electrical transmission line corridors crisscross the páramo. This area was the main routc across the Cordillera de Talamanca even before the construction of the Inter-American highway in the 1940s, and the vegetation has long been affected by tree cutting, human-set fires, and grazing (Horn, 1989a). Janzen (1973a, 1983) believes that low forest, rather than páramo, covered the Bucnavista peaks prior to extensive human disturbance.

In both physiognomy and floristics, the Taiamancan páramos resemble the more extensive priramos of the northern Andes (Wcber 1959). and most authorities consider them to mark the northern limit of páramo vegetation in the neotropics (Cuatrecasas 1979; Lauer 1981). The dwarf bamboo Chusquea (=Swallenochloa) subtessellata (Janzen 1983; Clark 1989; Horn 1989b) is a characteristic element within the páramos, forming monospecific stands in many areas (fig. 2). Woody dicots that grow intermixed with the bamboo include species in the Ericaceae, Hypericaceac, and Compositac families. A variety of herbaceous plants, many of Andean affinity, occur beneath the shrub canopy and in more open areas.

The oak Quercus costaricensis dominates the montane forests that arc found just below timberline in the Buenavista and Chirripó highlands. The upper limit of oak forest ranges in elevation from about 3150 m to 3300 m. In some areas, oak forest 20-30 m high gives way to l-3 m high bamboo- and shrub-dominated páramo along an abrupt boundary; elsewhere

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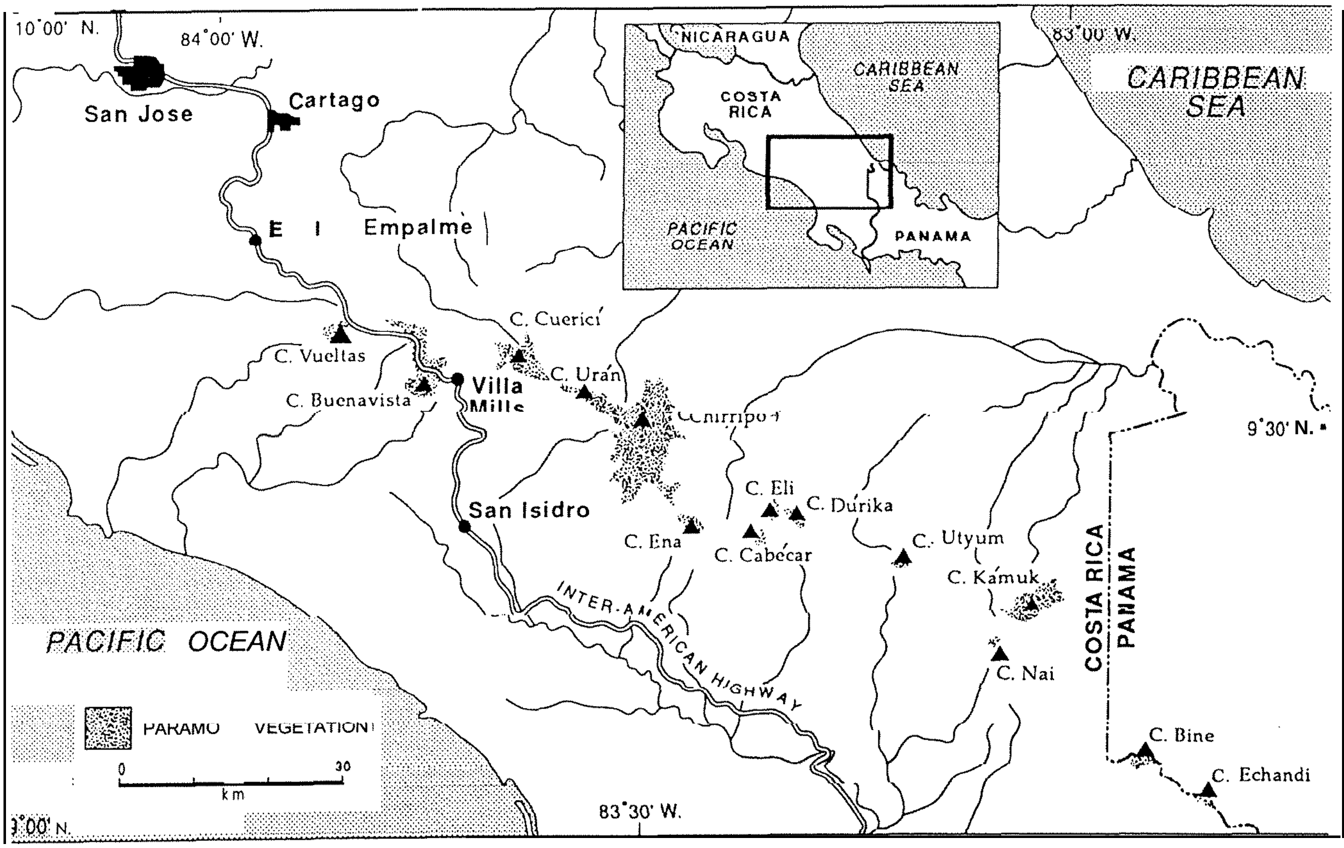

Figure 1. The páramos of the Cordillera de Talamanca, Costa Rica. The extent of páramo vegetation is based on Gómez (1986) and the 1:50,000 scale topographic maps published by the Instituto Geográfico Nncional.

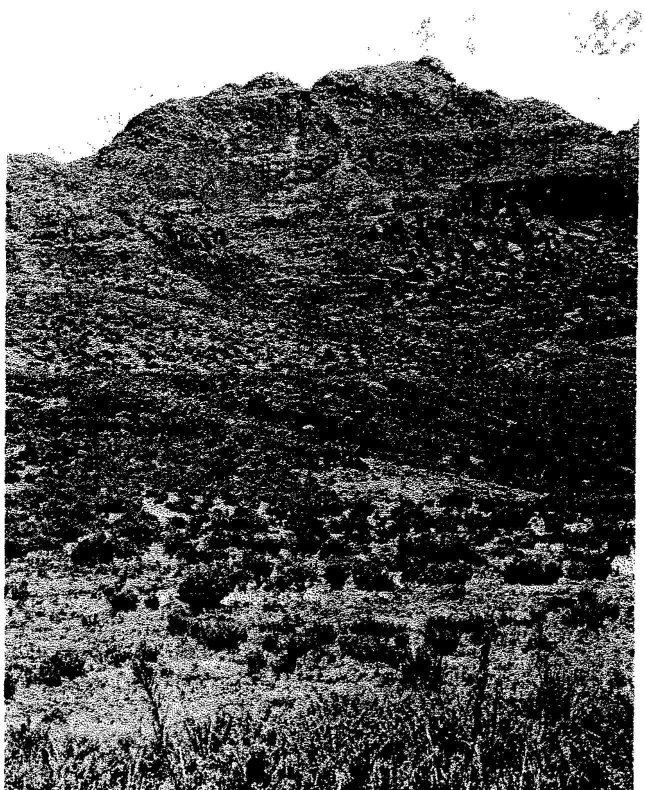

Figure 2. Nearly monospecilic stand of the bamboo *Chusquea subtesseltata* in the Valle de los Concjos in the Chirripó páramo.

the two communitics are separated by broad transitional zones supporting large shrubs and small trees other than oak. Some of the trees and shrubs within these transitional zones also occur, in mote stunted form, within the páramos of the high peaks.

Low annual temperatures and a highly seasonal precipitation regime characterize the climate of the Costa Rican páramos. Meteorological data from the Cerro Páramo station (3475 m) near Cerro Buenavista show a mean annual temperature during the period 1971-79 of 7.6" C (Instituto Costarricense de Elcctricidad, unpub. data). The warmest months, April and May, averaged only 1.4" C warmer than the coldest month, January.

Average annual precipitation during the period 1971-1984 at Ccrro Páramo was just over 2500 mm. Typically only about 10% of the total precipitation falls during the dry season or "vcrano" that lasts from mid-Deccmbcr to late April. Frost is frequent in the páramos, and hail has been observed (Dohrcnwcnd 1972), but there ate no reliable reports of snowfall (Coen 1983).

Clouds and mist frequently bathe the páramos, and must contribute an appreciable (though as yet unmeasured) amount of moisture. High atmospheric humidity moderates the dry season, but for days or weeks during this period the condensation belt lies below timberline, resulting in clear, dry wcalhcr on the high peaks. Somc perennial herbs die back at this time, and vegetation and litter quickly dry out, providing the fuel for fircs.

Although burning is most widcsprcad during the driest first 4 months of thc year, a secondary rainfall minimum (the "vcranillo") during mid-year may allow some burning. After a succession of several rainless days in July, 1981, cxpcrimcntal fircs started in the Bucnavista páramo ignited rcadily and burncd quickly to their fucl hrcaks (G.B. Williamson, pers. comm. 1982).

RECENT FIRE HISTORY

All of the Costa Rican páramos are protected within national parks or reserves, hut there arc no written fire records for these manngcmcnt areas. Documentary evidence of burning consists of occasional rcfcrcnccs in the scientific and popular litcralurc, and old photographs that reveal evidence of burning. Fires are often too small to bc readily apparent on satellite imagery, and aerial photograph coverage of the páramos is limited.

However, the slow rate of organic matter decomposition in thc continuously cool páramo environment (Janzcn 1973b) results in the long pcrsistcnce of ficld evidence of past fircs. Fire-killed shrub stems are conspicuous throughout the páramos, cven at sitcs that burned more than two decades carlicr. Some shrubs preserve evidence of multiple fires (fig. 3), and annual growth rings in living and fire-killed stems provide information on fire recurrence intervals (Horn 1986; Williamson and others 1986).

Ficld evidence indicates that all of the larger páramos cast of Cerro Utyum (Ddriia, Chirrip6, Urán, Cucricí, Buenavista, Vucltas; fig. 1) have burned since the mid-century (Weston 1981). Human carelessness, arson, helicopter and plane crashes, and escaped agricultural fires are among the ignition sourccs for recent burns (Horn 1986). Although human activity can explain all recent fires, lightning could also have played a role. Costa Rica has one of the highest incidcnccs of thunderstorms in the world (World Meteorological Organization 1953, 1956), and lightning has been observed striking both the forested slopes and treeless summit of Ccrro Chirripó (Horn 1989c). Such strikes might occasionally ignite fircs on the high Talamancan peaks, as they do on Mountain Pine Ridge, Belize (Kcllman 1975).

Many areas within the Chirripó and Bucnavista páramos have burned two or three times since 1950 (Horn 1986, 1989b, 1989c, 1990a). Fire recurrence intervals at specific sites have ranged from 6 to about 30 years. Given the slow rate of vegetation recovery within the páramos (see below), 6 years is probably close to the minimum fire recurrence interval possible. At least this many years of postfire growth is likely required to generate enough fuel to carry a second fire.

Figure 3. Resprouting shrub of Vaccinium consanguineum on Cerro Zacatales in the Bucnavista páramo showing stems killed in two successive fires. The larger, central stem was killed in the penultimate fire at the site, after which the shrub resprouted, producing the smaller dead stems that were killed by the last fire at the site. Following the last fire the shrub again rcsprouted. Older (twice-burned) stems can be differentiated based on position, degree of charring, presence or absence of bark, and extent of decay, and annual growth rings in dead and living stems can be counted to estimate fire recurrence intervals. The visible length of the tape measure is 60 em.

Recent fires in the Chirripd páramo have been much larger than fires in the Bucnavista páramo. Páramo fires along the highway route have tended to bum out relatively quickly because they encountered insufficient fuel, fuel that was too moist, or a fire break created by a road or electrical transmission line corridor. The larger Chirripó páramo has provided an extensive and continuous fuel bed not interrupted by roads or other fire breaks, and fires, once started, have tended to spread over hundreds or thousands of hectares.

The post-1950 fire record in the Buenavista and Chirripó highlands suggests a link between fire and drought. Figure 4 graphs monthly precipitation in the driest month and the two consecutive driest months during the period 1952-1985 at Ccrro Páramo (1971 onwards) and at the nearby Villa Mills station (3000 m) located just below timberline. I assume that these data reflect trends that would also have bccn evident at Cerro Chirripó, located 30 km to the east. Thc triangles denote known firc years in the Bucnavista and Chirripci highlands, and in the intervening Cuericí páramo (fig. 1), with the size of thc trianglcs indicating the total area above 3000 m clcvation estimated to have burned in that year.

Not surprisingly, the largest high-elevation fires have tcnded to occur during the driest years. Betwccn 1957, and 1985 therc were 3 years in which the driest month (February or March) recordcd less than 0.5 mm rainfall, and in each of thosc years a large (> 100 ha) fire occurred in the Chirripci páramo and surrounding montane forests. I f rainfall rccords for 1952-1985 are indicative of long-term trends, thc data suggcst that cxtremcly dry years conducive to widcspread burning may bc expcctcd to occur about oncc a decade in the Chinipci and Buenavista páramos.

LONG-TERM FIRE HISTORY

Charcoal fragmcnts in sediment cores from glacial lakes in the Chirripd highlands provide evidence of ancient fires in thc Costa Rican páramos. A short (I IO cm) scdimcnt corc rccovcrcd from Lago Chirripd (3520 m) in the Valle de los Lagos in 198.5 preserved two distinct layers of macroscopic charcoal and an ahundnncc of microscopic charcoal fragmcnts (Horn 1989c). The charcoal particlcs appcar to have bccn dcrivcd primarily from fircs within the watershed of the lake and in adjaccnt areas of thc Chirripci páramo. Charcoal concentrations varied with depth in the sediments, suggcsting tcmporal variations in fire frequency (fig. 5). Although absolute firc frequencies cannot be dctcrmincd from the charcoal data, variations in charcoal abundance may provide indications of rclativc fire frcqucncies. Howcvcr. the rclationship bctwccn fire history and scdimcntary charcoal concentrations is complicatcd (Clark 1983), and factors unrclatcd to burning also may have affcctcd thc charcoal curvc (Horn, 1989c).

Charcoal fragments were prcscnt in all of the samples from thc Lago Chirripci scdimcnt core, indicating that fircs have &ii thc lake basin and surrounding areas since the scdimcnts in the corc bcgan accumulating Some 4000 years ago. A longer core raised from a glacial lake in an adjaccnt valley in 1989 (Horn 199017, and in prep.) spans the last 10,000 ycars, and also contains charcoal, confirming and cxtcnding thc short core rccord. In the Chirripó páramo firc is clearly not a disturbance factor introduccd by modern human society; burning duc to human action or lighting has occurrcd for thousands of ycars.

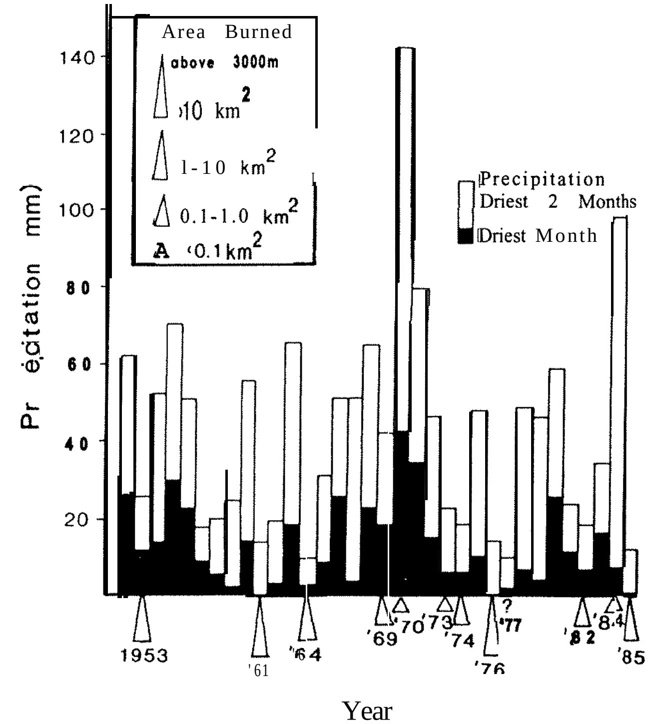

Figure 4. Dry season precipitation in the Buenavista highlands and the distribution of recent high clcvation fires. Rainfall data arc from the Cerro Páramo (1971 onwards) and Villa Mills stations (Instituto Costarriccnsc de Elcctricidad unpub. data). The qucstion mark in 1977 reflects uncertainty as to whcthcr a forest fire that burned near Ccrro Chirripd that year cxtcndcd above 3000 m elevation.

THE IMPACT OF BURNING

Effects of Fire on Soils and Hydrology

Soils within the Costa Rican páramos arc usually well-drained, rich in organic mattcr, and acidic, with pH values as low as 4.0 (Harris 1971). Soil samples collected by Leftwich (1973) within a recent burn area in the Buenavista páramo showed lower organic matter and C/N ratios, and higher cxchangcablc calcium and magnesium, than did samples from an ndjaccnt unburned arca. Field cvidcncc of erosion has been notcd on some bums, but soil loss following burning has not bccn quantilicd. No information is available on the hydrological impact of burning, though such studies would be of intcrcst given the important watershed function of the páramos and surrounding montane forests.

Effects of Fire on Páramo Vegetation

Postfire vegetation dynamics in the Buenavista and Chirripd páramos have been examined by Janzen (1973b, 1983), Chavcrri and othcrs (1976, 1977, and in prep.), Williamson and othcrs (1986) and Horn (1986, 1989b, 1990a). The slow rate of decomposition in thc Costa Rican páramos has been an asset in thcsc studies, as pcrsistent fire killed stems can be idcntificd and mcasured to provide information on thc species composition and stature of the preburn woody vcgetation.

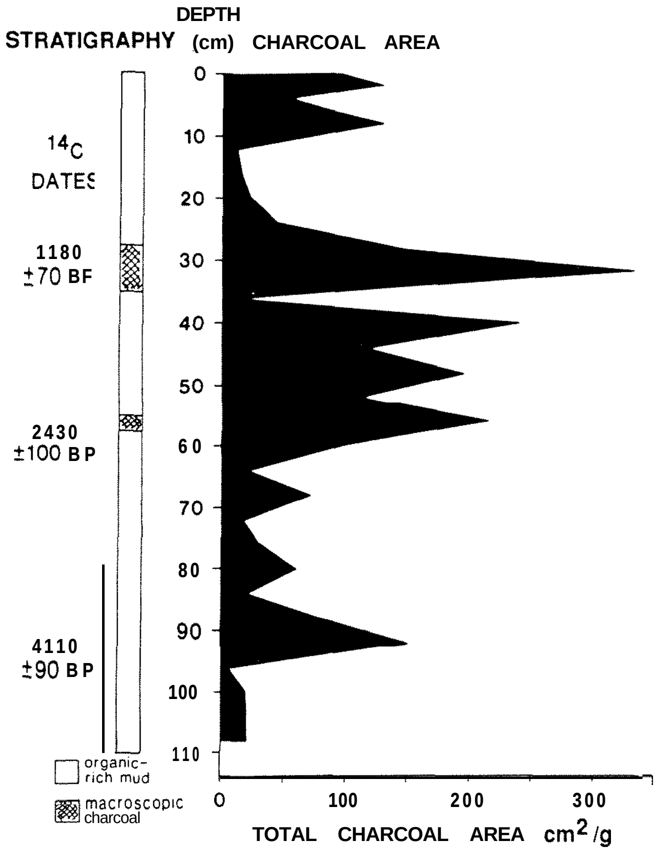

Figure 5. Diagram showing charcoal and sediment stratigraphy of a 110 cm core from Lago Chirripó.

Patterns of postfire vegetation recovery in the Costa Rican páramos support the initial floristics model of succession. The first species to colonize burned ground are the shrubs and herbs that comprise the mature vegetation. As appears to be the case in other tropical montane ecosystems (Smith and Young 1987), the Costa Rican páramos harbor no early successional specialists, and no invasion of plant species from outside communities takes place following disturbance.

Regeneration after and between tires is largely vegetative. Following fire, most woody plants and many herbs resprout vigorously, sometimes within just a few weeks of burning (Chaverri and others 1977). The bamboo Chusquea subtessellata and the ericaceous shrubs Vaccinium consanguineum and Pemettia coriacea show resprout rates of 90-100 percent following crown consumption and the death of aboveground stems (table 1). These vigorous sprouters rarely, if ever, establish seedlings, but may produce sprouts at new locations along widely diverging root and rhizome systems.

The common páramo shrub Hypericum irazuense supplements vegetative reproduction with seedling establishment. The species exhibited very low (4-14%) rates of basal resprouting at my study sites (table 1), but Janzen (1973b) noted abundant suckering by H. irazuense (his H. caracasanum) following a fire on Cerro Asunción in the Buenavista páramo. Fire intensity and antecedent soil moisture conditions may be important controls of resprouting ability in this shrub.

For the narrow leaf congener Hypericum strictum all tires may be lethal; the species showed complete mortality at the Tower 65 site (table 1) and may be an obligate seeder.

Postfire seedling recruitment by Hvoericum spp. and other shrubs and herbs relies largely on the influx of seeds from surrounding, unburned areas. The apparent lack of soil seed reserves (Horn 1989b) results in a very slow rate of seedling

Table 1. Percent basal resprouting by bamboo and shrubs following crown loss at paremo burn sites".

	BURN SITE			
	Tower 65	Conejos	Zacatales	Sábi La
Chusquea subtessellata	100	96		100
Vaccinium consansuineum	98	90	96	98
Pernettia coriacea	93		96	95
Escallonia poasana				57
Papanteat i e r i	25			15
Hyperi cum i razuense	4	6	12	14
Senecio firmipes	4			
Hypericum strictum	0			

[a]Only data for species with sample sizes greater than twenty at individual burn sites are included. For sample sizes and details on study sites and methods, see Horn (1989b).

Table 2. Postfire height and percent height recovery of resprouting shrubs and bamboo at paramo burn sites".

	SITE AND YEARS SINCE LAST FIRE			
	Touer 65 1 yr	Cone jos 9 yr	Zacatales 12 yr	Sábi la ≥ 12 yr
Chusquea subtessettata	34 cm (18%)	103 cm (98%)	141 cm (113%)	191 cm (129%)
Vaccinium consanguineum	22 cm (26%)	76 cm (71%)	67 cm (87%)	103 cm (88%)
Pernettia coriacea	16 cm (24%)		46 cm (97%)	65 cm (79%)
Hyper icum irazuense	20 cm (15%)	91 cm (64%)	65 cm (71%)	111 cm (70%)
Escallonia poasana	14 cm (17%)		96 cm (116%)	148 cm (171%)
Rapanea pittieri	19 cm (14%)		117 cm (78%)	172 cm (126%)

"Only the six most comnon woody species are Listed. See Horn (1989b) for samples sizes and standard deviations, and details on sites and field methods. Percentage height recovery equals mean maximum postfire plant height divided by the mean maximum height of unbroken, prefire (burned) stems. The prefire stature of plants partly reflects the time intervats between the Last and penultimate fires at the sites, which was ≥16 years at the Tower 65 site, 15 years at the Conejos site, ≥ 12 years at the Zacatates site, and ≥29 years at the Sábila site.

colonization and may delay significant seedling recruitment for several years following tire. Where potential seed sources are distant (as near the centers of large burns), recolonization by Hypericum irazuense and other fire-sensitive species may not occur until the rare individuals that resprouted following burning grow to maturity and begin seed production within the burn area.

Growth rates of seedlings and suckers are extremely slow (tables 2,3). The fastest growing woody species, the bamboo Chusquea subtessellata, requires about 8-10 years to regain its average prefire stature (Janzen 1983; table 2, this paper). Coupled with the slow pace of seedling colonization, the slow growth rates within the Costa Rican páramos result in the long persistence of gaps created by burning. Bare patches of ground from 0.1 m^2 to 0.5 m^2 or larger in size may persist for a decade or more following fires, particularly on large bums where seed influx is low.

The strong rcsprouting ability of most of the dominant woody species in the Costa Rican páramos minimizes compositional shifts following burning (Horn 1989b). If a site dominated by bamboo and ericaceous shrubs bums, postfire cover the first year, and for at least a decade afterwards, will be dominated by bamboo and ericaceous shrubs. Major shifts in woody species composition will be observed only at sites with a substantial cover of lire-sensitive shrubs. Herbaceous species may reach higher cover values on such bums, where greater shrub mortality results initially in less competition for space and other resources.

Based on observations at Cerro Zacatales in the Buenavista paramo, Williamson and others (1986) described a "fire cycle" in the Costa Rican páramos in which postburn cover is initially dominated by grasses and sedges, but reverts ultimately to shrub dominance if the site is kept free of fire for a sufficient period (perhaps 20 years). For reasons outlined above, such a cycle will be most evident where shrub mortality is very high, and where flowering plants are present nearby to reseed the bum site. When shrubs are closely spaced and suffer low mortality from fire, grasses and sedges may contribute little to postfire cover. Few graminoids or other herbs were present 3 years after a fire on Cerro Asunción (Janzen 1973b), a site characterized by high shrub survival and vigorous suckering. Greater herbaceous cover was observed 1 and 4 years after a fire at the nearby Tower 65 site, where 40% of the woody perennials (mostly Hypericum irazuense shrubs) died following crown loss. t even at this site, shrubs and bamboo dominated the initial postfire cover (Horn 1989b and unpub. data).

Graminoids, when abundant, will provide fine fuels that can support frequent fires. Williamson and others (1986) have suggested that when such fires occur at 5-10 year intervals

Table 3. Postfire stem diameters and percent diameter recovery for burned shrubs and bamboo at paramo burn sites".

	SITE AND YEARS SINCE LAST FIRE			
	Tower 65 1 yr	Cone jos 9 yr	Zacatales 12 yr	Sábi la ≥ 12 yr
Chusquea subtessettata	0.55 cm	0.73 cm	0.89 cm	1.20 cm
Vaccinium consanguineum	0.29 cm (13%)	1.98 cm (66%)	1.50 cm (70%)	2.13 cm (64%)
Pernettia coriacea	0.26 cm (26%)		0.68 cm (78%)	1.03 cm (63%)
Hypericum irazuense	0.16 cm (13%)	1.19 cm (58%)	0.86 cm (55%)	1.46 cm (60%)
Escallonia poasana			3.07 cm (125%)	3.52 cm (94%)
Rapanea pittieri	0.26 cm (15%)			2.92 cm (103%)

ªOnly the six most common woody species are tisted. Sample sizes and standard deviations are Listed in Horn (1986). Percentage diameter recovery equals mean maximum postfire stem diameter divided by the mean maximum prefire (burned) stem diameter. No data are available on prefire stem diameters of Chusquea subtessellata.

they may impede shrub recovery and facilitate the ultimate development of continuous grassland. Preliminary trials indicated that some shrubs may release allelopathic chemicals that inhibit the growth of grass and sedge seedlings and hence reduce the risk of fire (Williamson and others 1986).

CONCLUSION

Fire plays an important rote in most of the world's shrublands (Christensen 1985), and the páramos of Costa Rica Seem to be no exception. The low annual temperatures, high annual rainfall, and seasonal drought that characterize these tropical alpine habitats provide the fuel for periodic tires. Sedimentary charcoal evidence shows that fires have occurred in the Chirripó páramo for at least 10,000 years; fires due to human activity or lightning may be of similar antiquity in other páramo areas.

Postfire regeneration follows initial floristics and is principally vegetative for the major woody species, many of which resprout vigorously after fire. The ultimate origin of this resprouting ability is uncertain, but selection in the face of periodic fires may have reinforced the trait. Some shrubs rely on seedling recruitment to repopulate burned areas, but the lack of soil seed reserves gives the sprouters a strong advantage in the first few postfire years. Low rates of seed influx and slow growth following germination or sprouting result in exceedingly slow recovery rates; 10 years after burning, most shrubs will not have regained their prefire adult stature and bare patches of ground may still be conspicuous.

ACKNOWLEDGMENTS

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A SURVEY OF ABORIGINAL FIRE PATTERNS IN THE WESTERN DESERT OF AUSTRALIA

N.D. Burrows and P.E.S. Christensen'

Abstract.-Aborigines who occupied vast tracts of the Australian deserts used fire extensively for many purposes. The recent departure of Aborigines from traditional burning practices has coincided with an alarming decline *in native* mammal fauna. It *has* been postulated that the combined effects of a changed fire regime, predation by feral animals, and competition from feral herbivores has contributed to mammal decline in this otherwise pristine environment.

Black-and-white aerial photography, satellite imagery, and field observations have revealed that the size, distribution, frequency, and intensity of fires in a pan of the Western Desert have changed dramatically over the last 36 years. In an area west of Lake Mackay the average fire size has increased from 34 hectares in 1953 to over 32 COO hectares in 1986. The small-grainrd mosaic of burnt patches *of* varying ages that existed during Ahoriginal occupation of the land has been obliterated by large, intense and infrequent lightning-caused wildtires. Some evidence of a relationship between fire mosaic and the richness of flora and fauna was obtained during this survey.

INTRODUCTION

About one third of the Australian continent is classified as desert. In the State of Western Australia, the Great Victoria, Gibson, and Great Sandy Deserts occupy some 1.3 million square kilomelres and collectively are commonly known as the Western Desert (Tonkinson 1978). Rainfall is low (annual average ranges from 1.50 to 250 millimetres), and unpredictable, and long periods of drought are common. Rainfall is mostly from cyclones and thunderstorms and surface water is only abundant for short periods following rain. Summers are hot and winters arc cool. The Western Desert is well vegetated and surprisingly rich in wildlife. Highly flammable hummock grasslands, comprising species of *Triodia* and *Plectrachne* (spinifcx), dominate red sandy soils.

Aborigines first arrived on the Australian continent at least 50,000-60,000 years ago and occupied Ihc deserts of the interior by at least 30,000 years ago (Mulvancy 197.5; Flood 1983). Aboriginal people showed remarkable resilience and resourcefulness lo survive in a vast expanse of scattered food and water resources (Tonkinson 1978). They were highly mobile, were able lo exploit a variety of resources in different areas at different times, and developed a detailed knowledge of the environment. Fossil evidence suggests climatic and cultural continuities lasting at least 10,000 years (Gould 1971), so there has been a long period of Aboriginal influence on the desert biota.

The Western Australian Department of Conservation and Land Management (CALM) manages about Icn million hectares of desert conservation reserves which range in area from two hundred thousand hcclarcs two million hectares.

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The management priority for these reserves is the conservation of native flora and fauna. However, in spite of the apparent pristine nature of these reserves and the lack of direct European impact, a sudden and alarming decline in native mammals has been reported (Bolton and Latz 1978; Burbidgc 198.5; Burbidgc and others 1988; Burbidge and McKenzie 1989). Burbidge and Jenkins (1984) reported that about 33 percent of Western Australian desert mammmals are extinct or endangered. They noted that this decline had occurred over the last 30 to SO years. Burbidgc and McKenzie (1989) have shown that all declines and extinctions have been restricted to native mammals with a mean adult body weight in the range from 35 grams to 5,500 grams (critical weight range). The desert conservation reserves arc relatively pristine and have not been directly modified or disturbed by European activities. Generally, recent extinctions of wildlife have been associated with habitat destruction or modification by humans (Burbidge 1985).

Ngaanyatjarra Aborigines from the Warburton area of Western Australia believe that the "mitika", or burrowing bettong (*Bettongia lesueur*), had "gone to the sky because the country had not been cleaned up" (de Graaff 1976). "Clean up" is a term often used by Aborigines for burning the vegetation (Jones 1980). Kimber (1983) reported that Pinlubi Aborigines believed that perhaps a "big bushfire" caused the disappearance of the golden handicoot (*Isoodon auratus*). Scientists have proposed three main hypotheses to explain the decline and in some cases, extinction of desert mammals. Burbidge and Johnson (1983) proposed that changes in fire regime, predation by feral animals, and competition from feral herbivores as the main factors, acting either independently or in combination, leading lo mammal decline in the arid zone.

Recent changes in the fire regime were a direct consequence of the exodus of Aboriginal people from the desert to settlements in missions, outstations and other communities (Could 1971; de Graaf 1976; Latz and Griffin 1978; Kimber 1983; Saxon 1983). Latz and Griffin (1978) postulated that Aborigines created a stable ecosystem by "burning the country in a mosaic pattern". They claimed that mosaic burning reduced the extent and continuity of heavy fuels, and therefore reduced the occurrence of intense wildfires. They also stated that a second effect of mosaic burning was to create 3 range of "states" in the vegetation, from early postfire plant communities to old mature patches. They were of the opinion that such diversity of states would host a greater variety of plants and animals.

The extensive use of fire by desert Aborigines is well documented by early explorers (Warburton 1875; Carnegie 1898) and more recently by anthropologists and ecologists (Finlayson 1943; Jones 1969; Calaby 1971; Gould 1971; de Graaf 1976; Tonkinson 1978; Kimber 1983). Kimber (1983) observed that the Pintuhi used fire in a skilful and controlled manner for many reasons, such as hunting, signalling, to "clean up" the country, for ceremonies, and for fun. tic provides a general description of when fires were lit and the range in fire sizes. Hc also made some rough calculations of the proportion of country burnt and the approximate age since fire, based on information in the diaries of Davidson, who explored parts of the Tanami Desert in 1900. However, little quantitative information about the fire regime during traditional Aboriginal occupation of vast tracts of desert land is available.

Quantitative data on past and present fire regimes arc of considerable interest lo the Western Australian Department of Conservation and Land Management, which is engaged in a multidisciplinary study aimed at maintaining and improving the conservation status of desert ecosystems. As part of this project, experimental reintroductions of selected species of rare and endangered mammals to the Gibson Desert Nature Reserve will be attempted. Prior to reintroduction, prescribed fires will be used to recreate the kind of fire mosaic which is believed to have existed before the departure of Aborigines. Feral predators such as foxes and cats will also be controlled in the experiment.

The aim of this study is to define, as clearly as possible, the fire regime during the occupation of the Western Desert by Aborigines prior to European contact and to compare this with the present-day fire regime. In doing so, we could test the hypothesis of a recent and dramatically changed fire regime as proposed by Latz and Griffin (1978) and others. A knowledge of past fire regime would also greatly assist with the development of appropriate fire management strategies for desert conservation reserves.

We use Gill's (1981) definition of fire regime, which is the history of fire frequency, fire season (season in which fires burnt), fire intensity, and fire size.

METHODS

The departure of Aborigines from their desert homelands started with first European contact at the end of the nineteenth century. Amadio and Kimber (1988) present a summary of European exploration and contact with Aborigines of the northern portion of the Western Desert and also describe the movements of Aboriginal people away from their homelands and into European settlements. To reconstruct the fire regime, it was desirable to study fire on land from which Aboriginal people living a more-or-less traditional lifestyle, had most recently departed. We learnt that a very remote tract of land to the west of Lake Mackay. in Western Australia was probably the last homeland utilized in a traditional manner by Pintubi people (Richard Kimber pers.comm.). The study site of some 54,000 hectares is on the eastern edge of the Great Sandy and Gibson Deserts and lies between longitudes 128°35'E and 128°50'E and latitudes 22°8'S and 22°18'S (Figure 1). The area is arid, with an average annual rainfall

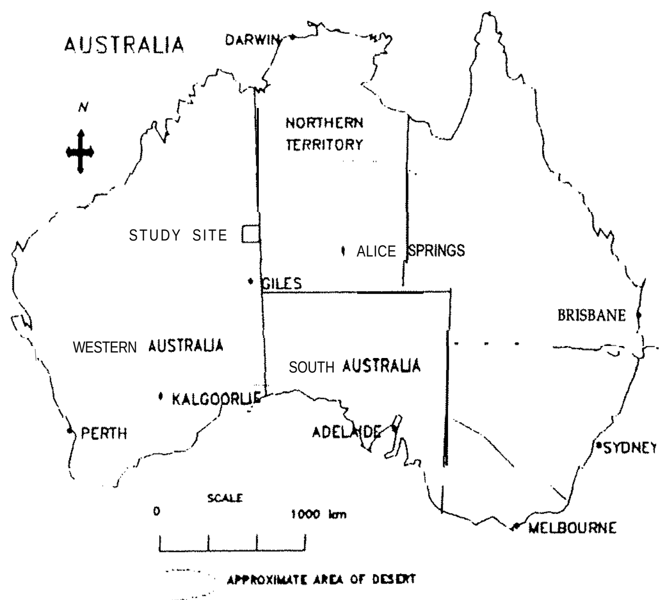


Figure 1: Location of the study area in the Western Desert The approximate boundary of land classified as desert is shown.

probably less than 225 mm (there arc no meteorological stations nearby). Most of the rain falls in summer but both the amount and the seasonal distribution arc highly variable. As a result of the arid climate and sandy soils, the vegetation consists mainly of hummock grasses (spinifex) and associated small shrubs, herbaceous plants and scattered low trees. The sand plain that covers most of the study area is crossed by longitudinal, stable dunes which trend east-west. There arc numerous saltpans, salt lakes, and claypans scattered throughout the area. The largest salt lake is Lake Mackay, which has a total area of about 3,500 square kilometres (Blake 1977). The salt lakes, saltpans, and claypans arc dry except after heavy rain.

The main exodus of Pintubi Aborigines from this area commenced in the early 1960s, with the last people coming into Kintore and Kiwirrikura communities in 1985 (Richard Kimber, pers. comm.).

We have used early black-and-white aerial photographs to help reconstruct the fire regime at the study site during Aboriginal occupation. This technique was supplemented by reviewing the literature and by talking with Pintubi and Pitjantjatjarra Aborigines to obtain information about their USC of fire. The current fire regime, particularly size and distribution of burnt patches, was inferred from Landsat satellite imagery and by visiting the study site and surveying fire history. While in the field WC also surveyed vascular plant and vertebrate animal abundance within areas of different age since the last fire.

Aerial Photographs and Landsat Imagery.

Details of aerial photography available for this area are presented in table 1. The earliest aerial photographic surveys were completed in 1953 by the Royal Australian Air Force as part of the Blue Streak rocket project during an era of cxtcnsivc rocket development and testing in Australia. Fire scars showed clearly on both aerial photographs and Landsat imagery. Fire scars that appeared on the 1953 photography were mapped onto a base map at a scale of 1:50,000.

Additional fire scars which appeared on successive photographs were mapped onto separate sheets, but at the same scale, to form a series of overlays. Landscape features such as salt lakes, claypans, saltpans, sand dunes, and major creeks and streams were also mapped The area and perimeter of each fire scar for each time series was calculated using a computer-linked digitizing board. Initially, an attempt was made to accurately age fire scars baaed on tonal intensities of the scars indicated on the photographs. The different toning of the fire scars represented different stages of revegetation, or fire succession, with recently burnt areas showing up as very bright patches. This proved to be difficult and fire scars could only be rated as "very recent" (up to 5 years old), or "recent" (5-10 years old), based on the extent of re-vegetation. While older scars were barely discernible, no attempt was made to map scars that did not have clearly visible boundaries.

Field Survey of Biological Indicators

The second technique used to obtain details about past and present fire regimes and the fire environment of the study area was ground survey of biological indicators. Some 50 kilomctrcs of line transect was surveyed by vehicle traverse in 1989 and visually discernible fire boundaries recorded. A fire boundary was defined as the boundary between two recognizable fuel ages or between vegetation burnt at different times. Within each fuel age encountered, the time since the last tire was estimated by counting "annual" growth rings from tree stem cross- sections. Because of the uncertainty of the period between growth rings, accurate aging of fire scars was not possible. Based on ring counts from areas of known fire history, stem analysis enabled the time since fire to bc estimated to within ± 1 1 percent of actual time. Measures of vegetation cover and height were also useful for estimating the age of vegetation. In some instances, the aerial photographs were used to estimate time since last fire. Circular plots of 100 metres radius were established in each fuel age and the number and abundance of plant species rccordcd. A list of animals utilizing each fuel age was compiled by searching for burrows, diggings, scats, and tracks. The Pintubi and Pitjantjatjarra guides who accompanied us, skillfully identified signs of animal activity and provided us with the Aboriginal names for the animals.

RESULTS

Aerial Photographs

There has been a dramatic change in the mean and median size of burnt patches in the period from 1953 to 1986. From table 1, it can be seen that thk area recently burnt (up to 10 years prior to photography) has increased from 23.6 percent of the study area in 1953, to almost 60 percent by 1986. Also, the mean size of fires has increased almost 1,000 fold

Table 1. Number, area, and perimeter statistics for burnt patches clearly visible on black-and-white aerial photography and on Landsat satellite imagery of a 53,483 ha study in the Western Desert, Western Australia.

Year	Number of burnt patches	Burnt patch size (ha)				Total burnt (ha)	Total perimeter (km)
		Maximum	Mean	Mode	Median		
1953	372	1,744	34	2	6	12,643	1,198
1973	27	13,534	645	5	197	22,600	412
1977	3	30,618	10,584	•	•	31,752	203
1988	1	32,164	.			32,164	272

over **the** same **period.** Habitat boundary, which is the total fire perimeter within the study site and is a measure of the boundary between vegetation of different ages since fire, has dccrcascd from 1,198 kilomctres in 1953 to 272 kilomctres in 1986. This indicates a substantial reduction in the diversity of fire ages or states of postfire succession within the landscape.

A visual inspection of the fire scars mapped for each time of photography rcvcalcd a number of obvious patterns. In 19.53, when groups of Pintuhi Aborigines were living on the land, there were some 372 individual, recent fire scars visible on the aerial photographs (see fig. 2). Many of **these** concentrated around major salt lakes, claypans, and saltpans. While it is not possible to determine the ignition source of all fires, many scars showed the classic shape of having been lit by a person dragging a firestick in a straight line. Many of the fires burnt between the sparsely vegetated sand dune crests. By 1973, some 11 years after Aborigines began to leave the land, the small-graincd mosaic of burnt patches that existed in 1953, had begun to be erased and to be replaced by large tracts of recently burnt country and large tracts long unburnt. The ignition source of these fires may have been lightning or the few Aboriginal people who remained in the area. This temporal trend of increasing fire size and increasing pyric homogeneity has continued until the present day, and there now exists vast tracts of country burnt at **the** same time by a single fire or multiple lightning strikes. There arc also vast areas which have not been burnt for in excess of 30 years. This pattern is repeated across the Western and Central Deserts (Griffin and Allan 1985).

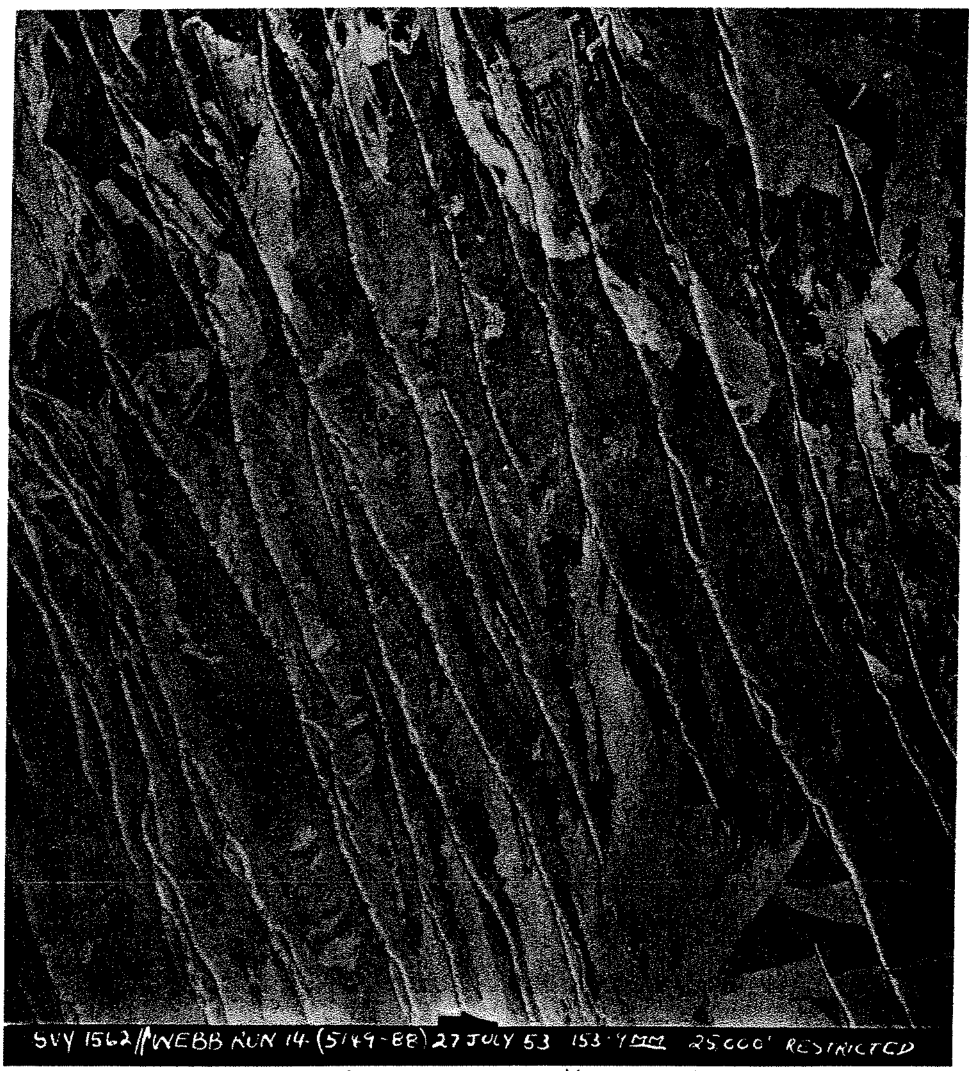


Figure 2. The patchwork of fire scars is clearly evident on early (1953) black-and-white aerial photographs. Most fires were lit by **Pintuhi** Aborigines Who occupied the desert at the time of photography.

The frequency distribution by size class of fire scars is shown in figure 3 for 1953 and 1973 photography. By 1977 and 1986, most of the study area had been burnt by only a few fires lit at about the same time.

A useful measure of the size and distribution of burnt and unburnt patches is the ratio of variance to mean patch size (Pcilou 1977). A random distribution produces a ratio of near 1. A uniform or over dispersed distribution results in a ratio greater than 1 while a clumped or contiguous distribution yields a value much less than 1. In 1953, the patchiness ratio of both burnt and unburnt areas was 0.62 to 0.72, or close to 1, indicating a random distribution of patches (figure 4). The mean length of continuous burnt vcgctation had increased from 467 mctrcs in 1953 to 2,570 meters in 1986 and the variance ratio had reduced to 0.43, indicating a clumped distribution of patch size and a significant increase in mosaic grain size (Peiiou 1977). Line transects across photographs of a portion of the study area to reveal a spectrum of recently burnt (< 10 years) and long unburnt vegetation are also useful for quantifying burn mosaics. This "substitute pattern" of alternating black and white stripes is a representation of the varying grain size of the two phase mosaic (Peilou, 1978). These patterns are presented in figure 5 for the 1953 photography and 1986 satellite imagery.

Field Survey of Biological Indicators

An example of the results of a vehicle traverse along a line transect showing the ages of vegetation since the last fire and the total number of animal species recorded from observations of tracks, diggings, and scats is shown in figure 6. The traverse was conducted in 1989. While most of the vcgctation, was about 6 years old at the time of the traverse, there were small patches of older vegetation which had escaped the most recent fire. Often, these patches were found on the leeward side. of natural fire barriers such as sand dune

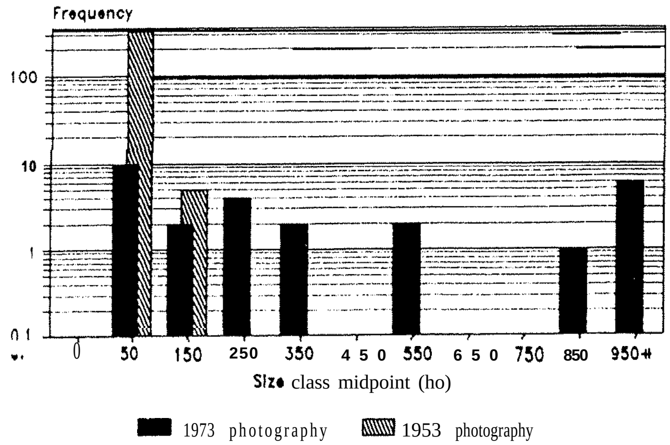


Figure 3. Frcqucncy distribution of size classes of recent fire scars evident on 1953 and 1973 black-and-white aerial photographs of a part of the Western Desert.

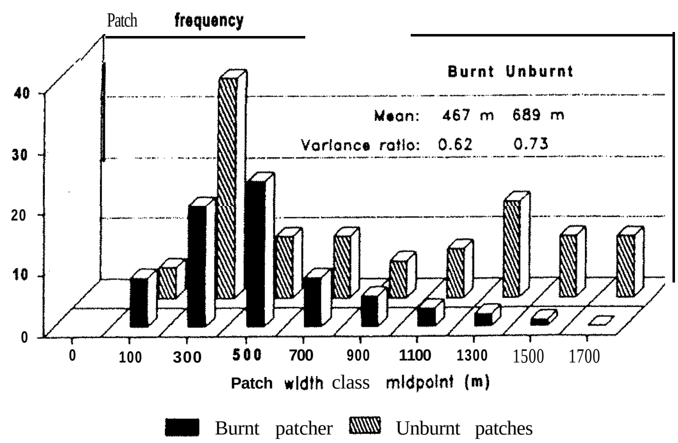


Figure 4. Percentage frequency distribution by size classes of the width of recently burnt (< 10 years) and long-unburnt patches of desert vegetation measured along transects across 1953 black-and-white aerial photographs.

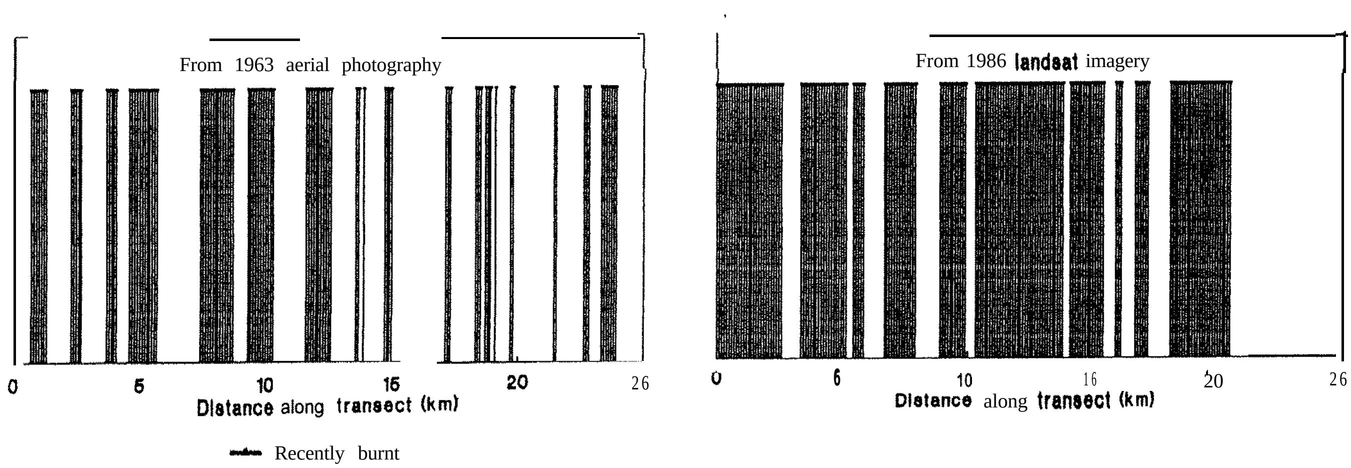
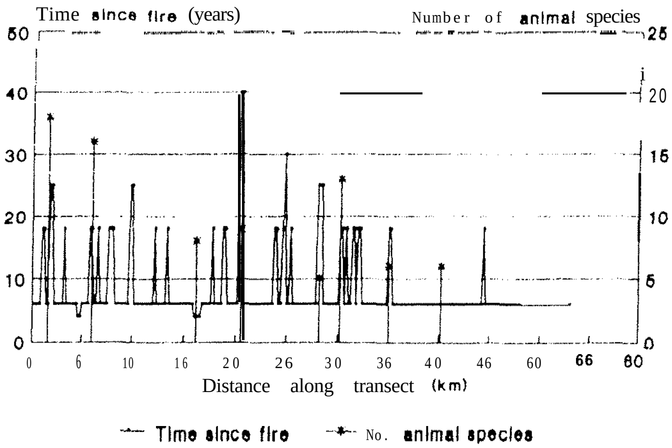


Figure 5. Substitute patterns for two two-phase mosaics (recently burnt and long-unburnt vegetation) of the Western Desert study arca from 1953 aerial photography and 1986 satellite imagery.



Figurc 6 Agc of vcgetation (time since firc) along a transect across dcscrt vcgctation in 1989. Also **shown** arc numbcrs of animal spccics cstimatcd from diggings, scats, and tracks at points along the transect.

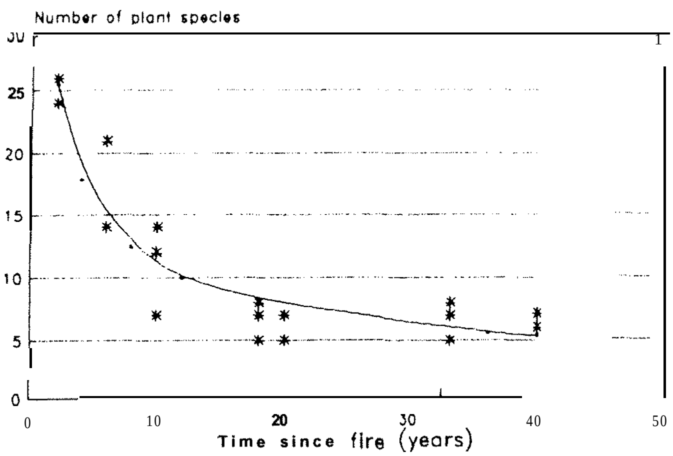


Figure 7. Numbers of plant spccics within a 100-mcter-radius plot with time since last fire in desert vegetation ncar Lake Mackay, Wcstcrn Australia.

crests, salt lakes, and claypans. Gcnerally , long unburnt patches were small (200-400 m across). The oldcst patch of vegetation was 40 $\pm$ 4 years. Animal species numbers, based on the survcy technique used here, were highest in areas where there was a mosaic of 6- and 1 S-year-old vegetation. Where vcgetation age was predominantly 6 years or less, or in one case where it was 40 years old, species numbers were low.

Plant species diversity dccreased with increasing time since fire (fig. 7). Older vegetation was dominated by large, senescent rings of spinifcx (*Triodia* species), whereas recently burnt patches contained a range of herbaceous plants as well as woody shrubs and spinifex. Many woody shrubs had resprouted from lignotubcr or regenerated from soil-stored seed following fire.

DISCUSSION

Fire size

Thcrc has bccn a dramatic and sudden increase in the mean and median size of burnt patches in the study area, coinciding with the dcparturc of Pintubi Aborigines. The average size of a burnt patch has increased from 34 hectares in 1953 to 32,184 hcctarcs in 1986 (table l). During Aboriginal occupation of this land, thcre existed a finc- graincd mosaic of burnt patches across the landscape as a rcsult of Aboriginal burning and natural lightning fires. The Pintubi and Pitjantjatjarra men who guided us around the study area, found old, long-unburnt spinifcx as aesthetically displeasing and continually fired the spinifcx as we travellcd across the dcscrt.

Kimbcr (1983) suggests that Aborigines had a good dcal of control ovcr firc, using wind, humidity and natural firc barricrs such as claypans and sand dunes to control thc size and intcnsity of fircs. Our observations of the concentration of firc scars around resource-rich arcas such as rockholes, crccks, claypans, and salt lakes is consistent with obscrvations madc by Kimbcr (1983). Thc fire regime around these resource rich landscape units may not be typical of arcas less utilized by Aborigines. Partitioning and utilization of thc landscape by Aborigines requires further investigation.

The abundance of small fire scars (< 10 hectares) visible on the 1953 aerial photography suggests either fircs wcre lit under mild conditions and burnt at a low intensity or cxtcnsivc tracts of heavy fuel Sufficient to sustain large and intcnse fires did not exist. Many generations of Aborigines frcquently burning off thc land would rcsult in a situation of discontinuous, patchy fuels, ranging from recently burnt to long-unburnt patches. However, when the human ignition sourcc was removed from the dcscrt, the fuels gradually accumulated over large, continuous tracts of land. Lightning strikes in summer have rcsultcd in the massive and intense wildfires obscrvcd today. The grain size of thc two-phase mosaic (recently burnt and long-unburnt) has increased over the last 36 ycars, as shown in figures 4 and 5.

Time of Year of Fires

Kimbcr (1983) believes that most burning by Aborigines was done in August to October and immediately prior to rains in December to February. Occasionally, large fires caused either by Aborigines or by lightning, burnt during the hot, dry, windy summer months. In the area studied here, it would not have been possible for a fire to become very large in 1953, as the fuels were discontinuous as a result of patch burning and natural fire barriers. Kimber suggests that the time of year for burning is not as important to Aboriginal people as the opportunity to bum. de Graaf (1976) observed that Aboriginal fires in the dcscrt were lit all year round and not seasonally.

Both de Graaf (1976) and Kimber (1983) reported that certain areas of the desert were not burnt by Aborigines because Aborigines did not visit these places for religious reasons, or feared that fire would destroy sacred objects. The season of burning was, however, very important in the monsoon forest regions of the Northern Territory (Jones 1980; Haynes 1985) and to Wadjuk Aborigines in the south-west of Western Australia (Hallam 1975).

Today the main ignition source in the remote deserts is lightning. Thunderstorms are common over the summer months and large, lightning-caused wildfires have been reported (Griffin and others 1983).

The benefits to Aborigines by way of increased food resources as a result of tiring the spinifex were evident during this study. Recently burnt country supported a diverse range of herbs and animals, particularly reptiles, whereas long-unburnt vegetation was generally less diverse in flora and fauna. However, we were unable to show any strong correlations between time since fire (fuel age) and total animal species numbers, although there was a trend between the spatial diversity of fuel age and animal numbers (fig. 6). Animal species numbers were based on visual observations of tracks, diggings, and scats, thus limiting the extent to which the data can be interpreted. More detailed studies of animal activity in relation to the temporal and spatial diversity of fuel age are needed. Postfire succession in spinifex communities has been described by Burbidge (1943) and Suijdendorp (1981). All authors report an increased level of plant diversity soon after fire. The range of tire-adaptive traits expressed by desert vegetation has also been reported by these authors.

As noted, there has been a significant change in fire regimes in recent times, and there is at least limited scientific evidence to suggest that this has contributed to the decline in mammal fauna. This change may have further predisposed mammals to predation by introduced predators. It is reasonable to accept the importance of temporal and spatial diversity within a landscape on resource levels and habitat opportunities (Latz and Griffin 1978, Pielou 1977). Saxon (1984) stated that "when large areas of a single landscape type are subjected to large uniform disturbances, they threaten the survival of wildlife species which depend on irregular boundaries of natural fire patterns to provide a fine grained mosaic of resources". Bolton and Latz (1978) have shown that a range of post-fire successional stages is important habitat for the western hare-wallaby (*Largorchestes hirsutus*). Burbidge and Pearson (1989) explain the lack of rufous hare-wallabies (*Largorchestes hirsutus*) in the Great Sandy Deserts as being due to the lack of frequency of small-scale bums and to high fox numbers.

It is somewhat ironic that modem mammal extinctions in the Australian deserts are in part due to the changed fire regime that resulted from the departure of Aborigines and consequent lack of Aboriginal burning. Tindale (1959), Merrilees (1968), and Jones (1968) have all concluded that extensive burning by Aborigines contributed to the extinction of the Pleistocene mega-fauna. Today, it is likely that the large, intense wildfires which occur throughout the deserts are placing extreme stress on some plant and animal communities (Griffin 1981). These intense fires are damaging vast areas of fire-sensitive vegetation such as marble gum (*Eucalyptus gongliocarpa*), desert oak (*Alocasuarina decaisneana*), and mulga (*Acacia anuera*) (Start 1986). There is sufficient evidence of the disadvantages of the current wildfire regime on the conservation status of desert reserves to warrant the development and implementation of managed fire regimes that mimic those in place during Aboriginal occupation of the land.

Today, aircraft are being used in parts of some Western Australian desert conservation reserves to set patch bums under carefully defined conditions to create a mosaic effect of burnt and unburnt patches across the landscape (Burrows and Thomson 1990). This study, together with detailed fire ecology studies currently in progress, will provide a basis for determining the frequency of fire and the size and distribution of burnt patches.

When the habitat has been rehabilitated through the prescribed use of fire and introduced predators controlled, then there will be an opportunity for translocating rare and endangered mammals to parts of their former range.

CONCLUSION

Black-and-white aerial photographs, Landsat imagery, and field observations have revealed that the size and intensity of fires in a part of the Western Desert west of Lake Mackay have increased dramatically over the last 36 years or so. This increase is attributed to the departure of Aboriginal people from the area during the 1960s. Aborigines used fire extensively for a multitude of reasons, resulting in a small-grained mosaic of burnt patches of different ages across the landscape. Such landscape diversity maximised resources available to Aborigines and to desert animals. Today, in the absence of frequent burning by Aborigines, fuels have accumulated over vast areas and when these fuels are ignited by lightning under hot, dry, windy summer conditions, large and intense wildfires sweep across the desert. This changed fire regime appears to have resulted in a lower diversity of animals and plants.

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INDIAN USE OF FIRE AND LAND CLEARANCE IN THE SOUTHERN APPALACHIANS

Michael S. DeVivo[1]

Abstract-The myth of an unbroken primeval forest, extending across eastern North America at the dawn of European settlement, has been perpetuated in the writings of both laymen and scholars throughout the present century. Accounts of sixteenth, seventeenth, and eighteenth century explorers, however, document vast amounts of cleared land held by aboriginal inhabitants, who likely populated the continent in much higher numbers than have been traditionally accepted. Fire was the principal tool used by the Indians to clear vegetation. Despite frequent historical reference to the Indian use of fire and the documentation of Indian old fields, the role of fire has been largely underplayed. Fire was implemented for forest management, driving game, and preparing land for agriculture. This paper examines the impact of fire and related anthropogenic disturbances on the southern Appalachian landscape hcfore white settlement.

INTRODUCTION

Man's role as an agent of landscape change has long been a theme of research in historical geography (Newcomb 1969). However, the influence of eastern North America's aboriginal inhabitants on the physical environment before European settlement has not been studied adequately. This paper offers an overview of anthropogcnic disturbances in the southern Appalachians during an era of aboriginal habitation, with especial reference to population, fire, land use, and land clearance.

ACCOUNTS BY EARLY EUROPEAN EXPLORERS

Permanent European settlement of the eastern United States did not occur until more than a century after the Columbus landfall in 1491, despite explorers' ventures along the coast, and subsequently into the interior of the continent, in pursuit of geographic knowledge during the sixteenth century (Brown 1948). Journals often contain comments as to the apparent presence or absence of forest, but other landscape traits were documented *as* well, such as aboriginal population and land use. In any event, the early explorers have provided documentation that refutes the popular notion of an unbroken virgin forest extending across eastern North America. Of particular significance were the travels of Giovanni Verrazano in 1524, Jacques Cartier a decade later, and Samuel dc Champlain in the early 1600s (Sauer 1971, 1980), but their accounts depict extensive areas of cleared land prior to European settlement only along the Atlantic coast and adjacent navigable waterways. Very few reports documenting conditions of the southeastern interior were provided by explorers. Narratives of the expedition led by Hcmando de Soto between the years 1538 and 1.543, however, offer illustrations of extensive maize fields, canebrakes, and open land on the southeastern coastal plain and in the southern mountains (Boume 1904; Hakluyt 1611; Rostlund 1957; U.S. 1939).

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ABORIGINAL, POPULATION AND DEPOPULATION

Supplementing explorers' accounts is a body of literature in which students have sought to determine the prc-contact aboriginal population of the Americas. These studies suggest that the aboriginal population of the southern Appalachians was much greater than heretofore accepted, and that it declined rapidly, largely because of disease transmitted by the Europeans following the Columbus landfall in 1492. Furthermore, aboriginal inhabitants were numerous enough and sufficiently advanced to significantly alter the region's vegetation.

Humans have occupied the southern Appalachians for over 12,000 years (Dickens 1976; Frizzell 1987). It is widely accepted that the Cherokees occupied the region at least since the protohistoric period; their settlement core was restricted to western North Carolina, cast Tennessee, north Georgia, and northwestern South Carolina, but claimed lands extending north to West Virginia and Kentucky (Dickens 1987; Frizzell 1987; Goodwin 1977). Mooney estimated the Cherokee population at no more than 22,000, a figure accepted and advanced by Kroeber (Dcncvan 1976; Goodwin 1977; Kroeber 1939; Swanton 1946).

Denevan, however, has indicated that the "authority of Kroeber has impeded serious consideration of North American aboriginal populations" and that "it is time for a reconsideration" (Denevan 1976). Unfortunately, one of the principal problems in estimating North American aboriginal populations, especially in the southeastern interior, is the lack of historical evidence. Historical documentation regarding Indian populations is more widely available for Latin American regions than for Anglo-America, largely because the former have a longer history of direct European contact and settlement. Because "estimates of aboriginal American populations have yielded a picture of small scale preconquest human population in the Western Hemisphere" (Dobyns 1966; Denevan 1976), anthropologist Henry Dobyns has proposed the use of depopulation ratios for calculating pre-Columbian populations. His estimates contrast rather sharply with Kroeber's; for example, Kroeber estimated a hemispheric total of 8.40 million, whereas Dobyns estimates 90.04-I 12.55 million (Dcnevan 1976), and Geographer Michael Williams has noted:

> There is the strong possibility that in the late fifteenth century the Western Hemisphere may have had a greater total population than western Europe. The implications of these figures for forest disturbance and destruction arc enormous (Williams 1989).

Today, Krocber's estimates arc considered much too low (Denevan 1976), and Dobyns' method offers an alternative, probably more realistic estimate. Dobyns' scheme employs a depopulation ratio, that is "the ratio of degree of decline from the time of contact to the population nadir" (Dencvan 1976). Dobyns determined the depopulation ratios for aboriginal tribes and regions for which relatively reliable information is available and derived an average of 20: 1 (a population decline of 95%). Borah further supports high figures; he estimated pre-contact New World native population at about 100 million, which suggests a depopulation ratio of up to 25:1 (Borah 1964; Dencvan 1976; Dobyns 1966).

The application of depopulation ratios is a viable means of estimating the pre-contact aboriginal population of the southern Appalachians. Archaeological evidence and travel accounts indicate rapid population decline during the sixteenth century, and the southeastern interior was not subject to exploration, description and interpretation, and settlement by Europeans until long after contact had been made on the East and Gulf coasts, as well as in the Southwest (Denevan 1976).

Figures of 7,400.10,000 are assumed to be rather reliable estimates for the Cherokees' nadir population during the early to mid-eighteenth century (Goodwin 1977), and application of a 20: I or 25: 1 depopulation ratio indicates that the pre-contact population would have been between 150,000 and 250,000. The difference between these estimates and Kroeber's estimate of 22,000 is highly significant. A quarter of a million inhabitants would have had a Iarger impact on the region's physical landscape than a comparatively scant population of 22,000 people. Obviously, the likelihood of extensive forest clearance would have been much greater.

The aboriginal inhabitants had been important components in the region's ecosystem. When disease, a major perturbation, decreased their numbers or eliminated them from some areas entirely, a major ecological change took place. Geographer Erhard Rostlund (1960) determined that buffalo (*Bison bison)* had not entered the Southeast by A.D. 1500, but migrated into the region after the middle of the sixteenth century, extending their habitat to the Atlantic and Gulf coasts. Historian Alfred Crosby suggests that the dramatic decrease in numbers of Amerindians opened up an econiche for the buffalo.

> Something had kept these animals out of the expanses of parklike clearings in the forest that periodic Amerindian use of fire and hoe had created. That something declined or disappeared after 1540. That something was, in all likelihood, the Amerindians themselves, who naturally would have killed the buffalo for food and to protect their crops (Crosby 1986).

ABORIGINAL SETTLEMENT FEATURES AND THE USE OF FIRE

Topography was probably the determining factor in the distribution of Indian settlements (Dickens 1976). Virtually all sites occurred along relatively extensive floodplains. A nucleated village of two or three acres appears to have been the predominant type of settlement, although sites could have been as small as a quarter of an acre, or as Iarge as six acres (Dickens 1976). Villages ranged in size from a few houses to perhaps as many as fifty houses surrounded by log palisades. Large areas of bottomland adjacent to the villages were probably maintained for agricultural activity (Dickens 1976). Fire was the principal tool used by Indians to clear vegetation. Despite Frequent historical reference to fire, and the documentation of "Indian old fields," Indian use of fire in North America has been greatly underestimated (Brown 1948; Day 1953; Gersmehl 1970; Goodwin 1977; Johanncssen and others 1971; Martin 1973a, 1973b; Maxwell 1910; Thompson and Smith 1970). Fire was useful in driving game and opening the forest "to increase visibility, improve forage, expose the mast, and help keep down the weeds" (Gersmehl 1970). In the Great Smoky Mountains, fires were set at frequent intervals to encourage the growth of certain plant species, such as blueberries (*Vaccinium vacillans*), which were useful for human consumption as well as wildlife habitat (Lindsay 1976).

The Cherokees, perhaps inadvertently, used fire for forest management. Plants having relatively little value, such as white pine, hemlock, birch, maple, and weeds were burned in order to encourage the growth of more valuable species. The Indians also burned the areas surrounding their villages to prevent catastrophic fires (Goodwin 1977).

Eastern Woodland Indians set fire periodically to bum accumulated litter and undergrowth and to encourage grassland (Thompson and Smith 1970). Periodic fire was especially important for the maintenance of prairies and canebrakes. Sondley has documented the existence of expansive grassland communities in the Asheville Basin at the dawn of white settlement.

> Most of the lands on and near the French Broad River . . were in prairies.... At the mouths of the smaller streams in that region tributary to the French Broad River were Iarge canebrakes extending for miles up those tributaries (Sondley 1930).

Ralph Hughes determined that under continuous protection from fire, cane stands "lose vigor, thin out and die" (1966). Moreover, cancbrake deterioration can be prevented with periodic fire. Thus, because cancbrakes were present in the Asheville Basin at the onset of pioneer settlement and because their maintenance requires frequent fire, it seems likely that the Cherokees set fire to the Asheville Basin at regular intervals in order to clear the land of brush and trees. Following pioneer settlement, a combination of fire suppression (or a decrease in burning), uncontrolled grazing, and cultivation of floodplains was probably responsible for the decline of the extensive canebrakes.

Perhaps the most widespread use of fire by the Indians was in the preparation of land for agriculture. Afer undergrowth was burned, larger trees were killed by girdling. Planting began when sunlight passed through the dead branches; maize, beans, and squash were usually planted in the same field (Brown 1948). Maize (*Zea mays*) was the most important staple in the Indians' diet, and may have been cultivated as early as 100 B.C. It was planted extensively on the floodplains of major streams and rivers. Corn was harvested in the late summer and early fall and was often processed into several different items. These most often included various flours and cakcs such as succotash, samp, hominy, hoecake, and ash-cake. Corn was also a principal ingredient in soups and stews. Beans (*Phaseolus*), probably introduced at a bout 800 A.D., were next in importance as a cultivated crop, (Yarncll 1976) and were usually planted alongside corn. In fact, cornstalks were often used as beanpoles. The usc of beans and corn in combination implied "complementation" in the natives' diet and as a result provided high nutrition. Squashes (*Cucurbitacae*) including pumpkins, gourds, and summer crookneck, were also an important staple. Certain squash varieties had been cultivated as early as 2300 B.C. The Clerokees planted squashes beside beans and maize. The sunflower (*Helianthus annus*) was probably domesticated during the. second or third millennium B.C. (Yarnell 1976) and had a multitude of uses. For example, its seeds yielded an edible table oil and flour that could be made into bread (Goodwin 1977).

Some wild edibles, such as spinach-like pigweed (*Amaranthus*) and gooscfoot (*Chenopodium album), grew* along wet ditches and streams. Blackberries (*Rubus argutus*), raspberries (*Rubus odoratus*), and blueberries (*Vaccinium vacillans*) were used. Nut-bearing trees provided the Indians with acorns, chestnuts, and walnuts, and sap from some trees provided sweetening agents such as maple syrup. Because Indians were strongly dependent on food from "wild" vegetation, some authorities believe that the Indians themselves were responsible for the wide distribution of certain trees, such a mockernut hickory (*Carya cordiformis*) and black walnut (*Juglans nigra*) (Goodwin 1977; Maxwell 1910).

The Cherokees obtained a number of fruits through contact with the Europeans in the sixteenth, seventeenth, and eighteenth centuries. These introduced fruits included watermelon (*Citrullus vulgaris*), peach (*Prunus persica*), apple (*Malus pumila*), and pear (*Pyrus communis*). Orchards were generally confined to moist, sandy soils at elevations below 3,000 feet; however, isolated stands of apple trees are at elevations up to 5,000 feet.

The Indians undoubtedly depended heavily on wild plants and animals for food, and they nurtured some forestland as a source of such foods. They also cleared large areas of settlements to provide fuelwood, and cleared extensive areas for agriculture.

Accounts of travelers document the abundance of cultivated fields and expansive grasslands throughout the Cherokee country. De Soto, in 1540, marched for a day through cultivated fields in southwestern North Carolina; subsequent explorers include De Luna in 1559-1561, Pardo in 15661568, Batt in 1667, Lederer in 1670, Needham and Arthur in 1673, Cuming 'in 1730, and Timberlake in 1762 (Boume 1904; Hudson and others 1985; Hudson 1987; Sondley 1977; Williams 1927, 1928). Perhaps Bartram's account of his travels of 1775 are the most informative. The botanist reported extensive open prairies and fields of corn along the Little Tennessee River Valley (Harper 19.58).

Virtually all permanent native settlements were limited to the floodplains of major streams and rivers for good reason. Their economy was based principally on agriculture, and it was impractical, if not impossible, to farm on steeply sloping terrain or at high elevations where erosion and microclimatic conditions were unfavorable. In southwestern North Carolina, the 3,000.foot contour follows the boundary between lands that are relatively suitable for agriculture and those that are not; the same is likely true for much of the southern Appalachians. Slopes are typically gentle below 3,000 feet and are typically much steeper at higher elevations.

Indians used fire to clear land for agriculture, and it is likely that some fires burned larger areas than intended. On shallow soils serotinous needleleaf conifers tended to dominate following a bum. On deep soils and open slopes hardwoods persisted or invaded after a fire. Deep-soiled, sheltered mesic sites were probahly less susceptible to burning.

Rostlund suggested that in the prc-contact southeastern United States, the aboriginal inhabitants' burning of vegetation resulted in maximum land clearance. After their depopulation and the concomitant reduction in frequent burning, the area of cleared land decreased and the proportion of forestland increased (Rostlund 1957). Silvical characteristics of some forested areas in the southern Appalachians seem to support

Rostlund's hypothesis. For example, in the contemporary Joyce Kilmer Memorial Forest in southwestern North Carolina, a high proportion of old-growth yellow-poplar dominates some mesic sites. Yellow-poplar is an aggressive pioneer species on fertile sites following drastic disturbance (McCracken 1978), and in examining the forest, Lorimer has determined that "disturbances as far back as 1.550 are almost certainly indicated by substantial numbers of the intolerant tulip tree in corresponding age classes" (Lorimer 1980). Possibly a fifteenth- to sixteenth-century Indian settlement that was located at sites now dominated by yellow-poplar (*Liriodendron tulipfera*) was a source of major disturbance. Perhaps after the European introduction and the subsequent diffusion of epidemic disease during the mid-fifteenth century, the settlement was largely depopulated, and pioneer species, such as yellow-poplar, invaded previously cleared areas.

CONCLUSION

The ability of aboriginal inhabitants to clear forest has often been grossly underestimated. In fact, anthropogenic perturbations over the last one or two millennia have accounted for much of the Southeast's forest composition, which is dominated by disturbance-initiated species (Buckner 1989). In the southern Appalachians, the Indians' livelihood depended on the use of fire to clear land and on the cultivation of crops along the floodplains of major rivers and their tributaries. Before contact with Europeans, the Cherokees and their ancestors probably cleared all bottomland in the region at one time or another. Moreover, the Cherokees' pre-Columbian ancestors may at one time have been numerous enough to clear all land below the 3,000-foot contour.

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SPECIAL SESSION: THE YELLOWSTONE FIRES

BARK BEETLE--FIRE ASSOCIATIONS IN THE GREATER YELLOWSTONE AREA

Gene D. Amman'

Abstract—The large forest tires in and around Yellowstone National Park in 1988 bring up many ecological questions, including the role of bark beetles. Bark beetles may contribute to fuel buildup over the years preceding a fire, resulting in stand replacement fires. Fire is important to the survival of scral tree species and bark beetles that reproduce in them. Without fire, seral species arc ultimately replaced hy climax species. Following fire, bark- and wood-boring beetles respond to fire-injured trees. Because of synchrony of the fires and life cycles of the beetles, beetle infestation in 1988 was not observed in fire-injured trees. However, endemic populations of beetles, upon emergence *in* 1989, infested large numbers of tire-injured trees. Of the trees examined in each species, 28 to 65 percent were infested by hark beetles: <u>Pinus contorta</u> (28 percent) by Ips pini; <u>Pseudotsuga menziesii</u> (32 percent) by <u>Dendroctonus pseudotsugae</u>; <u>Picee engelmannii</u> (65 percent) by <u>Dendroctonus rufipennis</u>; and Ahies <u>lasiocarpa</u> (35 percent) hy Buprestidae and Cerambycidac. Most trees infested by hark beetles had SO percent or more of their basal circumference killed by tire. Bark beetle populations probably will increase in the remaining tire-injured trees.

INTRODUCTION

Insects and diseases arc important in modifying the age structure and species composition of many forests. Their activities contribute to accumulation of dead fuels that make large-scale fires possible--resulting *in new stands of the* host tree. The stands arc then temporarily free of attack (Kilgore 1986). The mosaics of different-aged stands created as the result of fires assure survival of both trees and insects that infest them. However, fire is more. important to the survival of some ecosystems than others. Following fires, injured trees arc susceptible to infestation by bark beetles. Subsequent buildup of bark beetle populations can result in killing of uninjured trees.

In this paper I will discuss bark beetle ecology (1) as it may contribute to fuel buildup and fire intensity and (2) as it relates to fire-injured trees in the aftermath of forest fires. Lodgepole pine (<u>Pinus contorta</u> Douglas), the most prevalent tree species in the Greater Yellowstone Area (GYA) and one that we know the most about with respect to bark beetle-tree interactions, will be discussed more fully than other species.

BARK BEETLES AS CONTRIBUTORS TO FUEL BUILDUP

Pfister and Daubenmire (1975) recognized four basic successional roles for lodgepole pine: minor seral, dominant seral, persistent, and climax. Large areas of lodgepole pine in the GYA have almost no spruce-fir component. Despain (1983) concludes these arc essentially self-perpetuating climax lodgepole pine stands that often exceed 300 to 400 years of age, with no cvidcncc of fire since establishment.

Mountain pine beetle (MPB) infestation characteristics differ by lodgepole pine successional roles. In stands where lodgepole pine is seral and stands have been depleted by beetle infestations, lodgepole *will* be replaced by the more shade-tolerant species in the absence of fire. These shade-tolerant species consist primarily of Douglas-fir (<u>Pseudotsuga mcnzicsii</u> [Mirb.] Franco) at the lower elevations and subalpine fir (<u>Abies lasiocarpa</u> [Hook.] Nutt.) and Engelmann spruce (<u>Picca engelmannii</u> Parry) at the higher elevations. Starting with the stand generated by fire, lodgcpolc pine grows rapidly and occupies the dominant position in the stand. Fir and spruce seedlings also become established in the stand but grow more slowly than lodgepole pine.

Once the lodgepole reach susceptible size, MPB infestations kill 30 to over 90 percent of trees 12.7 cm and larger diameter at breast height (Cole and Amman 1980; McGregor and others 1987). After each infestation, both residual lodgcpolc pine and the shade-tolerant species increase their growth (Roe and Amman 1970). Infestations are repeated as the residual lodgcpolc pines reach size and phlocm thickness conducive to beetle infestation and survival (Amman 1977). This cycle is repeated at 20- to 40-year intervals, depending upon growth of the trees (Roe and Amman 1970). Although size and phlocm thickness arc: the variables necessary for beetle epidemics to occur, some authors (e.g., Berryman 1978) bclicvc trees must be weakened before MPB can infest them. Howcvcr, this has not been demonstrated, and will require detailed studies of beetle populations progressing from low level into the early phases of an epidemic (Schmitz 1988). Fuel levels and fire hazard continue to increase with each beetle infestation (Brown 1975; Flint 1924; Gibson 1943; Roe and Amman 1970) until lodgcpolc pine is

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eliminated from the stand, or until a fire occurs that kills most trees (including thick-barked, fire-resistant species), and the stand regcncratcs to lodgepole pine.

Where lodgcpole pine is persistent or climax (Pfister and Daubcnmire 1975), the association of lodgepole pine and mountain pine beetle is somewhat different. In these cases, the forest consists of lodgepole pine of different sizes and ages, ranging from seedlings to a few overmature trees. In these forests, MPB infests and kills many of the pines as trees reach large size. Openings created in the stand, as a result of the larger trees being killed, are seeded by lodgepole pine. The cycle is then rcpcatcd as other lodgcpole pines reach sizes and phloem thicknesses conducive to increases in beetle populations (Amman 1977).

Amman (1977) hypothesized that periodic MPB infestations continue the multi-aged nature of the stands. A mosaic of small clumps of different ages and sizes may occur. The ovcrall effect is likely to be more chronic infestation by the beetle because of the more constant source of food. Beetle infestations in such forests may result in the death of fewer trees per hectare during each infestation than would occur in even-aged stands (caused by stand replacement fires) and in those stands where lodgcpolc pine is seral. Fires in uncvcn-aged persistent and climax lodgepole pine forests should not be as hot as fires where widespread epidemics of beetles have occurred because smaller, more continuous deposits of fuel are added to the forest floor under chronic beetle infestations. Thus, with lighter accumulations of fuel, tires tend to eliminate or weaken some of the trees but do not cause total elimination and complete regeneration of the stand. An example is the situation described by Gara and others (1985) in south-central Oregon, where lodgepolc pine forms an cdaphic climax. Here, fires are slow moving, and the heat of smoldering Iogs scorches roots and sides of trees. Later these injured trees arc invaded by fungi that work their way up the roots into the trunks. Subsequently, mountain pine beetles arc attracted to and kill these trees. As the dead trees rot and fall over, the stage is set for another fire.

Most tires that occur in lodgepole pine arc either slow and smoldering or are rapidly moving, intense crown fires (Lotan and others 1985). High-intensity tires tend to favor lodgepole pine over such species as Douglas-fir (Kilgore 1986) and would likely occur following epidemic beetle infestations. Brown (1975) states that the major vegetation pattern found in lodgcpolc pine today was caused by stand replacement fires, although many uneven-aged lodgepolc pine stands result from lower intensity surface fires.

In south-central Oregon, Stuart and others (1989) have related lodgcpolc pine regeneration pulses to mountain pine beetle and fire disturbances. They observed that (1) stands that experienced periodic MPB epidemics accompanied by a fire had an even-aged structure; (2) stands that had periodic MPB epidemics and no fire had a storied, bimodal size structure; and (3) stands that experienced mortality by low level MPB populations, with or without low intensity fire, had multi-aged structure.

Rommc and others (1986) examined the effects of beetle outbreaks on primary productivity in forests dominated by lodgepole pine in northwestern Wyoming. They concluded that the mountain pine beetle does not regulate primary productivity. Even though MPB has drastic effects upon stands (considering the forest landscape comprises a mosaic of stands in various stages of succession), annual productivity for the landscape is relatively constant despite continual fluctuations of individual stands. The sudden and massive death of a large proportion of the biomass leads to only a brief drop in primary productivity and to a more equitable distribution of biomass and resources. Therefore, the primary function of large MPB infestations and the death of large numbers of lodgcpole pine appears to be survival of host and beetle by creating large amounts of fuel for fire that, when ignited, eliminate competing vegetation and regenerates lodgcpolc pine (Amman 1977; Roe and Amman 1970; Romme and others 1986).

The mosaic of stands of different ages created by the action of MPB and fire is ideal for MPB survival. Because stands arc coming into sizes conducive to continual MPB infestation and survival, a continual supply of food is provided. However, an ideal mosaic for MPB probably did not occur following the 1988 GYA fires because fire behavior was influenced more by drought and wind than by fuels. Virtually all forest age and fuel categories burned (Christensen and others 1989).

Romme and Despain (1989) state that the mosaic created by the 1988 fires will be more homogeneous than the mosaic created by fires in the early 1700's, and few ecological consequences will be incurred because succession is slow. One consequence is likely to be a major MPB infestation in 80 to 120 years because at this age many lodgepole pine stands sustain their first beetle outbreak, again creating a large amount of dead fuel in a relatively short period, setting the stage for another stand replacement fire (Roe and Amman 1970). The timing of MPB infestations, when lodgepole pine arc mature in seral stands, not only assures large amounts of fuel from the dead trees for a stand replacement fire but also adequate seed to regenerate the stand (Pctcrman 1978). Pctcrman suggests the ecological role of MPB could be to decrease the probability of lodgepole stands, with a high degree of serotiny, producing stagnant stands of offspring. By preventing the stand from getting too old, much less seed would be available. Such a mechanism could have evolutionary significance to lodgepole pine because stagnant stands do not reproduce well, and the stand following the stagnant stand could be outcompeted by climax tree species. Pcterman further points out that prevention of stagnant stands would bc advantageous to MPB because the beetle does not reproduce well in small, stagnant trees.

The contribution of dead fuel buildup, a result of the 391 000 ha infestation of MPB in Yellowstone National Park that was still active in 1982 (Gibson and Oakes 1987), to behavior of the 1988 fires was masked by the extreme fire conditions (Christensen and others 1989). Studies of small fires in portions of Yellowstone not involved in the 1988 tires probably would elucidate interactions of MPB infestations, dead fuel buildup, and tires. A relationship similar to MPB, lodgepole pine, and fire has been proposed for southern pine beetle (SPB) (D. frontalis Zimmermann) and pines in the Southern United States. There, pines arc replaced by hardwood tree species in the absence of fire (Schowalter and others 1981). Therefore, survival of SPB and its host in natural stands is dependent upon frequent fires.

Bark beetles infesting climax tree species would not have the same need for a close relationship with forest fires as those infesting seral species. The spruce beetle (SB) (D. rufipennis [Kirby]) and the Douglas-fir beetle (DFB) (D. pseudotsugae) usually kill small groups of trees. However, occasionally they also cause heavy mortality, favoring large trees over vast areas, after building up in windthrown trees. For example, SB killed millions of Engelmann spruce in Colorado between 1939 and 1951 (Massey and Wygant 1954) and white spruce (P. glauca [Moench] Voss) in Alaska between 1960 and 1973 (Baker and Kemperman 1974). Schmid and Hinds (1974) describe the scenario in spruce-fir stands in the central Rocky Mountains following spruce beetle infestations. Following a spruce-beetle outbreak, the percentage of subalpine fir in the stand increases, with fir dominating the stand. As fir reach 125 to 175 years of age, they begin to die, with the bark beetle Dryocoetes confusus Swaine being one of the mortality factors. Young spruce and fir increase their growth as overstory fir die. The less shade-tolerant spruce is then favored over fir as the original canopy fir are killed. Spruce becomes dominant as it outlives fir and gains greater size. Eventually, the cycle is repeated. Spruce beetle generally live in moist forests where fires are less frequent and intense because of moist, sparse fuels (Amo 1976). Small fires in the spruce-fir type would expose mineral soil and probably favor establishment of spruce.

The Douglas-fir beetle seldom creates widespread destruction in the Rocky Mountains, generally killing groups of dense mature Douglas-fir (Furniss and Orr 1978). These groups are usually widely separated, and the space created by death of some overstory trees usually regenerates to Douglas-fir.

These observations suggest coadaptive or coevolutionary relationships between bark beetles and their host trees, and the importance of fire in maintaining these relationships for seral tree species.

BARK BEETLE/FIRE-INJURED TREE ASSOCIATIONS

Following the 1988 GYA fires, large numbers of trees girdled or partially girdled by heat remained at the bum perimeter and are providing infestation opportunity to bark beetles. Beetles may increase to large numbers and infest uninjured trees after most of the fire-injured trees are killed.

The bark beetle situation in the GYA at the time of the 1988 tires shows that the species were at low population levels, except the DFB. The massive infestations of MPB that covered over 391 000 ha in Yellowstone Park in 1982 had declined to only 135 ha by 1986 (Gibson and Oakes 1987) and to no infested trees in 1987 (Gibson and Oakes 1988). In 1988, insect detection flights over the park were not made because of tire fighting efforts and smoke (Gibson and Oakes 1989). However, on the nearby Bridger-Teton National Forest, MPB infestation had declined from 1,296 ha in 1987 to 364 ha in 1988 (Knapp and others 1988).

Although no survey estimates are available for other bark beetle species in Yellowstone Park, surveys of adjacent areas showed only the DFB was increasing, whereas spruce beetle infestation was light (Knapp and others 1988) and pine engraver (Ips pini Say) populations had declined (Gibson and Oakes 1989).

The small populations of bark beetles in the GYA at the time of the 1988 fires, coupled with timing of the tires in relation to life cycles of bark and wood infesting beetles, resulted in few fire-injured trees being infested in 1988. The SB, DFB, and pine engraver all emerge to infest new material in the spring, prior- to occurrence of the fires. The MPB emerges in late July and early August, but few were in the GYA.

Studies were started in 1989 to determine bark beetle infestation of fire-injured trees and potential buildup of beetle populations. Observations were made in three areas: (1) near the Madison River, approximately halfway between Madison Junction and West Yellowstone (the North Fork tire); (2) along the John D. Rockefeller, Jr., Memorial Parkway, south of Yellowstone's South Gate (the Huck fire); and (3) in the Ditch Creek area of the Bridger-Teton National Forest (Hunter fire). In each area, variable plots (10 basal area factor) were established: area 1, three plots; area 2, nine plots; and area 3, seven plots. All trees in the plots were numbered so that survival of individual trees can be followed for several years. Survival of scorched trees can be predicted from volume of crown scorch (Ryan and others 1988). Peterson and Arbaugh (1986) found crown scorch and basal scorch were best predictors for lodgepole pine survival, and crown scorch and insect attack were most important as predictors of survival of Douglas-fir. However, the researchers did not identify the insects. I used the percentage of basal circumference in which the cambium was killed, rather than relating infestation to crown scorch, because of the high sensitivity of lodgcpolc and spruce to even light ground tire. Some bark was removed from trees infested by insects so that insects could be identified. Because our plots were mostly at low elevations (2 050 to 2 400 m), trees consisted mostly of lodgcpolc pine and Douglas-fir. The limited nature of our observations preclude their use for making predictions of bark beetle activity beyond our plots. Greater coverage of the burned area is planned in 1990.

Lodgepole Pine

Lodgepole pine is the most abundant tree in the samples. Overall, 28 percent of the trees were infested by the pine engraver (*Ips pini* Say) (table 1). infested, y one had not been scorched by fire. All others had 50 percent or more basal girdling (phloem killed by fire). Most commonly, trees infested by the pine engraver had 100 percent basal girdling (table 2). Many of these trees showed little evidence of scorch and looked healthy except for boring frass made by the beetles. Upon closer inspection, however, the trees were completely girdled at the base by a light ground tire. Geiszler and others (1984) also found most lodgepole pine infested by pine engraver were moderately to heavily injured following a fire in Oregon.

It is not surprising that a large number of trees were infested by pine engraver because they are able to reproduce in wind-broken material (including large branches) and in decadent trees near death (Sartwell and others 1971). There always seems to be plenty of such material available. Consequently, the engraver is almost always present in substantial numbers, although not necessarily causing noticeable tree mortality.

Only one tree containing MPB was observed (Hunter fire on the Bridger-Teton National Forest) and it was not on a plot. Observations over the years suggest that MPB is not strongly attracted to tire-scorched trees, so few trees would be infested even if a large population had been present in the GYA. The MPB seldom breeds in trees injured or killed by tire in numbers sufficient to cause an increase in the population. Hopkins (1905) found no MPB in tire-injured ponderosa pine in the Manitou Park area of Colorado. However, he did observe several secondary species, including the red turpentine beetle (*D. valens* Lec.). bsequent . publication concerning insect damage in the National Parks, Hopkins (1912) stated that forest fires contribute, to a limited extent, to the multiplication of certain species that breed in fire-scorched trees, but as a rule forest fires kill more beetles

Table 1.--Number of trees examined and the percentage infested by bark- and wood-boring beetles for plots Located in three fires in the Greater Yellowstone Area, 1989

Tree species	Fire: North Fork		Huck		Hunter		All fires	
	No.	Pct	No.	Pct	No.	Pct	No.	Pct
Lodgepole pine	0	0	67	24	59	33	125	28
Douglas-fir	34	18	25	50	4	25	63	32
Engelmann spruce	0	0	2	33	15	67	17	65
Subalpine fir	0	0	9		8	38		35
All species	34	18	103	31	85	38	2::	32

Table 2.--Number and percentage of trees infested by bark- and uood-boring beetles in different fire-injury categories, Greater Yellowstone Area, 1989

Tree species	Percentage of basal circumference killed by fire: 0		1-25		26-50		51-75		76-100	
	No.	Pct	No.	Pct	No.	Pct	No.	Pct	No.	Pct
Lodgepole pine / Douglas-fir	21 17	28	4 3	0	15 10	39 0	12 11	36 25	73 22	41 41
Engelmann spruce	0	0	1	0	0	0	0	0	16	69
Subalpine fir	0	0	0	0	0	0	0	0	17	31
All species	38	16	8	0	25	12	23	36	128	43

than they protect (by protect, he probably meant provide breeding habitat). Swaine (1918), referring to Canadian conditions, wrote that ground tires that injure and kill large numbers of trees may provide material for rapid development of bark beetles. He thought this was particularly true if fires occur year after year in neighboring localities. Apparently the proximity of fires would allow beetles to continue to build up their populations for several consecutive years. Blackman (1931), working on the Kaibab National Forest in northern Arizona, found MPB did not prefer fire-scorched trees. He thought the scorched phlocm did not offer favorable conditions for beetle offspring. The MPB has fairly limited requirements of phloem thickness and moisture in order to reproduce (Amman and Cole 1983).

In agreement with most observations in the Rocky Mountains that MPB are not attracted to fire-scorched trees, Geiszler and others (1984) observed MPB mostly in trees uninjured or lightly injured by fire, in direct contrast to pine engraver in moderate to heavily injured trees. Rust (1933) reported fire-injured ponderosa pine were infested by MPB the first year following a tire in northern Idaho; however, the infestation declined the next year.

The wood borers, both Buprestidae and Ccrambycidae, were found occasionally in fire-injured lodgepole.

Douglas-fir

Douglas-fir was the second most common tree found on the plots. Of the trees examined, 32 percent were infested by insects, mostly DFB and a few wood borer larvae of Buprestidae and Cerambycidae (table 1). Most infested Douglas-Iir had 50 percent or more girdling by fire (table 2). Some Douglas-firs that had needles and limbs completely burned were infested by DFB in the base where the bark was thick enough to protect the phloem from complete incineration or from drying so excessively that beetles would not construct egg galleries in it. Phloem in such trees was completely brown, and larvae probably will not complete development in such trees.

Fumiss (1965) studied the susceptibility of fire-injured Douglas-fir to bark beetle attack after a large fire in southern Idaho. He found 70 percent of the trees were infested by DFB 1 year after the fire. And even small or lightly burned trees attracted the beetles. He found incidence of attack increased with tree size and severity of crown and cambium injury by fire. However, infestation decreased sharply with outright tree killing by tire. Although beetles established brood in 88 percent of the trees, offspring numbers were small because of pitch invasion of the galleries and sour sap condition.

Fumiss (1965) did not report on DFB infestation in tire-scorched Douglas-fir beyond the first postfire year. However, following the Tillamook fire of 1933 in the coastal range of Oregon, DFB buildup in fire-injured Douglas-fir occurred. Beetles then killed large numbers of uninjured trees in 1935 and 1936, but the infestation soon subsided (Fumiss 1941). Fumiss thought beetles were able to increase because frequent tires in the Tillamook area provided large numbers of injured trees in which the beetles could reproduce.

Connaughton (1936) observed that delayed mortality of fire-injured Douglas-fir was mostly caused by insects (probably DFB) and fire damage to roots. He found Douglas-fir had a thick layer of duff around the trunk that burned slowly, heating the soil and badly injuring the roots. The evidence for root injury did not show up until a year or two after the fire in west-central Idaho.

Engelmann Spruce

Engelmann spruce constituted a small part of our tree sample, with only 17 trees examined. Spruce beetle infested 65 percent of the trees (table 1), and these were usually the larger diameter trees. Of the spruce, only those with 7.5 percent or greater basal girdling were infested (table 2). Some spruce burned similarly to Douglas-fir described by Connaughton (1936). Duff around the base resulted in a slow burning tire that often burned off the roots or so weakened them that the trees were easily blown over by wind. Windthrown trees with unscorched trunks created an ideal habitat for the SB, which shows a strong preference for windthrown trees (Massey and Wygant 1954; Schmid and Hinds 1974). Large numbers of spruce beetle larvae occurred in the spruce, as well as some larvae of Buprestidae and Cerambycidae.

Subalpine fir

Wood borers (Buprestidae and Cerambycidae) infested 35 percent of the 17 subalpine fir in the sample (table 1). All of the fir suffered 100 percent basal girdling. The bark was badly burned and not conducive to bark beetle infestation (table 2).

Whitebark Fine

Whitebark pine (P. albacaulis Engelm.), which is generally found at high elevations in GYA, did not occur in any of our plots. MPB infestations during the past 20 years caused considerable whitebark mortality (Bartos and Gibson 1990), but the number of infested trees was low at the time of the 1988 fires. Although MPB is not strongly attracted to fire-scorched lodgepole and pondcrosa pines in the Rocky Mountains, Craighead and others (193 1) state that it prefers weakened and tire-scorched western white pine (P. monticola Dougl.), one of the five-needle pines. Therefore, MPB may be more attracted to fire-injured five-needle pines, whitebark and limber (P. flexilis James), than to lodgepole pine.

CONCLUSIONS

Of the bark beetles in the GYA, MPB plays a significant role in converting live fuels to dead fuels in a relatively short period. This behavior probably promotes hot stand replacement tires that assure survival of lodgepole pine and, hence, survival of MPB. Fire is not as important in the ecology of bark beetles infesting climax tree species.

Although a limited number of fire-injured trees were sampled in the GYA, almost one-third were infested by bark beetles. Therefore, numbers of infested trees in the sampled areas likely will increase because of the remaining large numbers of tire-injured trees.

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YELLOWSTONE MEDIA MYTHS: PRINT AND TELEVISION COVERAGE OF THE 1988 FIRES

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Abstract-This paper draws on comments from 89 reporters who covered the fires, on comments from 146 of their news sources, and on evaluations of network television coverage by four groups of wildfire experts. The research also incorporates a content analysis of stories about the fires that appeared in Yellowstone-area and elite newspapers. The results suggest that repotters sometimes made serious factual errors, and often did a poor job of reporting on ecological issues and fire management policy. There were substantial differences in how the fires were covered by different news organizations.

INTRODUCTION

Molotch and Lester (1974, 1975), who examined hundreds of newspaper stories about the 1969 Santa Barbara oil spill, concluded that the contents of the news accounts were not determined by objective characteristics of the spill, but rather by a power struggle among various news sources who had vested interests in differing interpretations of the event. Only the local newspaper framed the story in the way it was perceived by Santa Barbara residents.

In the present paper, coverage of the Yellowstone fires by six newspapers and the three television networks is interpreted as a power struggle among sources offering two competing interpretations of the event: 1) Enlightened public land managers attempted to maintain the ecological integrity of a pristine national park by following a scientifically-based fire suppression regime which treated wildfire as a natural and necessary part of the biological process that shaped the ecosystem, and 2) Inept government bureaucrats allowed a national treasure to be destroyed because of their insensitivity to the beauty of Yellowstone forests and a cavalier attitude towards the fears of local residents and the right of local merchants to realize a fair return from investments in tourist-related business ventures.

BACKGROUND

On its surface, news can be viewed as an objective account of reality, as an impartial reflection of what happened. This is the newsgathering model offered by many journalists, and the goal described by various professional codes of journalistic ethics, which identify the search for truth as the most basic goal of all journalistic endeavors. In the real world of ncwsgathcring, however, reporters must make many value-driven choices that shape the ensuing stories. Who to interview? What questions to ask? Which facts to include at the expense of others that arc left out? What angles should be emphasized? What kinds of stories arc being written by competing reporters? What instructions have been received from an editor or producer? How much time is there before deadline? What kind of story will advance my career?

The impression made by news accounts is also shaped by editorial decisions that determine when the story is important enough for a newspaper or network to assign a reporter rather than relying on wire service accounts, the decision about who to assign to the story, and the decision about where to place the story and how much time or space to give it.

The Yellowstone fires were difficult to cover to the extent that they occurred outside the normal news routine. National reporters had to find their bearings in an unfamiliar place, and to seek information and identify new sources from scratch. Most of journalism has to do with routine stories covered from fixed locations through repeated contact with established sources. On the other hand, the urban fire is one of the most basic stories in the journalist's repertoire, and that made coverage easier because the urban fire model could be used as a model for covering wildfires. When reporters have little expertise about an event, they arc more likely to rely on their personal values to interpret it (Gans 1979), and more likely to borrow information and story angles from other reporters (Gitlin 1980). Research by Patterson (1989) and Wilkins (1987) indicates that disaster coverage tends to focus on immediate events rather than the context in which they occur, and suggests that these stories are often told in terms of cultural stereotypes and not as objective accounts of what happened. A study of news stories about environmental issues related to construction of the Tellico Dam (Glynn and Timms 1982) indicated that the snail darter fish itself, rather than the issues, dominated coverage.

Media scholar Gaye Tuchman (1978) says that journalists create news stories by transforming real events into a socially constructed "reality" that meets the organizational needs of news work. Some sources and facts are discarded, she observes, because of shared notions among journalists about what constitutes news. This process, according to sociologist David Altheide (1976), often distorts events by removing them from the context in which they occurred. "Journalists,"

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writes Altheide, "look for angles, interest, and entertainment value." Some of the ways in which reporters frame news, according to Gitlin (1980), "can be attributed to traditional assumptions in news treatment: news concerns the event, not the underlying condition; the person, not the group; conflict, not consensus; the fact that advances the story, not the one that explains it."

Many of the reporters who covered the Yellowstone tires were general assignment reporters rather than specialists in regional or environmental subjects. Herbert Gans (1979) observes that general assignment reporters "are like tourists, albeit in their own culture; they seek out what is memorable and perceive what clashes with the things they take for granted." Because of this, national news accounts of local stories are almost always inaccurate and exaggerated.

METHOD

This study is based on examination of 814 news accounts about the Yellowstone tires that appeared in 1988 in three elite American newspapers (the New York Times, Washington Post and Los Angeles Times), three Yellowstone-area newspapers (the Billings, MT Gazette, the Bozeman, MT Daily Chronicle and the Casper, WY Star Tribune), and in the evening newscasts of the three commercial television networks. The three elite newspapers are widely considered America's best and most prominent, while the three area newspapers circulate in Yellowstone and adjacent communities.

Yellowstone-area newspaper stories were obtained from the newspapers themselves, and newspaper employees judged whether individual stories should be categorized as being about the Yellowstone *fires.* Stories from the elite newspapers were obtained from the VuText and Nexis electronic databases, which allowed computer retrieval of all stories that contained the words "Yellowstone" and "tire" or "wildfire" (except for wire service stories in the Washington Post, which are not included in either database). Television stories were obtained from the Vanderbilt Television News Archive in Nashville.

The New York Times, which is published in the nation's media capital, received special scrutiny. It is widely read by journalists, and is often used by the networks and by journalists not only as a source of news, but also as a guide to the importance of stories and as a guide to how to cover stories (Gitlin 1980).

This paper also draws on two earlier studies by the author. One was about the Yellowstonc tires as seen by 68 print journalists who covered them and by 146 news sources for newspapers and news magazine stories (Smith 1989a). The other was based on evaluations of all 1988 evening network television stories about the fires by incident commanders, forest ecologists, wildfire behavior experts, and fire management policy experts (Smith 1989b).

RESULTS

Each of the six newspapers published its first account of the Yellowstone fires between July 1 and July 8. ABC and NBC television broadcast their first stories *on* July 25, after the evacuation of Grant Village. The first CBS story was broadcast on August 22, when soldiers joined the firefighting effort.

The Yellowstone tires were more newsworthy in the west than in the east. They made the front page of the Los Angeles Times 39 times, starting on July 18 with a news brief about wildfires in the west; the front page of the Washington Post three times, starting on September 8 after the fire's visit to the Old Faithful Geyser Complex; and the front page of the New York Times three times, starting on September 11 when the secretaries of Interior and Agriculture arrived in Yellowstone for an inspection. Stories about the fires appeared in the first five pages of the Washington Post 17 times, but only three times (the front page stories) in the New York Times.

The first Los Angeles Times story written by a full-time staff reporter for the paper (Tamara Jones) was published on August 24. The Los Angeles Times did not use freelance stringers to cover the fires. The Washington Post and New York Times, however, relied partly on outsiders. Freelancer Geoffrey O'Gara wrote seven stories for the Post. The first of these appeared on July 17. The New York Times also made use of material provided by stringers, starting with an August 10 article by Jim Robbins.

Although fire visited the Old Faithful geyser complex only on September 7, the geyser was a recurrent theme in news stories as a symbol of the park. Old Faithful is mentioned in 13 of 47 stories about the tires in the New York Times, in 13 of 41 stories about the tires in the Washington Post, and in 24 of 75 stories in the Los Angeles Times. The first stories on ABC and NBC also mentioned Old Faithful, and pictures of the geyser appeared in 18 network stories about the fires.

All of the Yellowstone tires were classified as wildfires on July 21, and were subjected to full suppression (Christensen 1989). However, I was unable to find any mention of this fact in any news report published or broadcast during July or August. Several news organizations did quote Interior Secretary Donald Hodel as saying on July 27 that all new tires would be suppressed (emphasis added), but many reporters retained the impression some fires were being allowed to burn unchallenged, and perhaps unmonitored, through all of August and into September.

Coverage in the New York Times

A free-lance story by Jim Robbins (1988a), published in the New York Times on August 10, said the abandoned natural-bum policy was still in effect, and was "the talk of the campgrounds and restaurants" in the Yellowstone area. Four days later, another story (Robbins 1988b) said that some tires were being fought, but that a dozen were being allowed to burn. On September I, yet another New York Times story (Wilson 1988), said "Some of the fires arc allowed to bum unchallenged as part of a philosophy that holds they arc a natural process." A September 10 article (Shabecoff 1988) described criticism of Yellowstone's natural-bum policy by Wyoming senators Alan Simpson and Malcolm Wallop without explaining that suppression of all fires bcgan in July.

Seven weeks after all fires were in suppression mode, the Nation's most influential and prestigious newspapcr thus continued to support the myth that some of the tires were being allowed to burn. A search through the Ncxis computer database for all 1988 New York Times stories containing the words "correction" and "Yellowstone" indicates that no corrections of this mistake were ever published.

The language used to frame New York Times stories about the tires sometimes encouraged the idea that they were being managed ineptly and insensitively. On August 14: "It may seem strange to a generation that grew up with stem admonitions from Smokey Bear, but the Park Service refuses to use words like 'damage' or 'destruction,' and instead describes how the tires will rejuvenate aging park forests and benefit wildlife" (Robbins 1988b, emphasis added). This clearly implies deviant behavior ("strangeness" and the "refusal" to use "reasonable" language). On September 11, in the Times' first front-page story about the tires: "(O)fficials could not keep up with reports of areas threatened by the blazes." "Evacuations were so numerous it was hard for park officials to keep track of them." (Robbins 1988d). The language here implies a park administration in disarray. On that same day, the major local paper, the Billings Gazette, had no trouble keeping track of the same evacuations. September 22: "(E)ven at the height of the fires, bulldozers were allowed into the park only on a case by case basis" (Egan 1988, emphasis added). The qualifying phrase tends to cast doubt on the management policy.

When Democratic Presidential candidate Michael Dukakis visited Yellowstone on September 15, the Times was the only elite newspaper to include an observation alluding to the Bambi myth that animals cope poorly with wildfire. The account describes Dukakis reading a letter from a firefighter received from a little girl who wrote, "I wish you could help the animals" (Toner 1988).

The kinds of factual errors described above continued in the second New York Times front-page story, published on Scptcmber 22 (Egan 1988). This story said that the government had a policy of allowing all naturally-caused fires in parks and wilderness areas to bum themselves out, and also that the Forest Service has a policy of fighting all fires in National Forests. The story said, incorrectly, that Interior Secretary Hodel had ordered on July 21 that all fires be fought.

A September 14 New York Times editorial supported the National Park Service by stating that the fires were not a disaster, as Interior Secretary Hodel had said they were, but helped perpetuate the myth that natural ecosystems arc static rather than dynamic, and supported the notion that it might have been possible to preserve Yellowstone forever as it was before the tires. "Yetlowstonc may take years," the editorial said, "to grow back exactly as it was" (emphasis added).

The first New York Times story about scientific aspects of [he 1988 wildfires (Malcolm 1988) was thoughtful and thorough, although it was not published until the end of September when the tires were largely under control. It contained interviews with Yellowstone research biologist Don Despain, with Cornell soil biologist Susan Riha, with tire-behavior expert Richard Rothermel, and with wildfire historian Stephen Pyne.

Coverage in the Washington Post

Stories in the Washington Post tended to be less judgmental than those in the New York Times, and tended to contain fewer factual errors. The first non-wire story (O'Gara 1988a) described fire as a positive influence on the forest, although it also helped establish the myth that Old Faithful was threatened by a "natural bum" fire when it attributed the human-ignited North Fork fire (the only one that ever threatened the Old Faithful tourist complex) to lightning. The second non-wire story (O'Gara 1988b) contained a reasonably good description of the natural-bum philosophy that later became controversial. The Post interviewed tire experts Don Despain and Richard Rothermel two months earlier than the New York Times (O'Gara 1988c). Unlike the New York Times and the three television networks, the Post specifically pointed out that the North Fork fire, which made the September 7 run on Old Faithful, and which caused all but one of the major evacuations in the park, was never subject to the natural-bum policy (Reid and Peterson 1988).

Coverage in the Los Angeles Times

The east-coast newspapers framed the fires as being more controversial than the Los Angeles Times. Although the New York Times mentioned controversy about Yellowstone's natural-bum policy on August 10, and the Washington Post first ran a story describing the controversy on August 9, the Los Angeles Times did not allude to any controversy about Yellowstone's natural-burn policy until September 1, and then only in an editorial endorsing the wisdom of that policy.

"Most of the complaints," the editorial said, "have come from a handful of landowners who have felt threatened by the raging fires and from business owners on the periphery of Ycllowstone who have suffered economic losses because of the fall-off of tourism." This frames the fires quite differently from the September 22 New York Times story that said the fires had led to unspecified but "widespread" criticism of the government's natural-bum policy (Egan 1988).

The Los Angeles Times carried a second editorial on September 13 that said the "unwarranted criticism of the Park Service, the U.S. Forest Service and environmental experts has reached a level of misinformed hysteria that is racing out of control, as the fires have done." This was followed by two op-ed columns supporting the scientific validity of the natural-bum policy, published on September 17 and September 26. On September 22, the Times carried an article that suggested officials were overreacting when they canceled a planned prescribed burn in the Santa Monica Mountains because of negative publicity about the Yellowstone fires (Fuentcs 1988).

Stories in the Los Angeles Times were presented in a way that interpreted the Yellowstone fires as more natural and less alarming than stories in the eastern elite newspapers. Yellowstone-area residents described in the New York Times and Washington Post tended to be critics of Yellowstone's tire management efforts. One of the very few local residents described in the Los Angeles Times, a merchant whose business was given a 25-percent chance of surviving one of the fires, was framed more positively. Ralph Glidden was quoted as saying "I'm trusting the professionals involved in this will do what they can do" (Los Angeles Times 1988).

The Television Networks

Like the New York Times, and perhaps following the example set by the Times, the three television networks continued to suggest that tires were being allowed to bum in Yellowstone long after that policy had been abandoned. The last such story on ABC was broadcast on August 25. NBC implied on September 6 that fires were still being allowed to burn, and CBS did so on September 7. The biggest difference in how the three networks framed the story was the differing ways in which they selected interviews with local residents and tourists. CBS and NBC focused on tourists and residents who were critics of Yellowstone's fiie management policy, but ABC did not carry a single critical comment on park policy by a local resident or tourist.

NBC and CBS lent credibility to the Bambi myth of animals fleeing from the tires; ABC did not. CBS, for example, implied large-scale fire-induced migration in a September 7 story that said some Yellowstone animals had been spotted 50 miles from their normal range. NBC twice focused on pictures of animals that appeared either to bc fleeing the flames (September 8) or to be confused by the thick smoke (August 25). ABC specifically said that moose didn't seem to notice the tires (August 25), and showed elk calmly grazing at Mammoth Hot Springs on September 9 as evacuation loomed.

The Yellow&one-Area Newspapers

Of the three daily newspapers in the Yellowstone area, one (the Casper, WY Star Tribune) circulates primarily outside the direct economic influence of Yellowstone Park. The other two (the Billings, MT Gazette and the Bozeman, MT Daily Chronicle), circulate heavily within the area directly affected by the Yellowstone tourist trade. Perhaps for that reason, the Casper Star Tribune carried virtually no stories about the effects of the Yellowstone fires on area businesses, while the Gazette and Chronicle carried many such articles.

The Star Tribune framed the tires as more natural and less disruptive than either of the Montana newspapers, and carried several stories and a column about the ecological benefits of the fires. The Bozeman Daily Chronicle adopted a relatively calm tone in describing the fires, but virtually ignored the scientific perspective about fire's biological role. The Billings Gazette carried far more stories about the fires than either of the other papers, and published many thoughtful and well-reported articles, especially those reported by Robert Ekey. But the Gazette also published many letters containing sharp attacks against the National Park Service and against specific officials of Yellowstone National Park, and published an editorial cartoon that ridiculed Yellowstone superintendent Robert Barbee. An August 29 Gazette editorial said "This fiasco is riddled with questions, and it's not too late for Congress to demand to know why Barbee blindly rode a dead policy into hell." A September 11 editorial called for the tiring not only of Superintendent Barbee, but also of the Director of the National Park Service and the Secretary of the Interior.

Coverage of the Natural Burn Policy

Because virtually all of the controversy that made the Yellowstone fires newsworthy centered around the policy that initially allowed many lightning-ignited fires to bum monitored but unsuppressed, it would have been reasonable to expect detailed articles about that policy's origins. Although most news organizations paid lip service to explaining the policy by explaining the role of fire in "cleansing" or "renewing" the forest, I was unable to locate a single article on the news pages of any of the newspapers published in 1988, or a single story in any of the evening network newscasts broadcast in 1988, that specifically mentioned the Leopold Report (Leopold 1963) that formed the philosophical foundation for the prescribed natural-fire policy. The Leopold Report was mentioned only once in all of the stories, in a December 11 New York Times Magazine article by Peter Matthiessen.

DISCUSSION

The New York Times and two of the three television networks lent considerable credence to the interpretation of the Yellowstone fires favored by local merchants and their elected representatives (including senators Wallop and Simpson of Wyoming and Baucus of Montana). This interpretation suggested that the National Park Service handled the fires ineptly. This reinforces findings by Molotch and Lester (1974, 1975), who predicted that business interests and federal officials would have more power to define the context in which the news media interpreted the tires than would environmentalists, scientists, or Yellowstone officials. However, the Washington Post and ABC television news framed the fires more neutrally, and the Los Angeles Times interpreted them as natural and as a somewhat positive event.

Myths About the Fires

For the purposes of this paper, there are two kinds of myths that help us define and explain features of the external world about which we have insufficient or incorrect knowledge. The first sort of myth is usually based on inadequate or inaccurate information, such as the idea inspired by the Disney film Bambi that animals flee in terror from forest fires. The second form of myth rises out of our effort to understand events that contradict cultural assumptions. For example, often assumed is that modem technology can extinguish forest tires. If the fires in Yellowstone are still burning, the reasoning goes, there must be some kind of conspiracy to mislead the public about tire suppression efforts. This myth probably gained credibility because of the initial policy not to suppress some of **the** naturally ignited fires.

The news media helped foster several myths about the Yellowstone tires. The most widely disseminated myth was that many of the fires were allowed to bum unsuppressed throughout August and into September. The New York Times and the three television networks also helped spread the myth that the most newsworthy of the tires, which was apparently started by a woodcutter's cigarette, spread because of the park's natural-bum policy. The North Fork tire was fought with available resources from the day it started.

By quoting park critics and tourists who lamented the fire-induced changes in Yellowstone ('*it won't be the same for a hundred years"), many media accounts supported the idea that Yellowstone is a static rather than dynamic ecosystem, and that it could be managed like a city park in which burned trees can be replaced by planting new ones, and in which elk can escape mortality if only they are provided with enough supplemental food. To a large degree, reporters failed to understand (or at least to communicate) the dynamic forces that shaped the way Yellowstone looked bcforc the 1988 tires.

Another myth, which has deep roots in the technological orientation of our culture, persisted despite minimal support from the media. This myth, that humans have the technology to control all wildfires, was regularly debunked by news accounts quoting firefighters and other officials who said only a change in the weather would put the fires out. This myth flourished in spite of the media.

The mythological way the media interpreted the fires is apparent in the fact that Old Faithful geyser was featured in about a quarter of all the stories in the elite press and on national television newscasts, despite the fact that only a small fraction of those stories dealt with the single day on which a fire actually made a run on the geyser. Other prominent Yellowstone features, such as Mammoth Hot Springs, Yellowstone Lake, and Ycllowstone Falls, were seldom mentioned. A person not familiar with the park could easily have gotten the impression that Old Faithful Geyser was the only real attraction in the park, and that virtually all of the Yetlowstone firefighting efforts in 1988 were part of a massive effort to save the geyser from destruction.

News as a Curriculum

Media scholar James W. Carey (1986) believes it is "unforgivably self-righteous" to criticize daily news accounts because they often fail to put news events into a perspective that explains how they happened and what they mean. He says news is a curriculum, and that it is unfair to expect the initial reports of any *event* to provide complete information about what happened. Considering the short deadlines under which daily journalists must operate, this perspective has some merit. But it does not explain why some interpretations of events are more likely than others, and does not explain why a major newspaper like the New York Times consistently failed to report that all Yellowstone fire were being fought.

All of the media organizations studied here published or broadcast thoughtful reports and analyses of the Yellowstone fires after they were brought under control in 1988, and all of the organizations continued to follow the story in 1989. Although these analyses were less prominently displayed than the initial dramatic stories about the tire's various runs, the persistent media consumer was eventually able to get a balanced picture of the fires, especially if she or he supplemented ordinary news sources with specialized magazines such as Audubon and Smithsonian. Media consumers without that kind of dedication, however, were likely to be misled by the high visibility of the stories that characterizcd the initial coverage. The panels of experts who evaluated all of the 1988 evening television stories about the Yellowstone fires rated the stories during the peak coverage period, when the fires got top-of-the-show coverage, as significantly less accurate than the stories that appeared earlier or later (two-tail t-test, p (0.001).

The lesson here is that the initial news coverage of any unanticipated natural event, such as the 1988 Yellowstone fires, is likely to contain many flaws. It may be unrealistic and even uncharitable to expect journalists to do a better job, but as long as the public has confidence in the news media, these shortcomings will continue to mislead newspaper readers and television viewers. These misinformed media consumers may support land-management decisions that are based on interpretations of events provided by special interests rather than on scientific research or long-term management goals.

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THE EVOLUTION OF NATIONAL PARK SERVICE FIRE POLICY

Jan W. van Wagtendonk[1]

Abstract—National Park Scrvicc policies concerning tire have changed over the years from no policy at all in the early ycars, through years of absolute fire suppression, to a period of experimentation and rcfinemcnt with a full spcctrum of intcgratcd fire managemcnt strategies. During much of this time, the Service was influcnccd by othcr agcncics and organizations but is now cmerging as a leader in the fire community.

Firc policics in thc National Parks have cvolvcd from no managcmcnt at all, through the full supprcssion of all lircs, to thc sophisticated application of scicntifically based firc managcmcnt stratcgics. Whcn Yoscmitc was sct asidc as a Statc rcscrvc in 1864 and Ycllowstonc as a national Park in 1872, thcrc wcrc no efforts to control lircs. An era of full firc supprcssion bcgan whcn managcmcnt of Ycllowstone passcd to thc U.S. Army in 1886 and to thc National Park Scrvicc in 19 16. Expcrimcntal prcscribcd burning was first conducted in Evcrgladcs National Park in 19.51. The Leopold Rcport (1963) influcnccd thc Park Scrvicc to rccvaluatc its firc policics. Revisions to the policics complctcd in 1968 pcrmittcd thc usc of firc as a management tool and led to the crcation of the first wildcrncss firc manngcmcnt Program, in Sequoia and Kings Canyon National Parks. To data, more than 2,000 lightning fircs have bccn allowcd to bum under carcfully monitorcd conditions in 46 Parks, and more than 1,000 prcscribcd bums havc bccn sct in 58 parks to mcct managcmcnt objcctivcs. Thc Ycllowstonc fircs in 1988 led to an cxamination of Scrvicc firc policy which affirmed current policy but rccommcndcd rcfincmcnts in implcmcntation.

THE ERA OF FIRE SUPPRESSION

In 1863, Prcsidcnt Lincoln sct asidc Yoscmitc Valley and the Mariposa Grove of sequoias as a Statc rcscrvc. This was the first fcdcral govcrnmcnt action spccifically designating an area for prcscrvation and is considcrcd by many to mark the beginning of the national Park idea. Although the native Amcricans who occupicd the Yoscmitc region had at least 4,000 ycars (Riley 1987) used firc for many cultural purposcs, it is doubtful that thcy practiced any fire supprcssion. Early Euro-American scttlcrs in the Yosemite rcgion uscd firc to clcar land and to improve grazing for shccp and cattlc. Their only firc suppression efforts were dirccted toward protecting structures. The State reserve cmploycd only onc guardian, who had little timc to fight lircs.

Ycllowstonc and Yoscmitc wcrc dcsignatcd as national Parks in 1872 and 1890.Howcvcr, no agcncy was assigned responsibility for thcir administration and their new status did not rcsult in thc implcmcntation of fire management. Although thcrc no firc managcmcnt policies or activitics during thcsc carly ycars, thc stage was sct for the beginnings of fire suppression.

The Army Years

Thc Unitcd States Army was assigned the responsibility for managing Ycllowstone in 1886 and Yoscmitc and Sequoia in 1891. Thc Policy of suppressing all fires bcgan in Ycllowstonc in 1886 (Agcc 1974) and was soon followed by similar policics in the other two parks. The Army built cxtcnsivc trail systcms to facilitate patrolling the new parks for shccp and timber trespass and for wildfires. As new parks wcrc cstablishcd, thc Army assumed control and dispatched the troops to extinguish all tires. Although there arc fcw records of the Army's efforts, fire scats were formed lcss frcqucntly during this period (Kilgorc and Taylor 1979). This could be intcrprctcd to mcan cithcr that there wcrc very fcw lircs or that the Army was very successful in extinguishing those that did occur.

The Years of Forest Service Influence

Whcn the National Park Service was cstablishcd in the U. S. Dcpartmcnt of the Interior in 1916, administration of the Parks passed into civilian hands. Many of the personnel who had prcviously scrvcd in the Army switched uniforms and bccamc thc first park rangers. Although thcy carried with thcm the lcssons and cxpcricncc of fire suppression, they had little formal training. Professional guidance of the fire program camc from the Forcst Scrvicc in the U. S. Dcpartmcnt of Agriculture (Pyne 1982). Established as a scparatc agcncy in 1901, the Forest Service had developed both a thcorctical basis for systematic fire protection and considcrablc cxpcrtisc in executing that theory. The supprcssion of all lircs became the official policy of the new National Park Service.

Since many of the Parks established during this period were originally parts of national forests, the Park Service inherited an infrastructure of firc control facilities and equipment. Fire stations, lookouts, and trails wcrc already in place. In addition, many of the new Park rangers came from the Forest Scrvicc and had forestry and fire backgrounds (Pyne 1982). The Forcst Service and thc Park Service joined together to form the Forest Protection Board, which advised agencies on firc policy and standards.

Although the Park Service dcvcloped a separate fire control organization, it relied heavily on thc Forest Service for cxpcrtisc, personnel, and equipment. Mutual-aid agreements allowed the two agencies to respond to fircs across boundaries

[1]Rcscarch Scicntist, National Park Scrvicc, Yosemite National Park, El Portal, CA.

and to share training and dispatching facilities. In most cases, however, the exchange was in the direction of the fledgling Park Service.

The CCC Yea-s

Professional fire protection began in the Park Service with the establishment of the Civilian Conservation Corps in 1933. A massive influx of personnel made it possible to expand firefighting facilities and deploy suppression forces throughout the parks. During the first 10 years, the program went from a single national fire officer, a special crew at Glacier National Park, and a fire guard at Sequoia to an organization of some 6.50 camps with over 7,000 employees (Pyne 1982).

The Park Service's fire policy was still identical with that of the Forest Service, which in 1935 adopted a policy of extinguishing any fire during the first burning period or, if that were not possible, by 10:00 a. m. the following day. Strict adherence to this policy required quick response time and numerous crews. Efforts were also directed toward developing better access to further reduce response times.

During this period, the Park Service greatly professionalized its approach to fire protection. Vegetation and fuel hazard maps were prepared from field surveys and response zones were delineated. Complete tire records were kept; each tire's cause and behavior were described, and the measures necessary to control each fire were detailed. These records did describe occasional large fires that might have exceeded the capabilities 0 f the suppression forces.

The War and Postwar Years

World War II caused a decline in fire protection throughout the nation. Skeleton crews were kept on to protect resources necessary for the war effort. Park Service: crews were practically nonexistent, although the fire records show that fires were still being suppressed successfully.

Demobilization after the war brought a new and different kind of influx to the fire lighting agencies. Although the Forest Service bad used bulldozers and smokejumpers before the war, airplanes, helicopters, tanks, and parachutes were products that the war had refined that were now available to fight the war against fire. Retardant drops, heliattack crows, bulldozers, and smokejumpers became the new tools of choice (USDA Forest Service 1960). The Park Service relied heavily on the Forest Service for this new technology, and shared support of aircraft and a smokejumper base at Yellowstone (Pyne 1982). The resulting fire-fighting force was very effective in continuing the policy of full fire suppression.

THE ERA OF FIRB MANAGEMENT

The effectiveness of fire protection was partly responsible for the beginnings of a shift in policy from fire control to lire management. As had long been recognized in the South, the absence of fire from an ecosystem that has evolved with fire can lead to unexpected, and often undesirable, results. Specifically, researchers found that periodic fires reduced accumulations of woody and brushy fuels and thinned thick understories of shade-tolerant species. Without tire, species composition shifted and fuel accumulations increased.

The Years of Revelation

Although the National Park Service's first experiments with the use of fire occurred in Everglades National Park in 1951 (Robertson 1962), impetus for a change in policy came later from outside researchers in California As early as 1959, Dr. Harold H. Biswell, of the University of California at Berkeley, advocated the use of prescribed fires to reduce the accumulation of debris underneath pondcrosa pine stands in the Sierra Nevada of California (Biswell 1959). His work was expanded upon by Dr. Richard Hartesvelt, from San Jose State University, who concluded that the greatest threat to the giant sequoia groves was not trampling by humans, but was catastrophic fire burning through understory thickets and unnaturally high accumulations of (Hartesvelt 1962).

In 1962, the Secretary of the Interior asked a committee to look into wildlife management concerns in the national parks. This committee, named after its chair, Dr. A. Starker Leopold, did not **confine** its report to wildlife, but took a broader ecological view that parks should be managed as ecosystems (Leopold and others 1963). They recommended that the biotic associations within a park be maintained or recreated as nearly as possible in the condition that prevailed when first visited by Euro-Americans. The report stated in an often quoted passage:

> When the forty-niners poured over the Sierra Nevada into California, those that kept diaries spoke almost to a man of the wide-spaced columns of mature trees that grew on the lower western slope in gigantic magnificence. The ground was a grass parkland, in springtime carpeted with wildflowers. Deer and bears were abundant. Today much of the west slope is a dog-hair thicket of **young** pines, white fir, incense cedar, and mature brush a direct function of overprotection from natural ground fires. Within the four national parks . Lassen, Yosemite, Sequoia, and Kings Canyon the thickets are even more impenetrable than elsewhere. Not only is this accumulation of fuel dangerous to the giant sequoias and other mature trees but the animal life is meager, wildflowers are sparse, and to some at least the vegetation tangle is depressing, not uplifting. Is it possible that the primitive open forest could be restored, at least on 3 local scale'? And if so, how? (Leopold and others 1963)

It was not a coincidence that Dr. Leopold's office was just across the street from Dr. Biswell's office. In fact, these gentlemen often discussed the ecological ramifications of fire exclusion over lunch and during seminars. Nor is it surprising that their graduate students would pursue fire-related Ph.D. dissertation topics and become Park Service scientists (Kilgore 1968; van Wagtcndonk 1972; Agee 1973; Graber 1981). The intellectual atmosphere at Bcrkeley invited students to challenge conventional approaches and practices.

The Turning Point

Only in 1968, after several false starts was the Leopold Committee report incorporated into policy. First the Secretary of the Interior had to find out whether or not the report's findings were acceptable to the public. A department underling was sent to the meeting where the report was being presented and found it to be overwhelmingly supported The Park Service was then directed to incorporate the report into its management policies. The entire report was included as an appendix and the section on fire management revised to reflect the new thinking (USDI National Park Service 1968). For the first time since 1916, the Park Scrvicc viewed fire as a natural process rather than as a menace:

> The prcscnce or absence of natural fire within a given habitat is recognized as one of the ecological factors contributing to the perpetuation of plants and animals to that habitat.
>
> Fires in vcgetation resulting from natural causes are recognized as natural phenomena and may be allowed to run their course when such burning can be contained within predetermined fire management units and when such burning will contribute to the accomplishment of approved vegetation and/or wildland managcmcnt objectives.
>
> Prescribed burning to achieve approved vegetation and/or wildland objectives may be employed as a substitute for natural fire (USDI National Park Service 1968).

The Years of Experimentation

As is often the case with the National Park Service, a policy change led to cxpcrimcntation. A prescribed natural fire program was initiated in Sequoia and Kings Canyon National Parks in 1968 (Kilgore and Briggs 1972), as were concurrent research studies of prescribed bums (Kilgorc 1971; Parsons 1976). At Yosemite National Park a similar prescribed natural fire program was started in 1972 (van Wagtcndonk 1978), and research concentrated on refining tcchniyues for prcscribcd burning (van Wagtcndonk 1974; van Wagtcndonk and Botti 1983). Experimental bums were ignited in several parks, and Yellowstone and a few other parks established prescribed natural fire zones (Romme and Dcspain 1989).

The Years of' Policy Refinement

As cxpcricncc with both prcscribcd burning and prescribed natural fire programs increased, interim guidelines were issued. Rcscarch also continued to contribute to the growing body of knowledge on both fire ecology and fire usc.

Contrary to Pyne's (1982) assertion, the National Park Service was a leader in the development of prescribed natural fire techniques. Although National Park Service personnel coopcrated with Forest Service managers and researchers in the same field, they did not need to look to the Forest Service for leadership.

The first revision of the 1968 fire policy came out in 1978 when all management policies for the National Park Service were rewritten (USDI National Park Service 1978). The policy staled:

> Fire is a powerful phenomenon with the potential to drastically alter the vegetative cover of any park.
>
> The prcscnce or absence of natural fires within a given ecosystem is recognized as a potent factor stimulating, retarding or eliminating various components of the ecosystem. Most natural fires arc lightningcaused and arc recognized as natural phenomena which must be permitted to continue to influence the ecosystem if truly natural systems arc to be perpetuated.
>
> Management fires, including both prescribed natural fires and prescribed bums, arc those which contribute to the attainment of the management objectives of the park through execution of predetermined prescriptions defined in detail in the Fire Management Plan, a portion of the approved Natural Resources Management Plan.
>
> All fires not classed as management fires arc "wildfires" and will bc suppressed. (USDI National Park Service 1978)

The policy further described the conditions under which fire could be used and specified that any management fire would be suppressed if it posed a threat to human life, cultural resources, physical facilities, or thrcatcncd or endangered species or if it threatened lo escape from prcdctcrmincd zones, or to exceed the prescription.

The Forest Service was also revising its fire policy to embrace fire management rather than fire control (DeBruin 1974). In 1978 it abandoned the 10:00 a. m. policy in favor of a new one that encouraged the usc of fire by prescription. The Forest Scrvicc's policy was also preceded by experimentation and research.

Thus, after a period of 10 years, policies of both the National Park Scrvicc and the Forest Scrvicc recognized the ecological role of fire and provided for its use. Pyne (1982) states, "Guided by the dazzling philosophy of the Leopold Report, the Park Service had advanced a policy too far ahead of its knowledge and technical skills; the Forest Service, with expertise and information in abundance, lagged in policy." While not entirely correct, his statement does point out the distinctive and synergistic roles the two agencies play.

In 1986, the Wildland Fire Management Guideline (NPS-18) was issued. It outlined in detail the procedures and standards to be used to manage wildfires, prescribed natural fires, and prescribed burns (USDI National Park Service 1986). With regard to prescribed natural fires, the new guideline specified that the condition limits under which naturally Ignited fires would be permitted to bum must be clearly stated. In addition, the ultimate size and boundaries of the fires must be preplanned and stated. Parks were also required to monitor each fire and to assess each burning day whether or not the fire should he allowed to continue to burn unimpeded.

Although there were no apparent problems with the Park Service's fire policies, they were revised again in March of 1988 as part of a lo-year comprehensive review of the management policies (USDI National Park Service 1988). The new policy emphasizes management objectives and plans:

> Fire is a powerful phenomenon with the potential to drastically alter the vegetative cover of any park. Fire may contribute to or hinder the achievement of park objectives. Park fire management programs will be designed around resource management objectives and the various management zones of the park. Fire-related management objectives will bc clearly stated in a fire management plan, which is prepared for each park with vegetation capable of burning, to guide a fire management program that is responsive to park needs.
>
> All fires in parks are classified as either prescribed fires or wildfires. Prescribed fires include fires deliberately set by managers (prescribed bums) or tires of natural origins permitted to burn under prescribed conditions (prescribed natural fires) to achieve predetermined resource management objectives. To ensure that these objectives are met, each prescribed fire will he conducted according to a written prescription. All fires that do not meet the criteria for prescribed fires are wildfires and will he suppressed. (USDI National Park Service 1988)

THE POST-YELLOWSTONE ERA

The fires of the Greater Yellowstone Area during the summer of 1988 brought fire policies of the National Park Service and the Forest Service under close scrutiny. The Secretary of Agriculture and the Secretary of the Interior appointed an interagency fire management policy review team to investigate the adequacy of national policies and their application for fire management actions in national parks and wilderness and to recommend actions to address the problems experienced during the 1988 fire season. With regard to policy, the review team recommended that:

> Prescribed fire policies be reaffirmed and strengthened.
>
> Fire management plans be reviewed to assure that current policy requirements are met and expanded to include interagency planning, stronger prescriptions, and additional decision criteria. (USDA and USDI 19891

A moratorium was placed on all prescribed natural fire programs until the agencies had complied with the recommendations of the review team. Although the National Park Service policies were determined to be adequate, implementation guidelines and fire management plans were found to bc in need of revision.

A task force was convened to rewrite NPS-18, the fire management guideline. The guideline was completely rewritten and addressed all of the operational recommendations of the review team report (USDI National Park Service 1990). Specifically, it requires approved tire management plans, established contingency plans, quantified prescriptions, monitoring procedures, fire situation analyses, and daily certification by the line manager that resources are available to manage the fire within the prescription. In addition, the prescription must include at least one indicator of drought and at least one definition of the maximum prescribed extent of the fire.

All the existing fire management plans were reviewed by teams of fire specialists from throughout the Park Service for compliance with the review team report and for adequacy of environmental documentation and public participation. Plans were sent back to the parks for revision. To date, three fire management plans have been approved Prescribed natural fire programs will be in effect in 1990 for Yosemite, Voyagers, and Sequoia and Kings Canyon National Parks.

National Park Service fire policies have evolved in a pattern of leaps forward followed by experimentation and refinement. The decentralized nature of the agency allows it to take advantage of new philosophical ideas and translate them into policy. The experience and expertise within the Service assures that it will continue to play that role.

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POSTER SESSION

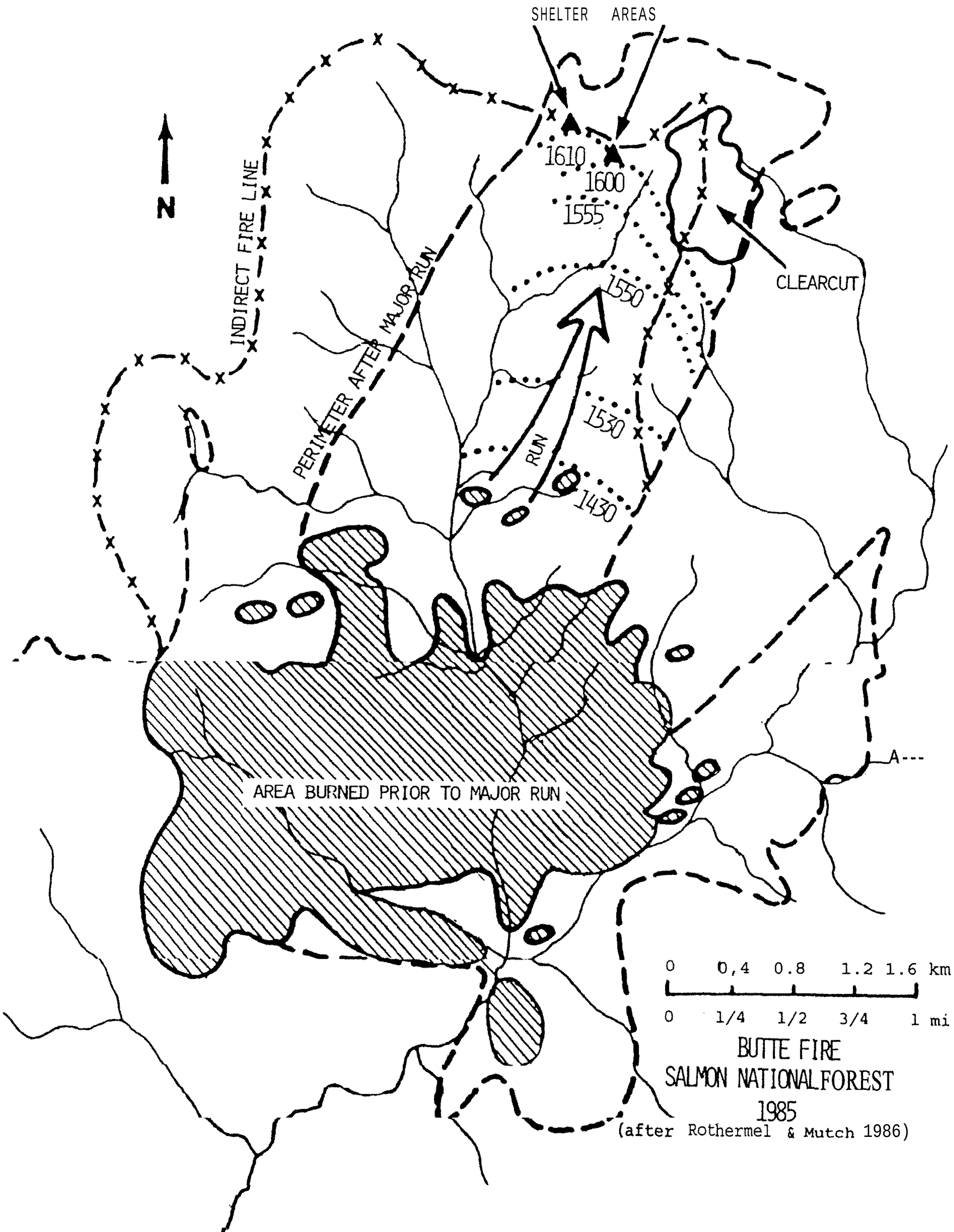

Figure I.--Fire progress map for the Butte Fire, Salmon National Forest, central Idaho, August 28-29, 1985 (after Rothermel and Mutch 1986).

THE 1985 BUTTE FIRE IN CENTRAL IDAHO: A CANADIAN PERSPECTIVE ON THE ASSOCIATED BURNINGCONDITIONS

Martin E. Alexander'

Abstract-During the afternoon of August 29, 1985, the Butte Fire made a high-intensity crown tire run, covering a distance of 2.22 km in one hour and 40 minutes, and forcing 73 tire fighters to deploy their protective tire shelters. This paper presents a retrospective analysis of the fire behavior in terms of the two major subsystems of the Canadian Forest Fire Danger Rating System. The fuel moisture codes (FFMC 94.6, DMC 172, DC 744) and tire behavior indexes (ISI 22.5, BUI 218, FWI 65) of the Canadian Forest Fire Weather Index System were indicative of extreme fire behavior and ignition potential. The predictions of headfire rate of spread (24.6 m/min or 1.48 km/h) and intensity (43,320 kW/m) based on the Canadian Forest Fire Behavior Prediction System were remarkably close to the observed tire behavior characteristics.

INTRODUCTION

The Butte Fire occurred on the Salmon National Forest toward the end of the 1985 fire season in central Idaho. Seventy-three fire fighters were forced to deploy their protective survival shelters in preestablished safety zones when the fire made a major run during the afternoon of August 39 (Mutch and Rothcrmel 1986). Fortunately, no one was seriously injured. As a result of this incident, the Butte Fire has attained a considerable degree of notoriety in the United States. Several published accounts of the fire's behavior (Aronovitch 1989; Mutch and Rothermel 1986; Rothcrmel 1991; Rothermel and Gorski 1987; Rothermel and Mutch 1986; Werth and Ochoa 1990) and the shelter dcployment (Jukkala and Putnam 1986; Turbak 1986) have already appeared, and there is a 33-minute videotape featuring interviews with those involved and photos taken during the episode (National Wildfire Coordinating Group 1989). Since a great deal of American information on wildland fire behavior finds its way into Canada, some means of relating it to Canadian conditions is oftcn desirable. Thus, this paper offers a hindsight analysis of the Butte Fire in terms of the two primary subsystems of the Canadian Forest Fire Danger Rating System (CFFDRS) (Stocks and others 1989). Emphasis is placed on the documentation of fuel moisture codes and fire behavior indexes of the Canadian Forest Fire Weather Index System and the quantitative prediction of forward rate of spread and frontal intensity based on the Canadian Forest Fire Behavior Prediction System. Some familiarity with the CFFDRS on the part of the reader is presumed.

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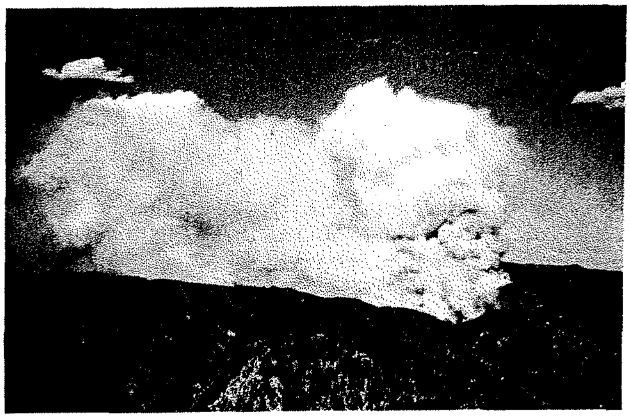
Table I.--Analysis of spread rates associated with the major run of the Butte Fire on August 29, 1985 (after Rothermel and Mutch 1986).

Time interval (hours MDT)	Elapsed time (min)	Forward spread distance (m)	Headfire Rate of Spread (m/min)	(km/h)
1430-1530	66	515	8.6	0.52
530-1550	20	772	38.6	2.32
550-1555	5	467	93.4	5.66
1555-1600	5	225	45.0	2.70
1600-1610	10	241	24.1	1.45
1430-1610	100	2,220	22.2	1.33

OBSERVED FIRE BEHAVIOR

The Butte Fire was started by lightning on July 20, 1985. By the afternoon of August 29 approximately 10,500 ha had been burned over and the northern perimeter of the fire was uncontaincd (fig. 1). On August 29 the Butte Fire made a forward advance of about 2,220 m between 1430 and 1610 hours Mountain Daylight Time (MDT) (table 1). This translates into an average headfire rate of spread (ROS) of 22.2 m/min or 1.33 km/h for the 100-minute run. For short time intervals, the main fire front travelled considerably faster. The maximum observed headfire ROS was 93.4 m/min or 5.6 km/h. The convection column associated with the major run (fig. 2a) was characterized by dense, black smoke and eventually reached an estimated height of nearly 5,000 m above the ground surface. The behavior exhibited by the Butte Fire on the afternoon of August 29 represents a classical case of a high-intensity crown fire event. Flames were observed to stand nearly vertical and greatly exceeded the height of the forest in which the tire was spreading (figs. 2b and 2c). A photograph taken from one of the shelter sites around 15.50 hours MDT, about 30 minutes before the arrival a



b

c

Figure 2.-- (a) convection column development over the Butte Fire at about 1520 hours MDT on August 29, 1985. (b & c) views of flame front approaching shelter areas (latter photo taken at about 1605 hours MDT). All photos reproduced from 35 mm slides taken by L. Duncan, USDA Forest Service.

of the active flame front, has been reproduced on the cover of Fire Management Notes Volume 46, issue Number 4 in 1986, and certainly attests to the severity of the tire behavior during the major run.

THE FIRE ENVIRONMENT

The fire environment is defined as "the surrounding conditions, influences, and modifying forces of topography, fuel and fire weather that determines tire behavior" (Merrill and Alexander 1987). The fire environment concept (Countryman 1972) as applied to the Butte Fire is described in the following sections.

Fuels

Forest cover types in the area of the major run consisted of Engelmann spruce (Picea engelmanii) - subalpine fir (Abies lasiocarpa) associations in the drainage bottoms and lodgepole pine (Pinus contorta) - subalpine fir stands at higher elevations (Steele and others 1981). The average tree height was considered to be about 18 m (Rothermel 1990) to 23 m (Patten 1990). Surface fuel loads (i.e., downed dead woody and forest floor materials) ranged from about 180 to 225 t/ha in the lower canyon slopes to about 55 to 90 t/ha in the midslope to upper slope areas (Rothermel and Mutch 1986). These figures appear quite reasonable in comparison to other areas in the Northern Rocky Mountains (Brown and Bevins 1986; Brown and See 1981; Fischer 1981). The forest floor depth varied from 2.5 to 10 cm (Patten 1990). Most of the tire area could be categorized by tire behavior fuel models 8 (closed timber litter) and 10 (timber-litter and understory) as described by Anderson (1982).

Topography

The general aspect of the fire area was southerly (fig. 2a). The tire swept up a well-defined north-south drainage that became progressively steeper near the shelter sites. Elevations from the start to the termination of the major run of the fire varied from 2,146 m to 2,341 m above mean sea level (MSL) (fig. 3). This vertical rise of 195 m coupled with the horizontal distance represents an average terrain slope of 9 percent (fig. 3). One unusual feature of the topography in the tire area was the dome-like nature of the upper slopes with continuous forest canopy cover.

Weather

In the spring of 1985, snow-free cover in the tire area probably occurred in early May (Finklin 1988, 1989). The area normally receives about 200 mm of precipitation between early May and late August according to data presented in Finklin (1988). A manually operated tire weather station at the Indianola Guard Station (GS), located 21 km east of the tire area at an elevation of 1,052 m MSL, reported 48.8 mm of precipitation in May but only 8.0 mm in June. The Skull Gulch remote automatic weather station (RAWS) (Warren and

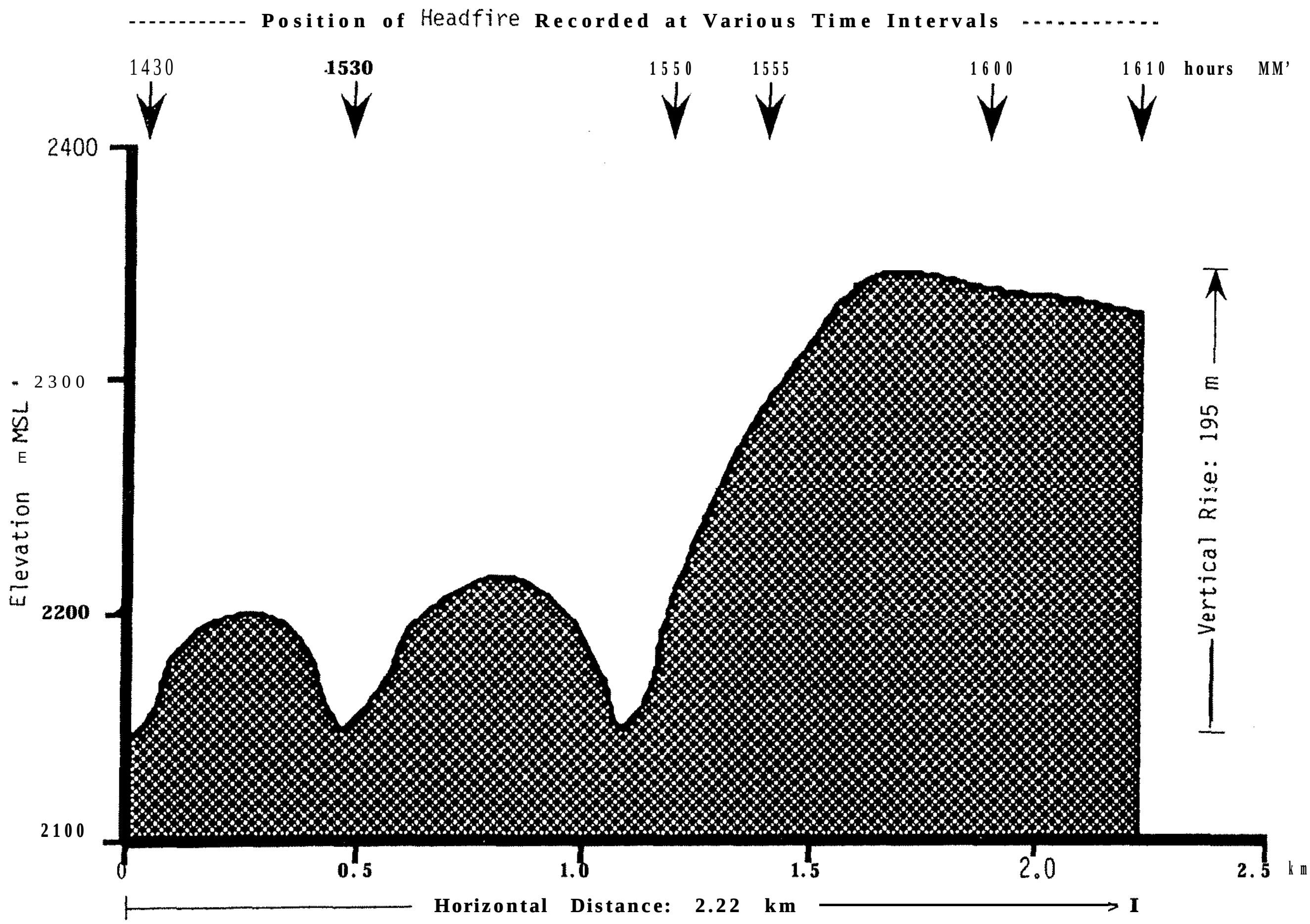


Figure 3.--Topographic profile for the area traversed by the major run of the Butte Fire on August 29, 1985. Note that the differences in the scales for the two axes (5.0:1) does accentuate the terrain relief.

Vance 1981), situated 14 km south of the fire area at 1,554 m MSL, recorded just 2.4 mm and 16.6 mm in July and August, whereas the Indianola GS measured 6.4 mm and 38.0 mm, respectively. The last significant rain (i.e., > 0.6 mm) received at the Skull Gulch RAWS prior to the major run of the Butte Fire occurred on August 2 and 3 (14.8 mm total). At Indianola GS, the last rainfall of any significance occurred on August 19 (3.0 mm).

A U.S. National Weather Service mobile fire weather unit was established at the base camp (2,256 m MSL), approximately 9.1 km southwest of the area involved in the fire run by the early afternoon of August 26. Weather observations taken prior to and on the day of the major run indicate relatively mild air temperatures with fairly low relative humidities but only moderately strong winds (table 2).

Table 2. --Dally 1399 hours MDT observations and extremes recorded at the temporary fire weather station at the Long Tao II Complex base camp (after Gorski 1999).

Calendar date (1985)	Dry-bulb temperature (°C)	Relative humidity (percent)	10-m open wind Speed[a] (km/h)	10-m open wind Direction (from)	24-h rain (mm)	Maximum temperature (°C)	Minimum relative humidity (percent)
August 26	21.7	27	14	south	0.0	23.9	21
August 27	22.8	23	13(31)	southwest	0.0	25.6	17
August 28	21.1	20	14	southwest	0.0	22.8	18
August 29	19.4	20	13(31)	southwest	0.0	23.9	17

[a]1-minute average. Reported gusts noted in parentheses.

Table 3.--Hourly observations recorded at the temporary fire weather station at the Long Tom II Complex base camp on August 29, 1965 (after Gorski 1990).

Local time (hours MDT)	Dry-bulb temperature (°C)	Relative humidity (percent)	10-P open wind Speed[a] (km/h)	Direction (from)
0800	12.2	49	3.7	east
0630	15.0	40	9.3	east
1000	16.1	38	5.6	east
1045	17.2	36	5.6	east
1120	18.9	32	0	
1290	21.1	29	13.0	southeast
1315	21.1	20	13.0(31)	southwest
1400	21.7	24	22.2(37)	south
1500	21.7	20	18.5	south
1535	22.2	19	23.1	south
1630	23.9	17	14.8	south
1720	23.3	19	15.7	sontb
1830	21.1	17	7.4	west
2030	16.7	31	11.1	sontb
2145	15.0	25	0	

*1-minute average. Reported gusts noted in parentheses.

According to the observations made on August 29 (table 3), the following weather conditions would have been rcprcscntative of the period during the fire run:

Dry-bulb temperature:	22°C
Relative humidity:	19 percent
1 O-m open wind:	20 km/h
Wind direction:	south
Days since rain:	26

Relative humidities as low as 6 percent and air temperatures of around 30°C were recorded at the Skull Gulch RAWS site during the major fire run (table 4); winds were generally southwest and averaged about 17 km/h.

Table 4.--Hourly fire weather observations recorded at tbe Skull Gulch RAW site on August 29, 1985.

Local time (hours MDT)	Dry-bulb temperature (°C)	Relative humidity (percent)	10-m open wind Speed[a] (km/h)	Direction (from)
0800	11.2	37	3.0	80
0900	13.9	37	0.7	206
1000	16.3	29	6.3	163
1100	19.8	24	10.0	149
1200	22.6	22	9.3	184
1300	25.7	18	10.7	145
1400	28.1	10	15.9	-[b]
1509	30.0	8	12.6	257
1666	30.4	6	23.0	281
1700	30.0	6	17.0	256
1899	29.6	6	19.8	292
1990	28.1	6	26.7	292
2060	25.3	7	18.0	300
2166	24.1	7	13.9	285
2266	21.8	9	0.2	196

[a] lo-minute average.
[b] Not available.

FIRE DANGER INDICES

The weather data used in calculations of CFFDRS values were obtained from the U.S. National Fire Weather Data Library (Furman and Brink 1975). These data were daily measurements of dry-bulb temperature, relative humidity, wind velocity, and 24-hour accumulated rainfall taken at 1300 hours MDT at Skull Gulch RAWS and Indianola GS during the 1985 fire season. Emphasis was placed on using the Skull Gulch readings since it was more representative of the fire area. The Indianola GS weather record was used for missing observations. Winds measured at the U.S. standard of 6.1 m or 20 ft in the open were adjusted to the CFFDRS standard of 10 m by means of the factor suggested by Turner and Lawson (1978). All weather data were converted to metric units. The standard components of the Canadian Forest Fire Weather Index (FWI) System (Van Wagner 1987) were calculated by computer program (Van Wagner and Pickett 1985). Calculations were begun on May 1, using the standard fuel moisture code starting values (Canadian Forestry Service 1984), and continued until August 25. Calculations for the subsequent period were based on weather observations made at the base camp immediately adjacent to the fire area.

Table 5.--Fire danger indices calculated at 1300 hours MDT for the temporary fire weather station at the Long Tom II Complex base camp.

Calendar date (1985)	Fine Fuel Moisture Code (FFMC)[a]	Duff Moisture Code (DMC)[a]	Drought Code (DC)[a]	Initial Spread Index (ISI)	Buildup Index (BUI)	Fire Weather Index (FWI)
August 26	94.5	160	723	16.4	207	53
August 27	94.5	164	730	15.6	211	52
August 28	[illegible]	[illegible]	[illegible]	[illegible]	[illegible]	54
August 29	[illegible]	[illegible]	[illegible]	[illegible]	[illegible]	65

[a]The FFMC, DMC, and DC at Skull Gulch RAWS on August 25, 1985, were 97, 157, and 716, respectively.

The three fuel moisture codes and three fire behavior indexes comprising the FWI System arc listed in table 5. Readers should consult Canadian Forestry Service (1984) for definitions of the six components. The moisture code values are all indicative of very low moisture contents for the types of fuels they are designed to represent. For example, litter and duff rcprcsented by the Fine Fuel Moisture Code (FFMC) and Duff Moisture Code (DMC) would probably have been 6.1 and 25 percent (Van Wagner 1987). A Drought Code (DC) value of 500 is generally considered to be a critical threshold for deep-drying conditions in forest floor and mineral soil layers (Muraro 197.5; Muraro and Lawson 1970). The Initial Spread Index (ISI), Buildup Index (BUI), and the Fire Weather Index (FWI) component itself, representing fire spread rate, fuel available for combustion, and fire intensity are all suggestive of extreme suppression difficulty (Alexander and De Groot 1988; British Columbia Ministry of Forests 1983; Muraro 1975). In most regions of Canada, an FWI value of greater than 30 would be rated as an extreme fire danger class (Stocks and others 1989). The fire danger ratings which prevailed on August 29 would also be considered extreme according to the criteria used by the British Columbia Ministry of Forests (1983).

PREDICTED FIRE BEHAVIOR

In an earlier paper by the author (McAlpine and Alexander 1988), a hindsight prediction of the August 29 run of the Butte Fire was made using the interim edition of the Canadian Forest Fire Behavior Prediction (FBP) System (Alexander and others 1984; Lawson and others 1985). This prediction was based on the ISI calculated for the Skull Gulch RAWS and the terrain slope. A crown fire spreading at 24.1 m/min was predicted. In the present paper, the basis for predicted tire behavior is a draft version of the first complete edition of the FBP System (Van Wagner 1989)[2]. The following predictions are based on the most representative FBP System fuel type (C-3: mature jack or lodgepole pine), a 9-percent slope, 10-m open wind of 20 km/h, FFMC 94.6, and BUI 219 for a line-source ignition spreading directly upslope with the prevailing wind during the late afternoon over a period of one hour and 40 minutes:

Headlire rate of spread:	24.6 mlmin or 1.48 km/h
Forward spread distance:	2,460 m
Fuel consumption:	58.7 t/ha (surface and crown)
Headlire intensity:	43,320 kW/m
Type of fire:	continuous crowning (> 99 percent crown fuel involvement)

In the above calculations, a foliar moisture content (FMC) of 105% (oven-dry weight basis) for lodgepole pine was used based on an on-site sample taken two days after the fire run by Rothermel (1990), and not the computational scheme for determining FMC according to calendar date, geographical location (i.e., latitude and longitude), and elevation. Fire intensity was computed, assuming a low heat of combustion value (reduced for fuel moisture) of 18,000 kJ/kg, on the basis of the predicted rate of advance and amount of fuel consumed (Alexander 1982).

DISCUSSION

The major runs of other well-documented wildfires in the United States such as the 1967 Sundance Fire in northern Idaho, the 1980 Mack Lake Fire in northern lower Michigan, the 1980 Lily Lake Fire in northeastern Utah, the 1983 Rosie Creek Fire in south-central Alaska, and the 1989 Black Tiger Fire in north-central Colorado, all occurred like the Butte

[2]To appear in final published form, authored by the Forestry Canada (ForCan) Fire Danger Group, as an Information Report issued by ForCan headquarters entitled "Development and Structure of the Canadian Forest Fire Behavior Prediction System" in 1991.

Table 6.--Canadian fire danger indices associated with five other major well-known U.S. wildfires.

Name of wildfire	Fine Fuel Moisture Code (FFMC)[a]	Daff Moisture code (DMC)[a]	Drought code (DC)"	Initial Spread Index (ISI)	Buildup Index (BUI)	Fire Weather Index (FWI)
Sundance"	96.1	318	752	23.8	318	68
Mack Lake[b]	94.6	35	59	43.2	35	50
Lily Lake"	95.0	66	107	562	66	77
Rosie Creek[d]	92.7	114	299	18.0	114	49
Black Tiger"	95.2	111	269	24.5	111	59

' Reference: Anderson (1968). The FWI System components were calculated on the basis of the 1606 hours Pacific Daylight Time (PDT) fire weather observations for the Priest Lake Experimental Forest, ID (winds were converted to 10-m open standard according to T-r and Lawson 1978); data were kindly provided by A.I. Finklin, Research Meteorologist (retired), USDA Forest Service! Intermountain Fire Sciences laboratory, Missoula, MT. Dry-bulb temperature and relative humidity were adjusted based on data presented in Fiaklia (1983), by (-) 1 .4°C and (+) 5.0 percent, respectively. in order to approximate the 1300 PDT values. The 1399 bows PDT fire weather observations on September 1, 1967, were: dry-bulb temperature 30.8°C; relative bamidity 18%; 10-m open wind 17 km/h; and 25 days since > 0.6 mm of rain.

'Reference: Simard and others (1983). The FWI System components were calculated on tbe basis of the 1300 hours Eastern Daylight Time (EDT) weather data given in Table 1 of Simard and others (1983) for Mio, MI (winds were converted to 10-a open staadard according to Tamer and Lawson 1978). The 1306 hours EDT fire weather observations on May 5, 1980, were: dry-bulb temperature 26.7°C; relative humidity 24%; 10-a open wind 33 km/h; and 6 days since > 0.6 mm of rain.

"Reference: Rothermel (1983, 1991). The FWI System components ware calculated oa tbe basis of the 1300 hours MDT fire weather observations for the Bear River Guard Station, Wasatch-Cache National Forest, UT (winds were converted to 10-m open staadard according to Tamer and Lawson 1978); these data are on file with the National Fire Weather Data Library (Furman and Brink 1975) under station number 420703. The 1300 boars MDT fire weather observations on June 23, 1930, mere: dry-bulb temperature 20.6°C; relative humidity 16%; 10-r open wind 37 km/h; and a minimum of 12 days since > 0.6 mm of rain.

[d]Reference: Juday (1985). The FWI System components were calculated on tbe basis of the 1300 hours Alaska Daylight Time (ADT) weather data for the international airport at Fairbanks, AK; data were kindly provided by P. Perkins, formerly with the USDI Bureau of Laad Management, Alaska Fire Service, Fairbanks, AIL The 1300 hours ADT fire weather observations on June 2, 1983, were: dry-bulb temperature 23.5°C; relative humidity 33%; 10-m open wind 21 km/h; and 4 days since > 0.6 mm of rain.

"Reference: National Fire Protection Association (1999). The FWI System components were calculated on the basis of the 1300 boars MDT weather data for tbe Betasso station operated by the Sherrif's Department, Boulder County, Co (winds were converted to lo-s opea staadard according to T- and Lawson 1978); these data are on file with the National Fire Weather Data Library (Furman and Brink 1975) under station number 056604. The 1300 boors MDT fire weather observations on July 9, 1989, were: dry-balb temperature 36.7°C; dative humidity 24%; 10-m open wind 20 km/h (based on an on-site estimate of add-flame wind speed): and 13 days since > 0.6 mm of rain.

Fire, under exceedingly severe burning conditions according to the FWI component (Table 6)'. Although extreme fire behavior was observed in all these cases, there were differences in forward spread rates and frontal intensities. This is to be fully expected given difference's in wind velocity, dead and live fuel moisture levels, fuel types (e.g., closed vs. open conifer stands), and the topographic situation (e.g., level ground vs. steep and complex mountainous terrain). The FWI System components offer a general indication of fire potential based largely on short- and long-term surface weather; presently there's no provision made in the FWI System to account for the influences of upper atmospheric conditions on tire behavior such as instability or low-level jet winds (Turner and Lawson 1978). It's the role of the FBP System, which depends in part on the FWI System for input, to consider the differences in fuel characteristics and topography on potential fire behavior.

The FBP System did, although admittedly in retrospect, a remarkably good job of estimating the actual fire behavior characteristics associated with the major run of the Butte Fire on August 29. Head fire ROS and forward spread distance are of course more easily verified in this particular case. A predicted spread rate approaching the maximum observed

'The major run of the 1971 Little Sioux Fire (Sando and Haines 1972) in northeastern Minnesota also occurred under an extreme fire danger rating; the FWI System componentsand 1200 hours local standard time tire weather observations are presented in Alexanderand Sando (1989).

value would be possible if a 45 percent slope were used (Van Wagner 1977b), as done by Rothcrmel and Mutch (1986), and if winds momentarily exceeded the mean value (Crosby and Chandler 1966). Patten (1990) has indicated that surface fuel consumption was perhaps 7.5 to 100 percent complete, and so the predicted value (47.3 t/ha) is quite reasonable given the fuel moisture levels and prcbum fuel loads. The predicted level of frontal fire intensity and type of tire are certainly indicative of a fully-developed crown fire (Alexander 1988; Alexander and Lanoville 1989; Anderson 1968; Simard and others 1983; Stocks 1987; Van Wagner 1977a) and certainly match the general impression obtained by viewing slides taken during the fire run.

CLOSING REMARKS

The CFFDRS can be, if properly applied, a useful tool in research and training for interpreting fire behavior information emanating from wildfire case histories completed in the United States. The present example has in fact been used since 1987 in the annual advanced fire behavior course held at the Forest Technology School in Hinton, Alberta. If the required weather observations are available, then it is possible to redescribe the burning conditions of previously documented forest fires in terms of the weather-dependent components of the FWI System, thereby permitting correlation with observed tire behavior and comparison with predicted fire behavior using the FBP System.

ACKNOWLEDGMENTS

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BEHAVIOR OF HEADFIRES AND BACKFIRES ON TALLGRASS PRAIRIE

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Abstract-Byram's fireline intensity model and time-temperature relationships at 0, 30, and 60 cm above the soil surface were used to characterize the behavior of headfires and backfires on tallgrass prairie during the spring. Weather and fuel parameters were used as independent variables in regression models of fire behavior. Byram's fireline intensity was greater in headfires than backfires. Residence time for backfires was not significantly different from residence time for headfires. Degree seconds for backfires was not significantly different from degree seconds for hcadfires. Maximum temperature at 60 cm was greater in headfires than backfires. Fire type (head fire or backfire) and variables related to fuel continuity, fuel loading, and fuel moisture were related to time-temperature relationships. Variables related to fuel load and weather were related to Byram's fireline intensity for headfires and backfires and for rate of spread for hcadfires.

INTRODUCTION

Data about the behavior of fire in the tallgrass prairie arc needed in order to increase our understanding of the interactions of fire behavior, fire environment, and fire cffccts. Previous studies of fire behavior have been confined primarily to wildfire in forests and shrublands and described mainly in terms of fireline intensity (Byram 1959; Wright and Bailey 1982). Because fire intensity is related to crown scorch of conifers (Van Wagner 1973) and because fireline intensity is used for describing wildfire behavior (Albini 1976), fireline intensity has been suggested for describing fire behavior in grasslands (Rothermc] 1972; Albini 1976) and for predicting scorch height on rangeland shrubs (Roberts and others 1988). Although fireline intensity accounts for the heat or energy released in the initial tire front, it does not account for energy released over the entire depth of the combustion zone (Tangrcn 1976; Alexander 1982). Combinations of fire temperatures and time-temperature relationships, rather than fireline intensity, have been used to quantify fire behavior in the residual combustion zone in tallgrass prairie fires (Engle and others 1989) and to relate fire behavior to tire effects on herbaccous vegetation (Stinson and Wright 1970; Wright 1971; Hobbs and Gimingham 1984; Ewing and Engle 1988; Bidwcl] and others 1990).

Research has not established whether grassland backfires or headfires produce higher maximum temperatures at the soil surface (McKell and others 1962; Daubenmirc 1968; Bailey and Anderson 1980). Because both fire types arc used in prescribed spring bums in the tallgrass prairie, it would enhance our knowledge of fire effects to elucidate their behavior. The fuel, topography, and weather in which a fire occurs dictates its behavior and may explain the contradictions regarding the behavior of headfires and backfires. Parameters of the fire environment that are easily measured by rangeland fire managers may also bc useful for predicting fire behavior in tallgrass prairie. Time-temperature relationships may be useful in describing differences between headfires and backfires by quantifying energy release in the entire combustion zone in tallgrass prairie fires. The objectives of our study were to compare the behavior of headfires and backfires and to explore the relationship between the behavior of tallgrass prairie tires and commonly measured variables of the fire environment.

STUDY AREA

Our study area is located on the Agronomy Research Range 15 km west-southwest of Stillwater, Oklahoma. Mean annual precipitation is 81 cm. The study area is a shallow prairie range site in the Central Rolling Red Prairies Land Resource Area (USDA Soil Conservation Service 1981). The soil is Grainola clay loam with a clay B horizon and is a member of the fine, mixed thermic family of Vertic Haplustalfs. Dominant grasses on the site include big bluestem (*Andropogon gerardii* Vitman), switchgrass (*Panicum virgatum* L.), indiangrass (*Sorghastrum nutans* (L.) Nash), and little bluestem (*Schizachyrium scoparium* (Michx.) Nash). The study area was grazed at a moderate to heavy stocking rate (2.4 AUM ha") from mid-July to mid-November in 1985 and 1986 before the treatments were applied in the spring of 1986 and the spring of 1987.

METHODS AND MATERIALS

We replicated treatments four times in a randomized complete block design. The 10 m by 20 m plots were located on nearly level terrain (< 2 percent slope), and oriented southeast to northwest so the direction of their long axes would correspond to the direction of the prevailing southeast spring wind. Each replication consisted of a headfired plot and a backfired plot. We used a drip torch to set the fires at plot borders. Burning treatments began in March and ended in April. Each replication was burned within a 4-hour burning period, and weather variables and fuel load were sampled

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immediately before each bum. Weather variables included ambient air temperature, relative humidity, and wind speed 2 m above the ground and were measured by means of a belt weather kit Fuel load was estimated by weighing clipped herbaceous material from five quadrats (0.5 m by 0.5 m) per plot. Clipped herbage was separated into standing tine fuel and fallen tine fuel (litter and mulch) before weighing. Fuel moisture, expressed on a dry weight basis, was determined after samples were oven dried at 70 °C for 72 hours.

We measured tire temperatures at 2 second intervals by means of high-temperature, chromel-alumel thermocouples at three stations per plot and at three heights relative to the soil surface (0 cm = soil surface; 30 cm = top of herbaceous canopy; 60 cm = above the herbaceous canopy). The thermocouple wire was 24 AWG with thermojunctions approximately 6 mm long and 1 mm in diameter and with 5- to 7-m leads overbraided with high temperature ceramic fiber insulation. An electronic data logger (Campbell Scientific model 21X with multiplexer) with tape data storage was used to record time-temperature data. Traces of time-temperature that were recorded for each thermocouple allowed estimates of degree seconds above ambient temperature (Potter and others 1983), maximum temperature, and residence time (the time from initial temperature rise to time of definite temperature drop) (Rothermel and Deeming 1980). A program in Turbo Pascal for IBM compatible microcomputers was used to generate each of these variables from the thermocouple data. A discrete summation algorithm was used to arrive at an estimate of degree seconds, which is the area above ambient temperature and under the time-temperature curve. The points of definite temperature rise and drop for computing residence time were determined numerically by sequential reverse progression through a IO-second interval of the time-temperature curve to points of 2 °C or greater departure from the postburn ambient temperature (Engle and others 1989).

Byram's (1959) fireline intensity model is expressed as I=Hwr, where I is fireline intensity, H is the fuel's low heat of combustion (LHOC)(kJ kg"), w is the weight of fuel consumed per unit area (kg m^{-2}), and r is the rate of spread (m s^{-1}). Low heat of combustion was determined by bomb calorimetry for the total fuel sample (standing and fallen). Fuel consumed was considered to be equal to fuel load because of the completeness of the bums. Rate of spread was reported in m min^{-1}. We measured rate of spread with a stopwatch and photographically with a 35 mm camera time-mode device in a manner similar to that employed by Britton and others (1977).

Approximate T tests were performed on tire behavior data (SAS, Inc. 1985). Differences between means of tire behavior variables were considered significant at the 1 0-percent probability level. Stcpwise multiple regression techniques were used to construct models of fire behavior (Table I) from the mean **values** of the environmental variables

Table 1. Fire behavior variables used in regression modets for relating fire environment to fire behavior on tat lgrass prairie

Variable	Code	Headfire (n=5) Min	Max	Mean	SE	Backfire (n=7) Min	Max	Mean	SE	P>F
Fireline intensity (Kw m^{-1})	BFI	75	2778	1167	445	31	146	97	16	0.02
Rate of spread (m min^{-1})	**ROS**	1	35	12	6	.5	2	1	.2	0.04
Degree seconds 0 cm (°C x s)	**DS0**	2504	**8260**	**5941**	**997**	1035	**31136**	9508	4001	0.48
Degree seconds **30 cm** ("C x s)	DS30	**993**	**10993**	**6096**	1727	2863	21523	10244	2731	0.27
Degree seconds 60 cm (°C x s)	**DS60**	551	6465	**4962**	1115	**198**	7767	4149	1249	**0.65**
Residence time 0 cm (s)	**RT0**	111	705	250	114	74	589	255	75	0.97
Residence time 30 **cm** (s)	**RT30**	**158**	553	337	a2	151	756	**465**	70	0.26
Residence **time** 60 cm (s)	**RT60**	92	779	**429**	139	116	749	487	77	0.70
Maximum temp. 0 **cm** (°C)	MT0	48	386	216	59	73	687	215	81	0.99
Maximum temp. 30 **cm** (°C)	**MT30**	67	537	307	a3	**108**	**634**	270	68	0.74
Maximum temp. **60 cm** (°C)	**MT60**	44	378	274	83	24	132	78	14	0.094

Table 2. Mean independent environmental variables from headfires (n=5) and backfires (n=7) used in multiple regression models

Variable	Code	Min	Max	Mean	SE
Relative humidity (%)	RH	18	51	34	2
Air temperature C"	TMP	15	26	21	1
Wind speed (km hr.')	WIND	3	24	10	1
Fuel load dry weight (kg ha")"	FLD	2372	5584	3570	260
Fuel load fresh weight (kg ha')	FLF	2720	6576	4707	298
Fuel moisture (standing) (%)	FMS	5	59	28	4
Fuel moisture (fallen) (%)	FMF	13	148	46	9
Fuel moisture (total) (%)	FMT	12	60	31	4
Quadrat fresh weight STD[b]	QFFS	18	56	42	3
Quadrat dry weight STD	QFDS	11	50	32	3
Quadrat fresh weight min.[c]	QFFMIN	15	122	69	8
Puadrat dry weight min.	QFDMIN	12	100	54	26
Quadrat fresh weight CV (%)[d]	CVF	12	74	36	17
Puadrat dry weight CV (%)	CVD	15	70	37	15

[a] All quadrat values (weight) in g $0.25m^{-2}$
[b] STD = standard deviation
[c] Minimum quadrat value within a plot
[d] CV = coefficient of variation

(Table 2) and tire type as a dummy variable (I for headfire and 2 for backfire) (SAS, Inc. 1985). Measures of variation associated with fuel load, standard deviation, and coefficient of variation for a plot, were included as measures of fuel continuity. Minimum quadrat sample values within a plot were included to provide an estimate of minimum fuel loading which, from observation, may affect fire spread over the fuel bed. Variation and minimum values of fuel loading were derived from five quadrats per plot. Five headlires and 7 backfires were used in the analysis because of wind shifts away from the longitudinal axis of the plots.

RESULTS

Byram's fireline intensity (BFI) of headlires averaged 1167 $\pm$ 445 kWm^{-1}, which was 12 times greater than fireline intensity of backfires (97 $\pm$ 16 Kw m^{-1}) ($P=0.03$). Rate of spread (ROS), the main influence on BFI, was 10 times greater for headfires (12.6 $\pm$ 6.0 m min^{-1}) than for backfires (1.0 $\pm$ 0.2 m min $^{-1}$) ($P=0.09$). Regression models explained more than 90 percent of the variation in BFI for both headlires and backfires and in ROS for headfires (Table 3). Degree

Table 3. Regression models relating environmental variables to Byram's fireline intensity (BFI) and rate of spread (ROS) on tallgrass prairie

Dependent Variable	b_0	b_1	X_1	b_2	X_2	R^2	P>F
Headfires							
BFI	-2274	0.39	FLF	24	FMF	0.94	0.0001
ROS	0.07	0.005	FMF	-0.604	RH	0.95	0.0001
Backfires							
BFI	497	-6	RH	-3	TMP	0.92	0. oool
ROS	0.04	-0.0006	FMS	-0.0001	QFDMIN	0.63	0.0001

seconds and residence time did not differ by fire type at any height (P > 0.10) (Fig. la, lc). Head fires produced greater maximum temperatures than backfires at 60 cm (P-0.004) (Fig. lb).

Parameters related to fuel load and fucl moisture were the most important variables entering regression models of time-tcmpcraturc relationships for all heights (Table 4). Fire type occurred as a first-entered variable in four of nine models. Environmental parameters and tire type explained at least 70 percent of the variation in each regression model (Table 4).

DISCUSSION

Behavior of Headfires and Backfires

Fireline intensity of headfires and backfires in our study were similar to those observed in other grassland fire studies (Englc and others 1989; Roberts and others 1988). The greatest fireline intensity we measured in a headfire, 2778 Kw m [1], was one-third as great as that observed in homogeneous grass stands in West Texas (Roberts and others 1988), but was comparable with the greatest fireline intensity in a summer head fire in a moderately grazed tallgrass prairie (Englc and others 1989). The magnitude of the difference between the ROS of our headfires and the ROS of our backfires is consistent with the rate of spread in two grassland communities in west Texas (Roberts and others 1988). Rate of spread and fuel consumption are the major variables in Byram's model of fireline intensity. Although fireline intensity and rate of spread were greater in headfires in our study, several time-temperature parameters did not differ between fire type because time-temperature profiles rcflect both the intensity (heat release rate) and heat transfer for the entire combustion period. Headfires are considered more intense than backfires because they consume fuel more rapidly and spread more rapidly than backfires (Lindcnmuth and Byram 1948; Trollope 1984). In contrast, the difference between degree seconds for headfires and degree seconds for backfires were much smaller. Degree seconds relate to the heat released over the entire combustion period, whereas fireline intensity represents only the rate of release of heat energy from the flaming front. Thus, the rate of release of heat energy is greater in headfires, but the two fire types release similar total amounts of heat energy.

Both fire types have been reported to be hotter above the herbaceous canopy (Fahnestock and Hare 1964; Bailey and Anderson 1980; Trollope 1984). Although mean maximum temperatures were not significantly different between headfires and backfires, they were highest at 30 cm in both backfires and headfires. Maximum temperature declined more from 30 cm to 60 cm in backfires. Maximum temperature above the herbaceous canopy is higher in head fires because the rate of energy release and convection are greater in headfires. Thus, differences between above

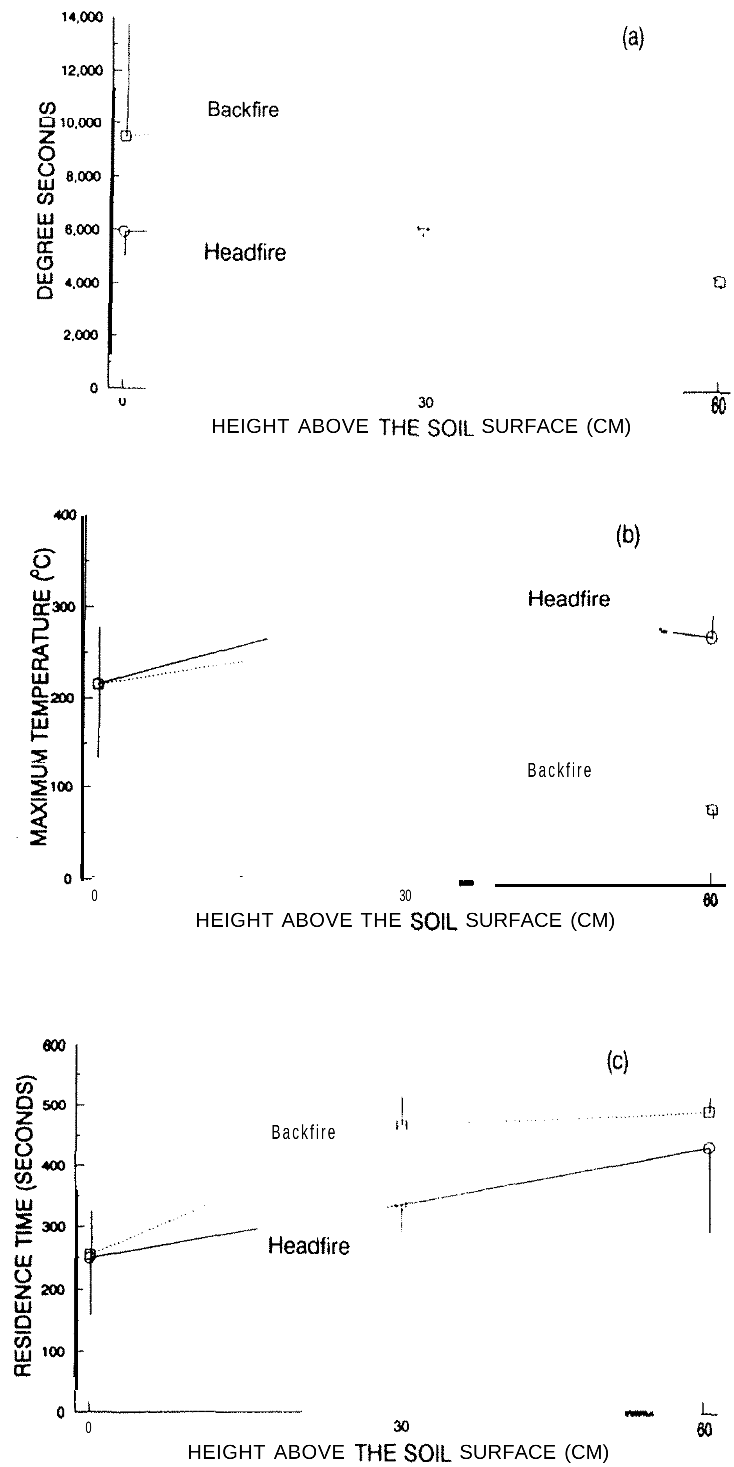

Figure 1. Time-temperature relationships in tallgrass prairie headfires and backfires for degree seconds (a), maximum tcmpcraturc (b), and residence time (c). Bars are plus or minus one standard error.

Table 4. Regression models relating fire type and environmental variables to time-temperature relationships of spring fires on tallgrass prairie'

	b_0	b_1	X_1	b_2	X_2	b_3	X_3	b_4	X_4	R^2	P>F
Degree seconds at 0 CM											
	23027	•462	FMT	1255	WIND	478	QFFS	•1049	QFDS	0.75	0.03
Degree seconds at 30 CM											
	27647	4561	TYPE	-2	FLD	-250	RH	25466	CVO	0.81	0.01
Degree seconds at 60 CM											
	17183	l	FLD	32	FMF	-212	RH	-140	TMP	0.88	0.02
Residence time at 0 CM											
	18	-1	FMF	48	WIND	2674	CVF	-3251	CVD	0.70	0.06
Residence time at 30 CM											
	2597	2.51	TYPE	•16	RR	-24	TMP	-1151	CVF	0.93	0.0004
Residence time at 60 CM											
	4048	212	TYPE	-27	RH	-31	TMP	•1362	CVF	0.89	0.002
Maximum temperature at 0 CM											
	1403	-0.19	FLD	-11	FMT	12	OFFS	•1798	CVD	0.79	0.02
Maximum temperature at 30 CM											
	1016	-0.08	FLF	2	FMF	-8	RH	-474	CVD	0.73	0.04
Maximum temperature at 60 CM											
	605	-206	TYPE	-9	RR	10	QFFS	-8	QFDS	0.87	0.003

'See Table 2 for descriptors

canopy maximum temperatures for prairie headfires and backfires reflect differences in rate of energy release much as did differences in fireline intensity between headfires and backfires.

Disagreement exists in the literature as to which tire type produces the hotter tire at the soil surface. McKell and others(1962) found that backfires produce higher temperatures at the soil surface than head fires do, but Daubenmire (1968) directly contradicts McKell and others Although maximum temperature was highly variable in both tire types we did not find that it differed significantly between fire types.

Fire temperature in the combustion zone is primarily dependent upon the quantity of fine fuel consumed (Stinson and Wright 1969; Engle and others 1989). Fine fuel load also has a pronounced effect on residence time, which increases proportionally to fuel load (Stinson and Wright 1969), especially with accumulation of mulch on infrequently burned tallgrass prairie (Engle and others 1989). The time required for active combustion was very nearly the same in both tire types in our study, and we would expect a difference in residence time measured by thermocouples placed on the soil surface only where there are differences in fuel loading or fuel consumption.

Regression Models

Fuel load and fuel moisture variables rather than weather variables were the tire environment parameters most strongly related to tire behavior. Fuel load was an important variable in our models. Fuel load accounted for 30 to 60% of the variation in fireline intensity in grassland fires in Africa (Trollope and Potgieter 1983). Our results were similar to those of other studies showing that fuel moisture affects ignition and combustion more than any other environmental factor (Byram 1957; Brown and Davis 1973).

Fuel continuity variables appeared in all but two models. Fuel continuity is a primary factor in tire behavior but is less important when heavy fuels arc available or wind speed is high (Brown and Davis 1973). Wind speed is an important influence on tire behavior including rate of spread (Rothermel 1972; Albini 1976), but fuel discontinuity may alter the influence of wind so much that mathematical tire models become poor approximations of tire behavior (Brown 1982). Mathematical models assume uniform fuel (Brown 1982), which seldom occurs in tallgrass prairie. Wind speed (< 24 km h^{-1}) was not an important variable in our tire behavior models (Table 3).

CONCLUSIONS

Fireline intensity and rate of spread, measures of 'fire behavior that relate to behavior of the flaming fire front and rate of energy release, indicate that headfires are at least ten times more intense than backfires in grazed tallgrass prairie. Time-temperature measures of fire behavior that account for energy released across the entire combustion zone indicate lesser differences between fire types. The behavior of backtires is more variable than headfires in discontinuous fuels.

In addition to tire type, fuel loading and fuel moisture were important variables in regression models of fire behavior. Fuel load largely determines the amount of energy available for combustion. Fuel continuity measures were important variables in regression models of fue behavior because they reflect the subtle fuelbed and microclimate differences associated with grazed fuel beds. Mosaics of discontinuous fuels or disturbed patches often result from spot grazing by large herbivores, soil disturbance by small mammals, soil heterogeneity, or natural spatial heterogeneity of tallgrass prairie vegetation (Loucks and others 1985). Our research indicates that fuel parameters together with fire type are major factors associated with variation in fire behavior. This should enable us to understand the role of fire as an disturbance factor in the tallgrass prairie plant community.

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FIRE AND AGRICULTURAL ORIGINS: PRELIMINARY INVESTIGATIONS

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Abstract—tested the hypothesis that hunter-gatherers used fire to manipulate wild cereal stands, in a process leading to agriculture. In Israel, wild emmer (*Triticum dicoccoides),* barley (*Hordeum spontaneum*), and oats (*Avena sterilis*) grow in mixed stands, so efficient harvest is difficult. Wild cereal grains are differentially protected against fire by hull thickness and efficiency of self-burial. Wild oat diaspores drill deeper into soil than wild barley and emmer, except next to rocks, where wild emmer drills best. Wild barley has thin hulls that protect against charring less than the thick hulls of wild emmer and oats. I predicted that wild barley is susceptible to tire, and wild oats more tolerant than wild emmer except around rocks. Distribution and survival of wild cereal grains after arson fire supported these predictions. Wild barley occurred almost exclusively in unburned spots. The percentage of charred wild oat grains was lower than that of wild emmer in open soil, but not adjacent to rocks. The results suggest that people could have set fires to reduce the importance of wild barley, and in rocky areas to favor wild emmer, but it is doubtful whether they would have desired to do so.

The Agricultural Revolution was one of the major watersheds in human history and cultural evolution, and led to unprecedented, ongoing transformations of global ecosystems. The causes of this change in human economy continue to be debated (Adams 1983; Binford 1968; Blumlcr and Byrne 1991; Byrne 1987; Cohen 1977; Flanncry 1965, 1969, 1973; Harlan 1975, 1986; Henry 1989; Reed 1977; Rindos 1984; Sauer 1952). Available cvidcncc suggests that environment was at least as important as cultural/technological state: early agricultural centers were characterized by semi-arid climates with a long dry season, which favored the evolution *of annual* grasses and legumes with unusally large seeds, and gcophytcs (Blumler 1984, 1987, 1991b; Byrne 1987); most if not all of the first crop plants were derived from such species (Blumlcr 1987, 199lb; Byrne 1987). The Fertile Crescent is an area of special interest since cultivation may have begun there first, by 10,000 B.P. (Hillman and others 1989; Zohary and Hopf 1988), or perhaps several thousand years earlier (Kislcv and Bar-Yosef 1988; Unger-Hamilton 1989). Some scholars have suggested that the initiation of farming in this region was connected to the USC of fire as a vegetation management tool (Lewis 1977,; Navch 1974, 1984; Pullar *1977).* They proposed that fire was employed to increase and maintain the abundance of wild cereal grasses, and may even have caused the evolution of these unique monstrosities (fig. 1) in the first place.

Ethnographic and historical evidence demonstratcs that hunter-gatherers often manipulate vegetation with fire (Burrows 1991; Day 1953; Hallam *197.5, 1989;* Lewis 1973; Reynolds 1959; Shipek 1989; Stewart 1956). Control of fire certainly predates the Epipalcolithic, and may be of great age in the Old World (Naveh and Dan 1973; Sauer 1956). Regular burning of vegetation is well-attested in early historical times in Israel, Greece, and Rome (Liacos 1973; Naveh 1974). Thus, it is reasonable to assume that Late

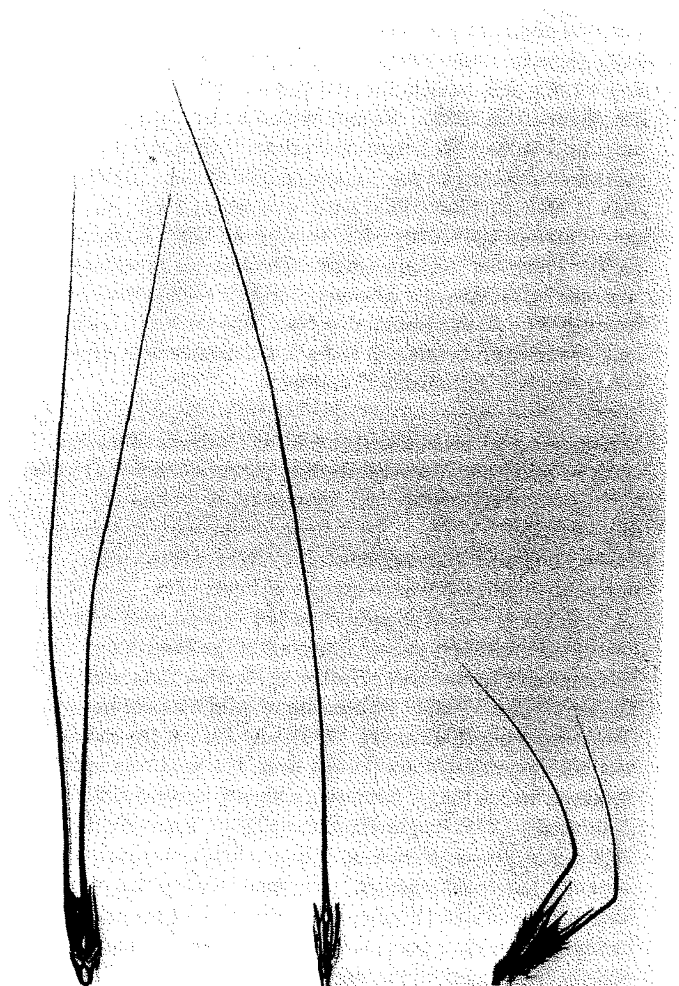

Figure 1.-Diaspores (from left to right) of wild emmer *(Triticum dicoccoides),* wild barley *(Hordeum spontaneum),* and wild oats (*Avena sterilis*). *(X* 2/3).

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Pleistocene peoples of the Near East would have set fires if the availability of wild cereals or other resources could have been enhanced by doing so. Noncthclcss, archaeologists and paleoecologists have been suspicious of the fire/domestication hypothesis, because archaeological/paleobotanical evidence offers no convincing support (Wright 1980). Furthermore, the study of Near Eastern fire ecology is in its infancy, and assertions about the response of wild cereals to tire have been pure speculation. The proponents of the fire/domestication hypothesis have generally assumed that the Fertile Crescent was naturally covered by dense woody vegetation, and that wild cereal stands could have formed only after the removal of this cover by disturbances such as fire. However, it is now well-established that steppic vegetation dominated the region during the Late Pleistocene, except possibly along the Levantine coast (Van Zeist 1969; Van Zeist and Bottema 1982), and that seedling establishment of woody species is extremely difficult on the fertile basaltic and terra rossa soils presently dominated by wild cereals (Berliner and others 1986; Blumler 1984, 1991b, c, d; Kaplan 1984; Litav and others 1963; Rabinovich-Vin and Orshan 1974; Rabinovitch-Vin 1983). Recent evidence also strongly suggests that the wild cereals arc primarily adapted to ungrazcd, undisturbed (i.e., unbumcd) conditions (Blumler 1984, 1991b, c; Litav 1965; Litav and others 1963; Naveh and Whittaker 1979; Noy-Meir and others 1989; Zohary 1969). Thus, the tire/domestication hypothesis in its original form can be dismissed _ although Naveh's (1984; Kutiel and Naveh 1987a, b) suggestion that Natufians on Mt. Carmcl used fire to open up maquis and allow establishment of wild cereal stands is reasonable if unproven.

In any case, the possibility that fire may have been used to favor wild cereals in certain specific circumstances, or to alter the mix of cereal grasses on a given site, is worthy of consideration. In Israel, for instance, wild emrner wheat, wild barley, and wild oats commonly form dense, mixed stands on good soils where grazing is light. The archaeological record suggests that wild emmer and wild barley were the first plants cultivated in the Near East (Blumler and Byrne 1991; Harlan 1975; Van Zeist and Bakker Heeres 1985).[2] Harvesting of wild oats seems to be time-consuming (Ladizinsky 1975), and the oat (*A. sativa*) was not domesticated until much later; but wild oats tend to dominate stands today, rendering efficient harvest difficult (Ladizinsky 1975; Unger-Hamilton 1989). Since the three wild cereal grasses mature at different times, they probably were not harvested together. Any manipulation that would have increased stand purity, especially of wild emmer or wild barley, should have been of benefit to the pre-agricultural peoples (Natufians) of the region.

While carrying out fieldwork in Israel during the 1987-8 rainy season, I undertook preliminary investigations into the effects of fire on the relative abundance of these three grasses. To this end, I considered diaspore characteristics that should affect fire tolerance, and tested my conclusions by measuring survival after fire. Ecological speculations arc rampant in the literature on agricultural origins, but experimcntal testing of ecological hypotheses is rare. This study was carried out in part to provide an illustration of the sort of research procedure that should prove fruitful in illuminating our presently rather sketchy understanding of the circumstances surrounding agricultural beginnings.

DIASPORE MORPHOLOGY AND FIRE

Several factors are likely to influence seed mortality during a bum. Grass fires arc typically fast-moving and not very hot, so seeds that are protected by a thick seed coat or other investing structures may escape with only minimal charring. During the passage of fire, a pronounced temperature gradient exists between the surface and deeper soil layers, so buried seeds should suffer much less mortality than those that lie on the surface or are caught up in the litter.

Wild cereal diaspore morphology is an adaptation that allows self-burial. I examined a number of sites early in the growing season, and found that depth of burial depends on microsite conditions and species (figs. 2 and 3).[3] Wild oats have hygroscopic awns, which cause burial, at approximately identical depth, under a wide range of conditions. Where there is no litter, wild emmer and barley tend to become oriented horizontally, which precludes self-burial except when diaspores are wedged up by rocks or soil cracks. Immediately adjacent to rock outcrops, wild emmer typically drills deeper than either wild barley or oats (in fact, it is occasionally found 15 or more centimeters beneath the surface).' Where there is abundant litter, wild barley diaspore rachis tips tend to weakly penetrate the soil surface, leaving the grain almost completely exposed, while wild emmer often remains suspended entirely above ground; nonetheless, seedling establishment of both species is excellent, presumably because litter shade reduces moisture stress. It may be no coincidence that wild emmer is found almost exclusively on hard limestone hills and basaltic plateaus (Harlan and Zohary 1966; Zohary 1969, pers. comm.), where rock cover is often so great that the seed rain from plants growing adjacent to rocks is spread throughout much of the open soil area, allowing rapid re-establishment away from rocks under favorable (i.e., high litter) conditions.

'Wild emmer is the progenitor of emmer (*T. dicoccum*), durum (*T. durum*), and most other tetraploid wheats, as well as bread wheat (*T. aestivum*); wild barley is the progenitor of all domesticated barleys (*H. vulgare*).

'Quantitative empirical data on drilling success in relation to microsite will be presented elsewhere (Blumler n.d.). Variability is great, of course; figures 2 and 3 are merely schematic diagrams illustrating the most characterisitc final positions of wild cereal diaepores under differing microsite conditions. Discussions of wild cereal self-burial in the literature (Aaronsohn 1910; Cook 1913; Zohary 1960, 1969; Zohary and Brick 1961) are generally in accord with my findings.

'In such cases, seedling emergence may be problematic

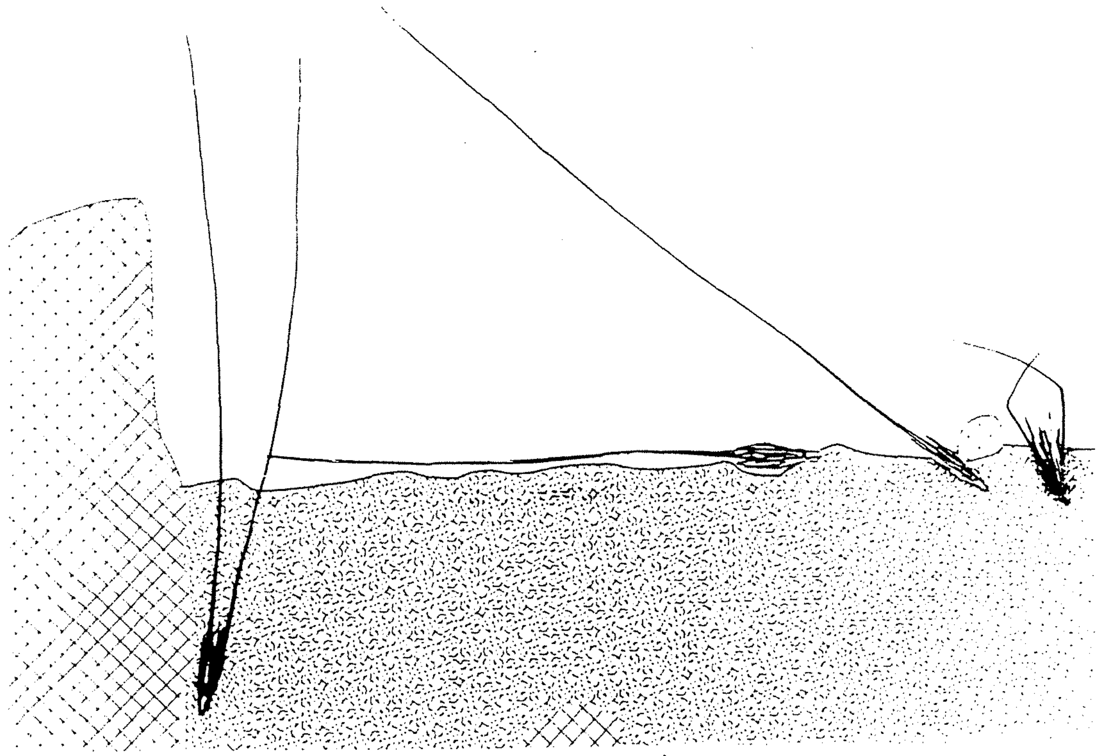

Figure Z.-Characteristic final position of wild cereal diasporcs in thc abscncc of plant litter (i.e., as a result of hcavy grazing or other disturbance). Wild emmcr and wild barley tend to lie prone, which prccludcs burial; whcn wedgcd upwards by rocks, however, wild emmcr, in particular, can drill very far bcncath the soil surface (lcft). Wild oats burics itself wcll, cven in open soil (right).

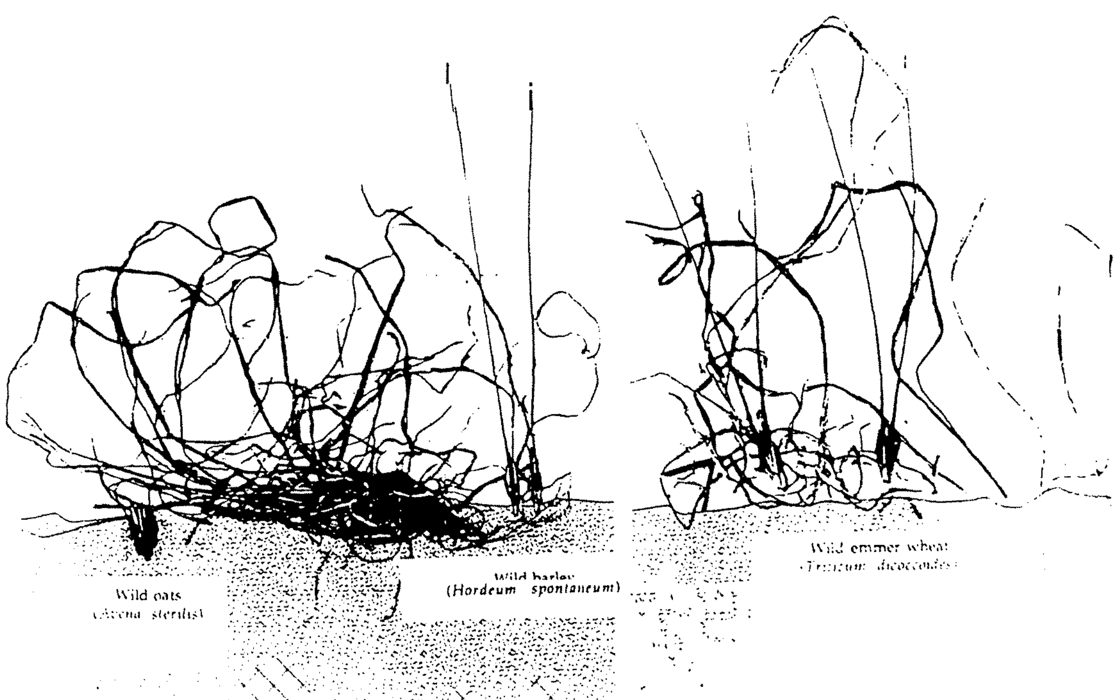

Figure 3.-Characteristic final position of wild cereal diasporcs whcrc thcre has bccn an accumulation of litter (under rclativcly undisturbed conditions). Wild oats is about as succcssful at self-burial as it is under no-litter conditions; wild barlcy barcly pcnctrates the soil surface with its rachis tip; while wild emmcr typically remains suspended just above the soil surface.

Table 1. Characteristics presumably influencing resistance of wild cereal diaspores to charring and heat caused by fire.

	Hull Thickness	Self-Burial: Open Soil	Self-Burial: By Rocks	Seed Dormancy
wild emmer	thick	very poor	best	less than 50 percent
wild barley	thin	poor	good	almost none
wild oats	thick	good	good	50 percent or more

Wild barley has relatively thin hulls, and no seed bank (table 1); hence, it is probably more vulnerable to fire than wild emmer or wild oats, which enclose their seeds in thick husks and have a large pool of dormant seeds that tend to become buried during the growing season, if not before.[1] Hence, I predicted that wild barley is susceptible to fire, and wild oats more tolerant than wild emmer except around rocks.

MORTALITY DURING FIRE

Distribution and survival of wild cereal grains after arson fire supported these predictions. The study site was a Cenomanian limestone/dolomite plateau (Givat Ha-Kerem) in Jerusalem, covered by terra rossa (Danin and others 1983). Since 1948, grazing has been excluded almost completely (there was no grazing at all by domestic animals during 1987-8), and as a result, the site is dominated by wild cereals (as are many other outcrops of the same rock type around Jerusalem). In recent years, fires have been set annually on Givat Ha-Kerem by schoolchildren celebrating the end of term; because of the extremely rocky terrain, however, fires arc spotty, so that some patches remain unaffected. To determine vulnerability of wild cereal diaspores to fire, I roped off small (IO-27 dm^2) plots in the burned parts of Givat Ha-Kerem in March, 1988 (9 months after the fire, and long after the completion of germination), and determined percent mortality as the ratio of number of spikelets without seedlings to total number of spikelets. Random samples of spikelets that failed to produce seedlings revealed that all grains were discolored or charred, and probably inviable: germination of spikelets in these species is normally on the order of 90 percent or more (Anikster and others 1988; Golcnberg 1986; Blumler unpublished data). I sampled plots in open ground, as well as adjacent areas by rocks. Because of the large size and only partial burial of wild cereal diaspores in open areas, all spikelets could be easily located. However, determination of wild emmer mortality rates adjacent to rocks was more problematic, because it often buries itself completely there. Wild barley occurred almost exclusively in **unburned spots,** such as ant nests and rockpiles; I was unable to locate it in burned areas until anthesis, at which point it was too late to sample the previous year's diaspores. Hence, only the effects of fire on wild emmer and wild oats could be studied and compared. However, the strong preference of wild barley for unburned spots suggests that it is intolerant of fire. Unfortunately, I developed back problems and was forced to terminate the experiment after only a small number of plots had been examined. Nonetheless, I obtained statistically significant results.

The results are presented in table 2. In open areas, wild emmcr spikelet mortality was high, on the order of 60 percent. Wild oat mortality was less than that of wild emmer

Table 2. Scorching of wild cereal diaspores by an arson fire on Givat Ha-Kerem, Jerusalem, 1987

	Wild Emmer			Wild Oats		
Plot[a]	Scorched	Seedlings	Pct Mortality[b]	Scorched	Seedlings	Pct Mortality[b]
S1	164	98	62.6	12	4.5	21.1
s2	39	42	48.1	47	140	25.1
s3	112	17	86.8		0	100.0
s4	61	63	49.2	4	11	26.1
S5	29	28	50.9	13	102	11.3
	405	248	62.0	77	298	20.5
RI	16	15	51.6	2		66.7
R4	19	26	42.2	2	4	33.3

[a] Plots Sl-S5 were located in open soil; plots R1 and R4 were located beside rocks, adjacent to S1 and S4, respectively.

[b] Percent mortality was calculated assuming that one and only one seedling arose from each viable diaspore. To the extent that seed predators may have removed diaspores, total mortality rates are likely to have been higher than calculated here.

[1] Wild emmer and wild oat diaspores typically contain a non-dormant grain and one or more dormant ones; the roots of the plant derived from the non-dormant grain tend to pull the diaspore down into the soil to some extent, thus increasing burial of dormant grains (Zohary pers. comm.).

at every open soil plot except S3, which contained only a single wild oat diaspore. If this plot is excluded because of the small wild oat sample size, wild emmer mortality was significantly greater than that of wild oats (t-test, $p < .01$). Most scorched diaspores lay on the surface or penetrated only slightly beneath it. Many surviving grains, on the other hand, were almost completely buried. Wild oats were rare adjacent to rocks, so comparison with wild emmcr in that microsite is problematic. There was no clear indication that the two species differed significantly in mortality around rocks. Wild emmcr survival was greater in plots R1 and R4 than in the adjacent open soil plots, but tests of statistical significance cannot be carried out on such limited data. Within R1 and R4, surviving wild emmcr diaspores were often complctely buried, and immediately adjacent to a rock, whereas the destroyed diasporcs tended to bc a few ccntimctcrs away from the rocks, and not so well-buried.

DISCUSSION AND CONCLUSIONS

These preliminary results suggest that fire could have been employed to selectively reduce the importance of wild barley, and perhaps also to favor wild emmer in rocky areas, but that wild oats would generally benefit most. At Givat Ha-Kcrcm wild oats dominates most open soil areas, for instance, while wild emmer shares dominance in open soil at Givat Ram, a nearby unburned site. However, further multi-year studies are needed to verify these conclusions, and other aspects of wild cereal demography need to be studied, such as the effects of fire on reproductive output and predation. Wild emmcr seems to be a poor competitor (compared to wild barley and wild oats), so reduction in seedling density might allow it to establish larger, more productive individuals. Harvester ants (***Messor semirufus***) are the major seed predators at many wild cereal sites, such as Givat Ha-Kcrem, and they forage more efficiently where litter has been removed. Observations indicate that ants take wild barley relatively easily, perhaps because its single awn and elongate shape increase its maneuverability; wild oats and emmcr, on the other hand, have spreading awns that catch in litter, and harvesting rates of these two species may increase dramatically on a bare surface. In the absence of empirical data, one cannot rule out the possibility that in open areas wild emmer mortality from charring and heat might be balanced by wild oat mortality from predation, although this seems unlikely. Also, while the distribution of wild barley is suggestive, other explanations for its rarity on the Givat Ha-Kerem bum should be considered. It is possible, for instance, that wild barlcy requires relatively fertile conditions, such as are found on the harvester ant nests, which happen also to be seldom burned. However, data on species composition before and after fire at other Israeli sites (Olsvig-Whittaker and others n.d.; Blumler unpublished data; Naveh and Whittaker unpublished data) indicate that wild barley suffers dramatically during the first season after a bum.

There is no archaeological record of use of wild oats, and wild oats are less efficient to harvest than wild emmer or barley (Ladizinsky 1975). Hence, it is by no means certain that Epipaleolithic people would have found it worthwhile to burn wild cereal stands. Nonethclcss, they may have burned dense maquis on Mt. Carmel and other Levantine hills to encourage wild cereal establishment, as proposed by Naveh (1984; Kutiel and Navch 1987a, b).

Another possibility, which has not been considered in the agricultural origins literature, is that fire may have been used to favor wild legumes (specifically, the progenitors of lentils, peas, and chickpeas). These plants are seldom abundant today, and thus seem to be poor food resources (Blumlcr 1991a; Ladizinsky 1987; Zohaty and Hopf 1973). Yet they, too, were cultivated at an early date-possibly as early as or even earlier than the cereals (Kislev and Bar-Yosef 1988).[6] Fire often seems to favor legumes (e.g., Barry 1971; Burrows 1991; DeSelm and Clebsch 1991; Masters 1991; Parsons and Stohlgrcn 1989). Future research should be directed towards testing the hypothesis that burning could have markedly increased the abundance of edible pulses, as well as to refining our understanding of the effects of tire on wild cereal grasses

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[6] Interpretation is problematic because of the diffuculty in distinguishing wild from domesticated legumes in the archaeological record.

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FLORISTIC AND HISTORICAL EVIDENCE OF FIRE-MAINTAINED, GRASSY PINE-OAK BARRENS BEFORE SETTLEMENT IN SOUTHEASTERNKENTUCKY

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Abstract-Several rarc plant species in Appalachian Kentucky have been found generally on sandy ridgcs of thc southern Cliff Section in native grassy roadside vegetation or young brushy pine-oak (*Pinus-Quercus*) woods, and almost never in arcas with less human disturbance. They include *Agalinis decemloba*, *Aster* ***concolor***, *Castanea pumila*, *Cirsium carolinianum*, *Eryngium yuccifolium*, *Gymnopogon ambiguus*, *Helianthus atrorubens*, *Liatris squarrosa*, *Lilium* ***philadelphicum***, *Oenothera perennis*, *Parthenium integrifolium*, *Phlox amoena*, *Polygala polygama*, *Rhynchosia tomentosa*, *Robinia hispida* var. *rosea*, *Sanicula marilandica* **(var.** *petiolulata*), *Schwalbea americana* (a candidate for fcderal protection) and *Sporobolus clandestinus*. Most are **concentrated in** the southeastern U.S.A., and scveral arc. typical of opcn pine or oak woods with frequent fire. Either these species have invaded roadsides and other disturbed arcas aflcr settlement, or they arc relicts from openings that wcrc maintained by fire, Indians and large hcrbivores bcforc settlement. Thc latter hypothesis is supported by the virtual absence of thesc spccies in reccnt clearings, suggesting low reproductive ratcs; some species have disappeared since 19.50. Also, therc are historical indications that firc did maintain some opcn pins-oak barrens, togcthcr with an associated Federally Endangered animal--thc red-cockaded woodpecker (*Picoides* ***borealis).***

INTRODUCTION

During ongoing invcntory of rarc spccies in thc Daniel Boone National Forest (DBNF) **and other** areas of Appalachian Kentucky (Palmer-Ball and others 1988, Campbcll and others 1989, 1990, 1991), it has become clcar that several rarc plants are largcly rcstrictcd to roadsides and other currently disturbed upland areas in the southern “Cliff Section” (Braun 1950). The purpose of this paper is to summarizc the distributions of these species, and to present the hypothesis that most arc relicts from woodland openings maintained by firc, which became supprcssed with the cstablishmcnt of DBNF in 1930-40 (Martin 1990).

The Cliff Section is a highly disscctcd, largely forested region with exposures of sandstone and, at low elevations, limcstonc (fig. 1). To the wcst, it merges with the Highland Rim in the south, or with the Knobs in the north. To the cast, therc is a transitional “Low Hills Bclts", then thc "Ruggcd Eastern Arca" of the Appalachian Plateau, then the Cumberland Mountains (Braun 19.50). The boundary of DBNF approximatcs that of the Cliff Section, plus some extensions into the southern Rugged Eastern Area. The Cliff Section has a much greater density of globally and regionally rare plant spccics than elsewhere in Appalachian Kentucky, except for some parts of the Cumberland Mountains.

The rare species can be divided into groups based on their typical habitats. These arc rocky banks of larger streams, seeping streamheads on broader ridges, overhanging cliffs with rockhouses, flattcr rock outcrops on clifftops and ridges, typical upland forest on moist to dry soil (with relatively few examples), and upland roadsides or disturbed woods. There is almost no overlap betwecn these species groups, except for about fivc of the roadsidc group that have also been found on rocky riverbanks (scc below). The “roadside” group mostly occurs in the southern half of the Cliff Section, especially in McCreary, Pulaski, Whitley and Laurel Counties (fig. 1). Most spccics are restricted to the southeastern U.S.A., and several are typical of open pine or oak woods with frequent firc. The term “barrens” *is* adopted in this paper for *the* putative firc-maintaincd, prescttlcmcnt vcgctation. This term has similar meaning to “savanna”, which has been used more widely in the southeastern Coastal Plain for particularly flat and grassy woodland with greater seasonal changes in hydrology. Species' nomenclature generally follows Kartcsz and Kartcsz (1980).

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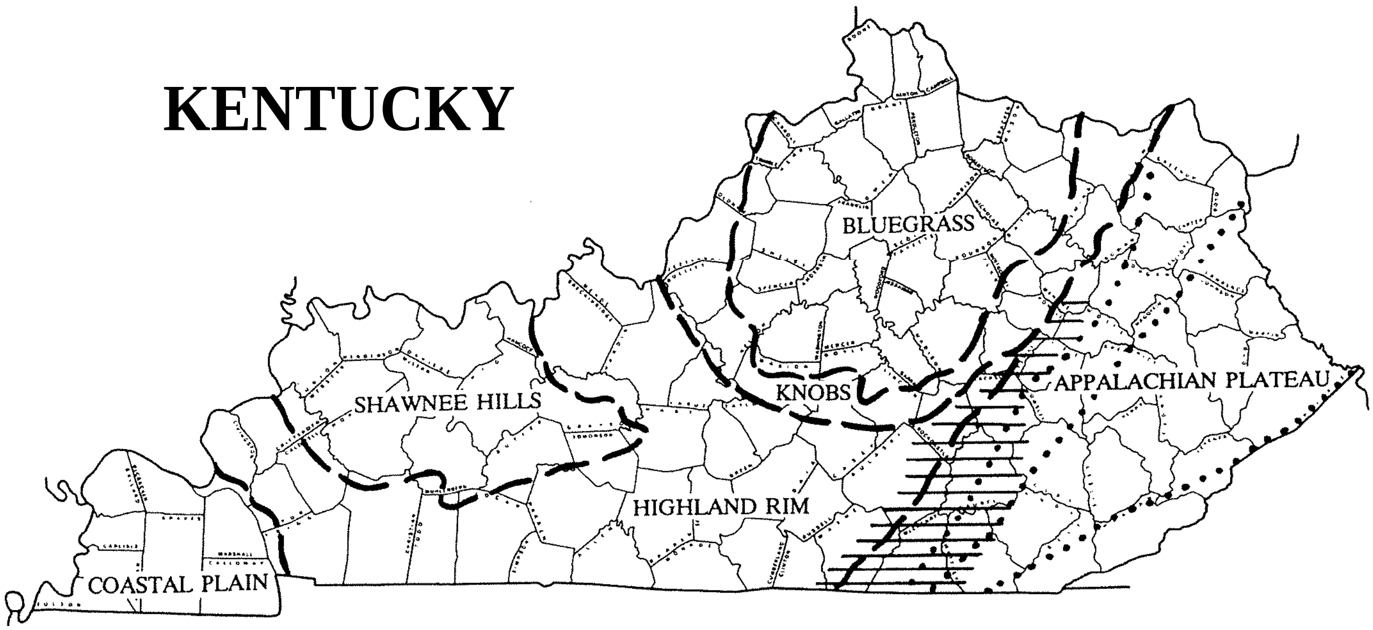

Figure 1. Map of Kentucky showing (a) major physiographic regions (dashed lines); (b) subsections (dotted lines) of the Appalachian Plateau and Mountains--from west to east, Cliff Section, Low Hills Belt, Rugged Eastern Area and Cumhcrland Mountains; and (c) the study area referred to in text (hatched)

SOURCES OF INFORMATION

Most of the data summarized below is in the database on rare species that the Kentucky State Nature Preserves Commission (KSNPC) has compiled during the past 15 years from all known publications on Kentucky plants, and from unpublished data shared by cooperating rcscarchers. The rare species list used by KSNPC is currently undcrgoing revision, and relevant unpublished data of the authors should be incorporated soon. Records from the Somerset and Stcarns Districts of DBNF, with which this paper is largely conccmcd, were increased 3-4 times during the 1987 and 1989 cooperative inventories (see Introduction). Most of the earlier records were referred to by Rogers (1941) and Braun (1943, plus her collection labels). The inventories included searches along most public roads in the study areas, and much exploration of remote forested areas. Although transects were not quantitatively sampled, statements below that certain species appear largely restricted to roadsides arc based on intensive local exploration. At most open sites with rare species, the adjacent forest was explored to see if these species extended into the shade. For general information on the effects of fire in recent history, D.D.T. interviewed several older USFS personnel and other residents familiar with the southern half of DBNF, especially Winnifer Freeman (Somerset), Clcmon Garrison (Mount Victory), Joe Planck (Somerset) and Ray Powell (McKee). The only other summary of fire-history in DBNF is the recent study of Martin (1990), which includes statistical details from USFS files.

NOTES ON RARE SPECIES OF ROADSIDES AND THICKETS

The list below includes all native species that generally meet two criteria.

(1) Within the Appalachian region of Kentucky, they are largely restricted to well-drained non-calcareous uplands of the Cliff Section (especially the southern half) and, in some cases, the Cumbcrland Mountains.

(2) Within this region, there are generally fewer than 10 records of each species (excepting *Helianthus atrorubens*, *Malus angustifolia* and *Phlox amoena),* and most records are from grassy roadsides, brushy areas or open woods disturbed by man within the past IO-20 years.

In parentheses after each species: name is the number of Kentucky counties where it is known, followed by the approximate total of known sites in the state. Unless stated, most records are post-l 950.

Agalinis decemloba (2/3). In Kentucky, this southeastern (mostly Piedmont) species of dry open woods and edges is known from only three collections. These were made in 1927 (oak-pine woods), 1934 and 1987 (grassy roadside), all on dry, sandy ridges in the southern Cliff Section.

Asclepias amplexicaulis (1 1/35). This widespread eastern species of dry openings is scattered over southern Kentucky. Almost all Appalachian records are from roadsides on ridges in the southern half of the Cliff Section.

Aster concolor (4/12). In Kentucky, this southeastern species of dry sandy barrens and open woods is known mostly from the southern Cliff Section (plus two western sites). All post-1970 records are from upland roadsides, with only 1-20 plants, except for one site on Kentucky State Route (KSR) 751 with several hundred. Two of the four older (1940-50) records are from "dry" or "open grassy pine" woods.

Aureolaria pectinata (7/10). In Kentucky, this southeastern species of dry woods and openings is known from scattered southern counties. There is one Appalachian collection, made during 1939 in the southern Cliff Section on a "dry wooded bank of Bridge Fork Pond" [an artificial pond]. The closely related, or sometimes combined, northern species, A. *pedicularia* is known from grassy openings on Pine Mountain (Harlan Co.).

Carex gravida (13/20). This widespread mid-western species of open ground is known from several scattered areas of Kentucky (except the Bluegrass Region), mostly as the more southern var. *lunelliana*. In the Cliff Section, it is known only from disturbed woods, edges and roadsides, on sandy and calcareous soils.

Carex physorhytzca (2 14). This southeastern species of sandy open woods was recently discovered in Kentucky, from a barrens in the western Highland Rim (R. Cranfill, pers. comm.), and from roadsides and adjacent young, open *Pinus virginiana* woods in the southern Cliff Section.

Castanea pumila (7/10). In Kentucky, this southeastern species of dry, sandy thickets and disturbed woods is largely restricted to the Cumberland Mountains, but there are a few, mostly old, western records. There are only two verified Cliff Section records, both southern, from 1935 and 1989 (a suppressed individual on a small stream terrace). Johnson (1989) noted "its ability to recover from fire and other disturbances, through rapid suckering and sprouting from the remaining stem at or below the ground level." Near Jasper, Georgia, M.E.M. observed abundant suckers after clearcutting, in association with *Robinia hispida* varieties (see below). In addition to fire-suppression, the chestnut blight (*Endothia parasitica*) has probably reduced this species.

Cirsium carolinianum (6/15). This southeastern species, generally of dry sandy open woods and edges, has been recorded in scattered areas of Kentucky. Most sites are in the Cliff Section with half these in a 50 km^2 area along the Cumberland River, generally on roadsides, except for one in a more natural grassy pine-oak woods (table 1), and another at the edge of a small calcareous prairie. Only 1-20 plants have been found at each site.

Digitaria violascens (414). This southern, pantropical species of open pineland has been reported in scattered non-calcareous regions of Kentucky. The two Appalachian records are from dry sandy roadsides on Cliff Section ridges. Whether it is truly native to open pine woods in this region may be doubted given the weedy nature of this gnus.

Eryngium yuccifolium (> 20/ > 50). This southeastern and mid-western species of prairies and barrens is frequent in naturally open vegetation of western Kentucky, but there is only one Appalachian record, from the southern Cliff Section on "a moist flat of pine-oak barrens" (Rogers 1941).

Gymnopogon ambiguus (7112). **In** Kentucky, this southeastern species of dry, sandy openings and open pineland (especially fire-maintained *Pinus palustris* woods on the Coastal Plain) is known from scattered southern areas. There are only three Appalachian records, from "sandy shores, South Fork of Cumberland River" in 1935, and, in the 1980s, from an open grassy roadside in the southern Cliff Section and a seasonally wet field in the southern Rugged Eastern Area.

Helianthus atrorubens (6/ > 50). In Kentucky, this southeastern and Appalachian species, generally of dry open woods and edges, is known only from the southern Cliff Section and the transition to the Low Hills Belt. The sites are mostly roadsides, and occasionally young open woods. It is locally dominant, with patches of several hundred plants, but only non-flowering plants occur in more shady areas.

Leiophyllum buxifolium (1/1). This species of Appalachian heath-balds, Coastal Plain sand-hills and pine-barrens, has been collected once in Kentucky, during 1939: "[a single individual] on top of dry [sandy] bank, Cumberland Falls, within park on S side of road going west, about 100-200 yards from park entrance" (McInteer 1940). The area is generally forested today, except for the roadsides. Another Appalachian heath-bald species, *Rhododendron minus,* still occurs in roadside woods near the park, but these may be planted. There is one other report of *R. minus* from Kentucky, on Pine Mountain (E. Carr, pers. comm.).

Table 1. vascular plants found on or near grassy roadsides with rare species Nomenclature generally follows Kartesz and Kartesz (1980); selected synonyms are in parentheses. Family arrangement follow Thorne (1961). + indicates species present; * indicates species locally abundant.

A-H: areas of ID-150 m^2 around patches of Aster concolor on KSR 751, from 1.5 km S of US 27 to 0.4 km N of railroad, Pulaski Co.

I: ca. 150 m^2 around patches of Aster concolor along dirt road, Bindsfield Ridge, 1.5 km SE of KSR 192, Pulaski Co.

J: ca. 100 m^2 around patches of Polygala polygama by Sand Hill Church on KSR 700, 8.5 km S of KSR 92, McCreary Co.

K: areas totaling ca. 50 m^2 around patches of Lilium philadelphicum on KSR 192, 1.15-2 km E of Craig's Creek Road, Laurel Co.

L: ca. 100 m^2 at "Bald Rock" on KSR 192, 1.7 km E of KSR 1193, Laurel Co.

M: ca. 3000 m^2 along KSR 700 and adjacent utility right-of-way, 1.5-3 km SW of sand Hill Church, McCreary Co.

N: ca. 200 m^2 around Polygala polygama along dirt mad by Roaring Paunch Creek, 0.8 stream-km N of KY-TN state-line, McCreary Co..

O: ca. 300 m^2 around Sporobolus clandestinus and Phlox amoena along KSR 791, 1.3 km W of KSR 92. McCreary Co.

P: ca. 2000 m^2 around outcrops at Dobbs Hill on KSR 1363, 0.8-1.1 km W of Pleasant Bill Church, McCreary Co.

Q: ca. 5ooo m^2 of pine-oak barrens (traversed by old road bed), 0.8 km NW of Barren Fork Cemetry, I.6 km SSE of Flat Rock, McCreary Co.

	A	B	C	D	E	F	C	H	I	J	K	L	M	N	O	P	Q
PTERIDOPHYTES																	
Pteridium aquilinum var. latiusculum				+	+									+			
Pellaea atropurpurea																	*
Asplenium platyneuron					+												+
Woodsia obtusa																	+
Polystichum acrostichoides					+						+						
GYMNOSPERMS																	
Pinus rigida								+			t						
Pinus virginiana	t	+	+	t	t	t		t			t		t			*	*
Pinus echinata				+	+	+				t	t		*			t	
Juniperus virginiana	+					t										*	t
DICOTELEDONS																	
Magnoliaceae																	
Liriodendron tulipifera				+	+						+		+				+
Annonaceae																	
Asimina triloba																	
Lauraceae																	
Sassafras albidum										+	+		+	+	+		
Ranunculaceae																	
Anemone virginiana											+						+
Aquifoliaceae																	
Ilex opaca																	
Clusiaceae																	
Hypericum (Ascyrum) hypericoides				+	+							+					+
Hypcricum gentianoides												+					+
Ericaceae																	
Oxydendron arboreum	+			+		+											
Gaylussacia brachycera																	+
Vaccinium stamineum (and neglectum)											+	+					
Vaccinium arboreum	+	+	+		+	+										+	+
Vaccinium vacillans	+				+					+	+						
Ebenaceae																	
Diospyros virginiana				+						+					t		t
Caryophyllaceae																	
E Cerastium glomeratum (viscosum)																	+
E Arenaria serpyllifolia																	+
E Dianthus armeria																	+
Silene virginica																*	+
E Silene antirrhina																	+
Pottulacaceae																	
Talinum teretifolium																	+
Ptiniulaceae																	
Lysimachia lanceolata												+					
Lysimachia quadrifolia																	
Polygonaceae																	
E Rumex acetosella																	

Table I. **(continued)**

	A	B	C	D	E	F	C	H	I	J	K	L	M	N	O	P	P
Geraniaceae																	
Geranium carolinianum																+	
Oxalidaceae																	
E Oxalis **fontana** ("europaea")																+	
Oxalis **violacea**																t	
Linaceae																	
Linum virginianum											+		+				
Polygalaceae																	
Polygala polygama										+			+				
Polygala **senega var.** latifolia													+				
Polygala **curtissii**																	
Polygala (verticillata **var.) ambigua**													+				
Celastraceae																	
Celastrus scandens																	
Euonymus americanus																*	
Violaceae																	
Viola **pedata**												t				t	
Viola hirsutula																+	
Viola **cf. emarginata**										t		t					
Viola **tripartita** var. glaberrima											t						
Viola **rafinesquii**																+	
Brassicaceae																	
E **Arabidopsis** thaliana																+	
E Cardamine hirsuta																	
E Erophila **verna** (Draba v.)																+	
Lepidium virginicum																	
Ulmaceae																	
Ulmus alata																+	*
Celtis tenuifolia																t	
Cistaceae																	
Lechea racemulosa																+	
Rhamnaceae																	
Rhamnus caroliniana							+								⊡	•	
Ceanothus americana													+		⊡		
Euphorbiaceae																	
Euphorbia corollata	t									t			t			t	
Anacardiaceae																	
Rhus **(Toxicodendron)** radicans											t				*	+	
Rhus glabra													+				
Rhus copallina		+	+	+	+			t	t	t			t			t	
Juglandaceae																	
Carya tomentosa									+						t		
Carya glabra													t				t
Carya pallida		?	?	+	?							t				t	
Aceraceae																	
Acer rubrum **var. r.**	+	+	+	+	+	+	+	+	t		t		*	+	+	+	
Fabaceae																	
Schrankia microphylla											t						
Cassia (Chamaecrista) nictitans									+	+		+	t				
E **Lotus** corniculatus																t	
E **Trifolium agreste** (procumbens)																+	
E Melilotus **officinalis**																t	
Desmodium nudiflorum													·				
Desmodium ciliare						▯					▯						
Desmodium marilandicum	+				▯								t			⊡	
Desmodium paniculatum (sensu stricto)										⊡	⊡			+		·	
Desmodium glabellum* **(not humifusum)**							ℓ		ℓ		ℓ		+				
Desmodium viridiflorum													t			⊡	
Desmodium laevigatum				+						t	t					▯	
Lespedeza repens						+											
Lespedeza virginica		t	+													t	
Lespedeza intermedia		t		t	t				t								
Lespedeza intermedia x virginica						+											
Lespedeza hirta		+			t	t						+	+			t	
E Lespedeza striata													+			?	
E **Lespedeza cuneata**						t							t				
Stylosanthes biflora		+										t					
Tephrosia virginiana var. v.	ttt				+							t				+	
Clitoria **mariana**										t			t				

Table 1. (continued)

	A	B	C	D	E	F	C	H	I	J	H	L	M	N	O	P	Q
Amphicarpa bracteata var. b.											+						
Galactia volubilis					+				+	+			+			+	
Hamamelidaceae																	
Liquidambar styraciflua																+	
Fagaceae																	
Quercus alba	+	+	+							+	+		*				
Quercus stellata			+			+			+				+				*
Quercus montana ("prinus")	+					+					+		+			+	+
Puercus falcata			+		+	+										+	
Quercus marilandica					+	+	+									+	+
Quercus velutina	+		+		+	+	+		+		+				+	+	
Quercus coccinea		+			+				+	+	+		+			+	
Betulaceae																	
Ostrya virginiana																+	
Rosaceae																	
Fragaria virginiana var. v.					+	+									*	+	
Potentilla simplex var. s.										+	+						
Potentilla canadensis var. c.	+	+				+						+	+			+	
Rubus Flagellares (group)		+	+							+	+		+			+	
Rubus allegheniensis var. a.														+			
Rubus Argutae (group)					+				+	+				+			
Rosa Carolina																+	
Prunus serotina		+	+														
Amelanchier arborea var. a.												+				+	
Crassulaceae																	
E Sedum acre																*	
Onagraceae																	
Oenothera biennis					+											+	
E Oenothera laciniata																+	
Rubiaceae																	
Bedyotis purpurea (Houstonia lanceolata)										+			+			+	
Bedyotis caerulea (Houstonia C.)																+	
Galium pilosum var. p.					+					+	+					+	
Apocynaceae																	
Apocynum cannabinum											+						
Asclepias amplexicaulis										+			+				
Asclepias exalatata																?	
Asclepias variegata													+				
Asclepias verticillata																	+
Gentianaceae																	
Gentiana villosa															+		
Scrophulariaceae																	
Agalinis decemloba					+												
Pedicularis canadensis					+								+				
Ambrosia artemisifolia													+			+	
E Achillea millefolium																+	
E Chrysanthemum leucanthemum (L. vulgare)	+	+								+	+		+			+	
Senecio anonymus (smallii)		+				+		+	+	+					+	*	
Chrysopsis (Pityopsis) graminifolia		+		+	+			+	+	+			+		*	+	+
Chrysopsis mariana	t	t		+	+	+	+	+				+	+				
Solidago hispida																?	
Solidago erecta	+++				+		+		++			++				+	+
Solidago nemoralis	+++				++			++				+	*				
Solidago arguta var. a.									+				+	+			
Solidago odora	t	t		+		+	+	+	+	+					+	+	
Solidago rugosa													+				
Solidago gigantea						?											
Solidago altissima		t		+									+				
Solidago (Euthamia) graminifolia						+											
Aster undulatus		+			+	+					+				+	+	
Aster patens var. patens	+	+	+		+	+	+		+	+			+	+		+	
Aster surculosus		+						+				+	+			+	*
Aster concolor	+	+	+	+	+	+	+	+	+								
Aster linariifolius		+		+	+										+	+	
Aster infirmus													+				
Aster umbellatus													+				
Aster paternus													+				
Aster solidagineus	+				+												
Aster pilosus var. pilosus		+	+			+						+				+	
Aster lateriflorus												+	+				

Table 1. (continued)

	Taxon	A	B	C	D	E	F	G	H	I	J	W	L	M	N	O	P	Q
	Aster dumosus var. coridifolies	+	+	+	+		+		+				+					
	Erigeron annuus										+	+					+	
	Erigeron strigosus													t			+	
	Conyza canadensis var. pusilla												+				+	
	Gnaphalium obtusifolium		+	+		+	t						+				+	
	Antennaria plantaginifoiia		?				?	?				+		t			+	
	Eupatorium fistuiosum				+	+								t				
	Eupatorium album	+					t						+					
	Eupatorium rotundifolium	t	t				t		t	t							t	
	Eupatorium aromaticum		+				+		+		+	+						
	Eupatorium rugosum var. r.			+														
	Liatris squarrulosa					+	+			+								
	Liatris microcephala																+	+
	Cirsium muticum														+			
	Cirsium carolinianum																	+
	Cirsium discolor																+	
	Elephantopus tomentosus													+				
	Prenanthes serpentaria					?	?											
	Lactuca canadensis																+	
	Hieracium venosum						+					+					+	
	Krigia virginica																+	
	MONOCOTYLEDONS																	
	Liliaceae																	
	Lilium philadelphicum var. p.				?							+		+				
	Uvularia perfoliata											?		+				
	Aleris farinosa													+				
	Smilax glauca	+	+			+				+	+		•		□		+	
E	Verbascum thapsus																♦	
E	Veronica arvensis																□	
	Plantaginaceae																	
	Plantago rugelii		+															
	Plantago virginica																+	
E	Plantago lanceolata																+	
	Acanthaceae																	
	Ruellia caroliniensis																+	
	Lamiaceae																	
	Prunella vulgaris var. lanceolata																	
	Salvia lyrata													+			+	
E	Satureja (Clinopodium) vulgaris												+					
	Pycnanthemum pycnanthemoides var. p.																+	
E	Mosla dianthera												+					
	Polemoniaceae																	
	Phlox amoena															*	*	
	Solanaceae																	
	Solanum carolinianum																+	
	Convolvulaceae																	
	Ipomaea pandurata													+				
	Convolvulus (Calystegia) spithameus															+		
E	Convolvulus (Calystegia) sepium var. s																?	
	Campanulaceae																	
	Lobelia puberula var. simulans	+		+			+		+			+		+	+			
	Lobelia inflata												+					
	Specularia (Triodanis) perfoliata																+	
	Vitaceae																	
	Vitis aestivalis										+			+			+	
	Vitis vulpina										+						+	
	Vitis rotundifolia													+		*	+	
	Parthenocissus quinquefolia										+			♦			•	
	Nyssaceae																	
	Nyssa sylvatica var. s.	+					♦♦				•			□				
	Cornaceae																	
	Cornus florida			+		+		+	+	+		+		+		+		
	Apiaceae																	
	Sanicula marilandica var. petiolulata													□				
	Sanicula canadensis											t						
E	Daucus carota										t			□			t	
	Caprifoliaceae																	
E	Lonicera japonica																+	

Table 1. (continued)

		A	B	C	D	E	F	G	H	I	J	K	L	M	N	O	P	P
	Asteraceae																	
	Helianthus atrorubens var. a.									?+				+				
	Angelica venenosa													+				
	Helianthus divaricatus					+					+			+				+
	Helianthus microcephalus		+			+					+	+		+		+		
	Helianthus hirsutus	+	+		+	+				+			+					
	Verbesina (Actinomeris) alternifolia														+			
	Verbesina occidentalis													+			+	
	Rudbeckia fulgida var. fulgida					+							+					
	Rudbeckia triloba var. t.	+															*	
	Coreopsis major	+	+	+	+	+	+	+	+	+	+	+	+				+	+
	Parthenium integrifolium var. i.										+			*				+
	Smilax bona-nox		+															
	Smilax rotundifolia				+			+				+		+	+			
	Dioscoreaceae																	
E	Dioscorea batatas													+				
	Juncaceae																	
	Luzula echinata																+	
	Cyperaceae																	
	Cyperus ovularis															+		
	C. gravida															+	+	
	C. artitecta (and C. physorhynca)															t	t	
	Carex umbellata (tonsa)																t	
	Carex nigro-marginata																t	
	Carex hirsutella (complanata var. h.)													t			t	
	Carex caroliniana													.				
	Carex suanii												8					
	Poaceae																	
	Stipa (Piptochaetium) avenacea										+		⊡	⊠		*	+	*
	Danthonia spicata		+										•	⊠	+		+	*
	Danthonia sericea																t	t
	Aristida dichotoma					+												
	Aristida longespica	+																
	Aristida purpurascens var. virgata		+															
E	Bromus japonicus																	
E	Poa compressa																	
E	Poa pratensis																+	
E	Festuca elatior (sensu lato)													+				
E	Festuca (Vulpia) octoflora																+	
E	Holcus lanatus																+	
E	Dactylis glomerata																	
	Elymus glabriflorus												+					
	Gymnopogon ambiguus	+				+												
	Eragrostis spectabilis															+		
	Sporobolus clandestinus															+	*	
	Erianthus alopecuroides					+								+				
	Andropogon (Schizochyrium) scoparius	*	+	*	*	*	+	+	t		t		+	*		+	*	+
	Andropogon gerardii	*		+	*													
	Andropogon virginicus var. v.					+	+			+	+		+		+			
	Andropogon ternarius					t	t									+	+	
	Sorghastrum nutans	*		+	*	+	+			t								
	Panicum flexile						t						+					
	Panicum anceps		+	*	+	+				t						*	+	
	Panicum (Dichanthelium) boscii	t	t						t					+			+	*
	Panicum (D.) commutatum														+		+	
	Panicum (D.) ravenelii													+		?		
	Panicum (D.) polyanthes												+		+			
	Panicum (D.) sphaerocarpon													+				
	Panicum (D.) laxiflorum										+		+	+	*		+	
	Panicum (D.) lindheimeri												+					
	Panicum (D.) acuminatum (sensu lato)					+			+				+	+	+		+	
	Panicum (D.) microcarpon										+							
	Panicum (D.) dichotomum (sensu stricto)	+				+	+						+	*			?	+
	Panicum (D.) depauperatum																+	
E?	Setaria geniculata																	

Lespedeza capitata (or hybrids) (>10/ > 20). In Kentucky, this widespread eastern species of dry openings (often sandy pincland) is scattered in western regions. Howcvcr, there arc only three Appalachian records, all from the Cliff Section during 1935-40 (Rogers 1941, Braun 1943). Braun's plants were idcntificd as *L. simulata*, which has been interpreted as a hybrid bctwccn *L. capitata* and ***L. virginica*** or ***L. intermedia***. Also, it was found during 1989 in a wet bottomland meadow half a mile south of the KY-TN state-line.

Liatris squarrosa (> 20/ > SO). In Kentucky, this widcsprcad southcastcrn species of dry opcnings is frequent in southern and wcstcrn regions, but there are only two Appalachian records. Both are from rclativcly undisscctcd ridges on the eastern side of the southcrn Cliff Section, in 1937 (dry open woods) and 1989 (dirt roadside). *Liatris* spccics that are more widcsprcad in the Cliff Section also occur on the roadsides, especially *L. squarrulosa*, less often *L. spicata*, occasionally *L. aspera*, and **on** rocky sites ***L. microcephalu.***

Lilium philadelphicum (9/21). In Kentucky, this species of dry to damp, acid openings, thickets and pine barrens is known only along the Cliff Section, as the largcly Appalachian var. *philadelphicum*. Most records are from roadsides on ridges, but one is from the edge of a calcareous prairie, and a 1940 record is from dry pine woods. It was locally common (Rogers 1941) but several populations have disappcarcd in the past 10-30 years. Currently, most sites have only l-5 plants; none cxcccd 20. In addition to firc-suppression, digging for ornament may have contributcd to its decline.

Malus angustifolia ***(61*** >25). This southeastern species of opcn woods and thickets is reported from scattered non-calcareous areas of southern Kentucky, mostly in the southern Cliff Scction. There may be some intermediates with the more northern ***M. coronaria*** (cspccially var. *lancifolia)*. Most sites have only 1-1 0 trees and are on dry ridges along roads and in young woods *of* ***Pinus virginiana*** or pine-oak. One population has at least 50 trecs (with much *Crataegus* spp. and *Prunus americana).* Less often, it has **been** found on rocky riverbanks, or on bottomland thickets extending into the Rugged Eastern Area.

Melampyrum lineare (3/3). In Kentucky, this northcrn species of dry to damp brushy grassland and open woods is known from a few records in the Cumberland Mountains, as var. *latifolium*, and in the Cliff Section, as ***var. americanurn.*** The latter variety is currently known only along the 75 m paved trail to Sky Bridge, with 400-500 plants at the edge of dry woods dominated by *Pinus rigida*, *Quercus* spp., *Gaylussacia baccata* and *Vaccinium vacillans*. An annual species, it often increases in woods after fire or logging (Swan 1970; Scheiner and Tecri 1981; Abrams and Dickmann 1984; Gibson and Good 1987).

Muhlenbergia torreyana (i/l). This southeastern species of "pine-barrens and meadows" is a candidate for federal protection. There is only one obscure Kentucky record (Hitchcock and Chase 1950), presumably based on a collection in the southern Cliff Section or the adjacent Highland Rim. It was formerly known from the oak-barrens region of Tennessee, in the transition from Highland Rim to Applachian Plateau (Svenson 1941).

Oenothera perennis (3/3). In Kentucky, this north-central species of wet, or occasionally (in the south) dry, natural openings is known from only thrcc southern records. The only Appalachian record, dated 1935, is from the southern Cliff Section, in "dry soil, open woods" along a ridge road, ***close*** to *Schwalbea atnericana* (Braun 1943 and collections). The only post-1960 record in Tennessee is from a physiographically similar roadside (Patrick and others 1983).

Orbexilum pedunculatum var. pedunculatum (Psoralea psoralioides **var. *p.)*** (4/1 0). In Kentucky, this southeastern variety of dry openings has been found in a few southern counties. The only Appalachian records are from the southern Cliff Section, either on ridgetop roadsides, or on open rocky riverbanks.

Panicum aciculare (415). In Kentucky, this southeastern (generally Coastal Plain) species of sandy pine woods is known from a few southwestern sites (as var. *aciculare*), and from two sites in the southern Cliff Section (as var. *angustifolium*) on ridges in dry, open, grassy pine woods near a road and a clifftop. As ***"P. angustifolium"***, this species was reported to increase greatly with frequent prescribed burning by DeSelm and others (1973). Some other *Panicum* spp. may have similar distributions and responses (e.g., *P. ravenellii*)

Parthenium integrifolium (> 20/ > 50). This mid-western and east-central species of dry openings is not rare in western Kentucky. Howcvcr, the only Appalachian records arc about 15 sites in the southernmost Cliff Section, mostly on ridgetop roadsides with only l-10 plants. At least 20 non-flowering plants were found in a more shady barrens (table 1).

Paspalum setaceum **var.** *longepedunculatum* (4/4). In Kentucky, this southeastern taxon of sandy openings with pine is known from a few open ridgetop sites in the southern Cliff Section, and as a railroad waif in Jefferson County.

Phlox amoena (7/ > 2.5). *In* Kentucky, this southeastern species of dry openings is known from the southern Cliff Section and the adjacent Highland Rim. It is locally frequent (with 100-1000 plants) along roads and adjacent forest edges, generally on sandy soils and occasionally above limestone. Only one site has frequent plants in more natural vegetation: the rather open, grassy, stunted *Pinus virginiana* woods on the sandstone outcrops *of* Dobbs Hill (table 1).

***Polygala polygama* (116).** In Kentucky, this northern and Coastal Plain species of dry, sandy openings is known only from the southernmost Cliff Section. Most sites are upland roadsides (with 10-50 plants), but a 1935 record is from "pine-oak" woods. It is a short-lived perennial that often increases after fire (Abrams and Dickmann 1984; Niering and Dreyer 1989).

Rhynchosia tomentosa (3/3). In Kentucky, this southeastern species of dry, sandy openings is known from only three records. The only Appalachian collection, dated 1949, is from the "edge of cut-over woods, 4 miles east of Cumberland Falls" (probably along KSR 92). Also, it was found in 1989 about 2 km south of the KY-TN state-line, on Big Island in the Big South Fork.

Robiniu ***hispida*** var. ***rosea*** **[=** ***R. boyntonii*]** (4/10). In Kentucky, this Appalachian taxon is known only from the Cumberland Mountains (with other varieties) and the southern Cliff Section, mostly in a 10 km^2 area around Day Ridge, McCreary Co. All sites are on relatively narrow sandstone ridges or knobs in dry pine or oak woods, especially young stands, thickets and edges. It occurs at low density in patches up to 30 m^2, with flowering observed only in the open.

Sanicula marilandica (1/2). In Kentucky, this north-central (to Rocky Mt.) species is known from only two localities, each with less than 10 plants, in the southern Cliff Section. They were found in edges and burned thickets along roads on rather broad ridges between the Cumberland River and the Big South Fork. The plants are var. ***petiolulata***, which typically occurs in "dry sandy pineland" of the southern Atlantic Coastal Plain (Fernald 1950).

Schwalbea americana (112). This species is a candidate for federal protection that is known from sandy, acid, damp to dry soil in open pine or oak barrens on the Atlantic Coastal Plain, plus a few old collections from Tennessee (including the "oak-barrens" of the Higland Rim-Cliff Section transition) and Kentucky. The Kentucky collections were from "dry sandy soil on knobs and sandstone plateau margins" in the southern Cliff Section, "with *Cleistes*" at one site (Braun 1937b, 1943). This species appears restricted to fire-maintained vegetation (S. Orzell, pers. comm.). Musselman and Mann (1977) noted: "particularly vigorous growth of *Schwalbea* was evident after early spring fire at the Horry Co. [SC] site. In such years, seed production was abundant." It is parasitic on a wide range of woody plants.

Sporobolus clandestinus (7/8). In Kentucky, this widespread eastern species of dry openings is known from scattered southern regions. The only Appalachian records are from the southern Cliff Section, on roadsides adjacent to sandstone outcrops, and on some limestone clifftops. It is locally dominant, with patches up to 300 m^2.

Tephrosia spicata (217). In Kentucky, this southeastern species of sandy openings is known only from the southern Cliff Section. Most records are from open rocky banks of the Cumberland River. Two others are collections dated 1941 and 1980 from disturbed areas on ridges near Barthell (McCreary Co.).

Viola fimbriatula **[=** ***V. sagittata*** var. ***ovata*]** (3/3). In Kentucky, this north-central species of disturbed woods (especially on mineral soil) is known from a few Cliff Section records. The only post-1950 record is from McCreary Co., along an old eroded roadside, with at least 30 plants.

NOTES ON ASSOCIATED VEGETATION

Some of the roadsides with rare species have a relatively high diversity of native species, and a low frequency of exotics. Table 1 lists the ca. 300 vascular species found at good examples of such vegetation and some nearby grassy woods. The most abundant species include several warm-season (C4) grasses, with *Andropogon scopurius* the most frequent dominant. The only typical native cool-season (C3) grass is *Stipa avenacea*, which is locally dominant in young pine woods adjacent to the roads. Other frequent species include composites, especially *Coreopsis major*, *Helianthus* spp., *Chrysopsis* spp., *Solidago* spp., *Aster* spp. and *Eupatorium* spp., and legumes, especially *Lespedeza* spp. and *Desmodium* spp. About 10 percent of the species present are exotics, but most of these were recorded only at the Dobbs Hill site, which is the only site adjacent to houses. The only exotic found at more than *2-3* sites is *Chrysanthemum leucanthemum* (= *L. vulgare*).

One of the most extensive native grassy areas, with several rare species (especially Aster *concolor)*, is along 3 km of KSR 751 between Burnside and Keno (Pulaski Co.). In addition to the typical dominants--Andropogon *scoparius, A. gerardii* and *Sorghastrum nutans,* frequent species here include *Tephrosia virginiana, Lobelia puberula, Helianthus hirsutus, Coreopsis major, Senecio anonymus, Chrysopsis mariana, C. graminifolia, Solidago erecta, S. nemoralis, S. odora, Aster patens, A. concolor, A. dumosus, Gnaphalirrm obtusifolium, Eupatoriron rotundifolium, E. aromaticum, Panicum anceps* and *Andropogon virginicus. Adjacent* forest is dominated by *Pinus* spp., *Quercus* spp. and *Acer rubrum*.

Typically the soils are hapludults with fine sandy or silty loam texture, A horizon pH of 4.5-5, and a depth of SO-150 cm to the sandstone or shale. On shallower hapludults or dystrochrepts near sandstone outcrops, the vegetation generally lacks taller species such as *Andropogon gerardii, Sorghastrum nutans, Helianthus atrorubens* and *Eupatorium* spp. An unusual variant with patches of *Sporobolus clandestinus* was found on some rocky sites, especially Dobbs Hill. In addition to abundant *Pinus virginiana* at this site, the trees included much *Juniperus virginiana,* which, together with some of the other frequent species (especially *Ulmus alata, Celtis tenuifolia, Rhamnus caroliniana, Phlox amoena, Rudbeckia triloba, Pellaea atropurpurea* and *Woodsia obtusa*), may indicate more base-rich soils (following Campbell 1987). *S. clandestinus* itself is also frequent on some limestone sites.

Apart from these roadside remnants, there is little information on the barrens or open forest that may have existed when annual burning was a common practice before DBNF was established. However, on drier ridges in the southern Cliff Section, Braun (1950, p. 102) notcd: "Instead of this pint-heath or pine-oak-heath community, some of the promontarics arc occupied by open pine woods (the three species of pine) with a grassy layer of *Andropogon scoparius* (little blucstcm), *A. glomeratus* (broom-sedge), and *Sorghastrum nutans* (Indian grass), in which arc a few scattcrcd forbs. Fires have modified most (perhaps all) of these pine summits, although the abundance of large *Cladonia* (lichen) mats is an indication that there has been *no* fire for many years."

Similar vegetation may have extended onto relatively moist sites, wherc droughts still occurred oflen enough to spread fires. The only direct information about such woods comes from Rogers (1941). He noted the following plants in "a moist flat of pine-oak barrens" along the road to Bauer: *Salix humilis* (vars. *humilis* and *microphylla), Hypericum punctatum, Eryngium yuccifolium**, *Liatris scariosa* [probably *L. squarrulosa**] and *L. spicata**; and he noted *Helianthus atrorubens** in "pine-oak barrens at the Tennessee State Line." Also near the Bauer Road, he noted several species typical of openings or edges, all in "woods" unless noted: *Andropogon gerardii* ("common"), *Robinia hispida**, *Lespedeza virginica, L. capitata**, *Polygala verticillata, Oxypolis* rigidior, *Angelica villosa, Cuscuta campestris, Solidago* caesia, *Aster patens* ("var. *phlogifolius*") and *A. solidagineus*; *Pycnanthemum pycnanthemoides, Helianthus hirsutus* and *Coreopsis major* var. *stellata* (all "dry woods"); *Anemone virginiana* ("dry pine-oak woods"); *Lobelia puberula* ("wet woods"); *Lilium philadelphicum** ("common along road"), *Coreopsis tripteris* var. *deamii* ("by the road"); *Hypericum frondosum* [*prolificum*?] ("low, moist, shaley soil in open thicket"). In *1987, Andropogon gerardii* is still common along the road to Bauer, but of the rarer species (shown by *), only *Helianthus atrorubens* was encountered.

Except for some of the roadsides, there are virtually no areas where a diverse native barrens vegetation remains. A few woodland-pastures today may bear some structural resemblance to prcscttlement barrens, and such areas were frequently burned by residents before the modem era of fire-suppression. However, grazing has been intensive, and exotic plants have often replaced the native flora in such pastures. None of the rare plants listed above have been found in actively pastured areas. A few dry, wooded areas near clifftops have an open grassy aspect, with occasional fires, but these have generally become too shady for most of the rare species noted here.

NOTES ON FIRE HISTORY

Archaeological evidence shows that, for at least 10,000 years before 1650, Indians lived in many parts of Appalachian Kentucky (e.g., Cowan 1985, Ison 1990). It is likely that they used fires extensively for managing game animals and clearing garden plots, especially on slopes near rockhouses. Lightning fires are relatively infrequent in DBNF, with only 10-15 per 1000 km^2 each year (Martin 1990), and they are probably not repeated often enough in the same locations to create open grassy vegetation. However, the role of Indians versus lightning in causing prescttlement fires must remain an open question.

There was almost no landscape description in the pioneer literature from the southern Appalachian Plateau in Kentucky and the adjacent Highland Rim. A few accounts suggest areas disturbed by fire, Indians or buffalo *(Bison bison).* Walker (1749) described an area in Jackson County where "The woods have been burnt some years past, and are now very thick, the only timber being almost all kill'd", and an area in Morgan County with "the only fresh burnt woods we have seen." He noted Indian trails and buffalo in several places. Near Pincville, he initially named Clear Creek as "Clover Creek", noting that "Clover and hop vines are plenty there"; the clover was probably *Trifolium stoloniferum*, which was associated with buffalo in Kentucky (Campbell and others 1988). Walker (1824), recounting his travels in 1775, noted "twenty miles, entirely covered with dead brush" in the Rockcastle and Laurel County area. This statement suggests the results of a large fire (see also McHargue 1941). Arnow (1960) noted descriptions of pioneers in the Cumbcrland River drainage that suggested "park-like" forests "with so little undergrowth a traveler could see a deer for 1.50 paces. There were, too, along the creeks and rivers, treeless glades and valleys, sometimes filled with cane...or only high grass..." Edwards (1970) described Wayne County (mostly in the eastern Highland Rim) during about 1775: "Three-fourths of the county was covered with virgin forests; the lowlands contained some cane, or tall grass as they preferred to call it...Price's Meadows [initially called the Big Meadow], near the mouth of Meadow Creek, contained very high grass. Corn could be planted without the forests being cleared."

Interviews with older residents provide a general historical view of the southern Cliff Section. Until DBNF was established in 1930-40, intentional fires were widespread, except perhaps for a few decades before 1910 when the Kentucky Landsharers Association had control over much land and restricted burning. Annual fires occurred in much of the area during 1910- 1930. They were generally set in February and March to promote grass and forb growth for cattle. Also, hogs ran in the forest, with about 0.5-I per km^2, and many became feral. In some years, a second set of fircs were set in October or November "to keep the woods open". Fires were generally started along roadsides on ridges and allowed to bum without control, unless property was threatened. In general, ridgetop forests contained much *Quercus coccinea* (ca. 50-60 cm dbh), *Q. velutina* and *Pinus echinata*, with scattered *Q. alba* (to 100 cm dbh) and *Liriodendron*. Most woody understory on ridges was removed, except for scattered Quercus spp. and *Liriodendron*, creating some savanna-like areas. The ground cover of blueberries and other low ericaccous shrubs, grasses and forbs was much thicker than today. Pink ladies' slipper orchids (*Cypripedium acaule*) were more frequent, but yellow ones (C. *pubescens*) were reduced by fire. Composites were more frequent, though concentrated along roads. Birds were generally more numerous, though wild turkey (*Meleagris gallopavo*), like deer (*Odocoileus virginianus*), had been much reduced by hunting.

By some accounts, fire would generally stop near the top of east and north slopes, but it would creep down west and south slopes, creating a scrub forest with such species as *Pinus rigida*, *Quercus marilandica* and *Kabnia latifolia*. However, by other accounts, the fire would often be blown onto cast slopes by prevailing winds, and it would seldom move down west and south slopes. Accounts agree that north slopes seldom burned and often had thick understories of *Acer saccharum* and *A. rubrum* below canopies of *Liriodendron* and *Quercus* spp.

Acquisition of land in DBNF by USFS began about 1933, bringing with it suppression of fire. Burning for forage generally stopped about 1945, though arson increased after 1970. All accounts agree that pine is more common today than 40-60 years ago. Abandoned fields and open woods grew back with much pine and *Liriodendron*. However, soils on and near ridgetops were often so worn-out that only scrub trees, mostly pines and oaks, grew back, and were called- "barrens". Fire was not generally set in this scrubby vegetation, which did not bum well. Remaining barrens of this type have much lichen (*Cladonia*) today, suggesting fire exclusion.

DISCUSSION

The restriction of several rare plants in southeastern Kentucky to roadsides and similar disturbed sites might seem paradoxical, because such artificial habitats arc generally considered to be dominated by common weeds and grasses. Two general hypotheses may explain this phenomenon: (a) these species have invaded the region along roads and other disturbed ground after settlement and forest-clearance; or (b) they arc relicts from natural openings that were maintained largely by fire, with old stable roadsides and adjacent areas offering them a continually open refuge.

The following two arguments favor the latter hypothesis (b), involving fire.

(1) These species generally do not appear invasive or weedy within this region, except perhaps a few of the more frequent ones (also *Digitaria violascens* and *Paspalum setaceum var. longepedunculatum*). Most have never been found in tree-fall gaps, clear-cuts, cropland, pastures, old-fields, artificial wildlife-openings or railroad rights-of-way. Moreover, some of them appear to have declined or disappeared in recent decades: *Aureolaria pectinata*, *Castanea pumila*, *Eryngium yuccifolium*, *Leiophyllum buxifolium*, *Lespedeza capitata*, *Lilium philadelphicum*, *Muhlenbergia torreyana*, *Oenothera perennis*, *Rhynchosia tomentosa* and *Schwalbea americana*. It seems most unlikely that the rarest species, especially those with disjunct records 100 miles or more further south (*Muhlenbergia torreyana*, *Schwalbea americana*) dispersed into Kentucky in the 200 years since settlement.

(2) As noted above, there is considerable evidence that fire has played an important role in the upland forests of this region within the past 100-200 years, before the modem era of fire-suppression. It is possible that death of large trees due to other factors in the presettlement forest created temporary gaps suitable for some of these species. However, rapid dispersal into such gaps would still be required, and, as already noted, these species generally have not dispersed into newly created openings.

Further support for the "fire hypothesis" comes from evidence in other southeastern regions. Even in the Appalachians, fire, mostly set by Indians, was probably a widespread factor maintaining open vegetation before European settlement (Devivo 1990), and early settlers continued frequent burning (Pyne 1982, Otto 1983). On the Coastal Plain, there is evidence of widespread prehistoric, anthropogenic fire (Myers and Peroni 1983). and several of the rare species noted above are typical of fire-maintained vegetation in addition to rights-of-way (S. Orzell, Florida Natural Areas Inventory, pers. comm.). Experimental use of fire in the Southeast has confirmed that barrens or savanna vegetation can be restored and maintained by decades of annual burning (Komarek 1974; Kulhavy and Connor 1986; DeSelm and Clebsch 1990).

One of the closest areas to southeastern Kentucky where extensive fire-maintained areas existed before settlement, and where substantial pieces still survive, is the "oak-barrens" region centered on Coffee County, Tennessee (Svenson 1941; Patrick 1979; DeSelm 1989; DeSelm and Clebsch 1990). The open grassy vegetation there is typically found in relatively flat areas, locally with fragipan soils, on the Appalachian Plateau and its residuum overlying the eastern Highland Rim. Almost all the rare species noted above in southeastern Kentucky occur in that region, plus several rare species of seasonal wetlands. Some species have no records in between, but there are a few well-known barrens sites with rare species on the Appalachian Plateau in northeast Tennessee (Patrick 1979; DeSelm 1989, and unpublished).

Other circumstantial evidence concerns the habitat of the red-cockaded woodpecker ***(Picoides borealis),*** a Federally Endangered Species. This species has declined drastically in Kentucky within the past 50-100 years, and only 10-15 birds are currently known (S. Phillips, pers. comm.). Optimal habitat for this southeastern species appears to be dry, open forest with large pine trees at least 75-100 years old. Given the admixture of hardwoods in DBNF, it is estimated that 400 ha may be required for a stable colony. In Kentucky, logging of large pines has probably been the major cause of the species' decline, rather than understory encroachment, and the birds may be more tolerant of closed forest than those on the Coastal Plain (P. Kalisz, pers. comm.).

Much of the older pine today, including most trees used by the woodpeckers, grew up during the period of frequent burning for woodland-pastures. During the last century (from County Court Deed Books; and Barton 1919), about 5-10 percent of the southern Cliff Section forest was composed of pine, mostly *Pinus echinata*. This percentage is probably much higher that which would exist without any fire. Without fire, current patterns of succession suggest that pine would be largely confined to the driest ridges and clifftops, where, despite extensive recent searches, signs of the red-cockaded woodpecker remain extremely rare. Therefore, before settlement, it seems likely that this species was largely dependent on fires to regenerate extensive areas of pine, especially the relatively fire-tolerant *Pinus echinata* (Martin 1990).

Although this paper focuses on the rare species of moderately dry soils in open habitats, several other uncommon or rare species in this region of Kentucky may have benefited from fire. Some of these are restricted to thin soil around rock outcrops with little or no woody cover, especially ***Arenaria glabra, Crotonopsis elliptica, Oenothera linifolia*** and *Talinum teretifolium*. Fires may have increased the openness of such places. Some more widespread species can persist in the shade of relatively undisturbed pine-oak forests, but clearly do better in the open. Such species persist mostly along trails and logging roads through the forest, but they are locally frequent in small clearings and burned areas. They include ***Cleistes divaricata*** (see also Komarek 1974; Gregg 1989), ***Danthonia compressa*** (see also Lindsay and Bratton 1979), *Isotria verticillata* (see also Baldwin and Wieboldt 1969) and *Porteranthus trifoliatus*. Such species are probably too widespread, well-dispersed and persistent in shade to be good indicators of presettlement barrens, though they may well have occurred in relatively moist or shady variants of such vegetation.

The "fire hypothesis" may also be extended to some uncommon or rare species of seasonally wet, flat ground in thin-canopied forest, thickets, small natural openings or adjacent old-fields. In Kentucky, some are largely restricted to streamheads in the southern Cliff Section: ***Carex joorii, Calamagrostis cinnoides, Calopogon tuberosus, Gratiola pilosa, Lobelia nuttalii, Platanthera cristata, P. integrilabia*** and ***Vernonia noveboracensis***. Others are mostly in broader valleys adjacent to, and in a few cases within, the Cliff Section, mostly in fill-sun: ***Bartonia virginica, Drosera brevifolia, Eryngium prostratum, Gratiola viscidula, Gymnopogon brevifolius, Hypericum canadense, Hypericum crux-andrewsii (Ascyrum stuns), Platanthera lacera, P. ciliaris, Polygala cruciata, Sabatia campanulata, Stenanthium gramineum var. micranthum, Trichostema setaceum*** and *Xyris torta*. Although the wetness of these sites is probably sufficient to maintain small openings, it is likely that fires formerly increased the openness. Forest succession appears to have eliminated some of these species from some sites, and excessive mowing from others (Campbell and others 1989, 1990). They generally have southeastern ranges, especially on the Coastal Plain (see also Braun 1937a,b), where several occur typically in wet, open pine barrens or savanna, where annuals fires promote unusually high diversity (Komarek 1974; Folsom 1979; Walker and Peet 1983).

In more rugged areas of Appalachian Kentucky cast of the Cliff Section, the general lack of rare species suggesting a fire history might be attributed to the general lack of extensive broad, flat ridges where fire might have frequently spread before settlement. However, the endemic *Silphium wasiotensis* (similar to *S. mohrii* of the "oak-barrens" in Tennessee and elsewhere) is restricted to young woods and roadsides on lower slopes, often in areas that have burned (Campbell and Medley 1990). Also, on Pine Mountain, there are small grassy openings with a few rare species that may have been maintained by tire: *Aureolaria pedicularia* (see also *A. pectinata* above), *Baptisia tinctoria* (see also Niering and Dreyer 1989, in relation to fire), *Danthonia compressa* and *Robinia hispida* var. *kelseyi* (see also var. *rosea* above).

In contrast, on more calcareous soils at or just beyond the western margin of the Cliff Section (and in cases indicated by * below, rarely on richer soils in the Rugged Eastern Area), there are several uncommon or rare species of roadsides or other openings that may indicate a tire-history. Some of these species still occur in relatively natural glades, grassland or brushy woods with occasional fire. They include *Apocynum medium* (or *A. androsaemifolium*), *Castilleja coccinea**, *Echinacea purpurea*, *Lathyrus venosus* var. *intonsus**, *Gentiana alba*, *Orbexilum onobrychis** (*Psoralea o.*), *Phaseolus polystachios**, *Polygala senega* var. *senega*, *Salvia urticifolia*, *Silene regia* (Rogers 1941), *Silphium terebinthinaceum* var. *brauniae*, *Solidago ridiga*, *S. speciosa* var. *speciosa** and *Veronicastrum virginicum**. For individual notes, see Campbell and others (1989, 1990; Palmer-Ball and others 1988).

In conclusion, although no definitive statement can be made based concerning the relationship of these rare species to tire within Kentucky, it is likely that fire was a major factor maintaining them on the presettlcment landscape. The exact nature of their barrens habitat will perhaps never be known, but it is likely that large areas were dominated by a relatively open canopy of pines (especially *Pinus echinata*), with some oaks, and much grass (especially Andropogoneae and *Stipa avenacea*). Although the evidence is circumstantial, biological interests, especially in the federally listed species, warrant serious attention to the following suggestions.

(1) It should be recognized that most of these rare species are probably relicts of a natural community that is virtually extirpated within Kentucky.

(2) A search should be made for similar vegetation remaining elsewhere, which might provide clues about ecological factors.

(3) The roadsides and other rights-of-way where these species survive should be managed to maintain their populations.

(4) Attempts should be made to reconstruct the original pine-oak barrens using fire, starting in areas adjacent to the roadsides with these species.

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LODGEPOLE PINE ARTHROPOD LITTER COMMUNITY STRUCTURE ONE YEAR AFTER THE 1988 YELLOWSTONE FIRES

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Abstract-Litter arthropod data was collected every 10 days from nine intensively burned forest stands, five lightly burned stands, and nine unburned forest stands. For burned forest stands (n=540 samples, there were decreases in insect density (87 percent), noninsect density (67 Percent), noninsect taxa (63 percent), and noninsect diversity (20 percent).

Burned stands with greater densities of tree seedlings had significantly ($P < 0.05$) lower arthropod densities. Those with greater densities of standing dead trees had significantly higher arthropod densities and lower diversities. Those with greater densities of litter had significantly higher arthropod densities and lower diversities.

INTRODUCTION

Remarks by E.O. Wilson (1987) at the opening of the invertebrate exhibit of the National Zoological Park in Washington, DC, states the importance of invertebrates to ecosystems. He remarked "if invertebrates were to disappear, I doubt that the human species could last more than a few months. Most of the fishes, amphibians, birds, and mammals would crash to extinction about the same time. Next would go the bulk of the flowering plants and with them the physical structure of the majority of the forests and other terrestrial habitats of the world. The earth would rot within a few decades the world would return to the state of a billion years ago, composed primarily of bacteria, algae, and a few other very simple multicellular plants." A researcher in forest resources, K.J. Stoszek (1988), noted "insects are among nature's most prominent agents of influence. Almost every process in forest ecosystems (for example, nutrient cycling), each developmental phase of forest stands, and every life stage of dominate and subordinate species of forest vegetation are subject to direct or indirect influences of feeding by insects. Without insects, current patterns of plant reproduction, growth, and death would not exist."

Arthropod communities located in unburned, lightly burned, and intensively burned forest stands were studied 1 year after the 1988 fires. The objectives of the study were (1) to determine the effects of fire on habitat structure, (2) to determine the effects of tire on litter arthropod diversity, density, richness and species evenness, (3) to correlate habitat factors (i.e., density of standing dead trees, herbaceous cover, seedling and sapling density, number of fallen trees, and litter biomass with arthropod ecological parameters, and (4) discuss management procedures for conservation of arthropod communities.

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METHODS AND MATEFUALS

Yellowstone National Park (YNP) occupies 8995 km^2 in the northwestern comer of Wyoming with small areas located in neighboring Idaho and Montana. Elevations range from 1500 m to over 3400 m. Present climatic patterns produce generally long, cold winters and short, cold summers. Annual precipitation ranges from 75 mm to 200 mm (Houston 1982).

Forested areas occupy 79 percent of YNP. Lodgepole pine (Pinus contorta) comprises 81 percent of the forested areas at elevations of 2300 m to 2600 m in which we have research sites (Houston 1982).

Nine randomly chosen unburned forest stands of at least 5 ha were paired with nine intensively burned forest sites and five lightly burned forest stands. Lodgepole pine stands were considered intensively burned if (1) all the trees lacked branches, (2) there was very little ground litter biomass, and (3) fallen trees were burned to some degree. Lightly burned forest stands produced as halo fire stands around intense fires, had a soil litter layer, fallen trees which were singed, and trees with branches and needles burned and brown but still intact.

Arthropods were collected during July to mid-September, 1989. Litter was collected from five 0.5 m^2 quadrates every 20 m (the initial starting point was randomly chosen each sampling period) along two of the three transects (the transects were randomly chosen before each sampling period) every 10 days in each forest site. Each litter sample collected for arthropod extraction was stored in a zip-lock plastic bag for not more than 12 hours before processing. These samples were then placed in Berlese funnels for 24 hours. Arthropods extracted from litter were stored in 70 percent ETOH.

Density of both live and standing dead trees, seedling density, fallen trees, and percent herbaceous cover were determined by making counts within a l-m swath along each transect on each site. Litter samples, which were collected for biomass weight, were placed into paper bags. Litter samples were oven dried for 24 hours at 25 °C and weighed to the nearest 0.01 gram.

Ecological parameters were of arthropod and annelid density, species richness, species evenness, and diversity. Density was based on the number of individuals per quadrate, species richness was based on numbers of individual species (Southwood 1978); evenness and diversity (Shannon-Weaver using the natural log) were analyzed using a program developed by Kotila (1986). Simple linear regression (STATGRAF) was used to determine biotic correlations at the $P < 0.05$ level of significance. Student's t-tests was used to identify statistical differences ($P < 0.05$) between ecological parameters for unburned, lightly burned and intensively burned stands.

RESULTS

Forest Arthropod Community

Undisturbed lodgepole pine forest stands (trees > 100 years) contained higher biotic variables than fire disturbed lodgepole pine stands (table 1). Litter biomass was decreased 19 percent in lightly burned stands and 62 percent in intensively burned pine stands. Downed trees decreased 24 percent in light bums and 30 percent in intensively burned pine stands. The percentage of herbaceous cover decreased 96 percent in light bums and 86 percent in intensively burned forest stands.

Litter mites were the dominate arthropod fauna in both density and species richness (table 2). Insect richness declined 63 percent in lightly burned mature lodgepole pine stands and 67 percent in intensively burned stands of that type (table 3). Insect community density was affected even more than richness with a 92 percent loss in light bums and an 82 percent dcchne in intensively burned stands. Community diversity declined more in light burns (66 percent) than in more intensively burned sites (40 percent).

Noninscct richness declined in both lightly burned stands (71 percent) and intensively burned stands (66 percent) (table 3). Community density declined 89 percent in light bums but only 67 percent in intensive bums. Noninsect diversity declined most in light bums (65 percent) and less (32 percent) in intensive bums.

Table 1. Biotic variables for unburned, lightly burned, and intensively burned lodgepole pine stands

Variable	Fire disturbance: None	Light fire	Intense fire
Litter biomass ($g/1m^2$)	153.20a[1,2]	123.60a	57.801,
Trees with needtes >1.82m ht. ($no./100m^2$)	42.82a	45.20a	0.00b
Tree seedlings <15.24cm ht. ($no./100m^2$)	24.42a	7.80b	17.64a
Trees with needles >15.24cm ht. and <1.82m bt.	40.50a	0.00b	0.00b
Standing dead trees no. needles and > 1.0m ht.	12.09a	14.20a	33.08b
Logs	38.11a	29.00a	26.73b
Herbaceous cover (percent)	50.19a	1.90b	7.24b

[1]Means followed by different letters differ at the P<0.05 level of significance according to Student's t-test.

[2]n=27 transects.

Table 2. Major arthropod groupings (percent) in unburned, lightly burned, and intensively burned lodgepole pine stands

Arthropod group	Fire disturbance: None	Light fire	Intensive fire
Density			
Mites			
Oribatids	42[1]	15	38
Others	21	48	47
Insects			
Collembola	24	31	5
Others	13	8	10
Species			
Mites			
Oribatids	16	23	17
Others	38	23	37
Insects			
Collembola	25	31	30
Others	23	23	16

[1]n=540 samples.

Table 3. Ecological parameters (avg/m²) for litter arthropods in unburned, Lightly burned, and intensively burned Lodgepole pine stands

Parameter	None	Fire disturbance Light fire	Intensive fire
		Insects	
Richness	18.80a[1,2]	7.00b	6.10b
Density	154.70a	13.00b	19.40b
Evenness	.82a	.97b	.87a
Diversity	2.39a	.82b	1.43c
		Noninsects	
Richness	27.80a	a.00b	9.40b
Density	222.70a	25.00b	74.10c
Evenness	.76a	.78a	.75a
Diversity	2.52a	.87b	1.70a

[1]Means followed by a different Letter differ at the $P<0.05$ level of significance according to Student's t-test.

[2]n=540 samples.

Factors influencing litter arthropod ecological parameters in unburned, lightly burned, and intensively burned forest.

The following are biotic variables which have been correlated ($P < 0.05$) with litter arthropod ecological variables.

Unburned forest stands.

Diversity was negatively correlated with tree seedlings ($r^2=0.91$).

Diversity was positively correlated with trees over 1.84m in height ($r^2=0.66$).

Diversity was negatively correlated with standing dead trees ($r^2=0.66$).

Density was positively correlated with logs ($r^2=0.66$).

Lightly and intensively burned forest **stands.**

Density was negatively correlated with tree seedlings ($r^2=0.66$).

Discussion. Biodiversity conservation is currently an important and intensively researched topic. Temperate regions should not be given any less attention than tropical regions. Temperate systems may not have the highly diverse fauna as found in the tropics, but temperate systems are important to world-wide biodivcrsity (Piclou 1979). Management decisions which may affect temperate forest arthropod communities cannot be based on research conducted in the tropics.

What is the goal of biodiversity conservation? Why should we study arthropod diversity and community structure? The goal may be to maintain a system by prescribed bums or the goal may be to allow system succession by naturally caused fire. The goals are determined by forest and national park resource managers. A number of studies indicate that natural disturbances are critical in maintaining species diversity (Horn 1974; Levins 1968; Tilman 1982).

E.O. Wison's address states the importance of understanding arthropod communities. Arthropod community structure is changed by habitat structure manipulation (Christiansen and others 1989; Schowalter 1985).

Forest Arthropod Community Structure

Habitat structure.

Habitat structure is important for arthropod community structure. One theory of preservation and conservation practices is that natural systems should be left alone and not cleaned except by natural phenomenon (Franklin 1988). Prescribed fire is one approach that is used to modify forest habitat structure. The degree of fire disturbance most certainly affects various forest components. Comparsons of unburned, lightly burned, and intensively burned lodgepole pine stands in Yellowstone National Park lead to the following conclusions.

If litter biomass and survival of older trees are important to conservation practices for the Yellowstone region, than light bums are a good strategy. However, intensively burned stands contained more tree seedlings, standing dead trees, and herbaceous cover than did lightly burned stands. Thus, intensive burns provide more potential resources, at least in the short term, for arthropod biodivcrsity than lightly burned pine stands provide.

The greatest loss of litter arthropod diversity in lodgepolc pine stands occurred in lightly burned stands. Lightly burned pine stands also had lower arthropod community densities than unburned or intensively burned stands had. These results are interesting since the largest litter biomass loss occurred in intensively burned stands. It is possible that light bums disrupt trophic webs enough to eliminate many arthropod's food resources but do not disrupt niches enough to make invader species competitive. lntensive bums drastically decreased litter biomass, which may have disrupted niche and food resources to such a degree that invader arthropod species can compete. Disturbance may benefit species diversity by opening habitats to fugitivc species that ephemerally invade disturbed habitats (Conncll 1978; Sprugal 1985).

Influence of habitat on arthropod ecological parameters. Litter arthropod diversity in unburned and burned lodgepole pine stands was lower in stands with higher tree seedling densities. It may bc that stands with higher than average seedling densities contained lower densities of mature trees. Thus, habitats, in both unburned and burned stands might have been more open than usual and might have had lower litter arthropod diversity for that reason. Many arthropods, such as Collcmbola (springtails) and many fitter mites, arc sensitive to moisture differences which could be affected by more open habitats. A study by Pontailler (1979) described moisture fluctuation due to tree thinning and establishment of forest gaps. A second possibility is that pine seedlings have not yet produced enough litter to support litter fauna. Both of these explanations are supported by the finding that higher than average arthropod diversities occurred in dense stands of mature lodgepole pines. These stands had closed canopies and greater litter biomass than found in dense stands of tree seedlings. Studies by Schowalter and others (1981) and Seastedlt and Crossley (1981) showed that many arthropod species became more abundant as debris increased, and that numbers of insect species that inhabit leaf litter increased enclosed-tree-canopy sites.

Lower arthropod diversity in unburned pine stands was correlated with greater numbers of standing dead trees. The occurrence of lower arthropod diversity in older stands may be inconsistent with the view that old-growth forests support high arthropod diversity (Franklin 1988). Older stands contained more fallen trees than did younger stands. A positive correlation existed bctwccn higher arthropod community density and grcatcr numbers of logs. This correlation indicates that old-growth pine stands support a few high density litter arthropod species.

Litter arthropod community density correlated negatively with pine seedlings. Higher pine seedling densities were found in intensively burned stands, which also had lower litter biomass. It is biologically significant that lower arthropod density would be correlated with reduced litter biomass; reduced litter biomass means reduced food and habitat resources for litter-inhabiting arthropods. Increasing densities of standing dead trees were corrclatcd with higher arthropod densities but lower diversity.

Arthropod density and diversity were low on burned sites with large numbers of logs. Lightly burned pine stands, which contained more litter biomass than intensively burned stands, contained higher arthropod community densities and lower diversity. This implies either that a few species were able to adapt and multiply in burned stands or that invader arthropod species entered the stands from nearby unburned pine stands.

Conclusions About Arthropod Diversity and Conservation

Habitat loss is one of the major factors in diversity reduction (Pimm and Gilpin 1989). The present study showed that litter arthropod diversity was lower in burned forest habitats than in unburned stands. However, species evenness increased in disturbed stands. Species evenness is an index describing the distribution of species in a community. Thus, an increase in species evenness suggests that competition is among more species than just those which compose the majority of the community. Chesson (1985) states that competing species that cannot. coexist in a constant environment may cocxist in the presence of environmental variation.

If the immediate management goal of burning is to maintain high litter arthropod diversity, either for biodiversity conservation or ecosystem functions, then high intensive bums are the suggested control method. Long-term management may be a different matter for diversity. Mature unburned lodgepole pine stands support greater diversity than found in very young (i.e., seedlings) pine stands. However, litter arthropod diversity in mature forests with high densities of standing dead trees is not as great as litter arthropod diversity in somewhat younger forest stands. These same older forests support greater densities of fewer litter arthropod species, however.

Results of our study indicate the complexity of conservation and management of litter arthropods in lodgepole pine forests. Our research has only provided preliminary data. There is clearly a need for more habitat and arthropod community structure interaction research on an ecosystem level if forest management goals arc to be achieved.

ACKNOWLEDGMENTS

We wish to thank C. Legg for her help in the laboratory. Wc also thank the University of Wyoming's National Park Research Center for support.

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SOME THOUGHTS ON PRESCRIBED NATURAL FIRES

Jack D. Cohen*

Abstract-Wildland tire is a significant component of nearly all North American ecosystems. High intensity, stand-replacement tires are normal in certain ecosystems, especially in the northern Rocky Mountains. Wilderness fire managers are obligated to let tire operate as a natural influence to the extent that this is possible. Where wilderness areas incorporate stand-replacement-type tire ecosystems, ecologically significant prescribed natural fires must reach stand-replacement fire intensities. However, because weather forecasting capabilities are limited, fire managers are unable to predict whether prescribed natural tires will escape prescribed boundaries. Moreover, the effectiveness of suppression actions decreases as wilderness tires increase in size. Thus, fire managers face the dilemma of managing for natural tire influences on ecosystems, with the consequence of increasing the potential for escaped fue situations.

Typically, thunderstorms develop over the northern Rocky Mountains in late spring and early summer. The following hypothetical situation begins on June 29, with lightning strikes peppering a broad area of the Intermountain region of the Northwestern United States.

A large wilderness area in the northern Rocky Mountains has a prescribed natural fire plan under which a few fires are allowed to bum within its boundaries. These fires are prescribed so that natural fire will continue to be a significant ecological influence in the wilderness area. Investigations have determined that this ecosystem normally experiences stand-replacement fires. Thus, to be ecologically significant, the prescription permits some high intensity burning, i.e., crown tires.

The fire managers use criteria based on the National Fire Danger Rating System's Energy Release Component (ERC) to decide whether or not to suppress lightning ignitions. However, using the ERC has had its uncertainties. Last season, the spring and early summer values were well above normal. All fires were suppressed. But the July and August weather had turned rainy and cool, and ERC's were below normal.

In the current season, the ERC's have closely followed normal values. The 30-day forecast predicts normal temperatures and precipitation for July. On July 1, a reported lightning fire is designated as a prescribed natural fire.

However, by July 11, there is a distinct trend toward hot, dry, weather conditions. From the end of July to August 5, the ERC's are consistently and increasingly above normal. The 30-day forecast for August calls for above-normal temperatures and below-normal precipitation. Although no new natural fires are allowed to bum, the July 1 fire continues. Even with the hot, dry conditions, the burned area is less than 10 acres (4 ha) as of August 5, well within the prescribed limits for size and proximity to boundaries. At this point, the prescribed natural fire is not yet an ecologically significant event.

On August 5, the National Weather Service forecasts the possibility that a dry cold front will move through the area in 3 to 5 days. The forecast does not change the designation of the prescribed natural fire. On August 8 and 11, two dry cold fronts produce strong winds. As of August 13, after the cold fronts pass, the fire perimeter contains over 8,000 acres (3,200 ha). Numerous spot tires and crowning occur. Within a period of 5 days, the fire has become ecologically significant.

The fire managers in this scenario have executed the policy of allowing fire to assume a more natural role in the wilderness and have accomplished an ecologically significant fire. But now they must deal with the uncertainty of the remainder of the fire season and thus the possibility that their management fire will escape predetermined boundaries. If undertaken at this time, an effective suppression action must be capable of containing an active 8,000-acre fire.

The Yellowstone Fires of 1988 represent one possible result of the implementation of a Park Service (U.S. Department of Interior) and Forest Service (U.S. Department of Agriculture) policy that attempts to return historic fire occurrence to designated areas. Research has established that fire is a significant component of most North American ecosystems (Amo and Brown 1989; Wright and Bailey 1982). The policy that allowed some tires to bum in the Greater Yellowstone area reflects management's desire to maintain processes vital to the ecosystem's existence.

An effective natural fire policy provides for the occurrence of ecologically significant fires. That is, the policy prescribes the occurrence of periodic fires that achieve a range of intensities over a large enough area so that tire will continue to be a significant ecosystem process. Normal ranges of ecologically significant fire frequencies and intensities vary

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among ecosystems. The spectrum ranges from ecosystems with frequent tires of low intensity to those with infrequent fires of high intensity (Bacon and Dell 1985). Much of the Greater Yellowstone area as well as other wilderness areas in North America have ecosystems where ecologically significant fires are of the infrequent, high-intensity type that result in the replacement of all surface vegetation in a portion of the burned area (Brown 1989; Heinselman 1985).

A problem arises when a natural fire program that requires high-intensity fires is implemented. Wilderness areas have administrative boundaries that separate the wilderness landscape from landscapes where uncontrolled high-intensity tires are undesirable. Much of the concern during and after the 1988 wilderness fires (the Yellowstone Complex and the Canyon Creek Fire in the Scapegoat Wilderness) has related to natural fires crossing wilderness boundaries and threatening or destroying outside values. The wilderness fire manager's goal is to let tire operate as a natural process and at the same time contain it within prescribed boundaries. But this goal can create dilemmas. The following discussion examines, in terms of scale characteristics, the various elements that contribute to this potential fire management dilemma. Although this discussion may relate to the initial bum-or-suppress decision, it does not specifically address that decision. Rather, this discussion focuses on the limitations of managing an ongoing prescribed natural tire.

THE DILEMMA

A natural prescribed fire may bum for several months. Due to the limits of weather predictability, a fire manager cannot predict fire behavior on the basis of extended-range weather forecasts. Thus, it is possible that any prescribed natural fire will exceed its prescription. There are ecosystems in which fires must be large and intense if they arc to be ecologically significant -- and such tires have the potential to out-scale fire suppression capabilities. By the time a fire manager realizes a fire will exceed its prescribed limits, suppression efforts may not be effective. Thus, managing for natural fire processes in wilderness areas also enhances the potential for escape.

ELEMENTS OF THE DILEMMA

The basis for the wilderness fire management dilemma can bc found in the response of fire behavior to the weather, limitations of weather predictability, and a limited fire suppression capability.

Wildland Fire

The conditions associated with fast-spreading, high-intensity tires (e.g., the 1988 Yellowstone Fires) are drought and high winds. Drought conditions promote drying in vegetative fuels, both dead and living, thus making the fuels more flammable. High windspeeds increase spread rates, which increases intensities and spot fire ignitions. Although drought conditions are commonly referenced in discussions of severe fire situations, drought alone is not sufficient to produce large, high-intensity tires. High windspeeds are required to drive fire to extreme behavior. Thus, given sufficient fuel loadings, severe fire situations depend on two different aspects of the weather.

Weather

Atmospheric phenomena can be generally characterized by distinct time scales. Dutton (1976) describes the temporal characteristics of several weather features as follows:

Atmospheric Dhenomena	Time scale
Planetary waves	1 yr- 1 mo
Synoptic systems	1 wk . 1 day
Mesoscale features	1 day . 1 hr

Trends in weather variables such as average monthly temperature and precipitation are related to the planetary scale, whereas, a specific day's temperatures and precipitation are related to synoptic and smaller scales. Drought conditions arc associated with planetary waves, and high winds are associated with synoptic systems and mesoscale features. Thus, weather conditions operating on two different atmospheric scales contribute to the conditions that produce severe fires. In general, severe fire behavior potential develops over a month or more as fuels become drier (drought), but has an episodic time scale of one week to one hour (high winds).

Weather Forecasting

The ability to forecast the weather depends on the atmosphere having the characteristic of predictability (Lorenz 1968). There arc indications that the atmosphere behaves in highly nonlinear ways, and that this limits weather predictability (Lorcnz 1963, 1969; Gleick 1987; Tennekcs 1988). In a mathematical modeling context (which is largely how weather forecasting is done), this non-linearity results in a loss of connectivity over time. That is, the state of the atmosphere at a given time (the initial conditions) loses its predictable relation with the state of the atmosphere at some later time.

I think the general conclusion can be drawn that weather conditions cannot be specifically predicted over time periods longer than their associated atmospheric time scales. That is, a forecast of weather variables associated with the mesoscale to synoptic time scale (e.g., specific precipitation or wind events) does not apply beyond about 3 days. Forecasts covering several weeks to several months, which predict general trends in weather variables (e.g., departures of mean temperatures and precipitation amounts from normal) are associated with planetary time scales. Extended-range forecasts that provide likelihoods of shorter time scale weather conditions are not an exception. in this case, the statistical characteristics associated with a planetary scale situation are described rather than a forecast of specific deterministic information about a smaller time scale condition.

Effective Fire Suppression

To be effective, a fire suppression action must be scaled to match a fire's spread and intensity. A fire suppression action must be capable of constructing a quantity of fire line comparable to the spread rates and fire growth, and the fire line must be sufficiently wide to prevent crossing caused by flames and firebrands. A fire suppression action can be ineffective because insufficient resources are committed to it or because the fire overpowers a maximum practical suppression effort.

Managing for ecologically significant, high-intensity fires requires that these management fires reach large sizes relative to fire sizes normally controlled as wildfires. To further complicate matters, weather forecasting limitations prevent the long term prediction of a fire's eventual size and intensity. Thus, an effective suppression force must be capable of constructing enough fire line to contain portions of or all of a management fire that could be potentially of high-intensity and extend over an area of several thousand acres. However, the larger a fire is, the less effective is a suppression action. That is, the fire's growth and/or intensity is more likely to out-scale suppression efforts. Commonly, after wildfires become large, suppression is not effective until a sufficient weather or fuel change occurs, causing a fire to return to a scale where suppression is effective. In the case of the 1988 Greater Yellowstone Fires and the Canyon Creek Fire, a sufficient weather change (to cool and damp) did not occur until mid-September.

CONCLUSION

Large, high-intensity fires are a natural occurrence in some ecosystems. Where these ecosystems occur in wilderness areas, fire managers are obligated to maintain natural processes for the existence of the ecosystem. Prescribed natural fires are a reflection of this obligation.

Because a prescribed natural fire can bum for an extended period, specific weather conditions cannot be selected. The fire burns under whatever conditions occur. But, for an ecologically significant fire, the weather must accommodate a high-intensity fire for some period of time. If, however, severe burning conditions are too persistent, the fire can exceed prescribed limits.

The fire manager does not have extended-range forecast information at the scale of specific fire weather conditions. This is due to the lack of predictability of the atmosphere at the scale responsible for high wind speeds. Thus, the eventual state of the prescribed natural fire cannot be predicted.

Fire suppression actions required to contain a prescribed natural fire within designated boundaries may not succeed. The achievement of an ecologically significant (large size, high intensity) wilderness fire reduces the likelihood of effective fire suppression, should suppression actions become necessary. Thus, the maintenance of fire as a natural process can lead to wilderness fires that escape prescribed boundaries to become costly wildfires. This is the essence of the prescribed natural fire dilemma.

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USE OF THE 1990 CENSUS TO DEFINE WILDLAND URBAN INTERFACE PROBLEMS

James B. Davis'

Abstract-Predicting the movement of people into rural wildlands previously has been limited to studies of population and housing growth in counties or other large geographical areas. In these studies, the areas of high tire danger that contain dispersed rural housing cannot be distinguished from the areas less vulnerable to wildfire (small towns and adjacent urban areas) because the data and the analysis procedures associated with the *1980 census* does did *not* give sufficient information about rural and wildland areas. Analysts studying rural demographics supplement census data with information From such sources as local tax assessors' records or building permit files.

INTRODUCTION

If we are going to effectively manage the wildland urban interface fire problem we need to know where people live, work and play within the interface area. Wc also must know something about their knowledge and attitudcs toward the environment and its mnnagcmcnt and protection (Davis 1990; Irwin 1987, 1988). We need to know where people obtain their knowledge and how thcy formulate their ideas and attitudes--a rapidly developing ficld called psychographics. If WC know what our customers want and what benefits &hey perceive, we can kc more effcctive in communicating with them.

We can help develop a firesafc community by influencing the location of housing within fire-prone areas and help regulate the design of homes and other structures in those locations most likely to bum. The intcrrclationship between the factors that result in a choice of a building site and the factors that matter in fire spread and suppression, such as vegetation type, slope class, and aspect, access and proximity to water and roads need to be understood. Reasonable estimates can be made of housing to be built five or more years in the future whcn these factors arc included in population projection modcls (Bradshaw 1987). These estimates can be mapped and overlaid on fire risk and hazard maps to allow a fire manager to display to local policy and planning officials detailed information on the areas likely to bc threatened by future wildfires and the homes and population that will be at risk unless mitigating measures are taken. The ability to display this information will enable fire managers to be proactive rather than reactive in their contacts with public leaders.

PROBLEMS

Rural Growth

Many foresters are surprised that they must cope with the most rapidly changing and dynamic segment of our nation's population. Although the increase in rural population has slowed somewhat since the 1970's when rural counties were growing three times as fast as the urban counties, population growth in many of the Nation's forest and range counties continues to exceed urban population growth and will probably continue to do so past the turn of the century (Long 1983; Rice 1987).

California, for example, has traditionally doubled its population every 20 years since statehood. However, it will not double its 1970 population again until at least 2020, a period of 50 years. Yet, the population of I.5 of its counties--all forested with the exception of one--is continuing to double in 20 years or less--tEe areas that are increasing in population most rapidly are those most prone to wildfires.

Problems Beyond Local Control

Another problem is that rural area population dynamics are influenced by socioeconomic factors well beyond the borders of the area involved. California has long appealed to Americans who move for one reason or another. By the late 1960's, however, California gained migrants from fewer states than previously, and it began to send migrants to Oregon, Washington, and Nevada, as well as to Oklahoma and Virginia (Hcrbcrs 1986). Between 1975 and 1980, California had a net loss of 420,000 people in migration exchanges with its ten western migration partners. But this loss was offset by a net gain of 534,000 people from the rest of the U.S., chiefly from the Northeast and Midwest because of the decline of the iron and automobile industries and because young professionals were being attracted to California's aerospace, computer and other high-tech industries, resulting in a total gain of 114,000 to California's population.

In the late 1970's Oregon residents sported bumper stickers asking Californians to visit but not to stay. By the mid 1980's such fears were allayed because Oregon began exporting people to California due to the decline in the woods products industry and rising unemployment in the Northwest (Sweeney 1979).

This continually changing economic situation has made population growth projections into fire prone areas difficult.

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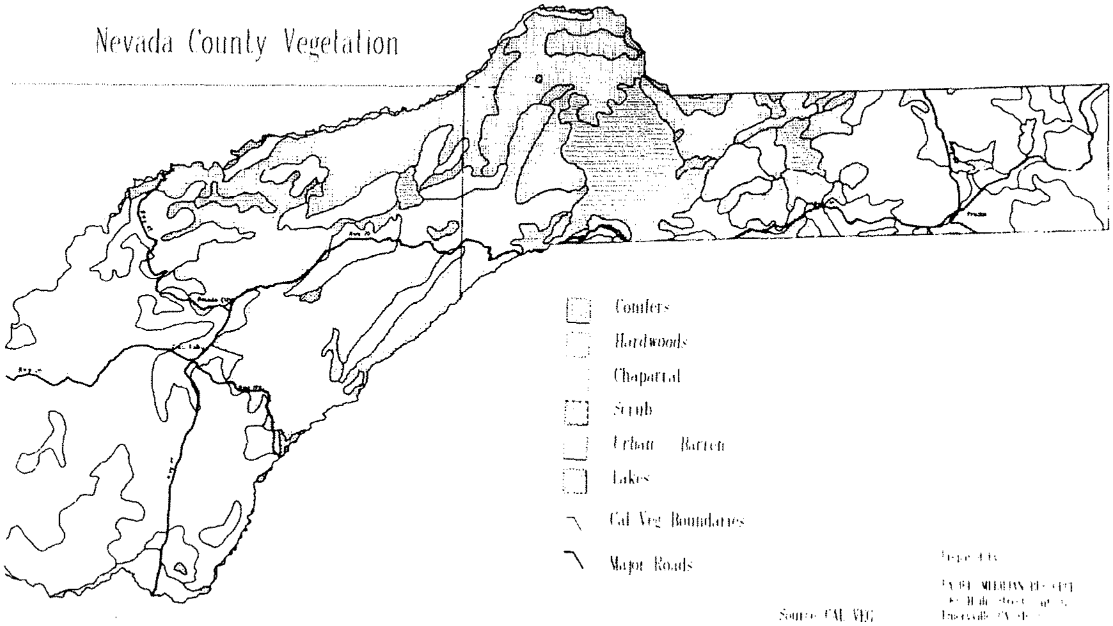

Figure 1--Broad categories of vegetation found in Nevada County. The vegetation changes from mostly grass, chaparral, and hardwoods at the lower elevations in the western part of the county, to heavy stands of conifers in National Forests covering the higher elevations in the east. Major highways are indicated on the map as well.

Limitations on Population Growth Estimation

For foresters, the 1980 census has not been helpful because of the poor level of resolution in wildland areas; the tracts are very large areas in most instances and often include diverse demographic characteristics within a tract. For example, the twelve tracts in Nevada County, California range from thinly inhabited National Forest Land to urbanized residential areas (Fig. I).

Analysis Models Are Limited

While knowing "what is there now" is difficult, it is only part of our battle. We need to know "what will be there in the future." The census information does the local planner little good unless it can be interpreted for his or her needs in both a temporal and spatial manner. This is a particularly difficult problem in the wildland urban interface because there have been few predictive models. Although such models exist for urban areas--the projections needed for the construction of a shopping center, for example--we know of no case in which they have been used to predict the location and number of households at risk from wildland fire. Population analysis in rural areas has usually been concentrated on estimating the movement of population to urban areas as farm and lumber industries decline, or in predicting the broad overall change in a county level population.

Extensive literature exists on the population change of particular rural counties, and permits extrapolation of this information to many potential rural growth situations (California Department of Forestry and Fire Protection 1988). However, virtually no micro-level studies have been done to understand where people in rural areas choose to live (Lindhult and others 1988). Much of the rural demographic research--if it has been done at all--has been in the northeastern United States where counties are generally small and relatively homogeneous. Counties in the western United States, on the other hand, are large, heterogeneous, and require a much more rigorous analysis.

Regional development models are similarly underdeveloped outside of metropolitan areas. Although they have been used to establish the regional growth within urban areas, they are less adequate for rural areas for which data and economic conditions are less well understood (Befort and others 1988). The interrelation between economic conditions and housing development has been posited in the literature, and evidence in rural areas indicate that people often commute long distances in order to take new jobs. Economic conditions go hand in hand with changes in the housing supply, but little is known about how economic growth affects the distribution of housing in wildland interface areas which often include a high number of retired persons.

RESEARCH ON WILDLAND DEMOGRAPHICS

The Riverside Forest Fire Laboratory, in cooperation with the University of California's Institute of Governmental Studies in Berkeley, is making an effort to solve the problem of wildland urban interface demographics. Dr. Ted Bradshaw is the principal investigator from the University for the project.

The cooperative effort is attacking the following three questions and is using Nevada County, California, as a field laboratory:

1. Can we identify reliable sources of demographic data for the wildlands.
2. Can we develop models to define and better understand the movement and eventual settlement of people in our wildland areas?

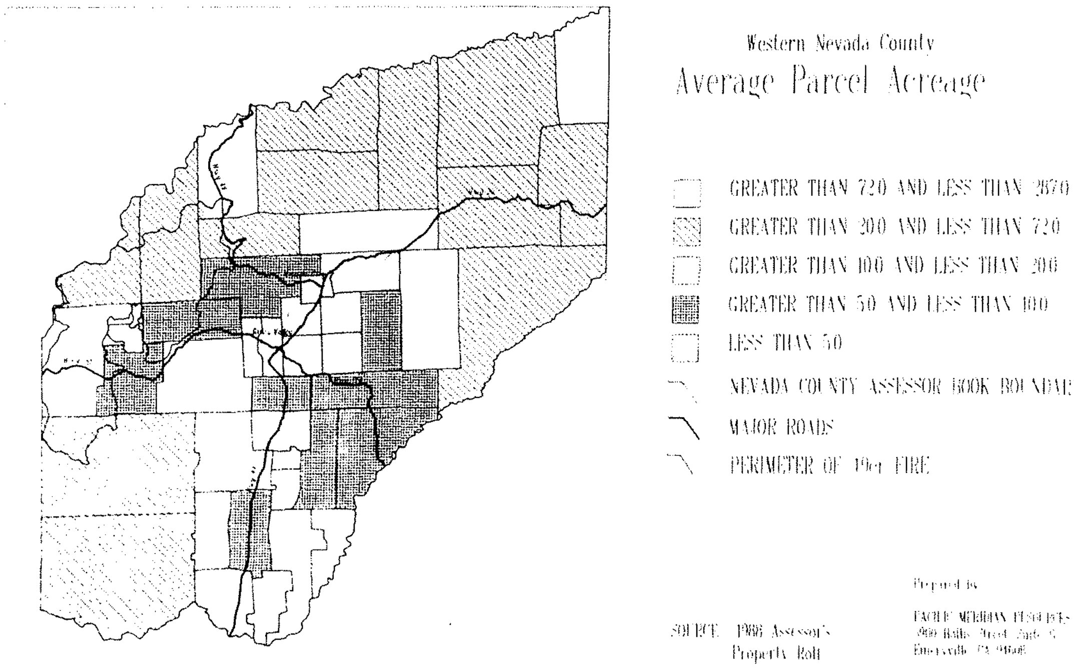

Figure 2--Patterns of population from the 1980 census. These data demonstrate the capacity of GIS to translate data on total population into density per acre. Current population densities arc significantly higher due to an average annual county population growth of 6 percent since the census.

3. What must we do to take advantage of the much higher resolution data that will be available from the 1990 census and the commercially available data and analysis software that will spin off from it?

Identification of Data Sources

The data we are using in our research come from various sources with different geographical boundaries:

- The vegetation data are from a statewide species mapping effort called "CAL VEG" based on land and aerial data.

- Fire severity zones in Nevada County are based on a series of maps prepared by the California Department of Forestry and Fire protection and determined from topographical, climatological, and fuel considerations (Phillips 1983).

- Census data from the 1980 census arc given by enumeration district (Fig. 2). In selected areas we are examining the existing pattern of settlement by using census data, current updates, and aerial photos, when possible, and are seeking methods to estimate the accuracy of the statistical data. A major source of information has been county tax assessor's records. We have found that data on building permits arc a key factor in determining patterns of growth since 1980.

- Property rolls contain information on the size, use, and value of property in the county. Thcsc data arc shown for geographical areas defined by the county assessors' books. Current research has enabled us to go from book to page level--a degree of resolution that normally will include 50 or fewer households. Also obtained from the assessors' records arc data on the per acre value of improvements. The areas with the highest values usually arc associated with residential construction and much of it is located in areas of high fire hazard.

Using assessors' data on average parcel size, trends in dcvciopmcnt arc easy to identify. Further analysis will show the characteristics of these parcels with regard to tire dangers, roads, and physical amenities.

Our rcscarch is extending the field of investigation to include groupings much smaller and more specific than the usual national or regional aggregates. We arc constrained by neither political nor administrative boundaries such as cities, villages, or natural regions, but our rcscarch is allowing us to study levels of human categories that arc not territorially well delincd (for example wildland urban rcsidcnts).

Development of Models

While current population and attitudinal information should be useful to foresters and fire managers, our long-range objective is to dcvclop predictive models. Wc arc conducting analyses to estimate parameters of various models that include growth as well as attitude toward forest land and its management and protection. The attitude and growth are related to factors such as vegetation type, slope, aspect, attractive physical features, proximity to urban settlements, employment, subdivisions, infrastructure, roads, etc. We will determine whether our models accurately estimate growth at reasonable levels of confidence by comparing our model determinations with patterns that have occurred over the past 5-10 years and with current building permit issuance and opinion surveys. We will select the most descriptive model and refine it as needed.

How do we take advantage of the 1990 Census?

Although we are making headway with non-census data, the 1990 census data, when they become available in 1992 or 1993, promise Lo create a "desktop computer revolution." Along with benchmark demographic data, the census includes a survey of housing and housing units. For our purposes the census of housing will be very important because it describes the location and demographic characteristics of the people living in each housing unit. It also details ownership, condition. and value of the property (Kirchner and Thomas 1989).

The 1990 census data will be available on four census computer "summary tape files" (STFs):

- STF- I and STF-2 will contain data on household type, race, sex, age, marital status, and detailed information on the residence obtained from the "short" census questionnaire sent to every home in the country. This information for the first time will provide good resolution in rural areas and will be traceable to the equivalent of a city block.

- STF-3 and STF-4 will contain the same basic data as the first two summary files, plus the information from the "long" census questionnaire. The long form will be answered by a 17 percent sample of households. This form will contain demographic information that fire planners may need, including income, educational background, migration, language, type and place of employment and housing information such as availability of 3 telephone.

In fact, this high level of resolution has created somewhat of a problem to the Census Bureau in maintainig confidentiality. In rural areas it might be possible to identify the source of some data--the income of a single ranch family for example- -and the Bureau has had to incorporate methods to screen out such information.

One objective of our analysis will be to determine whether the detailed 17 percent survey will give us all of the demographic information we need in very sparsely populated areas, or whether we will still have to depend to some degree on other sources such 3s building permits and assessors' records. Much of our research will be aimed at correlating information that we can obtain from census records with factors that cause people to move into the interface area.

Micro Computers and Laser Discs

Although the 1990 census data will be available in several forms from hard copy reports to computer tapes, the most exciting improvement for computer-wise foresters will be that the information will eventually be available on laser read-only memory compact discs known as CD-ROMs, reflecting a decade of changing computer technology. By putting census data on laser discs, the Census Bureau will make great quantities of information available to the individual with a good personal computer and the computer capability to use the information. Compact discs have enormous potential because each 4-S/8-inch disc can store as much information as three computer tapes or 1,500 floppy disks. An expensive mainframe computer is not required to process information contained on a compact disc. However, one problem may be "data overkill." There is likely to be so much information that determining what information to use and how to use it efficiently will be difficult.

With the addition of a laser disc reader--available to almost cvury forestry or fire management headquarters office for less than $1000--a microcomputer can become a desktop information system capable of printing STFs on demand.

However, despite the obvious advantages of compact discs, the Census Bureau is not releasing them as the basic medium for distributing 1990 census data because as yet, there is no standardization in disc technology. Until there is standardization as well as user-friendly software, much computer skill will be needed to use this new technology. To help users get started, the Census Bureau is making three CD-ROMS--Test Discs 1 and 2, and the 198.5 American Housing Survey--available now. The discs sell for $125 each and can be ordered from the Bureau's Customer Service Office.

TIGER Files

A recent innovation, that may prove very valuable for model development and testing and for understanding wildland urban interface population dynamics, is the automated mapping system known as TIGER (Topologically Integrated Geographic Encoding and Referencing). This system has enabled the Census Bureau, working with the U.S. Geological

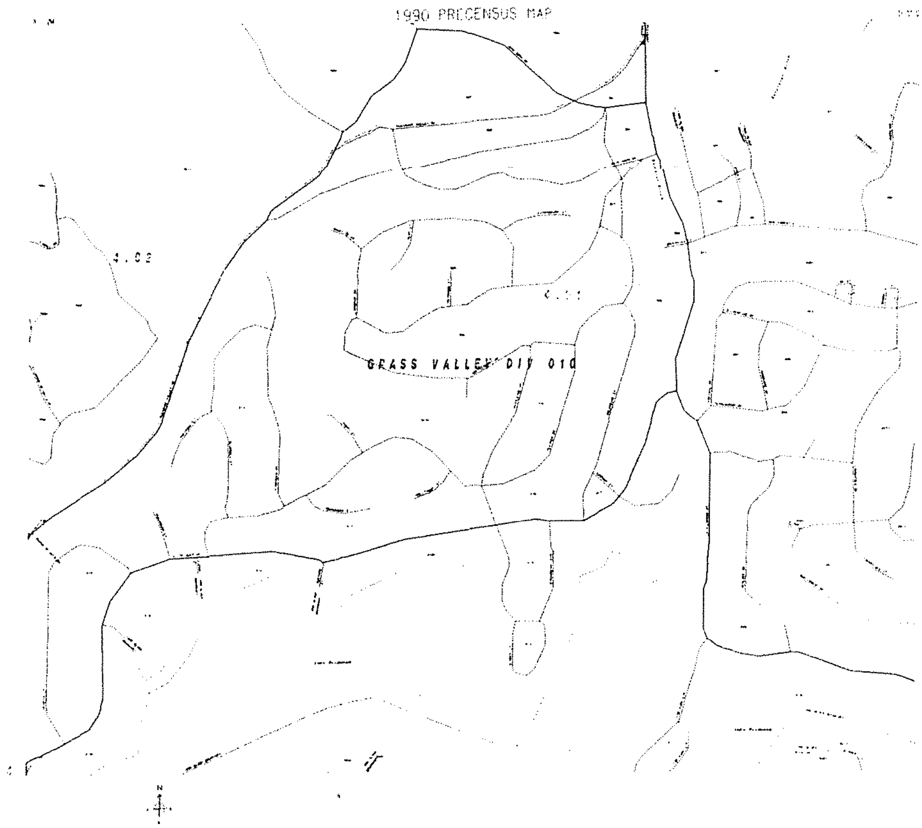

Figure 3--Section of preliminary map printed from the TIGER files. Map scale is about 4 inches to the mile. For many locations TIGER is capable of generating the most detailed and up-to-date maps available.

Survey, to develop computerized maps covering the entire United States (U.S. Department of Commerce 1985). TIGER is essentially a digital street map of the country (Fig. 3). The TIGER process uses geographical information system (GIS) technology that translates the intersection of boundaries of one type of information--census related information, for example--with information from another geographic feature.

The Bureau's preliminary plans envision TIGER boundary files for counties, census tracts, block numbering areas, and county subdivision. The road systems are so complete that forestry agencies should take a good look at them from the standpoint of updating their own transportation systems. The TIGER files currently are available only on magnetic tape, but the Bureau is looking at the possibility of releasing TIGER on CD-ROM as well.

As of now, TIGER files contain only geographical information--individual streets and other features digitally coded by latitude and longitude. They will not contain any 1990 census data. Several software companies are planning to combine the TIGER files with 1990 census data on compact discs.

Geographical Information System Technology

Desktop demographic systems become even more powerful when linked to geocoding and mapping software--geographical information systems (GIS). GIS technology and the proposed census data systems are virtually made for each other. Geographical information systems analysis can overlay many features about an area's population and urban development with data about the physical characteristics of the area (Thompson 1989). A GIS also provides a set of tools necessary to model and understand the flow of people, resources and commodities into and through the interface--essentially a depiction of the infrastructure.

CONCLUSION

The ability to assign a latitude and longitude to in-house records will be a fast effective link between census information and our wildland urban hazard reduction and fire prevention efforts. GIS technology will allow land and fire managers to superimpose population forecasts and trends, fire behavior factors, and even past fire occurrence records, enabling projections of fire problems years before they actually occur.

Although this paper has been oriented to the wildland urban interface fire problem, the potential for demographic research is much greater. The dynamics of populations and their attitude toward wildlands and their management affect all phases of forestry. We expect that many of the concepts and models that we are developing will apply equally to other forestry problems from wildlife management to watershed management.

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CLIMATE, FIRE, AND LATE QUATERNARY VEGETATION CHANGE IN THE CENTRAL SIERRA NEVADA

Eric G. Edlund and Roger Byrne[1]

Abstract-Fossil charcoal provides a record of significant changes in the importance of fire in the Central Sierra Ncvada over the past 15,000 years. Changes in fire regime appear to be related to regional shifts in climate. During the late Pleistocene (ca. 12.5—10 ka), minimal sedimentary charcoal influx is correlated with fossil pollen and macrofossil indicators of a moist climate, probably with deeper spring snowpack than the present. In the early Holocene (10—7 ka), macrofossils indicate a shift from white and lodgepole pines to more xeric ponderosa pine-dominated forest. Charcoal influx climbs rapidly to maximum values in this zone, in conjunction with increases in fir, oak, and dwarf mistletoe pollen, along with bracken fern spores. Charcoal declines to modem values between about 7-3 ka, by which time the modem mixed conifer forest became established.

Changes in the abundance of bracken, dwarf mistletoe, oak and ponderosa pine can be strongly correlated with charcoal influx. The late-Pleistocene interval of minimum charcoal influx is a period in which dense forest surrounded the lake, indicating that fire frequency was not directly a function of fuel availability. Increasing summer drought in the early Holocene made tire an important factor in vegetation change. The zone of rapid increase in charcoal ahundancc, beginning 10,000 years ago, is associated with abrupt changes in vegetation, including the first appearance of ponderosa pines and firs following dcglaciation.

INTRODUCTION

Concern over the potential impacts of climate change on natural ecosystems has demonstrated a need for long-term studies of vegetation-climate-fire relationships. Recent workers (Overpeck and others 1990; Clark 1988a) have suggested that prolonged warm-dry climatic intervals may lead to increases in fire frequency and intensity. The relationship is not a linear one, however; a shift to a warmer, drier climate could eventually reduce fire intensities as a function of decreased biomass available for burning.

The composition of forests in the Sierra Nevada has been strongly influenced by fire. Before twentieth-century fire suppression, fire frequencies in mixed conifer forests averaged about 7-10 years (Wagcncr 1961; Kilgorc 1973). Studics of Sicrran ponderosa (Weaver 1968), red fir, and sequoia-mixed conifer forests (Kilgore 1973) have demonstrated the extent to which the dominant montane tree species are adapted to periodic firc. The role of fire in higher elevation forests, where mountain hemlock and lodgepole pine dominate today, is less clearly understood (Rundel and others 1988). Before the twentieth century, natural fires in the Sierra Nevada are believed to have been of generally mild intensity and limited extent.

Computer modclling of forest responses to climate change (Overpeck and others 1990) has indicated that an increase in the rate of ecological disturbance accompanying potential CO,-induced climatic changes would produce greater changes in stand composition than would climate changes alone. Such impacts are of concern in modern forest management, particularly since palcoccological work has tied firc to climate changes on time scales of hundreds of years or less. Working on lake sediments from northwestern Minnesota, Clark (1988a) found high charcoal abundance from 1400-1600 A.D., a period of warm, dry climate in the area. Fire frequency during this interval is estimated at 44 years, compared to 85-90 years during the subsequent "Little Ice Age" of the 17th—19th centuries.

The role of climate in determining fire regimes may have been even more important during the major climatic shifts which have occurred over thousands of years since the last major ice age. Good evidence for a relationship between climate and fire frequency has been uncovered in midwestem North America. Early work by Waddington (1969) documented an increase in charcoal influx at Rutz Lake, Minnesota from 8-4,000 years ago, corresponding to a shift from oak to prairie vegetation. In southern Wisconsin, Winklcr and others (1986) recorded increased charcoal 6,500—3,500 yr. B.P., at a time of lowered lake levels and a shift from mixed mesophytic forest to oak savanna.

Paleoecological work in the Sierra Nevada provides a chronology of postglacial climate change inferred from shifts in vegetation (see fig. 1 for locations discussed below). Late Plcistoccnc pollen and macrofossil records show evidence of colder, drier conditions, with sagebrush (*Artemisia*) and juniper important components of the vegetation (Adam 1967; Batchelder 1980; Cole 1983; Davis and others 1985; Davis and Moratto 1988). At some sites, pine and fir forests developed bctween 12,500—10,000 years ago, probably responding to soil dcvclopmcnt and a wetter climate. In the

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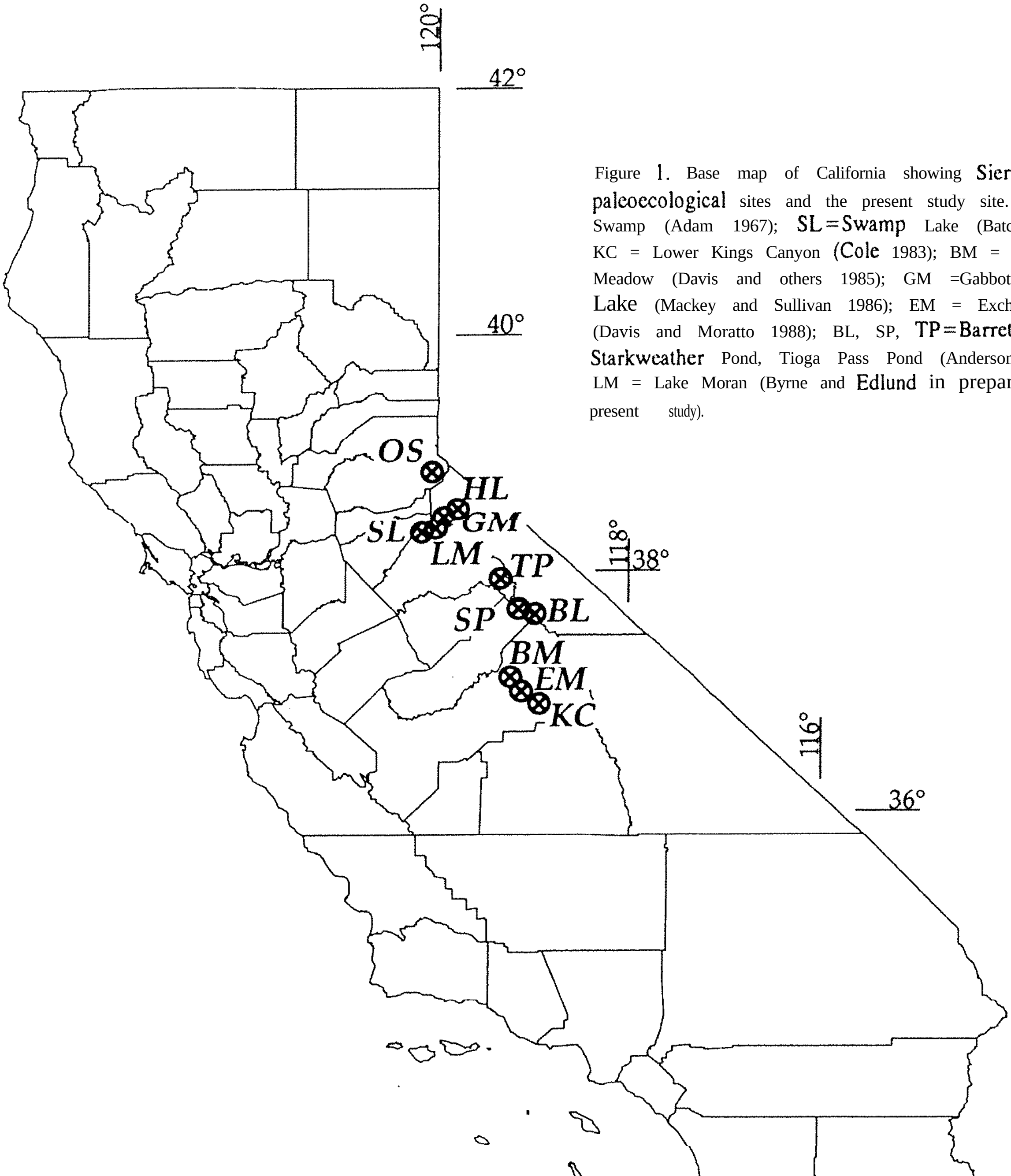

Figure 1. Base map of California showing Sierran paleoecological sites and the present study site. OS =Osgood Swamp (Adam 1967); SL=Swamp Lake (Batchelder 1980); KC = Lower Kings Canyon (Cole 1983); BM = Balsam Meadow (Davis and others 1985); GM =Gabbot Meadow Lake (Mackey and Sullivan 1986); EM = Exchequer Meadow (Davis and Moratto 1988); BL, SP, TP=Barrett Lake, Starkweather Pond, Tioga Pass Pond (Anderson 1990); LM = Lake Moran (Byrne and Edlund in preparation, and present study).

early Holocene, warming and drying conditions arc reflected in increases in oak and other pollen types, beginning ca. 10–9,000 yr. B.P., and lasting until at least 6,500–5,000 yr. B.P. (Mackey and Sullivan 1986; Byrne 1988; Davis and Moratto 1988; Anderson 1990). High elevation sites record increased effective precipitation beginning about 6,000 yr. B.P. (Anderson 1990). At some of the lower lake sites, warn-climate indicators persist until ca. 4,000–3,000 yr. B.P., when fir increased in response to the onset of Neoglacial cooling (Adam 1967).

The early Holocene xerothcrmic interval has been widely recognized in western North America. Mathcwes (1985) summarized work in British Columbia documenting xerothermic conditions 10–7,000 yr. B.P., following an interval of cool moist climate in the late Pleistocene (12-10 ka). He argued that douglas-fir, alder, and bracken fcm, which reach maximum levels in pollen records over this interval, are fire-adapted species which responded to increased fire frequency along with climatic warming.

In the Sierra Nevada, the existing evidence for changes in the role of tire during postglacial time is quite limited. At Exchequer Meadow (Davis and Moratto 1988), sedimentary charcoal reaches maximum abundance in deposits dated approximately 8,000–4,000 yr. B.P. At nearby Balsam Meadow (Davis and others 1985), macroscopic charcoal appears only after ca. 7,000 yr. B.P.

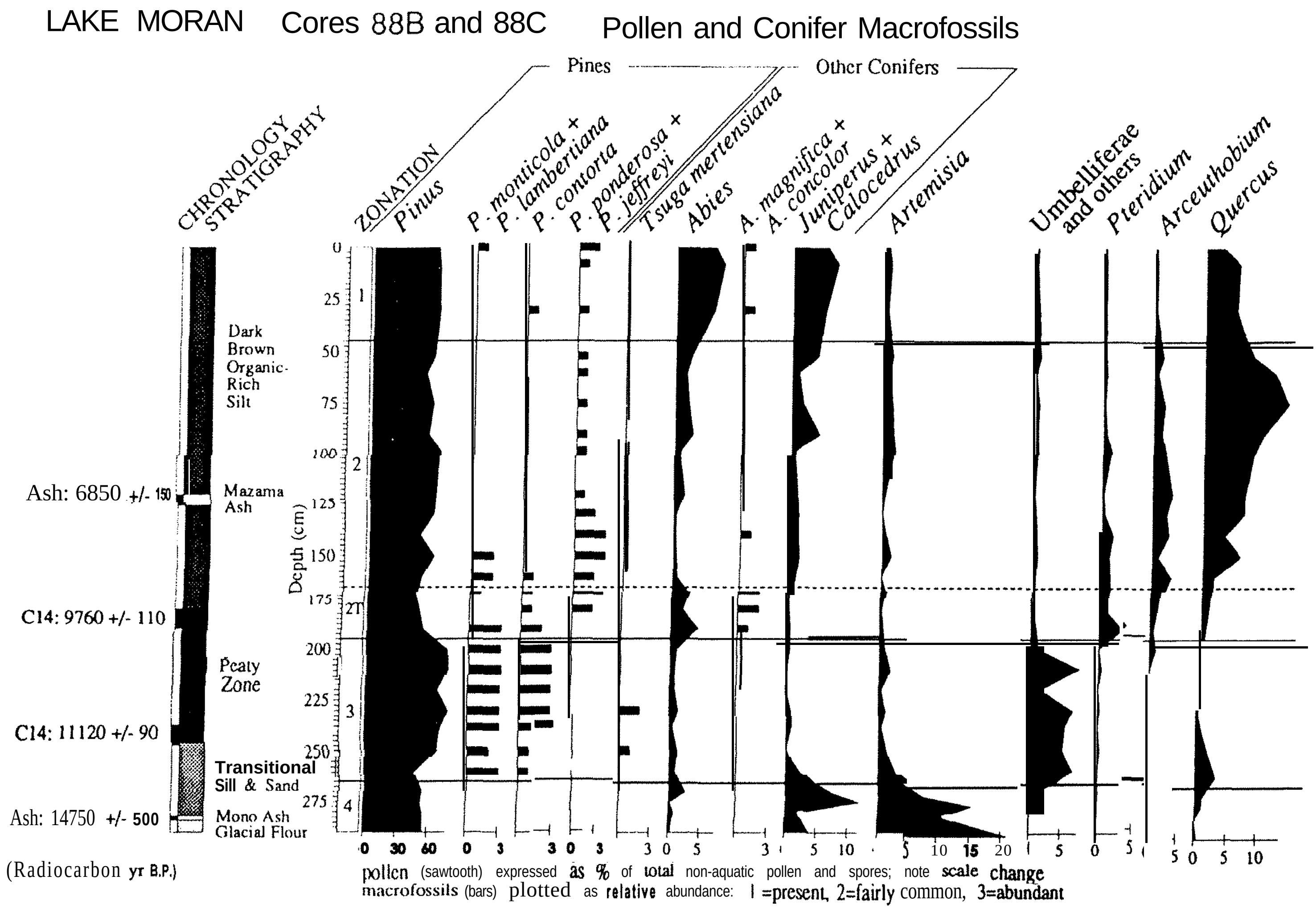

Figure 2. Selected pollen and macrofossils from Moran cores 88B and C. Sawtoothed pollen curves, labeled with genus or family name, indicate abundance as a percentage of total non-aquatic pollen grains and spores counted. Macrofossil bars, labeled by species, indicate relative abundance (see text). Zonation is by the authors. Radiocarbon dates were obtained from Beta Analytic, and volcanic ash layers were identified by Andrei Sarna-Wojcicki, U.S.G.S.

METHODS

The lake sediment samples analyzed for this project were taken in October, 1988, using a standard Livingstone piston corer with 2" diameter barrel. Two cores, 88B and C, were taken within 2 meters of one another, and correlated by depth and stratigraphy. Cores were transported intact to the U.C. Berkeley Pollen Lab. Core B was used for macrofossil analysis, while core C was subsampled at 10-20cm intervals for pollen and microscopic charcoal analysis. ^{14}C dates were obtained on two ten-centimeter segments of core 88C. Two volcanic ash layers present in the cores were identified by Andrei Sarna-Wojcicki at the United States Geological Survey in Menlo Park.

Extraction and preparation of pollen samples followed standard procedures as described by Faegri and Iversen (1975). Each pollen and charcoal sample underwent the same preparation, in order to eliminate differential effects of chemical treatments on the samples (Clark 1984). Pollen concentrations were calculated based on the ratios of *Lycopodium* control grains counted at each level. The curves of taxon abundance in Figure 2 are plotted as percentage of total non-aquatic pollen and spores counted; a total of at least 300 grains was identified at each level in the core. The diagrams were compiled using CALPALYN (Bauer and Orvis 1990).

Core 88B was sampled in measured segments of 5 or 10cm. Macrofossil sampling was based on standard procedures described by Birks (1980). Identification of pine needles was accomplished by comparing external morphology and thin sections with reference material and with Harlow's (1947) photographic key. For each sample, we calculated the total length of needle remains of each taxon. For Figure 2, length values were classified on a scale from 1, "present," with only one or two fragments per sample, to 3, "abundant," with total needle length exceeding 200mm per sample.

Microscopic charcoal was analyzed using 400x magnification and an ocular grid with squares 19.8 μm on a side. All charcoal fragments larger than one-half of a grid square were assigned to the appropriate size class, and the total area of charcoal was calculated. Calculated charcoal concentration

LAKE MORAN Core 88C Charcoal

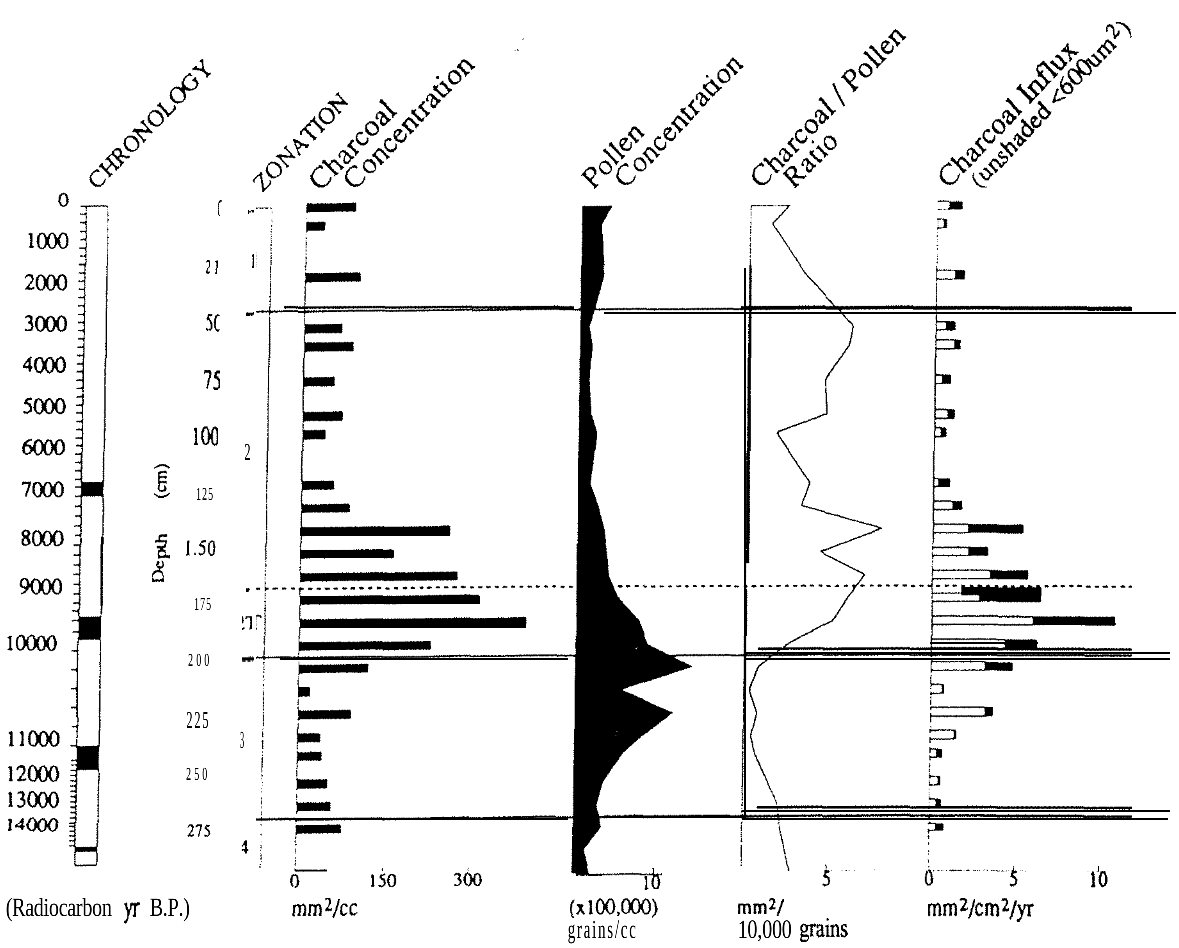

Figure 3. Charcoal and pollen indices for Moran core 88C. Sample depths and zonation are the same as in Figure 2 Calculated charcoal influx is divided into the share of fragments smaller (unshaded) and larger than $600\mu m^2$.

and influx to the lakebed are based on the ratio of counted charcoal to counted *Lycopodium* control spores (fig. 3). Sedimentation is estimated at a linear rate between radiocarbon and ash dates. Influx is not estimated for the glacial flour sample below the lowest tephra date. To control for variation in charcoal influx due to sedimentation processes, the ratio of charcoal to pollen was calculated for each sample (Cwynar 1978).

Charcoal Taphonomy

Questions of interpretation of sedimentary charcoal have been discussed in detail by Patterson and others (1987) and Clark (1988b). Charcoal in lake sediments ranges in size from fine dust, ≤ 2 micrometers in diameter, to macroscopic fragments centimeters in length. Like dust, small particles may travel long distances once entrained. Charcoal may reach high altitudes in the convective plume which rises from a fire. More intense tires produce higher plumes, resulting in longer-distance transport of all size classes of charcoal. The smallest sizes of charcoal may reflect a subcontinental source (Clark 1988b). As a result, even lakes in areas of 20th century fire suppression contain high levels of charcoal in recent sediments (e.g., Lake of the Clouds, MN, Swain 1973). Conversely, local tires, recorded as fire scars on trees, may go unrepresented in lake sediments if the plume convects most charcoal beyond the lake basin (Patterson and others 1987).

Lakes with small watersheds can accurately record local fires (Anderson and others 1986). Lake Moran is a small lake with no inlet stream, and a drainage basin of about 12 hectares. In this setting, it may be expected that larger microscopic charcoal fragments in the lake sediments originated from nearby fires, while the smallest fragments record events within a larger region. Figure 3 breaks microscopic charcoal influx into small fragments, $<600\mu m^2$ in area, and larger pieces, which ranged up to $>14,500\mu m^2$. We assume that the fires which produced the larger charcoal fragments probably occurred within 5-10 km of the lake (cf. Clark 1988b).

VEGETATION HISTORY AND THE RECORD OF FIRES AT LAKE MORAN

The Mono Craters and Mazama ash layers and radiocarbon dates provide a chronology for the Lake Moran sediments. The lake was exposed by ice retreat some 15,000 years ago This represents an early date for Tioga (late-Wisconsin) dcglaciation in the Sierra Nevada, confirming Batchelder's (1980) report from nearby Swamp Lake.

The earliest pollen assemblages (zone 4 on figure 2) reveal an open vegetation of sagebrush (*Artemisia*), juniper, and pines. This environment must have been effectively drier than the present study area, perhaps similar in appearance to higher subalpine environments today. No identifiable macrofossils are present in these sediments. Sedimentation rates were slow following the initial formation of the basin, and the calculated annual charcoal influxes arc among the lowest in the 'core. Some large charcoal fragments (> 2.500 μm^2 in area) are present, but it is uncertain whether these fragments blew into the lake from nearby fires, or were simply redeposited by the melting of the nearby glacier.

In late-Pleistocene zone 3, spanning ca. 12,500—10,000 yr. B.P., pine needles and high percentages of pine pollen indicate the establishment of a closed-canopy forest throughout much of the area (Figure 2). This forest signal is accompanied by increases in taxa normally associated with meadows, including members of the Umbelliferae, Liliaccac, Onagraccae, Malvaceae, and Ranunculaccac. (These taxa are grouped as "Umbellifcrae and others" in Figure 2). The presence of mountain hemlock (*Tsuga mertensiana*), white pines (*Pinus monticola* and *P. lambertiana*), and lodgepole pine (*P. contorta* ssp. murrayana) indicate a moister climate than that found in the Sierra Nevada today. The meadow taxa may have become established on sites too wet for tree growth. During this time, deep snowpacks may have persisted late into the summer, damping the effects of California's summer drought season. Charcoal concentration is generally lower in this interval than anywhere else in the core (Figure 3). The ratio of charcoal to pollen is consistently low, and larger charcoal fragments (> $600\mu m^2$) arc virtually absent. The evidence suggests that fires rarely burned the dense forest which surrounded the lake.

The Pleistocene/Holocene transition (subzone 2T) is marked by rapid increases in all measures of charcoal abundance (Figure 3). Charcoal concentration and influx reach maximum values here, at levels 5-10 times greater than in modern or late-Pleistocene scdimcnts. Lodgepole and white pines wcrc beginning to be replaced by yellow pincs (P. *ponderosa* and/or *P. jeffreyi*) (Figure 2). During this transitional period, persisting less than 1000 years, firs (both *Abies magnifica* and *A. concolor*) bccamc important constituents of the Moran forest.

The pollen record for zone 2 (Figure 2) shows a small relative decrease in pines, with increases in oaks and bracken fern (*Pteridium*). The evidence reveals a more open forest, with warmer summers allowing an increase in the relative importance of oaks. Dwarf mistletoe (*Arceuthobium*) pollen increases in this interval, perhaps indicating that conifers were under increasing ecological stress. Charcoal influx remains high until ca. 7,500 yr. B.P., dropping off to near-modem levels just above the Mazama ash layer (Figure 3).

Interestingly, both the charcoal/pollen ratio curve and the oak percentage curve (fig. 2) show continued high values through the middle Holocene (7,000—3,000 yr. B.P.). Since there is no evidence for changes in the lake's sedimentation regime, wc rely on the consistently low charcoal influx values to infer a decrease in the intensity of fires during this interval, compared to the early Holocene; however, the oak, bracken and sagebrush suggest that the forest remained drier and more open than it is today. By 3,000 yr. B.P., Neoglacial cooling and/or increasing moisture allowed fir, lodgepole, and sugar pine to reoccupy the lake margin. Fire has remained less important than it was in the early Holocene, although modem charcoal influx exceeds late-Pleistocene minimum values.

The important implication of the high charcoal values is that the early Holocene was a period of more extensive and more intense fires than either the late Pleistocene or the late Holocene. Indeed, it seems likely that an increase in the importance of fire was the mechanism by which the changing postglacial climate produced dramatic shifts in vegetation. The source of the early Holocene charcoal may well have been the dense late-Pleistocene forest, which began to burn under a more xcric climatic regime.

Temporal changes in several pollen and macrofossil types show a strong correlation with the charcoal signal. Bracken is highly correlated with total charcoal influx, as is fir to a somewhat lesser extent. The latter relationship is puzzling but important; fir is not normally considered a fire-adapted species, particularly compared to its associates in the Sierran montane forest. Red fir (*Abies magnifica*), the first species to appear in the macrofossil record, may have been able to effectively colonize fire-cleared patches, or it may have become established on soil fertilized by increased nutrient cycling under more frequent fires. In zone 2, increases in pondcrosa pine, oak and dwarf mistletoe lag behind charcoal influx, but generally the curvcs for these taxa parallel the changes in the charcoal/pollen ratio. Umbellifcrae and Cupressaceae pollcn arc negatively correlated with charcoal, as are lodgepole (*Pinus contorta*) and white pines (P. *lambertiana* and *P. monticola*).

CONCLUSIONS

Our results suggest that Sierran fire regimes have changed as a function of climate. In the Sierra Nevada, fire frequency and intensity are controlled by California's strongly seasonal precipitation regime. Today, summer drought makes fires increasingly likely in September and October, before the first winter rains. The Lake Moran record shows that from 12,000 to 10,000 years ago, summer drought was less pronounced than today, probably as a result of late-spring snowstorms and/or snowpack persisting into the summer. At that time, warm summers coupled with more abundant moisture led to the development of a dense mixed conifer forest, comprising several specks which no longer grow together in the Sierra Nevada. Fire was less important as an ecological factor in this late-Pleistocene forest than in the Holocene.

At 10,000 yr. B.P., in association with a rapid increase in fossil charcoal influx, the local vegetation changed abruptly. Red fir and ponderosa pine first appeared at this time. Conversely, western white pine disappeared from the site, and lodgepole and sugar pine began a more gradual decline. Bracken fern became much more important about 10,000 years ago, while oak and dwarf mistletoe pollen increased more gradually, reaching maximum pollen percentages between 9,000–3,000 yr. B.P. We infer rapid environmental changes beginning at the end of the Pleistocene. The Lake Moran record demonstrates the importance of fire as a determinant of vegetation during this climatically significant period.

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FIRE-HERBICIDE SYSTEMS FOR MANIPULATING JUNIPER

D.M. Engle and J.F. Stritzke[1]

Abstract-Fire exclusion results in extensive encroachment of fire-intolerant junipcrs, specifically eastern redcedar (Juniperus virginiana) and ashe juniper (J. ashei) into tallgrass prairies and other grasslands and hardwood forests of the central US. Control of both juniper species is possible with fire because they do not sprout from roots and so they are killed by intense tires that scorch the crown. However, low-intensity fires on sites with dense stands of junipers do not scorch crowns of larger juniper trees. One objective of our studies was to determine the effect of various paraquat treatments on juniper foliage water content, a primary factor affecting flammability of juniper crowns. A second objective was to determine if paraqoat pretreatment would increase crown scorch from broadcast tires in tallgrass prairie. We evaluated leaf water content of eastern redcedar treated individually with two rates (0.3 or 1.1 kg/ha) of paraquat, in two carriers (water, oil-in-water emulsion), and at two volumes (47 or 188 I/ha) of carrier. The oil-in-water emulsion carrier reduced leaf water content more consistently than the other carriers, so we used the oil-in-water emulsion carrier in a study of paraquat applied aerially in early August. Rate ranged from 0.3 to 3.4 kg/ha and included two volumes (47 or 94 l/ha). All aerial applications of paraquat reduced juniper leaf water content to 50 percent or less three weeks after treatment compared to 100 percent water in untreated junipers. Leaf water content in the late summer dry season will normally decline to about 80 percent, a level too moist for ignition of juniper foliage. Combining paraquat with burning almost doubled the crown scorch of large trees over either treatment alone and increased kill of large trees to more than SO percent compared to no trees killed with the burning-only or paraquat-only treatments.

INTRODUCTION

Tallgrass prairie and other grasslands of North American arc regarded as fire-tolerant systems which evolved with fire as a part of the natural disturbance *regime* (Anderson 1982). Protection from fire *results* in extensive encroachment of fire-intolerant woody species, including eastern redcedar and ashe juniper into tallgrass prairies as well as into other grasslands and hardwood forests of the central U.S. (Arend 19.50; Bragg and Hulbert 1976, Wright 1978). Juniper encroachment into these ecosystems has increased exponentially in recent years, modifying the Physiognomy of these ecosystems and reducing their value as rangelands (Wright and Bailey 1980; Snook 1985).

Tallgrass Prairies that become dominated by juniper are similar to other ecosystems that were previously grasslands or savannahs but now have successional processes driven by woody vcgctation. Upon reaching a transitional threshold Promoted by reduced fire frequency and intensity, new successional processes prevent reversion to grassland from overstory dominance by woody species, and fire is no longer an effective agent in reversing succession to grassland (Archer 1989; Bryant and others 1983; Walker and others 1981). Drastic anthropogenic modification (i.e., herbicides or mechanical manipulation) of the woody vegetation is necessary to convert the plant community to dominance by herbaccous Plants (Archer 1989; Engle 1987).

Fire kills both species of juniper if fine fuel loading is sufficient to produce an intense fire that completely scorches thc crown because neither species sprouts *from roots or stems* after topkill (Owensby and others 1973; Wink and Wright 1973). However, tree mortality decreases as tree size increases so fire intensity must increase to scorch the crown of taller trees (Bryant and others 1983; Dalrymple 1969; Engle and others 1988). Fire intensity and tree mortality are reduced further in dense stands of large trees because junipers reduce the production of fine fuel so that fires in these stands are of tow intensity or perhaps fires fail to carry at all (Bryant and others 1983; Engle and others 1987).

Junipers arc often controlled mechanically or by herbicidcs, but mechanical and herbicide methods of juniper control are generally either too expensive or are ineffective on large trees (Scifres 1980; Stritzke 198.5). Small trees can be controlled individually with some soil-applied herbicides and paraquat applied in hot weather can result in considcrabtc damage to trees (Engle and others 1988). Integration of several brush control treatments into a single set of treatments, i.e., integrated brush management, is one way to deal effectively with brush (Scifrcs 1980). Paraquat applied to individual trees in the summer has increased juniper crown damage from broadcast bums the following spring with light line fuel loading (Engle and others 1988). A combination of broadcast application of paraquat followed by a prescribed bum has bccn suggested as an integrated method to control severe infestations of juniper (Engle and others 1988). We

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conducted two studies, each with a separate objective: 1) to determine the effect of various paraquat treatments on juniper foliage water content, a primary factor affecting flammability of juniper crowns (Bunting and others 1983), and 2) to determine if aerial application of paraquat will reduce foliage water content and increase crown scorch from broadcast fires in tallgrass prairie.

METHODS

Study 1 was conducted on two sites approximately 15 km southwest of Stillwater, OK on the Oklahoma Agricultural Experiment Station's Agronomy Research Range. Shallow prairie was the primary range site of one location and red clay prairie was the primary range site of the other location. About 2800 kg/ha of fine fuel arc produced on both sites in a normal year. Herbaceous understory was dominated by tallgrasses and eastern redcedar canopy cover was less than 5 percent.

Paraquat was foliar applied to individual eastern redcedar trees on August 17-18, 1988 with a compressed-air hand sprayer using a T-3 conejet nozzle to simulate aerial application. Treatments were in a factorial arrangement of paraquat (0.3 or 1 .1 kg/ha), carrier (water or oil-in-water emulsion), and volume of carrier (47 or 188 l/ha). The design was completely randomized with ten replications on each of two sites located approximately 5 km apart. Eastcm redcedar foliage was sampled at weekly intervals (August 24, September 1, and September 8) following treatment, weighed in the field, and dried in a forced-air oven to determine foliage water content, which is expressed on the basis of dry weight.

Study 2 was conducted in Johnston County, OK, approximately 30 km northeast of Ardmorc, OK. The study area is located on a loamy prairie range site normally producing about 3900 kg/ha of forage in the absence of juniper interference. Ashe juniper, with canopy cover of approximately 25 percent, was the overstory woody dominant The under-story was a mixture of tallgrasses and midgrasses. Paraquat was applied aerially in the oil-in-water emulsion carrier to ashe juniper on August 9, 1989. Herbicide treatments included no herbicide treatment and paraquat applied at rates ranging from 0.3 to 3.4 kg/ha. Paraquat rate treatments were nested within two volumes (47 or 94 l/ha) of the carrier. Plots were 0.1 ha (30 X 30 m). The design was a randomized complete block with four blocks. Foliage was sampled three weeks after paraquat treatment, weighed in the field, and dried in a forced-air oven to determine foliage water content, which is expressed on the basis of dry weight.

Two blocks were burned with a headfire on September 1, 1989. At the time of the bum, air temperature was 38°C, relative humidity was 40%, average wind velocity was 8 km/h, and fine fuel loading within the juniper stand was visually estimated at 1500 kg/ha. Crown damage to trees and tree kill were determined in all treatment plots two months after the bum. Visual estimates of crown damage and tree kill were conducted on three size classes based on tree height: small (0.8 to 1.5 m), medium (1.5 to 2.5 m), and large (2.5 to 5.0 m).

We subjected the data from both studies to analysis of variance. Leaf water data from study 1 and study 2 were analyzed as repeated measures in time (Winer 1971). Means were separated by LSD subject to a protected F-test at the 95% probability level as suggested by Carmer and others (1989).

RESULTS

Study 1

Study site interacted ($P < 0.0001$) with treatment and sampling date so data from each study site were analyzed separately. No treatment effects were significant one week after paraquat application, but by two weeks after treatment, paraquat rate interacted with both carrier and volume of carrier ($P < 0.0224$ and $P < 0.0182$, respectively) at Site 2 (fig. 1). Water content of foliage with all treatments increased the second week after treatment, which we attributed to a change in available soil water in response to 20 millimeters of precipitation on August 28 between the first and second week after treatment. By the third week after treatment, leaf water had dropped drastically in trees in all trcatmcnts, with some below 50 percent leaf water. Paraquat treatment rate, volume, and carrier interacted at study Site 1 and 2 on the third week after treatment ($P < 0.0163$ and $P < 0.0002$, respectively). Foliage water content was lower in trees treated with the 1.1 kg/ha rate of paraquat and results were more consistent with the oil-in-water carrier (fig. 1).

Study 2

Leaf water content did not differ ($P > 0.1288$) between the two volumes of carrier on either sampling date so we averaged the rate data over the two volumes. Leaf water content was less in foliage of treated trees than in untreated trees two weeks after paraquat treatment and declined to 50 percent or less by three weeks after treatment with all rates of paraquat ($P < 0.0001$) (fig. 2). Although increasing rate of paraquat was not additive, all rates of paraquat desiccated ashe juniper foliage to the extent necessary to increase flammability by three weeks after treatment (Bunting and others 1983). Although juniper leaf water was higher in 1989 than in normal years (Engle and others 1988), leaf water content in the late summer dry season will normally decline only to about 80 percent, a level too moist for ignition of juniper foliage.

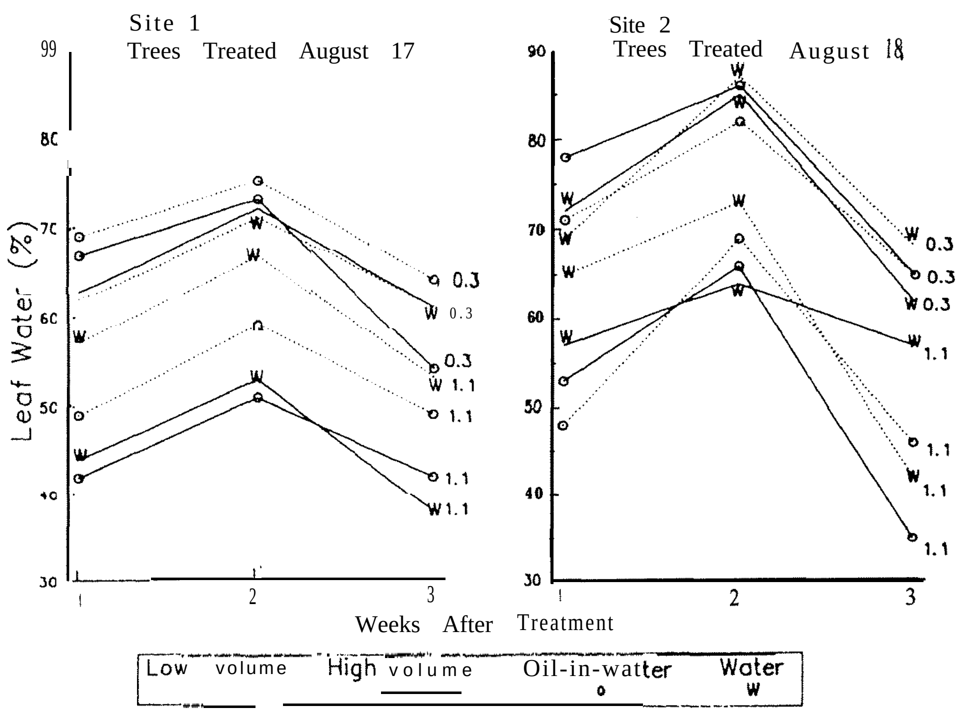

Figure 1.- Leaf water (%) after paraquat application at Site 1 and Site 2. Paraquat rate (0.3 or 1.1) is in kg/ha. Low volume of carrier is 47 l/ha and high volume is 188 l/ha. Values arc means of 10 trees. LSD,,,= 13 at Site 1 and LSD,,,=23 at Site 2 on week 3. Foliage of untreated trees was 80% water at both study sites on September 8.

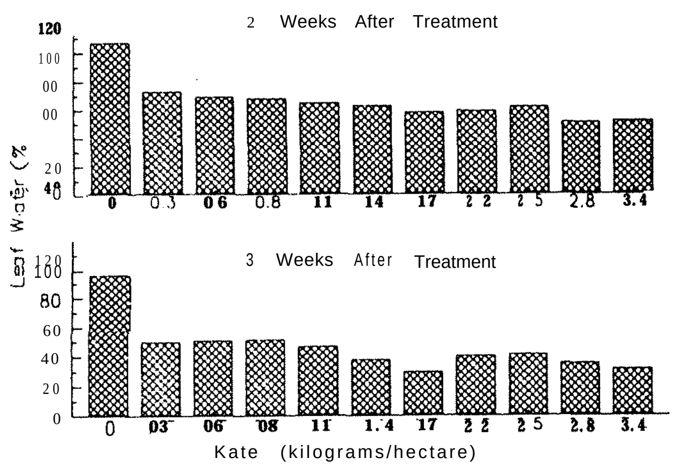

Figure 2.- Leaf water (%) two weeks and three weeks after paraquat was aerially applied on August 9, 1989. Valued art: means of 8 trees, averaged over volume of carrier ($LSD_{0.05}$= 10 on week 2 and $LSD_{0.05}$= 11 on week 3).

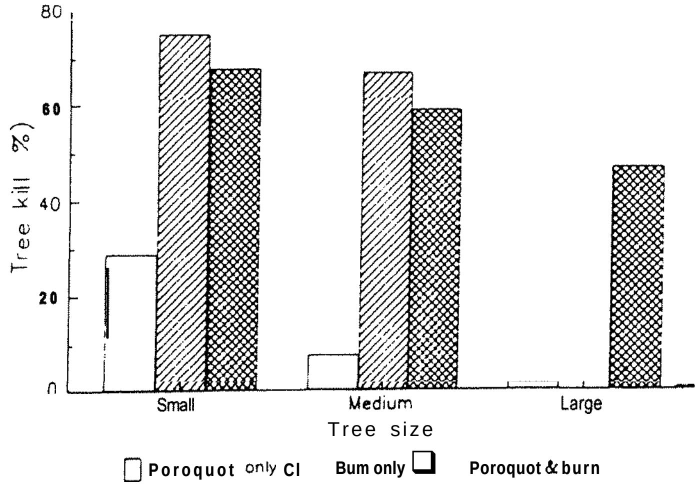

Figure 3.- Crown damage (%) two months after burning. Values for paraquat only treatments are averaged over rate Treatments were not different ($P > 0.0604$) for small and medium trees, but treatments were different ($P < 0.0319$) for large trees (LSD,,,, =27).

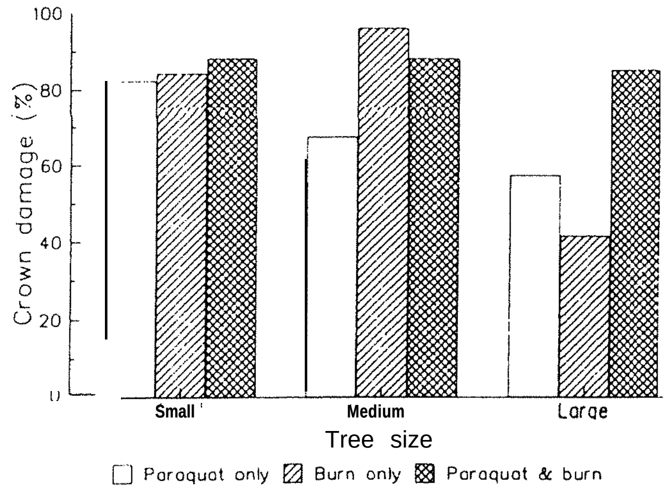

Figure 4.- Tree kill (%) two months after burning. Values for paraquat only treatments are averaged over rate. Treatments were not different ($P > 0.3031$) for small trees, but treatments were different ($P < 0.0392$) for medium and large trees ($LSD_{0.05} = 48$ and 31, respectively).

The amount of crown damage to trees was not affected by either carrier volume or paraquat rate within the paraquat plus bum treatment for any size of tree ($P > 0.1247$). Howcvcr, crown damage and tree kill of medium and large trees differed ($P < 0.0604$) among the three treatments (paraquat only, bum only, or paraquat plus bum) (fig. 3 and 4). Crown damage to trees from fire alone and paraquat alone was similar to the results of previous studies (Engle and others 1987). The effects of paraquat and fire on large trees were approximately additive in respect to crown damage which exceeded 80 percent in the paraquat plus bum treatment plots (fig. 3). Paraquat and burning appear to have a synergistic effect in killing large trees in that almost no trees were killed by the single treatment of either paraquat or burning but half of large trees were killed by combining the two treatments (fig. 4).

DISCUSSION

Paraquat applied in hot weather by hand to individual trees (study 1) or applied aerially in a broadcast spray (study 2) was effective in reducing juniper foliage water content to below the critical point of 60 percent water (Bryant and others 1983; Bunting and others 1983; H.A. Wright, pers. comm.). Desiccation of juniper foliage by paraquat applied aerially almost doubled the crown scorch and increased the kill of large trees from 0 to 50 percent. The results of this study arc in agreement with previous work in which desiccation of juniper foliage by treating individual trees with paraquat compensated partially for light fine fuel loading in cool-season fires the spring after summer paraquat application (Engle and others 1988).

We believe it is possible to use paraquat as a desiccant to promote crown fires in closed-canopied stands of juniper. Previous attempts to ignite crown fires have been unsuccessful in dense stands of ashe juniper in Texas (Bryant and others 1983) and in pinyon (Pinus cdulis) and juniper woodlands in Nevada (Bruner and Klebenow 1979) possibly because of high foliage water content, gaps in the tree canopy, and cool fire-weather conditions. Bryant and others (1983) evaluated igniting windrows of recently dozed ashe juniper to produce an intense fire with flames in contact with standing trees to produce a crown fire in dense stands of ashe juniper, but no sustained crown fire resulted. However, six standing trees were killed for every dozed tree by the windrow fire thereby reducing the overall cost of mechanical treatment of the juniper stand. Bruner and Klebenow (1979) were unable to obtain crown tires in closed stands (i.e., no understory) of pinyon-juniper and concluded that crown fires are possible only when burned under hazardous conditions.

Our research indicates an integrated approach using paraquat and tire can be used to reduce overstories of large junipers for restoring tallgrass prairie dominated by junipers. Further research is needed to determine if crown fires can be ignited from paraquat-desiccated strips in dense stands of junipers using aerial ignition with a helitorch.

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ELASTICITIES ASSIST TARGETING OF ARSON PREVENTION PROGRAMS

R.A. Kluender arnd L.C. Thompson'

Abstract-Elasticities (percentage change in one factor divided by percentage change in another factor) were calculated to demonstrate the effect that reducing the number of arson fires in Arkansas counties had on area burned. Elasticities are employed by economists and managers to determine which factors yield the greatest returns per unit of effort expended. Hypothetical reductions in area burned and average wildfire size were determined by randomly decreasing arson wildfires. Results showed that as arson rates increased above SO percent, disproportionately greater reductions in area burned accrued from reducing arson. Accordingly, counties with high elasticities for area burned should he the first targets for arson reduction campaigns. Elasticities for average fire size showed weak responsiveness to arson reductions. When the primary objective of a prevention program is reducing area burned, elasticities for area burned can provide another tool for wildfire prevention specialists to use in appraising where scarce resources can be best utilized.

INTRODUCTION

Achieving effective wildfire prevention and suppression programs requires proper analysis of available data and correct interpretation of results, which supports the generation and implementation of effective policy. Because of the tremendous task of acquiring, entering and extracting meaning from wildfire data, detailed analysis is commonly not performed. Instead, management targets (typically calculated statistics) or goals are used to gauge the yearly progression of control programs. Generally, these management targets are presented in a standardized form (as an example, on a "per thousand protected hectares" basis) because they provide a quick gauge across county and regional boundaries and can be used across state boundaries to weigh the success of innovative programs.

In Arkansas, state lands protected from wildfire by the Arkansas Forestry Commission (AFC) include all privately owned forest and pasture lands but not areas within incorporated town and city borders, row-crop lands, federal ownerships (such as national forest lands) and other public ownerships. The Associate State Forester for Protection in Arkansas has set four wildfire management targets for county and state level programs. They are: 1) less than 1.5 hectares per thousand protected hectares (TPH) burned per year; 2) not more than 0.37 fires per TPH per year; 3) an average fire size less than 4 hectares and; 4) less than 40 percent arson fires. The most important of these goals is the first, to keep burned area as low as possible. Counties below any particular target are considered within compliance relative to their wildfire prevention or suppression goals.

Factors such as weather, attitudes and prevention and suppression efforts all play a part in determining the final yearly toll to wildfires. However, increasing rates of arson (Arkansas Forestry Commission 1984-1989) have aroused concern. These trends were first reported by Kluender et al. (1988, 1989). Perhaps the most important findings of these studies were that local residents caused 70 percent of the arson tires and that arson fires had an average size twice that of other causes (8.4 vs. 4.2 hectares). Additionally, these studies indicated that while general state-wide trends were important, they did not provide the detailed information required to formulate county-level fire prevention and suppression plans. Preferably, prevention and suppression programs should be designed for local conditions to better target problems and reduce wildfire losses. Kluender et al. (1990) subsequently looked at county-level wildfire patterns and found considerable variability among Arkansas' 75 counties for arson rates and area burned. The data showed that of the 35 counties that exceeded the AFC target of 40 percent arson rate, 27 counties also exceeded the target for area burned; and, of the 39 counties that exceeded the target for area burned, 28 counties also exceeded the target for arson rate. Of the 48 counties that exceeded either target, 28 counties simultaneously exceeded both targets. So, the link between higher arson rates and higher area burned is well established among counties.

Wise management dictates that limited budget dollars be disbursed in the most effective way. Frequently, in wildfire prevention campaigns, managers may wonder whether money is being spent in the right place or on the right program. Numerous appraisal methods have been used by various agencies from time to time. Benefit-cost ratios express some measured benefit against the cost of obtaining it. Measures of cost to protect a known "value at risk" adopt an actuarial approach to the same problem. These measures, however, are static, cross-sectional statements of expected benefit for given expenditure. A more sophisticated concept is the use of elasticities to identify the percentage change of a dependent variable (like hectares burned) for a percentage change in an

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independent variable (like arson rate); thus, elasticities measure responsiveness to change. Elasticities are used by economists and policy makers to show how much change in a given factor to expect for a given change in a driving or causal variable (Pindyck and Rubinfeld 1981). Generally, those independent variables that produce the greatest changes in the dependent variable (highcst elasticities) are the best choices for policy manipulation.

Two factors point to the need to reduce the total number of arson tires in the South. First, arson is, in most southern states, the greatest single cause of wildfires (USDA Forest Service 1984). Second, arson tires tend to be larger than those from other causes. This paper demonstrates the use of elasticities as a measure of responsiveness to arson prevention based on hypothetical reductions in total area burned and average fire size. Our objective was to describe a tool that wildfire managers might use to help them select targets for arson prevention programs.

METHODS

We acquired the records of all reported wildfires on lands protected by the Arkansas Forestry Commission (AFC) for the period January 1983 through December 1988 (Arkansas Forestry Commission unpublished data). These data included records developed from Individual Fire Reports (AFC Form 24 10. 1) tilled out by suppression personnel immediately after a fire was investigated. Variables chosen for inclusion in this study were year, county, cause (for example, arson or debris burning) and hectares burned per fire. The data set included reports of 16,047 tires. A second data set was obtained from the AFC that contained total hectares and protected area per county. The data sets were sorted by county and merged. The number of fires per TPH, area burned per year per TPH, average fire size, and percentage arson tires for each county were calculated.

Because we (and the AFC) were interested in evaluating the effects of reducing arson fires in counties with high arson rates (by AFC definition, high includes those that exceeded the AFC management target of 40 percent arson rate), counties with an arson rate greater than 40 percent were retained in the analysis. Thirty-five of Arkansas' 75 counties (47 percent) remained in the active data set.

To simulate an effective prevention program that reduced arson rate to the AFC target, individual arson fires were randomly eliminated from each county data set until arson fires comprised 40 percent or less of the total fires. After randomly reducing arson fires to 40 percent, average tire size, area burned per TPH and number of fires per TPH were recalculated for each county. Elasticities for area burned per TPH (E_{AREA}), and average tire size (E_{AVSIZE}) were calculated for each county. For example, elasticity of area burned was calculated by:

$$E_{AREA} = \frac{(\Delta AREA \mid AREA)}{(\Delta ARSON \mid ARSON)}$$

Where:
$\Delta AREA$ = Change in area burned after reducing arson rate
AREA = Area burned before reducing arson rate
$\Delta ARSON$ = Change in percentage arson tires
ARSON = Percentage arson fires before reducing arson rate.

Elasticity of average fire size was calculated in a similar manner.

Three replicates of the random reductions in arson tires for each county were created and the calculations for E_{AREA} and E_{AVSIZE} performed for each county in each replicate. An ANOVA was performed on the E_{AREA} and E_{AVSIZE} data sets to determine if differences existed between replicates. To establish relationships, E_{AREA} and E_{AVSIZE} were regressed against the percentage of arson tires in each county before the reduction to 40 percent. Regression analysis was also used to ascertain what contribution the pre-reduction arson rate and average fire size of each county made to E,,.

A standard statistical package, SPSS (Norusis, 1988), was used to perform the initial data analysis and data sorting. The Quattro Pro spreadsheet (Borland International 1989) was used to calculate the elasticities, and SYSTAT (Wilkinson 1988) was used in the final comparison of pre- and post-reduction conditions and in the regression analysis. Statistical significance was accepted at the α = 0.05 level.

RESULTS AND DISCUSSION

The ANOVA showed no differences among the replicates for either E_{AREA} ($F_{2,102}$ = 0.085, p = 0.919) or E_{AVSIZE} ($F_{2,102}$ = 0.050, p = 0.951). Therefore, we used the mean value for each county's elasticities in all further analyses.

Elasticities for area burned per TPH (E_{AREA}) with respect to arson rate were greater than 1 .O (unity elasticity) for 23 of 35 counties; these counties were considered responsive to reductions in arson fires (Figure 1). When this elasticity was regressed against arson rate the linear function was:

$$E_{ARM} = 0.388 + 0.013 \times \text{ARSON RATE}$$

Both the constant and the slope were different from zero. While considerable variation was present in the data, general trends were obvious. The slope shows that the higher the initial level of arson tires, the greater will be the response of area burned to reductions in arson levels. Additionally, the best-tit line (R^2 = 0.435) rose above unitary elasticity at an arson rate of 49 percent. Accordingly, we reason that area burned per TPH would typically be responsive to changes when the arson rate in a county exceeds 49 percent, or 50 percent in round numbers.

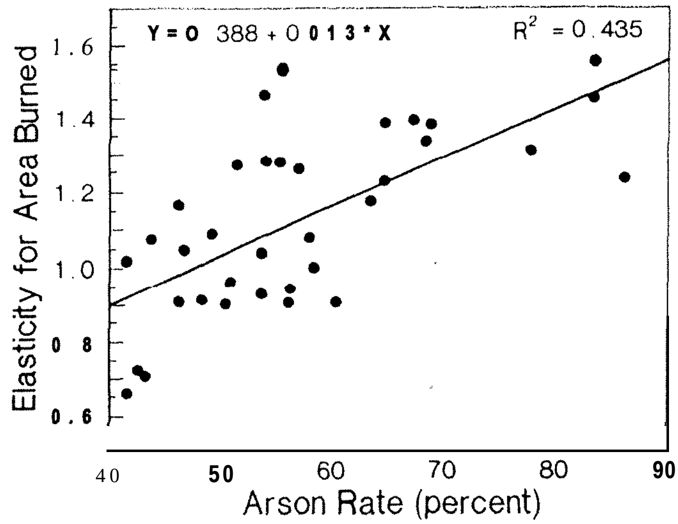

Figure 1. Elasticity for wildfire area burned compared with arson rate for 36 of 75 counties in Arkansas that exceeded the wildfire management target of 40 percent arson.

Elasticities for average fire size (E_{AVSIZE}) with respect to arson rate were ail below unity and, hence, considered unresponsive. Although average fire size decreased when arson fires were eliminated, the reduction was relatively small when expressed as a percentage. Hence, E_{AVSIZE} was typically low. When this elasticity was regressed against arson rate neither the constant nor the slope were different from zero.

Finally, we regressed the elasticities of area burned as a dependent variable against arson rate and average fire size The response surface was a quadratic function:

$$E_{AREA} = 0.187 + 0.014 \times \text{ARSON RATE} + 0.019 \times \text{AVERAGE FIRE SIZE} + 0.001 \times (\text{ARSON RATE})(\text{AVERAGE SIZE}).$$

However, only arson rate was different from zero; this surface had an R^2 of 0.549. Therefore, we conclude that arson rate alone, which had an R^2 of 0.435, is the best predictor for estimating E_{AREA}.

This analysis establishes that when the primary objective of a prevention program is reducing area burned, a good tool for choosing targets for the prevention program is the elasticity for area burned. For best results, ordinal ranking of counties should logically proceed from those with the highest to lowest elasticities. The best choice among counties with equal elasticities should be made by selecting the county with the largest average fire size.

We have shown that for Arkansas counties with high arson rates, disproportionately greater reductions in area burned can be expected when the number of arson fires is decreased. This is important because keeping burned area as low as possible is the most important goal of the AFC. Calculating and using elasticities for area burned will aid the efficient targeting of monies and other efforts for arson prevention programs. In areas where arson is not the primary cause of wildfires, perhaps other causes could be investigated in a similar manner.

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LONG-TERM IMPACTS OF FIRE ON COASTAL PLAIN PINE SOILS

William H. McKee, Jr.'

Abstract-Repeated burning of pine forests over long periods may have pronounced effects on the maintenance of soil fertility and soil development. Analyses of soils from four long-term prescribed burning studies in the Atlantic and Gulf Coastal Plain indicate that burning has had no effect on the total carbon and nitrogen level in the surface mineral soil. Winter burning increased the retention of nitrogen in the mineral soil over time. Changes in the carbon/nitrogen ratio in the forest floor with burning suggest that at least part of the increase in nitrogen in the mineral soil was due to pyrolysis of the litter. Available phosphorus was consistently increased in the surface 5 cm of soil by prescribed burning; however, the effect is less apparent on total phosphorus reflecting the low mobility of the nutrient. Concentrations of exchangeable bases in the surface soil increased with the frequency of burning. It is postulated that without burning, immobilization of calcium in the forest floor can lead in time to a magnesium:calcium imbalance and alteration of the soil formation processes.

INTRODUCTION

A concern about repeated burning is that it may reduce the "tilth" and productivity of soils in the long-term. Studying long-term effects of tire in the South is difficult, however, because burning has been part of the ecology of most fire sites there. Fire affects soil properties but measuring the cumulative effects on soil development takes many years and many fires. Data from some long-term burning studies in the Southeastern Coastal Plain offer clues about how fire influences soil forming processes.

Soil properties have been monitored in a number of Coastal Plain prescribed burning studies (McKee 1982, Ralston and others 1982, and McKee and Lewis 1983). In this paper I have combined the findings and updated the measurements for four major studies to develop indications of how burning may alter soil chemical properties and influence soil development.

MATERIALS AND METHODS

Study Areas

The four prescribed burning studies represent a wide range of soil textures, drainage classes, topographic positions, and understory vegetation. Brief descriptions of the four areas arc as follows:

Alabama. This study is near Brewton, Alabama, on the upper Coastal Plain. Soils are classed as coarse loamy siliceous thcrmic (Typic Paleudults), complexed with fine loamy siliceous thermic (Typic Palcudults), loamy siliceous thermic (Grossarenic Palcudults) and loamy skeletal siliceous thermic (Typic Hapludults). Overstory vegetation consists of 60- to 70-year-old longleaf pine (*Pinus palustris* Mill.). e index (age 50) ranges from 19 to 24 m, and understory vegetation consists of grasses, forbs, and small woody sprouts.

Treatments are replicated eight times on 0.16-ha plots. Treatments examined here are an un-bum control and biennial winter burning. The study was initiated in [illegible] and plots had been burned five *times* when data reported here were gathered. The last bum was applied about four months before sampling.

Florida. This study is on the Coastal Plain flatwoods in north central Florida. Soil on the study area is classed as sandy siliceous thermic (Aeric Haplaquods) with an organic pan between 46 and 61 cm deep. The overstory vegetation contains mixed, naturally seeded longleaf and slash (*P. elliottii* Engelm. var. *elliottii*) pines 60 to 70 years old. index (age 50) for the study area is 20 m, and average basal area is approximately 15.3 m^2 ha.

Treatments are replicated six times in a randomized block design on 0.81-ha plots. Treatments consist of an un-bum control, winter bum every four years, and annual winter burning. The annual winter bum was not imposed until six years after initiation of the study; at the time of sampling there had been 14 annual burns.

Georgia. The study is on a nearly level Coastal Plain site in south central Georgia. The site is poorly to somewhat poorly drained. Soils are loamy siliceous thermic (Arenic Paleaquults) covering about 2/3 of the site, loamy siliceous thermic (Arenic Plinthoquic Paleudults), fine loamy siliceous thermic (Plenthic Fraquidults), and loamy siliceous thermic (Arenic Paleaquults). Overstory trees are longleaf and slash pines from 25 to 30 years old.

Plots consist of pastures of about 19.7 ha on which grazing is also observed. Treatments consist of no burning, triennial winter burning, biennial winter burning, and annual winter burning. Treatments have been in force for 40 years except for a IO-year period 25 years previously when the stand was regenerated.

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South Carolina. This study is in eastern South Carolina on a Pleistocene terrace of the lower Coastal Plain. The terrain is nearly level and is poorly to somewhat poorly drained. Soils are clayey mixed thermic (Umbric Paleaquults), clayey mixed thermic (Typic Albaquults), clayey kaolinitic thermic (Typic Paleaquults), clayey kaolinitic thermic (Aeric Paleaquult), fine loamy siliceous thermic (Aquic Paleudults), tine loamy siliceous thermic (Aeric Paleaquults). The overstory is 70-year-old loblolly pine (P. tacda L.) and the site index (age 50) is 27 to 30 m. Burning treatments were begun 30 years prior to the current sampling.

Plots are 0.10 ha in area. Treatments consist of an un-bum control, periodic winter burning, periodic summer burning, annual winter burning, and annual summer burning. Periodic burning was done every seven years, or when hardwood stems were approximately 2.5 cm d.b.h.

Sampling Methods

On each plot in the four studies, the forest floor and mineral soil were sampled at 40 points. Samples collected at 10 points were combined to make four composite samples per plot each for the forest floor layer and the mineral soil. Forest floor samples consisted of all material, including 01, 02, and 03 (L, F, and H layers), collected from within a 15-cm square frame. The forest floor was mor humus with little mineral soil mixed in it. No attempt was made to separate the forest floor by layers. Two 2.5-cm soil cores were collected from within the square of each sample point. Depth of the cores was 0 to 5 cm and 5 to 10 cm on the South Carolina site and 0 to 8 cm and 8 to 16 cm on the other sites. Only the surface samples were used for soil reaction properties. Analysis of both depths were used in computing total nutrients and organic matter levels. The South Carolina and Georgia sites were sampled in mid winter while the Florida and Alabama sites were sampled in early summer.

Laboratory Methods

Forest floor samples were dried at 70" C for 24 h, weighed, and ground to pass a 40-mesh screen. Nitrogen content was determined by a modified micro-Kjeldahl procedure, and ammonia was determined by the salicylate-cyanurate procedure (Nelson and Sommcrs 1973). Other analyses were made on material dry ashed for 2 h at 450° C and taken up in 0.03 M HNO,. Phosphorus was determined by the molybdovanadate procedure (Jackson 1958); calcium and magnesium were determined by atomic absorption. A separate sample was dry ashed at 500° C for 2 h to determine mineral content, which was subtracted from the dry weight. Thus, weights of forest floor material represent only the component lost on ignition.

Soil samples were air dried and crushed to pass a 2-mm sieve. Organic matter was assayed by wet oxidation (Jackson 1958). Exchangeable bases were determined by atomic absorption on extracts made with 1M NH, OAC (Jackson 1958). Available phosphorus was determined by extracting 2.5 g of soil with 20 ml of Bray P2 solution (Bray and Kurtz 1945). Inorganic phosphorus was fractionated by the method of Chang and Jackson (1957), and organic phosphorus was determined by the procedure outlined by Olsen and Dean (1965). Total soil nitrogen was extracted by micro-Kjeldahl digestion, and ammonia was determined by the salicylate-cyanurate method (Nelson and Sommcrs 1973). Soil pH was measured with a glass electrode on a 1:2 soil:water mixture.

Total levels of nutrients were computed using the concentrations observed in laboratory analysis, the volume of soil obseved, and bulk densities considered typical for the series in the case of the South Carolina site measurements were made of individual plots.

RESULTS AND DISCUSSION

Mineral Soil Properties

Burning effects tended to be consistent over the range of Coastal Plain soils examined, with the degree of effect reflecting soil properties of specific sites (Table 1).

pH. Soil pH increased slightly but significantly with intensity of burning on the Florida and South Carolina sites. In the South Carolina on the plots burned annually in summer soil pH was significantly higher than on control plots on periodically burned winter plots. The difference was about 0.4 pH unit. Periodic summer and annual winter burning produced intermediate pH values, which did not differ significantly from those of other treatments. On the Alabama site, prescribed burning did not significantly alter pH values; however, on burned areas, readings averaged 0.2 pH units higher. Soil pH ranged from 3.5 to 4.4 among burning treatments on the Georgia site. The effects were not significant, but pH tended to increase with frequency of burning.

For a wide range of soils and burning techniques, soil pH has been shown to increase to some degree shortly after burning (Wells and others 1979). As time passes, acidity increases and soil pH returns to its prebuming value. Degree of pH increase and time needed to return to the prcbuming level depend on burning intensity, amount of forest floor consumed, soil organic matter and clay contents, rainfall, and internal soil drainage. Results of these studies indicate that periodic burning raises pH values slightly. On heavy soils, this effect is apparent for at least 7 years after burning. For the sandy soil on the Florida site, the pH value for the periodic burning treatment did not differ from that of the unburned control. This result suggests that the pH increase had a shorter duration on this site, which was last burned two years before sampling.

TABLE 1. Soil reaction, exchangeable bases and cation ratios for four sites uith different burning treatments.

Sample site and burn treatment	Soil pH	Exchangeable bases Ca	Mg	Ratio Mg/Ca
		mmol kg		
Alabama (0-8 cm)				
No Burn	5.2	1.278	0.20A	0.16
Biennial uinter	5.4	1.89A	0.148	0.07
Florida (0-8 cm)				
No Burn	4.1AB	0.256	0.22B	0.88
Periodic winter	4.08.	0.45A	0.35A	0.77
Annual uinter	4.2A	0.57A	0.38A	0.66
Georgia (O-S cm)				
No Burn	4.1	0.11	0.05	0.42
Triennial winter	3.5	0.18	0.05	0.31
Biennial winter	4.4	0.29	0.07	0.36
Annual winter	4.1	0.16	0.05	0.32
South Carolina (O-5 cm)				
No Burn	4.18	0.63	0.286	0.44
Periodic minter	4.18	0.79	0.33AB	0.42
Periodic summer	4.2AB	0.87	0.33AB	0.38
Annual uinter	4.2AB	1.23	0.51A	0.41
Annual summer	4.5A	1.08	0.38AB	0.35

Within col- and sites. values followed by the same letter or no Letter do not significantly differ at the 0.05 Level.

Samples collected after 10 years of burning on the South Carolina site averaged within 0.1 to 0.2 pH unit of the values after 10 years of burning (Metz and others 1961). An exception is the annual winter plots with a pH value of 4.2, compared with 4.6 at 10 years.

Exchangeable bases. On the Alabama site, burning increased exchangeable calcium in the surface 8 cm of soil by 0.62 mmol kg^{-1} (Table 1). Similarly, on the Florida site the burned plots had 0.30 to 0.32 mmol more exchangeable calcium than did the control plots. On the Georgia site, exchangeable calcium increased slightly but not significantly with increased burning intensity. The difference was only .05 to .018 mmol kg'.

Burning treatments did not significantly alter exchangeable calcium concentration on the South Carolina site, but calcium tended to increase with increased burning in the O-5 cm soil layer. Ten and 20 years of burning significantly increased calcium on this site (Metz and others 1961; Wells 1971).

Values for exchangeable calcium 10 and 20 years after initiating treatments were within the range of the 30-year values. These data do not indicate any long-term change in quantities of calcium in the mineral soil.

Generally, magnesium responses to burning were similar to calcium. Exchangeable magnesium increased with burning by 0.06 mmol kg^{-1} in the O-8 cm soil layer on the Alabama site. On the Florida site, magnesium content was 0.13 to 0.16 mmol higher on burned plots than on control plots. Burning treatments had no detectable effect on the exchangeable magnesium levels on the Georgia site. The annual winter bum had 82 percent more exchangeable magnesium than did the control plot on the South Carolina site. Magnesium values for other burning treatments did not differ from those for the control or annual winter burning.

An indication of the degree of soil development or weathering is the ratio of exchangeable magnesium to calcium (Buol et al. 1973). As soils become more weathered, relative magnesium levels increase and calcium levels decline. Within the period of these studies, this effect is found only in the surface Al horizon. Burning did not change the ratio of exchangeable magnesium to calcium at lower depths. Over an extended period, the effect of the accumulated forest floor or presence of organic acids and leaching of cations from the upper horizon--as observed by Herbauts (1980)--should appear at lower depths, including the B horizon. In any case, prescribed burning apparently slows the process of soil formation and may help maintain soil productivity at a higher level. A final proof of this hypothesis would require a timespan approaching several hundred years.

A more immediate problem, as proposed by Lyle and Adams (1971), is a nutrient imbalance caused by higher concentrations of magnesium than of calcium. These authors observed that because of “mass action effects,” magnesium in excess of calcium results in reduced loblolly pine root growth. According to this concept, the surface soil layer on unburned control plots on the Florida site is nearing this condition while that on burned plots is not. Such relationships also may bc important to microbial processes found on pine sites.

Heyward (1937) observed that the elimination of burning on Coastal Plain pine sites resulted in abrupt changes in the visual characteristics of a soil profile. His interpretation was that exclusion of burning accelerates soil weathering or development. With moisture conditions and parent materials he was observing, the end results would probably be a spodosol without fire and no spodic horizon with fire. Specific reports of such effects of fire on soil formation have not be published. It appears that the natural evolutionary pattern of soil development caused by water-soluble carbon, as observed by Herbauts (1980), can be moderated by burning. This conclusion is based on lysimeter studies on soils under forest cover where the degree of soil weathering was found to relate to the amount of soluble carbon moving through the profile. Nutrients moving from ash material after burning are alkaline (Raison and McGarity 1978), at least until the ash has dissolved and moved into the mineral soil. To some degree, burning destroys the substrate and either consumes or volatilizes organic acids produced in the forest floor. Such a change is assumed to be roughly proportional to the reduction of organic matter in the tire. The water-soluble carbon may be in a humic and fluvic acid (De Kimpe and Mattel 1976), in carbonic acid (McColl 1971), or in other soluble organic acids. Observations of Binkley (1986) on a site similar to that reported for on the South Carolina site in this report found that burning may convey resistance to soil acidification from atmospheric deposition or other sources by reducing pools of acid in the forest soils. Where sulfur dioxide from burning fossil fuels significantly acidities precipitation, the resulting sulfuric acid is a much larger factor than carbonic acid (Cronan and others 1978). The acid radicals react with the soil to form salts with alkali, alkali earth, and amorphic metals that move through the soil horizons in the process of soil development.

Yaalon and Yaron (1966) indicate that any man-caused activity such as adding fertilizer or changing the pH will change the metapedogenetic processes that retard podsolization; the rate of change depends upon the intensity of treatments. Bidwell and Hole (1965) also discuss human practices as dominant factors in altering soil formation by controlling organic matter buildup. Thus, burning pine sites on the Coastal Plain tends to maintain soils in a less developed state and probably in a better tilth. Historically, such has been the case for much of the Coastal Plain, where tire maintains the pine ecosystem.

Phosphorus fractions in the surface mineral soil. Phosphorus was fractioned into various chemical forms in the surface 5 to 8 cm to determine the effect of prescribed burning on the disposition of phosphorus and its availability for plant uptake.

The amount of available phosphorus in the soil was slightly higher on burned than on control plots on all four study sites, but the differences were significant only on the South Carolina site (Table 2). On the Alabama site, phosphorus levels were 0.1 to 0.2 mg kg^{-1} higher on burned plots.

TABLE 2. Distribution of soil phosphorus in available, mineral and organic forms for the surface sample Layer of soil on four sites.

Sampled sites burn treatment	Phosphorus Fractions			
	Available	Mineral	Organic	Total
	kg ha'			
Alabama (o-8 cm,!				
No Burn	2.3	11.6	20.18	31.70
Biennial. winter	2.4	12.0	23.3A	35.3A
Florida (O-8 cm)				
No Burn	7.1	15.4	18.0	33.48
Periodic uinter	9.5	19.0	21.6	40.6A
Annual winter	10.5	19.3	21.2	40.5A
Georgia (0-8 cm)				
No Burn	2.4			
Periodic winter	3.2			
Periodic summer	3.3			
Annual winter	2.7			
South Carolina (O-5 cm)				
No Burn	3.68	25.9	52.1	78.0
Periodic winter	4.5AB	27.5	45.8	73.3
Periodic summer	5.0AB	24.9	53.0	77.9
Annual tinter	5.4A	29.9	56.4	86.3
Annual summer	4.1AB	27.6	51.6	79.2

Within columns and sites, values for chemical fractions followed by the same letter do not differ significantly at the 0.05 Level. Available phosphorus represents both organic mineral fractions and is not used in computing the total phosphorus.

Annual and Periodic burns increased available phosphorus by 2.5 to 3.4 mg kg^{-1} in the O-8 cm depth on the Florida site. Available phosphorus ranged from 2.4 to 3.3 mg g^{-1} and increased with burning on the Georgia site.

Burning effect on individual mineral phosphorus fractions were relatively small and in most cases did not alter individual fractions, hence, the water soluble, aluminum and iron fractions are reported as mineral phosphorus. In general, the mineral phosphorus levels tended to increase with intensity of burning except on the South Carolina site where no trend was apparent. Changes in the available form of phosphorus or the magnitude of these changes do not appear to relate well to the phosphorus present in the mineral or organic forms.

Organic phosphorus accounted for 50 to 69 percent of total phosphorus in the surface 5 to 8 cm of soil on the three sites where measurements were taken. On the Alabama site, burning increased organic phosphorus by 16 percent or 3.2 mg kg'. On other sites, an apparent increase in organic phosphorus of 3 to 5 mg kg' was noted, but the increase was not significant. The amount of organic phosphorus was positively related to total phosphorus extracted from soils on these sites, accounting for 86 to 94 percent of the variation in total phosphorus.

Burning significantly increased total phosphorus (sum of the mineral and organic fractions extracted from sandy sites by 4 to 7 mg kg-‘. There is no apparent difference between the annual and periodic burning on the Florida site. Burning also tended to increase total phosphorus on the heavier soils of the South Carolina site, but the response was not significant. The large proportion of the total phosphorus in the organic form in soil indicates the need to investigate this form of the nutrient and to increase its availability to higher plants. Daughtrey and others (1973) found the release of organic phosphorus from a Coastal Plain soil was completely dependent on the activity of soil micro-organisms in decomposing organic matter. Nutrient release with burning would accelerate the breakdown of organic matter and release of organic phosphorus to the soil solution. Accumulation of organic matter and organic phosphorus is partly the result of small organic particles being washed from the forest floor into the soil.

Forest Floor Properties

Organic content. Prescribed burning predictably lowered the total weight and nutrient content of the forest floor on all four sites (Table 3). Across the range of sites, the unburned control plots contained from 13 to 59 T ha“ of organic matter. Annual and biennial burning reduced the weight of organic matter in the forest floor by 39 to 44 percent on the

Table 3.--Average weights of forest floor components after burning treatments on four study sites.

Site and burn treatment	Organic contest	N	P	Ca	Mg	C:N Ratio
	kg^{-1} X 1000			kg ha^{-1}		
Site #1						
No burn	13.86A	2264	8.7A	67.2~	9.3A	30:1
Biennial bum	5.440	276	3.18	29.28	3.58	100:1
Site #2						
No Burn	29.16A	131A	24.7A	115.0A	21.0A	111:1
Periodic winter	9.52A	378	6.78	40.08	11.0B	128:1
Annual winter	4.54C	7B	3.06	19.08	6.06	324:1
Site #3						
No burn	59.5	494	16.5~	83A	23.0A	60:1
Triennial burning	12.4	81	3.28	218	3.68	77:1
Biennial burning	7.48	56	2.28	15B	2.68	66:1
Annual burning	17.18	100	4.20	278	5.18	85:1
Site #4						
No burn	26.27A	408A	17.4	12.0A	19.0A	32:1
Periodic winter	18.468	3008	12.1B	91.08	19.0A	31:1
Periodic summer	17.566	2778	10.68	77.08	16.0AB	32:1
Annual tinter	10.48C	156C	7.28	52.0C	11.0BC	33:1
Annual summer	10.05C	129C	7.18	48.0C	6.0C	39:1

Within columns and sites, values followed bv the same letter do sot differ significantly at the 0.05 Level. Where no" letters am. shown, no significant differences are present.

Alabama site. Periodic burning on the Florida and South Carolina sites reduced the weight of the forest floor by 33 and 70 percent respectively. The forest floor reduction was 71 to 87 percent with burning on the Georgia site. Season of burning did not significantly affect the reduction total organic content.

Organic content of the forest floor on control plots is approximately the same for the South Carolina site as reported 10 years earlier (Wells 1971). On these plots the forest floor contained 18.57, 26.88, and 26.27 T ha" after 10, 20, and 30 years of measurements (Metz and others 1961; Wells 1971). Thus, in terms of weight, the forest floor reached an equilibrium between 10 and 20 years after initiating the study, when the pine trees were about 45 years old. At 30 years, the forest floor contained 18 to 39 percent mineral material (determined by dry ashing the combined L, F, and H layer samples).

Wells and Jorgensen (1975) found that forest floor biomass reaches its peak in loblolly plantations at about age 30 in Piedmont stands. The sites in this study had considerably older trees, which produced less needles, but greater production of litter by hardwoods and herbs probably compensated for lower needle production.

Nitrogen. Nitrogen content in the forest floor decreased by as much as 95 percent with annual tires and 72 percent with biennial fires. Part of this nitrogen loss was from leaching water-soluble components and tine particulates from the forest floor into the soil. The forest floor of periodically burned plots had nitrogen losses ranging from 26 to 32 percent on the South Carolina site which had not been burned for live years at sampling, to a 72 percent loss on the Florida site which had been burned the previous year. Wells (1971) observed that the periodic burning on the South Carolina site 10 years earlier resulted in a nitrogen loss of about 112 kg ha" by volatilization. The 408 kg ha^{-1} of nitrogen in the forest floor on the control plots appear to represent an "equilibrium" value for this nutrient under the conditions imposed by the stand and the climate (Wells and Jorgensen 1975). Values on control plots on the other sites probably also represent near-equilibrium levels. Of interest are the nearly equal amounts of organic matter on the control plots of the Florida and South Carolina sites but about a four-fold greater amount of nitrogen on the South Carolina site than on the Florida site. The amounts of nitrogen in the forest floor probably reflect species and site conditions specific to each location.

The C:N ratio is a major determinant of availability of nitrogen and potential decomposition of the forest floor. The ratio of carbon to nitrogen widened by 1- to 3-fold on the Alabama and Florida sites following annual or biennial fires. On the South Carolina site much smaller increases (5 to 20 percent) occurred after annual bums, and no increases occurred after periodic bums. An exact C:N ratio is difficult to obtain because much of the organic matter is charred after burning. The magnitude of observed change, however, reflects an apparent nitrogen mobilization that cannot be explained by degree of carbon reduction. A number of rains fell on all the burned plots between burning and sampling. Comparison of forest floor values on the South Carolina site after 20 years shows a similar trend.

Heyward and Bamette (1934) observed that the L layer had a C:N ratio 2 to 3 times as wide as that of the F layer. Wells and Jorgensen (1975) observed a similar relationship for loblolly pine plantations in the Piedmont, where the C:N ratio of litter narrowed over time. Because it is primarily the L layer that is consumed by tire, it is surprising that burning results in a wider C:N ratio. Apparently, low-intensity fires have a "mobilizing effect" on nitrogen in the F layer, which may in part account for the increased nitrogen concentration in the upper 5 to 8 cm of mineral soil. Nitrogen relationships are supported by findings of Klemmedson and others (1962), who showed that burning accelerated nitrogen movement into the mineral soil. Light burning in ponderosa pine (P. ponderosa Dougl. ex Laws.) stands caused movement of 12.4 kg ha^{-1} nitrogen per year into the surface 2.5 cm of mineral soil. Wells and others (1979) summarized a number of investigations which indicate that appreciable mobilization of nitrogen as well as volatilization of the forest floor occurs after burning.

Phosphorus. Biennial or annual burning reduced the amount of phosphorus in the forest floor by 42 to 88 percent on all four sites. Periodic burning resulted in a 39 to 73 percent decrease in phosphorus in the forest floor. The season of periodic or annual burning did not affect phosphorus loss, and there was no significant difference between annual and periodic tires.

Cakium. Annual or biennial burning reduced calcium in the forest floor by 50 to 92 percent. Periodic burning resulted in a 28 to 39 percent reduction on the South Carolina site. Season of burning had no effect on changes in calcium content of this site. Thus, prescribed burning accelerated the rate of calcium return to mineral soil. This movement probably results from cations moving in the soil solution, but ash conduction may also be a factor. Wells and Jorgensen (1975) indicate that without burning, calcium loss from the forest floor is slow compared to potassium or magnesium loss and that after eight years appreciable quantities of the nutrient remain in the forest floor from a given year's deposition. Quantitatively, 50 percent of the magnesium from a given year's accumulated litter is lost from the forest floor in less than one year, while three years are required to obtain this degree of calcium mineralization (Jorgensen and others 1980).

Magnesium. Amounts of magnesium in the forest floor were approximately 1/4 to 1/10 those of calcium. The mobilization of magnesium with burning appears to be similar to that of calcium; and 43 to 71 percent magnesium was lost from the forest floor with biennial or annual burning. Periodic burning on the Florida site reduced magnesium by 52 percent. The period between burns and the season of burning did not significantly affect magnesium losses from the forest floor on the South Carolina site. Based on values reported by Wells and Jorgensen (1975) for loblolly pine in the Piedmont, the forest floor on these sites contains about 1/2 to 1/3 as much magnesium as in the tree biomass, and the nutrient would be expected to move out of the floor faster than calcium.

Forest Floor Mineral Relationships

To understand the quantitative relationships of the forest floor and increased nutrient concentration, the total contents are presented together for the surface 10 to 16 cm of mineral soil and forest floor.

Organic Matter. Burning reduced the organic matter content of the forest floor but not of the mineral soil. In fact, burning may have actually increased organic matter content of the mineral soil for the Alabama and Florida site but the increase was not statistically significant (Fig. 1). The result was a rather small loss of total carbon from the system due to burning part of the forest floor. The study sites in Florida and South Carolina, which have poor drainage, appear to have higher organic content in the mineral soil than those of Alabama and Georgia.

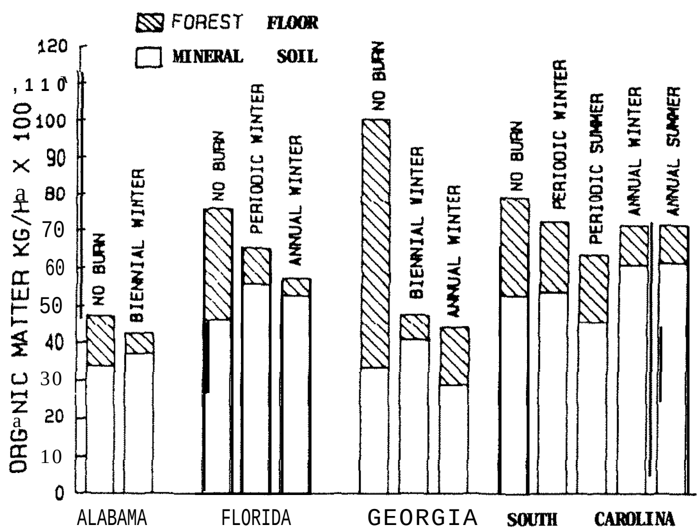

Figure 1. Organic matter content in the forest floor and soil after prescribed burning of coastal plain pine stands.

Nitrogen. On the sandy Alabama, Florida, and Georgia sites, burning caused a slight increase in nitrogen in the mineral soil despite a marked loss of forest floor weight after 8 to 40 years (Fig. 2).

Soil at the South Carolina site had been sampled after 10 years (Metz and others 1961), 20 years (Wells 1971), and 30 years (McKee 1982). Over 20 years between sample collection, total nitrogen changed little on the unburned control. After those treatments, there was a 20-year increase of 34 to 42 kg/ha--a four percent change for the periodic winter bum. The periodic summer bum (burned every seven years) resulted in a 128-kg nitrogen loss. The annual winter bum increased total nitrogen by 137 kg/ha. The striking effect was a 363-kg loss due to annual summer bum. Since both summer bums resulted in total nitrogen losses, summer burning apparently has a detrimental effect on the amount of nitrogen remaining in the surface mineral soil, while winter burning increases nitrogen. The exact cause is difficult to explain but may be related to a lack of nitrogen-fixing legumes that invade these treatment sites after summer burns.

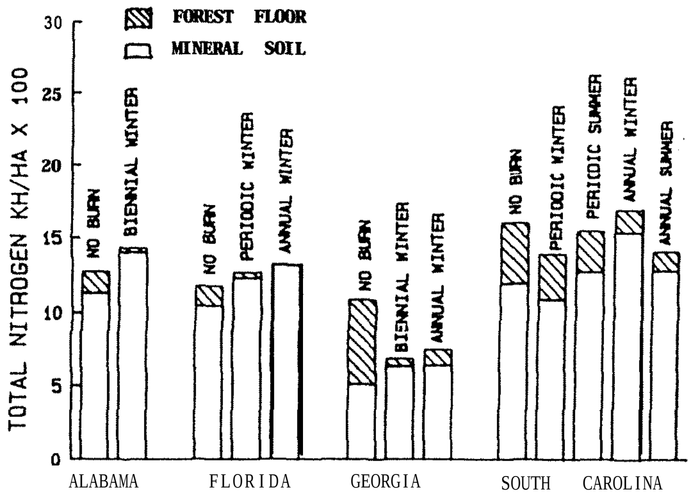

Figure 2. Nitrogen content in the forest floor and mineral soil after prescribed burning of coastal plain pine stands.

Phosphorus. Without burning, appreciable quantities of phosphorus were tied up in the forest floor on all four study sites (4 to 24 kg/ha) (Fig. 3). The amount of phosphorus in the mineral soil is difficult to relate to that of the forest floor. Available phosphorus was apparently increased by burning, but the quantities found did not relate well to frequency of burning. A standard chemical test suitable for all sites is difficult to select because numerous chemical forms of phosphorus are present. Available phosphorus levels appear to be slightly more responsive to treatments and are represented on all four sites.

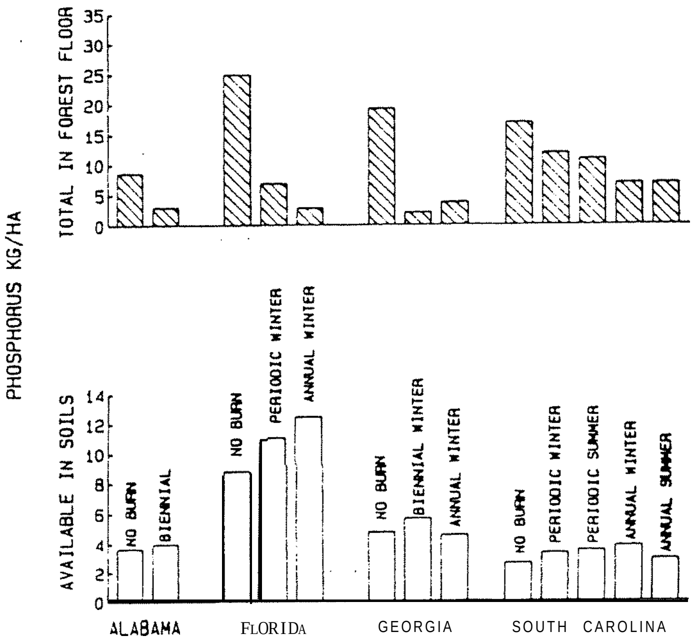

Figure 3. Phosphorus content of the forest floor and available phosphorus content in the mineral soil after prescribed burning of coastal plain pine stands.

Calcium. All the effects of burning on calcium appear to take place in the forest floor and the top 10 or 16 cm of mineral soil (Fig. 4). Without burning, 16 to 61 percent of the calcium present was in the forest floor. With burning, only 4 to 16 percent of the calcium was in there. The remainder was in the surface soil layers. These changes in calcium distribution logically account for the pH increase in the mineral soil associated with burning, and indicate a long-term effect on soil acidity. The amount of calcium in the forest floor decreased proportionally to the frequency of the burning on the Florida and South Carolina site. Earlier calcium observation on the South Carolina site, (Wells 1971) showed similar results, indicating little change in the calcium status of this site in the last 10 years.

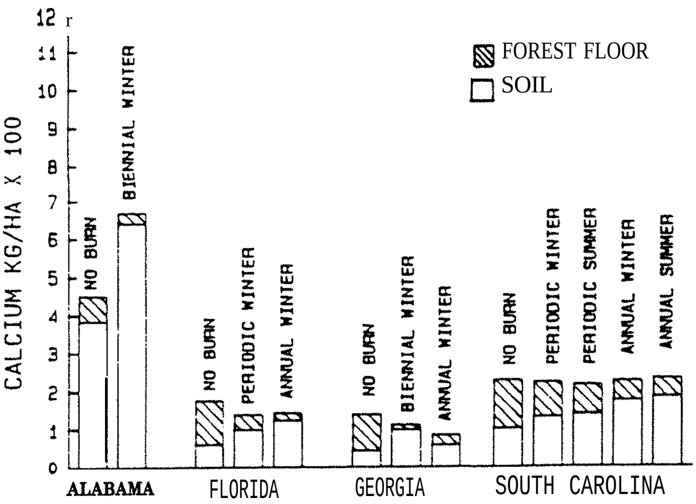

Figure 4. Calcium content in the forest floor and exchangeable calcium content in the soil after prescribed burning of coastal plain pine stands.

CONCLUSIONS

The most striking result of this analysis is the similarity of burning effects on soil properties despite obvious soil differences and probable differences in burning techniques, which were reported to be either low-tempcraturc flank fires, or backfires.

Organic matter consistently builds up faster in mineral soil on burned areas, and burning only reduces total nitrogen in the forest floor. On unburned plots, 6 to 11 percent of the nitrogen was in the forest floor. Over the range of study sites, annual burning reduced total nitrogen in the forest floor to 12 to 32 percent of the unburned levels, but burning did not appear to reduce total nitrogen in the mineral soil after up to 30 years of treatment. However, a balance sheet for the studies requires nitrogen data for the vegetation which may be causing an increase in soil nitrogen.

The consistent increase in available phosphorus in mineral soil caused by prescribed burning, may be one of the most beneficial effects of the treatment. No consistent pattern was found for burning effects on phosphorus fractions. The nature of compounds formed apparently represents specific pH conditions and mineral components in the soil. However, in all cases burning obviously accelerated mineralization.

The cation response was quite similar for all sites. The soil depth used appeared to represent complete cycling of calcium. Trends indicate that the unburned forest immobilizes a large proportion of the calcium altering the nutrient balance of the soil in some cases. Magnesium and calcium responded similarly to burning treatments, but their ratios indicate that magnesium recycles faster or at least accumulates in mineral soil in the absence of burning.

It is apparent that burning alters soil formation and long-term productivity over time. Evidence suggests that burning may improve soil by retarding soil development and, probably, formation of spodic layers in the profile. Proof of this observation would probably require five to six pine rotations to compare soil development with and without fire.

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FIRES, FORESTS, AND TRIBES IN THE NORTHERN PHILIPPINES: CULTURAL AND ECOLOGICAL PERSPECTIVES

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Abstract-The study was exploratory. The researcher utilized participant observation, case study, and interview-discussion methods to gather data. Purposive stratified multi-stage sampling guided initial selection of 103 respondents in 3 groups: kaingineros, school teachers, and government officials. The study is an attempt to explain forest burning from the perspective of the tire-setter. It is the first of a series of investigations that interpret forest burning as a lifeway of a people who inhibit a rugged environment and who possess a unique s&o-cultural temperament. Specifically, research focused upon the forest fire-setting behavior of the Ifugao, an ethnic tribe in the Cordilleras, a mountain range in Northern Philippines. This paper reports impacts of socio-culturally sanctioned indigenous forest burning practices the local economy, ecology, and society. Central to the issue of forest burning is the highly institutionalized Ifugao practice of muyung, or inherited private ownership of forests. Muyung greatly complicates governmental efforts to promote forest fire control, sound forest management, and sustainable forest development in Ifugao.

INTRODUCTION

Fire accounts for one-third of the damage done to the Philippines' critical watershed and forest lands. The problem continues to be addressed largely as a technical one, and forest management continues to he forestry-oriented following inception of the Social Forestry Program in the DENR. (The Philippine SFP, which was launched in 1982, is a radical departure from traditional programs, which put the forest before its occupants. Creation of the SFP is a mute admission that the conventional methods used to conserve and protect the forest have not succecdcd.) Ifugao, now legally recognized as the Cordillera Autonomous Region, was chosen for study because it had the highest rate of forest fires in the Northern Philippines and because its supposedly civic-minded and law-abiding people continue to bum forests despite the region's long history of Spanish and American religious endeavor and continuing government administration.

The study was designed to:

- document relevant demographic, economic, and social attributes of selected kaingincro respondents involved in forest burning practices in the area of study;
- determine level and type of knowledge (awareness), assessment (perception), and predisposition (attitudes) toward forests, forest fires, and forestry policies (including the prcscncc and role of the local forestry agency in forest fire prevention and managcmcnt),
- determine the existence and nature of beliefs relevant to forests and forest burning activities among thc natives in the area.;
- dctcrminc the role of revenge as a factor in forest burning;
- identify socio-cultural learning experiences that influence, reinforce, and institutionalize the practice of forest burning among the same; and
- review approaches adopted by the local forestry agency to prevention of forest fires.

LITERATURE REVIEW

Cruz (1985a, b) asserts that the fight against forest fire in the country is hampered by certain institutional and external problems compounded by public apathy toward forest protection due to the misconception that tire is the sole responsibility of the BFD. Misra (1983) described forest fire types, causes, uses, and prevention but did not focus on the personality of the fire-setter. Atabay (1978) and Binua (1978) argued for forest fire research to support forest protection, rcforcstation, and grassland management. Rabanal (1973) believes that the problem reflects a lack of knowledge on the part of those who regularly burn forests, who use fire to prepare land for planting and who do not fully understand the consequences of burning. Researcher like Duldulao, and others (n.d.) and Strasser (1970) stress the socio-economic angle, arguing that forest conservation consciousness cannot bc instilled among those involved in the destructive activity unless they are given an alternative way of earning their living. Social and cultural characteristics of people living near or within fire-prone forest areas, and attitudes of those people regarding forest burning, local forestry agencies, and their representatives, were identified by Bertrand and others (196.5) as factors relevant to the potential success of fire prevention programs. Forest fire has been attributed to plain ignorance of fire prevention practices, irresponsibility, carelessness, and grudges against forestry personnel.

METHODS

Place of Study

The study was conducted in the barangays of Bokiawan and Hucab in the municipality of Kiangan (1975 population

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15,985; area 443.3 km^2), and in the barangays of Panopdopan and Nayon in the municipality of Lamut (1975 population 9,516; area 104.5 km*). These sites were selected based on greatest fire incidence, kaigin activity, concentration of forested area, food or eating habits, sustainability and availability of land, weather conditions and facilities affecting agriculture as principal mean of livelihood.

Research Design

Upland farmers or kaingincro (shilling cultivators), who are termed munhabals in the Ifugao vernacular, served as the primary source of data; teachers and government officials served as secondary sources. The kaingincros were covered by complete enumeration, which yielded 60 respondents. The teachers and government officials group were criteria-selected using purposive stratified and multi-stage sampling. The Bureau of Public Schools (BPS) Form as of October 1978 and the Sangguniang Panlalawigan, Sangguniang Bayan, Barangay Council, and local BFD personnel were the sampling frames. Twenty respondents (10 from each municipality) from each group made up the sample. There were a total of 103 respondents (60 kaingineros, 20 teachers, 20 government officials, 2 municipal mayors and 1 BFD district officer--the last 3 treated as special case studies).

Collection of Data

Interview Method. Semi-structured interviews, mostly without the schedule on hand, were generally conducted inside offices, school premises, business establishment, and residential buildings in Bokiawan, Bolog, Ambabag, Baynihan, Baguinge, Cawayan, and Poblacion for Kiangan and in Panopdopan, Lawig, Nayon, Payawan, Mabatobato, Pieza, Bunog, Magullon, and Poblacion for Lamut. The term "forest fire" was somttimes-used in place of "forest burning behavior" to make it easier for the kringineros to grasp the concept.

Interviews were sometimes recorded on tape, particularly "huddled types" when one interviewee suddenly became several as wives, children, and other members of the household also sought attention. The recorder was introduced as a radio set to prevent selfconsciousness. Inquiry was directed primarily to the head of family, who was informed beforehand of the object of research. For the secondary sources, the average interview time took 1 hour and 40 minutes. Most of these were approached during their "break" periods and were not briefed as to the purpose of the interview.

Participant Observation or Case Study Method. "Live-in" observations were made during waking hours and upon return of the kaingineros from their kaingins in the evenings. Interviews were conducted in the morning before the subjects set out, when they invariably spend some time huddled together outside their huts as if awaiting the sunrise. Some return at noon in order to have an early start on the next day's work or because the kaingins are located some distance from the dwellings. The interviewers also observed the baki performed by the different households, and these occasions also presented opportunities for casual and natural talks with the natives. The babiis is an Ifugao native ceremony or rite performed to invoke the favorable intervention of spirits of dead ancestors and gods or deities on the occasion of sickness *in* the family or in token of thanksgiving for favors received. The last day was spent in the kaingins for a first-hand view of the situation.

FINDINGS AND DISCUSSION

The case of the kaingineros is highlighted in this report.

Socio-Economic Attributes

Rokiawan. The study centered upon the activities of an all native, almost SO-percent unconverted and non-formally educated majority in the sitios of Bunog and Kayapa. Farming is the main occupation, with almost 50 percent professing ownership of payo (ricefield), habal (kaingin), muyung, and animals? Per-family income is not computed. Most families are engaged in wood-carving business. Elementary school children earn an average of P21 .OO per week by carving pieces of spoon and fork figurines, while older children earn an average of P87.50 per week. In Bunog, all heads of families depend on sales of coffee for their monthly income. The average quantity brought by each family to the town proper is to 5 gantas of coffee valued at P40.00 at P1 .OO per chupa. Another source of money or cash income is the littuco (rattan fruit) which is harvested during September, October, and November and yields an average income of P250.00 at P25 per container. If they were to sell their palay, their income would be barely sufficient for daily needs. The *owners* of ricefields plant the traditional variety of palay and harvest only once a year. The yield is usually reserved for family consumption or for emergency barter in the adjacent or nearby barangays of Mungayang and Bayninan. Camote (sweet potato) harvested from the kaingins is the staple food. For those who own ricefields, rice is eaten alternately with camote.

Hucab. The study was centered in Hoba, where the kaingin system is the chief occupation. Camote is the main staple. The respondents also own pigs, chickens, and ducks. Cash income is derived from the sale of bananas and coffee grown in backyard gardens or in small orchards leased from other individuals (who arc not necessarily Ayangans). Unlike the other native respondents, the residents of Hucab do not own forests.

Nayon. The five primary subjects were predominately male, married, educated, but unconverted kaingineros of Binoblayan. All admit having a habal or patch of unirrigated agricultural land, but not all possess a payo and a muyung. The sale of bananas provides each family with an average weekly income of P24.00. Three to 400 pieces of unripe bananas are sold at an average of P6.00 per hundred. Camote and corn arc sold for about P20.00 per kerosene can.

Rice is harvested only once a year. The cash value of the palay harvested by a kainginero family averages P200.00 per year.

Panopdopan. The respondents have their own hospital, an elementary school building, and business establishments. Their characteristics are not basically different from those of residents of the other barangays studied. Those who were first to settle in the area are better off economically than those who came later. Panopdopan's forests were originally a public or communal forest site of the mother municipality of Kiangan. The residents of Panopdopan established ownership of these forests through the simple expediency of declaring them for taxation purposes and by claim of continuous and peaceful occupation. The privilege of developing these forests into ricefields or banana or coffee plantations has been exploited to the hilt. Residents leave their private forests well enough alone.

Cognitive Attributes

The findings about the kaingineros are generally applicable to all the groups studied. All kaingineros profess non-awareness of any government-promulgated law regulating kaingin. They believe that common law dictates that kaingins, are made in "open areas" regarded as "public land," and not on forests owned by private individuals. They do not understand why they should be prohibited from burning or utilizing fire as a tool in their kaingin practice. Forests are viewed primarily as private properties. The owner of a muyung is thought to be in the best position to care for it properly and manage it as a source of lumber, fuel, and the water that irrigates his ricefield. All claim that no one from the FNB had ever visited their areas. Non-observance of laws against burning accordingly stems in part from non-enforcement by the government. The local people arc scarcely aware of the presence of a local forestry service, and find it very difficult to conceive that the muyung could ever be placed under state control. All agree that public lands should bc distributed to the landless, who can develop these as sources of stable and adequate income and livelihood. All endorse stricter regulation and control of the activities of loggers and wood carvers in Ifugao, who arc held responsible for the wanton destruction of the public forests.

Beliefs Relevant to Forests and Forest Burning Activities

Two pervasive beliefs are relevant. The first has to do with forest ownership and seems to provide the key to the burning behavior. The natives know that the government has legal right over the forest, but they believe that the right belongs to the people who own the forests. Some natives explained that the government owned the public forests. The second belief relates to fires in the muyung. The natives do not regard these as forest fires but merely as a routine activity or tool for preparing the kaingin portion of the forest land for planting. The munhabal sets fire to what a non-native would consider as forest when there is no known claimant to the area and when customary law defines the area as public land primarily for kaingin. Burning preparatory to planting is indispensably customary.

Socio-Cultural Learning Experiences That Influence, Reinforce and Institutionalize the Practice

The native who lives in a more remote area learns to eat camote morning, noon, and evening, or, if he is luckier then the other kids in the neighborhood, camote alternated with rice. Camote is planted mainly in the family kaingin. Cleaning of rice fields starts in January, and rice planting is completed by March. The ricefields are then temporarily abandoned while the natives prepare their kaingins, which are usually located some distance away from the ricefields. Cutting down of vegetation starts by April. The grasses and trees are left to dry for at least a month, then the natives go back to bum them. Burning commences by May. Mongo is planted as soon as burning is completed, and camote is planted August after the mongo is harvested. The camote crop is harvested 5 months later. The habal is then left idle until April, when the grasses are cut, dried, and burned preparatory to planting activities. The process is repeated year in and year out unless, the place is totally abandoned in favor of another occupant. There is no room for idleness. Those who do not own any habal or payo earn their livelihood by helping clean and prepare rice fields for planting in consideration of wages in money, or by cultivating and planting someone else's ricefield in return for half of the harvest. The culture is highly animistic. The native believes in a supreme being whom he calls Maknongan, in a hierarchy of lesser deities, and in ancestral spirits (anito). When a baki is performed for a particular purpose, sacrificial animals are butchered and offered to the spirits. The number and kind of animals sacrificed depend upon the financial capacity of the family requesting the baki. The raising of animals is thus required by religious customs, and kaingins must be cultivated to provide food for the livestock.

Observing forest fires on mountainsides, especially during the night, is pani-o (taboo). Those who observe such fires accidentally are cautioned to keep the matter strictly to themselves. This taboo enables a public forest fire-setter in Ifugao to go about his way unchallenged.

Revenge as a Factor in Forest Burning

Envy, anger, or hatred were seen as motives for burning in only a few cases. Respondents suggested that laborers employed by the local BFD office in its nursery and plantation set reforestation projects on fire to get even for being laid off, for delayed payment of wages, or simply to ensure their period of employment. There were insinuations that the local forestry office was in cahoots with its laborers in perpetuating tires, especially in the reforestation plantations, to justify its continuing budgetary allocation for forest fire protection.

Approaches Adopted by the Local Forestry Agency to Prevention of Forest Fires

- Constant forest guard patrol in fire-prone areas before and during the dry season.
- Intensification of forestry information drives
- Constant dialogue with local leaders on forest conservation projects and programs.

Interestingly, the respondents interviewed in connection with this study declared non-awareness of these activities allegedly undertaken by the local BFD agency.

RECOMMENDATIONS

We recommend allocation of adequate funds for an intensive and extensive census to determine the number of people involved in kaingin-making throughout the country. The census should determine (a) the circumstances and factors that support kaingin-making, (b) the nature and extent of forest destruction resulting from fire and other causes, (c) local beliefs, customs, and practices pertaining to forests, their ownership, purpose, use, etc. in relation to govemment-promulgated forestry laws, rules, and regulations. The findings of the should be used to guide the repeal or amendment of existing forestry laws, which should be made compatible with local beliefs, customs, and practices.

We recommend allocation under title in favor of landless families solely dependent upon kaingin-making for survival of all available forest lands claimed under current ownership by reason of inheritance or succession or by actual, continuous, and peaceful possession for a period of more than 15 years provided that the recipients, their heirs, or their successors-in-interest shall not alienate their allocations or interests therein within 30 years from the date of allocation of title and, provided further, that no title of ownership shall be granted except after the lapse of 5 years from the date of allocation and upon proof of occupation, development, and improvement of his allocation particularly in tree planting of whatsoever kind suitable in the area. In the allocation of such forest lands, first priority should be given to the natives and second priority to local residents.

We recommend reorganization of the BFD, particularly on the district level. Personnel should be dedicated, competent, and active. Employment preference should be given to qualified applicants who are natives or residents of the districts served.

We recommend that adequate funds be allocated so that FEB offices throughout the country can regularly conduct information drives, and so that special educational efforts can be made in areas notorious for forest destruction.

The theoretical framework prcscntcd in this report should also receive further consideration. Some hypotheses worth testing are as follows:

- There is no relationship between level of education and level of information or knowledge about the destructive nature of forest fires.
- There is no relationship between level of information about the destructive nature of fires and forest burning behavior.
- There is no relationship between perception of the local forestry agency's role and forest burning behavior.
- There is no relationship between beliefs held about forest ownership and forest burning behavior.
- The indigenous institutions of the Ifugaos that may be related to forest burning should receive further consideration and study.
- Camineros, or road maintainers employed by the govemment, members of the local police force, out-of-school youths, and others should be officially consulted as respondents in studies of this nature; they can be sources of pertinent and valuable information.

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SMOKE MANAGEMENT: ARE RIGHTS INCLUDED WITH THE RESPONSIBILITY TO USE FIRE IN MANAGING PUBLIC LANDS?

Dean A. Shinn'

In the southeastern United States, the native biota of many natural ecosystems are adapted to periodic burning. It is generally believed that in Florida at the time of European intervention, these ecosystems were sustained as fire climax communities by relatively frequent fires, resulting both from natural and from cultural causes.

The preservation and where necessary the restoration of the aboriginal structure and function of biotic communities occurring on State Parklands is central to the mission of the Florida Park Service. In pursuit of this mission, park managers throughout the State have been charged with the responsibility of using fire and prescribed burning techniques to manage Florida's State Parks.

One might suppose that a public responsibility to use fire in this way would be accompanied by certain rights to produce and dispense smoke from the areas being burned. The premise of this paper is that such rights should naturally cmcrgc from that public charge. However, Florida Statutes and the contingencics that they govern arc not yet viewed in such a way that the practical circumstances associated with public land management can be administered form this point of view. In fact, potential liabilities, rather than rights, are among the most prominent aspects of the public responsibility to usc fire as a tool in modern management of commonwealth resources.

In Florida, many state parks are small and are defined by boundaries that are uneven and often broken by parcels of adjacent private land. Many parks contain private inholdings wholly within their boundaries. Many arc also situated in highly urbanized areas and are bordered, even surrounded, by high-value commercial and residential development. Very few of Florida's slate parklands arc located away from major roadways. In the Florida panhandle, large areas arc occupied by military installations, and many other areas arc traversed by military and commercial flight paths. In addition, a relatively high proportion both of the seasonal and of the year-round population is composed of elderly people, many of whom have respiratory problems.

As a result, the heat, smoke, and ashes emitted from prescribed fires are likely to affect people and property beyond state park boundaries. Because developed areas potentially affected by prescribed fires are so close to parklands, and in many places are a major component of a park's external environment, the weather conditions under which tire can be safely used in parks are seriously constrained. In many places, bum prescriptions can be written to accommodate only winds of a very specific speed and direction. Along the Panhandle Gulf Coast, many State parklands can be burned under prescription only after a winter cold front has passed. Under such circumstances, the wind blows rather predictably from the north for a relatively reliable period of time. This situation restricts the range of options open to park managers in their use of fire to restore and preserve Florida's original natural ecosystems.

It must be emphasized here that land managers employed by the Florida Park Service are well aware of the legitimate and compelling hazards associated with the smoke emitted from prescribed fires. They have been and will continue to be diligent in planning fire management activities to avoid traffic hazards along major transportation corridors and to protect public health.

Certain other problems, such as ash falling into nearby swimming pools, soot soiling laundry hung on clothes lines, or simply the unusual smell of burning vegetation, are also associated with the smoke and ash produced by prescribed fires. These problems can be characterized as nuisances rather than as genuine dangers or health hazards, however.

These problems should be addressed first by establishing open, good-faith communications between park personnel and local citizens. A conscientious public relations effort should be an integral feature of each park's fire management program. The park's neighbors need to be informed about the benefits of responsible fire management procedures and advised of the fire planning process prior to prescribed burning activities being undertaken. Adoption of seriously constrained prescribed fire management procedures as a means of avoiding inconvcnicnce, rather than genuine hazard, would be ill-advised and would likely not achieve anticipated long-range ecological objectives. Thorough, good-faith public relations efforts should be undertaken early in every prescribed fire and smoke management program.

Of course, even the best efforts to inform the public and to solicit the cooperation of all who might be affected may not be entirely successful. Some among a park's neighbors simply may not be reached or may not be persuaded to coopcrale. Litigation may result.

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This being the case, the author sought to determine if a line of legal reasoning could be set forth to advance the notion that a set of legal rights to produce and dispense smoke should be acknowledged as being implicit to the public responsibility to use fire as a resource management tool on public lands. However, the existing case law in Florida does not specifically address this issue. Certain judicial decisions and findings that can be grouped under legal classifications such as "nuisances," "negligence," "environmental rights and remedies," and "explosion and fire" pertain to this subject only in a very general sense. It is unlikely that a cogent legal argument in support of "smoke rights" could be derived from the existing case law. Indeed, the time invested in attempting to do so might be ill-spent in the absence of a test case well suited to the development of such an argument. However, it is not likely that the Florida Park Service would wish to promote circumstances under which suitable legal reasoning could be developed through litigation. Therefore, other means of establishing "smoke rights" were investigated -- means that would also serve to support the assertion of such rights, should a judicial defense of this concept become necessary.

Three options were examined: easements, land use plans, and original legislation.

Easements: Establishment of buffer zones around State parklands through the institution of conservation easements or other special land use agreements with owners of adjacent private lands can be used to codify mutual acceptance of specific fire and smoke management practices. This approach is most practical and most likely to be successful if undertaken while private lands surrounding parklands remain open and undeveloped. After residential or commercial development occurs, ownership -- and therefore decision-making authority -- is likely to be dissipated among many separate private interests. To achieve agreement concerning prescribed burning and smoke management contingencies with one, or with only a few, adjacent ranchers and timbermen can be a rather straightforward matter. On the other hand, reaching agreement among all potentially affected parties in an expanding area of mixed residential, commercial, and industrial land uses would be a much more ambitious undertaking. Therefore, conservation easements that acknowledge specific fire and smoke management rights and responsibilities should be established as early as possible in a region's development cycle. They should stipulate the conditions under which such rights and responsibilities can be exercised without risk of legal constraint and should attach in perpetuity to the land, with provision for conveyance with the deed to each succeeding owner.

Land Use Plans: In Florida. comprehensive growth management planning at the local level has been mandated by stats law (Chapter 163, Part II, Florida Statutes), and a local land use planning process has been established by administrative rule (Chapter 9J-5, Florida Administrative Code). In this way, each county and municipal government in Florida has been charged with the responsibility to develop and to implement a comprehensive land use and growth management plan. This planning process can be used to establish an explicit public acknowledgement of the need to use prescribed fire as a land management tool and of the implicit consequence of dispersing smoke from the areas burned. Local planning documents are fitting legal instruments in which to codify this public acknowledgement of "smoke rights" in association with established fire management responsibilities.

However, the local planning process is a long and open-ended procedure. Its results can vary widely from one county to another and also among the municipalities within a single county. Within certain general state-wide parameters established by rule and within certain basic standards set by each county, the specific provisions incorporated into any particular plan can be either favorable or very unfavorable with respect to fire management on public lands. The quality and strength of provisions addressing fire and smoke management on parklands as finally adopted in a plan depend largely on the commitment and tenacity of local park staff and of sympathetic citizens. The propriety of fire as an appropriate tool in modem land use management should be introduced early and should be re-enforced at ever stage of the planning process. Because land use planning is a cyclic and reverberative process, specific achievements can be rather transitory. Involvement at the local level must be thorough and continuing.

Legislation: The Florida legislature is now debating enactment of the "Florida Prescribed Burning Act." (After this paper was presented, the Legislature enacted this initiative as Chapter 590.026, Florida Statures.) This document states that prescribed burning contributes to public safety 1) by reducing fuels and the risk of wildfires; 3) by helping to maintain biotic diversity and the ecological integrity of native communities; and 3) by facilitating the revegetation, restoration, reforestation, and enhancement of public and private lands. The bill also authorizes public education and technical training programs, where appropriate, in order to assure general acceptance and proper use of fire as a land management tool. It then declares that prescribed burning, when properly authorized, is in the public interest and does not constitute a public or private nuisance. Most important with respect to smoke management, the bill finds that prescribed burning is a property right of the landowner and that the owner or his agent, when conducting an authorized bum, is not to be held liable for damage or injury resulting from tire or smoke, unless negligence is proven.

This bill introduces into legal debate the prospect of establishing certain smoke rights in association with an acknowlcdgcmcnt of related prescribed burning and fire managcmcnt rcsponsibilitics. Unfortunately, language in the bill that is pertinent to the concept of “smoke rights” is rather nebulous. It does not provide clear and precise guidance concerning assertions of negligence, especially where drifting or wind-driven smoke and ash arc concerned. Howcvcr, this draft legislation establishes a useful context for continuing public examination of the rights of land managers relative to fire and smoke management.

SUMMARY

Are smoke rights included with the responsibility to use fire in managing public lands? The premise of this paper is that certain rights to produce and disperse of smoke from lands subject to prescribed burning should be implicitly associated with the public responsibility to use tire as a land management tool.

The existing case law will not explicitly support such an assertion through legal argumentation, while public agencies are not inclined to promote litigation for the purpose of establishing favorable case law.

However, three alternatives exist for establishing such rights, or at least for developing legitimate public acknowledgement of the concept of such rights. To be practical and reasonably effective, casements specifying smoke rights should be instituted early in a region’s development history. Local land use planning processes can be used to develop explicit public acknowledgement of contingencies associated with the usc of tire as a land management tool, but involvement at the local level must be both consistent and persistent. In Florida, legislation specifically addressing fire and smoke management as a property right has been drafted (and was enacted as of October 1, 1990.) This latter alternative is a particularly straightforward approach. In each case, however, clear and direct communication with the public concerning the role of prescribed burning and smoke managcmcnt in public land management is ncccssary.

LITERATURE CITED

Chapter 163, Part II, Florida Statutes: Intergovernmental Programs, County and Municipal Planning and Land Dcvclopment Regulation.

Chapter 9J-5, Florida Administrative Code: Local Government Comprehensive Planning Regulations

Chapter 590.026, Florida Statutes: Florida Prescribed Burning Act.

SPATIAL DYNAMIC FIRE BEHAVIOR SIMULATION AS AN AID TO FOREST PLANNING AND MANAGEMENT

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Abstract-Mediterranean shrub communities dominate fire-prone landscapes in many parts of Portugal. It is feared that streamside anti-erosion buffers of natural shrub vegetation represent a fire hazard lo plantations of eucalyptus and other trees. The FIREMAP system, which can simulate fire behavior in spatially nonuniform environments, was used to assess the extent to which fire buffers can become main vectors of fire spread in Portuguese landscapes. FIREMAP predicted that tire will spread across landscapes consisting of Eucalyptus sp. plantations and streamside borders of natural Mediterranean vegetation much more rapidly than across landscapes consisting of Eucalyptus plantations and streamborders of planted Quercus sp.

INTRODUCTION

About 70 percent of Portugal's land area is unsuitable for agriculture. It has been suggested that these areas should be the subject of reafforestation programs (Grupo Coordcnador do Projecto Florestal 1986). Forests cover only about 36 percent of the country, so there is much room for this kind of initiatives, such as the ones recently attemptcd with support from organizations such as the World Bank and the European Economic Community.

Due to climatic and socio-economic factors, wildfires are a major threat to Portuguese forests. An average area of 42 000 ha burned yearly from 1973 through 1985 and in 1989 about 54 COO ha of forests werc destroyed by wildfircs.

A recent trend in Portuguese forcstry is the rapid expansion of plantations of exotic species for short rotation biomass production, for use by the paper and pulp induahies. The environmental impacts of those plantations have been a topic of heated debate, and legislation was issued regulating soil preparation and plantation and silvicultural practices, with special emphasis on minimizing soil erosion, hydrological disturbances, and loss of biological diversity.

The legislation that regulates these plantations requires that natural vegetation be lcft along stream channcls for erosion protection. The widths required dcpcnd on particular situations but are generally betwccn 20 and 60 m. Consequently, the majority of the projects generate a striped landscape where buffers of constant width indiscriminately marginate stream channels.

In this paper WC investigate thc possibility that wherc natural vcgetation consists of Mcditcrranean-type shrubs, this landscape structure may contribute to improved fire propagation by creating paths of faster spreading firc that make otherwise unavailable fuels more likely to bum. in fact, these shrub communities bum intensely and contribute to more effective preheating of the less easily ignited fuels in the neighboring plantation forest, thus setting the stage for larger, more intense fires.

The objective of this work was to usc a PC-based spatial analysis system that simulates the spread of fire in a spatially nonuniform landscape in discrete time steps (the FIREMAP system) to assess the extent to which anti-erosion buffers along streams can become the main vector of tire propagation, or on the other hand, work as barriers to the spreading fire.

We simulated structurally simple landscapes, not only to facilitate interpretation of the results, but primarily because this corresponds to the actual spatial structure of Eucalyptus plantations.

FIREMAP

This fire spread simulation system, designed at the University of Arizona (Vasconcelos 1988), estimates tire characteristics in spatially nonuniform environments, and displays areas burned on maps. FIREMAP consists of the integration of the DIRECT module from the BEHAVE system (Andrcws 1986) with a raster-based geographic information system, the Map Analysis Package -MAP- (Tomlin 1986), and allows distributed predictions of fire characteristics and simulation of fire spread.

In this framework, nonuniform fuels, wcathcr, and topography data arc encoded, stored, and manipulated on thematic maps, where the ficld is represented as cells of a grid corresponding to uniform parcels of land. Because the homogeneity assumptions are met, Rothermel's model can be used within each unit.

The dynamic process of a spreading tire is simulated through the use of the distance functions of MAP. Distance functions deal with the measurement of weighted distances, allowing

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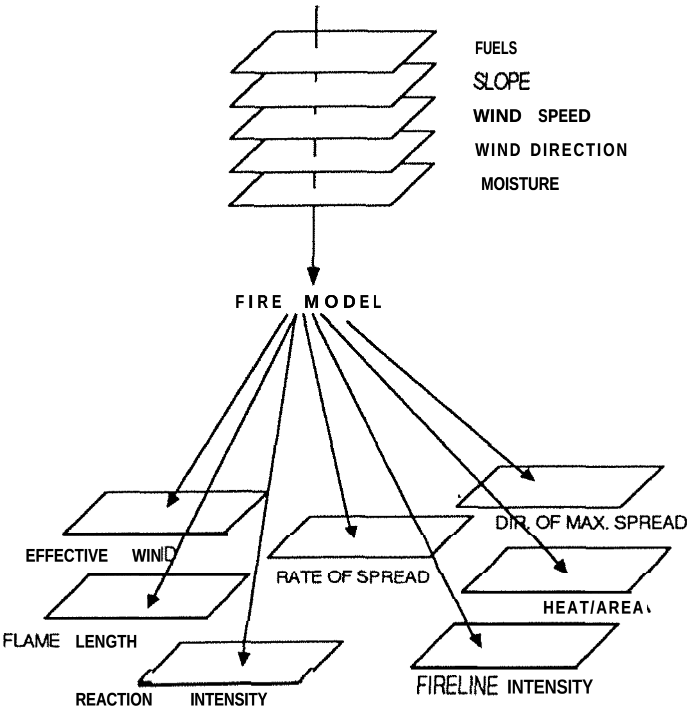
simulation of movement on a previously computed surface of "frictions". These are defined as the rate at which the fire spreads from one cell to its neighbors given the direction of the prevalent wind. The rates of spread overlay utilized depends upon weather conditions, and a new rates of spread overlay has to be utilized whenever there is a change in the weather. This is done by stopping the simulation at the end of a time step and proceeding with the spread process on a new rates of spread overlay that was calculated according to the new weather conditions.

Figure 1 illustrates how fire characteristics are computed for each cell of the data base and set of constant meteorological conditions, based on input overlays generated within MAP. These input overlays are created from the topography, vegetation, and weather data using the standard arithmetic and reclassification capabilities of MAP and the tables presented in Rothermel (1983).



APPLICATION

We ran four simulations to test fire sensitivity of four possible forest landscapes corresponding to four management alternatives for Eucalyptus sp. stands in a Mediterranean-type region. The simulations are for a 1.5 hour bum, in three 30-minute time steps, with a likely early summer weather scenario, which is summarized in table 1.

Table 1. --Weather conditions

	Temperature		Relative humidity	Wind	
	Dry	Uet		Speed[a]	Direction
	- Degrees F -		- Percent -	- Mph -	
First step	82	52	15	12	S
Second & third steps	84	51	12	12	S

[a] wind speed at midflame height

Some comments may be appropriate regarding the temporal and spatial scales we used. Under a normal weather scenario, it seems reasonable to assume constancy of temperature and relative humidity during 30 minute intervals. Windspeed and wind direction probably vary significantly at a finer time scale, and Fischer and Hardy (1972) indicate that the standard time for averaging windspeed is 10 minutes. However, Rothermel's model was designed to predict fire behavior under relatively uniform weather conditions (Rothermel, 1983), and although considerable weather changes in a 24-hour period should be expected, projection times of 2 to 4 hours, under constant meteorology are reasonable (Andrcws 1986). A more sophisticated treatment of the interactions between wind, terrain, and fire behavior will probably require expansion of the FIREMAP system to include a surface windflow model such as KRISSY (Fosberg and Sestak 1986).

Spatial resolution of the database is considered appropriate since it satisfies what we believe to be the two most important considerations. On the one hand, cell size is small enough to capture all essential landscape features and overall structure. On the other hand, the cells are large enough in comparison to average flame front depths to ensure that steady state spread conditions are almost always present (Catchpole et al. 1989).

The Digital Data Base

The digital cartographic data base corresponds to a 2280 ha area of undulating terrain. Altitudes range from 200 m to 600 m with aspect predominantly to the east, northeast, and southwest on steeper slopes. The scale is 1: 12,000 and there are 7.5 rows by 76 columns with a cell size of approximately 0.4 ha (1 acre). The data base consists of the following information layers: TOPOGRAPHY, STREAM CHANNELS, an LANDSCAPE1 , 2, 3, and 4.

The landscape maps correspond to different management alternatives for vegetation buffers along ephemeral streams in Eucalyptus plantations. LANDSCAPE1 represents a landscape of continuous Eucalyptus stands without any kind of stream buffering. LANDSCAPE2 and LANDSCAPE3 represent soil conservation alternatives favored by Portuguese environmental legislation regarding fast-growing plantation forests. In landscapes 2 and 3, natural vegetation is retained as stream side buffer strips of predetermined minimum width. Different stages of shrub development are considered in landscapes 2 and 3. Landscape 2 represents the case of having medium height evergreen sclerophyllous shrubs along the streams and landscape 3 the case of tall and dense stands of the same shrub types. In the fourth vegetation alternative (LANDSCAPE4) a more balanced and diverse landscape is considered with wider strips of deciduous Quercus sp. planted along stream banks. Figure 2 shows the stream channel pattern.

Simulation

Four sets of input maps for the fire model were generated as explained above. The FUELS overlays were created by reclassifying the vegetation types to one of the 13 standard fuel models (Anderson 1982) based on a correspondence presented in Barreto (1985). Barreto classifies most of the vegetation cover types found in Portuguese landscapes as the models included in the mentioned set of 13 models. The correspondence we used for this particular case is as follows:

Eucalyptus sp. stands- fuel model 5
Medium evergreen sclerophyllous shrub stands- fuel model 6
Tall, dense evergreen sclerophyllous shrub stands- fuel model 4
Deciduous oak stands- fuel model 9

FIREMAP generates maps with the expected fire characteristics for the 4 landscape alternatives in the 3 time steps and then simulates the spread of tire, with a given source point, for the alternatives considered. Maps of the expected flame lengths at each database cell are also provided.

RESULTS

The maps displaying the predicted areas burned and respective expected flame lengths are shown in figures 3 and 4. Table 2 shows the number of burned cells for each time step and landscape and Table 3 the number of cells in each flame length class for each of the 4 simulation scenarios.

eucalyptus stands

natural vegetation (stands of sclerophyllous shrubs)

Figure 2.--The stream channels.

Table 2.--Cells burned

	1st step	2nd step	3rd step
	. . . . . . & r		
Simulation 1	66 (26.4)[1]	81 (32.4)	48 (19.2)
Simulation 2	91 (36.4)	102 (40.8)	a2 (32.8)
Simulation 3	154 (61.6)	117 (46.8)	124 (49.6)
Simulation 4	38 (15.2)	20 (8.0)	31 (12.4)

[1] area burned, in hectares

Table 3.--Expected flame lengths

	0 - 4 feet	4-a 8 feet	- 11 feet	>11 feet
	. . . , . . . Number of ceils			
Simulation 1	4	153	38	
Simulation 2	4	216	55	
Simulation 3	6	218	87	84
Simulation 4	16	64	9	

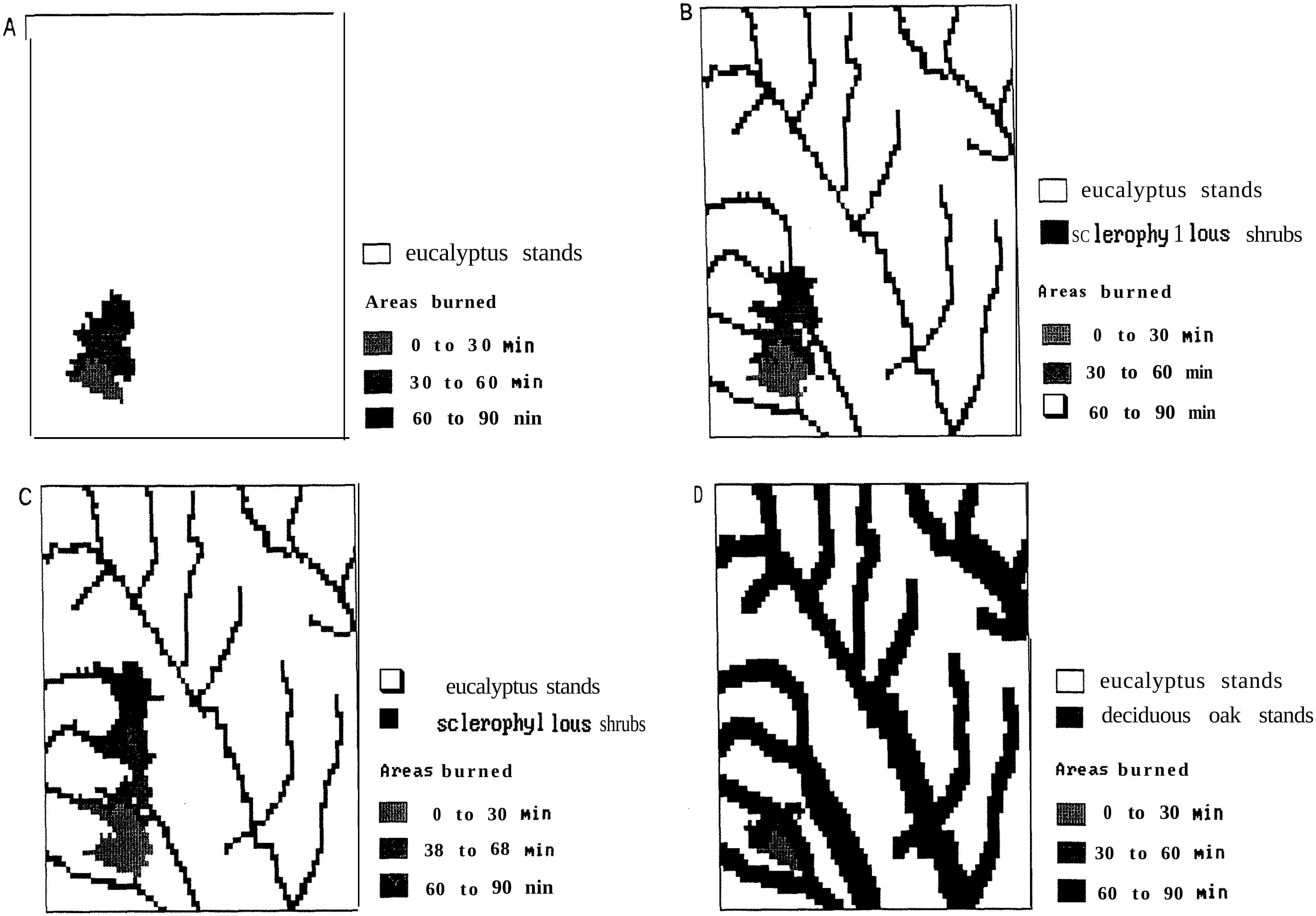


Figure 3.-Areas predicted to bum: (a) simulation 1, (b) simulation 2, (c) simulation 3, and (d) simulation 4.

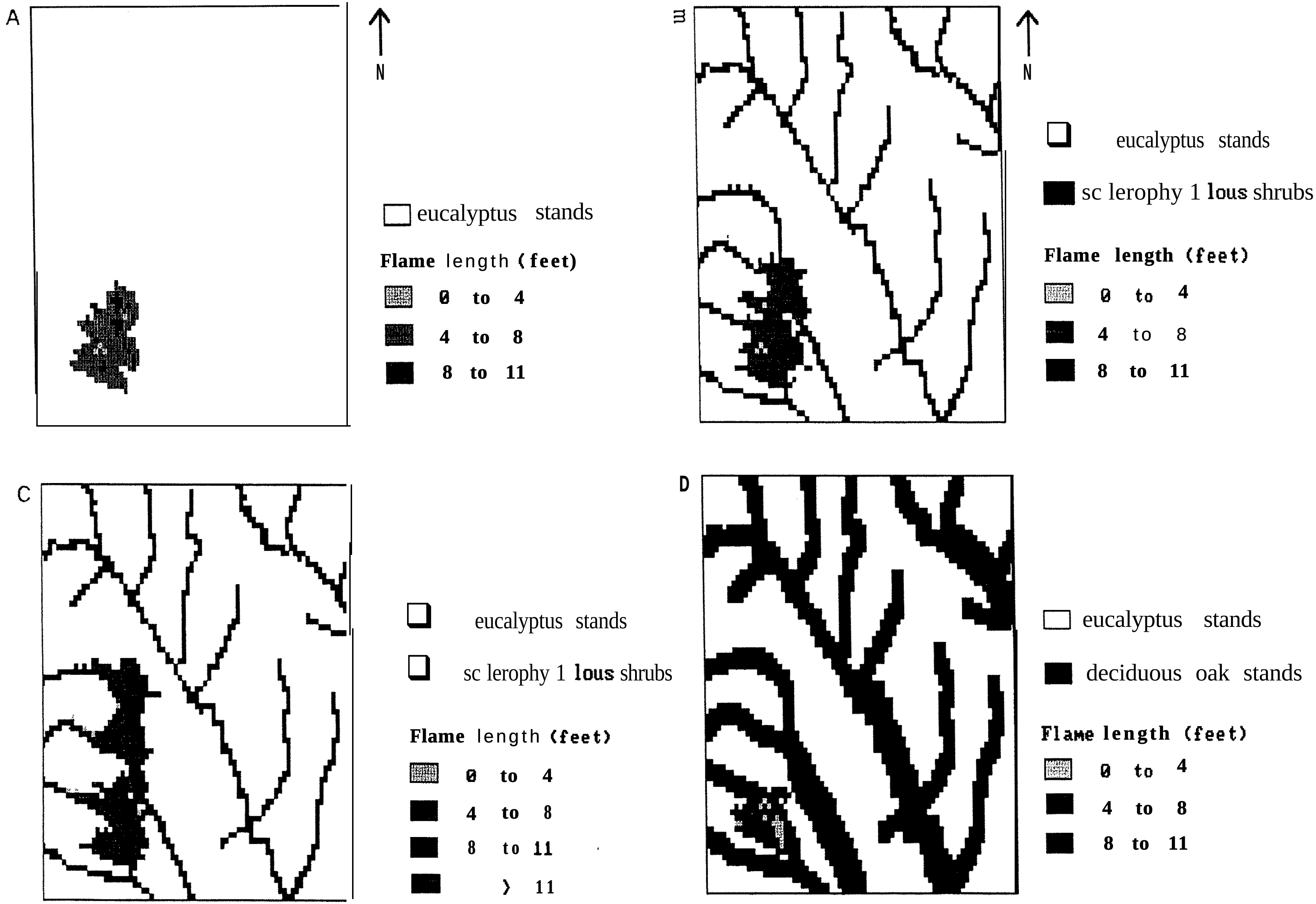


Figure 4.–Expected flame lengths: (a) simulation 1, (b) simulation 2, (c) simulation 3, and (d) simulation 4

DISCUSSION AND CONCLUSIONS

The results above indicate that under the simulation conditions, natural vegetation left along stream banks may actually contribute to increased fire problems. In fact, there was an increase of 41 and 102 percent of predicted burned area on the second and third alternatives (shrub stands along stream banks) relative to the first alternative (Eucalvntus sp. only)

The simulation outputs suggest that the striped patterns generate preferential paths for fire, creating spreading conditions that lead to higher perimeter-to-area ratios of the burning area, thus improving propagation chances by increasing the length of contact with unburned fuels. On the other hand, the deciduous oak stands have lower rates of spread and create zones of "higher friction" that retard propagation of fire to the Eucalyptus stands. There was a decrease of 46 percent of predicted burned area for this alternative when compared with the first alternative. Given the importance of convective heat transfer in deep fuelbeds such as those represented by shrub formations, a worst case scenario involving higher windspeeds would probably emphasize even more the rapid spread of fire along the stream buffers.

Landscape planning in areas of intensively managed plantation forests is a problem involving multiple, potentially conflicting objectives. Conflicts arise not only between economic production and environmental protection, but also between different ecological concerns. In the present case study, FIREMAP was used to emphasize one such conflict, between erosion protection and tire hazard, in a way that provides information about fire characteristics in a surrogate laboratory mode. These quantitative data can be integrated with other economic and environmental information to support cost-benefit or multiobjective decision analyses of forest management alternatives.

ACKNOWLEDGMENTS

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